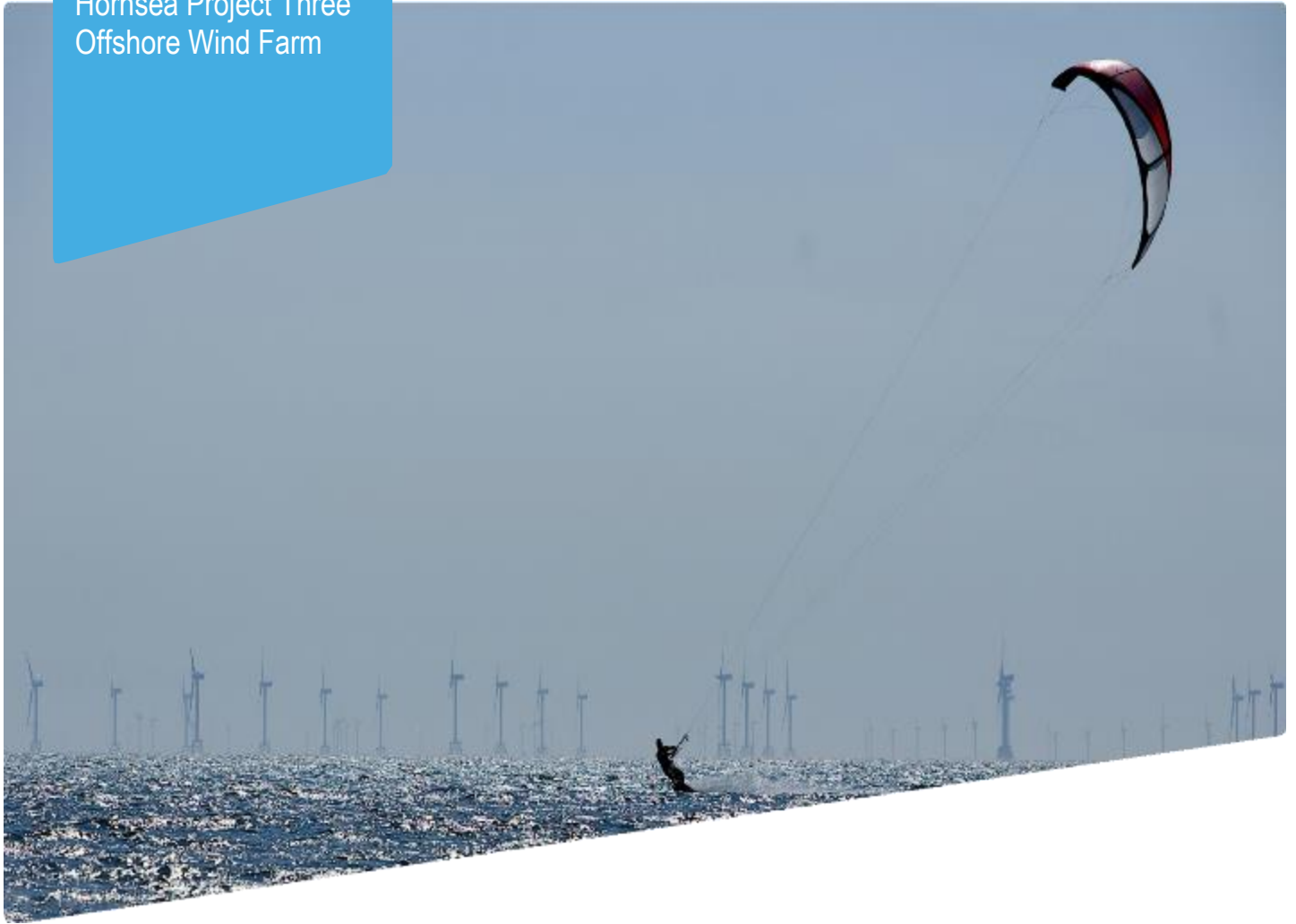


Hornsea Project Three  
Offshore Wind Farm



## Hornsea Project Three Offshore Wind Farm

Appendix 17 to Deadline 7 submission – Newell et al., 1998

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# 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

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**Abstract** The present review provides a framework within which the impact of dredging on biological resources that live on the sea bed ("Benthic" communities) can be understood, and places in perspective some of the recent studies that have been carried out in relation to aggregates dredging in European coastal waters. The impact of dredging works on fisheries and fish themselves, and on their spawning grounds is outside the scope of this review. We have, however, shown that empirical models for shelf waters such as the North Sea indicate that as much as 30% of total fisheries yield to man is derived from benthic resources, and that these become an increasingly important component of the food web in near-shore waters where primary production by seaweeds (macrophytes) and seagrasses living on the sea bed largely replaces that by the phytoplankton in the water column. Because dredging works are mainly carried out in near-shore coastal deposits, and these are the ones where benthic production processes are of importance in supporting demersal fish production, our review concentrates on the nature of benthic communities, their sensitivity to disturbance by dredging and land reclamation works, and on the recovery times that are likely to be required for the re-establishment of community structure following cessation of dredging or spoils disposal.

Essentially, the impact of dredging activities mainly relates to the physical removal of substratum and associated organisms from the sea bed along the path of the dredge head, and partly on the impact of subsequent deposition of material rejected by screening and overspill from the hopper. Because sediment disturbance by wave action is limited to depths of less than 30 m, it follows that pits and furrows from dredging activities are likely to be persistent features of the sea bed except in shallow waters where sands are mobile. Recent studies using Acoustic Doppler Current Profiling (ADCP) techniques suggest that the initial sedimentation of material discharged during outwash from dredgers does not, as had been widely assumed, disperse according to the Gaussian diffusion principles used in most simulation models, but behaves more like a density current where particles are held together during the initial phase of the sedimentation process. As a result, the principal area likely to be affected by sediment deposition is mainly confined to a zone of a few hundred metres from the discharge chute.

Our review suggests that marine communities conform with well established principles of ecological succession, and that these allow some realistic predictions on the likely recovery of benthic communities following cessation of dredging. In general, communities living in fine mobile deposits, such as occur in estuaries, are characterized by large populations of a restricted variety of species that are well adapted to rapid recolonization of deposits that are subject to frequent disturbance. Recolonization of dredged deposits is initially by these "opportunistic" species and the community is subsequently supplemented by an increased species variety of long-lived and slow-growing "equilibrium" species that characterize stable undisturbed deposits such as coarse gravels and reefs.

Rates of recovery reported in the literature suggest that a recovery time of 6-8 months is character-

istic of many estuarine muds where frequent disturbance of the deposits precludes the establishment of long-lived components. In contrast, the community of sands and gravels may take 2–3 yr to establish, depending on the proportion of sand and level of environmental disturbance by waves and currents, and may take even longer where rare slow-growing components were present in the community prior to dredging. As the deposits get coarser along a gradient of environmental stability, estimates of 5–10 yr are probably realistic for development of the complex biological associations between the slow-growing components of equilibrium communities characteristic of reef structures.

Most recent studies show, however, that biological community composition is not controlled by any one, or a combination of, simple granulometric properties of the sediments such as particle size distribution. It is considered more likely that biological community composition is controlled by an array of environmental variables, many of them reflecting an interaction between particle mobility at the sediment–water interface and complex associations of chemical and biological factors operating over long time periods. Such interactions are not easily measured or analyzed, but the results suggest that the time course of recovery of an equilibrium community characteristic of undisturbed deposits is controlled partly by the process of compaction and stabilisation that occurs following deposition.

Biological community composition thus reflects changes in sediment composition, but is also in equilibrium with seabed disturbance from tidal currents and wave action, both of which show spatial variations and interactions with water depth. The processes associated with compaction and stability of seabed deposits may, therefore, largely control the establishment of long-lived components of equilibrium communities and account for the dominance of opportunistic species in the initial stages of colonization in unconsolidated deposits of recently sedimented material after the cessation of dredging.

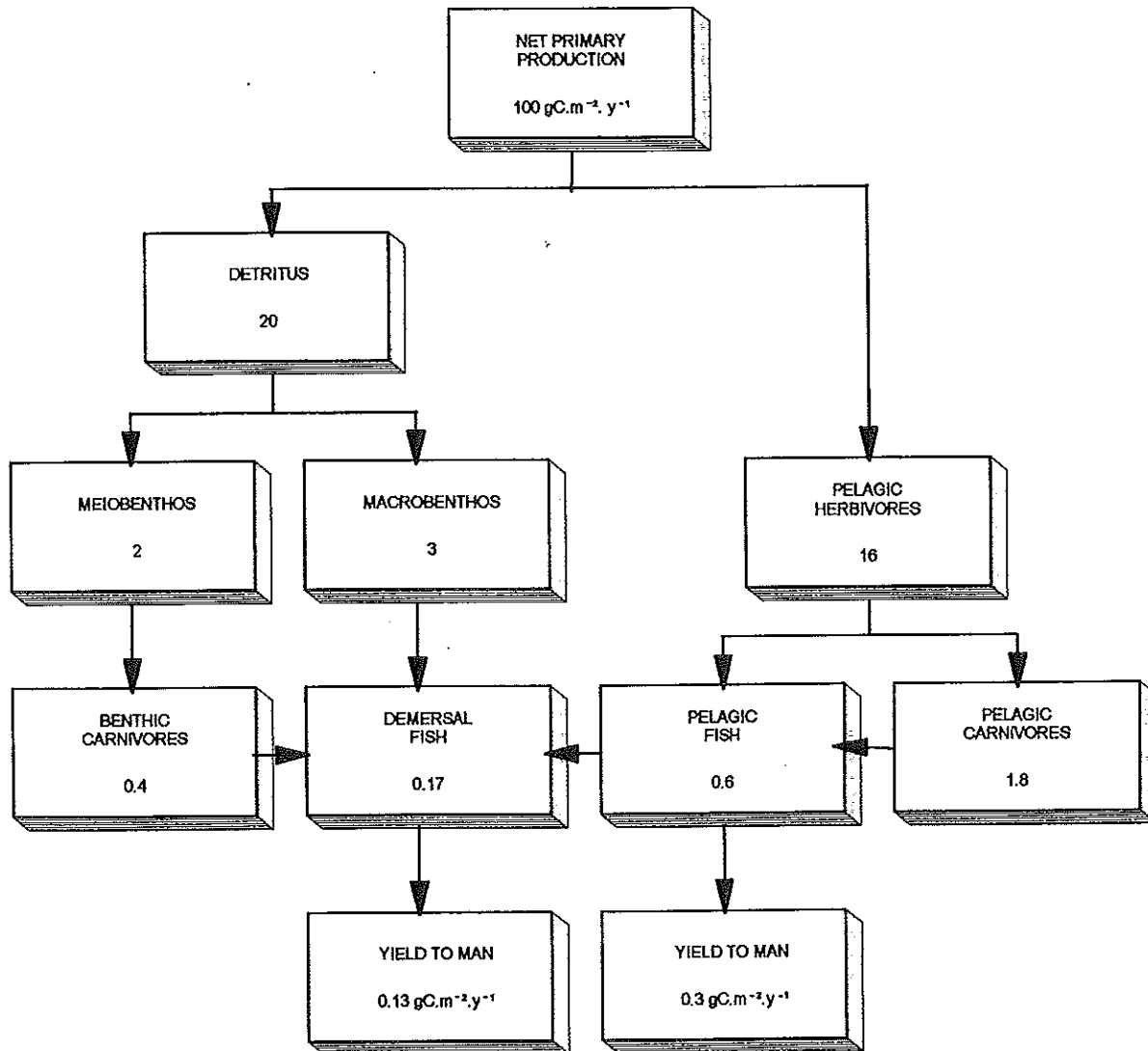
## Introduction

The importance of benthic communities in marine food webs leading to commercially exploitable yields of fish has been widely recognized. Early models for the North Sea (see Steele 1974) suggested that of net primary production by the phytoplankton, approximately 80% was consumed by pelagic herbivores such as copepods and euphausiids, and 20% fell to the sea bed as a detrital input to the benthic community. At each step of the food web, relatively large amounts (80–90%) of the material entering the consumers is remineralized and returned to the water column to support further primary production by the phytoplankton, leaving a small proportion incorporated into the biomass of the consumer.

Because of the complexity of marine food webs, and the major dissipation of energy at each step of the food chain, the empirical model proposed by Steele (1965, 1974) for the North Sea and shown in Fig. 1 indicates that out of  $100 \text{ g C m}^{-2} \text{ yr}^{-1}$  produced at the sea surface as net primary production by the phytoplankton, only  $0.3 \text{ g C m}^{-2} \text{ yr}^{-1}$  appears as yield to man through the pelagic food web, and approximately  $0.13 \text{ g C m}^{-2} \text{ yr}^{-1}$  from demersal fish. Despite the huge dissipation of materials that occurs at each step in the food web, however, sufficient carbon evidently flows through the detrital food web, even in plankton-based ecosystems such as the North Sea, for as much as 30% of total fish production to be dependent on conversion through the community which lives on the sea bed.

More recent analyses of the trophic structure and fluxes of carbon in shelf waters of the North Sea by Joiris et al. (1982) suggest that as much as 50% of the annual phytoplankton production sinks to the sea bed as detritus and is supplemented by faecal

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**Figure 1** Simplified empirical carbon flow diagram for the phytoplankton-based ecosystem of the shelf waters of the North Sea. Note that, of the  $100 \text{ g C m}^{-2} \text{ yr}^{-1}$  of sea surface produced by the phytoplankton, the yield to man through pelagic food webs and pelagic fish is estimated to be  $0.3 \text{ g C m}^{-2} \text{ yr}^{-1}$  whereas that through benthic food webs is  $0.13 \text{ g C m}^{-2} \text{ yr}^{-1}$ , or about 30% of the total exploitable fish yield to man. (Based on Steele 1965).

pellets of the zooplankton (see also Smetacek, 1984). The benthos is thus heavily implicated in carbon flow in coastal systems, and becomes of increasing importance in shallow waters where production by benthic algae (macrophytes) and seagrasses largely replaces that derived from the phytoplankton (see also Taylor & Saloman 1968, Thayer et al. 1975, Mann 1982, Moloney et al. 1986, Newell et al. 1988).

Benthic communities thus play a central role in the transfer of materials from primary production by the phytoplankton, benthic macrophytes and coastal wetlands through the detrital pool into higher levels in the food web, including commercially exploitable fish. Most estimates suggest that even in phytoplankton-based systems such as the North Sea, the yield to man through the benthos to demersal fish stocks is likely to approach 30–40% of that derived through the pelagic system. Partly for this reason, the populations of benthic communities which live on, and in, the deposits on the sea

bed have been widely studied in investigations of the integrated effects of disturbance from a variety of natural and other sources.

Early studies include extensive physiological-toxicological work on the potential impact of suspended sediments on commercially significant target organisms (Loosanoff 1962, Sherk 1971, Sherk et al. 1972, 1974, Bright & Ellis 1989, Jokiel 1989, for review, see Moore 1977). Such studies have been extended to include the potential impact of dredging works on the ecology of biological communities in coastal embayments and estuarine ecosystems (Jones & Ellis 1976, Morton 1977, Conner & Simon 1979, Johnston 1981, Ellis & Heim 1985, Ellis & Taylor 1988, Ellis & Hoover 1990, Giesen et al. 1990, Onuf 1994).

Comprehensive studies of the impact of dredging for marine aggregates and sand on marine communities in European waters have been carried out by Millner et al. (1977), Pagliai et al. (1985), Sips & Waardenburg (1989), van Moorsel & Waardenburg (1990, 1991), and Kenny & Rees (1994, 1996). Reviews of the impact of sand and gravel extraction include those of the International Council for the Exploration of the Sea (ICES, 1975, 1977, 1992a,b, 1993), Gayman (1978), de Groot (1986), Nunny & Chillingworth (1986), Hurme & Pullen (1988), Lart (1991) and Charlier & Charlier (1992). A recent review for the Minerals Management Service, US Department of the Interior containing a number of specific case histories on the impact of marine mining has been given in a C-CORE publication (1996; see also Ellis 1987).

Despite the work that has been carried out over the past 30 years, the non-biologist could be forgiven for being bewildered by the diversity of the results and the difficulties of making more than the most general predictions on the effects of dredging activities including the extraction of marine aggregates on biological resources. Essentially, most studies show that dredging itself is usually accompanied by a significant fall in species numbers, population density and biomass of benthic organisms. The rate of recovery is, however, highly variable depending (among other factors) on the type of community that inhabits the deposits in the dredged area and surrounding deposits, the latitude and the extent to which the community is naturally adapted to high levels of sediment disturbance and suspended particulate load.

In general, rapid rates of initial recolonization have been reported for some coastal deposits where the organisms are mainly mobile "opportunistic" species that have a rapid rate of reproduction and growth. Such organisms may also be able to recolonize the deposits by migration of the adults (see McCall 1976, Conner & Simon 1979, Saloman et al. 1982, Guillou & Hily 1983, Pagliai et al. 1985, van der Veer et al. 1985, Clarke & Miller-Way 1992, Rees & Dare 1993, van Moorsel 1994). In contrast, long-lived and slow-growing species, especially those in high latitudes may take several years before larval recruitment and subsequent growth of the juveniles allows restoration of the original community composition and biomass.

The process of "recovery" following environmental disturbance is generally defined as the establishment of a successional community of species which progresses towards a community that is similar in species composition, population density and biomass to that previously present, or at non-impacted reference sites (C-CORE 1996; see also Ellis & Hoover 1990). Typically, values range from up to one year in fine-grained deposits such as muds and clays (Ellis et al. 1995), although even in the fine deposits that characterize coastal ecosystems such as the Dutch Waddensea, van der Veer et al. (1985) report that recolonization takes 1-3 yr in areas of strong currents but up to 5-10 yr in areas of low current velocity. Longer recovery times are reported for sands

and gravels where an initial recovery phase in the first 12 months is followed by a period of several years before pre-extraction population structure is attained (van Moorsel 1994, Kenny & Rees 1996).

