Integrated spatial assessment of human pressures and impact on UK seabed habitats

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Declaration

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The research reported is my own original work which was carried out in collaboration with others as follows:

Chapters 1 and 6 – written by Jo Foden

Chapter 2 – Jo Foden was the lead author on a published paper:

Foden J, Rogers SI, Jones AP (2008) A critical review of approaches to aquatic environmental assessment. *Marine Pollution Bulletin* 56:1825–1833

Jo reviewed the literature and developed the categorisation system which built on preliminary work of the International Council for the Exploration of the Sea (ICES). Dr Stuart Rogers provided advice on the terms of reference of the ICES WGECO work. Jo wrote the paper and Drs Rogers and Andrew Jones reviewed drafts of the manuscript.

Chapter 3 – Jo Foden was the lead author on a published paper:

Foden J, Rogers SI, Jones AP (2009) Recovery rates of UK seabed habitats after cessation of aggregate extraction. *Marine Ecology Progress Series* 390:15–26

Jo manipulated and analysed the EMS aggregate extraction spatial data and marine landscape data in a GIS. She reviewed the literature and estimated recovery periods for UK marine landscapes affected by aggregate extraction. Jo wrote the paper and Drs Rogers and Jones reviewed drafts of the manuscript.

Chapter 4 – Jo Foden was lead author on a published paper:

Foden J, Rogers SI, Jones AP (2010) Recovery of UK seabed habitats from benthic fishing and aggregate extraction – towards a cumulative impact assessment. *Marine Ecology Progress Series* 411:259–270

Jo manipulated and analysed the VMS fishing vessel data and marine habitats data in a GIS. She reviewed the literature and estimated recovery times by habitat type. Jo combined aggregate extraction and fishing data and developed cumulative impact scenarios. Jo wrote the paper and Drs Rogers and Jones reviewed the drafts.

Chapter 5 – Jo Foden was lead author on a paper accepted for publication:

Foden J, Rogers SI, Jones (2011) Human pressures on UK seabed habitats – a cumulative impact assessment. *Marine Ecology Progress Series* 428:33–47

Jo collated spatial datasets for several human activities, and analysed them in a GIS. She reviewed the literature for recovery times from each activity and estimated recovery periods for cumulative activities, according to habitat type. Jo wrote the paper and Drs Rogers and Jones reviewed drafts of the manuscript.

Abstract

Understanding the pressures and impacts of human activities on the marine environment can help inform management responses.

A literature review was conducted of aquatic environmental assessments to identify examples of good practice. Where confusion was found in some assessment terminology, definitions were suggested. A new classification system was proposed, based on the environmental components considered, methodologies and nature of the linkages between components, and the inclusion or exclusion of socio-economic factors.

The spatial and temporal distribution of key activities causing direct pressure on the UK seabed in 2007 were analysed and their individual and cumulative impact were estimated according to habitat type. Activities were linked to five habitats, defined in terms of their particle-size ranges. Habitat sensitivity was determined from the scientific literature of recovery rates of the benthos following cessation of each human activity. Cumulative impacts were estimated as total recovery time under 4 scenarios; greatest, additive, antagonistic and synergistic impacts.

More than half (134 400 km^2) of the seabed was estimated to be directly affected by human activities. Benthic fishing accounted for 99.6% of this spatial footprint. Sensitivity to the pressures of human activities varied by activity-habitat combinations, with estimated recovery times from individual activities ranging from <1 month for otter trawling in sand, to >10 years for aggregate extraction in low-energy coarse sediments. However, the footprint of aggregate extraction was <0.01% that of benthic fishing. Multiple activities co-occurred in only 165 km^2 (<0.1% of seabed). The longest estimated cumulative recovery time was ~15 years for aggregate extraction and dredge material disposal in low-energy gravel habitat, but the footprint was small, <0.01% of seabed.

The finding that a limited number of activities were the predominant cause of widespread, long recovery times of benthic fauna suggests that when time and resources are limited, single sector assessment rather than detailed evaluation of cumulative effects, can still usefully guide management.

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CHAPTER 1

Integrated spatial assessment of human pressures and impact on UK seabed habitats

1.1 Managing conflict in the marine environment

Human demands on natural marine resources, such as fish, minerals, oil and gas are increasing (Glover & Smith, 2003). This is leading to increased degradation of the marine environment (UNEP 1982, Choi et al. 2005, Norkko et al. 2006, Halpern et al. 2008). This has resulted in a growing need to monitor and assess resource status in order to manage demands and militate against deterioration. Planning and control of human activities can secure long-term benefits for the whole of society and nature, with sustainable marine development as the desired outcome (Defra 2009). Driven by international commitments such European directives (EU 2000, EU 2008a) and the OSPAR Commission (OSPAR Commission 2008), there has been a recent move from sectoral planning to integrated strategic co-ordination of human activity, which is ecosystem-based. However, integrated marine assessment is a recent discipline and confusion exists in the terminology and methodology used in published studies (Foden et al. 2008).

There is a continuing need to develop improved and cost-effective ways to study and monitor ecosystems, and to assess the ecological significance of change in an integrated manner (Norkko et al. 2006). In this context it is imperative that assessments establish their terms of reference, use accurate terminology, present results with clarity, and focus on informing the development of management strategies for assessment. At present these imperatives are not being met for many assessments undertaken, limiting the utility of their findings. Ecosystem-based management is an approach that recognises the complex interactions within ecosystems, of which humans are a part. Yet to implement this approach necessitates knowledge of the pressures and impacts from major human activities on ecosystems of potentially differing sensitivities.

Pressures are stresses that human activities place on the environment, for example by extracting resources such as fish. Impacts are the resultant changes in the physical, chemical or biological state of the environment and the functioning of ecosystems, which can have ramifications for human health, society and the economy. However, there is a degree of inconsistency in the use of these terms. For example some studies have mapped the spatial distribution and intensity of human *pressures*, but have referred to this as 'impact' (e.g. Halpern 2008b). This can lead to misrepresentation of impact analyses (Heath 2008). For instance, some sea areas are subject to a high intensity of pressures, such as the southern North Sea, but if the receiving environment is resilient, the level of impact may be low. Greater understanding and quantification of pressures and impacts is required to better underpin effective environmental impact assessment and marine planning, and to provide the basis for integrated marine management (OSPAR Commission 2003, Eastwood et al. 2007, Borja et al. 2008, Foden et al. 2008, Halpern et al. 2008a).

Accurate information on the extant spatial distribution of pressures and their likely impact inform spatial planning, which will contribute to development that is socially, economically and environmentally sustainable (HMSO 2009). Whilst planning in the terrestrial environment has long established traditions, for example the Housing, Town Planning, Etc. Act of 1909 (Cullingworth & Nadin 2002), marine spatial planning (MSP) is an emergent discipline (e.g. Ehler & Douvere 2009, Trouillet et al. 2010, De Santo 2011) which lags a century behind. The EU recognises and promotes MSP as a key instrument to optimise the use of marine space for economic development and for the benefit of the marine environment by co-ordinating action between public authorities and stakeholders (EU 2008b). In deciding the objectives of MSP, value judgements need to be made, which may well change over time, e.g. 'maximise fish catches' could change to 'protect biodiversity'. At present the vision is for UK's oceans and seas to be "clean, healthy, safe, productive and biologically diverse" (Defra 2002), a vision which MSP can help deliver by implementing an ecosystem approach when planning the location of activities. However, to date there remain gaps in knowledge of the distribution of pressure, and in understanding the links between human activities and the receiving marine habitat type (Defra 2010). In particular there is little understanding of the cumulative impact of several activities in one area and the ability of species or habitats to recover once a pressure has been removed.

Developments in geospatial technology have significantly enhanced the ability of scientists and managers to quantify spatial and temporal footprints of human activities in the marine environment, and to begin identifying locations where cumulative effects are likely. Geospatial technology includes global positioning systems (GPS) and geographical information systems (GIS). A GPS-based vessel monitoring system (VMS) has been compulsory on European commercial fishing boats of ≥15 m since 2005 (EU 2003) and has been used as a surveillance and enforcement tool for fishing effort. The high-resolution locational data VMS provides have greatly enhanced fine-scale study of fishing effort distribution (Lee et al. 2010). Similar electronic monitoring systems (EMS) have also been fitted to all aggregate dredgers operating in UK (England and Wales) waters since 1993 to automatically record position and 'dredging status' (Crown Estate 2009). These positional data can be easily visualised in a GIS. There are several advantages of GIS for spatial planning: spatial patterns are easily identified, statistically robust integration of multiple spatial data layers is possible and spatial data can be presented at multiple scales (Foden et al. 2008). Furthermore, web-based GIS encourages stakeholder participation in MSP decisions, for example the UK's four regional marine conservation zones projects (http://www.mczmapping.org). This technology is becoming widely used for facilitating marine environmental science and research (e.g. Danz et al. 2007, Derous et al. 2007, Eastwood et al. 2007, Foden et al. 2010b).

The aim of this thesis was to conduct an integrated spatial assessment of human pressures and impacts on UK seabed habitats. Specific objectives were to quantify the spatial and temporal footprints of human activities directly affecting benthic habitats to estimate the distribution of individual and cumulative activities. The pressure from those activities, be they individual or cumulative, might then be estimated by linking their location, frequency and intensity to habitat type. This research directly addresses the recognised knowledge gaps by contributing a method and carrying out an assessment of the pressures and impacts of human activities directly affecting UK seabed habitats.

The following sections set the context for the thesis. Firstly, the main activities of the key sectors which cause pressure on the seabed are characterised. Secondly, the study area is delineated and ways in which marine seabed habitats can be classified are discussed. Thirdly, there are several approaches to measuring habitat sensitivity and the method chosen for this thesis is justified. Fourthly, methods for estimating cumulative impact are described, and four possible scenarios are defined. Finally, the structure of this thesis is presented.

1.2 Sectoral pressures

The Marine Strategy Framework Directive (MSFD, EU 2008a) requires Member States to analyse the predominant pressures and impacts on the environmental status of marine waters. Under the MSFD, human activities are grouped into generic pressure types. Many different activities directly affect the seabed and they can be grouped into four types of pressure: smothering, abrasion, obstruction (also known as sealing) and extraction. Some activities generate more than one pressure. For example, wind turbines are obstructions on the seabed and their influence on the local hydrodynamic regime can cause scour, which is an abrasion pressure. The main activities affecting the UK seabed and the pressures they constitute are described as follows.

Aggregate dredging for mineral resources constitutes extraction pressure. The UK is one of the largest producers of marine aggregate in the world (BMAPA 2008). The main method in the UK is trailer dredging of evenly distributed deposits (BMAPA 2006), producing shallow linear furrows approximately 1 to 3 m wide and 0.2 to 0.3 m deep per pass (Kenny & Rees 1994). A limited amount of static anchor dredging also occurs, where thick localised reserves are exploited to leave saucer-shaped depressions typically 8 to 10 m deep, but occasionally reaching 20 m in depth (Dickson & Lee 1972, Newell et al. 1998, Boyd et al. 2004). The degree of impact is a function of the environment that is dredged and the intensity and longevity of extraction effort (Cooper et al. 2005).

Benthic fishing is a source of abrasion pressure and it is the most important human activity in terms of its spatial extent and level of impact in UK waters (e.g. Collie et al. 1997, Rijnsdorp et al. 1998, Dinmore et al. 2003, Stelzenmüller et al. 2008). The main types of fishing gear used in the UK are benthic beam trawls, otter trawls and shellfish dredges. Each gear sweeps the seabed at different widths and depths, which determines the area of disturbance (Collie et al. 2000). For example, mechanised hydraulic suction dredges are approximately three metres wide, whereas the overall width of a twin-rig beam trawl is generally 24 m (Hall et al. 1990, Rijnsdorp et al. 1998, Kaiser et al. 2006).

Offshore telecommunication cables represent the vast majority of cable types on the UK continental shelf. Most offshore cables are buried below the seabed to protect them from ship anchors and benthic trawl fishing damage, but a small proportion lies on the surface of the seabed (Carter et al. 2009). On rock or boulder sediments, e.g. the rocky seabed off North Cornwall, burying is not feasible and exposed cables are armoured, representing an obstruction pressure. In softer sediments of mud, muddy-sand, sand or gravel the majority of cables are successfully buried. The impacts of buried cables include loss or disturbance of sea bed habitat, representing an abrasion pressure. Burial is by sea-plough or water jet (Allan 1998, Carter et al. 2009, Drew & Hopper 2009). The overall disturbance strip produced by a plough-share and its skids in direct contact with the seabed ranges from 2 to 8 m wide, depending on plough size. Disturbance zones associated with jetting (liquefaction and coarse sediment redeposition) are typically about 5 m wide (Carter et al. 2009).

Oil and gas production infrastructure includes platforms, well head protective structures and pipelines, which are all obstruction pressures. When an oil or gas well is drilled, waste cuttings are separated on the platform and are normally discharged to the seabed (OSPAR Commission 2009), creating a smothering pressure. The size of cuttings piles depends on whether cuttings are discharged close to the seabed or at the surface and on the type of cuttings discharged (Daan & Mulder 1996, Zuvo et al. 2005, de Groot 1996, Eastwood et al. 2007). Discharges of untreated cuttings contaminated with oil-based drilling fluids and the use of organic-phase or synthetic-based drilling fluids, or cuttings contaminated with these fluids, have ceased (OSPAR Commission 1993, 2000). Consequently, for more than a decade only water-based fluids (WBM) have been used during the first stage of drilling (“spudding”), when discharges to sea cannot be prevented. The use of oil- or synthetic-based mud remains an option for drilling the deeper well sections, during which the contaminated mud is recycled and reused, then shipped back to shore for reprocessing (R.S. Rowles pers. com. 2010).

Wrecks and wind turbines represent two types of obstruction pressure. Wind turbines typically consist of monopile foundations and often scour protection is included at the base, which increases the direct spatial contact at the seabed (OSPAR Commission 2006). Waves and tides cause sediment transportation away from wind turbine foundations, scouring the seabed, particularly in areas of mobile sediments (Rees 2006). Poorly designed scour protection can even lead to secondary scour associated with the protection installation.

Seabed dredging occurs in UK waters for three purposes, capital, maintenance and aggregate dredging each of which produce characteristic material for disposal to sea. Capital dredging takes place mainly in harbour berths and approach channels and removes previously undisturbed material the nature of which is mainly consolidated material such as rock and stiff clay (A. Birchenough pers. comm.). Maintenance dredging is often a regular, long-term activity, carried out in order to maintain harbour berths, marinas and approach channels and is usually concerned with the regular removal of silt and clay or sand, deposited as a result of natural siltation. Some disposal material is also generated through aggregate dredging, this is material that has been dredged but is unsuitable for aggregate use and is therefore disposed at sea. This operation, often called silt washing, only contributes a small amount of the overall material disposed to sea and is only licensed to take place at 3 disposal sites in the UK (Birchenough et al. 2010).

There are over 150 sites designated for the disposal of dredged material around the coast of England and Wales. On average 25–40 million wet tonnes of dredged material are disposed of annually to these sites (Bolam et al. 2006, Birchenough et al. 2010). The number of sites used in any one year and the amount disposed to those sites can vary significantly. The licensed area boundaries define the area in which disposal occurs, with little under- or over-estimation (A. Birchenough pers. comm.). Material is deposited predominantly from hopper barges where dredged material is deposited through vessels' bottom opening doors. Licensees are generally guided to dispose material in the centre of the site in an attempt to restrict plumes. Furthermore some sites have licence conditions which restrict disposal operations, e.g. to certain states of the tide or to areas within a disposal site, to reduce impacts (S. Pacitto pers. comm.). There is an annual programme of disposal site monitoring (Birchenough et al. 2010).

To determine the spatial and temporal extent of the direct seabed pressures these various activities constitute, firstly requires their location and frequency to be ascertained. The following section describes the hydrodynamics and habitat characteristics of the UK marine environment within which the pressures are apparent.

1.3 Study area and seabed habitats

The study area for this thesis comprised the marine waters of the UK (England and Wales). The boundary of this area was determined for environmental status reporting under Charting Progress (Defra 2005) (Figure 1.1). Water depths range from coastal mean high water to: 100 m in the North Sea; 180 m in the English Channel; 200 m in the Irish Sea, and; >1000 m in the western approaches.

Habitats have been classified in several ways. For example, the 'marine landscapes' that Connor et al. (2006) characterised by sediment type, depth and natural hydrodynamic regime, are particularly relevant to studies of impact from aggregate extraction pressure. In studies of bottom-fishing impact on the benthos, habitats are generally defined more simply, using particle-size ranges (Kaiser et al. 2006, Pitcher et al. 2009), with natural disturbance being implicit in these habitat types. This is because disturbance, such as that caused by near-bed currents and wind-induced waves, is related to depth and grain size. Using British Geological Survey sediment types (Folk 1954), five habitat types were distinguished and mapped (Figure 1.1), based on the largest proportion of constituent particle size: mud, muddy sand, sand, gravel and reef (including biogenic habitats constructed or composed primarily of living biota). Together the five habitats constitute 97.4% of the entire UK seabed, the remainder being diamicton.

The impact of a human activity is dependent on the sensitivity of the receiving habitat. Habitat sensitivity is a complex concept. The different methods for measuring sensitivity are explained in the following section, with a focus upon cumulative impact assessment, as this approach was adopted in the material presented in subsequent chapters of this thesis.

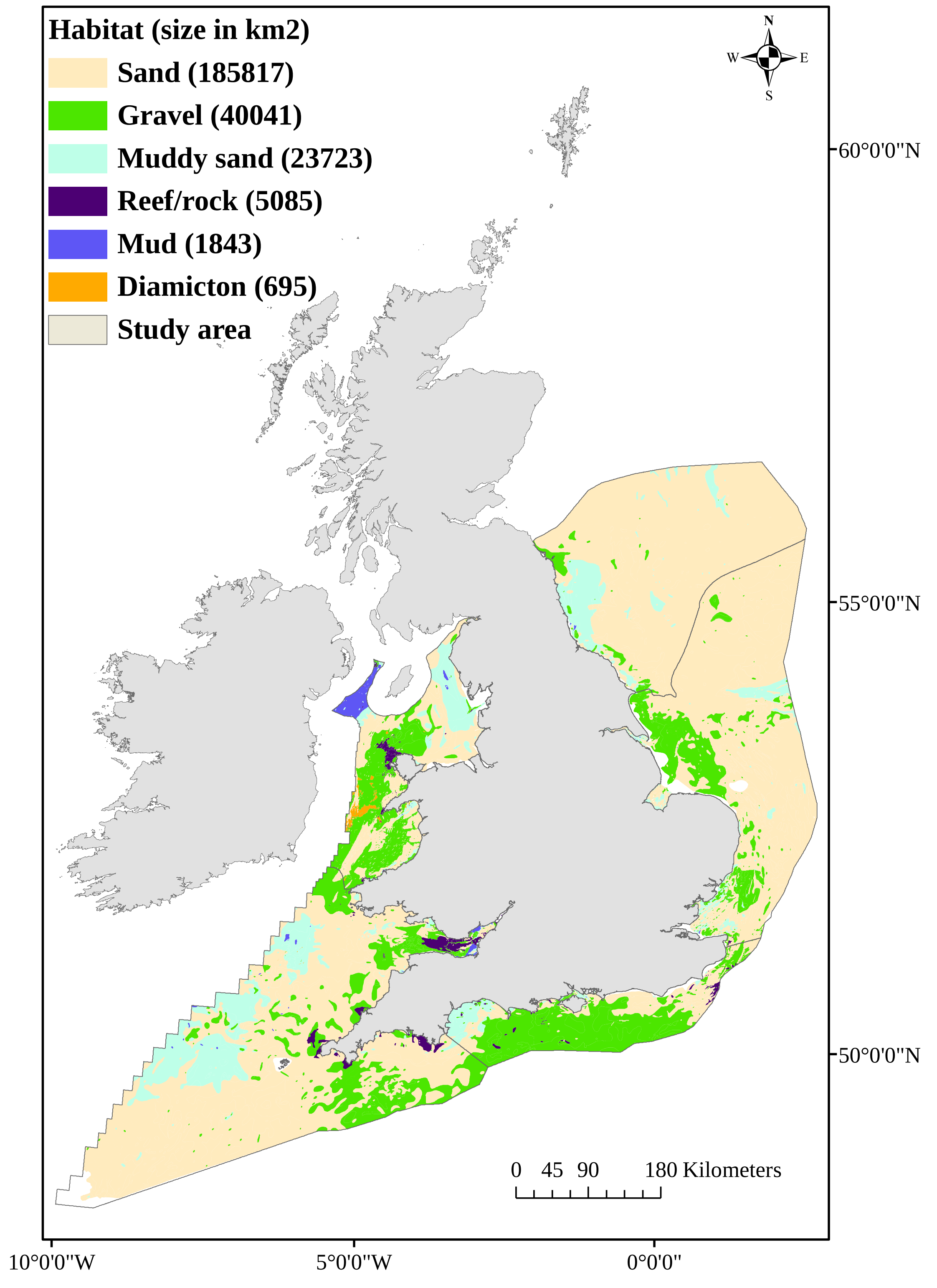

Figure 1.1: Study area. UK (England and Wales) seabed habitat types – Charting Progress reporting area.

1.4 Habitat sensitivity

Sensitivity has been defined as the degree to which features of the environment respond to deviations in environmental conditions beyond the expected range (Zacharias & Gregr 2005). Measurements of sensitivity sometimes have been based on expert judgement or scoring indices (e.g. Tognelli et al. 2009, Scheurle et al. 2010, Teixeira et al. 2010). Whilst these approaches may be suitable when comparing markedly contrasting environments, they are less so when considering the range of more closely related habitat types found in UK waters. In this case a more appropriate method would be to quantify sensitivity, for example by measuring the physical and biological recovery time of the benthic community, following cessation of a pressure of defined magnitude and duration (Hiddink et al. 2007). The assumption is, where recovery times are long, sensitivity is considered to be high.

A number of issues needed to be considered in adopting this approach. Perhaps the most important is defining ‘recovery’. Ideally this would be determined as the time-period for the re-establishment of a biological benthic community to a condition that was virtually indistinguishable from surrounding, non-impacted reference sites (Cooper et al. 2005). However, the study area has a long history of high levels of activity, for example trawlers repeatedly target the same grounds year after year (Kaiser et al. 2002, Hiddink et al. 2006). Therefore, the point at which recovery was deemed to have occurred was, for some habitats, a point in a constant disturbance cycle, and not disturbance of a pristine benthic community. Furthermore, some sites which are subjected to direct and intense physical pressures, where community structure is permanently altered, may never recover because new physical conditions will determine a different biological community (McCauley et al. 1977, van der Veer et al. 1985).

Despite these limitations, the estimation of recovery rates of the benthos is widely used as an indication of marine habitat sensitivity (e.g. Conlan et al. 2010, Shephard et al. 2010). Therefore it was considered a suitable approach to adopt for this thesis, as the findings could be directly compared with, and could build upon, a plethora of other studies which use this method.

There are parts of the UK seabed where more than one activity occurs (Eastwood et al. 2007). Coincidental activities can be relatively easily mapped and linked to habitat type. Yet a bigger question is how to estimate the cumulative impact these pressures represent. Habitat sensitivity, in terms of recovery rates of the benthic community from human activities, will influence the overall degree of impact. There are several possible methods for cumulative impact assessment that have been used in previous studies. These and the approach adopted for this thesis are described below.

1.5 Cumulative impact assessment

An assessment of cumulative effects first requires locations to be identified where there are several pressures, or a single pressure affecting a site repeatedly. Such spatial and temporal inquiry is facilitated by using a GIS. Just as there are several ways habitat sensitivity has been estimated, as described above, estimations of cumulative impact from coincidental activities have been conducted in a number of ways. For example cumulative impact has been estimated using modelling (e.g. Stelzenmüller et al. 2010), expert judgement (e.g. Teck et al. 2010), scoring indices (e.g. Ban et al. 2010), biological recovery rates (e.g. Kaiser et al. 2006), or combinations of these (e.g. Hall et al. 2008). For reasons explained in section 1.4, recovery rates of the benthic community from individual activities were used in this thesis and combined using hypothetical scenarios, as described below.

Cumulative impact may be additive or interactive in nature. Additive effects means the overall impact is the simple sum of the impact from individual pressures, whereas interactive effects occur when the response of the receiving habitat is non-linear. The latter could lead to overall effects that are less than (antagonistic effects), or more commonly greater than (synergistic effects), the sum of their parts (Smit & Spaling 1995, Cefas 2001). Although there are some data on the particular type of cumulative effect that is likely to result from specific combinations of coincidental human activities in certain habitat types, this is not the case across the matrix of all potential cumulative pressure-habitat groupings. The alternative is to use a range of scenarios to estimate total recovery time for pressure-habitat combinations, such as greatest, additive, antagonistic and synergistic. This allows determination of the sensitivity of the findings to different measures of impact estimation, for different habitat types.

1.6 Thesis structure

Chapter 2 presents a critical review of the diversity in approaches to aquatic environmental assessment (Foden et al. 2008). The motivation was to examine the different ways in which marine habitat condition is investigated, using examples from around the globe. Particular attention was paid to (a) the definitions of 'assessment', (b) the spatial and temporal scales, (c) the methodologies, and (d) the purpose and target audiences of the outcomes. The findings of the literature would provide a guide to best practice and inform the manner in which an assessment of individual and cumulative impacts on UK marine habitats would be conducted.

Marine aggregate extraction and benthic fishing are the two largest causes of physical disturbance to the UK seabed (Eastwood et al. 2007, BMAPA 2008, Foden et al. 2009, Foden et al. 2010a). Chapter 3 considers the distribution and impact of aggregate extraction on UK seabed habitats, in 2007. In a GIS, aggregate extraction activity and intensity were mapped and spatially linked to seabed landscape types. This enabled determination of footprint as a proportion of habitat. The sensitivity of habitats to this activity was determined by recovery rates of the benthic community, following cessation of extraction. Recovery rates were determined from a literature review of the physical and biological effects of aggregate extraction from marine habitats similar to those of the UK. Linking information on habitat recovery potential to marine landscapes and aggregate activity can provide a practical tool for use in marine spatial management.

Chapter 4 investigates the distribution of benthic fishing activity in 2007 and the sensitivity of habitats to this sector, by adopting a similar approach to that in Chapter 3 (Foden et al. 2010). Data were separated by towed bottom-fishing gears (scallop dredges, beam and otter trawls) and linked to habitat. A literature review provided estimates of recovery rates for habitat-gear combinations. Based on the distribution and frequency of benthic fishing, the proportion of seabed able to recover at 2007 levels of fishing effort could be estimated. Cumulative impacts were estimated for locations where aggregate extraction and benthic fishing activities coincided. To determine the most and least sensitive habitats total recovery time was estimated under four scenarios; greatest, additive, antagonistic and synergistic impacts.

In Chapter 5 the distribution of multiple human activities affecting the UK seabed, including aggregate extraction and benthic fishing, were mapped for 2007. Together these constitute four pressure types as defined by the MSFD (EU 2008a); smothering, abrasion, obstruction (sealing) and extraction. The spatial extent and intensity of individual and cumulative activities were quantified by habitat type. Cumulative impact was estimated using published recovery times under the same four scenarios (Foden et al. accepted).

Chapter 6 discusses the main conclusions of this work and suggests further research.

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CHAPTER 2

A critical review of approaches to aquatic environmental assessment

Foden J, Rogers SI, Jones AP (2008) A critical review of approaches to aquatic environmental assessment. Mar Pollut Bull 56:1825–1833 (see Declaration, page 2)

Abstract

As demands on aquatic resources increase, there is a growing need to monitor and assess their condition. This chapter reviews a variety of aquatic environmental assessments, at local, national, international and global scales and finds confusion in the terminology used to describe assessments. In particular the terms 'ecosystem' and 'integrated' are often misused resulting in lack of clarity. Therefore, definitions of some assessment terminology are suggested, consolidating existing proposals and simplifying future applications.

A conclusion from the review is that a new classification system for types of environmental assessments is required. The categorisation system proposed builds on preliminary work of the International Council for the Exploration of the Sea (ICES). Assessment classification is based on the environmental components considered, methodologies and nature of the linkages between components, and the inclusion or exclusion of socio-economic factors. The assessment terminology and categorisation system provided could in future simplify the way that assessments are defined and used to inform development of management strategies.

