ICES Journal of Marine Science, 57: 1428–1438. 2000
doi:10.1006/jmsc.2000.0926, available online at http://www.idealibrary.com on IDEAL®

Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short- and long-term post-dredging restoration

Michel Desprez

Desprez, M. 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short- and long-term post-dredging restoration. – ICES Journal of Marine Science, 57: 1428–1438.

Sediment and associated macrofauna of an industrial extraction site off Dieppe have been monitored during a 10-year period. The original heterogeneous substrate of the shingle bank, characterized by gravels and coarse sands, was progressively dominated by fine sands deposited in dredging tracks. The maximum impact on benthic macrofauna was a reduction by 80% for species richness and 90% for both abundance and biomass. The structure of the community changed from one of coarse sands with *Branchiostoma lanceolatum* to one of fine sands with *Ophelia borealis*, *Nephtys cirrosa*, and *Spiophanes bombyx*, with local dominance of the opportunistic, sessile *Pomatoceros triqueter* on bare shingles. Impact of overflowing sands on benthic macrofauna in the surrounding deposition area proved equally large as in the dredged area. Early stages of recolonization were studied from 1995 to 1997 after cessation of dredging. Species richness has been fully restored after 16 months, while densities and biomass were still 40% and 25%, respectively, lower than in reference stations after 28 months. Nevertheless, community structure differed from the initial one corresponding to the new type of sediment. Impact within and around the dredging site was classified according to three levels. Exploration of a former experimental site (CNEXO) dredged in the 1970s provided an example of long-term restoration.

Key words: dredging, English Channel, France, marine aggregates, physical and biological impact, recolonization.

Michel Desprez: GEMEL-Picardie, Station d'Etudes en Baie de Somme, 115 quai Jeanne d'Arc, 80230 Saint-Valery-sur-Somme, France.

Introduction

Because of increasing environmental pressure, the number of exploitation sites and the volume of reserves of land-based aggregates are decreasing rapidly in France. Consequently, coastal areas should provide more and more constructional material in the next few years. As conflicts of interest are high between the dredging and fishing industries, the French government is granting production licences slowly (ICES, 2000). Therefore, marine aggregate extraction is not yet highly developed. Annual production stabilized at around 3.6 million tonnes, much lower than in countries such as Great Britain or the Netherlands, where about 20 million tonnes (15% of total demand) are dredged annually either for the construction industry or for coastal defence and major reclamation schemes (Arthurton *et al.*, 1997).

Most marine sediments extracted is sand from the Atlantic coast. Dieppe represents an exception by being a major extractor of gravels and shingles (60%). There is no monitoring of impact on the Atlantic sites, and the effects on the marine environment are largely unknown. However, the dredging company in Dieppe invested in a monitoring programme in 1979, which has provided fundamental data on the spatial and temporal effects of industrial coarse aggregate extraction (Desprez, 1997).

Although a substantial literature exists on the environmental effects of dredging (Arntz and Rumohr, 1982; Bonsdorff, 1983; De Groot, 1979a; Hily, 1983; Norden Andersen *et al.*, 1992; Van Dalfsen and Essink, 1998), most studies describe fine-sediment community responses that are not directly applicable to impact on gravel communities. The lack of research on the biological effects of marine gravel extraction is understandable considering the small number of countries possessing exploitable gravel resources. The United Kingdom is one of the world's largest producers and consequently has been responsible for much of the environmental

1054–3139/00/051428+11 $30.00/0

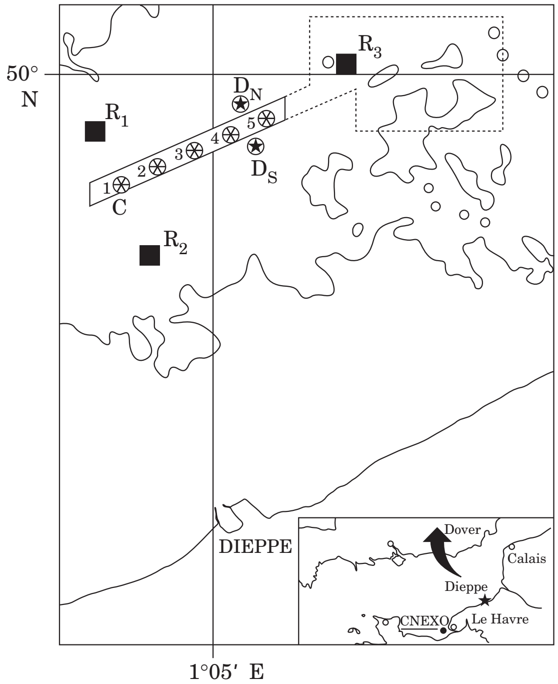

Figure 1. Location of the former (drawn box) and present (hatched box) dredging site off Dieppe with the monitoring stations (dredging control stations C1–C5; deposition control stations Dn–Ds; reference stations R1–R3).

research. During the 1970s, the impacts of suction-anchor dredging on Hastings shingle bank in the English Channel have been examined (Shelton and Rolfe, 1972), while Millner *et al.* (1977) examined the impacts of suction-trailer dredging off Southwold in the southern North Sea. More recently, Lees *et al.* (1990) reported on the impacts of suction-trailer dredging at a licensed extraction area off the Isle of Wight. However, due to difficulties of sampling coarse sediments, these studies were unable to quantify accurately the initial impacts on the benthos. A notable exception was a controlled dredging study off north Norfolk, initiated in 1992 to investigate the process of recolonization upon cessation of dredging (Kenny and Rees, 1996). Effects of aggregate extraction were also quantitatively examined on the Cleaver Bank in southern North Sea (Van Moorsel, 1994). Both studies used quantitative sampling techniques before and after dredging to describe the physical and biological impacts.