Even longer times may be required for biologically-controlled communities that characterize coarse deposits (see Garnett & Ellis 1995), although the evidence is conflicting for coral reef communities. Some studies report long-term damage to coral resources from sedimentation associated with dredging (Dodge & Vaisnys 1977, Bak 1978, Dodge & Brass 1984, Madany et al. 1987, Hodgson 1994; for review see Maragos 1991). Other studies suggest that corals themselves may be tolerant of short-term increases in siltation associated with dredging (Marszalek 1981, Brown et al. 1990) but that modification of community structure of other components of reef communities such as fish species are detectable after multivariate analysis of species composition (Dawson-Shepherd et al. 1992).

Recovery times following disturbance from a variety of sources, including dredging work, may be extended in colder waters at high latitudes where communities typically comprise large slow-growing species that may take many years for recolonization and growth. In a Swedish fjord system, for example, a recovery that was indistinguishable from natural variations was established only after 8 yr following closure of a pulp mill (Rosenberg 1976), whereas de Groot (1979; see also Wright 1977, Aschan 1988) reports that recovery of communities within the Arctic Circle may take more than 12 yr compared with estimates of approximately 3 yr for deposits off the coast of the Netherlands. Similar extended timescales for recolonization by the benthic community have been reported for Antarctic waters by Oliver & Slatterly (1981).

The concept of "recovery" of biological resources is itself not an easy one to define for complex communities whose composition can vary over time, even in areas that remain undisturbed. Whether a community is identical in species composition and population structure following cessation of dredging thus to some extent begs the question of whether the biodiversity would have remained stable over that period in the absence of disturbance by dredging. Probably a more practical approach to the question of "recovery" will be the recognition of the establishment of a community that is capable of maintaining itself and in which at least 80% of the species diversity and biomass has been restored.

This implies a substantial restoration of the carrying capacity of the benthic food webs leading to fish, even though the precise species composition may not be identical to that recorded in the pre-dredged system. This issue of whether biological resources have been restored, and how this should be assessed, is of considerable importance in areas such as Canadian coastal waters where recovery of seabed resources forms part of a Statutory obligation following cessation of mining (D. V. Ellis pers. comm.).

Despite the complexity of the results for specific dredged areas, some firm general principles governing community structure following environmental disturbance have emerged in recent years and these appear to be generally applicable to a wide variety of communities both on the land and on the sea bed. The application of such concepts to coastal communities allows some credible predictions on the scale of impact of environmental disturbance such as that imposed by dredging and dredged spoils disposal and, more important, gives some insight into how long it might take for recovery in dredged areas and the surrounding deposits once dredging has ceased.

The object of the present review is to provide a framework within which the biological impact and subsequent recovery of benthic resources can be understood, with

examples drawn mainly from the impact of dredging works in near-shore waters and estuarine systems.

### General features of community structure

Most general models of community structure are based on the concept that biological communities do not form a series of distinct groups or assemblages along an environmental gradient, but show a corresponding gradient in community composition. Species that colonize habitats with unpredictable short-term variations in environmental conditions at one end of an environmental gradient of stability are subject to frequent catastrophic mortality. Such conditions occur in many shallow-water, intertidal and estuarine habitats and are characterized by populations which tend to have a high genetic variability that allows at least some components of the population to survive environmental extremes (see Grassle & Grassle 1974, Guillou & Hily 1983). Such organisms are thus selected for maximum rate of population increase, with high fecundity, dense settlement, rapid growth and rather a short life cycle. They are well suited to rapid invasion and colonization of environments where space has been left by a previous catastrophic mortality, whether this has been induced by natural factors or disturbance by man.

Such components have been designated "*r*-strategists" in a pioneer work by MacArthur & Wilson (1967; see also Pianka 1970), although we prefer to use the term "opportunists" for all such early colonizing species. Opportunists rely on a large investment in reproductive effort, rather than on mobility, for success in colonizing habitats made available by the catastrophic destruction of the previous community (see Gadgil & Solbrig 1972, McCall 1976).

Many communities living in unstable environments may comprise small, highly mobile species that are able to take advantage of recently created empty habitats quickly and to colonize them with large populations. These mobile colonizers are often associated with frequently-disturbed habitats (see Osman 1977). We distinguish these as "mobile opportunists" (see also MacArthur 1960, Grassle & Grassle 1974). All such mobile opportunists are *r*-strategists with life-cycle traits of small size, high fecundity, rapid growth and high mortality.

Under the stable conditions that occur at the other end of the environmental continuum, the community is controlled mainly by biological interactions, rather than by extremes of environmental variability. Here the organisms have an "equilibrium strategy" in which they are selected for maximum competitive ability in an environment that is already colonized by many species and in which space for settlement and subsequent growth is limiting. Such organisms are designated "*K*-strategists" or "equilibrium species" and devote a larger proportion of the resources to non-reproductive processes such as growth, predator avoidance and investment in larger adults (MacArthur & Wilson 1967, Gadgil & Bossert 1970, McCall 1976). Between these two extremes are communities whose species may be intermediate between those that occur at the extremes of the environmental gradient and have different relative



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**Table 1** Table summarizing the population characteristics of *r*-selected opportunists and *K*-selected equilibrium species (based on Pianka 1970, McCall 1976, Rees & Dare 1993, Holt et al. 1995).

Early colonizing species <i>r</i> -selected	Equilibrium species <i>K</i> -selected
<ol style="list-style-type: none"> <li>1. Mainly opportunistic species               <ol style="list-style-type: none"> <li>(a) early reproduction</li> <li>(b) many reproductions per year</li> <li>(c) rapid growth</li> <li>(d) early colonizers</li> <li>(e) often catastrophic mortality</li> </ol> </li> <li>2. Small body size</li> <li>3. Generally surface deposit feeders</li> <li>4. Short life span; generally &lt;1 year</li> <li>5. Population size variable, usually well below carrying capacity of environment and recolonized frequently</li> <li>6. Brood protection with investment of energy into larval food provision (lecithotrophic)</li> </ol> <p> <i>Streblospio benedicti</i>  <i>Capitella capitata</i>  <i>Owenia fusiformis</i>  <i>Ampelisca abdita</i>  <i>Scolecopsis fuliginosus</i>  <i>Chaetozone setosa</i>  <i>Jassa marmorata</i> </p>	<ol style="list-style-type: none"> <li>1. Equilibrium species               <ol style="list-style-type: none"> <li>(a) delayed reproduction</li> <li>(b) few reproductions per year</li> <li>(c) slow growth</li> <li>(d) late colonizers</li> <li>(e) low death rate</li> </ol> </li> <li>2. Large mobile animals</li> <li>3. Deposit and suspension feeders</li> <li>4. Long life span; several to many years</li> <li>5. Fairly constant in time; saturated community in equilibrium with carrying capacity of environment. No recolonization necessary</li> <li>6. No brood protection; larvae widely distributed in the plankton.</li> </ol> <p> <i>Nephtys incisa</i>  <i>Ensis directus</i>  <i>Sabellaria spinulosa</i>  <i>Arctica (Cyprina) islandica</i>  <i>Echinocardium cordatum</i>  <i>Nephrops norvegicus</i>  <i>Melinna cristata</i>  <i>Nucula sp.</i>  <i>Amphiura filiformis</i>  <i>Terebellides sp.</i>  <i>Virgularia mirabilis</i>  <i>Gart fervensis</i>  <i>Tellina crassa</i>  <i>Venerupis rhomboides</i>  <i>Dosinia exoleta</i>  <i>Scoloplos armiger</i>  <i>Abra alba</i> </p>

proportions of opportunistic *r*-strategists and equilibrium *K*-strategists. The characteristics of *r*-selected and *K*-selected equilibrium species are summarized in Table 1 (based on McCall 1976, Rees & Dare 1993), although it should be emphasized that the distinction is to some extent an arbitrary one, and is blurred in habitats that are subject to only mild environmental disturbance.

Changes in the structure and physical size of the infauna along a gradient of environmental conditions have been described in relation to organic pollution by Pearson & Rosenberg (1978) and in relation to physical disturbance by Rhoads et al. (1978), Oliver et al. (1980) and by Gray & Pearson (1982). These are illustrated in a schematic diagram in Fig. 2. Essentially, such studies show that community composition of benthic infauna (those that live within the deposits) along an environmental gradient is the result of a complex interaction between physico-chemical factors that

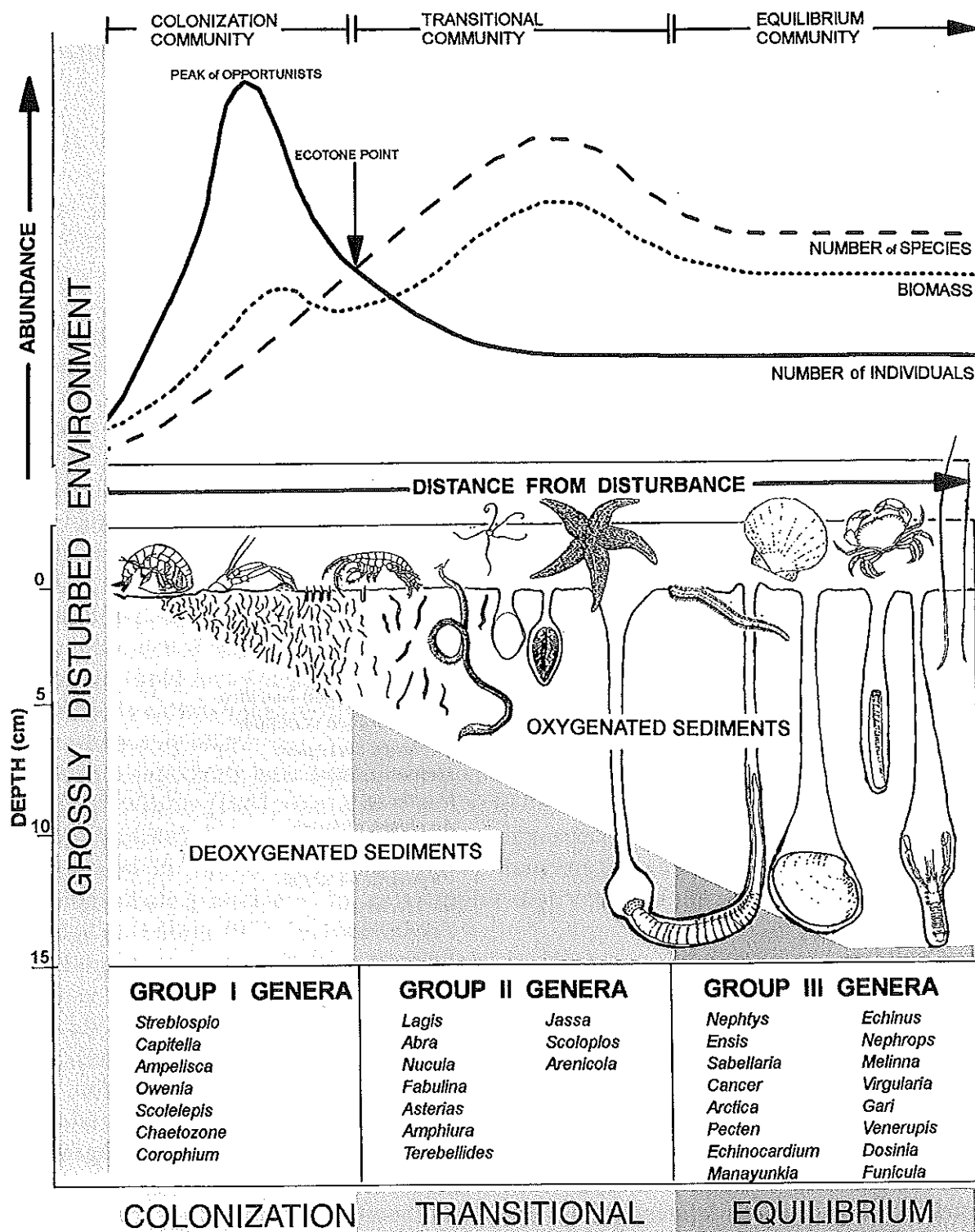


Figure 2 Pictorial diagram showing the ecological succession that characterizes benthic communities through a gradient of environmental disturbance. Note that in highly disrupted environments (on the left side of the diagram) few organisms may be capable of survival. In polluted or semi-liquid muds the sediments are colonized by few (resistant) species but which can attain very high population densities. As the stability of the environment increases, these opportunistic *r*-selected species are replaced by increased species variety, including slower-growing *K*-selected species. Finally in environments of high stability the community is dominated by equilibrium species with complex biological interactions between members of the community. (Based on Pearson & Rosenberg 1978, Rhoads et al. 1978).

operate at one end of the gradient and biologically-controlled interactions under the more uniform environmental conditions that occur in deeper waters (see Sanders 1969, Boesch & Rosenberg 1981).

The large species that comprise the burrowing infauna of stable habitats and those with low organic content maintain oxygen levels in the deposits down to considerable depths (see Flint & Kalke 1986) and often have complex interactions with neighbouring species, including smaller species whose survival depends on their association with large burrowing components (Fig. 2). The importance of bioturbation in both enhancing species diversity and in exclusion of potentially competitive species has been widely documented (Gray 1974, Rhoads 1974, Lee & Swartz 1980, Carney 1981, Rhoads & Boyer 1982, Thayer 1983). Comprehensive reviews by Pearson & Rosenberg (1978) and Hall (1994) summarize the impact of disturbance by a wide variety of factors including storms, dredging, fishing and biological activities on benthic community structure.

Biological interactions may also control community composition on the surface of the deposits. The presence of surface-dwelling bivalves, for example, may allow colonization by barnacles, ascidians and other epifaunal species that would not otherwise occur in the surface of the sediments. In other stable habitats, the activities of suspension-feeding mussels produce consolidated silt deposits that then allow deposit feeders such as the polychaete *Amphitrite*, the burrows of which in turn provide specialized shelter for the commensal scale worm, *Gattyana* (Newell, 1979).

Several studies have shown that the activities of the infauna may also inhibit, rather than facilitate, the occurrence of potential competitors for space. In an important study by Rhoads & Young (1970), it was shown that the benthic environment may be significantly modified by the burrowing and feeding activities of deposit-feeding organisms. This bioturbation results in the production of an uncompacted surface layer of faecal material that may result in the transfer of fine material to the sediment-water interface by turbulent mixing (Wildish & Kristmanson 1979, Snelgrove & Butman 1994) and may lead to the exclusion of potential competitors by deposit feeders (Woodin 1991, Woodin & Marinelli 1991). This inhibition of one type of population by the activities of another has been termed "amensalism" by Odum & Odum (1959) and has since been described in many habitats (Aller & Dodge 1974, Nichols 1974, Driscoll 1975, Eagle 1975, Johnson 1977, Myers 1977a,b, Brenchley 1981, de Witt & Levinton 1985, Brey 1991, Flach 1992).

Loss of these "key species" in *K*-dominated equilibrium communities following disturbance by dredging or other activities can lead to a collapse of the entire biologically-accommodated community even though individual species may be apparently tolerant of environmental disturbance. The colonial polychaete *Sabellaria spinulosa*, for example, provides a complex habitat that is associated with a wide variety of species which would not otherwise occur (see Holt et al. 1995). This polychaete undergoes a natural cycle of accretion and decay along with the associated community with a periodicity of from 5–10 yr (Wilson 1971, Gruet 1986). Disturbance of communities that are dominated by *K*-strategists may therefore take many years for recovery of their full community composition even though recolonization by individual components may occur comparatively rapidly.

As the amount of organic matter in the sediments increases along a gradient towards the fine silts and muds that characterize estuarine habitats, the larger species and deep-burrowing forms are replaced by large numbers of relatively inactive small suspension-feeding and surface deposit feeders including polychaete worms, bivalves and

holothurians. This reduction in the species diversity and extent of sediment bioturbation results in an increased sediment stability and a restriction of the oxygenated layer to the surface of the sediments. Species in the intermediate parts of the environmental gradient shown in Figure 2 are thus relatively smaller than their counterparts in deeper waters and comprise a "transitional community" that is confined to a restricted habitat in the surface oxygenated layer of sediment and includes components that have many intermediate characteristics between typical *r*-selected opportunists and *K*-selected equilibrium species.

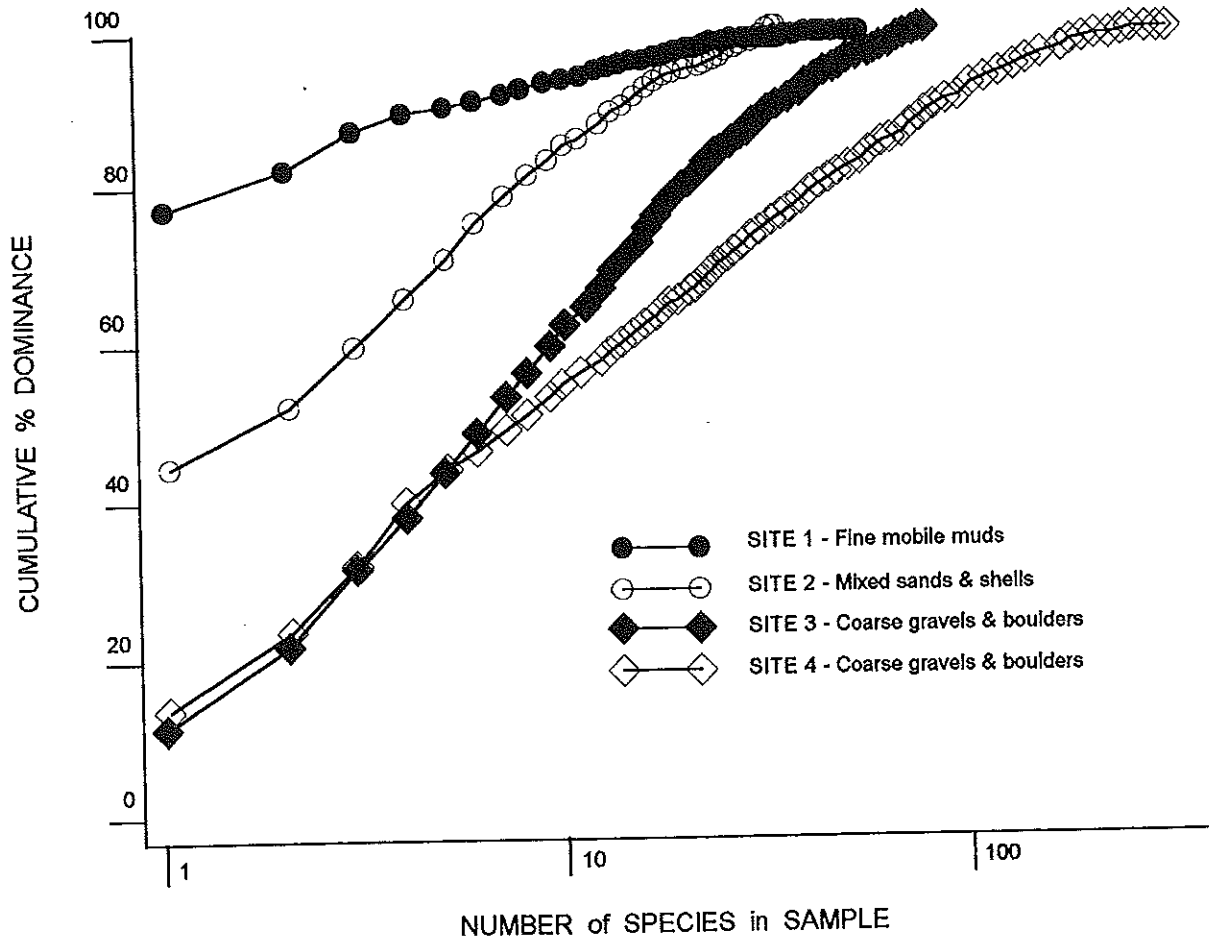
Because the *K*-selected components in the community live for longer, the individuals must be able to tolerate short-term changes in environmental conditions including siltation. They therefore have generally wider limits of physiological tolerance than *r*-selected opportunistic species that respond to environmental change by selection of genetically adapted components of the population during each of the many reproductive cycles per year.

