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Recovery rates of UK seabed habitats after cessation of aggregate extraction

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ABSTRACT: Marine aggregate extraction and benthic fishing are the 2 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 to 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 6% of estuarine areas and <1% of all other landscapes. Ninety-six percent of extraction occurred in sand or coarse sediment. Fifty percent targeted coarse sediment plains with moderate tidal stress, which had the longest period of T_{Phys} (20 yr) and the second longest T_{Bio} (8.7 yr). Shallow coarse sediments with weak tidal stress had the longest mean T_{Bio} (10.75 yr), but 21% of the habitat supported high intensity dredging. The most intense dredging (>90 h) was in estuaries, which have the shortest recovery times: T_{Phys} 1.67 yr and T_{Bio} 5.25 yr. At present, licensed areas do not appear to be located to avoid the most sensitive marine landscapes nor to target the least sensitive areas. 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

INTRODUCTION

As demands on marine resources increase, there is a growing need to 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. Such commitments include 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 (CEC 2000, CEC 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 as well as other complex forms of cumulative impacts (Smit & Spaling 1995, Cefas 2001, CCW 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. Approximately half the area of con-

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tinental shelf habitats are trawled annually (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 to 223 km^2 of seabed dredged annually, 1998 to 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, e.g. sublittoral muddy sands, are less sensitive to stress than those of more benign, less variable environments such as coarse sand and gravel in deep water (Bax & Williams 2001, Bolam & Rees 2003, Hiddink et al. 2007). In general, the more resilient areas 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 recolonise 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 a site disturbed by aggregate extraction and subsequently abandoned, 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. 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 (t) 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 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).

This paper describes temporal and spatial variation in aggregate extraction pressure and impact in UK marine waters since 2001. To quantify recovery periods, a review of published literature on physical and biological impacts and rehabilitation following cessation of dredging was undertaken. 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 gears.

MATERIALS AND METHODS

Literature review of marine aggregate extraction impacts. A review was conducted of scientific literature that measured recovery time following cessation of dredging. Studies were identified using computer database search engines of peer-reviewed literature, such as Scopus and ASFA, as well as 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 (i.e. case studies and government, government agency or industry published reports) were examined. Each reference was given 1 of 6 ‘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 indicates a randomised controlled study; II-1 is a controlled study without randomisation; II-2 is a comparison of differences between sites with and without controls; II-3 is multiple time series evidence; III is qualitative field evidence, descriptive studies or expert opinion; IV indicates inadequate evidence.

Physical recovery (T_{Phys}) from aggregate dredging 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 original 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), data on sediment grain size, depth, hydrodynamic regime and dredge intensity were quantified and standardised (see Table 1). 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 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: as near-bottom current shear-stress in Newtons m^{-2} (N m^{-2}), as depth-integrated current velocity, or as 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 3 categories: weak tidal stress = 0 to 1.8 N m^{-2}, moderate = 1.8 to 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 the 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 yr was chosen, which is supported by unpublished biological recovery data (Sánchez-Moyano in Guerra-García et al. 2003).

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 s. 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 × 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), 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 to 2007, inclusive.

Spatial and temporal pressure of UK marine aggegate extraction. The locations of aggregate dredging activity were spatially joined to UK marine landscapes characterised by a combination of sediment type,

Table 1. Marine landscape types in UK waters, targeted by the aggregates sector. Wave base is 50 to 70 m. Tide stresses: weak = 0 to 1.8 N m^{-2}, moderate = 1.8 to 4.0 N m^{-2}, and strong is >4.0 N m^{-2}. UKCS: UK continental shelf (from Connor et al. 2006). Slope is negligible (<2 %) for the shallow and shelf plain. Estuary has a strong salinity gradient from riverine inputs

Marine landscape type (depth) Substratum	Tide stress (currents)	Abbrev.	Area (km^2)	Prop. of total UKCS
Estuary (0 to 30 m)				
Mainly soft sediment, limited rock	Variable; moderate to strong in channels	ES	2881	0.3
Shallow plain (coastline to wave base)				
Coarse sediment	Weak	SCSW	33 694	3.9
	Moderate	SCSM	16 745	1.9
	Strong	SCSS	7869	0.9
Mixed sediment	Moderate	SMSM	2021	0.2
Sand / muddy sand	Variable	SS	48 218	5.5
Shelf plain (wave base to 200 m)				
Coarse sediment	Moderate	SHCM	17 433	2.0
	Strong	SHCS	2840	0.3
Mixed sediment	Moderate	SHMM	2260	0.3
Sand / muddy sand	Variable	SHSP	215 215	24.7

depth and tidal stress (Connor et al. 2006). The categories listed in Table 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, T_{Phys} and T_{Bio} rates 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 approximately 30 vessels operating in UK waters vary depending upon the pump size, power and age as well as vessel capacity (currently 880 to 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 5000 t in 3 hours (BMAPA 2006), which would equate to >2100 t at high intensity dredging, or <1700 t at low intensity. This would, however, 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 2). For each cell the dredge duration mid-points for all 7 years were summed to estimate cumulative dredging time.