Methods

The industrial dredging site of Dieppe (50°00′N 1°05′E) is an area of 1.5 km^2 located 3 miles offshore, along the French coast of the eastern English Channel (Fig. 1), at a mean depth of -15 m below chart datum. Dredging activity began in 1980 to provide sand and gravel for the construction of a nuclear power station. The amounts extracted were maximum from 1980 to 1985 (0.4–0.8 million tonnes per year) and then decreased to an average annual of 0.1 million tonnes until 1994. Extraction was stopped in late 1994, allowing for the study of the restoration process.

After an initial survey in 1979, the biological and sedimentological monitoring of 12 stations, realized between 1980 and 1993, showed the absence of impact 1 km off the dredging site and allowed a detailed map of the surface sediments and associated macrofauna to be obtained. Three types, referred to as communities hereafter, were identified (Fig. 2): stones with sessile epifauna in the eastern sector; gravels with the lancelet *Branchiostoma lanceolatum* and sessile epifauna in the southern sector including the dredging site; gravely sands with *Branchiostoma* in the northern sector. A fourth community (fine sands with *Ophelia*) was found in the extraction site as a consequence of dredging. This community is naturally present only further East.

Ten stations were sampled within and around the dredging site: three reference stations (R1–R3), located about 1 km away from the site, providing information on natural fluctuations of benthic communities; five control stations (C1–C5), located within the extraction area and providing information on the impact and on recolonization rates; in 1996 and 1997, two other stations were sampled 200 m North (Dn) and south (Ds) of the eastern most impacted part of the site to quantify the effect of deposition by overflow.

From 1980 to 1991, sampling was done with a Rallier dredge providing qualitative data only, while from 1993 onwards quantitative data were collected using a Van Veen grab (0.1 m^2). Three replicates were sieved on board over a 1 mm mesh size and the retained macrofauna fixed in 4% formaldehyde solution. Identification was performed at a species level for each infaunal group. Samples were dried at 60°C during 48 h and burnt at 450°C for 2 h to get ash-free dry weight biomass. Per station, two subsamples of sediment were analysed for particle size distribution and for classification in sediment types.

Physical images of the seabed at the extraction site were obtained post-dredging using a sidescan sonar.

Results

Physical impact

Several years of more or less intensive dredging seriously affected the seabed surface, which showed a disturbed topography with large furrows (up to 5 m deep) separated by crests of shingles (Fig. 3). These furrows and depressions were more or less filled in with sand coming either from overflow (before late 1994) or from the large natural transport of sediments resulting from

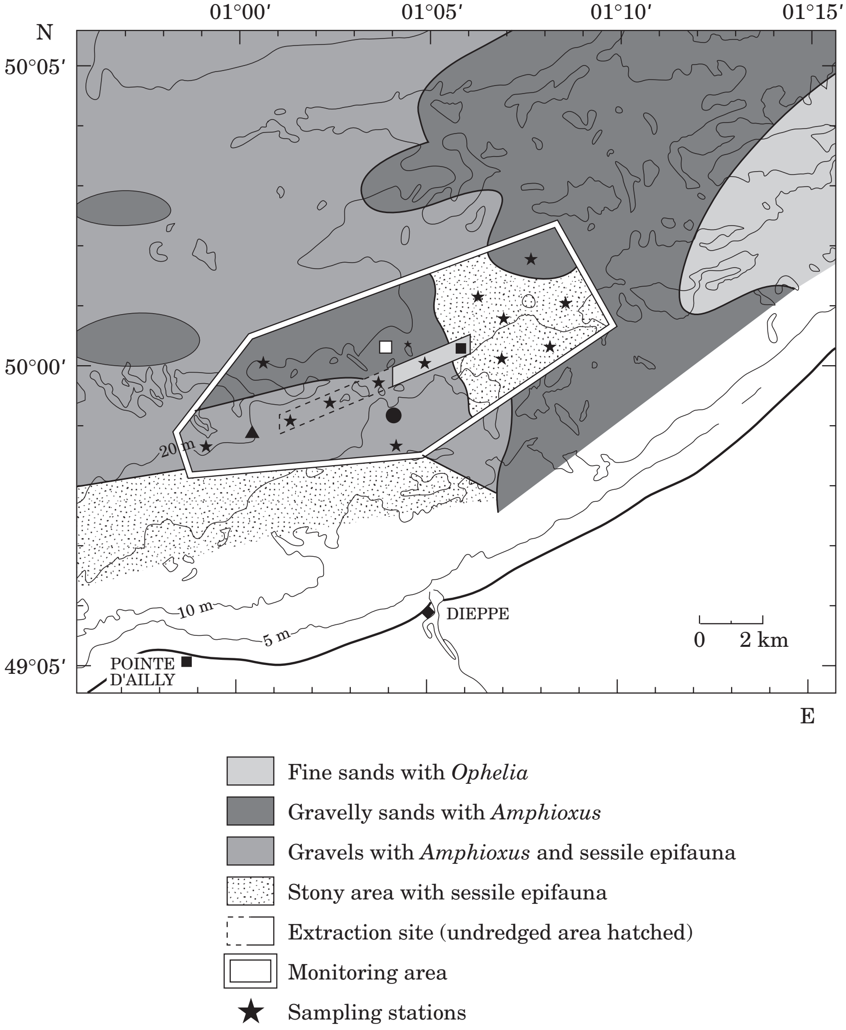

Figure 2. Map of the bio-sedimentary assemblages off Dieppe (after Cabioch and Glaçon, 1977) with modifications for the monitoring area.