The transitional community comprises more species than the equilibrium community shown in Figure 2 because of invasion by opportunistic species, but the species variety and mean size rapidly decline as the organisms are increasingly crowded into the upper oxygenated layer at the sediment – water interface. The region between this transitional community and those dominated by large populations of a restricted variety of small opportunists has been referred to as the "ecotone point" by Pearson & Rosenberg (1978) and is shown in Figure 2.

Finally at the extreme end of the physical gradient shown in Figure 2, there is a further restriction of habitat space to the upper oxygenated layer of sediment. This results in a progressive elimination of species and to communities dominated by opportunistic *r*-strategists that are selected for small size, high fecundity and an ability to recolonize rapidly following catastrophic mortality (see Pearson & Rosenberg 1978, Gray & Pearson 1982). Very high population densities of these *r*-selected opportunists can occur (the "Peak of Opportunists" in Figure 2) before these decline as organic pollution or high environmental disturbance eliminates even these rapid colonizers.

A useful tool for determining the extent of impact of environmental impact from a variety of sources is a plot of the proportional contribution of each species in the community to the overall population density of the assemblage as a whole. These curves have been designated "K-dominance curves" by Lamshead et al. (1983) and have been widely used in environmental impact studies in recent years (Warwick 1986, Clarke & Warwick 1994). Obviously the equilibrium communities characteristic of undisturbed (or unpolluted) environments have a high species diversity and each component species makes a relatively small contribution to the overall population density. Conversely, as a point source of disturbance is approached the (sensitive) species are replaced by large numbers of those (resistant) members of the community that are capable of survival. This can lead to as much as 80–90% of the population being dominated by only one or two opportunists or *r*-selected species at the Peak of Opportunists shown in Figure 2.

A typical set of results taken from one of our surveys of coastal communities in the eastern English Channel is shown in Fig. 3 (Newell & Seiderer, 1997c). From this it can be seen that as much as 78% of the community in unstable, unconsolidated, mobile deposits at Site 1 was represented by just one species, the opportunist amphipod crustacean *Ampelisca brevicornis*, and that additional species each made only a relatively small contribution to the population. Further along the gradient of sediment stability



**Figure 3** A set of typical *K*-dominance curves showing the proportional contribution of individual species to the overall community in fine mobile muds, in mixed sands and shells and in stable habitats of coarse gravels and boulders off the Kent coast at West Varne in 1996. The fine mobile muds are dominated by the opportunist amphipod crustacean, *Ampelisca brevicornis*, whereas the more stable deposits have a higher total species complement each of which makes a relatively small contribution to the overall population density. (Based on Newell & Seiderer 1997c).

in mixed sands and shells at Site 2, the dominance by one species (*Sabellaria* sp.) alone was approximately 45%. Finally, in the stable environmental conditions of coarse gravels and boulders at Sites 3 and 4 there was a very large species variety of over 300 and a relatively uniform species distribution with dominance values of only 12–15%.

Estimation of *K*-dominance curves adjacent to dredging works and other point sources of environmental disturbance is thus potentially useful because it can be used as a relatively simple index to define the area of immediate impact. It can also be used to determine whether this is enlarging or decreasing with time, without the necessity of the complex analysis of community structure that is required for interpretation of the wider impact on community structure in the transition zone.

These distinctions between the lifestyles and adaptive strategies of opportunists and *K*-selected equilibrium species are of fundamental importance because they go some way towards accounting for the differences in the rate of recovery that has been recorded for biological resources following disturbance by episodic events such as dredging.

Clearly, the species composition and rate of recovery of biological communities following cessation of dredging will depend to a large extent on whether the original communities were dominated by opportunists or equilibrium species, and on the time that is required to develop the complex associations which characterise interactions between the *K*-dominated equilibrium community. Knowledge of the key faunal components and their lifestyle thus allows some predictions on the impact of dredging and spoils disposal on biological resources and on the subsequent rate of recovery of marine community composition following cessation of dredging.

### Ecological succession and the recolonization process

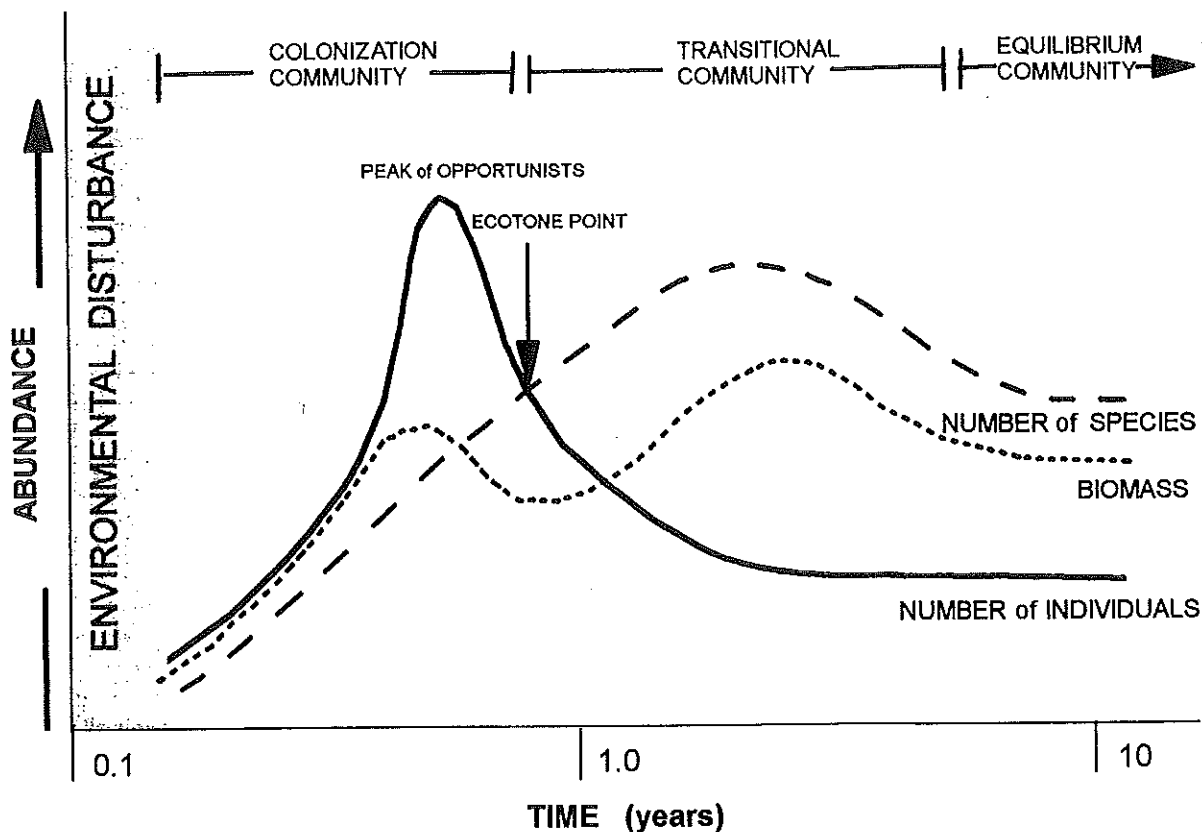
These general features of the structure of benthic communities apply not only to successional stages along a gradient of environmental variability, but also to the successive sequence of populations that recolonize deposits after the cessation of environmental disturbance. McCall (1976) and Rees & Dare (1993) have recognized the occurrence of three main types of benthic components of marine communities based on the distinction between *r*-selected opportunistic species and *K*-selected equilibrium species. Group I species comprise those that colonize first after a community has been removed by disturbance. They comprise large populations of small sedentary tube-dwelling deposit feeders that have rapid development, many generations per year, high settlement and death rates. Examples include the polychaete worms *Streblospio*, *Capitella capitata*, and *Owenia fusiformis* as well as the amphipod *Ampelisca*. That is, the Group I community comprises mainly *r*-strategist opportunistic species.

Group II species comprise mainly bivalve molluscs such as *Tellina*, *Nucula* and *Abra*, the tube worm *Lagis* (= *Pectinaria*) and the common starfish (*Asterias rubens*). There is no absolute distinction between this community and the primary colonizers, but the components attain a lower peak abundance than the smaller opportunistic species and have slower recruitment and growth rates. Finally Group III species comprise larger slow-growing *K*-strategist equilibrium species such as the polychaete *Nephtys*, the reef-forming Ross worm (*Sabellaria*), razor shell (*Ensis*), sea urchins such as *Echinocardium* and *Echinus*, scallops (*Pecten*), the ocean quahog (*Arctica islandica*), the edible crab (*Cancer pagurus*) and larger burrowing crustaceans such as *Nephrops* and *Callinassa*.

The changes in species variety, abundance of individuals and biomass during the recolonization process are shown in Fig. 4. Inspection of this figure shows that initially the sediments are almost devoid of benthic macrofauna.

The initial colonizing species are few, but the number of individuals (population density) increase rapidly with time to a peak of (Group I) opportunistic species. As time passes, the short-lived opportunistic species (*r*-strategists) decrease in numbers and biomass as more species invade the area. This transition point where the community is poor in population density and biomass is the same ecotone point shown on the spatial gradient in Figure 2.

Prior to this, the community is characterized by large populations of a few small opportunistic species; after this time the species variety increases, as does the biomass, but the population density declines. This Group II community is a transitional one where the maximum number of species has invaded the newly-available space, and is



**Figure 4** Schematic diagram showing a colonization succession in a marine sediment following cessation of environmental disturbance. Initial colonization is by opportunistic species which reach a peak population density generally within 6 months of a new habitat becoming available for colonization after the catastrophic mortality of the previous community. As the deposits are invaded by additional (larger) species, the population density of initial colonizers declines. This ecotone point marks the beginning of a transitional community with high species diversity of a wide range of mixed *r*-selected opportunistic and *K*-selected equilibrium species. This period may last for 1–5 yr depending on a number of environmental factors, including latitude. Provided environmental conditions remain stable, some members of this transition community are eliminated by competition and the community as a whole then forms final equilibrium community comprising larger, long-lived and slow growing species with complex biological interactions with one another. (Based on Pearson & Rosenberg 1978).

followed by a phase where some species are eliminated by competition and the community returns to the (somewhat lower) species composition and biomass characteristic of the undisturbed Group III community.

The sequence shown in Figure 4 indicates that colonization is likely to follow a definite time course of progressive invasion by large numbers of opportunistic species in the first instance, followed by a wider species diversity during the transitional phase and finally by a consolidation phase when competition between the *K*-strategist equilibrium species for the limited space available results in the elimination of some of the transitional colonizers (see also Warwick et al. 1987). The biological diversity in any particular community will then reflect the frequency of disturbance and represent a balance between invasion and subsequent growth of colonizers, and losses by extinction and displacement (see Huston 1994). In areas where environmental disturbance is unevenly distributed, this may lead to a mosaic of communities, each at different stages

of the successional sequence shown in Figure 4 (see Johnson 1970, Grassle & Sanders 1973, Whittaker & Levin 1977, Connell 1978), and may partly account for the patchiness of marine communities in dredged areas.

The time taken for recovery of the full species composition and for subsequent exclusion of some of the transition community following the growth of larger *K*-strategist equilibrium species in a particular area will depend largely on the components that occur under natural conditions. In shallow water and estuarine conditions, where the community is in any case dominated by opportunistic species, recovery to the original species composition may be very rapid and coincide with the Peak of Opportunists point in Figure 4. In the stable environmental conditions of deeper waters, the replacement of the initial colonizers in the transitional community following complex biological interactions between the *K*-selected equilibrium species may take several years.

## The physical impact of dredging

### *Impact within the dredged area*

The increased exploitation of marine deposits and the physical impact of dredging works has been widely reviewed (see Dickson & Lee 1972, Shelton & Rolfe 1972, Cruikshank & Hess 1975, Eden 1975, Millner et al. 1977, de Groot 1979, van der Veer et al. 1985, Glasby 1986, Lart 1991, Gajewski & Uscinowicz 1993, ICES 1993, Land et al. 1994, Whiteside et al. 1995, Hitchcock & Drucker 1996). Most of the sea-going aggregate dredgers are self-contained and use a centrifugal suction pump to lift the aggregates from the sea bed into a hopper where the material is screened before being transferred to the hold. The *in situ* reserves for economic exploitation normally range from 15–55% gravel. Unless the material is otherwise suitable for direct use as a beach feed or landfill (see Hess 1971), the sand : gravel ratio in the final cargo is adjusted to between 50 : 50 and 65 : 35 depending on customer requirements, local geology and ship performance (A. Hermiston pers.comm).

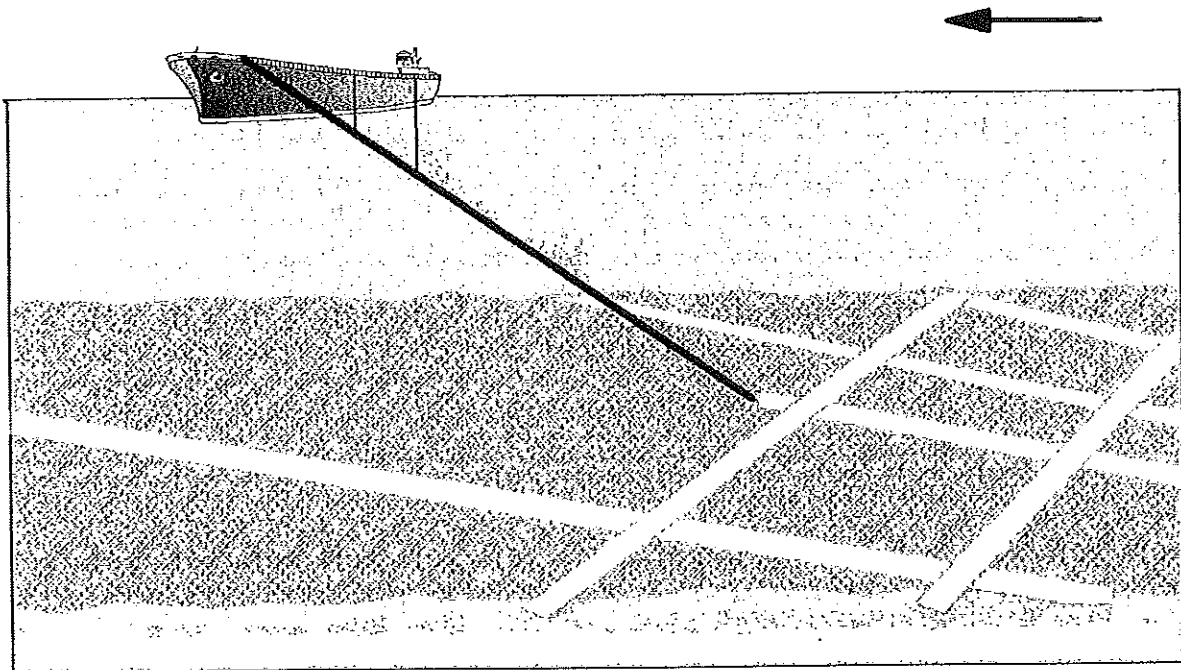
A proportion of the dredged deposits may therefore be returned to the sea bed through reject chutes when there is a larger proportion of fine material than is required for a commercially viable cargo. In most aggregate deposits the fines comprise only 1–2% of the total and are dominated by the silt fraction although significant quantities of sand may also be discharged in the immediate vicinity of the dredger to increase the gravel component of the cargo. Overspilling of water via spillways from the hopper will also contain some fine sands that are maintained in suspension by the turbulence within the hopper.

Essentially the physical impact of dredging works is dependent partly on the method of dredging, and partly on the amount and grade of deposits rejected by screening (if used) and overspill from the hopper. Two main methods of dredging are used for gravels extraction in European coastal waters. These are anchor hopper dredging and trailer suction hopper dredging and are illustrated in Fig. 5. In anchor dredging the vessel is stationary and dredges the deposits from a sequence of specific points on the sea floor and can therefore leave pits or depressions on the sea bed that may reach as much as 20 m in depth and 75 m in diameter (Cruikshank & Hess 1975, Dickson & Lee 1972).