Table 2. Annual dredge duration time ranges and calculated mid-points used to describe dredge intensity

Time range (decimal h)	Mid-point (h)
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

RESULTS

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 1 or 2 yr for comparable habitats (Cooper et al. 2008b).

The aggregate dredge sites reviewed are summarised in Table 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 3 orders of magnitude: 10^3 to 10^6 t km^{-2} yr^{-1}. T_{Phys} ranged from 0.5 yr at Kwinte Bank and Graal-Müritz to decades at Tromper Ost and Hastings. T_{Bio} varied from 0.75 yr in the Bristol Channel to decades in high intensity dredged sections of Thames Area 222.

Mean 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 (Fig. 1). Aggregate dredge site sizes varied from 0.3 to 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 from 0.1 m^2 to 0.1 km^2 in area, but not in larger sites (Guerra-García et al. 2003), i.e. not at the scale of sites in Table 3. All sites reviewed were in water depths <50 m and were in 6 landscape types; sand plain, estuary, or shallow coarse and mixed sediments in strong, moderate and weak tidal stress (see Table 1, first 6 landscape types for abbreviations). Different landscape-specific recovery rates at high and low levels of dredge intensity were not calculable as there were too few data at

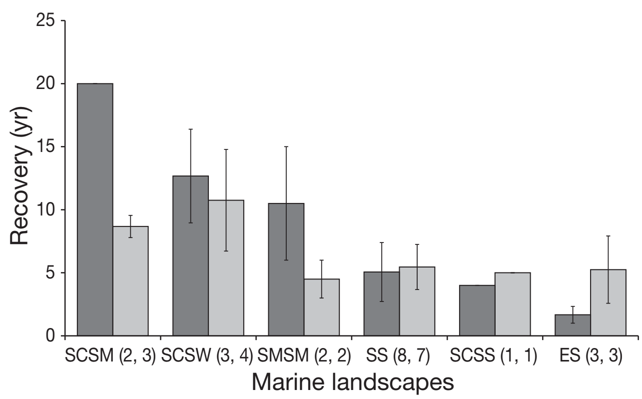

Fig. 1. Predicted recovery times of marine landscapes from dredging based on the literature review. Physical recovery (T_{Phys}): dark grey; biological recovery (T_{Bio}): light grey. N for each bar shown in brackets (T_{Phys}, and T_{Bio}) with standard error (σ_M) bars. Marine landscape codes as given in Table 1

Table 3. Summary of (a) physical recovery (T_{Phys}) and (b) biological recovery (T_{Bio}) rates for different habitat types following cessation of dredging. 'Hydrodynamics', unless otherwise described, are classified as follows: weak = 0 to 1.8 N m^{-2}, moderate = 1.8 to 4.0 N m^{-2}, and strong is >4.0 N m^{-2}. *: estimated value, na: data not available, LW = low water. Quality of evidence categories based on Pullin & Knight (2003); Categories I to 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} (yr)	Examples	Source (& 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	2850	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-Müritz, Baltic Sea	Diesing et al. (2006) (II-2)
Medium sand	Strong	4 to top of bank	SS	23 000	151.8	0.5[a]	Kwinte Bank Zone 2 (2001 max. intensity yr)	Vanaverbeke et al. (2000) (II-2), Degrendele et al. (in press) (II-3)
Sand	Strong and moderate–strong	<20	SS & ES	n/a	na	≤1	Bristol Channel, UK	Newell et al. (1998) (III)
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	91500	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	na	na	>4	Suffolk coast, UK	Millner et al. (1977) (I)
	Moderate–weak	28–34	SMSM	80 000	6.1	Decades[b]	Southwold, Area 430, UK	Andrews Survey (2004) (I), Newell et al. (2004b) (I). MES (2007) (I)