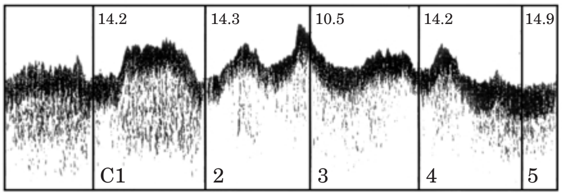

Figure 3. Longitudinal bathymetric transect of the dredging site off Dieppe with location and depth (metres above chart datum) of the control stations (soundings from "Arco Thames", July 1995).

the strong tidal currents characterizing the area (1 m s^{-1} during spring tides). Sidescan sonar revealed the presence of sand megaripples in big depressions between areas of gravels and shingles where dredge tracks were clearly visible.

Sediment granulometry (Table 1) confirmed the heterogeneity of the seabed topography. Sediments of the reference area were homogeneous, with neither very fine sands nor silts and a predominance of gravels and coarse sands. In the dredging area, sediments were heterogeneous and co-dominated by fine sands and shingles, with a notable proportion of very fine sands. Extraction progressively eliminated the original sandy gravel, exposing the underlying shingles on the edge of the drag-head tracks (up to 50%), while fine sand derived from mobile sand-ripples and from overflow had filled the furrows (up to 70% in the most intensively dredged eastern sector). In the deposition area, sediments were also homogeneous and largely dominated by fine sands. The outer limit of this area, sampled in 1997, was characterized by slightly muddy gravels.

Table 1. Comparison of sediment granulometry and main population parameters for the three sampling areas of the dredging site off Dieppe (in 1996).

	Deposition area	Dredging area	Reference
Shingles and gravels	11	26	47
Coarse sands	12	8	34
Fine sands	63	54	18
Very fine sands	13	19	1
Silts	1	1	0
Biomass (g m^{-2})	0.3	2.4	6.8
Density (ind. m^{-2})	230	810	1440
Species richness	17	44	39

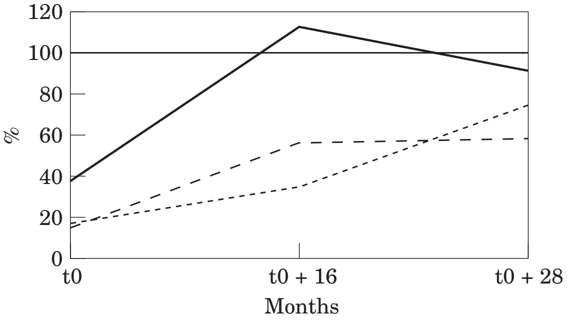

Figure 4. Evolution of biomass (stippled line), abundance (hatched line), and species richness (drawn line) in percentage of the values for the reference stations after cessation of extractions (t0) at the dredging site off Dieppe.

Biological impact

Biological monitoring in the 1980s indicated that the amplitude of natural fluctuations of densities at the control stations was caused by the intensity of local hydrodynamic processes, leading to a strong instability of surface sediments (presence of shifting sand ribbons moving eastwards). The intense hydrodynamics explain why the effects of dredging were almost absent in the westernmost control station (C1), while a gradient of increasing impact was always observed from West to East (from C2 to C5) within the extraction site. The maximum effect registered in the eastern stations (C4 and C5) was a decrease by 80% for species richness and 90% for densities in 1993. Because of the presence of the sea urchin (*Echinocardium cordatum*), biomass was only 83% lower than in reference stations.

Meanwhile, the benthic community had changed from one of coarse sands with *Branchiostoma* to one of fine sands dominated by polychaetes (*Ophelia borealis*, *Nephtys cirrosa*, and *Spiophanes bombyx*), with *Echinocardium* as complementary characteristic species. The structure of the community had thus fundamentally changed after several years of intensive extraction, with decreased densities of crustaceans, echinoderms and bivalves and an almost complete dominance by errant annelids (in abundance) and by echinoderms (in biomass).

For the whole dredging area (four control stations), the biological effect of extraction could be summarized (1993 survey) as decreases by 63% in species richness, 86% in abundance and 83% in biomass. Biomass was limited by the presence of echinoderms of the eastern station and of decapods (*Galathea*), molluscs (*Spisula*, *Natica*) and echinoderms (*Ophiura*) at the western station.

The effect of sands depositing from overflow around the extraction site was studied in 1996 and the results showed that the deposition area was biologically more disturbed than the dredged one (Table 1). The mean number of species was 60% lower than in the reference area, whereas the dredging area had fully recovered; the mean density was 86% lower, against 53% in the dredged area; and the mean biomass was only 3%, compared with 28% in the dredged area. The community was dominated by two characteristic species of fine sands, the bivalve *Tellina pygmaea* (29%) and the annelid *Nephtys cirrosa* (22%), while other sand-dwelling species, such as the annelids *Scoloplos armiger* (3%) and *Spiophanes bombyx*, were also observed. Dominant species of coarse sands, *Echinocyamus pusillus* and *Amphipholis squamata*, were poorly represented (1%) and the characteristic species of gravels and shingles were absent. These results show that the indirect impact of sands depositing in the vicinity of the extraction site on macrobenthic fauna can be as significant as the direct effects.

Recolonization

Data collected in 1996 and 1997, respectively 16 and 28 months after cessation of dredging, showed that species richness had been fully restored by as early as 1996, abundances had recovered to up to 56% of the reference value in 1996 and stabilized in 1997 (Fig. 4). Biomass recovered less rapidly (35%) but increased faster during the second year (up to 75% of the reference value in 1997). Thus, total recovery was almost achieved 28 months after cessation of dredging activity, except for densities.