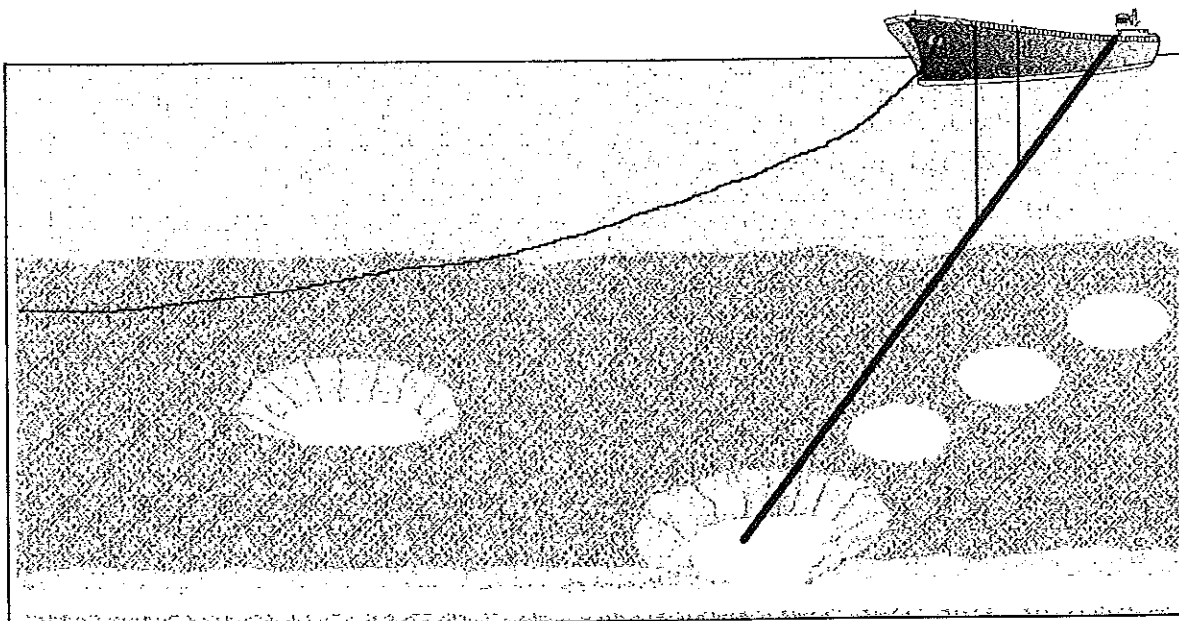
Because the deposits required for marine aggregates are coarse, and sediment disturbance by wave action is in any case limited mainly to depths of less than 30 m even



TRAILER DREDGING



ANCHOR DREDGING



**Figure 5** Diagram showing the two principal methods of dredging for marine aggregates in European coastal waters. Upper diagram shows the furrows left on the sea bed by trailer suction dredging while the vessel is under way. In this case the sea bed is crossed by a series of tracks which are 2–3 m wide and up to 50 cm deep. Lower diagram shows the pits left on the sea bed by anchor dredging. In this case the vessel is anchored and the dredged pits may reach as much as 75 m in diameter and 20 m in depth. (Based on Dickson & Lee 1972).

during storm conditions, it follows that not only is the fauna likely to be removed in patches from the dredged areas, but such pits are likely to be persistent features of the sea bed topography for several years except in areas where the sands are mobile (Eden 1975). Dickson & Lee (1972) studied the recovery of test pits dug by anchor dredge in gravel deposits in the Hastings Shingle Bank off the south east coast of England. They found that the pits were very slow to fill and were still visible after two years. In another study, van der Veer et al. (1985) described the recovery of pits in sandy substrata in the Dutch Wadden Sea. They showed that in this instance pits in channels with a high current velocity filled within one year, but those in the lower current velocities which occur in tidal watersheds took 5–10 yr to fill whereas those in tidal flat areas were still visible after 15 yr.

Such sediment movement as does occur is mainly through slumping of the sides of the furrows and subsequent infilling by fine particulates transported by tidal currents into the furrows that reduce current velocity and act as sediment traps. This can lead to heavily anoxic sediments and to colonization by a community which differs considerably from that in the original deposits (Dickson & Lee 1972, Shelton & Rolfe 1972, Kaplan et al. 1975, Bonsdorff 1983, Hily 1983, van der Veer et al. 1985, Hall 1994).

A second method of marine aggregate dredging is for the ship to extract deposits by suction through one or two pipes deployed while the vessel is slowly under way (Figure 5). In this case, side-scan sonar records show that the sea bed within the boundaries of licenced extraction areas in the southern North Sea is crossed by a series of dredge tracks that are 2–3 m wide and up to 50 cm deep (van Moorsel & Waardenberg 1990, Kenny & Rees 1994) although deeper troughs of up to 2 m were recorded from areas where the dredge head had crossed the area several times. Davies & Hitchcock (1992) reported dredge cuts of between 20–55 cm depth and 3–3.8 m width in commercially exploited deposits of the Bristol Channel. Somewhat deeper troughs of up to 70 cm were reported for the Baltic (Gajewski & Uscinowicz 1993). In this case removal of the surface 0.5 m of deposit would be sufficient to eliminate the benthos from the deposits in strips, the total removal depending on the intensity of dredging at a particular worked site.

Despite the shallower depth of removal, the evidence suggests that infilling of the troughs from trailer suction dredging takes at least 12 months in the Baltic and is achieved partly by slumping from the sides and partly by transport of fine material by bottom currents into the sediment traps formed by the dredged furrows (Kaplan et al. 1975, Hily 1983, van der Veer et al. 1985, Gajewski & Uscinowicz 1993). Progressive removal of the original sandy gravel and its replacement by fine sand has also been reported for the sediments off Dieppe by Desprez (1992). In the case of experimental furrows dredged by trailer suction in gravel deposits of the southern North Sea off the Suffolk coast of England, even shallow depressions of only 20–30 cm depth were still visible on side-scan sonar records made up to 4 yr later (Millner et al. 1977). In contrast, dredge furrows in the Bristol Channel have been reported to disappear within 2 to 3 tides because of high sediment mobility (pers. obs.).

Rather unexpectedly, Kenny & Rees (1994, 1996) found an increase in the particle size of deposits in the dredged areas, possibly reflecting the exposure of coarse deposits at depth below the surface gravel layers. In this study, which was carried out in the southern North Sea, the dredged furrows were visible with side-scan sonar even after 2 yr. Similar results have been reported for dredging tracks off the French coast at

Dieppe (Desprez 1992), although winter storms obliterated tracks within a few months on the Klaver Bank in the Dutch sector of the North Sea (Sips & Waardenberg 1989, van Moorsel & Waardenberg 1990, 1991). In general, dredge tracks will persist for varying times depending on the rate of local sediment fluxes, recent measurements suggesting 1–4 days only for the Norfolk Banks, but periods as long as 1–4 yr for more stable deposits off the Owers to the east of the Isle of Wight (A. Hermiston pers.comm.).

Thus both anchor dredging and trailer suction dredging have an important potential impact on the biology of the dredged areas, since no benthos is likely to occur below the dredged depth. This can be expected to lead to a patchy distribution of organisms, reflecting the differences between the dredged furrows and the intervening undredged surfaces. Such recolonization as occurs within the dredged areas is likely to be by migration of adults through transport on tidal currents (Rees et al. 1977, Hall 1994); by transport in sediments slumping from the sides of the pits and furrows (McCall 1976, Guillou & Hily 1983); by the return of some undamaged components through outwash from the chutes and spillways (see Lees et al. 1992, Ministry of Agriculture, Fisheries and Food 1993); and by colonization and subsequent growth of larvae from neighbouring populations. In this case, a clear succession of colonizing species is to be anticipated, leading to the establishment of definite clusters or patches in benthic community composition, depending on the type of deposits that have infilled the dredged areas and the time since the recolonization sequence started.

### *Impact adjacent to the dredged area*

Although a good deal of concern has been expressed about the possible impact of marine aggregate extraction on coastal resources (see ICES 1992a,b), the possible scale of impact outside the immediate dredged area from the settlement on the sea bed of fine material temporarily suspended by marine aggregates dredging is poorly understood.

Hitchcock & Drucker (1996) have summarized values for material lost through the hopper overflow spillways and from the reject chutes during the screening process on a typical modern trailer suction dredger of 4500 t hopper capacity operating in UK waters off the coast of East Anglia. Table 2 shows the size distribution for the material lost through the reject chute and spillway, while the total screened load quantities are

**Table 2** Size distribution of overspill and reject material from a typical modern trailer suction dredger of 4500 t hopper capacity (based on Hitchcock & Drucker 1996).

Particle size (mm)	Proportion in discharge (%)	
	Spillway	Reject chute
<0.063	38.0	1.0
0.063–0.125	14.0	0.9
0.125–0.250	5.7	8.9
0.250–0.500	12.9	31.4
0.500–1.000	9.2	27.3
1.000–2.000	3.3	12.0
>2.000	16.9	18.5

summarized in Table 3. These show that during a recorded average loading time of 290 min, 12 158 t of dry solids and 33 356 t of water were pumped by the dredge pump. The data show that 4185 t of dry solids are retained as cargo, while 7223 t of dry solids are returned overboard because of rejection by screening, and a further 750 t from overspill.

It is also clear that some 1338 t of material  $>2.0$  mm (representing 18.5% of the 7235 t in the reject chute – see Table 2) is lost overboard through the reject chute and a further 126 t (representing 16.9% of the 750 t in the spillway – see Table 2) from the spillway. This equates to a loss rate of  $76.9 \text{ kgs}^{-1}$  of particles  $>2.0$  mm from the screening reject chute and  $7.2 \text{ kgs}^{-1}$  from overspill. Assuming that the dredger moves at an average speed of 1 knots, the flux of material  $>2.0$  mm entering the water column is  $39.6 \text{ kgs}^{-1} \text{ m}^{-1}$  from the screening reject chute and  $3.7 \text{ kgs}^{-1} \text{ m}^{-1}$  from the spillways. Much of this material is in the size range 2.0–10.0 mm and falls rapidly to the sea bed with little horizontal displacement during screening. Video recordings during normal loading operations show that such material deposits on the sea bed directly under the dredge vessel (Davies & Hitchcock, 1992).

Finer sand and silt fractions discharged during dredging and screening amount to 5824 t from the reject screening chutes and 338 t from overspill. This is equivalent to a deposition rate of  $334 \text{ kgs}^{-1}$  and  $19.4 \text{ kgs}^{-1}$ , respectively. In addition to the sand fraction, up to 213 t ( $12.2 \text{ kgs}^{-1}$ ) of muddy sediment ( $<0.063$  mm diameter) may be lost through the rejection process and 285 t ( $16.4 \text{ kgs}^{-1}$ ) from overspill. The material may be expected to settle more slowly than the sand-sized fraction and has a typical settling velocity of  $0.1\text{--}1.0 \text{ mm s}^{-1}$ .

In its simplest form, the settlement velocity and residence time of such particles in the water column can be estimated from Stoke's Law. If the residence time of particles in the water column is known, the duration and speed of currents will then determine the excursion pattern before settlement. Estimates of dispersion of fine material based on these Gaussian diffusion principles suggest that very fine sand particles may travel up to 11 km from the dredge site, fine sand up to 5 km, medium sand up to 1 km and coarse sand less than 50 m (H. R. Wallingford 1994, cited in Hitchcock & Drucker 1996).

Similar estimates based on the settlement velocity of fine silt-sized particles ( $<0.063$  mm diameter) suggest that this material could remain in suspension for up to 4–5 tidal cycles and be carried for as much as 20 km on each side of a point source of discharge. Most recent studies made on the dispersion of sediment plumes generated from dredging operations suggest, however, that the area of impact of outwash from dredging activities is smaller than estimates based on Gaussian diffusion models, especially where the proportion of silt and clay in the deposits is low. This appears to be due to complex cohesion properties of the discharged sediment particles that settle to

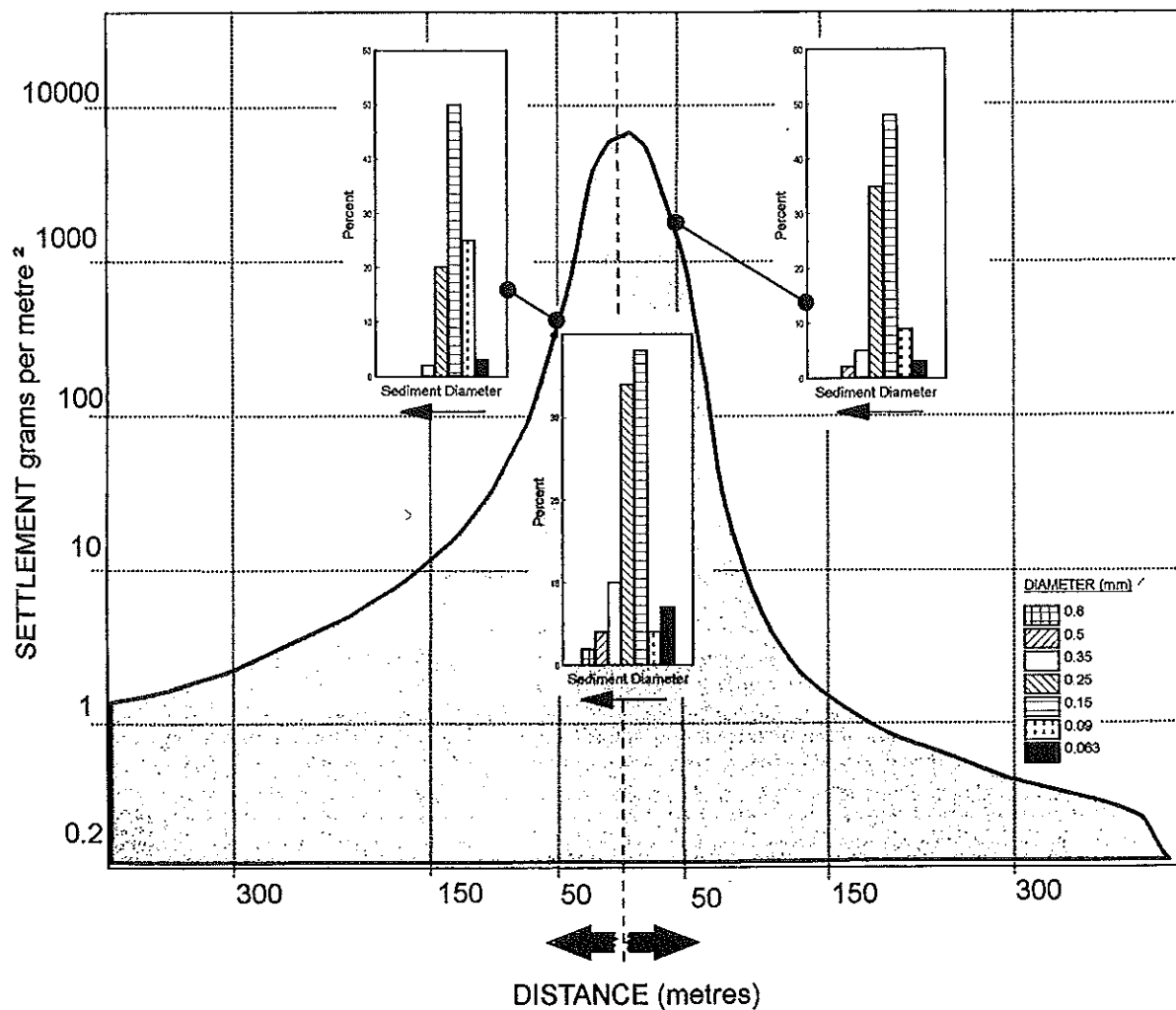
Table 3 Typical screened load quantities for a suction dredger of 4500 t hopper capacity (based on Hitchcock & Drucker 1996).

	Dry solids (tonnes)	Water (tonnes $\pm 5\%$ )
Quantity pumped	12 158	33 356
Quantity retained	4185	874
Quantity reject through screening	7223	13 499
Quantity lost through overspill	750	21 387
Total losses	7973	34 886

the sea bed as a density current and do not conform to settlement rates based on the specific gravity and size of the component particles themselves.

A detailed study by Gajewski & Uscinowicz (1993) in relation to trailer dredging in the Baltic showed that the width of the plume, as determined by the light extinction in the water, did not exceed 50 m. Settlement of the suspended matter onto the sea floor was measured by a series of sediment traps deployed approximately 1 m above the sea bed. When the dredger was discharging mainly fine sand (0.25–0.125 mm) at a concentration of approximately  $11000 \text{ mg l}^{-1}$ , it was found that deposition of  $7500\text{--}15000 \text{ g m}^{-2}$  were recorded in the troughs during dredging. The settlement in traps deployed 50 m away from the dredger's route was, however, less than  $1220 \text{ g m}^{-2}$ . At distances greater than 50 m the amount of material settling on the sea floor decreased rapidly (Fig. 6).

More recently, Acoustic Doppler Current Profiling (ADCP) techniques have been used to determine plume dispersion in relation to spoils dispersal from both commercial aggregate dredgers (Hitchcock & Dearnaley 1995, Hitchcock & Drucker 1996)



**Figure 6** Diagram showing the settlement of sand during dredging operations from trailer dredging in the Baltic. Particle size profiles for the sediments deposited in the track of the dredger and 50 m on each side of the dredger are also shown. Note that the main deposition of sand from flow-off from dredging operations was confined to distances within 150 m on each side of the track of the dredger. (After Gajewski & Uscinowicz 1993).

and in relation to capital dredging works and sand mining (Land et al. 1994, Whiteside et al. 1995). Remote airborne and satellite imagery has also proved to be a useful tool in defining the contours of sediment dispersal (Whiteside et al. 1995).

These studies confirm that the initial sedimentation of material discharged during outwash from dredgers does not, as had been widely assumed, disperse according to the Gaussian diffusion principles used in most simulation models, but behaves more like a density current where particles are held together by cohesion during the initial phase of the sedimentation process. As a result, the principal area likely to be affected by sediment deposition is much less than the "worst case" scenarios predicted from conventional Gaussian diffusion simulation models, and is mainly confined to a zone of a few hundred metres from the discharge chutes.