Key words: Environmental assessment; aquatic; assessment terminology; classification; review

2.1 Introduction

The diverse nature of aquatic environmental assessments and the lack of a coherent terminology to differentiate the various assessment types, are issues of concern raised by United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) (UNEP 2007) and the Working Group on Ecosystem Effects of Fishing Activities (WGECO) of the International Council for the Exploration of the Sea (ICES) (ICES 2005). In assembling reports, projects and programmes to provide an understanding of global and regional assessments concerning the marine environment, as well as the processes for undertaking these activities, the UNEP-WCMC "Assessment of Assessments" Group of Experts has begun to compile statistics on the diversity of assessments. Of the 258 examples currently listed, the group found the amount and detail of information contained was highly variable and in some cases quite limited: only 7% of entries are regarded as broad-based assessments, and 20% are classified as thematic (or narrow) assessments, focussing on particular features such as fisheries, biodiversity or specialised habitats (UNEP 2007). The purposes of the assessments were also wide ranging, as were the methodological approaches employed and the audiences at whom the assessments were targeted.

In conducting an environmental assessment, assessors need to pre-determine three components: their definition of what the 'assessment' is to comprise, the methods to be used, and the standards against which measured parameters are judged. Jennings (2005) supports the need to determine these components for fisheries assessment and highlights the importance of setting management objectives, identifying appropriate indicators, developing methods for monitoring, and setting reference points. So that management systems can improve, the need for repetition and review in the process is implicit. However many attempts at aquatic environmental assessment fail to define their terms, methods, or reference points, and their outcomes may thus depend upon the subjective viewpoint of the assessors or the commissioners of the assessment. Indeed the UNEP-WCMC group recognised the importance of defining 'environmental assessment' and stated their intention to devote particular attention to the development of a definition (UNEP 2007). Choi et al. (2005) is one of few examples of a marine environmental assessment that explicitly defines the term 'assess'.

After establishing the definition and terms of an environmental assessment, the next considerations are the choice of methodology and the standards against which a component or study area is to be judged. Adopting a recognised national or international framework with inherent and explicit protocols and standards has the advantages of openness, clarity and objectivity. For example, the ecological status of a water body can be assessed following the guidelines of the European Water Framework Directive (WFD) or eutrophication can be assessed under the Oslo Paris Convention's (OSPAR) Comprehensive Procedure. Conducting an assessment with an explicit purpose in mind, such as assessing North Sea cod stocks or assessing an estuary's ecological status, can result in information that translates into improved management, maximising the assessment's utility. A good example of this is the USEPA report on the United States' estuaries (Bricker et al. 1999). This study comprehensively assessed the scale, scope, and characteristics of estuarine nutrient enrichment and eutrophic conditions, using informal information and data brought together in a rigorous manner. The results contributed to the development of a national strategy to control nutrient enrichment problems affecting national coastal waters. For assessments to be of practical value they should help inform and guide the systems under which they are managed, leading to clear management responses.

This chapter critically reviews a variety of aquatic environmental assessments available in the scientific literature, at a range of scales, with two aims. The first aim is to identify the purpose of the assessments, together with an evaluation of the methods of analysis and presentation techniques which authors have employed. The second is to investigate and address the problems of diverse interpretation of key assessment terminology. To this end definitions are suggested which aim to consolidate existing proposals and a system is proposed for categorising assessments, building on the work of ICES WGECO. The assessment classification incorporates the purpose of the assessment, the environmental factors considered, the methodologies and nature of the linkages between components, and the inclusion or exclusion of socio-economic factors.

2.2 Methods

Aquatic environmental assessments were identified using computer database searches of specialist peer-reviewed research holdings, such as Scopus and ASFA, and general Internet search engines. Search terms included: aquatic, marine, environment, ecology, ecosystem, assessment, integrated, offshore, coastal, transitional, estuary, human impacts, sustainable, fishery, and management. The reference lists of identified studies were also reviewed for additional studies. Peer-reviewed academic papers and grey literature (e.g. case studies and government, government agency or industry published reports) were examined. Environmental assessments are often in non-peer-reviewed grey literature and difficulty can be encountered accessing such information. Studies were included if they provided an assessment of some aspect of an aquatic environment, be it biological, abiotic or both. Assessment was defined as an estimation of the value, magnitude or quality (Oxford Dictionary of English 2005) of the aquatic environment. The magnitude of an effect in the aquatic environment has a spatial or temporal component, the sizes of which can be defined and their impact on the ecological and environmental components can be determined.

To reflect the most recent work in this field, only assessments published since 2000 were reviewed. Studies that were included in the review were assessments of aquatic (generally marine) environments in a geographically specified location. Terrestrial environmental assessments were excluded, as were studies that investigated an aquatic environment without making either an objective or subjective assessment of its value, magnitude or quality. A total of forty studies were identified as relevant to the review, ranging in geographical extent from single estuaries to large, regional seas. The majority of the environmental assessment studies reviewed here were conducted in Europe (17) and North America (15), with only four from Australia, two from Far East, one from Africa and one from South America.

2.3 Motivations for undertaking assessments

Aquatic environments are assessed for a wide variety of reasons that are generally determined by the commissioners or authors of the assessment. The purpose of an assessment will dictate its size, scope, spatial extent and frequency. In general it was found that the aquatic environmental assessments reviewed herein had one of five main purposes and these are briefly discussed below.

The most common assessment relates to the sustainability of fish stocks and the purpose is the need to report on stock status, condition of habitats and threatened or declining species (ICES 2006a, Diaz-Uribe et al. 2007, DFO 2007a, Environment Australia 2007). Such assessments focus principally on species of concern and use historic data to analyse stock trends. Their spatial scope is determined by the geographical extent of the stock or the fishery being considered. These assessments are frequent, often conducted annually, and publications usually follow established formats.

A second purpose encompasses coastal and estuarine condition assessment for monitoring water quality impacts on biota. These are on the scale of individual countries, published using nationally recognised methods and standards. To this end, the United States Environment Protection Agency (USEPA) has begun to regularly assess their national coastal and estuarine condition (USEPA 2001, USEPA 2004, USEPA 2006) building on the pioneering US estuarine condition report by Bricker et al. (1999), and the USEPA's latest report is due in 2008. Similarly, Environment Australia regularly produces state of the environment reports (e.g. Barratt et al. 2001).

Increasingly, national governments and conventions are required to report on the condition of the whole marine ecosystem in response to human pressures. These large scale assessments are less frequent. They tend to draw on expertise from many parties, using information and data from several sectors, and cover a large spatial extent. For example, the UK's Department for the Environment, Food and Rural Affairs (Defra) commissioned 'Charting Progress' (Defra 2005), a national scale assessment to evaluate the state of the UK's seas, predominantly offshore on the continental shelf. This broad scale assessment included reports on environmental quality and processes, biology, habitats and climate. In an international context, a similar exercise was undertaken by

Contracting Parties to OSPAR in 2000, and a revised and updated Quality Status Report of the entire OSPAR area is to be published in 2010. More recently OSPAR members assessed eutrophication status in their maritime waters (OSPAR Commission 2003) and publication of the results for a second round of these assessments is due in 2008. The US Clean Water Act (CWA), formally known as the Federal Water Pollution Control Act of 1972, requires states, tribes and territories to monitor their waters and report biennially to the USEPA. It has evolved over the last decade attempting to shift from specifically physical, chemical and biological programmes to more holistic watershed-based strategies (Keller & Cavallaro 2008). The WFD requires European member states to assess ecological status in all their fresh, transitional (i.e. estuaries) and marine waters, with reporting cycles every six years. The newly developed European Marine Strategy Framework Directive (MSFD) (CEC 2005) has a similar objective to achieve good environmental status in European seas. It is a thematic strategy which will require status analyses of habitats, biology, physico-chemical and hydro-morphological characteristics, in marine waters on the seaward side of the baseline from which the extent of territorial waters is measured, extending to the outmost reach of the area where a Member State has jurisdictional rights. Member states will also have to include an economic and social analysis of use and cost of the marine environment; this review found such analyses to be rare.

A fourth purpose of assessment is the need to predict potential environmental impacts of future projects, programmes and policies. Acceptance of the need for this approach is widespread amongst European Union member states (Bond & Wathern 1999). For example, the European Environmental Impact Assessment (EIA) Directive requires an analysis of the likely effects of an individual programme on the environment (CEC 1985). As part of the more recent Strategic Environmental Assessment (SEA) Directive, there is an obligation for member states to carry out a SEA to ensure that environmental consequences of certain plans and programmes are identified and assessed during their preparation and before their adoption (CEC 2001). The spatial scope of a SEA is determined by the size of the proposed plan or programme and the area it is likely to affect. SEAs are conducted in response to plans such as offshore renewable energy generation (BMT Cordah 2003), aggregate extraction (East Channel Association 2003) or fossil fuel exploitation (DTI 2002). SEAs have been more widely adopted and the North Eastern Sea Fisheries Committee (NESFC) is in the process of conducting a non-mandatory pilot fisheries SEA for their shellfishery (Mott MacDonald 2008). The Habitats Directive requires an appropriate assessment to be made of any plan or programme, within a member state's territories, likely to have a significant effect on the conservation objectives of a site designated as a special area of conservation (CEC 1992). Member states have translated these Directives into national laws, or are in the process of doing so.

Finally, there are one-off assessments that are generally restricted in spatial extent, and often commissioned to investigate the impact of a particular activity in an estuary or a specific area of coastline. Widdows et al. (2007) and Bale et al. (2007), for example, carried out a detailed assessment of the effects of dredging in the Tamar estuary. Aubry & Elliott (2006) also conducted a single estuary assessment and investigated the use of environmental integrative indicators to assess seabed disturbance in the Humber. Mangi & Roberts (2006) investigated the effects of fishing gear on 11 Kenyan coral reefs. Diaz-Uribe et al. (2007) evaluated the trophic impacts of small-scale fisheries as a whole on the marine ecosystem of La Paz Bay, Mexico.

2.4 Analysis and presentation techniques

Table 2.1: Summary of analysis and presentation techniques used in assessments, with examples of three advantages and three disadvantages of each

Analysis and presentation technique	Advantages	Disadvantages
Plain text	• Simple to write • No technical expertise required of authors or readers • Easy to produce repeated reports (e.g. annual assessments)	• No visual representation of spatial data • Difficult to present complex spatial relationships • Difficult to cross-reference different aspects of an assessment
Plots of time-series	• Presentation of temporal trends • Simple for authors to produce and readers to interpret • Correlations between two datasets can be presented	• Difficult to illustrate complex relationships between multiple datasets • Relies on long-term datasets • Time-consuming to investigate all potential statistical relationships
GIS	• Spatial patterns are easily identified • Statistically robust integration of multiple spatial data layers is possible • Spatial data can be presented at multiple scales	• All data need geographical references • Requires a degree of expertise in compiling, analysing and presenting multiple data layers • 2-dimensional maps are not dynamic
Traffic-light system	• Provides readers with general overview of temporal change • Simplified presentation of potentially complex quantitative • data • Easy to track temporal changes	• Categorisation into colour scheme by authors may be subjective • Risk of losing fine-scale changes in a few broad categories • Can be influenced by the number and ranges of colour categories chosen
Computer-based modelling	• Possible to run multiple scenarios • Feedback mechanisms and loops can be built in to consider cumulative impacts • The user interface may be simplified for non expert users	• Computationally intensive requiring technical expertise • Data quality can affect final outcomes • Difficult to convey degree of uncertainty

The techniques used by authors for analysing and presenting results varied, depending on the quantity and complexity of data and the nature of their target audience. Five commonly used techniques found in the assessments reviewed are summarised in Table 2.1, with examples of advantages and disadvantages of each, and are briefly discussed below.

Plain text is possibly the simplest choice of technique, with discrete sections for each environmental factor or pressure considered. This approach has been used by Environment Australia (2003) and in some SEAs, e.g. BMT Cordah (2003) and East Channel Association (2003). However, it can be difficult for a reader to get an overall impression of all the environmental pressures affecting an area at one time and it is not intuitive (Table 2.1).

Plots of time-series data are commonly used, particularly for fish stock assessments. For example Allen et al. (2007), DFO (2007a), Diaz-Uribe et al. (2007), Environment Australia (2007) and González et al. (2007) all employ this method to present trends in fish catches over time. It is also a useful method for presenting temporal trends in environmental data, such as flow rates and suspended sediment (e.g. Bale et al. 2007), or NAO index (e.g. Widdows et al. 2007). This method is not ideal where multiple datasets are being analysed to identify potential statistical relationships (Table 2.1).

Geographical information systems (GIS) can provide important spatial context, for example by presenting a series of overlay maps and a final cumulative pressure or impact map. Table 2.1 summarises some of the potential advantages and disadvantages of GIS. Use of GIS is specifically recommended by the Countryside Council for Wales (CCW 2002) for cumulative effects assessment. GIS were used by Cefas (2001), BMT Cordah (2003), East Channel Association (2003), Danz et al. (2007), Derous et al. (2007) and Eastwood et al. (2007). One disadvantage of their application is that complex interactions can be difficult to display in two dimensions.

A ‘traffic light’ system has been used by authors in a variety of ways. A traffic light management approach is proposed by Caddy (2002) to judge the status of a fishery. As increasing numbers of limit reference points are infringed in the fishery, red lights accumulate and the quantity of these dictates the severity of the management response in terms of either quota or effort limitation. This response remains in effect until some or all of the limit reference points are no longer infringed, i.e. the red lights turn green again. Choi et al. (2005) also adopt a colour coded traffic light scheme, and the thresholds between colours are quantitative, representing anomalies in standard deviation units from long-term means. It is a method for providing the audience with a general overview of temporal changes, whilst attempting to maintain quantitative detail.

Choi et al. (2005) and Kenny et al. (2006) plot the principal component scores of multi-variate analyses using colour schemes. Link et al. (2002) summarise 5-year averages of abiotic, biotic, and human metrics to ascertain the qualitative status of ecosystem characteristics. Colour codes correspond to the different quintiles that these metrics exhibit with respect to the historical time-series. The traffic light approach has been adapted for the Charting Progress report (Defra 2005) to indicate whether the current environmental status is ‘acceptable’, ‘unacceptable’ or there is ‘room for improvement’. The judgements expressed in the report are subjective estimates based on available evidence and were reached in consultation with experts. Such colour coding schemes are useful for showing trends but interpretation is generally subjective and can be influenced heavily by the selection of components and the number and ranges of colour categories chosen (Table 2.1).

Computer based modelling systems are used by some authors. For example, Chang et al. (2008) utilise a dynamic Decision Support System (DSS). These models have a series of subsystems containing loops and feedback mechanisms to consider cumulative impacts. The advantage of such models is that whilst the user interface is straightforward with drop-down boxes and default variables, it is possible for users to explore the way the model has been constructed and to adjust algorithms and interactions built into the system if necessary. Computer based models can be very complex requiring technical expertise by either the developers of the model or the users. The advantages and disadvantages of this are summarised in Table 2.1.

2.5 Terminology and categorisation

Building on the WGECO's descriptions of the diverse types of assessment in the marine environment (ICES 2007), Figure 2.1 proposes six main categories of assessment (highlighted boxes) within a flow chart to illustrate how these categories have been identified. Further explanations of the key features of the six assessment types are given in Table 2.2, along with definitions of terms.

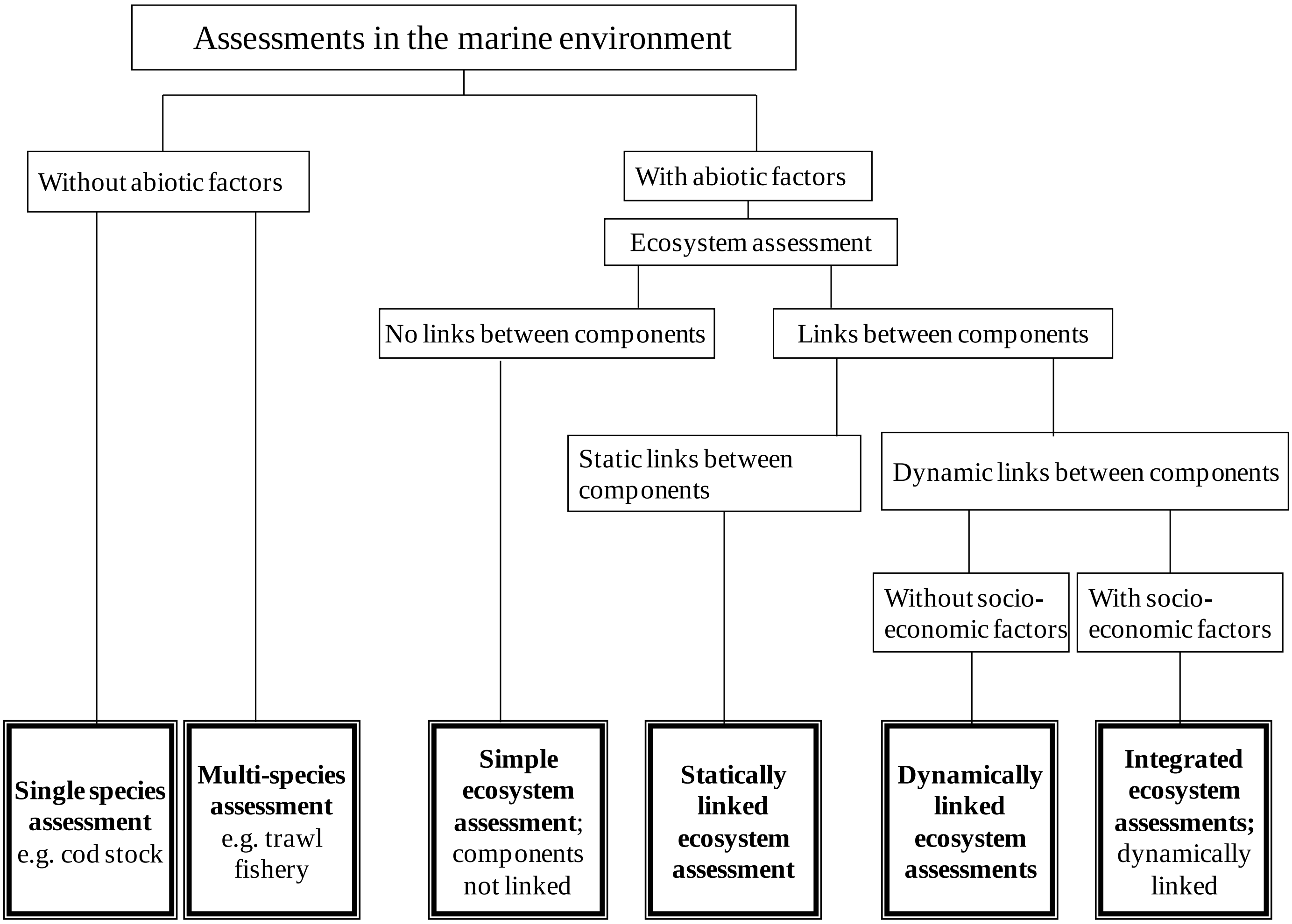

Figure 2.1: Aquatic environmental assessments, classified by their key features. The six assessment categories are in highlighted boxes

Table 2.2: Explanation of the key features of proposed aquatic environmental assessment types

<table>
<tr><th>Assessment type</th><th colspan="2">Key features</th></tr>
<tr><td>Single species assessment</td><td>Single component assessment, such as fish, of a single species; e.g. cod stock assessment

Single component assessment, such as fish, of multiple species; e.g. a mixed fishery assessment</td><td>These two categories include so-called ‘ecosystem’ assessments that omit the environmental (abiotic) components and consider only species and their interrelationships. In some instances abiotic factors may be included, but provide only limited background information. Consideration of the influences between abiotic and biotic factors is more properly an ecosystem assessment (see below)</td></tr>
<tr><td>Simple ecosystem assessment</td><td colspan="2">An ecosystem is the complex of a community of organisms (animals, plants, bacteria) and its environment (physical and chemical) functioning as a unit, as defined by UNEP (2007)

More than one trophic level is included; e.g. phytoplankton, zooplankton, molluscs and fish are all considered, along with abiotic factors. However the linkages between trophic levels or biology (biotic) and environmental (abiotic) factors are merely presented, possibly with some subjective and limited discussion of links</td></tr>
<tr><td>Statically linked ecosystem assessment</td><td colspan="2">The trophic levels and abiotic factors are analysed. The forcing and interactions between them are investigated quantitatively or qualitatively, but such linkages are considered as being ‘static’ because there is limited or no analysis of feedback mechanisms or cumulative effects. Qualitative links must be robust and rigorous, e.g. adhering to national or internationally approved assessment methodologies</td></tr>
<tr><td>Dynamically linked ecosystem assessment</td><td colspan="2">Dynamic links are likely to be modelled or calculated in algorithms, investigating multi-directional forcing and feedback mechanisms between the biota, environment and anthropogenic factors. This approach also involves an analysis of cumulative effects. Data usually presented mathematically or in geographical information system (GIS) maps

Cumulative effects:
• Additive effects means identifying areas where there are several pressures acting at the same time. The overall pressure is the simple sum of the individual pressures
• Interactive accumulation effects are the results of multiple activities accumulating non-linearly; i.e. causing lesser or more commonly greater effects than the sum of their parts (Smit & Spaling 1995, Cefas 2001, CCW 2002). These may be called compounding effects

Socio-economic factors do not inform the dynamic modelling</td></tr>
<tr><td>Dynamically linked, fully integrated ecosystem assessment</td><td colspan="2">An integrated assessment is a cohesive and comprehensive set of principles, criteria and methods, forming a linked, quantitative system (Leadbitter & Ward 2007). To be integrated, an ecosystem assessment should not only incorporate biotic and abiotic factors, but also socio-economic factors, with an analysis of how these factors interact (UNEP 2007) – i.e. the same type of assessment as dynamically linked ecosystem assessments, but with socio-economic factors incorporated

Socio-economic factors are the social, cultural, economic and political processes in and around the aquatic environment. These may include issues such as food security, livelihood opportunities, monetary and non-monetary benefits of resources and their equitable distribution, sustainable resource use, or local cultures’ perceptions and awareness of aquatic resources and processes</td></tr>
</table>

2.5.1 Review of environmental assessments

In this section the categories of assessment shown in Figure 2.1 and Table 2.2 are discussed in more detail, using examples of assessments drawn from the literature to illustrate the characteristics of each category. The purpose, authors, audience, available resource (e.g. funding, researchers, time) and potential impacts of an assessment report can all influence the choice of assessment methods, the criteria included and excluded and the interpretation of results. No single assessment system is likely to be perfect because of inherent uncertainties and complexity and it would be impossible for every conceivable parameter to be incorporated in a quantitative and relevantly weighted manner. Hence, choices have to be made to ensure an assessment is fit for its purpose. During this review it was found that many authors did not explicitly define the terms of their assessment; rather these were implied by the purpose, methods and components of the study. Others were misleading in their type-descriptions. This made the categorising process problematic and so classification of all assessments was based on their degree of similarity with one of the types shown in Figure 2.1, rather than relying on the definition provided by the authors.

2.5.1.1 Single or multi-species assessments

Whilst single or multi-species assessments can involve complicated algorithms, for example population or food web dynamics modelling, they were the most simple of those reviewed because of the absence of links with abiotic factors. These types of assessment were predominantly for commercial or non-target finfish and shellfish species. Such assessments are often regularly repeated and have the explicit purpose of aiding management practice, for example through the provision of advice with regard to the status of stock levels (e.g. ICES 2006a, ICES 2006b, Tidd & Warnes 2006, Pawson et al. 2007), setting total allowable catches (ICES 2006b, DFO 2007a) or establishing stock reference points and conservation limits (Environment Australia 2003, Braccini et al. 2006, DFO 2007a). Reports of current stock status usually involve trend analysis of previous years' data (e.g. Allen et al. 2007, DFO 2007a Environment Australia 2007) and may include predictions of future status, such as that of fish biomass in ICES reports (2006a, 2006b). For data-poor species, qualitative and/or semi-qualitative data can still be effectively used and modelled to help make competent management decisions (e.g. Caddy 2002). Expert knowledge has been used in Bayesian (e.g. Martin et al. 2005, Michielsens et al. 2006) and fuzzy logic modelling (e.g. Mackinson 2000, Cheung et al. 2005) and in hierarchical risk assessment approaches for data-poor species (e.g. Braccini et al. 2006).

The incorporation of broader ecosystem considerations has not yet become widespread in fisheries management (Link et al. 2002), although there have been some recent moves towards it. Whilst the majority of single or multi-species assessments do not include abiotic factors, a few examples were found which did. Mangi & Roberts (2006) for example examined the impact of fishing gear on fish and the level of structural (i.e. abiotic) damage to coral reefs, without considering other environmental data.

Two ICES reports (2006a, 2006b) present abiotic data on pollution trends, seabed topography, substrates, fishing activity related damage to reefs, circulation patterns, nutrients, temperature and salinity. Whilst these data provide the environmental context, the reports are essentially compiled based on a series of individual or multi-species fish stock assessments. The ecosystem overview is provided as useful contextual information, but not linked to biology. Other than specific stock data, all other parameters were discussed in general, using qualitative terminology. It could be argued that the environmental data presented in these ICES reports classify them as simple ecosystem assessments. However, their predominant focus on providing fish stock data precludes other ecosystem components which are needed to be considered as an ecosystem study.

Similarly, Diaz-Uribe et al.'s (2007) multi-species assessment of small-scale fisheries in La Paz, Mexico, compiled highly complex connections, utilising quantitative trophic mass balance models to analyse interactions between many biological components of the food web and the different effects of fishing gear on stocks. The study's classification as a multi-species assessment is justified by the absence of abiotic factors, which means however complex the linkages are between trophic levels it cannot be classed as an ecosystem assessment.

The single or multi-species reports reviewed did not include an analysis of the socio-economic factors, as defined by UNEP (2007), other than referring to the inherent economic incentive for commercial fishing. However, the ‘traffic light’ system for fisheries management proposed by Caddy (2002) will allow other indices to be incorporated (e.g. net earnings per trip), levels of by-catch of protected species, rising prices or rising demand for a product and other socio-economic factors.

2.5.1.2 Simple ecosystem assessments

An ecosystem approach to the management of human activities is increasingly becoming incorporated into international policy making, and it aims to manage these activities with greater consideration of ecosystem health and sustainable use (e.g. OSPAR Commission 2003, ICES 2006b, HELCOM 2007). The term ‘ecosystem’ is defined in Table 2.2, but some assessments were inappropriately described as such (see Section 3.1.1). An assessment that included only biota was a single or multi-species, rather than ecosystem, assessment; i.e. it was pertaining to the study of interrelationships between biotic organisms of the food web.

Of the ecosystem assessments reviewed, the simpler ones presented biotic and abiotic data but did not link these together. For example, Munawar et al. (2003) carried out what they termed to be an assessment of ecosystem health of a national marine park in Lake Huron. The authors applied a complex, multi-trophic suite of structural and functional techniques to evaluate the food web and measure the ecosystem’s health against pristine reference sites. Abiotic parameters were judged against advisory board guidelines and their presentation in the study sets the ecosystem context. However, there are no qualitative or quantitative analyses of linkages or forcing between abiotic parameters and the elements of the food web. Whilst this is a very thorough, holistic assessment of the entire pelagic food web, it is not integrated with the abiotic ecosystem. Furthermore, there is no discussion of socioeconomic pressures.

Eastwood et al. (2007) quantified the overall ‘footprint’ (i.e. spatial extent) of a number of direct, physical pressures on the seabed caused by humans. The footprint constituted the additive pressures of oil and gas exploration and production, windfarms, cables, aggregate extraction, waste disposal fishing and wrecks. As such the authors provide an environmental rather than an ecosystem assessment. The impacts of these pressures on biology and the environment would be needed for this to become an ecosystem assessment. Similar to Munawar et al. (2003), there were no linkages between the different pressures, rather they were examined separately and their additive effect was calculated. Nor were socio-economic factors included.

2.5.1.3 Statically linked ecosystem assessments

There was a divergence amongst the assessments reviewed which separated those that included links between their measured biotic and abiotic parameters, and those that did not (Table 2.2). Fish were the only biotic component included in all assessments. Other variables considered were plankton, seabirds, benthic fauna, macrophytes and marine mammals (e.g. DTI 2002, HELCOM 2007). Abiotic factors driving biological change included water quality elements such as nutrients, oxygen, temperature, water clarity and the North Atlantic Oscillation (NAO) (e.g. USEPA 2001, Widdows et al. 2007). These abiotic drivers were linked semi-quantitatively or qualitatively to the biota, without discussion of how feedback mechanisms may operate. Hence they were classed as static links.

The majority of statically linked ecosystem assessments were conducted in fulfillment of national (e.g. USEPA 2001, USEPA 2004, USEPA 2006, Defra 2005, DFO 2007b, DFO 2007c) or international legislation and conventions (e.g. HELCOM 2007) and are therefore constrained by the agreed protocols of each. The SEAs carried out by the DTI (2002), East Channel Association (2003) and BMT Cordah (2003) conform to both a European directive and UK national law. The characteristic features of these reports are discussed in more detail below.