Species composition of the new community was closely linked to sediment type. In the western part, where intrusion of mobile coarse sands occurred, the community was dominated by several species characteristic of the reference area, such as the echinoderm *E. pusillus* and the annelids *Polycirrus medusa*, *Notomastus latericeus*, *Syllis* sp., but also by the opportunistic annelid *Pomatoceros triqueter*. In the eastern part, characterized by shingles and fine sands, the community was dominated by some opportunistic species belonging to the sessile (*P. triqueter*, hydrozoans) and mobile (crustaceans *Pisidia longicornis* and *Galathea intermedia*) epifauna, accompanied by some sand-dwelling species like the amphipods *Urothoe elegans* and *Cheirocratus sundevallii*, and the annelids *S. bombyx* and *N. cirrosa*. Finally, the community in the deposition area, largely dominated by clean fine sands, was characterized by the annelids *S. bombyx*, *N. cirrosa*, *Ophelia bicornis*, the bivalve *T. pygmaea* and the amphipod *Urothoe brevicornis*.

Four levels of impact could be distinguished in 1997 (Table 2) from west to east: 0, reference site (dominance of coarse sand-dwelling species); 1, recolonization nearly achieved in the western sector (dominance of the opportunistic annelid *P. triqueter*) and in the northern deposition area (co-dominance of coarse/fine sand-dwelling species); 2, recolonization in progress in the median sector of the dredging site (lower abundances and biomass; dominance of opportunistic epifauna on bare shingles; increasing number of coarse sand species and decreasing number of fine sand species); 3, maximum impact in the eastern sector (lowest species richness, abundance and biomass; dominance of sand-dwelling species in the fine sediments linked either to dredging or to overflow).

The former CNEXO experimental dredging site in the Seine estuary (Fig. 1) was surveyed again in 1995, 15 years after the cessation of dredging activity. Sampling of superficial sediments and benthic macrofauna at 10 stations located inside and outside this site provided information on the long-term physical and biological restoration (Table 3). Natural infill of the dredging site was limited and depended on bottom morphology. In the deepest and narrowest part of the site (dredged to 12 m below original bottom level), strong tidal currents had prevented any restoration, whereas in the larger and shallower part (maximum deepening of 5 m) fine sediments were deposited that were five times more silty than in the reference area. The community was two to three times richer in number of species, abundance and biomass. Mud-dwelling species were dominant on the dredging site (68%) whereas the reference area was dominated by sand-dwelling species (65%). The dominant groups within the dredging site were amphipods (density) and sea-urchins (biomass), while polychaetes (density) and holothurians (biomass) predominated in the reference area. The community of the dredging site was linked to the muddy fine sand communiuty (with *Abra alba* and *Pectinaria koreni*), whereas the community in the reference area was intermediate between the medium clean sand and the muddy fine sand community. Some 20 years ago, the site was located in medium sands with *Ophelia* (Gentil, 1976). Since dredging stopped, the evolution of the substratum was more rapid within the site than outside.

Although the exploration in 1995 yielded valuable information on the potential for restoration of a marine aggregate extraction site in relation to bottom topography, it is impossible to deduce how many years had been required to reach this level of restoration.

Discussion

Physical impacts

The most serious physical impacts of marine aggregate extraction relate to substratum removal, alteration of the bottom topography and re-deposition of material (De Groot, 1996; Newell *et al*., 1998). The impact is site specific and depends on numerous factors, including extraction method, sediment type and mobility, bottom topography, and current strength.

Water depth may change rapidly. Kenny and Rees (1996) estimated that over a five-day period depth had increased by approximately 2 m in areas where the draghead had followed the same path several times. Infilling of the pits and furrows depends upon the type of substratum and the ability of local currents to transport the surrounding sediment. Except in areas of mobile sands, the process tends to be very slow (Eden, 1975; Millner *et al*., 1977; Desprez and Duhamel, 1993; Kenny and Rees, 1996; Newell *et al*., 1998): furrows that are only 30 cm deep when formed are still clearly visible on sidescan sonar records several years later, and even the strongest currents are unable to transport gravel from adjacent areas. Therefore, infilling occurs largely through fine sediment fallout, either from suspension or from transport over the sea floor. Off Norfolk, the uniform and stable sea floor was transformed after five days of dredging into a bottom traversed by numerous furrows (Kenny and Rees, 1996). Where sediment transport was larger (presence of sand-ripples), this led to a significant erosion of these furrows, but they were still visible after three years.

Long-term dredging of a stable gravelly area will result in the sea floor being cratered, with removal of a significant thickness of sediment (Winterhalter, 1990; De Groot, 1996). The uneven sea floor topography can cause snagging of bottom trawls in tracks or around boulders exposed by removal of the surrounding substrate (Cruickshank and Hess, 1975). A further implication of the formation of these depressions is a local drop in current strength associated with the increased water depth, resulting in deposition of finer sediments than those of the surrounding substrate (Kaplan *et al*., 1975; Hily, 1983; Van der Veer *et al*., 1985; Desprez and Duhamel, 1993). The former CNEXO experimental dredging site provides an example, but also shows the opposite effects in a deep channel where an acceleration of tidal bottom currents prevented any sediment deposition. To minimize modifications of bottom

Table 2. Classification of the sampling stations according to an increasing impact gradient and biological synthesis in 1997, two years after cessation of dredging.