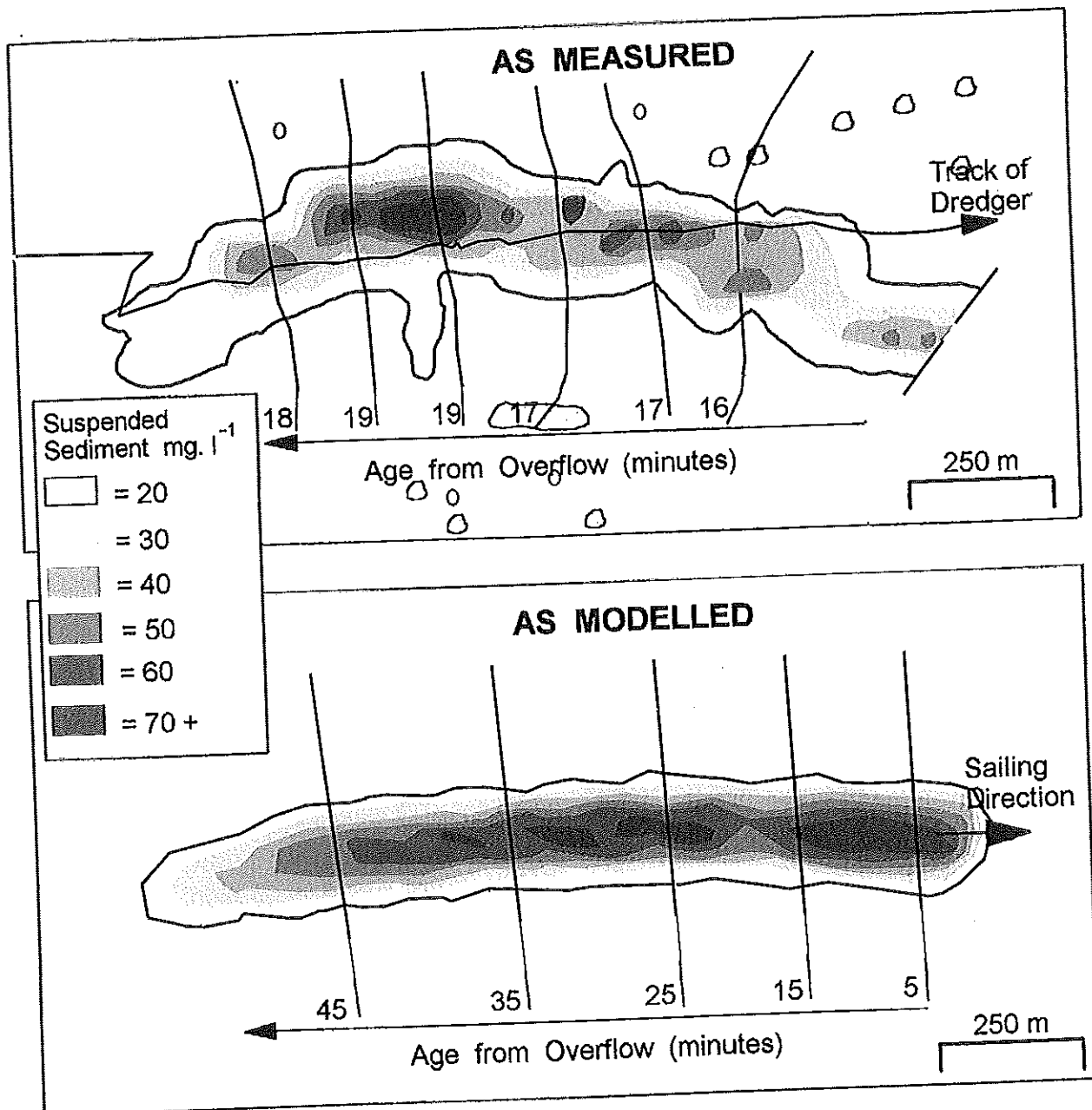
A recent study by Whiteside et al. (1995; see also Johanson & Boehmer 1975, Gayman 1978) has shown that the behaviour of plumes discharged during sand dredging can best be regarded as comprising an initial "dynamic phase" during which the sediment-water mixture descends rapidly to the sea bed as a density current jet at a rate that depends on the overflow density, the diameter of the discharge pipe, the water depth, the velocity of discharge and the speed of the dredger. During its passage through the water column and following impact with the sea bed the sediment is dispersed into the water and forms a well defined plume astern of the dredger. This second longer phase has been referred to as the "passive phase" of dispersion by Whiteside et al. (1995) and starts approximately 10 min after outflow. During this phase the material behaves in a relatively simple settling mode according to Stokes' Law, the plume then decaying to background levels after a period of 2-3 h.

Their study showed that approximately 100 m (corresponding to approximately 3 min from the overflow) astern of a dredger working in Hong Kong waters the plume surface sediment concentrations were from 75-150 mg l<sup>-1</sup>. Levels were halved in 10 min and reduced to 20-30 mg l<sup>-1</sup> after 30 min. This approached the recorded background suspended solids concentration of 10-15 mg l<sup>-1</sup> and indicated that only a relatively small proportion of the fines category (<63 µm) remained in the water column at the start of the passive phase of dispersion 10 min after discharge. Even then, their data suggest that the settlement rate of the plume continued to be more rapid than simple particle settlement would suggest.

A plume dispersion model developed by Whiteside et al. (1995) for the surface layer (the upper 8 m of the water column) for up to 40 min after discharge is shown in Fig. 7 and compares well with summed plume decay measurements in the vicinity of the dredger. The contours for sediment deposition evidently remain as a narrow band extending for approximately 100 m on each side of the track of the dredging vessel, much as recorded by Gajewski & Uscinowicz (1993) for Baltic waters.

Very similar rapid rates of deposition and decay of sediment outwash plumes have been recorded by Hitchcock & Drucker (1996), who studied plume generation and decay from four dredge vessels ranging in capacity from 2000-5000 t during normal loading operations off the coast of East Anglia, UK. During the plume tracking exercise peak current velocities reached 0.6 m s<sup>-1</sup> and the water depth was approximately 22 m. The concentration of total suspended sediment discharged was approximately 2500 mg l<sup>-1</sup> comprising mainly sand-sized material and with <30 mg l<sup>-1</sup> mud (<0.063 mm diameter).

The total concentration of suspended solids in the water column at different depths and distances from the dredger measured by water sampling and optical transmis-



**Figure 7** Diagram showing contours for suspended sediment concentrations astern of a trailer dredger operating in Hong Kong waters. Upper diagram shows the contours as measured in the upper 8 m of water across the plume at various arbitrary time intervals during field studies of a sediment plume from 16–19 min after discharge. Lower diagram shows the output of a simulation model developed for sediment dispersion based on rapid initial sedimentation during a dynamic phase and a second longer passive phase which starts approximately 10 min after outflow. (After Whiteside et al. 1995).

someters is summarized in Fig. 8. The corresponding values for silt-sized material ( $<0.063 \text{ mm}$ ) are shown in Fig. 9. These data show that concentrations of sand-sized material are reduced to background levels over a distance of only 200–500 m from the point of release into the water column and that the concentration of even silt-sized particles is reduced to background values of  $2\text{--}5 \text{ mg l}^{-1}$  over a similar distance. This very rapid reduction in suspended sediment concentrations is similar to that reported by Whiteside et al. (1995).

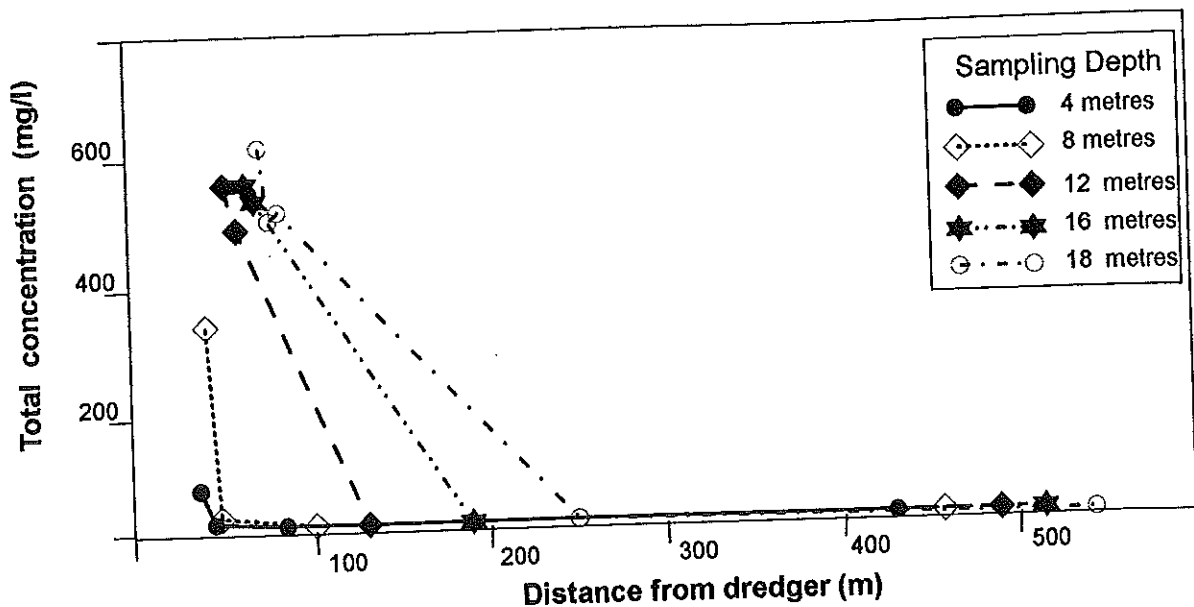


Figure 8 The total concentration of suspended solids in the water column at different depths and distances from the dredger measured by water sampling and optical transmissometers. (After Hitchcock & Drucker 1996).

Although suspended sediment concentrations in the plume were not significantly different from background levels beyond 400–500 m from the point source of discharge using conventional water sampling and optical transmissometer techniques, it is interesting to note that it is possible to track the plume using ADCP techniques over a distance of up to 3.5 km. A series of typical sections across a discharge plume at

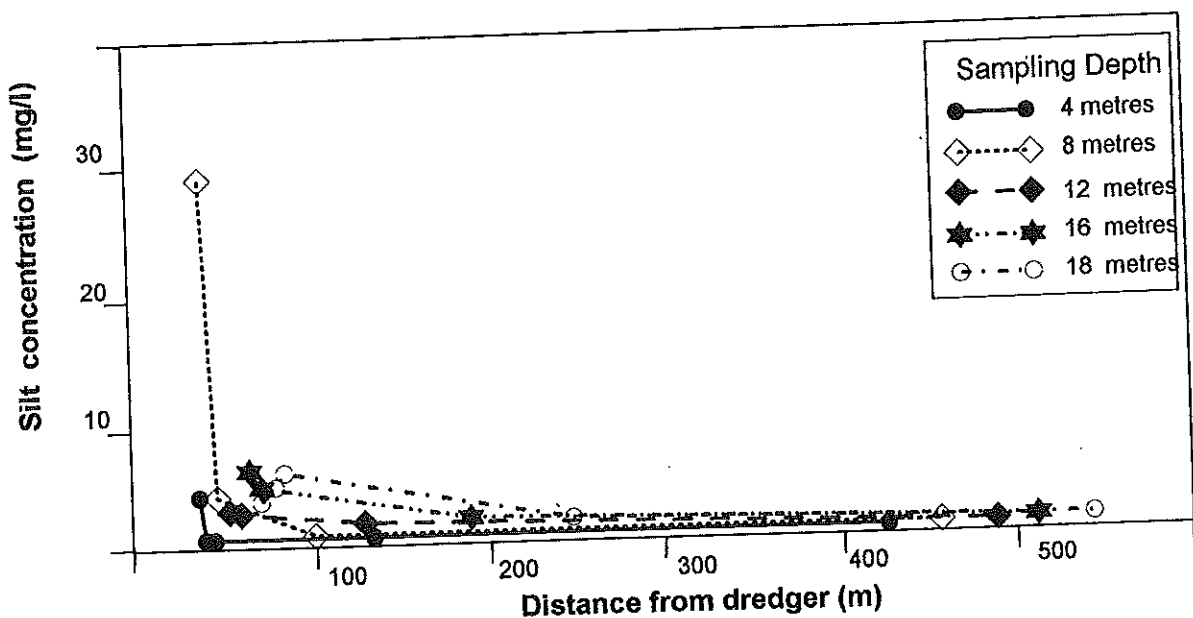
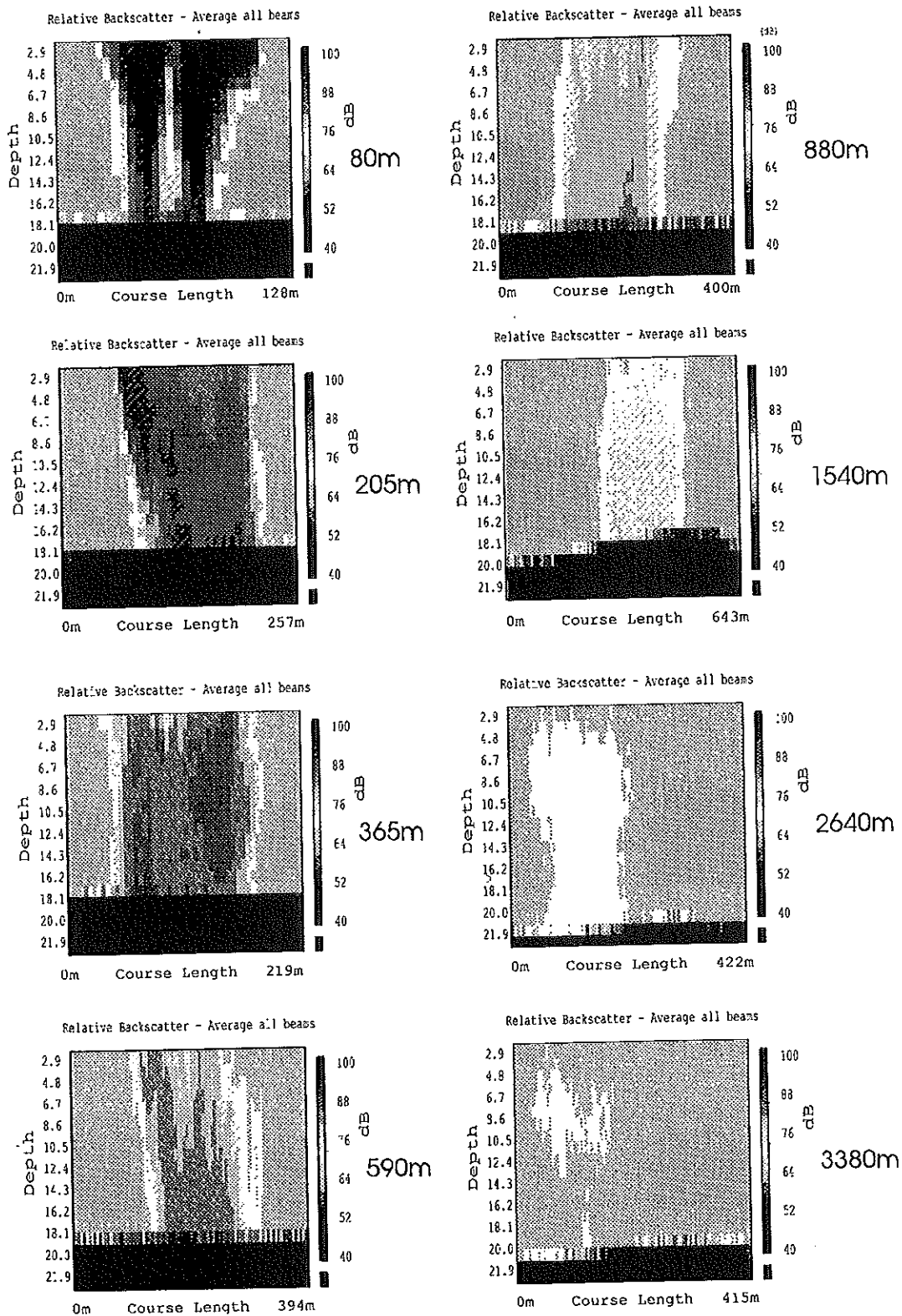


Figure 9 The concentration of silt-sized material (<math><0.063\text{ mm}</math>) in the water column at different depths and distances from the dredger measured by water sampling and optical transmissometers. (After Hitchcock & Drucker 1996).



# IMPACT OF DREDGING WORKS IN COASTAL WATERS



**Figure 10** Acoustic Doppler Current Profiler (ADCP) acoustic backscatter across the plume of a dredger at varying distances downstream (m). Water speed  $100 \text{ cm s}^{-1}$ . Note that although sedimentation is achieved within 880 m, the relative backscatter suggests an additional “plume”, perhaps representing entrained air bubbles, biochemical precipitates or organic matter released from the sediment, which extends up to 3380m astern of the dredger. (After Hitchcock & Drucker 1996).

varying distances from 80m up to 3335m away from a dredger is shown in Fig. 10. This shows clearly the decay of the plume to background levels at approximately 400m, but that a residual impact is detectable up to 3.5km by ADCP methods. Whether this residual impact is caused by air bubbles or organic matter becoming entrained into the water column during the dredging operation is at present unknown. However, it is noteworthy that there is now a good deal of evidence that suggests that disturbance of marine sediments may release sufficient organic matter into the water column to enhance benthic production.

These results for the dispersion of sediment in the water column thus suggest that sedimentation is rapid and is confined to the immediate vicinity of the discharge. They confirm earlier studies of Poiner & Kennedy (1984; see also Willoughby & Foster 1983) who reported that sediment deposition generated from dredging activities in Moreton Bay, Queensland was confined to the immediate vicinity of the dredging works. Concentrations of suspended sand-sized material were reported to decay to background levels over a distance of only 200–500m from the point of release into the water column from a commercial aggregate dredger. They estimated that the sediment deposition 500m outside the boundary of the dredged area was  $29.6 \text{ kg m}^{-2}$  ( $23 \text{ mm m}^{-2}$ ). At 1km deposition was  $21.2 \text{ kg m}^{-2}$  ( $16 \text{ mm m}^{-2}$ ), at 1.5 km it was  $15 \text{ kg m}^{-2}$  ( $12 \text{ mm m}^{-2}$ ), at 2 km it was  $10.7 \text{ kg m}^{-2}$  ( $8 \text{ mm m}^{-2}$ ) and finally at 2.5 km from the boundaries of the dredged area the estimated deposition was less than  $7.6 \text{ kg m}^{-2}$  ( $6 \text{ mm m}^{-2}$ ).