Charting Progress (Defra 2005) described and evaluated monitoring data from the UK seas. It is organised into four sector reports which are then drawn together into eight regions. Expert judgement is used to present a ‘traffic light’ indication of the current status of the components. The variety of components examined makes this an ecosystem assessment, but with no quantitative connections between components and no analysis of cumulative effects or feedback mechanisms they are classed as statically linked.

Canada's regional seas reports (DFO 2007b, DFOc) are also ecosystem studies, containing biotic and abiotic components. However, they are not true assessments in that no judgements are made to the overall condition of the environments examined. Whilst there is some trend analysis of individual components (DFO 2007c), the links between components are static, in a similar manner to the Defra (2005) report. For example, the cause for the decline in shallow inventories of nitrate is only speculatively linked to changes in productivity, water column structure and influence of volume transport of the Labrador Current. As in the Defra report (2005) there is a lack of quantification, or consideration of cumulative effects and feedback systems.

The three USEPA reports (2001, 2004, 2006) are similar to the UK national reports in the nature of the linkages. A variety of biotic and abiotic ecosystem components constitute five primary indices of estuarine or coastal condition: water quality, sediment quality, habitat loss, benthic and fish tissue contaminants. These are assigned a rating and thresholds are set on the percentage of a region's coast in good, fair or poor condition for each index, similar to the 'traffic light' system used by Defra (2005). Overall condition is a weighted average of the regional index scores. The semi-quantitative connection between components still does not allow for consideration of cumulative effects and feedback mechanisms within the ecosystem. As such, there is more emphasis and information in the reports on the effects rather than the causes of water body condition.

The report for offshore windfarms by BMT Cordah (2003) is a typical example of a SEA. The assessment combines descriptive and quantitative information, converting the former into a semi-quantitative risk analysis, using a scoring system calculated from the consequence and likelihood of different scenarios, and the latter into a series of GIS maps. The term 'integrate' is used in the document to refer to the presentation of data as a series of GIS overlay maps; the maps were created to assist with the description of the existing environment and economic activity and to facilitate identification of areas of high and low constraints. The data are not dynamically linked in terms of the investigation and quantification of drivers and feedback mechanisms between the measured parameters. Cumulative impacts, in an additive (as opposed to interactive) form, are discussed. Results are qualitative and often subjective because of gaps in knowledge, and are based on professional judgement, supporting evidence, and expert opinion. Judgements are made of cumulative impact being ‘significant’, though what this means is not clearly defined. Rather than statistical significance, it is likely to be ecological or environmental, but no thresholds for significant impact have been defined. Other SEA reports (DTI 2002, BMT Cordah 2003, East Channel Association 2003) were similarly limited in detailed analysis of specific impacts.

The multi-sectoral approach by HELCOM (2007) in the Baltic Sea was the best example found of a national or international assessment which most thoroughly examines links between components. A broad range of parameters were measured under a series of thematic assessments (e.g. eutrophication, hazardous substances, biodiversity, conservation and maritime activities), against explicit targets and ecological objectives. The report cross-references between themes and objectives. For example failure to reach the objectives for eutrophication will impair the achievement of favourable status of biodiversity. At the same time the management objectives for airborne nitrogen emissions from shipping and nutrient inputs from ships’ untreated sewage are also relevant for reaching the objectives with regard to eutrophication. However, such cross-referencing still does not fully consider cumulative effects, nor did the report take into account feedback mechanisms between components of different thematic assessments.

Assessments that were carried out for non-legislative reasons generally covered a smaller geographic area. Widdows et al. (2007) focussed on the Tamar estuary and did not consider socio-economic factors. Aubry & Elliott (2006) did include socio-economics in the application of their proposed environmental indicators to the Humber estuary, as did Xue et al. (2004) in their case study of Xiamen harbour, China, and González et al. (2007) in an analysis of the artisanal longline hake fishery in the San Matías Gulf, Patagonia.

2.5.1.4 Dynamically linked and fully integrated ecosystem assessment.

Eight of the assessments included dynamic linkages that used models or algorithms to investigate multi-directional forcing and feedback mechanisms between the biota, environment and anthropogenic factors. Danz et al. (2007) aimed to create a tool for environmental research and management, modelling the effects of anthropogenic stressors on a variety of ecological variables, whilst Derous et al. (2007) deliberately excluded anthropogenic effects in order to model the intrinsic biological value (expressed as diversity) of marine zones to facilitate provision of a greater-than-usual degree of risk aversion in management of activities in such areas. The aim of the study by Kenny et al. (2006) was to understand the relationship between the causes of change at different scales so as to set targets for the management of human pressures. Consequently, it had a wider scope, modelling the interactions between anthropogenic effects, biota and abiotic components. The cumulative environmental impacts of marine aggregate extraction on fisheries and the seabed environment were investigated by Cefas (2001). All of these studies analysed cumulative impacts, either additively (Danz et al. 2007, Derous et al. 2007) or interactively (Cefas 2001, Kenny et al. 2006), but none incorporated the UNEP (2007) defined socio-economic factors.

The methods and factors that make an assessment ‘fully integrated’ (Table 2.2) informed only four of the reviewed assessments, probably because of the difficulty in establishing and calculating those indicators and setting thresholds against which to measure them. The ability of even highly integrated ecosystem assessments to indicate causality and to predict future scenarios can be highly dependent on the approach adopted by the assessors, which can have subsequent implications for management practices. Culp et al. (2000a, 2000b), Link et al. (2002), Choi et al. (2005) and Chang et al. (2008) all used multiple parameters, including socio-economics, to model present and future states, but their assessments were variable in terms of producing quantitative or qualitative conclusions. Culp et al. (2000a) monitored the anthropogenic, biotic, environmental, socio-economic and political components that best indicated the environmental state relative to objectives articulated by people living within their study area. In this way, these authors were able to propose cause-effect mechanisms of multiple anthropogenic stressors on the ecology. The components’ interactive effects were modelled and a framework was built allowing feedback and management responses (Culp et al. 2000b). Chang et al. (2008) developed a decision support system model, dynamically integrating socio-economic, environmental, biological and management sub-systems, to build scenarios and inform management strategies. Their model was designed to show a reasonable long-term trend rather than precise quantification.

In contrast to the above, Link et al. (2002) analysed more than 30 biotic, abiotic and human metrics, empirically and through statistical models. The conclusion of those authors, that many of the metrics are correlated, but that the strength and interdependence of those relationships was unknown, is an important one as it emphasises the complexity in conducting a baseline integrated assessment and drawing accurate conclusions regarding cause and effect of the component parts. A solution to this would be to develop a means of weighting the indicators, as in the USEPA reports (2001, 2004, 2006). However, caution needs to be exercised in assessing the interactive cumulative impacts of multiple pressures as they may accumulate non-linearly (Table 2.2). Mechanistic assessment models may be incapable of incorporating this feature.

The most spatially extensive marine ecosystem assessment was conducted by Choi et al. (2005), on the Eastern Scotian Shelf, Canada. The authors present an integrated analysis of temporal changes in anomalies of 55 primary and secondary biotic, abiotic and human parameters from their long-term mean. The parameters were ordered in the sequence of the primary gradient from a multi-variate ordination. Inspection of the sorted matrix identified changes over time and parameters were then colour coded to indicate the degree and direction of change using a ‘traffic light’ system (see Section 2.4). The study is multi-sectoral and includes socio-economic parameters, but with a strong focus on fisheries. The authors’ model is capable of identifying the forces contributing to the stability of the alternate state including both top-down processes and bottom-up processes. The improved comprehension of the dynamics and contributing factors in the shelf system generated by these authors allows for potential early-warning indicators of systemic change to be identified and exposed, thereby better informing the decision-making processes aiding effective management to avoid deterioration and to work towards improving the ecosystem’s health.

2.6 Discussion

Assessment of aquatic ecosystems is necessary because of the near ubiquity of human pressure on them (Glover & Smith, 2003). Managers and conservationists need to identify the most vulnerable ecosystems in order to prioritise mitigation of the pressures to which they are subject. For these reasons it is important for an assessment to have a clearly defined purpose and to use accurate terminology. This chapter has provided an approach to more accurate usage of assessment terminology and a system of categorisation, which could in future simplify the way that assessments are defined and used.

The majority of the reviewed assessments did specify their terms of reference, purpose and methodology. In particular, the single and multi-species fish stock assessments reported extant condition against historic data and scientifically established reference levels and thresholds. This work was predominantly carried out by government sponsored agencies, or international bodies, such as ICES, forming part of long-established annual reporting cycles to provide stock advice for management purposes. However, there was a high degree of variability in the interpretation of some terminology, with the term ‘integrated’, as defined in Table 2.2 (Leadbitter & Ward 2007, UNEP 2007), being the most misrepresented.

Leadbitter & Ward (2007) identified the following features for effective assessment systems: comprehensiveness (including stock condition, environmental impacts, and socio-economics), transparency and accountability, and the nature and quality of data and information. However, there are numerous difficulties in creating an assessment system that adequately achieves all these. Whilst there are scientifically accepted and robust, quantitative methods of stock assessments (e.g. Environment Australia 2003, Braccini et al. 2006, DFO 2007a), references to the inclusion and measurement of other factors which will make an assessment comprehensive are more elusive. Some assessments did not have quantitative thresholds, with Danz et al. (2007) and Derous et al. (2007) employing relative values. To quantitatively determine how environmental resources will respond to additional impacts requires knowledge of the ‘carrying capacity of the environment’ (Cefas, 2001). The carrying capacity can be defined in environmental terms as the maximum number of organisms that can be supported in a given area (Cohen 1997), or in socio-economic terms as ecosystem goods and services (Elliott et al. 2007). However, determining at what point an activity such as aggregate extraction might reach a critical intensity beyond which, for example, a fish population may be reduced to sub-commercial levels, is difficult to establish. This is because there is often no basis for presuming a simple additive relationship for example between dredging activity and the size of stock (Cefas, 2001). Objective conclusions cannot easily be drawn from these studies, limiting their usefulness in the development of management strategies in the short term. The value of such studies, however, is they help establish baseline data from which spatio-temporal change can be monitored and cumulative impacts can be quantified in the future.

In other studies environmental impacts were measured and judged against standards established under the auspices of national or international policies and directives (e.g. OSPAR, WFD), a process having the advantage of transparency and objectivity. The greater difficulty comes in gathering the necessary data for accurately using such external standards. Openness in initially establishing the system to be used, and openness of the results to peer-review scrutiny will go a long way to reaching the goal of a thorough, scientific ecosystem assessment. It is noteworthy that much material for this review is from non-peer-reviewed grey literature and it is not always clear whether there has been a process of openness to public or peer scrutiny in the assessment. The public and stakeholder involvement in some of the assessments reviewed (e.g. DTI 2002, BMT Cordah 2003) is a positive move towards transparency and accountability.

Link et al. (2002) conclude that a suite of metrics is required to assess a marine ecosystem and limiting an assessment to just a few components may be misleading. The number of components used in the assessments reviewed varied between one, such as single fish stock assessments (e.g. Environment Australia 2003, DFO 2007a), and >100 (Danz et al. 2007). Points of reference and reference direction can be identified but it may be difficult to inform managers of the magnitude of required changes. Furthermore, greater complexity is not necessarily better because it can become more difficult to unpack the information provided in multi-parameter integrated assessments, so as to identify discrete problems and failings which need tackling through improved management. This adds to the difficulty of compiling an all-encompassing assessment and helps explain their relative scarcity. The series of papers written or edited by Culp et al. (2000b) is probably the most comprehensive integrated ecosystem assessment reviewed, whilst remaining amenable to identification of specific problems, around which management strategies can be designed.

As degradation of the aquatic environment is escalating on a global scale (UNEP 1982, Choi et al. 2005, Norkko et al. 2006, Halpern et al. 2008), there is a continuing need to develop improved and cost-effective ways to study and monitor these ecosystems, and assess the ecological significance of change (Norkko et al. 2006). Required methods include, *inter alia*, population and community studies, population and ecosystem modelling and biochemical techniques (GESAMP 1995). In this context the establishment of terms of reference, accurate terminology, clarity in the presentation of results and a focus on informing the development of management strategies for assessments, are imperative. At present these imperatives are not being met for many assessments undertaken, limiting the utility of their findings.

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CHAPTER 3

Recovery rates of UK seabed habitats after cessation of aggregate extraction

Foden J, Rogers SI, Jones AP (2009) Recovery rates of UK seabed habitats after cessation of aggregate extraction. Mar Ecol Prog Ser 390:15–26 (see Declaration, page 2)

Abstract

Marine aggregate extraction and benthic fishing are the two largest causes of physical disturbance to the UK seabed. Aggregate dredging is a damaging but highly spatially heterogeneous pressure, with a footprint <1 % of bottom fishing (2001–2007). To understand the impacts of aggregate extraction, international literature was reviewed for recovery rates of the seabed following cessation of dredging, in a range of habitats, hydrodynamic conditions and dredge intensities. Physical recovery (T_{Phys}) and biological recovery (T_{Bio}) were determined as the mean time-period for recovery to pre-dredge or reference site conditions. Recovery times were then estimated for marine landscapes targeted by the aggregate industry in UK waters – maintenance dredging data were not included.

Aggregate extraction affects less than one percent of most landscapes and six percent of estuarine area. Ninety-six percent of extraction occurred in sand or coarse sediment. Fifty percent targeted coarse sediment plains-moderate tidal stress, which had the longest period of T_{Phys} (20 years) and the second longest T_{Bio} (8.7 years). Shallow coarse sediments-weak tidal stress had the longest mean T_{Bio}, 10.75 years, but 21 % of the habitat supported high intensity dredging. The most intense dredging (>90 hours) was in estuaries, which have the shortest recovery times – T_{Phys} 0.17 and T_{Bio} 5.25 years. At present licensed areas do not appear to be located to avoid the most sensitive marine landscapes, nor to target the least sensitive. Linking information on habitat recovery potential to marine landscapes and aggregate activity provides a practical tool for use in marine spatial management.

Key words: Geographical Information Systems (GIS); marine aggregate extraction; physical recovery; biological recovery; marine landscapes; UK; trawling

3.1 Introduction

As demands on marine resources increase there is a growing need monitor and assess the condition of the marine environment. Understanding and quantifying pressures and impacts from major human activities is necessary to underpin effective environmental impact assessment and marine planning, and to provide the basis for integrated marine management (OSPAR Commission 2003, Eastwood et al. 2007, Borja et al. 2008, Foden et al. 2008, Halpern et al. 2008). Indeed, current political commitments require countries to consider an ecosystem approach to marine planning and management; e.g. the World Summit on Sustainable Development (UN 2002), Fisheries Management (FAO 2003), creation and maintenance of networks of marine protected areas (MPAs) under the UN Regional Seas Programme (UNEP-WCMC 2008), European Directives (EU 2000, EU 2005) and the UK Marine Bill (Defra 2007). While some descriptions of the spatial extent of pressures are available (e.g. Eastwood et al. 2007), the lack of spatially resolved information on ecosystem attributes and their response to pressure makes it difficult to quantify impacts. It is also necessary to account for additive and synergistic interactions between pressures, and other complex forms of cumulative impacts (Smit & Spaling 1995, Cefas 2001, Oakwood 2002, DTLR 2002, Foden et al. 2008).

Of the direct physical pressures causing disturbance to the seabed, bottom-trawling for fish is amongst the most widespread. More than 50 % of many shelf habitats are trawled annually (Hall 2002, Watling & Norse 2008) and in the UK up to 55,500 km^2 (21.4 %) of the seabed is affected annually (Eastwood et al. 2007, Stelzenmüller et al. 2008). Aggregate dredging for mineral resources in the UK is the second most widespread pressure, with approximately 135–223 km^2 of seabed dredged annually, 1998–2007 (BMAPA 2008). Although demersal fishing is a much more widespread pressure than aggregate extraction, there are some general similarities in their impacts on the physical environment and the ecology of the seabed (Hall 1994, Kaiser et al. 2006, Diesing 2007, Halpern et al. 2008). These impacts can be quantified by measuring the physical and biological recovery time of each habitat that is affected following a pressure of defined magnitude and duration (Hiddink et al. 2007). The period needed for physical recovery of the seabed depends on hydrodynamics, sediment particle size and the intensity of the activity. The length of time that furrows, depressions or mounds remain as distinctive features depends on the ability of tidal currents or wave action to erode crests or transport sediments into them (Newell et al. 1998). Except in areas of mobile sands, this process is slow and even the strongest currents are unable to transport gravel from adjacent areas (Millner et al. 1977, Newell et al. 1998, Desprez 2000). Such alterations to the seabed, especially changes in sediment characteristics, potentially affect the biological recovery of impacted sites (de Groot 1986).

The more resilient communities of naturally dynamic environments are less sensitive to stress than those of more benign, less variable environments such as deep water, coarse sand and gravel (Bax & Williams 2001, Bolam & Rees 2003, Hiddink et al. 2007). In general they show more rapid recovery, though exceptions have been noted (e.g. Kenny et al. 1998). Some sites subjected to direct and intense physical pressures, where community structure is permanently altered by changes in the physical environment, may never recover (McCauley et al. 1977, van der Veer et al. 1985). Although the resulting community may return to its pre-impact abundance levels, it may never regain its structure and internal integrity (McCauley et al. 1977). In addition, the seabed is a dynamic environment, with naturally changing faunal assemblages (Matthews et al. 1996). If the fauna available to re-colonise a disturbed area of seabed have undergone natural change a pre-impact community structure may be replaced by a sustaining ecological succession. That is, at an abandoned site a new suite of species becomes consistently abundant over time, either through breeding or repeated settlement from pelagic larvae (Ellis 2003).

Marine aggregate extraction is a spatially heterogeneous activity that affects both the physical environment and the ecology of the seafloor, and the degree of impact is a function of the environment that is dredged and the intensity and longevity of extraction effort. The UK is one of the largest producers of marine aggregate in the world, and 23.1 million tonnes of sand and gravel were extracted from 134.7 km^2 of English and Welsh seabed in 2007 (BMAPA 2008). The main method of dredging in UK waters is trailer dredging of evenly distributed deposits (BMAPA 2006), producing shallow linear furrows approximately 1–3 m wide and 0.2–0.3 m deep per pass (Kenny & Rees 1994). A limited amount of static anchor dredging also occurs, where thick localised reserves are exploited to leave saucer-shaped depressions, typically up to 8–10 m deep, but that may reach up to 20 m in depth (Dickson & Lee 1972, Newell et al. 1998, Boyd et al. 2004).

This chapter describes the temporal and spatial variation in aggregate extraction pressure and impact in UK marine waters since 2001. To quantify recovery periods, a review was undertaken of published literature on physical and biological impacts and rehabilitation following cessation of dredging. Results of the review were then used to estimate recovery times for the types of marine landscape targeted by the aggregate industry in UK waters. A brief comparison is made with impacts on the seabed caused by fishing with demersal gear.

3.2 Method

3.2.1 Literature review of marine aggregate extraction impacts

A review was conducted of scientific literature which measured recovery time following cessation of dredging. Studies were identified using computer database search engines of peer-reviewed literature, such as Scopus and ASFA, and general Internet search engines. Search terms included: marine, aggregate, extraction, dredging, intensity, benthos, seabed, physical, environmental, biological, habitat, recovery, rehabilitation and colonisation. The reference lists of identified publications were also reviewed for additional studies. Peer-reviewed academic papers and grey literature (e.g. case studies and government, government agency or industry published reports) were examined. Each reference was given one of six 'quality of evidence' categories, based on Pullin & Knight (2003). These represent a decline in quality of evidence, based on the type of research undertaken: Category I, randomised controlled study; II-1, controlled study without randomisation; II-2, comparison of differences between sites with and without controls; II-3, multiple time series evidence; III, qualitative field evidence, descriptive studies or expert opinion; IV, inadequate evidence.

Physical recovery from aggregate dredging, T_{Phys}, was considered complete when previously extant dredge tracks and scours were no longer detectable by imaging techniques and where sediment composition was similar to either pre-dredge conditions or local reference sites (Boyd et al. 2004). Biological recovery, T_{Bio}, was defined as the establishment of a community that was virtually indistinguishable from surrounding, non-impacted reference sites, determined using both uni- and multi-variate analysis techniques (Cooper et al. 2005). Where numeric data were available to describe species number, abundance, and/or biomass in pre-dredge and post-dredge conditions, a return to 90 % of the values was considered to indicate recovery (Hiddink et al. 2006). The periods of time needed for T_{Phys} and T_{Bio} to take place after cessation of dredging were identified for each extraction site.

To categorise dredge sites into marine landscape types (Connor et al. 2006) (Table 3.1), data on sediment grain size, depth, hydrodynamic regime and dredge intensity were quantified and standardised. Descriptive sediment types were converted into grain size diameters (mm) using the Wentworth (1922) scale, and dredge intensity values were standardised to a rate of extraction, expressed as tonnes (t) km^{-2} yr^{-1}. HELCOM's recommended conversion factors were used for converting m^3 to t (Schneider 1996). Authors of the reviewed literature recorded hydrodynamic regimes in differing ways; for example as near-bottom current shear-stress in Newtons m^{-2} (N m^{-2}), as depth-integrated current velocity, or velocity 1 m above the bed (m s^{-1}). To standardise these data, current velocities were converted to N m^{-2} at 1 m above the seabed. Where authors provided depth-integrated velocity, the velocity at 1 m above the bottom was calculated using Soulsby (1997 p 52) and converted to tidal stress N m^{-2} for different bottom sediment types (Soulsby 1997 p 55). These standardised near-bed stress values were then grouped into three categories; weak tidal stress = 0–1.8 N m^{-2}; moderate = 1.8–4.0 N m^{-2}; and strong >4.0 N m^{-2} (Connor et al. 2006).

Where a range of years was given by the authors for recovery period of a habitat, T_{Phys} and T_{Bio}, the upper limit was used as a precautionary approach. Where recovery was reported in the literature as 'decades', a value of 20 years was chosen, which is supported by unpublished biological recovery data (Sánchez-Moyano in Guerra-García et al. 2003).

Table 3.1: Marine landscape types in UK waters, targeted by the aggregates sector. Wave base is 50–70 m. Tide stress; weak = 0–1.8 N m^{-2}, moderate = 1.8–4.0 N m^{-2}, and strong >4.0 N m^{-2}. UKCS = UK continental shelf (from Connor et al. 2006)

Marine landscape type	**Abbrev.**	**Substratum**	**Depth range (m)**	**Tide stress (currents)**	**Slope and additional descriptors**	**Area (km^2)**	**Proportion of total UKCS (%)**
Estuary	ES	Mainly soft sediment, limited rock	0–30m	Variable–moderate	Strong salinity gradient from riverine	2,881	0.3
Shallow coarse sediment plain-weak tidal stress	SCSW	Coarse sediment	Coastline to wave base	Weak	Negligible slope (<2 %)	33,694	3.9
Shallow coarse sediment plain-moderate tidal stress	SCSM	Coarse sediment	Coastline to wave base	Moderate	Negligible slope (<2 %)	16,745	1.9
Shallow coarse sediment plain-strong tidal stress	SCSS	Coarse sediment	Coastline to wave base	Strong	Negligible slope (<2 %)	7,869	0.9
Shallow mixed sediment plain-moderate tide stress	SMSM	Mixed sediment	Coastline to wave base	Moderate	Negligible slope (<2 %)	2,021	0.2
Shallow sand plain	SS	Sand and muddy sand	Coastline to wave base	Variable	Negligible slope (<2 %)	48,218	5.5
Shelf coarse sediment plain-moderate stress	SHCM	Coarse sediment	Wave base to 200 m	Moderate	Negligible slope (<2 %)	17,433	2.0
Shelf coarse sediment plain-strong tide stress	SHCS	Coarse sediment	Wave base to 200 m	Strong	Negligible slope (<2 %)	2,840	0.3
Shelf mixed sediment plain-moderate tide stress	SHMM	Mixed sediment	Wave base to 200 m	Moderate	Negligible slope (<2 %)	2,260	0.3
Shelf sand plain	SHSP	Sand and muddy sand	Wave base to 200 m	Variable	Negligible slope (<2 %)	215,215	24.7

3.2.2 Study area and data

Electronic Monitoring Systems (EMS) have been fitted to all aggregate dredgers operating in UK (England and Wales) waters since 1993 to automatically record position and 'dredging status' every 30 seconds. It would have been desirable to also analyse data on maintenance dredging of shipping lanes and estuaries. However, the maintenance process involves the removal of recent unconsolidated sediments, as opposed to the mineral deposits targeted by the aggregate sector. Furthermore, maintenance dredging data were not available for inclusion. Annual EMS data (hours dredged per year) from the UK Crown Estate were provided in 50 x 50 m (2500 m^2) cells. EMS data were clipped to the relevant licence boundary for each year using the ArcGIS Geographical Information System (ESRI Inc), to remove out-of-area dredge events and incorrectly transmitted EMS codes. The annual spatial records of all areas dredged were then combined and a cumulative dredging footprint was created in the GIS for the period 2001–2007, inclusive.

3.2.3 Spatial and temporal pressure of UK marine aggregate extraction

The locations of aggregate dredging activity were spatially joined to UK marine landscapes characterised by a combination of sediment type, depth and tidal stress (Connor et al. 2006). The categories listed in Table 3.1 are those in which aggregate dredging occurs in the UK. Each landscape's extent and the proportion of the UK continental shelf it constitutes are also given. Using the results of the literature review, physical and biological recovery rates, T_{Phys} and T_{Bio}, were estimated for those UK marine landscapes affected by aggregate extraction.

The intensity of dredge effort in the UK is measured as hours dredged annually (h yr^{-1}). Only a very general approximation can be made of quantities dredged because production rates of the ~30 vessels operating in UK waters vary depending upon the pump size, power and age, and vessel capacity (currently 880–8800 t). This is further complicated by environmental variables such as water depth and the seabed sediment composition which also affect efficiency, so that a vessel's extraction rate can vary each trip. High intensity dredging is defined by the aggregate extraction industry as >1.25 h yr^{-1} (BMAPA 2008) and low intensity dredging is <1 h yr^{-1} (Boyd et al. 2004, Cooper et al. 2005, 2007a). Larger dredgers are able to remove up to 5,000 t in three hours (BMAPA 2006), which would equate to >2,100 t at high intensity dredging, or <1,700 t at low. However, this would be a gross over-estimate for smaller vessels.

As details of individual sea-trips for every vessel were not available, the number of hours dredged was used as a proxy for intensity. Intensity of dredge effort was available only as categorical data, so the mid-point of each category was used (Table 3.2). For each cell the dredge duration mid-points for all seven years were summed to estimate cumulative dredging time.

Table 3.2: Annual dredge duration time ranges and calculated mid-points used to describe dredge intensity

Time range (decimal hour)	**Mid-point (decimal hour)**
0.01–0.24	0.12
0.25–0.49	0.37
0.50–0.99	0.75
1.0–1.49	1.25
1.5–2.49	2.00
2.5–4.99	3.75
5.0–7.49	6.25
7.5–9.99	8.75
10.0–12.49	11.25
12.5–14.99	13.75
15.0–17.49	16.25
17.5–19.99	18.75
20.0–29.99	25.00
30.0–40.00	35.00
40.0–50.00	45.50

*Table 3.3: Summary of (a) physical recovery (T_{Phys}), and (b) biological recovery (T_{Bio}) rates for different habitat types, following cessation of dredging. Descriptions of 'Hydrodynamics'; weak = 0–1.8 N m^{-2}, moderate = 1.8–4.0 N m^{-2}, and strong >4.0 N m^{-2}. * = estimated value, n/a = data not available. Quality of evidence categories based on Pullin & Knight (2003); categories I–IV represent declining strength of evidence.*

(a) Physical recovery following cessation of dredging

Habitat type	Hydrodynamics (tidal stress)	Depth (m)	Marine landscape	Intensity; rate dredging (t km^{-2} yr^{-1})	Area (km^2)	T_{Phys} (years)	Examples	References (& quality of evidence category)
Fine sand	Strong tidal current estuaries	Shallow subtidal <10	ES	617,500 mean rate	~ 1*	1–3	Waddenzee – subtidal channels, NL. Fastest T_{Phys} in fastest current flows.	van der Veer et al. 1985 (II-2).
	Low tidal current estuaries	Just below LW	ES	1,045,000 mean rate	~ 1*	1	Waddenzee – Oosterbierum tidal watershed, NL.	van der Veer et al. 1985 (II-2).
	Non-tidal, energetic wave dominated	10–12	SS	552,900	1.1	7–8	Wustrow, Baltic Sea – 'high intensity'.	Diesing et al. 2006 (II-2). Diesing 2007 (II-3).
	Non-tidal, weak wave environment	19–21	SS	329,300	0.6	Decades	Tromper Wiek Ost, Baltic Sea.	Diesing et al. 2006 (II-2).
	Very weak seasonal erosion	40–42	SS	1,520,000	1.0	>2.5	Northern Adriatic.	Simonini et al. 2005 (II-1). Simonini et al. 2007 (II-1).
Fine to medium sand	Seasonally strong tide & wind driven current	20–23	SS	2,850	1.4	>4	Terschelling, NL.	van Dalfsen et al. 2000 (II-2).
	Energetic wave dominated, non-tidal	8–10	SS	178,125	1.6	~ 0.5	Graal-Mürtiz I, Baltic Sea.	Diesing et al. 2006 (II-2).
Medium sand	Strong	4 – to top of bank	SS	23,000	151.8	0.5	Kwintebank Zone 2 (2001 max intensity year).	Vanaverbeke et al. 2000 (II-2). Degrendele et al. 2007 (II-3).
Sand	Strong and moderate–strong	<20	SS & ES	n/a	n/a	≤ 1	Bristol Channel, UK.	Newell et al. 1998 (III).