Impact level	0	1	1	2	3
Area	Reference	Dredged	Deposition	Dredged	Dredged/Deposition
Stations	R1/R2/R3	C1/C2	Dn	C3/C4	C5/Ds
Species richness	50	53	66	48	27
Abundance (ind. m^{-2})	1940	1360	1090	895	625
Biomass (g m^{-2})	8	9	5	6	2
Dominant species	*Polycirrus medusa*[1]	*P. triqueter*[2]	*P. medusa*[1]	*Pisidia longicornis*[2]	*S. bombyx*[3]
	Syllis amica[1]	*P. medusa*[1]	*N. latericeus*[1]	*P. triqueter*[2]	*Phyllodoce groenlandica*[3]
	Echinocyamus pusillus[1]	*S. hyalina*[1]	*S. bombyx*[3]	*Urothoe elegans*[3]	*Nephtys sp*[3]
	Harmothoe ljungmani[1]	*N. latericeus*[1]	*Chaetozone setosa*[3]	*P. medusa*[1]	*P. triqueter*[2]
	Syllis hyalina[1]	*S. kefersteini*[1]	*H. alatus*[1]	*Actiniaria*[2]	*Bathyporeia guillamsoniana*[3]
	Notomastus latericeus[1]	*Heterocirrus alatus*[1]	*L. impatiens*[1]	*L. impatiens*[1]	*P. medusa*[1]
	Staurocephalus kefersteini[1]	*Spiophanes bombyx*[3]	*Pholoe minuta*[1]	*Cheirocratus sundevalli*[3]	*L. impatiens*[1]
	Glycera sp[1]	*H. ljungmani*[1]	*Nephtys* sp[3]	*Eulalia sanguinea*[1]	*Urothoe brevicornis*[3]
	Pomatoceros triqueter[2]	*L. impatiens*[1]	*Nebalia bipes*[3]	*Galathea intermedia*[2]	*Glycera* sp[1]
	Eulalia tripunctata[1]	*E. pusillus*[1]	*Glycera* sp[1]	*Amphipholis squamata*[1]	*Ophelia borealis*[3]
Relative abundance (%) of species characteristic of:					
[1]Coarse sands	17	10	10	5	4
[2]Shingles	3	30	3	35	4
[3]Fine sands	1	3	6	2	35

Table 3. Summary of physical and biological characteristics of the inner (control) and outer (reference) areas of the CNEXO experimental dredging site.

	Control	Reference
Depth (m)	21.7	17.2
Sediment composition (%)		
Gravels	9.4	1.4
Fine sands	54.1	68.9
Silts	2.9	0.6
Benthic macrofauna		
Species richness (mean)	30	12
Number of mud species	19	8
(Relative abundance; %)	(68)	(24)
Number of sand species	16	11
(Relative abundance; %)	(20)	(65)
Mean density (ind. m^{-2}	1510	538
Dominant group	Amphipods (28%)	Polychaeta (48%)
Dominant species	*Urothoe elegans* (13%)	*Nephtys hombergii* (31%)
	Cheirocratus sundevalli (5%)	*Lumbrineris impatiens* (15%)
Mean biomass (g m^{-2} AFDW)	17.5	8.7
Dominant groups	Echinoderms (50%)	Echinoderms (61%)
	Cnidaria (21%)	Polychaeta (27%)
	Bivalves (14%)	Cnidaria (2%)

topography, sediments and currents that may have a drastic impact on benthic fauna, deep extractions should be avoided. In contrast, shallow depressions may be recolonized by richer communities than the original ones.

A recent study indicated that the bulk of a dredge plume (approximately 80% by weight of the total discharged sediment is composed of sand-sized particles) collapses to the seabed within a few hundred metres of the dredger (Newell *et al.*, 1998). Hitchcock and Drucker (1996) demonstrated that suspended sediment (>0.063 mm) decayed to background levels over a distance of 200 m to 500 m from the point of release, which is in accordance with our observations. Consequently, the gravelly substratum is gradually evolving to a sandy one with accumulation of fine sand and mud in the furrows of the dredging area and deposition of fine and in the close vicinity of the site.

New sediment at the extraction site may accumulate by one or more of the following processes (ICES, 1992): (1) through bedload transport of mobile sand; (2) by natural deposition of fines from the water column; (3) through slumping of the pit walls; and (4) by deposition of outwash fines and sand from the dredger. Of these, infilling by transport of mobile sand is the most rapid and dominant process if the dredge site is located in an area of active sand transport, like off Dieppe. Complete and rapid regeneration of a dredged deposit to its former state (topography and sediment type) may then occur. By contrast, pits outside of such areas are generally filled very slowly (Eden, 1975), as observed at the CNEXO site where infilling was mainly the consequence of slumping of the pit walls.

The current regime is frequently altered after dredging, with consequences for the sedimentation process. In the case of gravel deposits on the northeast North American and northwest European continental shelves, deposited by fluvial and fluvio-glacial processes during the Quaternary, the tidal currents are too weak to transport the pebbles and only silt or sand may accumulate in its place (Shelton and Rolfe, 1972; Millner *et al.*, 1977; Desprez and Duhamel, 1993). In areas of active sedimentation, dredging may sufficiently alter sea-floor topography or increase water depth to slow down currents over the tracks or pits, thereby causing deposition of finer sediment (Kaplan *et al.*, 1975; Hily, 1983; Van der Veer *et al.*, 1985). Our data support these conclusions.

Biological impacts

Animals constituting benthos are living upon or inside the sediment. They are mainly invertebrates such as shellfish, worms, crustaceans and sea urchins. Several species of fish are also more or less closely linked to the bottom where they may live, feed or spawn (Westerberg *et al.*, 1996). Links between the sea floor and the water column are evident when considering the planctonic larval stages of benthic invertebrates. According to the importance of these links between species and sediment, aggregate extraction will have direct (suction or infaunal or epifaunal species) or indirect (disturbance, loss of food) effects.