There is a good deal of evidence from other surveys that disturbance of sediments by dredging may release sufficient organic materials to enhance the species diversity and population density of organisms outside the immediate zone of deposition of particulate matter. Disturbance of the sediments may thus enhance benthic production outside the immediate zone of deposition provided that contaminants from polluted sediments are not associated with the disposal of spoils. Stephenson et al. (1978) and Jones & Candy (1981) both document the enhanced diversity and abundance of benthic faunas near to dredged channels. Poiner & Kennedy (1984) showed that there was an enhancement of benthic biota close to dredged areas at Moreton Bay, Queensland and that the level of enhancement decreased with increasing distance from the dredged area up to a distance of approximately 2 km. They ascribe this to the release of organic nutrients from the sediment plume, a process which is well known from other studies (Ingle 1952, Biggs 1968, Sherk 1972, Oviatt et al. 1982; Walker & O'Donnell 1981).

The results reviewed above thus suggest that the impact of dredging activities mainly relates to the physical removal of substratum and associated organisms from the sea bed along the path of the dredge head and to the impact of subsequent deposition of sediment from outwash during the dredging process. The evidence from direct studies on the sedimentation of particulate matter suggests that the impact of sedimentation on biological resources on the sea bed is likely to be confined to distances within a few hundred metres of the dredger where the deposits are sands and gravels. It should be remembered, however, that discharge of dredge spoils from maintenance and capital dredging works in estuaries may result in much larger dispersion plumes that reflect the dominantly fine particles and strong current flows that occur in estuaries, and that the same processes, which result in the release of dissolved organic matter, can also result in the release of bound surface contaminants from the sediments into the water column.

## The impact of dredging on biological resources

### *Sensitivity to Disturbance*

The impact of disturbance by the dredge head during marine aggregate dredging has been reviewed on pp. 140–43. The effects of sediment deposition and spoils disposal outside the immediate boundaries of dredged areas in coastal waters has also been widely studied and includes extensive physiological–ecological work on a wide variety of animals including plankton, benthic invertebrates and fish species (for reviews, see Sherk 1971, Moore 1977). Early studies by Loosanoff (1962) showed that different species of commercially significant filter-feeding molluscs were differently affected by suspended sediment. Subsequent studies by Sherk (1971) and Sherk et al. (1974) included both plankton and fish species. They showed that, as in the case of bivalves, fish species have varying tolerances of suspended solids, filter-feeding species being more sensitive than deposit feeders and larval forms being more sensitive than adults (see also Matsumoto 1984).

Many of the macrofauna that live in areas of sediment disturbance are well adapted to burrow back to the surface following burial (see Schafer 1972). Studies by Maurer et al. (1979) showed that some benthic animals could migrate vertically through more than 30 cm of deposited sediment, and this ability may be widespread even in relatively deep waters. Kukert (1991) showed, for example, that approximately 50% of the macrofauna on the bathyal sea floor of the Santa Catalina Basin were able to burrow back to the surface through 4–10 cm of rapidly deposited sediment. A good deal of the apparent recolonization of deposits following dredging or spoils disposal may therefore reside in the capacity of adults to migrate up through relatively thin layers of deposited sediments (see also Ellis & Heim 1985), or to migrate in during periods of storm-induced disturbance (see Hall 1994).

There is good evidence that the activities of filter-feeding bivalves, in particular, can play an important part in controlling the natural phytoplankton and seston loads in the water column (Cloern 1982), to an extent that food may become a limiting resource in the benthic boundary layer at the sediment–water interface (Wildish & Kristmanson 1984, Fréchette et al. 1989, 1993; see also Dame 1993, Snelgrove & Butman 1994) as well as on coral reef flats and in cryptic reef habitats (Glynn 1973, Buss & Jackson 1981). Because the suspension-feeding component is evidently highly effective in removing particulate matter from sea water, the release of large quantities of suspended matter can lead to a loss of suspension-feeding components through clogging of the gills. This has led to a corresponding increase in the community of deposit feeders in some areas such as St Austell Bay off the southwest coast of England (Howell & Shelton 1970).

In general, however, most recent studies of filter feeders that live in coastal waters show that bivalves, in particular, are highly adaptable in their response to increased turbidity such as can be induced by periodic storms, dredging or spoils disposal and can maintain their feeding activity over a wide range of phytoplankton concentrations and inorganic particulate loads (Shumway et al. 1985, 1990, Newell et al. 1989, Newell & Shumway 1993, Iglesias et al. 1996, Navarro et al. 1996, Urrutia et al. 1996).

Although these studies on the physiology of individual species can give some insight into the differing susceptibilities of the macrofauna to increased turbidity, or to burial from dredger outwash, in general it is difficult to make predictions of the impact of

dredging on whole communities from the results of studies on individual species. Partly for this reason, and because the interactions between the components of natural populations are complicated in space and time, most recent studies on the impact of spoils disposal and dredging works have been carried out on whole communities, rather than individual species. Such studies have concentrated on three main features of benthic communities, namely the number of individuals (population density), number of species (the diversity) and the biomass (to give an index of the growth following recolonization).

Sampling is conventionally carried out by means of a grab that allows collection of a sediment sample from a known area of seabed deposits, which are then eluted through a 1-mm mesh sieve to extract the macrofauna. Sediment samples from fine deposits such as occur in coastal embayments, lagoons and estuaries are relatively easy to obtain by means of equipment such as the van Veen and Smith-McIntyre grabs, the Ponar grab (Ellis & Jones 1980), or the Day grab whose jaws are held closed by the tension of the wire from which the grab is suspended rather than by a spring-loaded mechanism (see Holme & McIntyre 1984). Sampling of coarser gravel deposits is complicated, however, by the fact that the larger stones become trapped between the jaws of conventional grabs, leading to extensive losses through "washout" from the grab. Partly because of this problem, most work on coarser deposits has been carried out with semi-quantitative dredges such as the Anchor dredge (Forster 1953, Holme 1966, Kenny et al. 1991) or the Ralier du Baty dredge used by Davoult et al. (1988).

More recently, however, Sips & Waardenburg (1989) and Kenny & Rees (1994) have used a Hamon grab for quantitative studies on the fauna of gravels and sands. This grab takes a scoop out of the seabed deposits, rather than relying on the closure of opposing jaws (see Holme & McIntyre 1984). This greatly reduces the problem of fauna losses through "washout" during the sampling process and the Hamon grab is now widely used in the quantitative evaluation of the benthos in coarse sands and gravels.

Such studies emphasize that the macrofauna may vary considerably even over relatively short distances, and that a proper understanding of the distribution of benthic communities is necessary if damage to potentially important communities is to be avoided during dredging operations. Figure 11 shows, for example, the distribution of two important members of the benthic community in mixed gravel, sand and muddy deposits off the coast of East Anglia in August 1996 (Newell & Seiderer 1997b). Inspection of this figure shows that the main population of the reef-building tubeworm, *Sabellaria spinulosa* (called Ross by the fishermen) occurs in the northwestern part of the survey area, and corresponds with a localized patch of coarse stones and cobbles that give sufficient stability to support a rich reef community. This species may be predated upon by the pink shrimp (*Pandalus*) and is potentially important as a feeding ground for a variety of demersal fish species (see Warren, 1973).

In contrast, the populations of the comb worm (*Lagis* = *Pectinaria koreni*) occur in mobile muddy sands in the southwest of the survey area. This species is an important prey item for sole (*Solea solea*), dab (*Limanda limanda*) and plaice (*Pleuronectes platessa*) (see Lockwood 1980, Basimi & Grove 1985, Carter et al. 1991, Horwood 1993, also Peer 1970) and therefore represents a food resource within the survey area that requires conservation.

IMPACT OF DREDGING WORKS IN COASTAL WATERS

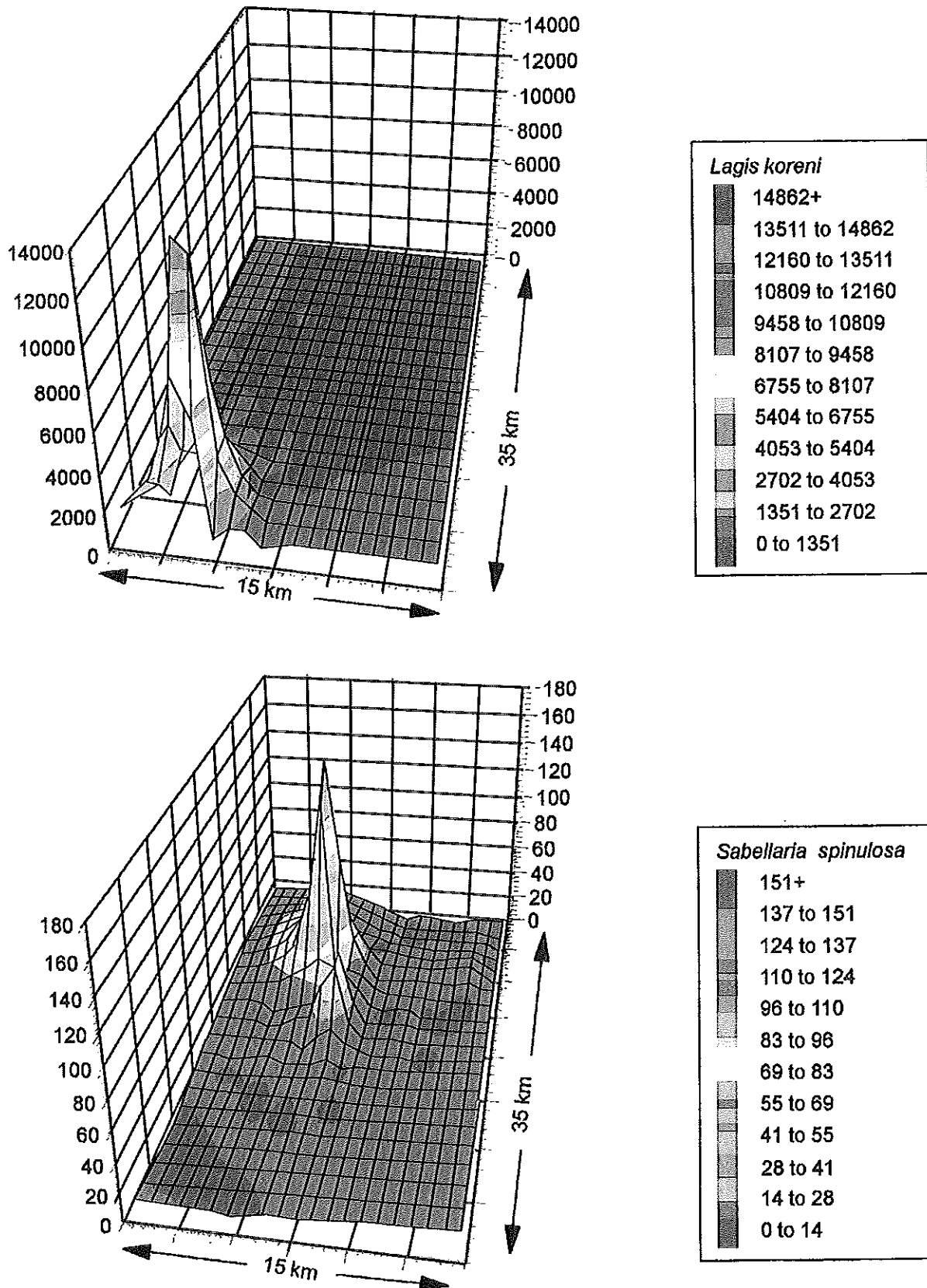


Figure 11 Schematic diagram of a survey area in the southern North Sea off Suffolk showing the distribution of the comb worm (*Lagis koreni*) in fine deposits of the survey area, and that of the colonial "Ross" worm (*Sabellaria spinulosa*) in areas where coarse boulders provide a stable environment for development of reef-forming species. Population density in numbers of individuals per 0.25 m<sup>2</sup> Hamon grab sample. (Newell & Seiderer 1997b).

*Impact of dredging on diversity and abundance*

The impact of dredging on benthic communities varies widely, depending, among other factors, on the intensity of dredging in a particular area, the degree of sediment disturbance and recolonization by passive transport of adult organisms (see Hall 1994) and the intrinsic rate of reproduction, recolonization and growth of the community that normally inhabits the particular deposits.

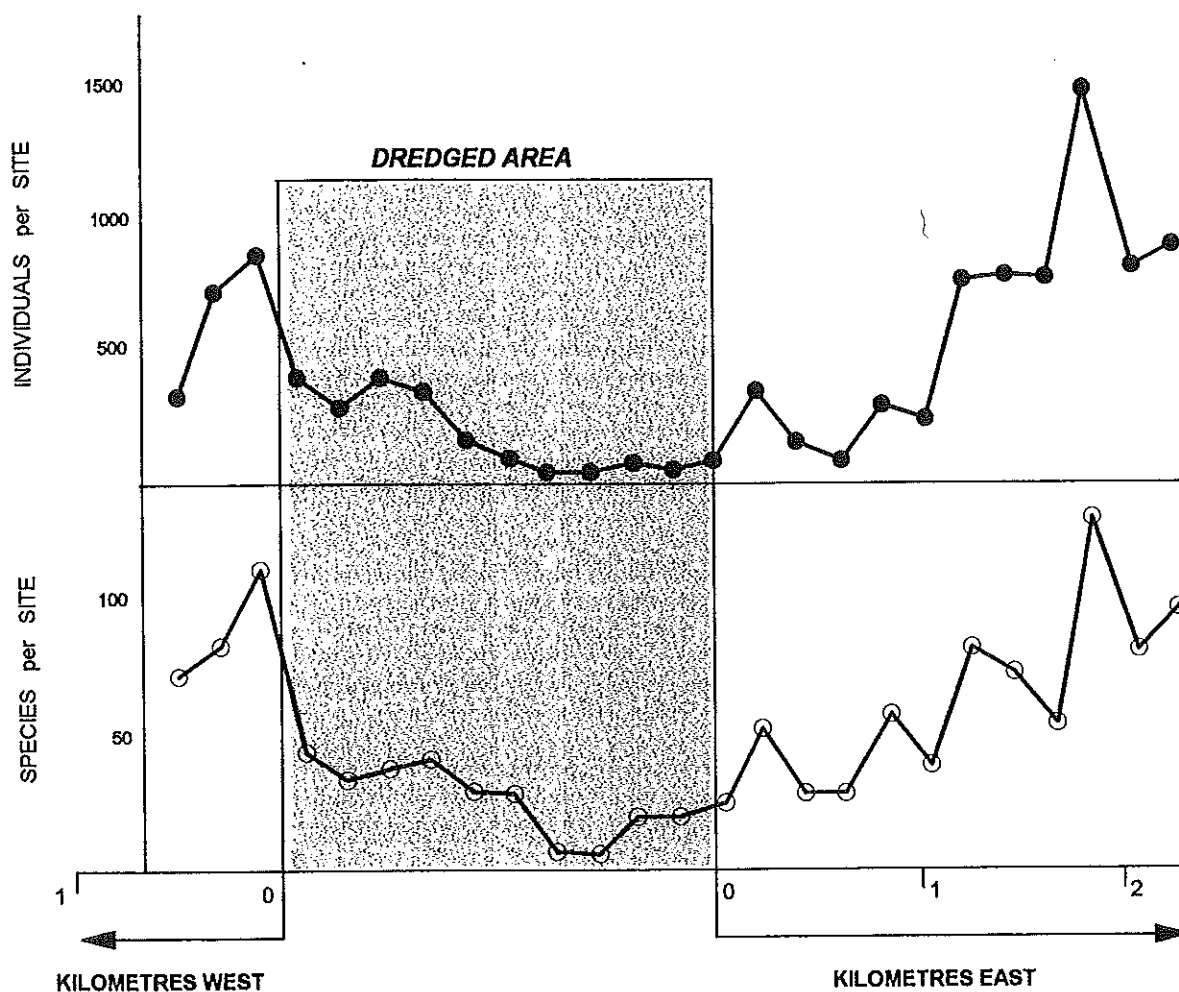
Some examples of the impact of dredging on the species variety, population density (number of individuals) and biomass of benthic organisms from a variety of habitats ranging from muds in coastal embayments and lagoons, to oyster shell deposits, and to sands and gravel deposits in the southern North Sea are summarized in Table 4. This shows that both maintenance dredging and marine aggregates dredging can be expected to result in a 30–70% reduction of species diversity, a 40–95% reduction in the number of individuals, and in a similar reduction in the biomass of benthic communities in the dredged area.

Despite the major impact of dredging on benthic community composition within dredged areas, there is little evidence that deposition of sediments from outwash through the chuteways during the dredging process has a significant impact on the benthos outside the immediate dredged area. Poiner & Kennedy (1984) showed that the population density and species composition of benthic invertebrates adjacent to dredging works on sandbanks in Moreton Bay, Queensland, Australia increased rapidly outside the boundaries of the dredged area, as might be anticipated from the relatively small amounts of sediment that are deposited beyond a few hundred metres of the dredger trail (see Figs 6 and 7 and p. 150). The population density and species diversity recorded from a transect across a dredged area in Moreton Bay in July 1982 by Poiner & Kennedy (1984) is shown in Fig. 12.