[1]Morphological recovery rate from dredging furrows only. There are considerable changes in sediment characteristics, but a causal relationship with sand extraction is unproven for the whole area (Vanaverbeke et al. 2007)

Table 3.3 continued

Habitat type	Hydrodynamics (tidal stress)	Depth (m)	Marine landscape	Intensity; rate dredging (t km^{-2} yr^{-1})	Area (km^2)	T_{Phys} (years)	Examples	References (& quality of evidence category)
Medium to coarse sand	Wave dominated delta front with seasonally strong winter storms	15	SS	~ 570,000*	1.5	1–1.5	Tordera river, Catalan, western Mediterranean. Dredge depth 20 cm*.	Sardá et al. 2000 (II-2).
Very coarse sand	Weak–moderate	27–35	SCSW	733,300	0.3	Decades	Thames Area 222, UK, southern North Sea.	Boyd et al. 2004 (I). Cooper et al. 2007a (I).
Sand and sandy gravel	Weak	20–25	SCSW	Up to 365,000	2.6	>5	Coal Pit, Area 408, UK.	Coastline Surveys Europe 2002 (I). Cooper et al. 2005 (I). Robinson et al. 2005 (II-1).
Coarse sandy-gravel	Non-tidal, energetic or mixed-energy wave dominated	9–13	SCSW	91,500	0.4	5–10	Tromper Wiek I, Baltic.	Diesing et al. 2006 (II-2).
	Moderate	15–21	SCSM	960,000	1.35 (in 1996)	Decades	Hastings Shingle Bank Area X, UK.	Boyd et al. 2004 (I). Cooper et al. 2007a, (I).
		16–25	SCSM	400,000	3.1	Decades	Hastings Shingle Bank Area Y, UK.	Boyd et al. 2004 (I). Cooper et al. 2007a (I).
Gravel	Moderate–strong	12–46	SCSS	75,000	107.0	~ 4	Cross Sands, East Anglia, UK – conglomerate of several dredge sites.	Kenny & Rees 1994 (I), 1996 (I). Kenny et al. 1998 (I). Cooper et al. 2007b (II-1), 2008a (II-3).
Mixed – mud to gravel	Moderate	20–30	SMSM	n/a	n/a	>4	Suffolk coast, UK.	Millner et al. 1977 (I).
	Moderate–weak	28–34	SMSM	80,000	6.1	Decades	Southwold, Area 430, UK.	Andrews Survey 2004 (I). Newell et al. 2004b (I). MES 2007 (I).

[2]Changed sediment type has slowed recovery (Newell et al. 2004)

Table 3.3 continued

(b) Biological recovery following cessation of dredging

Habitat type	Hydrodynamics (tidal stress)	Depth (m)	Marine landscape	Intensity; rate dredging (t km^{-2} yr^{-1})	Area (km^2)	T_{Bio} (years)	Example	References
Fine sand	Strong tidal current estuaries	<20	SS & ES	n/a	n/a	0.5–0.75	Bristol Channel, UK.	Newell et al. 1998 (III).
		Shallow subtidal <10	ES	617,500 – mean rate	~ 1*	>1– >3	Waddenzee – subtidal channels, NL. Fastest recovery in fastest current flows.	van der Veer et al. 1985 (II-2). Newell et al. Table 6, 1998 (II-2).
	Low tidal current estuaries	Just below LW	ES	1,045,000 – mean rate	~ 1*	5–10	Waddenzee – Oosterbierum tidal watershed, NL.	van der Veer et al. 1985 (II-2). Newell et al. Table 6, 1998 (II-2).
	Non-tidal, energetic wave dominated	10–12	SS	552,900	1.1	>1	Wustrow, Baltic Sea.	Krause et al. 2010 (I).
	Non-tidal, weak wave environment – with infrequent seasonal storms	16–20	SS	3,230	1.5	5–10	Costa Daurada, Mediteranean.	van Dalfsen et al. 2000 (II-2).
	Very weak seasonal erosion	40–42	SS	1,520,000	1.0	>2.5	Northern Adriatic.	Simonini et al. 2005 (II-1). Simonini et al. 2007 (II-1).
Fine to medium sand	Seasonally strong tide & wind driven current	20–23	SS	2850	1.4	4	Terschelling, NL.	van Dalfsen et al. 2000 (II-2).
Medium to coarse sand	Wave dominated delta front with seasonally strong winter storms	15	SS	~ 570,000*	1.5	11–14 (for full bivalve growth)	Tordera river delta, Catalan, western Mediterranean. Dredge depth 20 cm*.	Sardá et al. 2000 (II-2).
	Seasonally strong tide & wind driven current	16–18	SS	950	0.5	4	Torsminde, Denmark.	van Dalfsen et al. 2000 (II-1).

Table 3.3 continued

Habitat type	Hydrodynamics (tidal stress)	Depth (m)	Marine landscape	Intensity; rate dredging (t km^{-2} yr^{-1})	Area (km^2)	T_{Bio} (years)	Example	References
Very coarse sand	Weak–moderate	27–35	SCSW	733,300	0.3	Decades	Thames Area 222, UK, southern North Sea. High intensity extraction.	Boyd et al. 2004 (I). Cooper pers. com. (III).
Sand and sandy gravel	Weak	20–25	SCSW	Up to 360,000	2.6	>10	Coal Pit, Area 408, UK.	Coastline Surveys Europe 2002 (I). Cooper et al. 2005 (I). Robinson et al. 2005 (II-1). Cooper pers. com. (III).
Coarse sandy-gravel	Moderate	15–21	SCSM	960,000	1.35 (in 1996)	>7	Hastings Shingle Bank Area X, UK.	Boyd et al. 2003 (II-1), 2004 (I), 2005 (I). Cooper et al. 2005 (I), 2007a (I), pers. com. (III).
		16–25	SCSM	400,000	3.1	8–9	Hastings Shingle Bank Area Y, UK.	Boyd et al. 2004 (I). Cooper et al. 2005 (I), 2007a (I).
	Moderate–weak	27–35	SCSM	33,000	0.3	7	Thames Area 222, UK, southern North Sea. Low intensity extraction.	Limpenny et al. 2002 (II-1). Boyd et al. 2003 (II-1), 2004 (I), 2005 (I). Cooper et al. 2005 (I), pers. com. (III).
	Weak	18–20	SCSW	65,000	7.1	4	Off Humber estuary, Area 106, UK.	Andrews Survey 2004 (I). Newell et al. 2004b (I).
Gravel	Strong	15	SCSS	67,000	1.5	~ 3	Dieppe, English Channel.	Desprez 2000 (II-1).
	Weak	30–40	SCSW	n/a	n/a	>2	Klaverbank, Dutch North Sea.	van Moorsel & Waardenburg 1991 (II-1). van Moorsel 1993 (II-1), 1994 (II-1).
Mixed – mud to cobbles & stones	Moderate	10	SMSM	150,000	1.0	3	East of Isle of Wight, Area 122, UK.	Newell et al. 2004a (I). Boyd & Rees 2003 (II-1).
		20–30	SMSM	n/a	n/a	>4	Suffolk coast, UK.	Millner et al. 1977 (II-1).

3.3 Results

3.3.1 Literature review of marine aggregate extraction impacts

The literature review found that researchers used a range of indices for assessing recovery of the macrofaunal community. Typical indices were biomass, species richness and species composition. Despite this, there was broad agreement between them with calculated recovery times differing only by one or two years for comparable habitats (Cooper et al. 2008b).

The aggregate dredge sites reviewed are summarised in Table 3.3. Dredging activity was focussed on soft sediments of fine sand to gravel and was undertaken in widely differing tidal stress regimes, from very strongly tidal estuaries (e.g. Bristol Channel) to weak non-tidal environments (e.g. Tromper Ost). Dredge intensity ranged across three orders of magnitude; 10^3–10^6 t km^{-2} yr^{-1}. T_{Phys} ranged from 0.5 years at Kwintebank and Graal-Mürtiz, to decades at Tromper Ost and Hastings. T_{Bio} varied from 0.75 years in the Bristol Channel to decades in high intensity dredged sections of Thames Area 222.

Mean (M) T_{Phys} and T_{Bio} recovery rates and standard error of the mean (σ_M) were calculated for marine landscapes using the results of the literature review (Figure 3.1). Aggregate dredge sites size varied from 0.3–152 km^2. Recovery time is considered proportional to the spatial scale of a dredge site, as colonisation is more rapid at smaller than larger sites. This is evident in sites 0.1 m^2–0.1 km^2 but not in larger sites (Guerra-García et al. 2003), i.e. not at the scale of sites in Table 3.3. All sites reviewed were in water depths <50 m and were in six landscape types; sand plain, estuary, or shallow coarse and mixed sediments in strong, moderate and weak tidal stress. Different landscape-specific recovery rates at high and low levels of dredge intensity were not calculable, as there were too few data at the landscape scale. However, the standard error bars offer an indication of the range in recovery rates in each marine landscape. The error bars overlap for some landscape types, as mean values were calculated from few studies. Coarse sediments in moderate (SCSM) and weak tidal stress (SCSW) landscapes showed the longest period for T_{Phys} and T_{Bio}, respectively. The shortest period for T_{Phys} was 1.7 years in estuaries. Biological recovery was most rapid (4.5–5.5 years) in shallow coarse and mixed sediments of moderate or strong tidal stress, in sand plains and estuary landscapes. The mean values are considered reasonable estimates of recovery rate in UK sites.

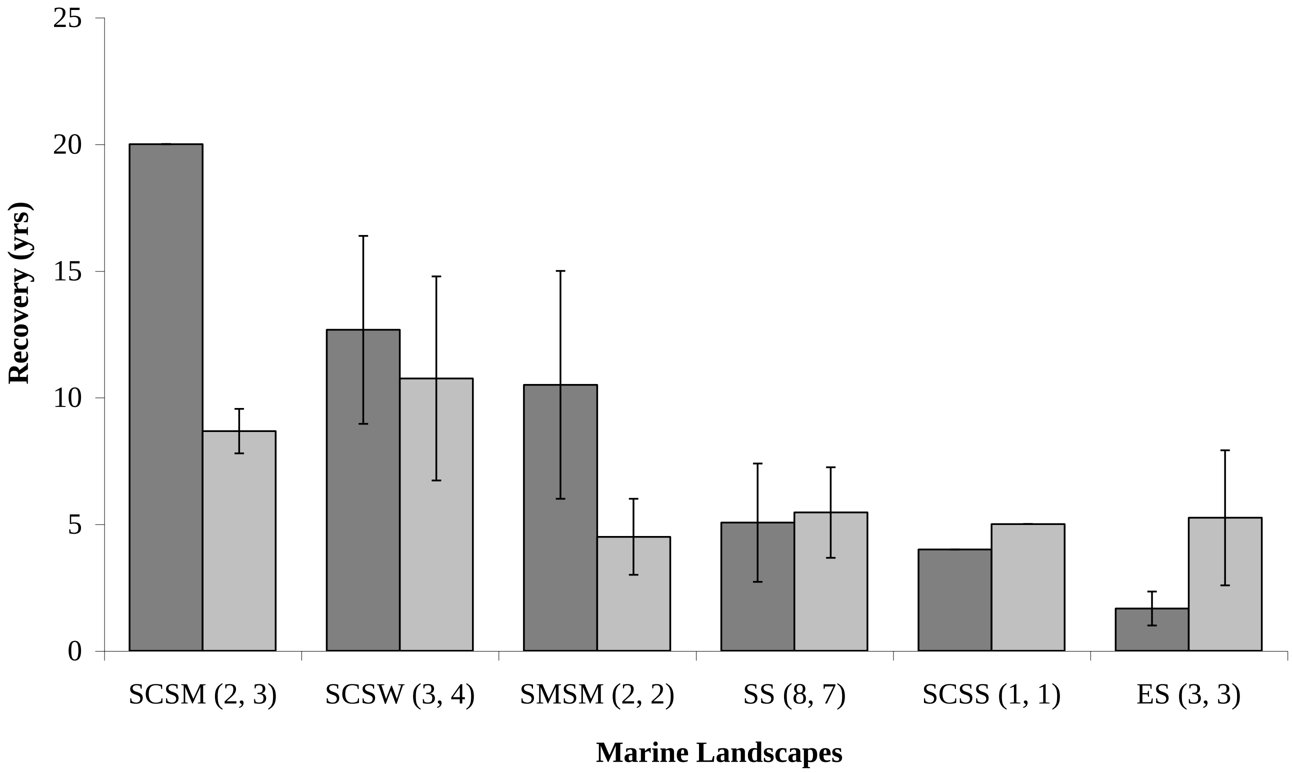

Figure 3.1: Predicted recovery times of marine landscapes from dredging based on the literature review. T_{Phys} = dark grey, and T_{Bio} = light grey, and N for each bar shown in brackets (T_{Phys}, and T_{Bio}), with standard error (σ_M) bars. Marine landscape codes as given in Table 3.1.

3.3.2 Footprint of UK marine aggregate extraction

The total footprint of UK dredge activity since 2001 was 321.7 km^2, confirming summary statistics reported by BMAPA (2008). The area of discrete dredge sites ranged between 0.0025 km^2 (i.e. isolated 50 x 50 m cells) to 21 km^2, the latter measured as the sum of contiguous cells at a site off the Suffolk coast. The isolated cells are likely to be indicative of vessels surveying, or ballasting (Royal Haskoning, pers. comm.) and are not discussed further herein. There were 63 main dredge sites in licensed areas ranging from 0.05–21.36 km^2 in area, 86 % of which were >1 km^2.

3.3.3 Spatial and temporal pressure of UK marine aggregate extraction

When spatially joined to the UKSeaMap marine landscapes (Table 3.1), 22 % of the EMS records were within four of the 'shelf' categories (>50 m deep), adjoining the boundaries of the 'shallow' categories. Typically dredgers work in depths 10–40 m (BMAPA 2006). Twenty seven of the 29 dredgers in UK waters have a maximum operating depth of <45 m and two of 50 m or less. Consequently the categorisation of some dredge sites as shelf was considered to be an artefact of the more coarse resolution of the UKSeaMap data. These dredge sites were grouped with the adjacent shallow landscapes of the same sediment size and tidal regime. For example dredging activity in shelf sand plain was combined with the shallow sand plain data (Figure 3.2).

Six main landscape types are targeted in the UK by the aggregate extraction sector. Figure 3.2 illustrates (a) the extent of these landscapes on the UK continental shelf and the proportion of each that is dredged, (b) the size of dredged area and levels of dredge intensity, by landscape type. Sand plains constitute the largest landscape (Figure 3.2a), and only 0.01 % is dredged for aggregate. Fifty percent of all aggregate extraction in the UK occurred in shallow coarse sediment plain-moderate tide stress (Figure 3.2b). This landscape has an estimated mean physical recovery rate from aggregate extraction of 20 years, the longest T_{Phys} of any landscape type (Figure 3.1). It also has the second longest T_{Bio}, of 8.7 years. The shallow coarse sediment plain-weak tide stress landscape has the longest mean T_{Bio}, 10.75 years, and a significant proportion of its dredge area (30 %) was exploited at moderate or high intensity (Figure 3.2b). Estuaries have the shortest T_{Phys} and T_{Bio} recovery periods (1.7 and 5.25 years, respectively) and aggregate dredging activity is undertaken in six percent of this landscape (Figure 3.2a). This is an under-estimation of total dredging activity, as maintenance dredging also occurs in many UK estuaries. However, the two processes are different; maintenance dredging removes recent unconsolidated sediments, whereas aggregate extraction removes minerals to a maximum permitted depth (DCLG 2007).

The majority of dredging effort in the UK was at low intensity (Table 3.4). At least 89 % of the total area dredged in any one year was worked at low intensity effort (of less than 1 hour). Over the entire seven year period, 81.7 % (263 km^2) of the whole dredging footprint was always worked at <1 h yr^{-1} and cumulative dredging in this area was less than five hours in all locations.

Table 3.4: Dredge intensity (h) and extent (km^2) for the UK marine aggregate extraction area. Data are categorised by low, moderate and high intensities. The seven year cumulative dredged time (h) for these intensities is also shown (rows).

Cumulative dredge time (h) for total period 2001–2007	**Dredged area (km^2)**		
	Low intensity, <1 h yr^{-1}	Moderate intensity, 1–1.25 hours in one or more years	High intensity, >1.25 hours in one or more years
<1	218.02	0.00	0.00
1–3	43.63	12.48	0.00
3–5	1.36	4.98	19.68
5–10	0.00	0.41	15.20
>10	0.00	0.00	6.07
TOTAL (km^2)	**263.0**	**17.9**	**41.0**

Between 2001 and 2007 only 4.9–6.7 % of the total area was dredged at high intensity of >1.25 hours in any one year. The area continually exploited at this intensity for every year was 0.03 km^2 (0.009 % of the footprint). Very high intensity dredging of >40 hours was rare and the most heavily impacted site was in the Mersey estuary in a small area of only 0.0013 km^2, where >90 hours of dredging occurred in 2001–2007.

Coarse sediment and sand plain landscapes were heavily targeted by the aggregate industry; together constituting 96 % of UK extraction area (Figure 3.2b), though the area that was dredged was <0.5 % in each landscape (Figure 3.2a). The industry dredges in all strengths of tidal stress. The relative proportions of dredge intensity within landscape types (Figure 3.2b) shows that the majority of dredging activity (>70 %) was at low intensity. Moderate or high intensity dredging formed a larger proportion of dredging effort in shallow coarse sediment-weak tide stress, 29 %. Moderate and high intensity dredging affected the smallest proportion (<11 %) of shallow mixed sediment plain-moderate tide stress landscapes.

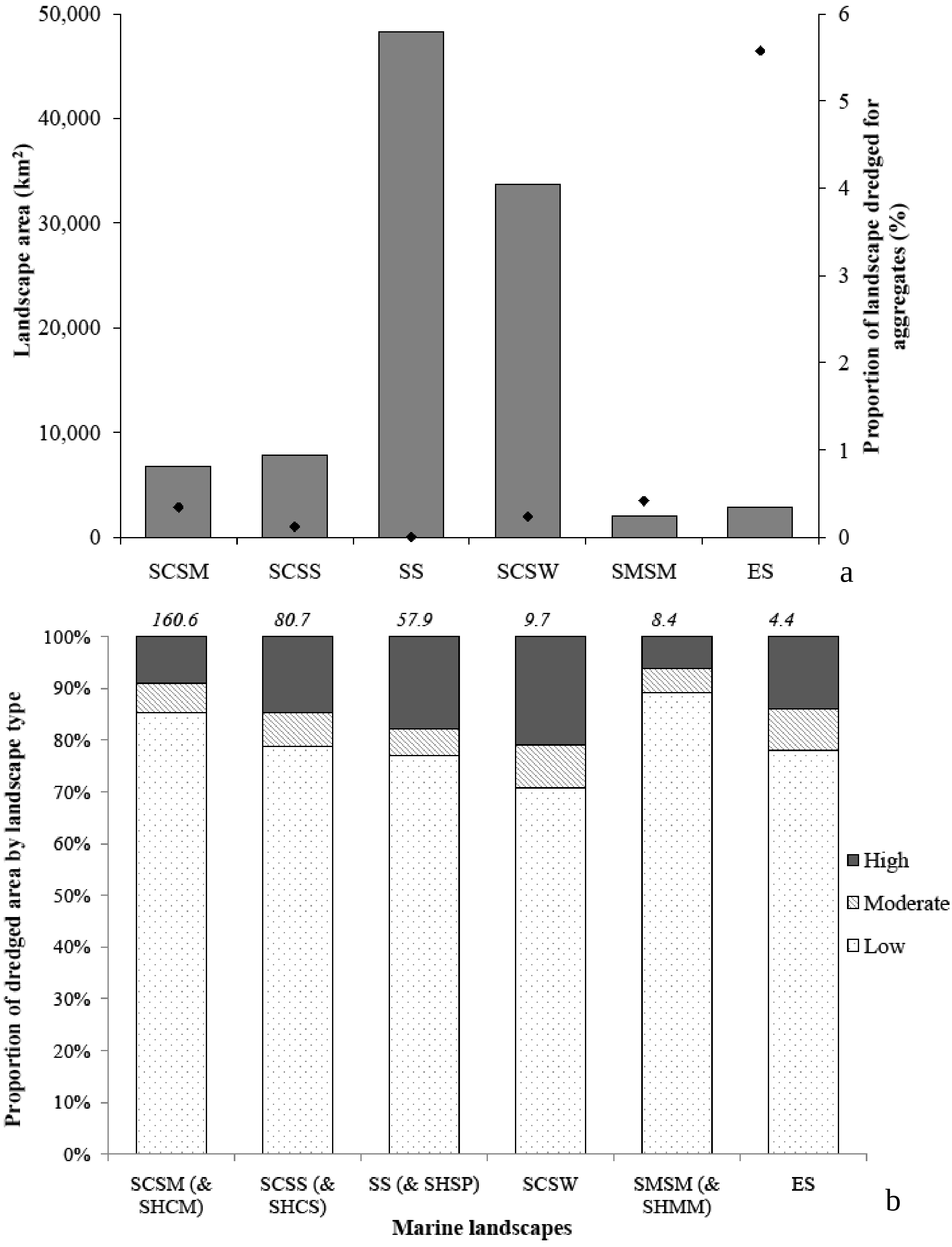

Figure 3.2: UK marine landscapes targeted for aggregate minerals (landscape codes as in Table 3.1). (a) Extent of marine landscapes in UK (bars) and proportion dredged for aggregate (points). (b) Proportions of dredge intensity in marine landscapes; High >1.25 h yr^{-1}, Moderate 1 – 1.25 h yr^{-1}, Low <1 h yr^{-1}. Values in italics above columns are the total area dredged (km^2) in each landscape type.

3.4 Discussion

The literature review highlighted the influence of environmental characteristics such as sediment type and hydrodynamics on recovery rates, following cessation of disturbance by dredging. The generalised model of macrofaunal species recovery following aggregate dredging proceeds from initial colonisation beginning within days, to recovery of diversity within months, recovery of population density after several months and biomass recovery after one or more years (ICES 1992, Newell et al. 2004a). Whilst this sequence is likely to be similar in a wide range of deposit types, the absolute rates are likely to vary. For example stable, low-energy environments, or coarse sediments are likely to have a biological community with a high proportion of long-lived, slow-growing and slow-reproducing species (Newell et al. 1998). These species may take longer to recover than biological communities in naturally more dynamic regimes.

Where sediment characteristics, topography and the natural hydrodynamic regime do not differ before and after dredging, re-establishment of a similar biological assemblage is probable (van Moorsel & Waardenburg 1991, van Moorsel 1993, 1994, van Dalfsen et al. 2000, Boyd et al. 2004, Robinson et al. 2005, Cooper et al. 2008b). In such situations recovery can be rapid, for example 24–30 months in the North Adriatic (Simonini et al. 2007). As expected, the least sensitive habitats, i.e. with the lowest T_{Phys} and T_{Bio}, occurred in estuaries, highly mobile sands in shallow waters and conditions of strong tidal stress (Millner et al. 1977, Bax & Williams 2001, Bolam & Rees 2003). Estuaries are relatively heavily exploited as six percent of their area is dredged for aggregate, in comparison with <1 % in the other landscape types. If topography and sediment composition are permanently altered and previously stable sediments are not re-established, communities remain at an early developmental stage (Boyd et al. 2004) and biological recovery can take more than 10 years (Cooper et al. 2005). The identification of physical and biological recovery rates for marine habitats should allow managers to further reduce impacts by targeting aggregate deposits in environments that can recover the most quickly.

Aggregate extraction activity in UK waters is restricted to sites licensed by the UK Crown Estate. At present these licensed areas do not appear to be located to avoid the most sensitive marine landscapes, nor to target the least sensitive. For example, the majority of aggregate dredging was found to take place in the shallow coarse sediment-moderate tide stress landscape, even though this environment has the longest estimated T_{Phys} and the second longest T_{Bio} from aggregate extraction (20 and 8.7 years, respectively). New statutory regulations for aggregate licensing came into force on 1st May 2007; Environmental Impact Assessment and Natural Habitats (Extraction of Marine Minerals by Marine Dredging) (England and Northern Ireland) Regulations (DCLG 2007). These replace the previous voluntary, informal, non-statutory Government View and will require Crown Estate to consider the Habitats Directive in their decisions. Consequently, future decision-making on site licences is more likely to be move towards an ecosystem approach.

The landscapes most heavily fished by towed benthic gears are those representing soft seabed with weak or moderate tide stress (Stelzenmüller et al. 2008), which increases the likely occurrence of cumulative impacts with aggregate extraction. Seabed penetration of fishing gear, such as hydraulic dredges, causes physical impacts similar to those of single passes by aggregate dredgers (Gilkinson et al. 2003). Other benthic trawl gears are less penetrative, but can still generate tracks on the seabed of 1.5–12 m width and 1–60 cm depth, depending on sediment and gear type (Churchill 1989, Hall et al. 1990, Nédélec & Prado 1990, Hall 1994, Vanstaen et al. 2008). The gears are associated with detrimental impacts on the marine benthos (Currie & Parry 1999, Shephard et al. 2008) and studies of epifauna at marine aggregate extraction sites have been found to show similar recovery rates to those for towed bottom-fishing gears in similar environments (Løkkeborg 2005, Smith et al. 2006). The impact of fishing gear varies significantly among habitats (Kaiser et al. 2006) and L. Robinson et al. (unpubl.) found the most sensitive marine landscapes to be the shelf sand plains and shallow coarse sediment plains of weak, moderate and strong tidal stress. Together, the area dredged for aggregates in these sensitive marine habitats constitute 213.85 km^2, representing 67 % of the total dredged area in UK waters. Further research of fishing impacts fishing on marine landscapes is underway, using spatial analyses similar to the methods herein.

The intensity of dredging is an important issue in landscapes with longer recovery times. The first anthropogenic sediment disturbance (e.g. dredging or trawling) in a previously unaffected site generates the highest mortality of biota, and subsequent repeat activity will result in relatively less damage per dredge (Jennings & Kaiser 1998, Kaiser et al. 2002, 2006). Most of the macrobenthos lives in the top 30 cm of the sediment, so mortality rates are directly related to the surface area of extraction (van Dalfsen et al. 2000). This is the depth to which most UK dredgers remove surface sand and gravel (MARINET 2004). Pranovi et al. (1998) demonstrated that fauna can recover after 15 days from dredging that penetrates only 7–13 cm into a sand and silt sediment, but if the penetration is 20 cm, recovery does not start until after 60 days (Sánchez-Moyano et al. 2004). The removal of commercial aggregate repeatedly over the course of a year has a cumulative effect slowing recovery further and increasing mortality (Boyd et al. 2004, Cooper et al. 2005).

The majority of dredging activity in the UK 2001–2007 was at low intensity, although for some habitats the proportion impacted at moderate and high intensity could be substantial. Although it was not possible to quantitatively analyse the effect of intensity on recovery in this study, modifying the intensity of dredge activities is likely to minimise the risk of permanently changing the physical characteristics of a site, which would slow its recovery. The shallow coarse sediment plain-weak tide stress landscape was subject to 30 % of dredging at moderate or high intensity (>1 h yr^{-1}). This landscape has the second longest mean T_{Phys} and the longest T_{Bio} of all landscapes. Faunal recovery after high intensity (>5 hours), repetitive dredging can require up to 10 years for recovery after cessation of dredging (Boyd et al. 2004).