Recent results from experimental dredging sites in the North Sea show that benthic macrofauna may be rapidly and strongly reduced by extensive trailing dredging. On the Cleaver Bank (van Moorsel, 1994), an extraction of 20% of the seabed during two months created furrows of limited depth without any modification of the sediment. The number of benthic species decreased slightly (30%) whereas abundance and biomass of the community were markedly reduced (72% and 80%, respectively). In the experimental study initiated off Norfolk (Kenny and Rees, 1996), dredging of about 70% of the site during a five-day period created a pattern of furrows with sand ripples along the base of the dredge tracks. Biological observations showed a significant reduction in the average number of species (66%) and large decreases in densities (95%) and biomass (99%). These two examples show that extensive dredging may only modify the sediment slightly but alter benthic communities seriously; the results obtained at the industrial site off Dieppe support this conclusion. This suggests that it would be preferable to concentrate dredging in small areas. However, prolonged extraction in limited sites affects seabed morphology and sediment quality more seriously, and consequently, the type and recolonization rate of benthic communities whose trophic value needs to be ascertained.

The magnitude of the effect of sediment redeposition upon a benthic ecosystem not directly affected by dredging itself and depends largely on the nature of the indigenous fauna, the deposition rate and the relative increase in water turbidity. Of the different forms of impact generally recorded (ICES, 1992), a virtually complete defaunation was observed in Dieppe, and recolonization did not progress rapidly.

To select the least harmful way of dredging for the marine environment, one criterion may lie in the recovery potential of the benthic community. Alterations are only temporary and the readjustment of the benthic community after cessation is linked to parameters such as the nature and stability of the new sediment exposed or accumulated at the extraction site, the larval and adult pool of potential colonizers, and the nature and intensity of stress usually endured by the community (ICES, 1992; Newell *et al.*, 1998).

The character of the sediment exposed or which subsequently accumulates after dredging controls the structure and composition of the new benthic community. In soft sediment, colonization by a range of infaunal species (especially polychaetes and bivalves) will occur within weeks or months depending on season, largely through larval recruitment. Recolonization by populations of motile epifaunal browsers and predators, especially decapod crustaceans and echinoderms, will depend on the availability of suitable food, but may occur opportunistically through migration of adults into the area or *via* larval recruitment (Rees, 1987).

Sediment filling the dredge tracks is generally finer than the original one and the fauna colonizing the new substrate may differ from the biota present in adjacent undredged coarser substrates (Shelton and Rolfe, 1972; Millner *et al.*, 1977). The difference between the two is proportional to the refinement of sediment, which is itself linked to the intensity of dredging. In the dredging experiment on the Cleaver Bank (Van Moorsal, 1994), some opportunistic sand-dwelling species invaded small deposits of sand in the furrows. By contrast, the seafloor of the industrial extraction site off Dieppe became characterized by fine sands and was colonized by a new benthic community.

In the Öresund between Denmark and Sweden, rocks, gravel, and calcareous concretions discarded from dredgers were colonized by epifauna and species composition was different inside and outside the dredging area (Norden Andersen *et al.*, 1992). The CNEXO experimental site provides another example: the intensive deposition of fine muddy sands in most of the area subsequently enabled colonization by a mud-dwelling community with more species and higher densities and biomass than the surrounding communities of sand-dwelling species.

Complete recovery of the benthos in a dredged area may take from one month to five years or more, depending upon the stocks of potential recolonizing species and their immigration distance (Bonsdorff, 1983). The recolonization rate is inversely proportional to dredging intensity (McCauley *et al.*, 1977; Millner *et al.*, 1977), a low number of furrows being rapidly repopulated by organisms from undisturbed hummocks of sediment.

On the Cleaver Bank (Van Moorsel, 1994), total density recovered in eight months and the majority of species had returned after one year. However, large bivalves, which are more important in terms of biomass, did not show signs of recovery after two years. The more extensive experiment of north Norfolk (Kenny and Rees, 1996) confirmed that recolonization processes proceed rapidly after cessation of dredging and that substantial progress was made towards recovery within the first 12 months, particularly in terms of densities and species richness. Again, the community differed substantially from its pre-dredged state in terms of biomass. In Denmark, the benthic fauna appeared to be close to total recovery in numbers as well as in biomass 17 months after massive trailing suction dredging had removed up to 2 m of sand (Norden Andersen *et al.*, 1992). Annual species had mostly recolonized the area, but perennial bivalves lagged behind, explaining the lower biomass in the dredging area.

Further examples (McCauley *et al.*, 1977; Bonsdorff, 1983) also indicate that the recovery potential for macrobenthic fauna following mechanical disturbance is high. Recolonization rates were higher in a high-energy environment (Wissant, Dover Straight) where no impact could be seen after five years (Davoult and Richard, 1986). Some evidence of recolonization (increase of species richness and densities) could even already be observed off Dieppe during the period of lower extraction intensity (Desprez, 1995).

These examples indicate that, even though biological and physical responses to dredging are site specific, dredging of gravel deposits in areas of relatively high natural disturbance may have relatively short-term (approximately three years) biological significance because of the rapid recovery of the physical environment and the biological communities. Indeed, communities of high-stress areas (e.g., shallow areas exposed to strong tidal currents and periodic storm disturbance) are more adept to readjustment to the impact of dredging operations than more stable communities and recovery may be largely completed within one year. On sandbanks, excavated sites are filled with sediment similar to that removed by the dredger and hence the colonizing benthic community is similar in composition to that originally present (Van der Veer *et al*., 1985).