**Table 4** Table showing the impact of dredging on benthic community composition from various habitats.

Locality	Habitat type	% Reduction after dredging			Source
		Species	Individuals	Biomass	
Chesapeake Bay	Coastal embayment muds-sands	70	71	65	Pfitzenmeyer 1970
Goose Creek, Long Island, NY	Shallow lagoon mud	26	79	63–79	Kaplan et al. 1975
Tampa Bay, Florida	Oyster shell	40	65	90	Conner & Simon 1979
Moreton Bay, Queensland, Australia	Sand	51	46	–	Poiner & Kennedy 1984
Dieppe, France	Sands-gravels	50–70	70–80	80–90	Desprez 1992
Klaver Bank, Dutch Sector, North Sea	Sands-gravels	30	72	80	van Moorsel 1994
Lowestoft, Norfolk, UK	Gravels	62	94	90	Kenny & Rees 1994
Hong Kong	Sands	60	60	–	Morton 1996
Lowestoft, Norfolk, UK	Sands-gravels	34	77	92	Newell & Seiderer 1997a

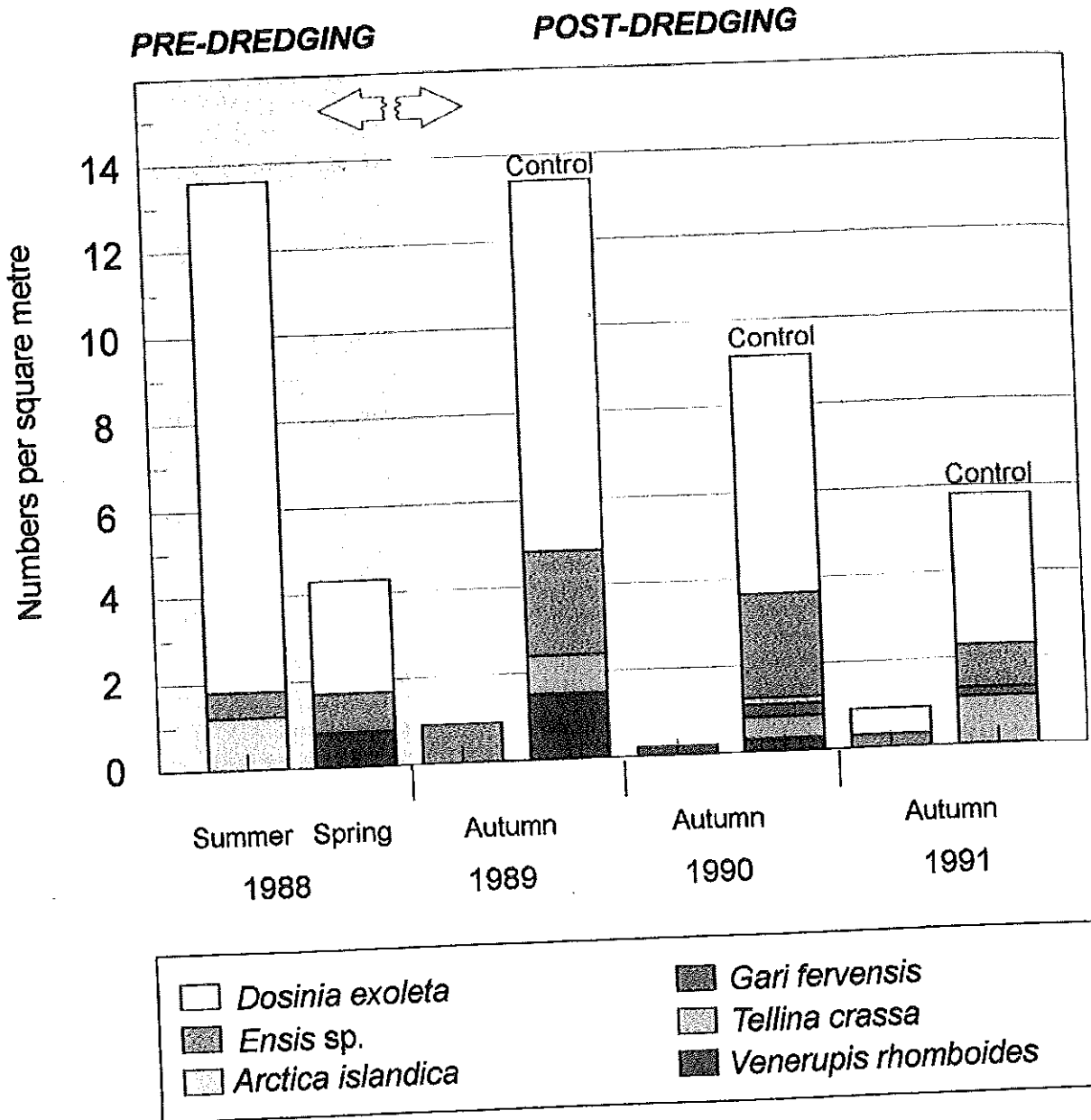
## IMPACT OF DREDGING WORKS IN COASTAL WATERS



**Figure 12** Diagram showing the number of individuals and species of benthos recorded in July 1982 on a transect crossing a dredged area on a sublittoral sandbank in Moreton Bay, Queensland, Australia. Note that species variety and population density increased rapidly outside the immediate boundaries of the dredged area. (Based on Poiner & Kennedy 1982).

Other than this study, there is surprisingly little detailed information on the precise boundaries of biological impact or "footprint" surrounding areas that have been dredged for sands and gravels. The circumstantial evidence from the boundaries of sediment deposition suggest, however, that biological impact is likely to be confined to the immediate vicinity of the dredged area.

One of the problems with assessing the impact of dredging works and the recovery of benthic communities over time is that biological communities are often subject to major changes in population density and community composition, even in areas that are apparently unaffected by dredging. Variations in the population density and species composition of the large bivalve population recorded between 1988 and 1991 in the sand and gravel deposits of the Klaver Bank in the Dutch sector of the North Sea by van Moorsel (1994) are shown in Fig. 13. This shows the major change in population of the bivalve, *Dosinia exoleta* between the summer of 1988 and the spring of 1989 and the loss of the large bivalve, *Arctica islandica* from the deposits even before aggregate extraction had taken place.



**Figure 13** Diagram summarizing the changes in population density and species composition of large bivalves on the Klaver Bank in the southern North Sea between 1988 and 1991. Pre-dredging values in 1988 show major seasonal changes in density and species composition. After dredging in the summer of 1989, large differences in population density and species composition emerged between dredged and control areas, and these differences persisted for at least two years. (After van Moorsel 1994).

A short period of aggregate extraction was carried out in the study area on the Klaver Bank in the summer of 1989. Thereafter, clear differences emerged between the large populations of bivalves in control areas outside the dredged zone and those within the dredged area, despite the natural variations in species composition and population density that evidently occurred in the deposits of the survey area. These differences persisted until the end of the survey period in autumn of 1991, suggesting that this slow-growing component of the benthos remains impacted for at least 2 yr after cessation of dredging.



## The process of recolonization and recovery

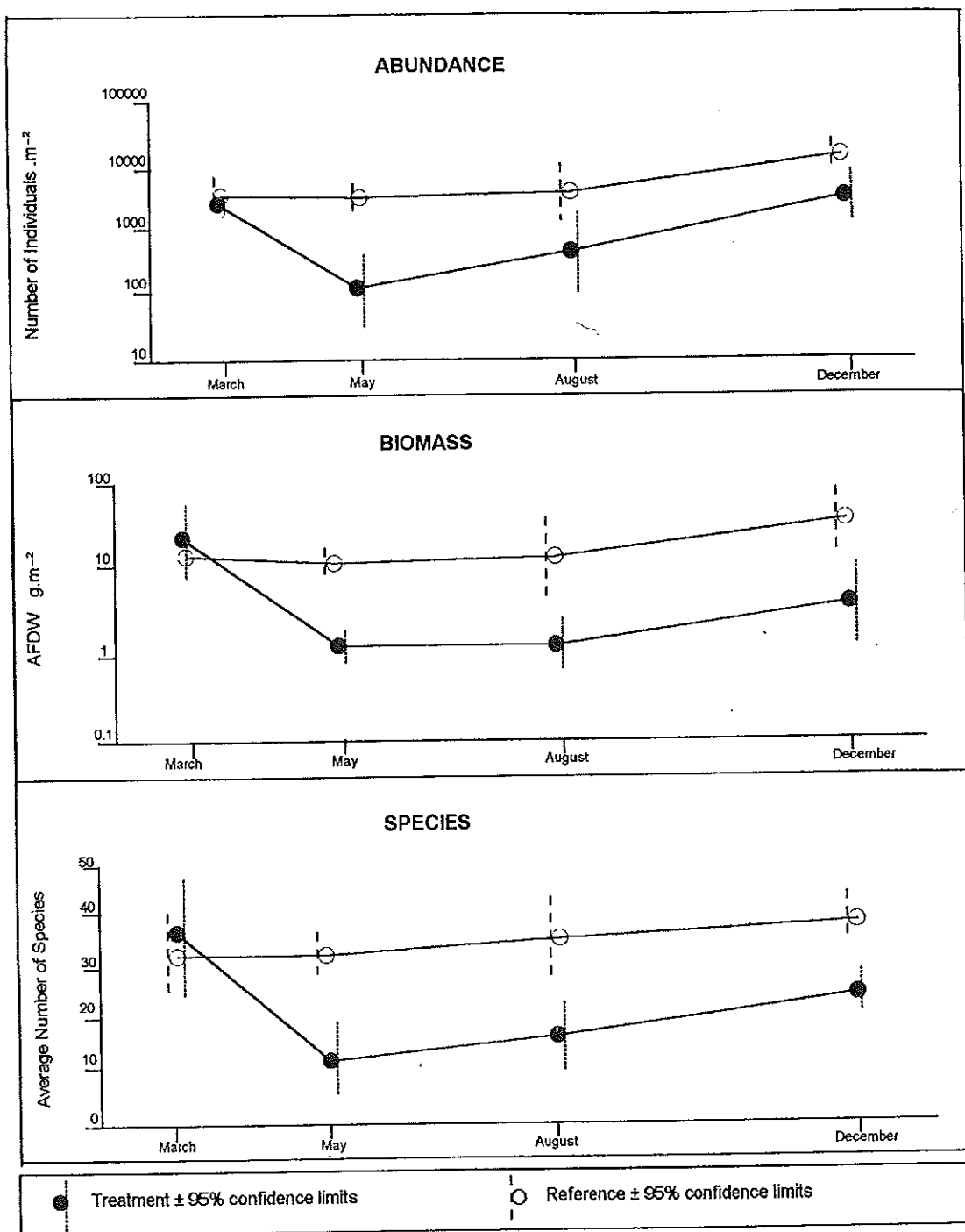
These complex changes in community structure following dredging, and which occur during the recovery process, are difficult to assess by mere inspection of the data for species composition, population density and biomass. Most recent studies on community structure in relation to environmental gradients, therefore, whether these are natural or induced by man, use relatively sophisticated analytical techniques that incorporate the type of species as well as their individual population densities and biomass to assess changes in community structure. The use of these techniques is beyond the scope of this review, but useful accounts for the biologist are given in Kruskal (1977), Hill (1979), Field et al. (1982), Heip et al. (1988), Magurran (1991), Warwick & Clarke (1991), Clarke & Ainsworth (1993), Clarke & Warwick (1994, and references cited therein).

Probably the most widely-used method is detrended correspondence analysis (DECORANA), an ordination technique that arranges stations along axes according to their similarity in species composition (Hill 1979). This is often used in association with two-way indicator species analysis (TWINSPAN) to identify species that characterize particular parts of an environmental gradient such as might be imposed, for example, by dredging or spoils overspill, or communities in relation to wider spatial gradients (see Eleftheriou & Basford 1989).

A second approach is the use of non-parametric multivariate analyses of community structure as outlined by Field & McFarlane (1968), Field et al. (1982) and Clarke & Warwick (1994). This procedure has recently become available in a convenient software package PRIMER (Plymouth Routines in Multivariate Ecological Research) and is now widely used in the analysis of benthic community structure in European coastal waters.

Despite problems in the interpretation of long-term studies on the abundance and composition of marine communities, studies that are carried out over even relatively short time periods can give important information on the recovery process following cessation of dredging. The most comprehensive analysis of the impact of dredging on community composition and on the process of recolonization and recovery in mixed gravel deposits is that of Kenny & Rees (1994, 1996). They carried out an intensive dredging programme by suction trailer dredger in an experimental area off Lowestoft, Norfolk in the southern North Sea and subsequently monitored the recovery over a period of 8 months in the first instance, although this was increased to 2 yr in an extended study of the recolonization process (Kenny & Rees 1996). Dredging occurred in April 1992, during which the suction dredger SAND HARRIER removed a total of 52 000 t of mixed aggregates from an area measuring 500 m by 270 m, an estimated 70% of the surface deposits down to an average depth of 0.3 m having been removed from the experimental area. The species variety, population density and biomass in the experimentally-dredged site was then compared with that in a reference site nearby over the 8-month period between March and December 1992.

The results from their study are summarized in Fig. 14. This shows that the number of species in the dredged site declined from 38 to only 13 species following dredging, whereas the number of species remained at about 35 during the 8-month period at the reference site. The number of species in the dredged area subsequently increased somewhat in the following 7 months, suggesting that some recolonization occurs even over this relatively short time.



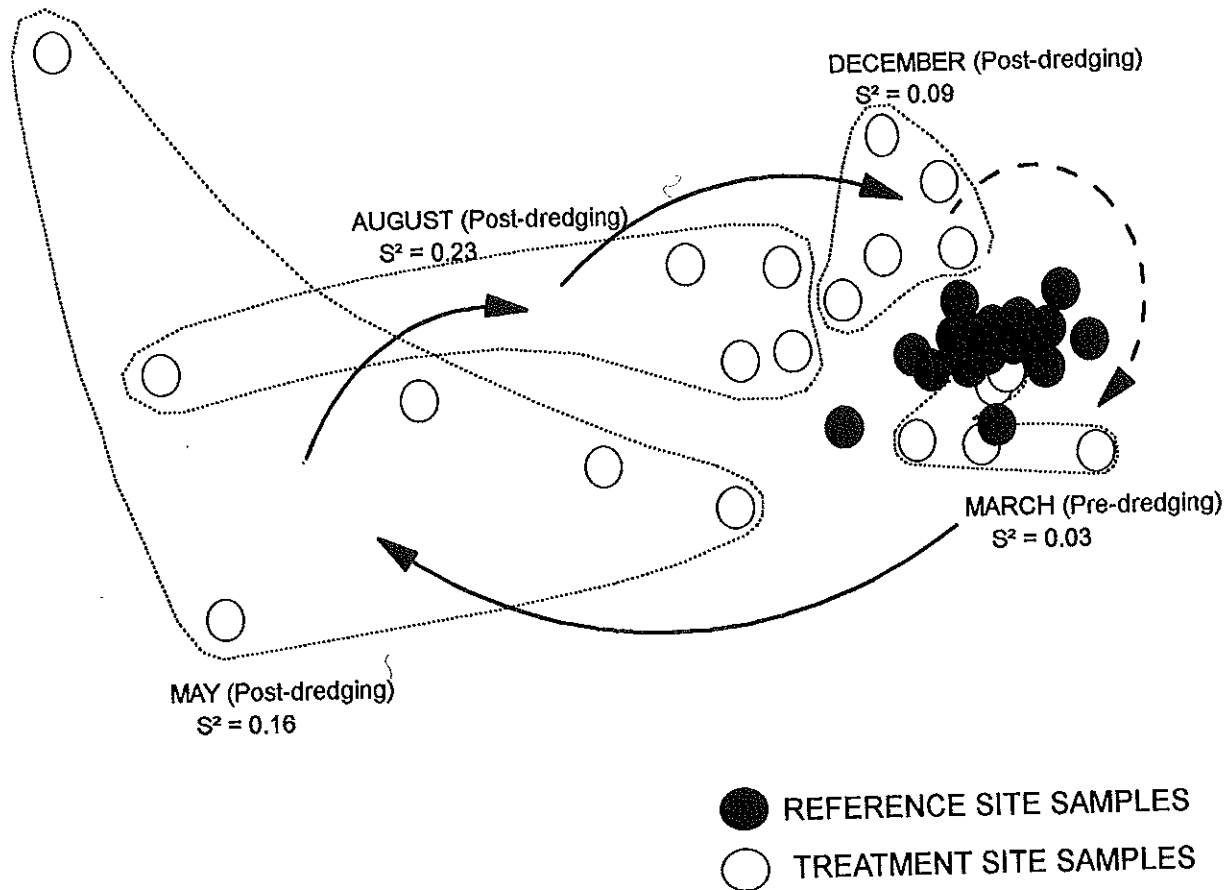
**Figure 14** Graphs showing the mean values for the abundance of individuals (No. per  $m^2$ ) from five Hamon grab samples each of  $0.25 m^2$  taken in a dredged site and at a reference site. Dredging occurred in April, and samples were taken in the pre-dredged deposits in March 1992 and through to December 1992. Values for the biomass are expressed as  $g$  AFDW  $m^{-2}$  from five Hamon grab samples. The average number of species in each of the five Hamon grab samples is also shown. The 95% confidence limits are indicated as bars. Note that there was a significant increase in species variety and abundance during the 7-month post-dredging period, but that the biomass increased only slowly. This indicates that recruitment was mainly of small individuals by larval settlement. Despite this recolonization, it is clear that population density, biomass and species variety had not recovered at the end of the 7-month post-dredging period. (After Kenny & Rees 1994).

## IMPACT OF DREDGING WORKS IN COASTAL WATERS

The average population density for all taxa of 2769 individuals recorded by Kenny & Rees (1994) prior to dredging was reduced after dredging to only  $129 \text{ ind. m}^{-2}$ , compared with a relatively uniform invertebrate population density of  $3300 \text{ ind. m}^{-2}$  in the reference site. Again, the population density showed a significant increase in the 7 months after dredging had ceased.

Inspection of Figure 14 shows that the high biomass of  $23 \text{ g AFDW m}^{-2}$  was reduced to only  $1 \text{ g AFDW m}^{-2}$  after dredging. This reflects the removal of relatively large macrofaunal species, such as the mussel *Modiolus modiolus*, from the dredged sediments and was followed by a slower rate of increase in the post-dredging period than that recorded for population density. This implies that recolonization was initially by small individuals that then grew relatively slowly during the 7 months after dredging had ceased.

Figure 15 shows the output of a non-metric multidimensional scaling (MDS) ordination (see Kruskal 1977, Kruskal & Wish 1978, Field et al. 1982) of the data for the macrofauna sampled in gravel deposits before dredging of the experimental site off



**Figure 15** Two-dimensional multidimensional scaling (MDS) ordination for the benthic communities in a survey area off Norfolk in March 1992 prior to dredging, and in May, August and December 1992. Note that dredging of the experimental area resulted in an initial impact on community structure which differed from that in control areas and to that in the deposits prior to dredging. In the following months community structure became more similar to that in the undredged deposits, but was still distinct at the end of the 7-month post-dredging period. (After Kenny & Rees 1994).