Overall the dredging activity of the UK's marine aggregate sector has a relatively small spatial footprint (<1 % of the size of the area trawled for fish), but the size of individual dredge sites can affect the re-establishment of the macrobenthic community (Guerra-García et al. 2003). Recovery rates of the biological community following dredge cessation will be more rapid at small dredge sites (e.g. <100 m^2) than at large sites (e.g. >1000 m^2) because larger patches possess a smaller edge/surface area ratio, restricting the potential for immigration by colonists (Guerra-García et al. 2003). However, 86 % of the main marine dredge sites in the UK were >1 km^2 in size (mean

4.6 km^2), which is a large spatial scale relative to the range summarised by these authors. Consequently, the most rapid recovery predicted would be a minimum of six months, and more probably would be in the order of years, which corresponds with the temporal range found in the literature review (Table 3.3).

A limitation to this study was the inability to consider the effect of aggregate overspill and screening creating sediment plumes. These techniques may significantly alter the substratum, extending the period for recovery (de Groot 1996), or the sediment plumes may lead to smothering effects (de Groot 1996, Sánchez-Moyano et al. 2004). To accurately quantify variation in spatial extent and impact of sediment plumes would require site-specific studies to quantify concentration of suspended sediments above background levels, but there are too few studies of these factors (Eastwood et al. 2007) and they were not incorporated into this study.

In this study both the physical and biological impacts of dredging have been found to be spatially variable and frequently site-specific, as observed by previous workers (e.g. Newell et. al. 1998, De Grave & Whitaker 1999, Boyd et al. 2004). This site-specificity of dredge impact complicates the prediction of likely effects and recovery periods at both extant and prospective extraction areas (Boyd et al. 2004). Furthermore it supports the findings of ICES (1992) in highlighting the importance of site-specific studies in the future. Research into effects of different intensities of dredging by landscape type is needed. The general pattern of response to aggregate extraction needs to be tested further to establish its general validity in all environments, particularly in areas which have been exposed to high intensity dredging over many years (Boyd et al. 2004). Long-term studies of sites disturbed by either the aggregate or fishing sectors are few in number (Bradshaw et al. 2002). Detailed studies into the cumulative effects of anthropogenic activities, e.g. mobile bottom fishing and aggregate dredging, are urgently required for the offshore zone to be better understood, so that stakeholders can make sound decisions for marine planning.

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CHAPTER 4

Recovery of UK seabed habitats from benthic fishing and aggregate extraction – towards a cumulative impact assessment

Foden J, Rogers SI, Jones AP (2010) Recovery of UK seabed habitats from benthic fishing and aggregate extraction – towards a cumulative impact assessment. Mar Ecol Prog Ser 411:259–270 (see Declaration, page 2)

Abstract

Assessing cumulative impacts of multiple pressures on the marine environment can help inform management response. This requires understanding of the spatial and temporal distribution of human pressures and their impacts. Quantifying seabed recovery rates from two significant pressures in European waters, benthic fishing and aggregate extraction, is a significant step towards assessing sensitivity and cumulative impact.

Vessel Monitoring Systems data were used to estimate the distribution and intensity of benthic fishing in UK marine waters (2006 to 2007). Data were separated by towed bottom-fishing gears (scallop dredges, beam and otter trawls) and linked to habitat in a GIS. Recovery periods of seabed habitats were estimated by literature review, for gear types and fishing intensity. Recovery rates generally increased with sediment hardness and habitats required longer periods of recovery from scallop dredging than from otter or beam trawling. Fishing pressure across the habitat-gear combinations was such that 80% of the bottom-fished area was estimated to be able to recover completely before repeat trawling, based on mean annual trawl frequencies. However, in 19% of the UK's bottom-fished seabed, scallop dredging in sand and gravel and otter trawling in muddy sand and reef habitats occurred at frequencies that prevented full habitat recovery.

In 2007 benthic fishing and aggregate extraction occurred together in an estimated 40 km^2 (<0.02%) of UK seabed. Cumulative impacts were estimated as total recovery

time under four scenarios; greatest, additive, antagonistic and synergistic impacts. Recovery from aggregate extraction required much greater periods than from benthic fishing, and gravel was identified as a more sensitive habitat than sand.

Key words: towed bottom-fishing, recovery, cumulative effects, fishing intensity, marine habitats, aggregate extraction, UK seabed, vessel monitoring system

4.1 Introduction

Marine ecosystems provide a range of goods and services of value to humans (e.g. Costanza et al. 1997, Foley et al. 2010). Human activities can alter ecosystems, changing energy flows and biological communities, potentially leading to regime shifts (e.g. Choi et al. 2004, Hinz et al. 2009) which could undermine the supply of marine resources. Sustainable human use of marine resources requires an integrated approach to marine management, with knowledge of the location of activities and habitats, and accurate assessment of the condition of the marine environment. Mapping the distribution and intensity of pressures on different habitats, and assessment of cumulative effects are relevant for spatial planning and conservation objectives, as required under legislation such as the Marine and Coastal Access Act (HMSO 2009) and the European Marine Strategy Framework Directive (EU 2008).

At present there are gaps in our knowledge and understanding of the impacts from major human activities and the interactions between them (OSPAR Commission 2003, Eastwood et al. 2007, Borja et al. 2008, Foden et al. 2008, Halpern et al. 2008b). Such interactions can create cumulative impacts, which need to be understood and quantified for more informed management decisions. Understanding cumulative impacts may be particularly important as repeated impacts may, for example, limit the ability of an ecosystem to recover over time. Furthermore, the combined impacts of many different activities might lead to different ecosystem responses from those of just one type. Cumulative effects can be comparative, additive or interactive in nature. A comparative effect predicts the overall impact to be equal to that of the single worst or dominant stressor (Bruland et al. 1991).

Additive effects are simply the sum of the individual pressures (Halpern et al. 2008a), which is judged to be the case for pairs of pressures (Crain et al. 2008). Interactive effects are the results of multiple activities accumulating non-linearly; i.e. causing lesser (antagonistic) or greater (synergistic) effects than the sum of their parts. Antagonistic effects describe the total impact of multiple pressures as being less than the algebraic sum of the individual impacts and synergistic effects occur when the total impact is greater than the sum of the individual pressures (Smit & Spaling 1995, Folt et al. 1999, Cefas 2001, CCW 2002, Crain et al. 2008).

Recent ecological studies and meta-analyses have begun to address the theories and concepts of cumulative effects caused by multiple human activities on the seabed (e.g. Vinebrooke et al. 2004, Darling & Côté 2008, Halpern et al. 2008b). However, there have been fewer attempts to robustly quantify those impacts and there is a gap in knowledge regarding environmental sensitivity and response to multiple pressures (e.g. Halpern et al. 2008a). Investigations of seabed recovery rates following disturbance provide one method of quantitatively estimating habitat sensitivity (Desprez 2000, Cooper et al. 2007, Foden et al. 2009). If a pressure occurs too frequently for a habitat to recover, the benthic community's biomass and productivity decline (Hiddink et al. 2006a) and sustainability of human use of the resource may be jeopardised. By geospatially modelling the process of cumulative impacts of six different pressures, using empirical data, Stelzenmüller et al. (2010) made an important advancement in the practical application of cumulative impact concepts. Using two major pressures currently affecting the UK seabed these concepts were put into practice using quantitative data on habitat sensitivity and the effects of individual and cumulative impacts.

Whilst oil and gas extraction, wind farms, dumping, pipelines and cables can be important site-specific pressures acting on the UK (England and Wales) seabed, the major sources of seabed disturbance are near-bed currents, wind-induced waves (Hall 1994), aggregate dredging for mineral resources and bottom-trawling for fish (Jennings & Kaiser 1998, Eastwood et al. 2007, BMAPA 2008, Stelzenmüller et al. 2008). The UK is one of the largest producers of marine aggregate in the world, and 23.1 million tonnes of sand and gravel were extracted from the English and Welsh seabed in 2007 (BMAPA 2008). Recovery rates of the benthic community after cessation of disturbance by marine aggregate dredging have been used as a proxy of habitat sensitivity (Desprez 2000, Cooper et al. 2007, Foden et al. 2009). Equivalent examinations of impacts caused by other sectors will be required to fully assess habitat sensitivity and to estimate the cumulative effects of all pressures operating on the UK seabed.

A broad range of approaches to integrated assessments and cumulative impacts assessments have been used, with varying degrees of success (Foden et al. 2008). A sector by sector approach to analysing human pressures on UK seabed habitats was adopted, with a view to combining these in a multi-sectoral integrated assessment, and the work presented here contributes to this. The intention is that our approach will help inform decision makers, facilitating an ecosystem approach to marine management.

The most important human pressure in terms of its spatial extent and level of impact on the UK marine environment results from fishing (Collie et al. 1997, Rijnsdorp et al. 1998, Dinmore et al. 2003, Eastwood et al. 2007). Taking an ecosystems approach in the context of fisheries management requires, *inter alia*, information on the response of marine habitats to fishing (FAO 2003). Fishing practices that have a direct physical impact on the seabed such as trawl and dredge gears were considered (Rijnsdorp et al. 1998). Bottom-fishing directly affects the seabed and benthic communities and has been recorded as occurring in 75% of the global continental shelf (Kaiser et al. 2002). The degree of disturbance is dependent on three main factors: the type of fishing gear deployed, the intensity of fishing activity and the sensitivity of the habitat (Collie et al. 2000, Kaiser et al. 2006).

Benthic trawl gears can generate tracks on the seabed of 1.5 to 12 m width and 0.01 to 0.6 m depth, depending on sediment and gear type (Churchill 1989, Hall et al. 1990, Nédélec & Prado 1990, Hall 1994, Vanstaen et al. 2008). These gears are generally associated with detrimental impacts on the marine benthos (Currie & Parry 1999, Shephard et al. 2008). For example, to increase fish catches, beam trawls are usually fitted with chains designed to penetrate the top few cm of the substratum and infauna and epifauna can be damaged as the chains pass through the sediment (Kaiser &

Spencer 1994, 1995). Typically, shellfish dredges and flatfish beam trawls disturb the seabed more intensively than otter trawls (Hall 1994).

Biomass, abundance and cover of macro-fauna and -flora in sand and gravel environments are known to be negatively correlated with trawling intensity (Svane et al. 2009). In previous work responses and recovery rates of different habitats to towed bottom-fishing gears have been reviewed (Kaiser et al. 2006, Pitcher et al. 2009). Kaiser et al. (2006) found fishing activities had significant negative, but short-term, impacts on biota in the highly energetic, shallow, soft-sediment habitats, with abundance recovering in <50 days (d). Whereas in more stable gravel sediments, taxa were still ~40% reduced after 50 d. Collie et al. (1997) found recovery from scallop dredging in shallow gravel sediment at the St George's Bank to take five to ten years and in the most stable environment of biogenic reefs, biota showed no signs of recovery after more than four years.

Vessel monitoring system (VMS) data have been used successfully for estimating the spatial and temporal distribution of fishing effort (Deng et al. 2005, Murawski et al. 2005). This study analysed 2006 and 2007 VMS data for UK and foreign vessels deploying bottom-fishing gear in the UK (England and Wales) reporting region. The four objectives were to: (i) quantify the spatial extent and annual intensity of benthic fishing activity in 2006 and 2007 at a high resolution; (ii) use data on recovery periods from published, peer-reviewed studies to estimate the proportion of fished habitats in which recovery would be possible at recent (2007) levels of fishing effort; (iii) combine the results with published spatial data on aggregate dredging in UK waters (Foden et al. 2009) in order to investigate where the two pressures coincided during 2007, potentially giving rise to cumulative impacts on the seabed; and (iv) estimate overall recovery times for the four cumulative effects scenarios proposed by Stelzenmüller et al. (2010) – greatest, additive, antagonistic and synergistic. The latter objective allowed us to determine the sensitivity of the findings to different measures of impact estimation, for different habitat types. Our key aims are for this information to contribute to an integrated assessment of cumulative impacts from simultaneous pressures acting on the UK seabed, and to help in the application of an ecosystem approach to marine management.

4.2 Method

4.2.1 Study area and VMS data

The study area comprised the marine waters of the UK (England and Wales) (Figure 1.1). Regions were delineated using boundaries determined for environmental status reporting under Charting Progress (Defra 2005).

Satellite-based Vessel Monitoring Systems (VMS) have been fitted to fishing vessels ≥15 m since 2005 to automatically record identity, position, speed and heading data at an average frequency of ~2 h. Vessels exclusively fishing within 12 nautical miles of the shoreline or that undertake fishing trips of less than 24 h are exempted (Dann et al. 2002). Positional data of UK and European vessels (Belgium, Denmark, Holland, France, Ireland and Spain) in the study area for the two year period covering 2006 and 2007 were made available by the Defra Marine Fisheries Agency. Speed rules were applied to determine fishing or steaming activity (Lee et al. 2010). Benthic fishing gears were identified using UK logbook data or the European vessel register and these were aggregated to three bottom-fishing gear types; beam trawls, otter trawls and dredges.

4.2.2 Swept area of fishing gears

The swept area of trawls or dredges depends on the width of that part of the gear which is in contact with the seabed. There are three main bottom-fishing gear types deployed in UK waters; beam trawls, scallop or hydraulic dredges and otter trawls. A subset of the latter includes *Nephrops* (prawn) trawls. Beam trawlers generally operate twin rigs with an overall width of 24 m (Rijnsdorp et al. 1998, Dinmore et al. 2003) and this width was used as the swept area.

Scallop dredges are often deployed from beam trawl vessels in the UK, with a maximum number of nine dredges (~1 m wide) per beam. As two beams are generally deployed, 18 m was used as an estimate of swept area. The use of hydraulic suction dredges for fishing shellfish is geographically restricted in UK waters. Mechanised dredges of this nature generate tracks on the seabed up to three metres wide (Hall et al. 1990, Hall 1994), which was the area used in our calculations.

Because the ecological impacts of these gears are comparable the spatial extent and fishing intensity of each were calculated separately and subsequently combined as 'scallop dredges'.

To account for the active region of a benthic trawl, 24 m was used as the swept area of an otter trawl (Carrothers 1980, Hiddink et al. 2007). The area for *Nephrops* trawls was taken from Hinz et al. (2008) whose side-scan sonar observations showed tracks up to 60 m wide in the northeast Irish Sea. These gears were combined and presented as 'otter trawls'.

The locations and swept areas for the different bottom-fishing gears were calculated for both 2006 and 2007. A chi-square test (χ^2) was conducted to analyse whether the particular gears deployed in cells in 2006 tended to be the same as those in 2007.

4.2.3 Fishing distribution and intensity

The exact tracks of fishing vessels are not recorded by VMS and previous studies have either analysed VMS records as point data (Rijnsdorp et al. 1998) or estimated vessel tracks between points (Eastwood et al. 2007, Mills et al. 2007). Point data were used in this study and the spatial extent of the potential surface area fished around each point was calculated for 2006 and 2007. The extent was estimated from each VMS record based on vessel speed, VMS interval and width of fishing gear. For example, for a beam trawl 24 m wide fishing at 4.5 knots (8.334 km h^{-1}) one VMS record (two hour interval) corresponds to a fished surface area of 0.024 x 8.334 x 2 = 0.4 km^2.

Trawling effort derived from VMS records may be represented using high resolution grids, with cell sizes of <3 km^2 (Mills et al. 2007) and 1 nm^2 or l km^2 having been used for fishing effort analyses (e.g. Rijnsdorp et al. 1998, Hinz et al. 2009). For this study 1 km^2 cells were used as an appropriate compromise between the high resolution VMS point data and the uncertainties in the calculations of surface area fished around each point. Gridded intensity of fishing activity for each year was calculated from the annual number of trawls estimated to have passed through each cell and their associated fished surface area. An intensity score of 0.4 therefore represented a fished area of 0.4 km^2 yr^{-1}. A fished intensity of one could describe a situation where the entire area of a l km^2 cell was fished once a year, assuming the trawl distribution was homogeneous. Intensity scores greater than one indicate a cell or parts of a cell were fished more than once annually. To test the similarity of fishing intensity per cell in 2006 and 2007 Pearson correlation coefficients were calculated.

To identify the spatio-temporal distribution of bottom-fishing effort by habitat type VMS records were spatially joined to British Geological Survey (BGS) sediment type data using the ArcGIS Geographical Information System (ESRI). This linked bottom-fishing effort to the sediment classification scheme of Folk (1954). These classes were grouped into five habitat types, based on the largest proportion of constituent particle size in the BGS classification scheme. Mud, muddy sand, sand, gravel and reef are the habitat descriptors widely used in studies of bottom-fishing impact on the benthos (e.g. Collie et al. 2000, Kaiser et al. 2006, Pitcher et al. 2009). Biogenic habitats constructed or composed primarily of living biota are classed as rock in the BGS system. The potential for recovery of the seabed in UK waters following bottom-fishing was then calculated using published recovery rates of each habitat type, based on the distribution and intensity of fishing activity.

4.2.4 Recovery from benthic fishing

Published studies have found the effects of bottom-fishing on habitats were most strongly related to types of gear and sediment, with sediment type itself correlated with depth (Kaiser et al. 2006, Pitcher et al. 2009). Natural disturbance, such as that caused by near-bed currents and wind-induced waves, is also related to depth and so it is implicit in habitat type and recovery. Therefore habitats are generally defined in terms of their particle-size ranges and this approach was adopted here. A review was conducted of scientific literature for recovery rates of the benthos in different habitats, after bottom-fishing by the three main benthic gear types. Studies were identified using computer database search engines of peer-reviewed literature, such as Scopus and ASFA, and general Internet search engines. In our study area primary and secondary production are large with strong seasonal patterns, so data from studies conducted in this, or similar, areas were used. The review by Kaiser et al. (2006)

provided the majority of the recovery periods and the results of four additional studies were used where particular habitat-gear combinations were not given in that review.

The term recovery was used, as defined by Kaiser et al. (2006); recovery was deemed to have occurred when the abundance, species richness or biomass of benthic biota was equivalent to a 20% reduction or less in the pre-impact value. The study area has a long history of high levels of fishing activity, with trawlers tending to repeatedly target the same grounds year after year (Kaiser et al. 2002, Hiddink et al. 2006b). Therefore, the point at which recovery is deemed to have occurred is, for some habitats, a point in a constant disturbance cycle, and not disturbance of a pristine benthic community.

An index of recovery (Ind_{Rec}) was calculated to estimate whether the frequency of bottom-fishing allowed habitats enough time to recover between fishing events. The fishing intensity scores (Section 2.3) were used to estimate the mean number of days (d) between fishing events, per cell, per year. For example, a fishing intensity score of 0.4 represents 912.5 d between sweeps by fishing gear. To provide an index of recovery (Ind_{Rec}) for each 1 km^2 cell, the recovery periods established from the literature review were divided by the mean number of days between fishing events. Where more than one gear was deployed in a cell the longest recovery period was used for the portion of the cell affected by that gear. Where $Ind_{\mathrm{Rec}} = 1$, fishing intensity was equivalent to the required number of days for a habitat to recover; >1, the habitat was unable to fully recover; <1, recovery can be expected before the next fishing event. Benthic fishing fleets are active throughout the year in UK waters, although effort is noticeably reduced in December. The mean sizes of fished area per month by gear type, excluding December, were: beam trawling, 8375 km^{-2} (σ_M = 322); otter trawling, 7331 km^{-2} (σ_M = 279); and scallop dredging, 279 km^{-2} (σ_M = 49). During December effort is between ten and 30% of the other 11 months. Given the consistency of fishing activity a homogeneous distribution of pressure during the year was assumed. This also facilitated comparison of fishing effort with aggregate extraction records, the latter only being available as annual data.

4.2.5 Cumulative impact assessment

To investigate cumulative impacts of aggregate extraction and fishing pressures, locations where these activities coincided were identified. Electronic Monitoring Systems (EMS) have been fitted to all aggregate dredgers operating in UK (England and Wales) waters since 1993 to automatically record position and 'dredging status' every 30 seconds. Aggregate extraction effort in the UK is measured as hours dredged annually (h yr^{-1}). EMS data provided in 50 x 50 m (2500 m^2) cells were analysed by Foden et al. (2009). Electronic monitoring system data for 2007 and the gridded 1 km^2 fishing cells for the same year were spatially combined. The locations and habitats where both pressures coincided could then be identified. The sediment descriptors from Foden et al. (2009) were cross-referenced and these were associated with the habitat types used herein for recovery from benthic fishing (Table 4.1). Where the pressures were concurrent in 2007, the recovery times of each habitat type from aggregate extraction and fishing were estimated. For recovery from aggregate extraction (T_{BioAgg}) values from Foden et al. (2009) were used; recovery was defined as the mean time necessary for establishment of a benthic community virtually indistinguishable from surrounding, non-impacted reference sites (Cooper et al. 2005), or when species number, abundance and/or biomass of a site had returned to 90% of the original values (Hiddink et al. 2006b). Recovery from fishing was estimated for each habitat-gear combination as the mean fishing intensity in 2007 x maximum recovery period (from Table 4.4).

To quantify cumulative impacts where benthic fishing occurred in active aggregate extraction sites, impact was estimated as total recovery times under four scenarios. The premise for scenario I was the single worst or dominant pressure (i.e. the pressure requiring the greatest period for the benthos to recover) takes precedence over the others in determining combined effects; with lesser pressures have no additional impact (Bruland et al. 1991). Previous investigations of marine landscapes' sensitivity to a range of human activities have found sand, coarse and mixed sediment environments to be more sensitive to extraction pressures (aggregate removal) than to abrasion pressures (benthic fishing) (Stelzenmüller et al. 2010). Therefore aggregate extraction was considered to be the dominant pressure. For scenario II, multiple pressures were assumed to act independently within the system and therefore overall recovery time was the sum of both aggregate extraction and benthic fishing (e.g. Crain et al. 2008, Halpern et al. 2008a).

The purpose of scenarios III and IV was to show a range in the sensitivity of habitats to impacts that interact. Scenario III estimated cumulative impacts as the antagonistic effects of multiple pressures (Darling & Côté 2008); if pressures are applied consecutively to marine habitats, then the impact of the first pressure may pre-condition the habitat to be less sensitive to the second pressure. The total recovery period for scenario III was estimated as recovery time from the primary pressure + 50% recovery time from the secondary pressure. Recovery times were expected to be between those of scenario I and II. In scenario IV synergistic effects were assumed, in which the impact from accumulated pressures was greater than the sum of the individual parts (Cefas 2001, CCW 2002), the assumption being the first pressure lessens the resilience of a habitat, making it more sensitive to subsequent pressures. Therefore in scenario IV estimated total recovery time was recovery from the primary pressure + 150% recovery time from secondary pressure, with the expectation of total times greater than the other three scenarios.

4.3 Results

4.3.1 Fishing distribution and intensity

In 2006 122 000 km^2 (47.2%) of the English and Welsh seabed was estimated to be affected by benthic fishing, increasing in area to134 000 km^2 (51.8%) in 2007. Fishing activity was patchily distributed with strong spatial variation in annual fishing intensity in UK waters. In locations where towed bottom-fishing was recorded by VMS, intensity scores ranged from 0.0002 to 39 in 2006 (i.e. a 1 km^2 cell may have been fished up to 39 times per year) and 0.0002 to 30 in 2007. In 2006 42% of the UK's bottom-fished area was fished more frequently than once per year, this value decreasing slightly to 40% in 2007.

There was a strong spatio-temporal correlation in fishing intensity between 2006 and 2007 (Pearson's $r = 0.405$, $p < 0.001$), i.e. the spatial pattern of fishing intensity remained broadly unchanged. In particular, there was a consistent pattern of intense activity near the northeast coast of England, and low or no activity to the north and west of Anglesey in the Irish Sea. As the spatio-temporal patterns of fishing pressure in UK waters have been mapped previously by Stelzenmüller et al. (2008), the data have not been reproduced here.

The spatial distribution of bottom-fishing coincided with 17 sediment classes grouped into five habitats (Table 4.1). Mean fishing intensity per habitat type ranged between 1.1 and 6.7 yr^{-1}. Mud was the most intensively fished habitat in both 2006 and 2007. The habitats most commonly targeted for bottom-fishing were sand and gravel (Table 4.2). These constitute relatively large habitats in UK waters (226 000 km^2 in total) and in 2006 and 2007 fishing affected approximately 50% of their areas. In contrast mud is the smallest habitat of the five (2 000 km^2) in UK waters, but 77% of its area was bottom-fished in 2006, increasing to 80% in 2007. A large proportion (63% in 2006 and 76% in 2007) of muddy sand was bottom-fished, but only 12% of reef habitat in 2006 and 2007.

Table 4.1: British Geological Survey (BGS) sediments affected by bottom-fishing activity in UK waters 2006 and 2007. Sediments were grouped to habitat types for this study. Mean (M) intensity of fishing per year and standard error of the mean (σ_M).

BGS sediment type (Folk 1954)	**Habitat type**	**Intensity (times fished yr^{-1})** *2006* M	(σ_M)	*2007* M	(σ_M)
Sand Gravelly sand Slightly gravelly sand	Sand	1.14	0.005	1.12	0.005
Gravel Muddy gravel Muddy sandy gravel Sandy gravel	Gravel	1.75	0.018	1.46	0.012
Muddy sand Sandy mud Gravelly mud Gravelly muddy sand Slightly gravelly muddy sand	Muddy sand	2.3	0.021	1.92	0.019
Rock	Reef (biogenic)	1.72	0.122	1.77	0.128
Mud Slightly gravelly mud Slightly gravelly sandy mud	Mud	4.67	0.073	6.91	0.122

Table 4.2: Seabed habitats subjected to bottom-fishing in UK waters; total habitat size, mean areas bottom-fished and unfished in 2006 and 2007. Proportion (%) of habitat fished shown in italics.

	Habitat type				
	Sand	**Gravel**	**Muddy sand**	**Reef**	**Mud**
Total habitat size (km^2)	185817	40041	23722	5085	1843
Unfished Area (km^2)	94139	22264	7181	4458	394
Fished Area (km^2)	91677	17776	16541	627	1448
% habitat fished	*49.3*	*44.4*	*69.7*	*12.3*	*78.6*

Fishing gears were deployed in different proportions in each habitat (Table 4.3). However, there was a strong spatial overlap in the locations where particular gears were deployed in 2006 and 2007 (χ^2 p <0.001). Scallop dredgers were the gear deployed in the smallest proportions of all habitats, ranging from 0.4% of mud to 11% of reef habitats.

Table 4.3: Proportions (%) of the bottom-fished area by gear type in 2006 and 2007.

	Habitat type									
	Sand		**Gravel**		**Muddy sand**		**Reef**		**Mud**	
	2006	*2007*	*2006*	*2007*	*2006*	*2007*	*2006*	*2007*	*2006*	*2007*
Dredge only	2.5	1.7	8.2	8.1	3.4	2.3	8.7	11.0	0.7	0.4
Beam only	41.7	28.8	24.3	8.2	13.5	6.7	42.0	22.7	1.2	0.1
Otter & Nephrops only	31.9	41.5	33.6	40.4	64.0	76.8	30.8	34.2	93.5	97.9
>1 gear type	23.9	28.0	33.9	43.3	19.1	14.2	18.5	32.1	4.6	1.6

4.3.2 Recovery from benthic fishing

Table 4.4: Maximum recovery times for habitats by fishing gear types. d: days, mo: months, yr: years, n/d: no data.