Although initial recovery is rapid, it may take five years or more for a shallow water benthic community to stabilize, because many sessile biota require several years to reach maturity (Rees, 1987; Newell *et al*., 1998). Several studies (Kaplan *et al*., 1975; Simon and Dauer, 1977; Hily, 1983; Kenny and Rees, 1996) suggest that the response may be divided into three phases: (i) an initial recolonization (within 12 months) by the dominant opportunistic taxa; (ii) a reduced community biomass for up to three years caused by continued disturbance through sediment instability; (iii) recovery of the community biomass. Surprisingly, biomass had recovered more than abundance after two years at the site off Dieppe.

Impact on fisheries

In general, fish will be less affected than shellfish resources by dredging activities because they may evade the disturbed area. However, particular species may be particularly vulnerable if aggregate extraction activities coincide with areas where they breed or spawn. The prime risk of redeposition is smothering of fish eggs on spawning grounds, such as those of herring and sandeel (Westerberg *et al*., 1996). Suffocation of filter-feeding benthos such as mussels (Collinson and Rees, 1978) represents another danger. Also, many demersal fish species may be affected by the removal of benthos that provides an important source of food (Daan *et al*., 1990). However, although recovery of a dredging site will take about three years (de Groot, 1979a), benthic animals will colonize the area much sooner and may serve as a resource for demersal fish species, if the dominant organisms are acceptable as food (Millner *et al*., 1977).

Dredging may also create new habitats (presence of boulders, higher heterogeneity of sediment) and favour an increase in the richness of benthic macroinvertebrates and fish alike. A video survey at the CNEXO site revealed the presence of fish (dab, haddock, dragonet), cuttlefish, swimming and edible crabs under the outcrops of clay, whereas hermit and spider crabs dominated in the more silty sand areas.

The uneven seafloor topography created by dredging may interfere with bottom trawl gears. Trawls may be snagged around boulders exposed by removal of the surrounding substrate (de Groot, 1979b) and steep slopes of dredged furrows may reduce trawling efficiency. Both problems have been encountered at the two French dredging sites.

For finfish fisheries, dredging causes more of a loss of access to traditional grounds rather than to a direct loss of fish. If areas support important, seasonal fisheries of migrating fish, any redistribution of fish may have economic consequences for local vessels. In this case, the best approach is to time extraction operations so as to permit access to fishermen during this seasonal window (ICES, 1992). At Dieppe, dredging activity is stopped from October to February to enable the herring fishery to recover.

Fishing has also impacts on the benthic community. Trawling leaves visible marks on the sea floor and many invertebrates may be killed or damaged. Even in a dynamic environment with an adapted infaunal community, Eleftheriou and Robertson (1992) observed selective elimination of the fragile and sedentary components of the infauna, and the destruction of the large epifaunal and infaunal organisms. Long-term disturbance effect of trawling may lead to a re-structuring of the benthic system, with an increase of biomass, a dominance of opportunistic short-lived species and a decline of long-living sessile organisms (Lindeboom and de Groot, 1998). The disturbance of the seabed by fishing activity is far greater than that of all other activities: it has been estimated that marine aggregate extraction disturbs annually 0.03% of the seabed in the North Sea, whilst fishing disturbs about 54% (ICES, 1992). In view of the present scale of dredging activity, there is no substantiated evidence of any licensed activity having an observable deleterious effect on the ecology of a fishery (ICES, 2000).

Acknowledgements

I am grateful to all members of the ICES Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem, particularly to its former Chair Dr S. J. de Groot, for their assistance in improving my work, and my English, during the past 10 years. I also thank Dr Daridon (GIE Graves de Mer) who kindly gave permission to use the data acquired in baseline surveys off Dieppe on his behalf since 1986; he has also shown a keen interest in obtaining a better understanding of the possible environmental impact of gravel extraction. I also acknowledge all GEMEL members who contributed to data collection over the years.

References

Arntz, W. E., and Rumohr, H. 1982. An experimental study of macrobenthic colonization and succession, and the importance of seasonal variation in temperate latitudes. Journal of Experimental Marine Biology and Ecology, 64: 17–45.

Arthurton, R. S., Harisson, D. J., Laban, C., Leth, J. O., and Lillywhite, R. P. 1997. Marine and gravel in North-West Europe. A regional view of demand and resources. European Conference on Mineral Planning: 7 pp.

Bonsdorff, E. 1983. Recovery potential of macrozoobenthos from dredging in shallow brackish waters. *In* Proceedings 17th European Marine Biology Symposium, Brest, France, 27 September–1 October, 1982, pp. 27–32. Oceanologica Acta, No. SP.

Cabioch, L., and Glaçon, R. 1977. Distribution des peuplements benthiques en Manche Orientale du Cap d'Antifer à la Baie de Somme. Comptes Rendus Académie Sciences Paris, 285(D): 209–211.

Collinson, R. I., and Rees, C. P. 1978. Mussel mortality in the Gulf of La Spezia, Italy. Marine Pollution Bulletin, 9: 99–101.

Cruickshank, M. J., and Hess, H. D. 1975. Marine sand and gravel mining. Oceanus, 19: 32–44.

Daan, N., Bromley, P. J., Hislop, J. R. G., and Nielsen, N. A. 1990. Ecology of North Sea fish. Netherlands Journal of Sea Research, 26: 343–386.

Davoult, D., and Richard, A. 1986. Analyse du benthos au large du site des caps. Recherche de l'impact d'une extraction de granulats. Rapport Station Marine Wimereux, 40 pp.

De Groot, S. J. 1979a. An assessment of the potential environmental impact of large-scale sand-dredging for the building of artificial islands in the North Sea. Ocean Management, 5: 211–232.