Table 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} (yr)	Examples	Source (& quality of evidence category)
Fine sand	Strong tidal current estuaries	<20 Shallow subtidal <10	SS & ES ES	na 617 500 mean rate	na ~1*	0.5–0.75 >1–>3	Bristol Channel, UK Waddenzee, subtidal channels, NL. Fastest recovery in fastest current flows	Newell et al. (1998) (III) van der Veer et al. (1985) (II-2), Newell et al. (1998), Table 6 (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. (1998) (II-2), Table 6
	Non-tidal, energetic wave dominated	10–12	SS	552 900	1.1	>1	Wustrow, Baltic Sea	J. C. Krause et al. (unpubl.) (I)
	Non-tidal, weak wave environment, with infrequent seasonal storms	16–20	SS	3230	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)
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), (2005) (I) K. M. 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), K. M. 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), K. M. Cooper 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), K. M. Cooper pers. comm. (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	na	na	>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 20–30	SMSM SMSM	150 000 na	1.0 na	3 >4	East of Isle of Wight, Area 122, UK Suffolk coast, UK	Newell et al. (2004a) (I), Boyd & Rees (2003) (II-1), Millner et al. (1977) (II-1)

[a]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); [b]Changed sediment type has slowed recovery (Newell et al. 2004)

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 yr in estuaries. Biological recovery was most rapid (4.5 to 5.5 yr) 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.

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 × 50 m cells) and 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 (K. O'Shea pers. comm.) and are not discussed further herein. There were 63 main dredge sites in licensed areas ranging from 0.05 to 21.36 km^2 in area, 86% of which were >1 km^2.

Spatial and temporal pressure of UK marine aggregate extraction. When spatially joined to the UKSeaMap marine landscapes (Table 1), 22% of the EMS records were within 4 of the 'shelf' categories (>50 m deep), adjoining the boundaries of the 'shallow' categories. Typically, dredgers work in depths from 10 to 40 m (BMAPA 2006). Twenty-seven of the 29 dredgers in UK waters have a maximum operating depth of <45 m and 2 dredges up to 50 m depth. Consequently, we considered the categorisation of some dredge sites as shelf sites as being an artefact of the 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 (Fig. 2).

Six main landscape types are targeted in the UK by the aggregate extraction sector. Sand plains constitute the largest landscape (Fig. 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 areas (Fig. 2b). This landscape has an estimated mean physical recovery rate from aggregate extraction of 20 yr, the longest T_{Phys} of any landscape type (Fig. 1). It also has the second longest T_{Bio} of 8.7 yr. The shallow coarse sediment plain-weak tide stress landscape has the longest mean T_{Bio}, 10.75 yr, and a significant proportion of its dredge area (30%) was exploited at moderate or high intensity (Fig. 2b). Estuaries have the shortest T_{Phys} and T_{Bio} recovery periods (1.7 and 5.25 yr, respectively) and aggregate dredging activity is undertaken in 6 percent of this landscape (Fig. 2a). This is an underestimation of total dredging activity as maintenance dredging also occurs in many UK estuaries. However, the 2 processes are different; maintenance dredging removes recent unconsolidated sediments, whereas aggregate extraction removes minerals to a maximum permitted depth (DCLG 2007).