Gear type	Habitat type				
	Sand	**Gravel**	**Muddy sand**	**Reef**	**Mud**
Beam trawl	<6 mo[a]	n/d	<8 mo[b]	n/d	n/d
Otter trawl	0 d[b]	<12 mo[d]	>7 mo[c]	>8 yr[b]	8 d[b]
Scallop dredge	>8 yr[b e]	>8 yr[b]	1.6 yr[b]	3.2 yr[b]	n/d

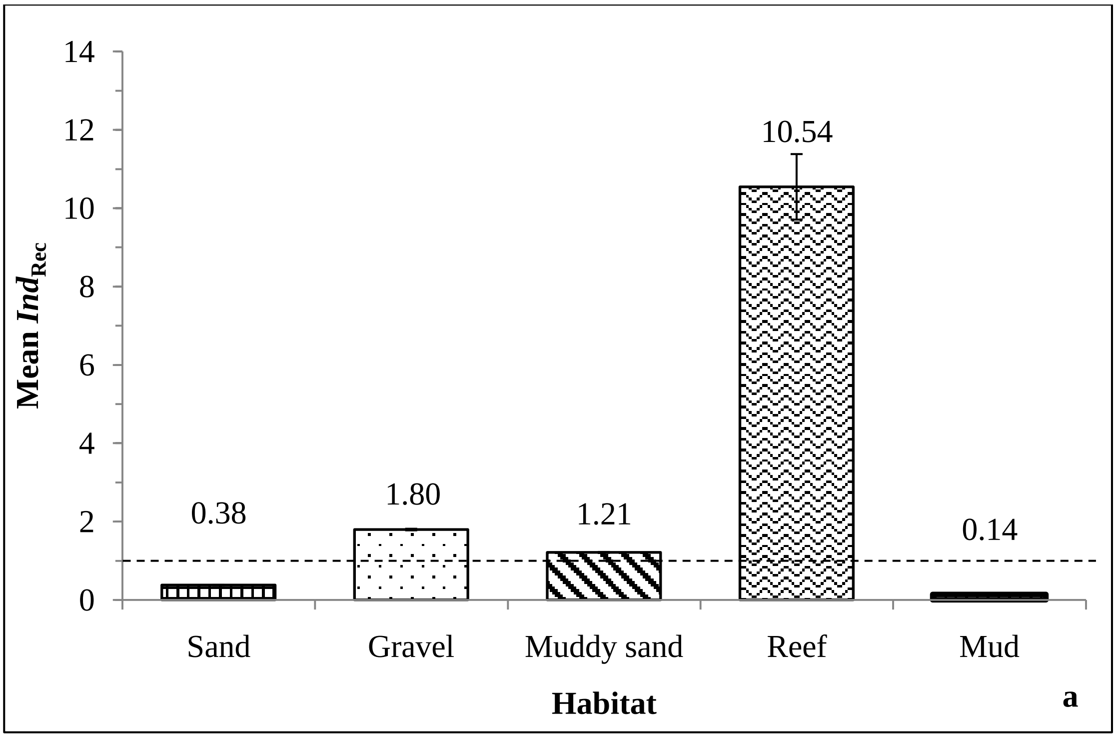
[a]Kaiser et al. (1998); [b]Kaiser et al. (2006); [c]Ragnarsson & Lindegarth (2009); [d]Kenchington et al. (2006); [e]Gilkinson et al. (2005)

Table 4.4 summarises the recovery periods of five habitats after fishing by three types of benthic gears. No published data were found for recovery rates following beam trawling in gravel, biogenic reef and mud habitats, nor from scallop dredging in mud habitats. However, these four habitat–gear combinations only represent approximately one percent of the total area subject to bottom-fishing in 2007 (Table 4.5). The recovery times in Table 4.4 were used to estimate an index of recovery (Ind_{Rec}) for each 1 km^2 cell. As there were no statistically significant spatial or temporal differences in gear types or fishing intensity between 2006 and 2007, recovery at 2007 fishing pressure levels was estimated. The results are presented in Figure 4.1 as a mean Ind_{Rec} for each habitat and as mean Ind_{Rec} for each habitat-gear combination.

Table 4.5: Percentage and area (km^2 in parentheses) of habitats estimated to recover / not recover from 2007 levels of fishing intensity.

Recovery status	Habitat type				
	Sand	**Gravel**	**Muddy sand**	**Reef**	**Mud**
Recovered	89.9 (84 485)	50.3 (10 005)	63.1 (11 412)	20.0 (129)	99.5 (1469)
Not recovered	10.1 (9 462)	41.5 (8 263)	36.9 (6 676)	57.4 (371)	0.0 (0)
Undetermined	0.0 (0)	8.2[a] (1 626)	0.0 (0)	22.7[a] (147)	0.5[a b] (8)

[a] No recovery period for beam trawl; [b]no recovery period for scallop dredge



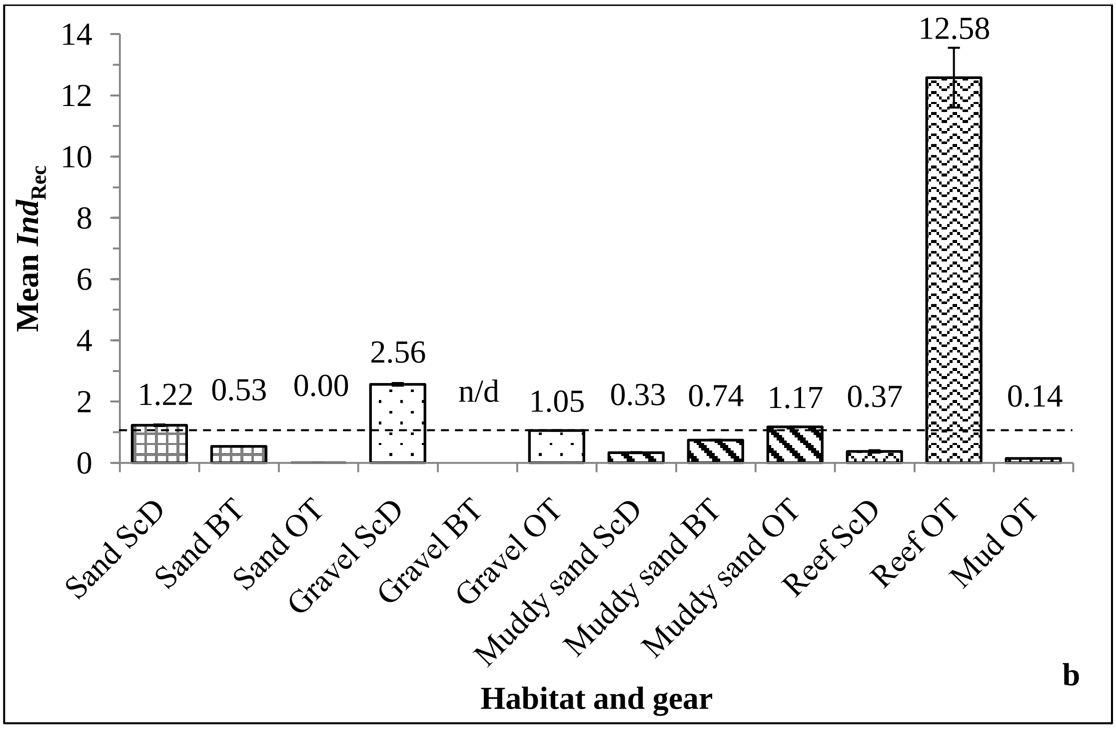


Figure 4.1: Mean Ind_{Rec} 2007 for (a) habitats regardless of gear type and (b) gear-habitat combinations. BT: Beam trawl; OT: otter trawl; ScD: scallop dredge. Error bars are standard error of the mean (σ_M). At Ind_{Rec} = 1 recovery period equals fishing frequency (horizontal broken line on each plot), at <1 fishing frequency is less than predicted recovery period and >1 fishing frequency exceeds recovery period.

In 2007, at an average fishing intensity for gear types, sand and mud habitats appeared to be able to fully recover, whereas gravel, muddy sand and reef habitats were fished at frequencies in excess of estimated recovery periods (Figure 4.1a). However, Ind_{Rec} (Figure 4.1b) varied significantly with individual habitat-gear combinations. Ind_{Rec} was less than 1 in six habitat–gear combinations and was ~1 for otter trawling in gravel. High fishing intensities in four habitat-gear types were responsible for Ind_{Rec} exceeding one. On average scallop dredgers were deployed in sand too often to allow for the habitat's recovery. Scallop dredgers affected sand habitat 1.22 times more frequently than the estimated period required for recovery between sweeps. Beam and otter trawls are more commonly deployed gears in sand habitats (Table 4.3) and had an Ind_{Rec} less than one in 2007, resulting in a mean Ind_{Rec} of 0.38 for the whole habitat. Scallop dredgers in gravel and otter trawls in muddy sand and reefs were also deployed at frequencies unlikely to allow sufficient time for the habitat to recover.

Recovery was possible in 90% or more of the bottom-fished area of sand and mud, decreasing to approximately two-thirds of the fished area of muddy sand and half of gravel habitats (Table 4.5). In reef habitat 20% (129 km^2) was fished at low enough intensity to allow for recovery. For 147 km^2 of reef no estimate could be made of whether recovery was possible.

4.3.3 Cumulative impact assessment

Aggregate extraction took place in 135 km^2 of UK seabed in 2007; in 40 km^2 of which benthic fishing also occurred. The pressures were found to be concurrent in sand (20.3 km^2) and gravel (19.8 km^2), in which scallop dredges, beam and otter trawls were deployed. Table 4.6 shows the size of areas for particular habitat-fishing gear combinations in aggregate extraction sites. The intensity score of benthic fishing in these locations was less than 1.0 in most habitat-gear combinations, so estimated recovery times km^{-2} from fishing were lower than the maximum (Table 4.4), except for otter trawling in gravel. The recovery times from aggregate extraction alone were estimated at 7.3 years (σ_M 2.39) in sand and 9.0 years (σ_M 2.10) in gravel (Foden et al. 2009).

Table 4.6: Cumulative impacts of benthic fishing at aggregate extraction sites in the UK, in 2007. See Methods for explanation of recovery times from fishing and aggregate extraction (T_{BioAgg}). Cumulative impacts for four scenarios; I: longest recovery period; II: additive recovery period (fishing recovery + T_{BioAgg}); III: antagonistic recovery period (T_{BioAgg} + 50% fishing recovery); synergistic recovery period (T_{BioAgg} + 150% fishing recovery). d: days, BT: Beam trawl, OT: otter trawl, ScD: scallop dredge.

Habitat fishing gears in aggregate extraction sites	**Area (km^2)**	**Maximum recovery after fishing (from Table 4.4) (d)**	**Mean fishing intensity**	**Estimated recovery time after fishing (d)**	**Estimated recovery time after agg extraction (d) (T_{BioAgg})[a]**	**Cumulative impact (d)** SCENARIOS I	II	III	IV
Sand BT	18.34	182	0.89	163	2666	2666	2829	2747	2911
Sand OT	1.45	1	0.38	0.0	2666	2666	2666	2666	2666
Sand ScD	0.55	2922	0.19	561	2666	2666	3227	2947	3508
Gravel BT	6.36	n/d	0.62	n/d	3287	3287	3287[b]	3287[b]	3287[b]
Gravel OT	3.46	365	1.26	459	3287	3287	3746	3516	3976
Gravel ScD	9.99	2922	0.35	1017	3287	3287	4304	3795	4813

[a] Rates from Foden et al. (2009)

[b] No recovery period for beam trawling in gravel, therefore values are aggregate T_{BioAgg} only.

The cumulative impact of aggregate extraction and benthic fishing was estimated from the number of days required for the seabed to recover from these pressures at current levels (Table 4.6). In scenarios III and IV the primary pressure was aggregate extraction and the secondary pressure, fishing, because sand and gravel habitats are more sensitive to the former (Stelzenmüller et al. 2010). Under the four scenarios total recovery times were estimated between 7.3 and 13.2 years. The estimated T_{BioAgg} from aggregate extraction dominated the cumulative recovery time calculations. Gravel habitat required longer periods of recovery than sand habitat from the cumulative impacts of aggregate extraction and benthic fishing, regardless of fishing gear. In gravel habitat cumulative recovery time was greatest from scallop dredging and aggregate extraction under scenarios II, III and most especially from the modelled synergistic effects in scenario IV. Deployment of otter trawls in sand at aggregate extraction sites did not result in a cumulative effect of increased recovery time. The impact of beam trawls and aggregate extraction in gravel could not be estimated (Table 4.4).

4.4 Discussion

The objectives of this study were to calculate, at a fine resolution, the spatial extent and intensity of benthic fishing in UK waters, and to estimate the proportion of fished habitats in which recovery would be possible at 2007 levels of fishing effort. Furthermore the intention was to identify locations where benthic fishing and aggregate extraction coincided, in order to conduct a cumulative impact assessment under the scenarios of greatest, additive and interactive (antagonistic and synergistic) effects.

4.4.1 Fishing distribution and intensity

Benthic fishing is the most widespread pressure acting on the continental shelf, with half to three-quarters trawled annually (Watling & Norse 1998, Kaiser et al. 2002). In mapping the spatial extent and intensity of benthic fishing, it was found that approximately half of the UK seabed was affected in 2006 and 2007. The estimated 134 000 km^2 total bottom-fished area in 2007 is considerably higher than some previous authors' estimates (e.g. Eastwood et al. 2007, Stelzenmüller et al. 2008). This is because the swept area of otter trawling was calculated to be the entire gear width (i.e. 24 m) (Hiddink et al. 2007), rather than only the 2 x 2m trawl doors. Similarly, the area of impact for *Nephrops* trawls was calculated from a gear width of 60 m, as observed by Hinz et al. (2008).

The availability of high resolution VMS data was fundamental to our study, but VMS records have two notable limitations. Firstly, point data do not record the exact tracks of fishing vessels. Calculating the spatial extent of the potential surface area fished around each point will inevitably include some locations that were not fished and exclude areas that were. Nevertheless, this has been an established approach used by others (e.g. Rijnsdorp et al. 1998). Secondly, total fishing effort close to the coast is expected to be underestimated. Vessels of ≥15 m are able to fish within 12 nautical miles, provided they adhere to local Marine and Fisheries Agency (MFA) and Sea Fisheries Committees restrictions, but the majority of their effort occurs in waters further offshore. However, it was not possible to consider the effort by vessels less than 15 m in length, fishing predominantly within 12 nautical miles, because they are not required to carry VMS. A project is underway to build a dataset of inshore fisheries effort from Sea Fisheries Committees' observations, and to compare and integrate this information with that available from vessels carrying VMS. Preliminary results suggest fishing effort in inshore areas is comparable to levels of fishing effort in the adjacent offshore waters (K. Vanstaen, pers. comm.). Further effort will be required to identify habitat-specific recovery rates for these data.

The landscapes identified as most heavily bottom-fished are those representing soft seabed with weak or moderate tide stress (Stelzenmüller et al. 2008), as scallop dredges, beam and otter trawls are generally deployed in soft sediments where the chances of gear loss are small (Rijnsdorp et al. 1998, Piet et al. 2000). In keeping with these findings it was established that the majority of benthic fishing by area was in sand habitats. Fishing intensity varied considerably between habitats, being lowest in sand and highest in mud. The spatial distribution of effort remained similar between 2006 and 2007, an annual consistency noted by Kaiser et al. (2002). Fishing intensity and inter-annual change in intensity between the two years were similar to those found by Stelzenmüller et al. (2008). These authors analysed spatio-temporal patterns of fishing pressure in UK waters since 2001 and found it to vary spatially by region, but within regions, patches of high fishing pressure remained centred at the same locations. Inevitably fishing activity occurs where target species are most abundant (Swain & Wade 2003), resulting in patchy distribution of fishing effort (Rijnsdorp et al. 1998, Stelzenmüller et al. 2008). This also helps explain why the deployment of specific gears broadly remains spatially unchanged over time, as fishing métiers are designed to target particular species.

4.4.2 Recovery from benthic fishing

The intensity of bottom-fishing was such that the seabed habitats in 80% of the area were estimated to have enough time to recover between fishing events. Bottom-fishing at frequencies greatly in excess of estimated recovery time in 19% of remaining habitats means benthic recovery is unlikely to be achieved. Recovery times could not be estimated from our literature review for beam trawling in mud, gravel and reef habitats, or for scallop dredging in mud. However, this was not a serious limitation as these habitat-gear combinations only accounted for approximately one percent of the entire fished area. Notable patterns were identifiable in recovery periods for the five different habitats, and for particular habitat-gear combinations.

The recovery time of habitats was influenced by gear width, gear penetration, fishing frequency and sediment grain size. The literature review showed recovery periods increase with habitat stability or sediment grain size; in general soft sediment habitats recover more rapidly than hard (gravel or reef) habitats. For example, otter trawls were the predominant gear deployed in mud, and benthic recovery after otter trawling is relatively rapid (i.e. days) in this environment (Sanchez et al. 2000, Kaiser et al. 2006). In contrast, recovery from benthic trawls may take several years in hard habitats (Kaiser et al. 2006, Kenchington et al. 2006). Scallop dredging in sand and gravel, and otter trawling in muddy sand and reef habitats had recovery times greater than one, leaving the seabed area unable to recover from bottom-fishing at 2007 levels of fishing frequency. It is hoped this detailed information on habitats' sensitivity to particular gears and levels of intensity will help inform marine managers when making marine planning decisions.

An important area for ongoing research is the chronic modification of benthic communities by long-term bottom-fishing (Jennings & Kaiser 1998, Kaiser et al. 2000; de Juan et al. 2007), which was not considered in our study. Infrequent, pulsed otter trawling has been found to allow fauna to recover more quickly with an overall lesser effect than more intense, chronic, long-term trawling (Henry et al. 2006, Hinz et al. 2009). However, continental shelves are heavily exploited creating a dearth of historically unfished reference sites with similar environmental characteristics to fished sites (Watling & Norse 1998, Kaiser et al. 2002, de Juan et al. 2007). This may severely restrict the possibility of accurately quantifying both acute and chronic impacts of bottom-fishing across the range of seabed habitats.

4.4.3 Cumulative impacts

Aggregate extraction in UK waters is spatially restricted but highly damaging in nature (e.g. Cooper 2005, Eastwood et al. 2007, Vanstaen et al. 2008, Foden et al. 2009, Stelzenmüller et al. 2010). Aggregate was extracted from a seabed area 10^3 less than that affected by benthic fishing in 2007. However, the impacts in terms of time required for the seabed to recovery after cessation of extraction are more severe (Foden et al. 2009).

The fishing industry has expressed particular concern over the potential for cumulative effects in areas where there are local concentrations of marine aggregate extraction licences. Cooper (2005) noted that in general in the UK there is avoidance of areas licensed for aggregate extraction by static gear fishers, due to the potential for gear damage. The spatial pattern of fishing effort in 2007 was found to support this finding; benthic fishing occurred in less than 30% of the area from which marine aggregate was extracted.

There has been little research quantifying the cumulative impacts of bottom-fishing and aggregate extraction on benthic communities and the wider ecosystem (Vanstaen et al. 2008) and our study is a contribution to this area of cumulative effects assessment. Although limited to only two pressures, the four scenarios of cumulative impacts provide useful insights of how benthic fishing and aggregate extraction in combination might affect the seabed. All scenarios showed that, in site-specific activity on sand and gravel habitats, aggregate extraction appeared to be a greater pressure than benthic fishing, regardless of gear type. Estimated recovery from aggregate extraction (T_{BioAgg}) constituted 68 to 100% of the cumulative recovery time for each scenario, where calculations could be made for both pressures. The additive impact modelled in scenario II may be the most appropriate for quantifying cumulative impact for a pair of pressures (Crain et al. 2008). Synergistic effects in scenario IV estimated the longest recovery times for all habitat-fishing gear combinations in aggregate extraction sites. The weighting schemes of scenarios III and IV (Stelzenmüller et al. 2010) should ideally encompass several more pressures, in which aggregate extraction and benthic fishing would be part of a rank order. It is hoped that future empirical data from observations or experiment will enable application of a more complex weighting scheme, making estimations of total impact more comprehensive.

Many other human activities in the marine environment are absent from this study, such as oil and gas extraction, renewable energy structures, dumping, pipelines and cables. Nevertheless, our findings contribute to increased understanding of differing marine habitat sensitivities, and the individual and cumulative impacts on habitats from two major human pressures. Further research into the impact of abrasion, obstruction, smothering, and extraction pressures on the marine environment is underway, using spatial analyses similar to the methods herein. This will enable integrated assessments of cumulative impacts, which can begin to estimate the effects of the multitude of pressures acting simultaneously in many offshore locations.

4.5 Acknowledgements

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CHAPTER 5

Human pressures on UK seabed habitats – a cumulative impact assessment

Foden J, Rogers SI, Jones (2011) Human pressures on UK seabed habitats – a cumulative impact assessment. Mar Ecol Prog Ser 428:33–47 (see Declaration, page 2)

Abstract

European Member States are required to assess the status of marine waters, including analysis of cumulative effects. We developed a methodology for evaluating the impact of several human activities that constitute four direct pressures on the UK (England and Wales) seabed community: smothering, abrasion, obstruction (sealing) and extraction. The method was tested by mapping the spatial extent of individual and cumulative activities of 2007 by habitat type, quantifying the intensity of activities, and estimating impact using published recovery times.

More than half (134 400 km^2) of the seabed was directly affected by human activities, of which only 165 km^2 (<0.1%) was occupied by multiple activities. Benthic fishing accounted for 99.6% of the spatial footprint. Sensitivity to the pressures of human activities varied by habitat type, with estimated recovery times ranging from <1 month for otter trawling in sand, to ~15 yr for co-occurring aggregate extraction and dredge material disposal in low-energy gravel habitat.

Fully integrated, dynamically-linked environmental assessments are generally considered desirable for greater scientific understanding of an ecosystem. The methodology we present to quantify cumulative effects is a step towards this. However, our findings indicate that a limited number of activities were the predominant cause of widespread, long recovery times of benthic fauna. This suggests that when time and resources are limited, single sector assessment rather than detailed evaluation of cumulative effects, can still usefully guide management. As the observed cumulative effects were primarily related to a few activities, it might reasonably be argued that

management effort should be focused on spatially extensive activities, such as benthic fishing to mitigate most of human impact on the UK seabed.

Key words: Abrasion, cumulative impact, extraction, marine habitats, obstruction, recovery, smothering, UK seabed

5.1 Introduction

The UK has commitments to monitor and assess the condition of the marine environment under several international conventions and Directives, including the United Nations Convention on biological diversity (UNEP 1992), the OSPAR Convention for the protection of the North-East Atlantic, and the Habitats (EU 1992), Birds (EU 2009) and Water Framework (EU 2000) Directives. Added to these is the new European Marine Strategy Framework Directive (MSFD), which requires member states to make an assessment of their marine waters with a requirement to achieve Good Environmental Status by 2020 (EU 2008). The recently published Charting Progress 2 (CP2) report was compiled to address many of the obligations under these initiatives and regulations by providing robust evidence for the current and projected state of the marine environment (Defra 2010). However, the conclusions that could be drawn were limited. The authors reported that they had low confidence in the assessment of shallow subtidal sediment habitats and that this was because of a lack of knowledge, data and assessment tools. These habitats, defined as sand, gravel, mud and mixed sediments that are affected by wave action, constitute the majority of the seabed of England and Wales (Figure 1.1). Only 10 % of the seabed was assessed using habitat maps and the remainder was modelled. Furthermore, the authors highlighted the severe lack of understanding regarding the links between human activities and the marine environment. Most significantly there was little understanding of the cumulative impact of several activities in one area and the ability of a species or habitats to recover once a pressure has been removed.

In this chapter some of these knowledge gaps were addressed, with particular reference to the requirements of the MSFD. The Directive requires member states to assess predominant pressures and impacts, including cumulative and synergetic effects. Apart from naturally occurring near-bed currents and wind-induced waves (Hall 1994), the major sources of seabed disturbance in UK waters are caused by human activities (Foden et al. 2010). Under the MSFD, human activities are grouped into generic pressure types, which are useful because ecosystems respond to types of pressure rather than specific activities. Pressures directly affecting the seabed are physical loss (smothering and obstruction) and physical damage (siltation, abrasion and extraction). The MSFD defines and lists examples of activities causing such pressures. From these the following four pressures caused by 12 activities which occur in UK (England and Wales) waters were considered:

- Smothering: covering the natural seabed habitat with a layer of material which, under some circumstances, might be expected to disperse. Smothering activities include disposal of dredged material and cuttings from oil and gas exploration
- Obstruction (termed ‘Sealing’ in the MSFD): permanent structures fixed on the seabed. Obstruction activities include oil and gas platforms, well heads, oil and gas pipelines, telecommunication and power cables, wind turbines and wrecks
- Abrasion: scouring and ploughing of the seabed. Abrasion activities include benthic fishing using trawl gear, burying activity during telecommunication and power cable laying and wind turbine scour
- Extraction: exploitation by removal of seabed resources. Aggregate extraction is the only activity in this pressure type.

Habitats vary in their sensitivity to disturbance from different pressures. Investigations of seabed recovery rates following disturbance provide a method of quantitatively estimating habitat sensitivity (Desprez 2000, Cooper et al. 2007, Foden et al. 2009, Foden et al. 2010). Habitats requiring long recovery periods might be considered more sensitive than those with more rapid recovery rates. If a pressure occurs too frequently for a habitat to recover, the benthic community’s biomass and productivity decline (Hiddink et al. 2006a) and sustainability may be jeopardised. Defining benthic recovery from any type or scale of pressure is problematic. Ecosystem recovery is complex with a range of definitions and metrics used, and existing scientific studies have limitations in their scope (Gilkinson et al. 2005, Hall et al. 2008). This is because complete recovery would be the return of an ecosystem to its original, pre-disturbance state, whereby the abundance, diversity, structure and functioning of the biological community are the same as prior to the disturbance (Hiscock & Tyler-Walter 2006). However, this is unrealistic and most studies focus on the recovery of the key species, assemblages and components of the ecosystem (Hall et al. 2008).

The cumulative effects of coinciding pressures can be additive, antagonistic or synergistic. Antagonism is a cumulative impact value lower than the sum of individual impacts and synergy a value greater than the sum of individual impacts (Folt et al. 1999). These can be difficult to predict (Crain et al. 2008, Darling & Côté 2008). Consequently, with a few notable exceptions (e.g. Stelzenmüller et al. 2010), most previous regional and global scale studies have been limited to assuming cumulative pressures are additive and have presented relative rather than actual impacts (e.g. Halpern et al. 2008b). Quantifying the capacity for habitats to withstand pressures has been identified as a critical step for better understanding of ecosystem resilience (Ban et al. 2010) and will help inform decision-makers in facilitating an ecosystem approach to marine management.

Our study builds on the innovative earlier work of Eastwood et al. (2007) who mapped human activities in UK (England and Wales) waters, and Stelzenmüller et al. (2010) who analysed spatial pressures and marine habitat sensitivity by running scenarios to estimate risk of cumulative impacts. Our objective was to develop a method for examing whether cumulative effects are of spatial or temporal concern in UK waters. To do this, a 'dynamically linked ecosystem assessment' (Foden et al. 2008) was conducted for a range of different sectors, by: (1) mapping the spatial of extent human activities in 2007 at a high resolution; (2) using data on habitat recovery periods as indicators of sensitivity and estimating the proportion of habitats in which recovery would be possible at 2007 levels of activity; (3) investigating where pressures coincided, potentially giving rise to cumulative impacts on the seabed; and (4) where pressures overlap, estimating overall recovery times for four cumulative effects scenarios – greatest, additive, antagonistic and synergistic (e.g. Crain et al. 2008, Darling & Côté 2008, Foden et al. 2010, Halpern et al. 2008a).

5.2 Methods

5.2.1 Study area and habitats

The study area (Figure 1.1) comprised the marine waters of the UK (E & W), as delineated for environmental status reporting under Charting Progress (Defra 2005). Five habitat types were identified, based on the largest proportion of constituent particle size: mud, muddy sand, sand, gravel and reef (including biogenic habitats constructed or composed primarily of living biota). These incorporate European Nature Information System (EUNIS) habitats A5.1, A5.2, A5.3, A5.4, A5.5 and A5.6 (EEA 2004). The habitat types are relevant to the impact of human activities on the seabed (Collie et al. 2000, Kaiser et al. 2006, Pitcher et al. 2009, Foden et al. 2010). Together the habitats constitute >99% of the UK seabed with diamicton (matrix of large and fine grains) or unclassified sediment accounting for the remainder.

5.2.2 Spatial data and processing

To conduct a pressure assessment of human impacts on the seabed spatial data were collated for four pressures and associated activities listed above (Table 5.1). Data for 2007 were used for compatibility with previous impact assessment work (Foden et al. 2009, Foden et al. 2010). Records for each activity were joined to British Geological Survey sediment types (Folk 1954) using the ESRI ArcGIS Geographical Information System (ESRI), and grouped to the habitat types listed above.

Table 5.1: Pressures and activities affecting the UK seabed. Data provided in WGS 84 (world geodetic system 1984) projection in decimal degrees (6 decimal places) (*), or British National Grid eastings and northings at ordinate resolution 0.0001 m (#) – see footnotes.