De Groot, S. J. 1979b. The potential environmental impact of marine gravel extraction in the North Sea. Ocean Management, 5: 233–249.

De Groot, S. J. 1996. The physical impact of marine aggregate extraction in the North Sea. ICES Journal of Marine Science, 53: 1051–1053.

Desprez, M., and Duhamel, S. 1993. Comparison of impact of gravel extraction on geomorphology, sediment and macrofauna in two areas: Klaverbank (NL) and Dieppe (F). ICES CM 1993/E: 7, 17 pp.

Desprez, M. 1995. Biological and sedimentological impact of a marine aggregate extraction site (Dieppe) along the French coast of the English Channel. Preliminary results on post-dredging recolonization. ICES CM 1995/E: 5: 73–79.

Desprez, M. 1997. Exploitation des agrégats: bilan, enseignements et menaces sur les biocénoses. *In* Les Biocénoses Marines et Littorales Françaises des Côtes Atlantique, Manche et Mer du Nord. Synthèse, menaces et perspectives. Ed. by J. C. Dauvin. Museum National Histoire Naturelle. pp. 189–195.

Eden, R. A. 1975. North Sea environmental geology in relation to pipelines and structures. Oceanology International, 75: 302–309.

Eleftheriou, A., and Robertson, M. R. 1992. The effects of experimental scallop dredging on the fauna and physical environment of a shallow sandy community. Netherlands Journal of Sea Research, 30: 289–299.

Greening, J., and Kenny, A. J. 1996. The diet of fish and shellfish from gravel areas off England and Wales. Report Crown Estate and MAFF: 22 pp.

Hily, C. 1983. Macrozoobenthic recolonization after dredging in a sandy mud area of the Bay of Brest enriched by organic matter. No. sp. Oceanologica Acta: 113–120.

Hitchcock, D. R., and Drucker, B. R. 1996. Investigation of benthic and surface plumes associated with marine aggregates mining in the United Kingdom. Oceanology International, 2: 221–234.

ICES. 1992. Report of the ICES Working Group on the Effects of Extraction of Marine Sediments on Fisheries. ICES Cooperative Research Report, No. 182, 78 pp.

ICES. 2001. Effects of Extraction of Marine Sediments on the Marine Ecosystem. ICES Cooperative Research Report. In press.

Kaplan, E. H., Welker, J. R., Kraus, M. G., and McCourt, S. 1975. Some factors affecting the colonisation of a dredged channel. Marine Biology, 32: 193–204.

Kenny, A. J., and Rees, H. L. 1996. The effects of marine gravel extraction on the macrobenthos: results two years post-dredging. Marine Pollution Bulletin, 32: 615–622.

Lees, R. G., Rees, H. L., Lambert, M. A., Rowlatt, S. M., and Limpenny, D. S. 1990. Benthic studies in relation to dredging activity off the Isle of Wight, Southern England. ICES CM 1990/E: 48, 11 pp.

Lindeboom, H. J., and de Groot, S. J. 1998. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. NIOZ-Rapport 1998-1, RIVO-DLO report C003/98: 404 pp.

MacCauley, J., Parr, R. A., and Hancock, D. R. 1977. Benthic infauna and maintenance dredging: a case study. Water Research, 11: 233–247.

Millner, R. S., Dickson, R., and Rolfe, M. S. 1977. Physical and biological studies of a dredging ground off the east coast of England. ICES CM 1977/E: 48, 11 pp.

Newell, R. C., Seiderer, L. J., and Hitchcock, D. R. 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. Oceanography and Marine Biology: an Annual Review, 36: 127–178.

Norden Andersen, O. G., Nielsen, P. E., and Leth, J. 1992. Effects on sea bed, benthic fauna and hydrography of sand dredging in Køge Bay, Denmark. *In* Proceedings of the 12th Baltic Marine Biologists Symposium, Helsingør, Denmark, 25–30 August 1991, pp. 1–6. International Symposium Series. Olsen & Olsen, Fredensborg.

Rees, H. L. 1987. A survey of the benthic fauna inhabiting gravel deposits off Hastings, Southern England. ICES CM 1987/L: 19, 9 pp.

Shelton, R. G. J., and Rolfe, M. S. 1972. The biological implications of aggregate extraction: recent studies in the English Channel. ICES CM 1972/E: 26, 13 pp.

Simon, J. L., and Dauer, D. M. 1977. Reestablishment of a benthic community following natural defaunation. *In* Ecology of Marine Benthos, pp. 139–154. University of South Carolina Press, Columbia, South Carolina, USA.

Van Dalfsen, J. A., and Essink, K. 1998. The effects on macrozoobenthos of subaqueous sand extraction North of the Island of Terschelling, The Netherlands. ICES CM 1998/E: 5, 7 pp.

Van der Veer, H. W., Bergman, M. J. N., and Beukema, J. J. 1985. Dredging activities in the Dutch Wadden Sea: effects on macrobenthic fauna. Netherlands Journal of Sea Research, 19: 183–190.

Van Moorsel, G. 1994. The Klaverbank (North Sea), geomorphology, macrobenthic ecology and the effect of gravel extraction. Bureau Waardenburg bv, Culemborg, NL, Rep. 94. 24: 65 pp.

Westerberg, H., Ronnback, P., and Frimansson, H. 1996. Effects of suspended sediments on cod eggs and larvae and on the behaviour of adult herring and cod. ICES CM 1996/E: 26.

Winterhalter, B. 1990. The Baltic marine environment as a source of aggregates and as a recipient of dredged material. Ocean Resources, 1: 153–158.