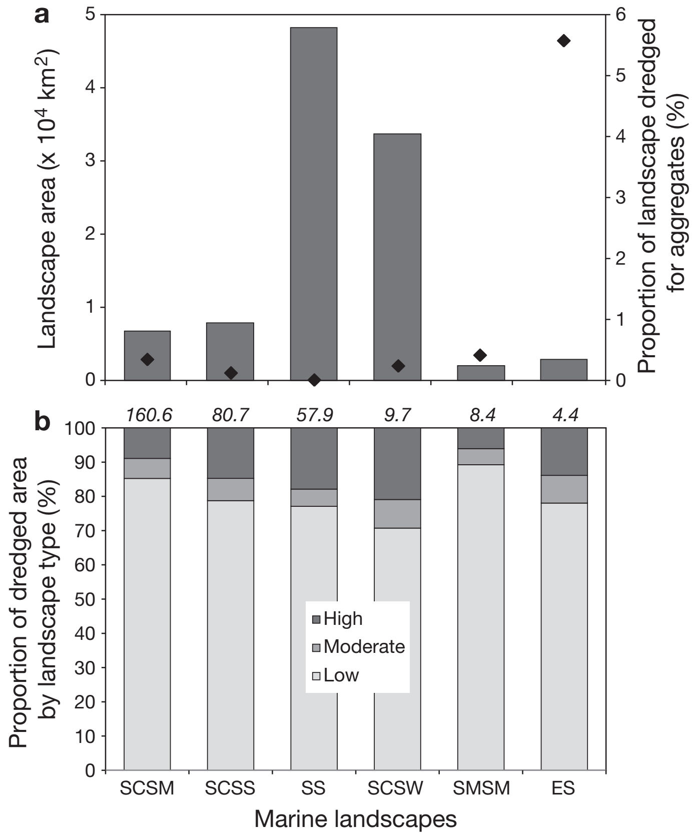

Fig. 2. UK marine landscapes targeted for aggregate minerals (landscape codes as in Table 1). (a) Extent of marine landscapes in UK (bars) and proportion dredged for aggregate (points). (b) Proportions of dredge intensity in marine landscapes: High: >1.25 h yr^{-1}; Moderate: 1 to 1.25 h yr^{-1}; Low: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

The majority of dredging effort in the UK was at low intensity (Table 4). At least 89% of the total area dredged in any one year was worked at low intensity effort (less than 1 h). Over the entire 7 yr period, 81.7% (263 km^2) of the entire dredging footprint was always worked at <1 h yr^{-1} and cumulative dredging in this area was less than 5 h in all locations.

Between 2001 and 2007 only 4.9 to 6.7% of the total area was dredged at high intensity of >1.25 h in any one year. The area continually exploited at this intensity every year was 0.03 km^2 (0.009% of the footprint).

Table 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 7-yr 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 h in one or more yr)	High intensity (>1.25 h in one or more yr)
<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

Very high intensity dredging of >40 h total 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 h of dredging occurred in 2001 to 2007.

Coarse sediment and sand plain landscapes were heavily targeted by the aggregate industry, together constituting 96% of the UK extraction area (Fig. 2b), though the area that was dredged in each landscape was < 0.5% of the total (Fig. 2a). The industry dredges in all strengths of tidal stress. The relative proportions of dredge intensity within landscape types shows that the majority of dredging activity (>70%) was at low intensity (Fig. 2b). Moderate or high intensity dredging formed a larger proportion (29%) of dredging effort in shallow coarse sediment-weak tide stress areas. Moderate and high intensity dredging affected the smallest proportion (<11%) of shallow mixed sediment plain-moderate tide stress landscapes.

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.

Where sediment characteristics, topography and the natural hydrodynamic regime do not differ before and after dredging, reestablishment 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: 24 to 30 mo in the North Adriatic, for example (Simonini et al. 2007). As expected, the least sensitive habitats, i.e. those 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 6% of their combined area is dredged for aggregate, compared with <1% in the other landscape types. If topography and sediment composition are permanently altered and previously stable sediments are not reestablished, communities remain at an early developmental stage (Boyd et al. 2004) and biological recovery can take more than 10 yr (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 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 yr, respectively). New statutory regulations for aggregate licensing came into force on 1 May 2007: Environmental Impact Assessment and Natural Habitats (Extraction of Marine Minerals by Marine Dredging) (England and Northern Ireland) (DCLG 2007) Regulations. These replace the previous voluntary, informal, non-statutory Government View and will require the 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 gears, 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 to 12 m width and 1 to 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 shown similar recovery rates to those for towed bottom-fishing gears in similar environments (Løkkeborg 2005, Smith et al. 2006). The impact of fishing gears varies significantly among habitats (Kaiser et al. 2006, L. Robinson et al. unpubl.) have 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 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 (such as 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 live 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 also 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 d from dredging that penetrates only 7 to 13 cm into sand and silt sediment, but if the penetration is 20 cm, recovery does not start until after 60 d (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, 2005, Cooper et al. 2005).

The majority of dredging activity in the UK from 2001 to 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 h), repetitive dredging can require up to 10 yr for recovery after cessation (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 (i.e. <100 m^2) than at large sites (i.e. >1000 m^2) because larger patches possess a smaller edge to 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 6 mo, and more probably would be in the order of years. This corresponds with the temporal range found in the literature review (Table 3).

A limitation to this study was the inability to consider the effect of aggregate overspill and screening. These techniques may significantly alter the substratum, extending the period for recovery (de Groot 1996), or the 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. There are too few studies of these factors, however (Eastwood et al. 2007), and they were not incorporated into this study.

In the present 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 the effects of different intensities of dredging by landscape type is needed. The general pattern of response to aggregate extraction needs to be further tested 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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