Pressures and activities	**Data source & description**	**Activity description**	**Manipulation in GIS and footprint area**	**References**
Smothering				
Disposal of dredged material	Cefas[a], SPIRE[b]. Licence area polygons	Disposal occurs within >150 licensed sites. Licensees deposit material over the centre of sites	Dimensions of licensed disposal sites, without buffers	Bolam et al. (2006), Birchenough et al. (2010), S. Pacitto (pers. comm.)
Cuttings from oil & gas exploration	SPIRE[b], UK DEAL[c]. Point data	Cuttings are produced during drilling. Cuttings are separated and disposed to sea. Since 2000 only water-based fluid has been permitted for drilling	Circular buffers of 100 m radius applied to platform and well head data. Area of ~31500 m^2 per point	OSPAR Commission (1993, 2000, 2008) Daan & Mulder (1996), de Groot (1996), Currie & Isaacs (2005), Eastwood et al. (2007), R.S. Rowles (pers. comm.)
Obstruction (sealing)				
Oil and gas platforms	SPIRE[b], UK DEAL[c]. Point data	Four- or six-leg steel structures, each 2 m diameter. Plus associated drilling and production gear	Circular buffers of 7.5 m radius applied. Area of 180 m^2 per platform	UKOOA (2002), Eastwood et al. (2007)
Well heads	SPIRE[b], UK DEAL[c]. Point data	Protective structures built over well heads	Circular buffers of 25 m radius applied to point data. Area of ~2000 m^2 per well head	Eastwood et al. (2007)
Oil and gas pipelines	SPIRE[b], UK DEAL[c]. Line data	Pipelines resting on the surface of seabed	Exact dimensions of pipelines	Eastwood et al. (2007)
Telecommu-nication and power cables	SPIRE[b], SeaZone[e], UKHO[f]. Line data	Exposed cables on rock are armoured to a maximum diameter of 50 mm. Approx 20% cables in soft sediment are not buried	Buffers 50 mm wide for cables on rock substrate and 25 mm on soft sediment	Kogan et al. (2006), Carter et al. (2009), R. Hill,(pers. comm.), UKCPC[d]

Wind turbines	Crown Estate, SPIRE[b], SeaZone[e]. Point data	Monopile foundations 4–5 m in diameter with scour protection of 30 m diameter	Circular buffers of 15 m radius applied. Area of ~700 m^2 per turbine	OSPAR Commission (2006), Rees (2006)
Wrecks	SPIRE[b], SeaZone[e], UKHO[f]. Point data	Sizes of individual wrecks unknown. Nominal spatial extent used	Circular buffers of 17.5 m radius applied. Area of 962 m^2 per wreck	Eastwood et al. (2007)
Abrasion				
Benthic fishing using trawl gear	Cefas[a]. Point data	Vessels ≥ 15 m. Satellite-based Vessel Monitoring Systems (VMS) point data. UK logbook data and the European vessel register for type of gear deployed; grouped as otter trawls, beam trawls or shellfish dredges	Estimates of spatial extent of fishing for each VMS record based on vessel speed, VMS interval and width of fishing gear. Data gridded in 1 km^2 cells. Intensity calculated from annual number of trawl passes	Collie et al. (1997), Rijnsdorp et al. (1998), Dinmore et al. (2003), Stelzenmüller et al. (2008), Foden et al. (2010)
Telecommunication and power cables	SPIRE[b], SeaZone[e], UKHO[f]. Line data	Fibre-optic cables 17–21 mm diameter, protected to a total diameter of 30 mm. Buried in soft sediment by sea plough or water jet	Buffers 5 m wide representing the mean width of trench disturbance	Allan (1998), Carter et al. (2009), Drew & Hopper (2009), UKCPC[d]
Wind turbine scour	Crown Estate, SPIRE[b], SeaZone[e]. Point data	Waves and tides around turbines cause scour pits in mobile sediment, up to 10 times the diameter of the obstruction	Circular buffers of 50 m radius applied. Area of ~7850 m^2 per turbine, minus area of scour protection (see Obstruction)	Rees (2006)
Extraction				
Aggregate extraction	Crown Estate, Cefas[a], SPIRE[b]. 50 x 50m polygons	Electronic Monitoring Systems (EMS) data in 50 x 50 m (2500 m^2) cells showing location and hours dredged per year	EMS 50 x 50 m cell locations and dredge intensity	Dickson & Lee (1972), Newell et al. (1998), Boyd et al. (2004), Kenny & Rees (1994), BMAPA (2006), BMAPA (2008), Foden et al. (2009, 2010), K. O'Shea (pers. comm.)

[a]Centre for Environment Fisheries and Aquaculture Science (Cefas)*, [b]Shared Spatial Information Services (SPIRE) (https://secure.services.defra.gov.uk/)*, [c]United Kingdom Offshore Oil & Gas Industry, Common Data Access Ltd (www.ukdeal.co.uk)[#], [d]United Kingdom Cable Protection Company (UKCPC) (www.ukcpc.org.uk), [e]British Crown and SeaZone Solutions Limited*, [f]United Kingdom Hydrographic Office (UKHO)*

The two main causes of smothering in UK waters are the disposal of material from harbour dredging (creating dredging spoils) and the discharge of drill cuttings at oil and gas platform drilling rigs. Disposal occurs in defined licensed areas and licensees are generally guided to dispose of material in the centre of the site in an attempt to restrict plumes (S. Pacitto pers. comm.). When an oil or gas well is drilled, waste cuttings are separated on the platform and are normally discharged to the seabed (Kingston et al. 1987, Breuer et al. 2004). By January 2001 oil- and synthetic-based muds could no longer be released into the environment (OSPAR Commission 2000, 2009). The assumption was that during the intervening decade, recovery from drilling with these muds would have occurred (Daan & Mulder 1996). Consequently only the effects of water-based muds (WBM) cuttings were considered for well heads and platforms in operation during or since 2001. Although cuttings piles will vary in size and shape, WBM-contaminated cuttings have been reported to reach approximately 100 m from the well (Daan & Mulder 1996, Currie & Isaacs 2005, Zuvo et al. 2005), which we used as a standard dimension.

Potential causes of obstruction in UK waters include: oil and gas platforms, well head protective structures, pipelines, exposed cables, wind farm turbines and wrecks. Individual platforms, well head structures, wind turbine scour protection and wrecks vary in size and shape, but as specific information was not available standard dimensions were used to generate representative footprints (spatial extent estimates) for these activities (Table 5.1). Dimensions were available for individual pipelines and armoured telecommunication cables overlying rock so their footprints could be accurately represented. In soft sediment telecommunication cables are generally buried, but to account for a proportion which cannot be buried, ~20% was assumed to be exposed and a standard cable width was used (R. Hill pers. comm.).

Abrasion in UK waters is caused by benthic fishing, wind turbine foundation scour and burial of power cables. The most important human pressure, in terms of spatial extent and level of impact, results from fishing using benthic trawl gear such as beam trawlers, otter trawlers and shellfish dredges (e.g. Collie et al. 1997, Rijnsdorp et al. 1998, Dinmore et al. 2003, Stelzenmüller et al. 2008). Recovery from fishing is gear-dependent and may also depend on frequency of trawl passes (Kaiser et al. 2006, Hall et al. 2008). Published estimates of spatial variability and intensity of fishing activity in UK waters in 2007 were used (Foden et al. 2010). Intensity was accounted for as follows: e.g. if a beam trawler sweeps the entire area of a 1 km^2 cell four times a year, fishing intensity was set to be 4.0 and the mean recovery time for the cell was estimated as the recovery time from one pass x 4 (*sensu* Foden et al. 2010). Abrasion caused by hydrodynamics around individual turbine foundations can create scour pits of 100 m diameter (Rees 2006). This was used as a standard footprint for all turbines. The majority of offshore cables in UK waters are buried using sea-ploughs or water jets (Allan 1998, Carter et al. 2009, Drew & Hopper 2009). The overall disturbance strip ranges from two to eight metres (Carter et al. 2009) and the mean width was used for all buried cables.

Aggregate dredging for mineral resources constitutes extraction pressure in UK waters. Published estimates of aggregate extraction effort in UK waters during 2007 were used (Foden et al. 2009).

With GIS, estimations were made of the location and areas of seabed habitats affected by individual and by coincidental activities. Activities were also grouped by the four pressure categories to estimate the location and areas affected by individual or cumulative pressures. The footprint estimates of each activity were attributed a confidence rating on a scale of 1 to 3; 1 indicating the highest confidence rating in which location and extent of an activity's footprint were accurately known, 2 indicating known location but estimated extent, and 3 indicating the lowest confidence based on estimations of location and extent (*sensu* Eastwood et al. 2007).

5.2.3 Recovery

Seabed habitat sensitivity to different anthropogenic activities was estimated by determining recovery rates of the benthic community following cessation of an activity, and based on the activity's distribution and intensity. Recovery was characterised as having occurred when the abundance, species richness or biomass of benthic biota was equivalent to a 20% reduction or less in the pre-impact value (Kaiser et al. 2006), or a return of benthic resources to either a baseline (pre-impact) or reference condition (Wilber et al. 2008). Recently published estimates of habitat recovery after aggregate extraction (Foden et al. 2009) and benthic fishing (Foden et al. 2010) were used. A review was conducted of scientific literature for recovery of the benthos from the remaining human activities. Our study area is in temperate waters where primary and secondary production are large with high seasonal patterns, so data from studies conducted in this, or similar areas, were used. UK waters have a long history of high levels of human activity, with many pressures tending to repeatedly target the same grounds year after year (Kaiser et al. 2002, Hiddink et al. 2006b). For some habitats therefore, the point at which recovery is deemed to have occurred is a point in a constant disturbance cycle, and not disturbance of a pristine benthic community.

For some activities the date of occurrence can be important when determining a site's stage of recovery. The timing of different activities was known with varying levels of precision (e.g. day or month) for four activities: dredge material disposal, fishing, cable burial and aggregate extraction. This information was used to estimate the degree of recovery already reached by 2007 and to filter out activities old enough for full recovery to be assumed. Drilling dates were not available for well heads and the spatial extent of resultant cuttings piles is likely to be an overestimation, as recovery was probably well underway at sites where dispersal had occurred. Date of installation is irrelevant for areas of the seabed permanently sealed by some obstruction activities, as no recovery is possible for the duration of the activities' presence. Similarly, date of wind farm construction was not relevant for scour pits associated with turbines, as they represent a constant abrasion pressure.

5.2.4 Cumulative impact

The size and location of multiple activities and pressures were identified, as described above. Activities representing obstruction pressure were considered to be exclusive of in-combination effects. Where the seabed has been effectively sealed by an installation, benthic recolonisation is prevented and extra activities cannot have further impacts. For example, disposal may occur on top of a wreck, or a benthic fish trawl may pass over a well-head but they can create no more damage. Within each habitat the size and location of areas where activities coincided were estimated; the estimation of total recovery times is described below.

Where activities were coincident, cumulative recovery times were estimated according to the intensity of the activity and habitat in which they occurred. Estimates were made under four different cumulative effects scenarios: single greatest, additive, antagonistic and synergistic (Halpern et al. 2008a, Foden et al. 2010). This allowed us to determine the sensitivity of the scenarios to different measures of impact estimation, according to habitat type. The premise for Scenario I was that the single worst or dominant pressure takes precedence over the others in determining combined effects; with lesser pressures having no additional impact. For Scenario II, multiple pressures were assumed to act independently within the system and therefore overall recovery time was the sum of all pressures (e.g. Halpern et al. 2008a, Ban et al. 2010).

The purpose of Scenarios III and IV was to show a range in the sensitivity of habitats to impacts that interact. Scenario III estimated cumulative impacts as the antagonistic effects of multiple pressures. Previous investigations have found marine landscapes to be more sensitive to some human activities than others. If pressures are applied consecutively to marine habitats, then the impact of the primary pressure may pre-condition the habitat to be less sensitive to the secondary pressure. To estimate total recovery time, a linear calculation was used *sensu* Stelzenmüller et al. (2010); recovery time from the primary pressure + 50% recovery time from the secondary pressure, + 0% from the third pressure. Total recovery times were expected to be between those of Scenarios I and II. In Scenario IV synergistic effects were assumed, in which the impact from accumulated pressures was greater than the sum of the individual parts, the assumption being the first pressure lessens the resilience of a habitat, making it more sensitive to subsequent pressures. Therefore in Scenario IV we estimated total recovery time using the same linear relationship: recovery from the primary pressure + 150% recovery time from secondary pressure + 200% from the third, with the expectation of total times greater than for the other three scenarios. A rank order of pressures needed to be determined for Scenarios III and IV. Stelzenmüller et al. (2010) scored the sensitivity of UK marine landscapes to a range of pressures and found that in general landscapes were most sensitive to extraction, and were slightly more sensitive to smothering than abrasion pressures.

Table 5.2: Estimates of spatial extent of human activities and pressures affecting the UK seabed (km^2) in 2007. Percentages of habitat and seabed affected are in italics. Confidence in spatial data; 1: known location and extent, 2: known location and estimated extent, 3: estimated location and extent.

Pressure	**Human activity**	**Confidence**	**FOOTPRINTS PER HABITAT**					**FOOTPRINTS ON UK SEABED**	
			Sand	Gravel	Muddy sand	Reef	Mud	Per activity	Per pressure (overlapping activities merged)
Smothering	Dredge material disposal	3	110.8 *0.06*	89.6 *0.22*	61.0 *0.26*	21.9 *0.43*	0.2 *0.01*	283.5 *0.11*	346.01 *0.14*
	Cuttings from well-heads & platforms	2	52.6 *0.03*	6.2 *0.02*	3.6 *0.02*	0.03 *<0.01*	0.1 *0.05*	62.6 *0.02*	
Smothering per habitat (overlapping activities merged)			163.4	95.8	64.5	21.9	0.34		
Obstruction	Oil & gas platforms	2	0.8 *<0.01*	<0.1 *<0.01*	<0.1 *<0.01*	0.0 *0.00*	0.0 *0.00*	0.8 *<0.01*	21.1 *<0.01*
	Well-heads	2	4.2 *<0.01*	0.5 *<0.01*	0.3 *<0.01*	<0.1 *<0.01*	<0.1 *<0.01*	5.0 *<0.01*	
	Oil & gas pipelines	1	3.1 *<0.01*	0.7 *<0.01*	0.2 *<0.01*	0.0 *0.00*	<0.1 *<0.01*	4.0 *<0.01*	

Table 5.2: continued

Pressure	Human activity	Confidence	FOOTPRINTS PER HABITAT					FOOTPRINTS ON UK SEABED	
			Sand	Gravel	Muddy sand	Reef	Mud	Per activity	Per pressure (overlapping activities merged)
Obstruction	Submarine cables	1	0.2 *<0.01*	<0.1 *<0.01*	<0.1 *<0.01*	<0.1 *<0.01*	0.0 *0.00*	0.3 *<0.01*	
	Wind turbines	1	0.1 *<0.01*	0.1 *<0.01*	0.1 *<0.01*	0.0 *0.0*	0.0 *0.00*	0.2 *<0.01*	
	Wrecks	2	6.7 *<0.01*	3.4 *<0.01*	1.7 *<0.01*	0.4 *<0.01*	0.1 *<0.01*	12.4 *<0.01*	
Obstruction per habitat (overlapping activities merged)			14.4	4.4	1.9	0.4	0.1		
Abrasion	Benthic fishing	2–3	93946.2 *50.56*	19893.4 *49.68*	18088.2 *76.25*	647.3 *12.73*	1324.7 *71.89*	133899.7 *52.2*	133909.59 *52.20*
	Wind farm scour pits	2	0.6 *<0.01*	0.6 *<0.01*	0.5 *<0.01*	0.0 *0.00*	0.0 *0.00*	1.7 *<0.01*	
	Submarine cable burial	2	12.3 *<0.01*	4.2 *<0.01*	1.4 *<0.01*	0.0 *0.00*	0.1 *<0.01*	18.0 *<0.01*	
Abrasion per habitat (overlapping activities merged)			93,952.3	19,896.8	18,089.3	647.3	1,323.9		
Extraction	Aggregate extraction	1	51.9 *0.03*	92.4 *0.23*	1.9 *<0.01*	0.0 *0.00*	0.0 *0.00*	146.3 *0.05*	146.3 *0.05*
Footprint of all pressures, per habitat			94182.0 *50.69*	20089.5 *50.17*	18157.6 *76.54*	669.6 *13.17*	1324.3 *71.87*		

5.3 Results

5.3.1 Spatial distribution of pressures

Aggregate extraction and three obstruction activities – pipelines, cables and wind turbines – were all at the highest confidence level (Table 5.2), because their location and extent were available from the data source (Table 5.1). The location of dredge disposal is only accurate to licence areas not the exact dumping site and this was assigned the lowest confidence rating. The location of benthic fishing vessels is provided at a 2 hourly frequency from VMS. However, in this time vessels can cover up to 12 nautical miles whilst fishing (Lee et al. 2010) and, as an estimate of the exact tracks of vessels was required for this work, confidence was rated as 2 to 3 (intermediate-to-low). Data for the remaining activities were at the intermediate confidence level because their locations were known, but the extent of impact was estimated.

The total area affected by human activity was 134 400 km^2, constituting 52% of the UK seabed (Table 5.2). Abrasion was the main pressure. Specifically, benthic fishing accounted for most of the abrasion pressure, affecting an area up to three orders of magnitude greater than for any other activity in any habitat. The total area affected by obstruction, extraction and smothering pressures was only 513.4 km^2, constituting ~0.2% of the study area. Smothering was the second largest pressure, mainly accounted for by dredge material disposal. The majority of mud and muddy sand habitats were affected by human activities, with an estimated >70% of their area affected. Human activities occurred in approximately half of the area of sand and gravel and only 13% of reef habitats. As benthic fishing was the major cause of human pressure but its confidence rating was two-three, then in general overall confidence in the location and extent of human activity on the UK seabed in 2007 could be classed as intermediate-to-low.

In total only an estimated 166 km^2 (0.07%) of UK seabed was affected by cumulative pressure (Table 5.3). Smothering, abrasion and extraction pressures were coincident in relatively small proportions of habitat areas. Smothering and abrasion accounted for the largest areas of in-combination pressures. These two pressures coincided in 71

km^2 (<0.1%) of sand, and 45 km^2 (0.2%) of muddy sand. In all other cases, two or three combined pressures were coincident in <0.1% of habitat areas.

Table 5.3: Estimated areas of coincidental pressures (km^2) affecting UK seabed habitats (percentages of habitat area affected are in italics).

Coincidental pressures	**HABITAT**				
	Sand	Gravel	Muddy sand	Rock	Mud
Smothering + abrasion	71.1 *0.04*	10.2 *0.03*	44.6 *0.19*	0.7 *0.01*	0.1 *0.01*
Abrasion + extraction	13.6 *0.01*	18.5 *0.05*	0.0 *0.00*	0.0 *0.00*	0.0 *0.00*
Extraction + smothering	2.2 *<0.01*	3.7 *0.01*	0.0 *0.00*	0.0 *0.00*	0.0 *0.00*
Smothering + extraction + abrasion	0.1 *<0.01*	0.9 *<0.01*	0.0 *0.00*	0.0 *0.00*	0.0 *0.00*

5.3.2 Recovery

For reasons stated above, no recovery estimates were made for areas where the seabed was sealed by obstructions. Therefore recovery estimates are not given for locations in which oil and gas platforms, wellheads, pipelines, wind turbines, wrecks or surface laid cables were present. The precision of published recovery rates of the benthic community from the other commonly occurring human activities was variable, e.g. quoted as days, months or years. These were rationalised as months or years, so that recovery rates in Table 5.4 were in the range less than one month to nine years.

Table 5.4: Recovery times for habitats by seabed pressure (smothering, abrasion or extraction) and activity. Strong hydrodynamics: shallow (≤20 m), strong wave action or tidal currents, low residence time (~days), high turbidity and sediment movement. Weak hydrodynamics: >20 m deep, non-turbulent, low circulation sites. mo: months, yr: years, n/d: no data available, n/r: no recovery until the activity ceases, n/a: not applicable (the activity does not occur in the habitat, in UK waters).

Pressure	**Human Activity**	**Environment**				
Smothering	Dredge material disposal			Estuarine (polyhaline)	Coastal (euhaline)	
		Strong hydrodynamics		1–9 mo[a,b]	≤1 yr[a,b]	
		Weak hydrodynamics		≤2 yr[a]	1–4 yr[a,b,c,d]	
		Habitat				
		Sand	*Gravel*	*Muddy sand*	*Reef*	*Mud*
	Oil & gas cuttings	< 1 yr[e,f]	> 11 mo (well advanced)[g]	n/d	n/d	n/d
Abrasion	Beam trawling	< 6 mo[h]	n/d	< 8 mo[i]	n/d	n/d
	Otter trawling	< 1 mo[i]	< 1 yr[j]	> 7 mo[k]	> 8 yr[i]	< 1 mo[i]
	Shellfish dredging	> 8 yr[i,l]	>8 yr[i]	1.6 yr[i]	3.2 yr[i]	n/d
	Wind turbine scour	n/r	n/r	n/r	n/a	n/a
	Cable burial	1–12 mo[m,n]	< 1 yr[n]	< 1 yr[n]	n/a	< 4 mo[o]
Extraction	Aggregate extraction	7.3 yr[p]	9.0 yr[p]	n/d	n/a	n/a

[a]Bolam & Rees (2003), [b]Wilber et al. (2008), [c]Borja et al. (2009), [d]Wilson et al. (2009), [e]Daan & Mulder (1996), [f]Daan et al. (1994), [g]Currie & Issacs (2005), [h]Kaiser et al. (1998); [i]Kaiser et al. (2006); [j]Kenchington et al. (2006); [k]Ragnarsson & Lindegarth (2009); [l]Gilkinson et al. (2005), [m]Guerra-García & García-Gómez (2006), [n]Andrulewicz et al. (2003), [o]Sparks-McConkey & Watling (2001), [p]Foden et al. (2009)

The impacts on the benthos from smothering are site-specific. Recovery from dredge material disposal is context dependent and thus does not conform to a single ecological model (Bolam & Rees 2003, Bolam et al. 2006, Whomersley et al. 2010). In general, communities adapted to strongly hydrodynamic environments recover significantly more rapidly than those in weakly hydrodynamic environments (Bolam & Rees 2003, Bolam et al. 2006, Wilber et al. 2008). The recovery rates quoted in Table 5.4 are based on salinity, hydrodynamics and water depth of the receiving environment. Our estimates of recovery from cuttings contaminated with WBM were based on recovery from physical smothering, because these are quantified in the literature. However, WBM drilling wastes may contain free oil, dissolved aromatic hydrocarbons, heavy metals and radionucleides, and recent studies suggest the response of the benthic community may be through oxygen depletion (e.g. Trannum et al. 2010). However, benthic recovery has yet to be quantified. Recovery rates from physical smothering depend on particular combinations of sediment characteristics, the local hydrodynamic regime, receiving habitat and benthic community (Kröncke et al. 1992, Daan et al. 1994, Holdway 2002, Kröncke & Bergfeld 2003, Breuer et al. 2004). Although recovery rates were only available for sand and gravel environments, 94% of cuttings piles in the UK are in these habitats (Table 5.2).

Abrasion pressures may be caused by regularly-occurring, constant, or one-off activities (Table 5.4). Recovery rates from bottom-fishing were summarised from Foden et al. (2010), who estimated recovery of the benthos by fishing gear-type and intensity. The evidence for the effects of fishing frequency is mixed, i.e. within a single habitat type the same frequency of trawl events can lead to differing responses of the benthos in different locations. Therefore a minimum recovery time of one year in Table 5.5 was used, which also allows for seasonality in the ability of the benthic community to recover (Hall et al. 2008, CLJ Frid pers. comm.). The habitat-gear combinations for which there were no recovery rates only represent approximately one percent of the area subjected to benthic fishing in 2007. Scour pits around the foundations of wind turbines are likely to be a constant abrasion pressure. Routine sampling is not possible within 50 m of a turbine (OSPAR Commission 2008), so it was assumed that no benthic recovery is possible from scour during the lifetime of wind farms. Wind farms are licensed for 25 years (A. Judd, pers. comm.) and to date none have been decommissioned in UK waters. Outside the scour zone, between turbines, there is little or no evidence for benthic disturbance caused by the wind farm (NPower

Renewables 2008, Degraer & Brabant 2009, Cefas 2010). Cable installation is of limited spatial and temporal extent, unless a submarine cable is damaged, and recolonisation may be rapid (Guerra-García & García-Gómez 2006). No data were available specifically on recovery from cable burial in mud habitats. However, a study of low intensity benthic trawling effects in a low energy 60 m deep mud environment was appropriately comparable as it mirrors the mud habitat in UK waters.

The only activity representing extraction pressure is aggregate dredging. Recovery estimates in Table 5.4 for this activity were taken from Foden et al. (2009).

Table 5.5: Impact from human activities in UK waters, in 2007. Recovery period estimates for individual activities and cumulative impact recovery estimates for four scenarios; I: longest recovery period; II: additive recovery period; III: antagonistic recovery period (100% 1st pressure + 50% 2nd pressure + 0% 3rd pressure); IV synergistic recovery period (100% 1st pressure + 150% 2nd pressure + 200% 3rd pressure). Fishing gear, mean fishing intensity (where 1 = entire 1 km^2 cell is fished yr^{-1}) and period of recovery from fishing are taken from Foden et al. (2010), with a minimum recovery period of 1 yr (C.L.J. Frid pers. comm.). BT: Beam trawl, OT: otter trawl, SD: shellfish dredge, yr: years.

Habitat	Cumulative pressures	Recovery estimates for individual activities	Affected area (km^2)	Recovery estimates for cumulative impact -------- scenarios (yr)------			
				I	II	III	IV
Sand	Smothering + abrasion	Disposal (estuarine strong dynamics) 0.75 yr, BT (intensity 1.86)[a] 1 yr[b]	4.0	1	1.8	1.3	2
		Cuttings & disposal (coast strong dynamics) ≤1 yr, BT (intensity 0.81)[a] 1 yr[b]	18.3	1	≤2	≤1.5	≤2.5
		Disposal (coast weak dynamics) 1–4 yr, BT (intensity 1.11)[a] 1 yr[b]	35.6	≤4	≤5	≤4.5	≤5.5
		Cuttings & disposal (coast strong dynamics) ≤1 yr, OT (intensity 1.07)[a] 1 yr[b]	8.2	1	≤2	≤1.5	≤2.5
		Disposal (coast weak dynamics) 1–4 yr, OT (intensity 1.41)[a] 1 yr[b]	3.3	≤4	≤5	≤4.5	≤5.5
		Cuttings & disposal (coast strong dynamics) ≤1 yr, SD (intensity 0.18)[a] 1.4 yr	1.8	1.4	≤2.4	≤1.7	≤3.1
	Extraction + abrasion	Extraction 7.3 yr, BT (intensity 0.89)[a] 1 yr	11.6	7.3	8.3	7.8	8.8
		Extraction 7.3 yr, OT (intensity 0.38)[a] 1 yr	1.5	7.3	8.3	7.8	8.8
		Extraction 7.3 yr, SD (intensity 0.19)[a] 1.5 yr	0.6	7.3	8.8	8.1	9.6
	Extraction + smothering	Extraction 7.3 yr, Disposal, coast strong dynamics ≤1 yr	2.2	7.3	8.3	7.8	8.8
	3 pressures	Extraction 7.3 yr, Disposal, coast strong dynamics ≤1 yr, BT (intensity 0.27)[a] 1 yr[b]	0.1	7.3	≤9.3	≤7.8	≤10.8

Table 5.5: continued

Gravel	Smothering + abrasion	Cuttings & disposal (coast strong dynamics) ≤1 yr, BT (intensity 1.16)[a] 1 yr[b]	0.2	1	≤2	≤1.5	≤2.5
		Disposal (coast weak dynamics) 1–4 yr, BT (intensity 0.85)[a] 1 yr[b]	2.9	≤4	≤5	≤4.5	≤5.5
		Cuttings & disposal (coast strong dynamics) ≤1 yr, OT, (intensity 0.72)[a] 1 yr[b]	1.0	1	≤2	≤1.5	≤2.5
		Disposal (coast weak dynamics) 1–4 yr, OT (intensity 1.58)[a] 1.6 yr	5.9	≤4	≤5.6	≤4.8	≤6.4
		Cuttings & disposal (coast strong dynamics) ≤1 yr, SD, (intensity 0.40)[a] 3.2 yr	0.2	3.2	≤4.2	≤2.6	≤5.8
		Disposal (coast weak dynamics) 1–4 yr, SD (intensity 0.01)[a] 1 yr[b]	<0.1	≤4	≤5	≤4.5	≤5.5
	Extraction + abrasion	Extraction 9 yr, BT (intensity 0.62)[a] 1 yr[b]	5.9	9	10	9.5	10.5
		Extraction 9 yr, OT (intensity 1.26)[a] 1.3 yr	3.0	9	10.3	9.7	11
		Extraction 9 yr, SD (intensity 0.35)[a] 2.8 yr	9.6	9	11.8	10.4	13.2
	Extraction + smothering	Extraction 9 yr, disposal (coast strong dynamics) ≤1 yr	1.9	9	10	9.5	10.5
		Extraction 9 yr, disposal (coast weak dynamics) 1–4 yr	1.9	9	13	11	15
	3 pressures	Extraction 9 yr, disposal (coast strong dynamics) ≤1 yr, BT (intensity 0.18)[a] 1 yr[b]	0.9	9	11	9.5	12.5
Muddy sand	Smothering + abrasion	Disposal (coast strong dynamics) ≤1 yr, OT (intensity 1.5)[a] 1 yr[b]	44.6	1	≤2	≤1.5	≤2.5
Reef	Smothering + abrasion	Disposal (coast weak dynamics) 1–4 yr, SD (intensity 0.15)[a] 1 yr[b]	0.7	≤4	≤5	≤4.5	≤5.5
Mud	Smothering + abrasion	Cuttings only (no data), OT (intensity 1.03)[a] 1 yr[b]	<0.1	1[c]	1[c]	1[c]	1[c]

[a]Foden et al. (2010).

[b]Minimum recovery period of 1 year for all fishing gears and fishing intensity.

[c]No recovery period for one of the activities, therefore values for Scenarios II, III and IV are for one pressure only

5.3.3 Cumulative impact

Co-occurring activities were found in all habitat types (Table 5.5). The footprints of coinciding activities were largest in sand (87.2 km^2) and smallest in mud (<0.1 km^2) habitats. However, these constitute <0.5 % of each of the five habitat types. Values are stated to an accuracy of 0.1 km^2 to reflect the variation in confidence levels in the locational accuracy of activities and the spatial extent of their impact (from Table 5.2). Some combinations of activities were not found. For example, in weakly hydrodynamic estuarine environments disposal did not occur where there were extraction or abrasion pressures. Wherever abrasion coincided with extraction or smothering, benthic fishing was the causative activity of abrasion pressure. Submarine cables and wind farm scour pits were exclusive of any other activity.

Cumulative recovery rates across the four scenarios ranged from less than one to 15 years. In Scenario I, recovery time estimates were less than four years for all combinations of smothering and abrasion activities, but estimates doubled (7.3–9 years) where extraction comprised one of the co-occurring activities. A similar pattern was repeated for Scenarios II, III and IV; aggregate extraction consistently accounted for the longest cumulative recovery times. Consequently, sand and gravel habitats were estimated to have the longest recovery periods for Scenario IV, at up to 10 or 15 years, respectively, because there was no aggregate extraction in the other three habitats. Recovery estimates were most rapid for all scenarios in muddy sand habitats (up to 2.5 years), where disposal and otter trawling in a strongly hydrodynamic environment were the only coincidental activities. From the values given in Table 5.4 reef habitats might have been expected to recover slowly, but recovery from coincidental activities was estimated at up to 5.5 years. The likely explanation is that scallop dredging was at the very low intensity of 0.15 (i.e. approximately once per km^2 every seven years), which could allow for the benthos to begin recovering between trawls and dredge material disposal events. Mud habitat was least affected by cumulative activity (<0.1 % of habitat). Benthic recovery from drill cuttings in mud was not known, so cumulative times could not be estimated for combined smothering and abrasion pressures in this habitat. However, this mud habitat only represents an area <0.1 km^2.

5.4 Discussion

This study builds on previous assessment of human activities causing direct, physical pressure on the UK seabed (Eastwood et al. 2007) in three key ways: by quantifying the intensity of relevant activities, by linking the spatial extent of activities to habitat type, and by estimating their cumulative impact using published recovery times. Our methods and findings are relevant to several European and global obligations to assess and report marine environmental status, principally those commitments requiring greater knowledge and understanding of individual and cumulative effects from different human activities. Using a typical year's data a snapshot of the spatial extent of human activities acting on the UK seabed in 2007 was provided, by applying the framework for evaluating individual and cumulative impacts proposed by Foden et al. (2010). Where activities were coincidental four impact scenarios were applied to assess the range of possible consequences.

In 2007 cumulative activities were relatively rare, in total affecting an area of only 166 km^2 (<0.1 % of the study area). The majority of the footprint of human activities was caused by the activity of single, rather than multiple, sectors. Abrasion had the largest spatial extent of the four pressures and just one activity, benthic trawling, accounted for 99.99% of abrasion by area. Benthic fishing affected more than half of UK (E & W) waters, as compared with 0.2 % affected by all the other 11 activities combined. Inter-annual change in this pattern is predicted to be small. Previous work examining temporal changes in fishing pressure in UK waters found strong spatio-temporal correlation in fishing intensity between 2006 and 2007 (Pearson's $r = 0.405$, $p < 0.001$) (Foden et al. 2010). The other human activities are of more restricted spatial extent, e.g. pipelines, dredge disposal licence areas, oil and gas platforms. Therefore the locations and sizes of coincident activities are unlikely to be highly variable over time. To control the consequences of human pressures on the marine environment it could reasonably be argued that, in terms of extent, the assessment and control of spatially limited cumulative impacts is relatively unimportant. The remaining concern relates to consequences of these combined activities on benthic recovery, and the extent to which they are sustainable.

Estimates of recovery rate for single sector activities were less than one month to nine years, while recovery ranged from one to 15 years for cumulative activities under the four scenarios. The largest activity-habitat combinations were beam and otter trawling in sand and gravel, where recovery of the seabed community might reasonably be expected within one year. In contrast, the recovery rates from aggregate extraction were substantially greater, although the spatial footprint was very restricted, comprising only <0.01 % that of benthic fishing. In the small areas where cumulative effects occurred, abrasion and smothering in sand and muddy sand habitats accounted for the majority of coinciding pressures. These pressure-habitat combinations had recovery estimates of up to 5.5 years, while other cumulative impact scenarios suggested the benthic community would require a decade or more to recover. Wherever recovery estimates were ≥9 years, aggregate extraction accounted for the largest proportion of that time period.

These results provide quantitative estimates of spatial extent and recovery times of habitats, which are an important addition to the assessment of UK marine ecosystems already undertaken (Defra 2010). This national assessment of benthic habitat condition was based on expert judgement and drew upon limited evidence from monitoring studies and research. Its conclusion states "large areas of subtidal sediments in most regions have been adversely affected by mobile fishing gears such as bottom trawls and dredges … [and] locally, extraction of aggregates has damaged the seabed in the Eastern Channel and Southern North Sea" (Defra 2010, p 31). Given the large footprint of benthic fishing and the slow recovery rate estimates from aggregate extraction, it might be reasonable, at least in the short term, for management measures to focus on these two activities if impact on the seabed is to be mitigated and marine status improved. Indeed, aggregate extraction is already highly regulated and very spatially restricted (DCLG 2002). New statutory regulations for aggregate licensing (DCLG 2007) require the Marine Management Organisation to consider the Habitats and Environmental Impacts Assessment Directives (EU 1992, EU2003) in their decisions. Consequently, decision-making on site licences is already moving towards an ecosystem approach. Furthermore, it has been argued by some commentators that the loss of seabed can be balanced by the socio-economic need for aggregate, especially when sites are selected within an ecosystem-based management system (Petterson 2008, Rabaut et al. 2009). This arguments is more difficult for extensive activities such as fishing, which is amongst the most important factors affecting the ecological state of many marine ecosystems (Jennings & Kaiser 1998, Watling & Norse 1998), and for which management is widely considered to be a high priority (e.g. Pauly et al 2005, Pitcher & Lam 2010). However, in the UK there has been a recent move toward regionalisation with the establishment of Regional Advisory Councils, which have the potential to improve fisheries management.

The methodology for analysing individual and cumulative impacts of human activities on the seabed and its application presented herein, are likely to be appropriate for other locations. Indeed the literature reviews of recovery rates, here and from previous work (Foden et al. 2009, 2010), included international studies with similar environmental conditions to the UK, based on this *a priori* assumption. Waters of the wider European region have comparable pressures as well as environmental characteristics to the UK, and similar responses to those pressures would be expected. The basic framework would also be amenable for use in examining the impact of different kinds of pressures affecting regions of contrasting environmental conditions, although literature reviews or empirical studies of pressure effects would need to be focused on the same characteristics of the region under investigation.

Cumulative impact assessment as a discipline is still in its early stages. Recent studies have mapped spatial extent and intensity of multiple human activities (e.g. Ban & Alder 2008, Halpern et al. 2008b, Selkoe et al. 2009), but few have considered their impact on the receiving habitat (e.g. Ban et al. 2010). These studies are at a variety of scales and in all cases identified greater proportions of overlapping pressures than in this study, generally because they considered pressures on the entire marine ecosystem such as shipping, recreation, aquaculture, and pelagic fishing. Our detailed, quantitative study is an important step on which a holistic ecosystem assessment could be based. However, the more complex and highly integrated the assessment, the more difficult it can be to indicate causality and to predict future scenarios (Foden et al. 2008). For example, recent assessments using multiple parameters to model present and future states are variable in producing quantitative or qualitative conclusions (e.g. Culp et al. 2000, Link et al. 2002, Choi et al. 2005, Chang et al. 2008), which can have consequences for management practices. A compromise is needed between the complexity of a fully integrated ecosystem assessment and its utility for directing management.

There are limitations to this work, for example related to the quality of broad scale habitat maps, and the lack of ground-truthing of recovery times, which have been faced by many previous regional scale assessments (e.g. Halpern et al. 2009, Ban et al. 2010). Recovery rates for single activities, summarised in Table 4, are based on empirical evidence, but there are too few studies of recovery from the effects of combinations of activities to estimate recovery. For this reason we presented four scenarios of cumulative effects from multiple activities. Ideally the scenarios would also have been run to investigate cumulative impact from repeat events of the *same* activity. However, it was not possible to present this within the confines of a single paper, although it would be an interesting area for future study. The use of confidence ratings for our data has been a useful step, particularly for those activities for which the exact locations or the spatial extent of their impact was unknown. Confidence was generally low where generic spatial dimensions were used, e.g. for cuttings piles (*sensu* Eastwood et al. 2007). In reality a markedly patchy distribution of cuttings around production platforms has been noted, depending on site-specific sediment grains size and local hydrography (Kingston et al. 1987). Comparative *in situ* observations of cuttings from oil and gas production would size improve estimates of area affected and biological impacts from smothering, by habitat type. Similarly, the development of scour pits around turbines will be in the direction of local, prevailing hydrodynamics and such site-specificity could not be taken into account for each structure.

Finally, it was assumed that recovery of the natural benthic community could not take place where obstructions were present. However, it is possible that some structures, such as surface-laid cables and pipelines, have either self-buried, or are themselves used as a structure for future colonisation. Nevertheless, they might also be subject to damage and therefore be re-exposed for maintenance and it was not possible to account for these differences in our analyses. Other semi-permanent structures such as well-heads and turbines might provide a hard substrate offering new habitat for benthic fauna and flora (Kogan et al. 2004, 2006, Danish Energy Authority 2006). Such subtle changes in habitat can have implications for higher trophic levels by providing shelter and feeding opportunities for predators. There is some evidence that the artificial reefs created by obstructions such as offshore wind farms can enhance fish populations in two ways, by increasing the availability of prey species living on the turbines and excluding some types of fishing activity from the region, effectively creating a no-take zone (Punt et al. 2009, Reubens et al. 2011). Consideration of these types of trade-off among different activities may be an interesting extension of cumulative impact assessment. Nonetheless, establishment of fauna associated with these hard substrates represents changes in populations, not re-establishment of the ambient benthic community which we defined as recovery.

The response of the benthic community to all human activity was found to be strongly dependent on the type of receiving habitat. This highlights the necessity for accurate, high resolution habitat maps, essential for more precise estimations of impact for effective management and control (Defra 2010). Similarly, higher resolution spatial data would more closely link activities to the receiving habitat. For example, whilst two-hourly VMS signals from fishing vessels are suitable for management purposes at the UK scale, for estimating cumulative effects more frequent data would improve the accuracy in locating coinciding activities. Further scientific observations of the immediate and long-term footprint on the benthos from all human activities are necessary to determine which of the four scenarios is the most likely in the natural environment.

This work is a rigorous, quantitative assessment of direct individual and cumulative anthropogenic impacts on the seabed. It contributes towards understanding the links between humans and the marine environment by assessing the spatial and temporal effects of anthropogenic activities on the benthos. For future assessment and management purposes, in terms of the impact we have examined in this UK-based study, perhaps the most significant finding is that cumulative pressures on the seabed were very spatially restricted in 2007. Recovery times were comparable to some single sectors; therefore single sector activities remain the predominant cause of direct human impact on the benthos. Nevertheless, there is still a need to develop scientific understanding of the linkages across those aspects of the marine environment not considered herein; e.g. water column pollution, pelagic fisheries, shipping and underwater noise. How such considerations might modify our findings is yet to be determined. To implement a full ecosystem-based approach to management would require a coherent and holistic assessment of all such interconnections, some of which may lead to cumulative impacts greater than those considered herein (Foden et al. 2008). Nevertheless, some of the knowledge, data and assessment tool gaps identified in recent national status assessments (Defra 2010) have been addressed, with regards to human pressure and impact on the benthic community. We hope that this approach to assessing the state of marine habitats will enable a more quantitative approach in future reporting against European and international commitments.

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CHAPTER 6

Discussion

If humans are to exploit marine resources in a sustainable way, ecosystem integrity must be preserved. This means the ecology remains diverse, long-lived, productive and resilient despite the human pressures exerted upon it. In recent years there has been a proliferation of national and international policies and regulations designed to deliver sustainability in the marine environment, including the European Union's Environmental Impact Assessment (EU 1985), Habitats (EU 1992), Water Framework (EU 2000) and Marine Strategy Framework Directives (MSFD) (EU 2008a), and the UK's Marine and Coastal Access Act (Defra 2009). Specifically, the MSFD commits member states to achieve Good Environmental Status, providing "ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions" (Article 3(5) MSFD, EU 2008a). Furthermore the use of the marine environment should be "at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations". To achieve the goals of sustainability requires effective marine management, which is based on sound knowledge and understanding of the impact of pressures upon the marine environment.

The research presented in this thesis has developed and applied a methodology for analysing the effect of human activities on the UK seabed. The research was undertaken with the aims of investigating the spatial and temporal pressures and impacts of both individual and cumulative activities on different seabed habitats, by estimating recovery rates of benthic communities following cessation of a pressure of defined magnitude and duration. This chapter draws on the findings of previous chapters to propose a method for conducting an integrated assessment of human pressures and impacts on UK seabed habitats. The implications of the research are examined, its strengths and weaknesses discussed and recommendations for future work explored.

6.1 Summary of principal findings

The purpose of Chapter 2 was to guide development of a practical programme for conducting an effective marine environmental assessment, which would then be used to steer subsequent research for the thesis. The review examined a broad range of aquatic environmental assessments to identify best practice. A key finding of the chapter was that in conducting an environmental assessment, assessors need to pre-determine three components: their definition of what the 'assessment' is to comprise, the methods to be used, and the standards against which measured parameters are judged. However, commonly used terms were often defined or interpreted differently between assessments. Indeed, fewer truly integrated ecosystem assessments have been conducted than the literature search would suggest. The chapter provided an approach to more accurate usage of assessment terminology and a system of categorisation which could simplify the way that assessments are defined and used.

At first more complex, dynamically-linked, fully integrated ecosystem assessments may seem preferable to simpler assessments, but this is not necessarily the case. More important is for every assessment to have a clearly defined purpose and to use accurate terminology, than to be complex. For example, relatively straightforward single and multi-species fish stock assessments, which regularly report extant condition against historic data and scientifically established reference levels and thresholds, provide valuable long-term datasets. Managers are able to take action on the findings of these reports and their long history means there is good awareness of their strengths and limitations. In contrast one-off, highly complex, fully integrated marine ecosystem assessments are difficult to conduct, are time and resource hungry and the resultant report can be complicated to interpret. Often relative rather than quantitative values and thresholds are used, reducing transparency. Managers may face many challenges when trying to act on the findings of such reports, as there are fewer clear messages to use as guidance.

Whilst Chapter 2 was in large part a review of existing studies, the lessons learned in regard to definitions, methods and standards were applied to subsequent chapters. For example, in analysing the pressure and impact of marine aggregate extraction on UK seabed habitats (Chapter 3) terminology and the scope of the assessment were carefully defined. The method of measuring habitat sensitivity in terms of benthic recovery rates were clearly set out and this is a recognised standard previously established in studies of a similar nature. The key findings were that between 2001 and 2007 the spatial extent of this pressure was small, but the environmental impact was high. In total less than 0.1% (321.7 km^2) of seabed was subject to aggregate extraction, but recovery rates of the physical environment and the benthic community were estimated to take decades in the more sensitive marine landscape types. Linking information on habitat recovery potential to marine landscapes and aggregate activity provides a practical tool for use in marine spatial management. Managers could be guided when licensing aggregate extraction activities to avoid the most sensitive marine landscapes and to target the least sensitive.

A similar assessment of pressure and impact from benthic fishing (Chapter 4) was conducted and taken a stage further by combining the findings with those of Chapter 3. Guided by the principal components for a practical and successful integrated environmental assessment set-out in Chapter 1, this was a step towards a cumulative impact assessment. There were three key findings. Firstly, in 2007 benthic fishing was a far more extensive pressure on the UK seabed (~50% of the study area) than aggregate extraction, but exerted less environmental impact. The shorter recovery-time estimates suggest 80% of the fished area would be able to recover between trawl events at the levels of activity recorded in 2007. Secondly, aggregate extraction and fishing were estimated to be coincident in a very small area, only 40 km^2 (<0.01%). Four scenarios of cumulative impact were presented, because there are few empirical studies of recovery from these combined pressures, across the range of habitat types. The third finding was that the impact of extraction accounted for the majority of the estimated recovery times in all scenarios.

In Chapter 5 the following human activities that directly affected the seabed in 2007 were added to the aggregate extraction and benthic fishing data, and investigated for their individual spatial extent and impact: oil and gas production activities, pipelines, telecommunication and power cables, disposal of dredged material, wind turbines and wrecks. The activities were grouped into four pressure types, as specified in the MSFD: smothering, abrasion, obstruction (also known as sealing) and extraction (EU 2008a). They were incorporated into the cumulative impact scenarios proposed in Chapter 4 and there were four key findings. Firstly, more than half of the seabed was affected by human activities and recovery rates across the matrix of activity-habitat combinations ranged from months to decades. Secondly, benthic fishing had the greatest spatial scale of all activities, accounting for 99.6% of the seabed area subjected human pressure in 2007. However, the evidence for fishing frequency effects is mixed, so a minimum recovery time of 1yr was used, which also allowed for seasonality in the ability of the benthic community to recover (Hall et al. 2008, CLJ Frid pers. comm.). Thirdly, the impact on the benthos of smothering did not conform to a single ecological model and studies of recovery have found ecological impacts to be site specific (Bolam & Rees 2003, Bolam et al. 2006, Whomersley et al. 2010). In general, the composition of benthic communities is naturally related to levels of hydrodynamic stress, rather than habitat types per se, and communities adapted to strongly hydrodynamic environments recover significantly more rapidly (e.g. less than one year) than those in weakly hydrodynamic environments (Bolam & Rees 2003, Bolam et al. 2006, Wilber et al. 2008). Deep, non-turbulent, low circulation sites show a slow benthic community recovery rate because ecosystem communities are slower growing and are considered less resilient. Fourthly, recovery of the benthos was not feasible in the presence of some activities. Where the seabed has been sealed the benthic community cannot recover until the obstruction is removed and the seabed becomes available once more. Therefore recovery estimates were not given for locations in which oil and gas platforms, wellheads, pipelines, wind turbines, wrecks or surface laid cables were present. Wind turbine scour is a constant source of pressure, making recovery impossible until removal of the turbines. Finally, less than 0.1% of the seabed was affected by multiple activities. Wherever aggregate extraction was coincident with other activities, extraction accounted for the greatest proportion of estimated recovery time.

In conclusion, a limited number of activities were the predominant cause of widespread, long recovery times of benthic fauna. This suggests that when time and resources are limited, single sector assessments are still useful guides for managers. With the current distribution of activities it might reasonably be argued that effort should be focussed on spatially extensive activities, such as benthic fishing, to mitigate most of human impact on the UK seabed. Single sector planning can still be ecosystem-based, if knowledge of habitat sensitivity influences management decisions.

Cumulative impact maps can inform planning decisions where reductions in human-induced stressors are an explicit goal, and thus it is important that these techniques continue to be tested and refined to provide meaningful results (Ban et al. 2010). Detailed evaluation of the effects on cumulative activities contributes to a more comprehensive picture of pressure and impacts. As demands on the natural resources of the marine environment are predicted to rise, cumulative effects are likely to increase. The work presented in this thesis provides a method for assessing such cumulative impacts.

6.2 Strengths and limitations

Inevitably there are both strengths and limitations to the research presented in this thesis. The main strengths are: a method has been proposed for robustly quantifying, at a high resolution, the location and intensity of activities directly affecting the seabed; the spatial extent of activities has been linked to habitat type, and individual and cumulative impacts have been estimated using published recovery times. Many of the strengths have been discussed at length in the preceding chapters and will not be repeated here.

A particular strength is that the thesis has provided a way of combining spatial data from a variety of sources at different resolutions. The use of geospatial technology enabled human activities to be visualised, combined and quantified, and was found to be an especially valuable tool for identifying cumulative impacts. The method was tested using data from 2007 and is transparent and reproducible. Therefore, similar analyses can be conducted for more recent data in order to carry out an assessment of temporal change in individual and cumulative pressures. An additional strength is that estimations of habitat sensitivity were based on data from previously published studies, which were independent of the work conducted for the thesis. Drawing on information from a broad base of national and international work makes the estimations of benthic community recovery rates more robust. Finally, Chapter 2 contributes to the clarification of environmental assessment methodologies, including the relatively new discipline of cumulative impact analysis. Cumulative pressure maps have already been devised for some regions of the sea to help inform planning decisions (e.g. Halpern et al. 2008, Ban et al. 2010). The approach proposed herein takes mapping a stage further and offers a quantitative method to estimate a range of potential impact scenarios. The main limitations to the work and the ways in which these were addressed are discussed below.

Two different classifications of 'habitat' were used in the thesis. Aggregate activity was linked to marine landscape types in Chapter 3 because aggregate extraction recovery rates are determined by a combination of depth, tidal stress and grain size (Foden et al. 2009), which are characteristics that classify marine landscape types (Connor et al. 2006). Impacts from fishing, however, relate primarily to sediment grain size (Kaiser et al. 2006, Pitcher et al. 2009) and this single characteristic formed the classification of the habitat types used in Chapters 4 and 5. Nevertheless, these two can be easily cross-referenced through the common attribute of grain-size. Habitat types were mapped from British Geological Survey data because more accurate habitat maps from survey data currently only cover ~10% of the UK continental shelf (Defra 2010).

The literature reviews undertaken for Chapters 3, 4 and 5 found there to be a limited number of studies of recovery of the benthic community following cessation of some human activities. The estimates of recovery times were not ground-truthed, which is a problem faced by many regional scale assessments (e.g. Halpern et al. 2009, Ban et al. 2010). These problems can complicate the quantification of recovery rates. Some published studies estimate recovery from low impact activities to be less than one year. However, the ability of the benthic community to recover from impacts is seasonal (Hall et al. 2008). Seasonality has less effect on recovery periods estimated to be of several years, but it can be important when recovery is estimated as ≤ 12 months. To account for this, a minimum of one year was assumed for all recovery periods (CLJ Frid pers. comm.).

Confidence rating was low for the activities which had inexact locations or where the precise spatial extent of impact was unknown (Chapter 6). For example, the location of benthic trawl vessels is usually recorded at 2 hourly intervals. This is generally adequate for a regional scale assessment (Lee et al. 2010), but at finer scales used for identifying co-occurring activities (e.g. m^2), the actual footprint of vessels' trawl-gear has to be estimated. Confidence was also low where generic spatial dimensions were used, e.g. for cuttings piles and scour pits associated with wind turbines and wrecks (*sensu* Eastwood et al. 2007). However, the spatially restricted nature of these latter pressures means there will be a small effect on the overall spatial estimates.

As described above, there was an assumption that recovery of the natural benthic community could not take place where obstructions were present. However, it is possible that structures such as surface-laid cables and pipelines have self-buried, but it was not possible to account for these differences in the analyses. Other structures might provide a hard substrate offering new habitat for benthic fauna and flora. For example, anemones (*Actiniaria*) were abundant on an exposed cable that traversed soft sediment which normally would be unsuitable for such animals (Kogan et al. 2004, Kogan et al. 2006). The common mussel (*Mytilus edulis*), common starfish (*Asterias rubens*) and a macroalgal community became abundant on turbines and scour protection, where previously they had been absent (Danish Energy Authority 2006). Such alterations can have implications for higher trophic levels; fish, marine mammals and seabirds, but establishment of species associated with hard substrate represents changes in population, not recovery of the ambient benthic community.

Some activities were not included in the assessment either because data were not available, or only available in an inappropriate format, such as dredging for navigational purposes, potting, fixed-net and inshore fishing. Total fishing effort is likely to be underestimated within 12 n miles of the shore, because the majority of fishing effort here is by vessels less than 15 m in length, which are not required to carry VMS (Foden et al. 2010). However, data that were used were prioritised so that those representing the greatest pressures and impacts were considered. Whilst the inclusion of extra data would technically offer more comprehensive coverage, their restricted nature means they would have a negligible influence on the final outcomes. Nevertheless, accurately quantifying the potential influence of more minor activities would be an interesting area for future study.

6.3 Recommendations for future work

As previously noted, accurate habitat maps currently only cover approximately 10% of the UK continental shelf (Defra 2010). Accurate habitat maps for the whole study area would be a significant improvement on the British Geological Survey data used herein. Linking the location of human activities to more accurate habitat maps would improve confidence by reducing the chance of over- or under-estimation of habitat sensitivity and recovery rates. Nevertheless, this level of detail may only be necessary for fine-scale assessments, for example in locations where there is a risk of multiple pressures coinciding, or at the boundaries between contrasting habitat types. In the absence of marine habitat maps generated from survey data, modelled marine habitats have been used as a successful alternative in a range of recent scientific studies (e.g. Ierodiaconou et al. 2010, Kloser et al. 2010, Robinson et al. 2010).

Recovery from some activities is site-specific, dependent on particular combinations of habitat characteristics, the local hydrodynamic regime and benthic community (Breuer et al. 2004), all of which complicate the prediction of potential long-term environmental impacts (Holdway 2002). Comparative *in situ* observations, for example of cuttings from oil and gas production or scour around wind turbines, would improve estimates of size of area affected and biological impacts from pressures, by habitat type.

Another issue worthy of future study is how best to reconcile past one-off pressures that still exist, with those which are ongoing. In Chapter 5 recovery of the benthos was not considered feasible in the presence of existing obstructions. On the other hand, as noted in that chapter, it is possible that some structures, e.g. wrecks or surface-laid pipelines, have either self-buried, or are themselves used as a structure for future colonisation. Therefore the impact of these structures is likely to be considerably lower than for repetitive pressures such as bottom-fish trawling. To take this into consideration would require recovery estimates to be tempered by the duration of time since initial impact. The concept proposed in this thesis of estimating impact under a range of scenarios, could be a model for estimating the possibility of ‘recovery’ from obstructions; ‘recovery’ being the time taken to self-bury or to colonise wrecks and pipelines. Recovery is likely to vary with habitat type and although some data are available in the published literature on self-burial and colonisation (e.g. Guerra-García & García-Gómez 2006, Kogan et al. 2006), ideally the scenarios would be ground-truthed.

6.4 Conclusions

This thesis was concerned with developing practical approaches to environmental assessment. Best practice for conducting marine environmental assessments was identified in Chapter 1. A relatively simple assessment of impact caused by a single sector was carried out in Chapter 2, based on this best practice; i.e. the scope, the methods and standards against which parameters were measured were all clearly set out. Chapters 3 and 4 were increasingly complex, in becoming 'dynamically linked ecosystem assessments' (Foden et al. 2008). The most complex type of assessment, 'dynamically linked and fully integrated', was not conducted because there is still a need to develop scientific understanding of the linkages across aspects of the marine environment not considered. Human activities cause pressure to other parts of the marine ecosystem, such as water column pollution, pelagic fisheries, shipping, underwater noise and siltation caused by plumes from disposal of dredge material or aggregate screening might be considered. To implement a full ecosystem-based approach to management would require a coherent and holistic assessment of all such interconnections, some of which may lead to cumulative impacts greater than those considered in this thesis (Foden et al. 2008). Nevertheless the focussed approach presented in this thesis accepts the limitations of the data outlined above, but it still derived significant and important outputs on which future work can be built. The approach has the advantages of clarity and reproducibility, and it delivers analyses which can guide management of those activities specifically affecting benthic ecosystems.

Some of the knowledge, data and assessment tool gaps identified in recent national status assessments (Defra 2010) have been addressed in this thesis, with regards to human pressure and impact on the benthic community. The spatial data used were readily available and after manipulation and preparation of the different datasets, analysis was possible in a GIS. The methods described in previous chapters for combining the data are relevant to marine spatial planning (MSP). The EU recognises and promotes MSP as an instrument for optimising the use of marine space to benefit the marine environment (EU 2008b). Marine spatial planning can aid delivery of the vision for UK's oceans and seas to be "clean, healthy, safe, productive and biologically diverse" (Defra 2002). It is hoped that the method for assessing the state of marine habitats will enable a more quantitative approach in future reporting against European and international commitments.

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