Norfolk, and in the 7 months after dredging (after Kenny & Rees 1994). Their multivariate analysis of community structure prior to dredging and in the months following dredging shows a number of important features of the recolonization process that highlight the general principles of succession outlined in Fig. 2 (p. 134).

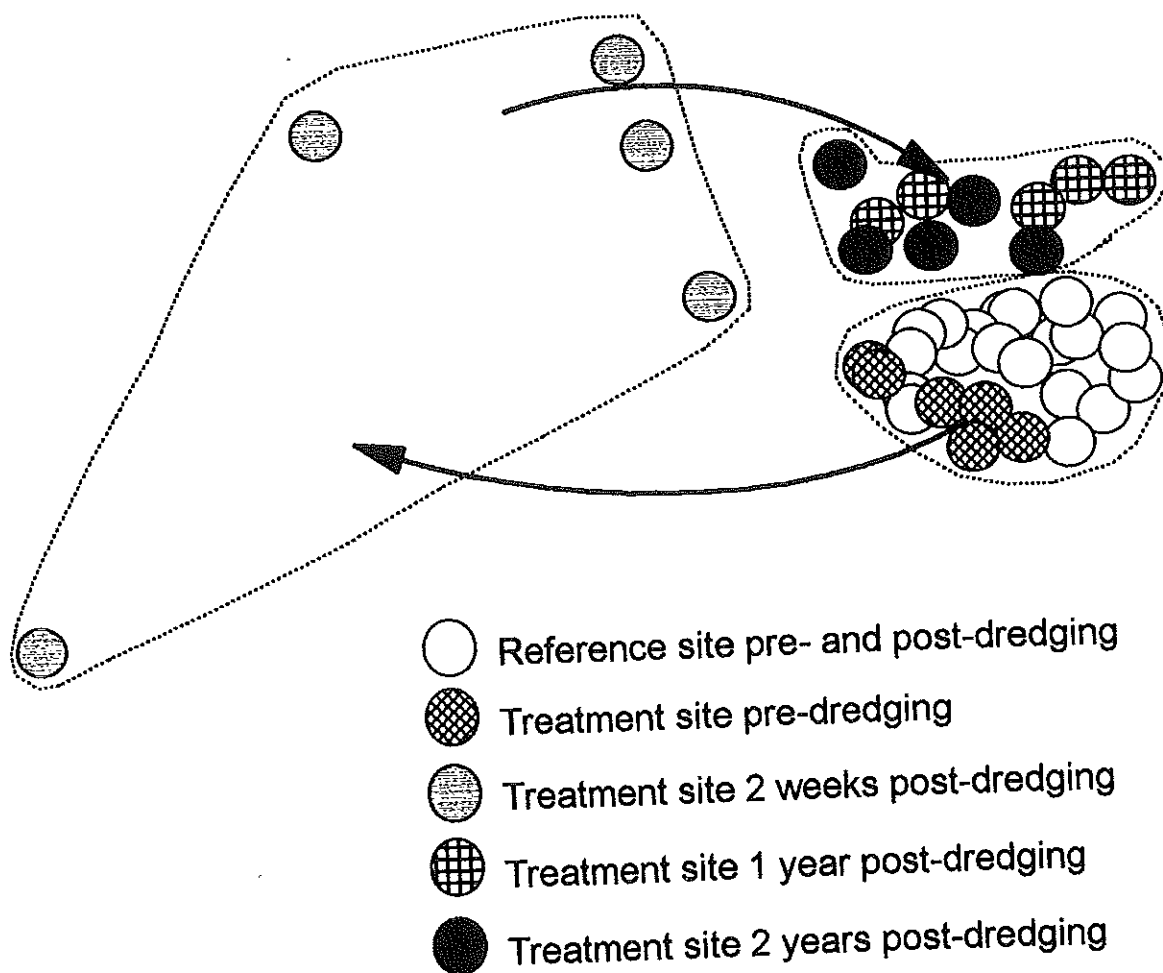
The first point that is clear from their results is that the community within the dredged site prior to dredging in March 1992 formed a small "cluster" on the MDS ordination. This indicates that the communities sampled within the experimental site were similar to one another, and were also evidently very similar to those in the reference site since they are close together on the MDS ordination.

The experimental area was again sampled in May 1992, 1 month after completion of dredging. At this stage it can be seen from Fig. 15 that dredging had resulted in two important changes in community structure. First, the communities in all the samples from the dredged site were well separated in the MDS plot from those in March and from those in the reference site. This implies a major change in community composition following dredging. Secondly, the communities at each of the sampling sites within the dredged area were different from one another. This is indicated by the fact that they have an increased derived variance ( $S^2$ ) and no longer form a tight "cluster" on the MDS ordination shown in Fig. 15 (see also Warwick & Clarke 1993). This increased variance would be expected when some samples were taken from the dredged furrows themselves whereas others were from areas between furrows.

One of the interesting features of this study is that it shows that much of the initial process of colonization of the gravel deposits off the Norfolk coast was accomplished within the following 7-month period. Inspection of Fig. 15 shows that the community in the dredged area became more similar to those in the surrounding deposits of the reference area and to those in the pre-dredged site, and also had a closer internal similarity to one another ( $S^2$  reduced to 0.09) in the months following cessation of dredging. This shows that many of the commoner species present in the dredged area in March 1992 prior to dredging had recolonized by December 1992. The clear difference from both the reference site and the community prior to dredging suggests, however, that many of the rarer components of the community had not yet colonized the dredged area in the following 7 months.

The study was then extended to include data for a 2-yr period following dredging. These results are reported by Kenny & Rees (1996). They showed that although recruitment of new species, especially *r*-selected species such as the barnacle *Balanus crenatus* and the ascidian, *Dendrodoa grossularia* had occurred by December 1992, even at the end of a 2-yr period both the average species abundance and biomass for the dredged area were lower than those in the reference site.

It is also clear from their work that the community composition in the dredged area was not restored even 2 yr after dredging. Inspection of Figure 16 shows the tightly clustered samples from the reference site and from the pre-dredged experimental site in March 1992. The marked shift in community composition and the increased variation between samples taken in May 1992 shortly after dredging is shown, as well as data collected in May 1993, 1 yr post-dredging and in May 1994, 2 yr after dredging. It is apparent from Figure 16 that despite the significant recolonization that had evidently occurred within 7 months of dredging, the community in the dredged area remained distinct from that in the reference area and from that in the deposits prior to dredging, even after 2 yr. Whether this reflects residual differences in the nature of the deposits following dredging, or the long time period required for establishment of the rarer



**Figure 16** Two-dimensional multidimensional scaling (MDS) ordination for the benthic communities in a survey area off Norfolk in March 1992, and for following 2 yr post-dredging. Note that despite the increasing similarity of the community in the dredged area to those in the surrounding sediments over the 2 yr post-dredging period, recovery had not been fully accomplished even after 2 yr. (After Kenny & Rees 1996).

components of the original community is not yet known.

The results that have been reviewed above thus show that the process of recolonization involves two distinct phases; first, recolonization of species composition and population density by settlement of small individuals as larvae and juveniles; secondly, a period of growth during which the biomass approaches that in the undisturbed deposits. Inspection of Figure 16 shows, however, that in the gravel deposits of the southern North Sea this process had only entered its initial phase of partial restoration of community structure in the 7-month period that followed cessation of dredging, and that full recovery may take several years, much as would be anticipated for typical equilibrium communities on the sea bed (see Figure 4, p. 139).

### The rate of recovery of biological resources

The rates of recovery of biological resources following capital and maintenance dredging, disposal of dredged spoils and marine aggregate dredging have been widely

studied in other habitats and conform with the general principles of ecological succession shown in Figures 2 and 3. That is, communities that inhabit fine semi-liquid and disturbed muds comprise opportunistic *r*-selected species that have a high rate of recolonization and which can reach high population densities within weeks or months of a catastrophic mortality. Conversely, communities that inhabit less disturbed deposits of deeper waters or coarse substrata have complex associations and are characterized by large slow-growing species that are selected for maximum competitive advantage in a habitat where space is already crowded. These large, slow-growing *K*-selected equilibrium species recolonize only slowly following disturbance and may take several (or many) years for recovery of full species composition and biomass.

Table 5 shows the rates of recovery of the benthic fauna following dredging in various habitats. We have included semi-liquid muds from freshwater tidal areas and have arranged the data along a gradient of increasing environmental stability and predictability through estuarine and coastal muds to sands and gravels and coral reef assemblages. Inspection of the data summarized in Table 5 shows that recovery of the benthic fauna in highly disturbed semi-liquid muds can occur within weeks. This is associated with an ability for species such as *Limnodrilus* spp., *Ilyodrilus*, *Coelotanypus* sp. and *Procladius* to migrate through the surrounding deposits and to recolonize dis-

**Table 5** Table showing the rates of recovery of the benthic fauna following dredging in various habitats. Note that highly disturbed sediments in tidal fresh waters and estuaries that are dominated mainly by opportunistic (*r*-strategist) species have a rapid rate of recovery. Recovery times increase in stable habitats of gravels and coral reefs that are dominated by long-lived components with complex biological interactions controlling community structure. Longevity and slow growth are also associated with slow recolonization rates in sub-arctic seas. Examples have been arranged along a gradient from disturbed muds of freshwater-tidal estuarine conditions to stable reef assemblages.

Locality	Habitat type	Recovery time	Source
James River, Virginia	Freshwater semi-liquid muds	± 3 wk	Diaz 1994
Coos Bay, Oregon	Disturbed muds	4 wk	McCauley et al. 1977
Gulf of Cagliari, Sardinia	Channel muds	6 months	Pagliai et al. 1985
Mobile Bay, Alabama	Channel muds	6 months	Clarke et al. 1990
Chesapeake Bay	Muds-sands	18 months	Pfitzenmeyer, 1970
Goose Creek, Long Island, NY	Lagoon muds	>11 months	Kaplan et al. 1975
Klaver Bank, Dutch Sector, North Sea	Sands-gravels	1-2 yr (ex-bivalves)	van Moorsel 1994
Dieppe, France	Sands-gravels	>2 yr	Desprez 1992
Lowestoft, Norfolk, UK	Gravels	>2 yr	Kenny & Rees 1994, 1996
Dutch Coastal Waters	Sands	3 yr	de Groot 1979, 1986
Tampa Bay, Florida	Oyster shell (complete defaunation)	>4 yr	US Army Corps of Engineers 1974
Tampa Bay, Florida	Oyster shell (incomplete defaunation)	6-12 months	Conner & Simon 1979
Boca Ciega Bay, Florida	Shells-sands	10 yr	Taylor & Saloman 1968
Beaufort Sea	Sands-gravels	12 yr	Wright 1977
Florida	Coral reefs	>7 yr	Courtenay et al. 1972
Hawaii	Coral reefs	>5 yr	Maragos 1979

turbed muds as adults. A similar recolonization of disturbed deposits in dredged channels may also account for the relatively fast recolonization of some muds and sands in near-shore waters, especially those where tidal currents may transport juveniles into the dredged area (see Hall 1994).

Inspection of the recolonization rates reported in the literature and summarized in Table 5 suggest that a period of 2–4 yr is a realistic estimate of the time required for recovery in gravels and sands, but that this time may be increased to more than 5 yr in coarser deposits, including coral reef areas. Interestingly, the data for areas in Tampa Bay, Florida that had been dredged for oyster shell, suggest that a period of as much as 10 yr may be required for recovery following complete defaunation whereas a recovery time of only 6–12 months is required for recovery following partial dredging and incomplete defaunation (see Benefield 1976, Conner & Simon 1979). This suggests that areas of undisturbed deposits between dredged furrows may provide an important source of colonizing species that enable a faster recovery than might occur solely by larval settlement and growth (see also van Moorsel 1993, 1994).

Other more complex environmental factors also evidently affect the rate of recovery of dredged areas. Studies in the Dutch Wadden Sea by van der Veer et al. (1985) show that the recovery of species composition and biomass of benthic organisms was related to the speed of infilling of dredged pits. These data are summarized in Table 6, which shows that even 16 yr after cessation of dredging no recovery of the benthos had occurred on a tidal flat at Terschelling Sand. On a tidal watershed at Oosterbierum a partial recovery of 85% of the species and 39% of the biomass had occurred after 4 years. This is typical of recolonization by small individuals that were in the process of growth towards the original biomass levels of the undisturbed deposits, a process which would clearly take several more years.

In the tidal channels, both the rate of infill and recolonization were related to the speed of currents. A partial recovery of 57% of the species and 67% of the biomass was recorded after 3 yr in a tidal channel at Paesensrede (see Table 6), with greater recovery and shorter time periods being recorded in areas of faster current. Even then, it will be noted that the species composition had not recovered and that the biomass evidently became dominated by fewer species of relatively large size compared with those in the surrounding deposits.

The likely recolonization rates for the benthic community of estuarine muds, sands, gravels and reef areas have been superimposed onto a generalized colonization succession in Figure 17, which allows some predictions to be made on the rates of recovery of deposits following dredging. The fine muds that characterize coastal

**Table 6** Table showing the percentage recovery recorded in a variety of habitats in the estuarine Dutch Wadden Sea following dredging up to 15 yr previously. Based on van der Veer et al. 1985.

Area	Habitat	Time interval since dredging	% Recovery	
			Species	Biomass
Terschelling sand	Tidal flat	16 yr	0	0
Oosterbierum	Tidal watershed	4 yr	85	39
Paesensrede	Tidal channel	3 yr	57	67
Holwerderbalg	Tidal channel	2 yr	64	100
Kikkertgat	Tidal channel	1 yr	88	116

embayments, estuaries and lagoons are likely to be colonized by large populations of a relatively restricted variety of opportunistic *r*-selected species, which are capable of rapid colonization within months of space being made available for colonization and growth. Because such deposits are subject to regular disturbance under natural conditions prior to dredging, the ecological succession recovers to the colonization phase shown in Figure 17, but does not proceed to the development of *K*-selected slow-growing equilibrium species within the community. Recovery of the "normal" community in disturbed deposits such as muds, therefore, can be achieved within months of cessation of dredging, or disposal of spoils.

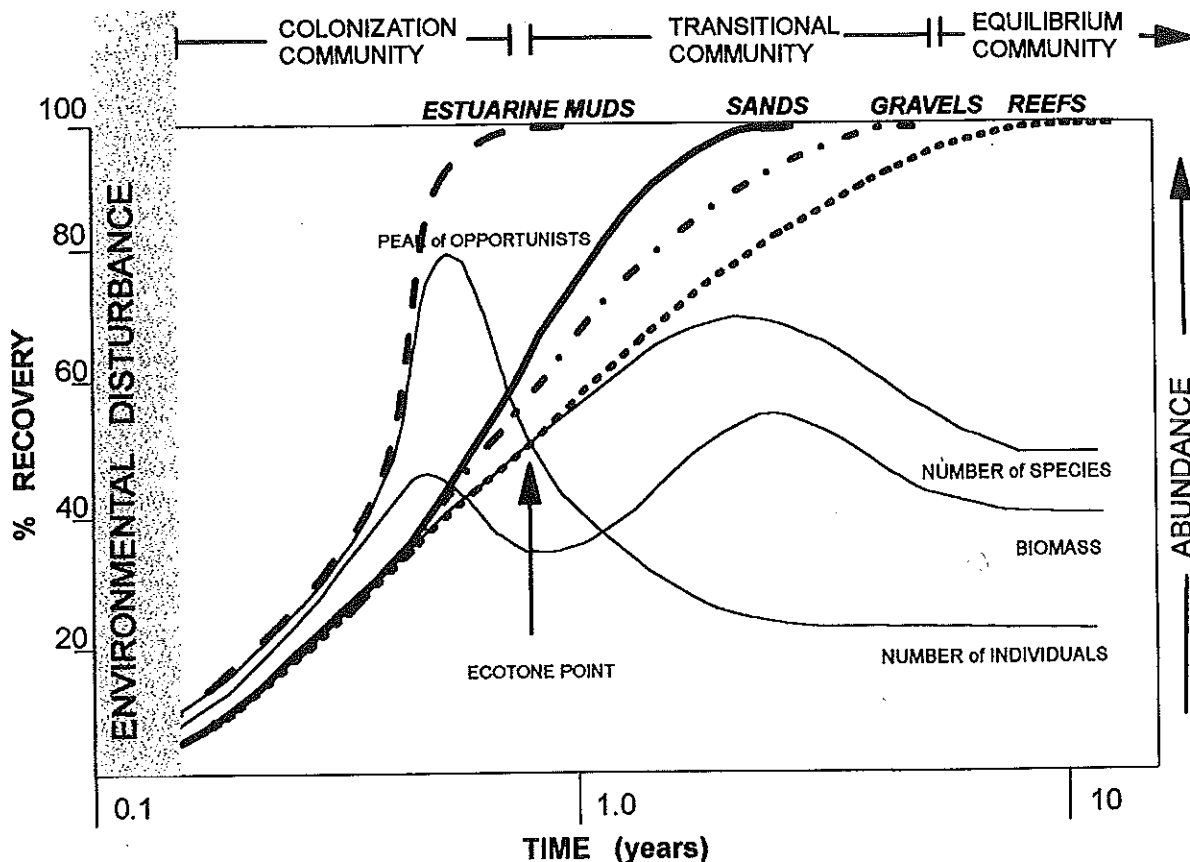


Figure 17 Schematic diagram showing the likely recolonization rates for the benthic community of estuarine muds, sands and reef areas. The curves for recovery have been superimposed onto a generalized colonization succession and allows some predictions to be made on the rates of recovery of deposits following dredging. Note that the fine muds that characterize coastal embayments, estuaries and lagoons are likely to be recolonized by a relatively restricted variety of opportunistic *r*-selected species within months of space being made available for recolonization and growth. Because such deposits are subject to regular disturbance, the succession recovers to the colonization phase, but does not proceed to the development of long-lived slow-growing *K*-selected species. The natural communities of gravels and sands, however, contain varying proportions of slow-growing *K*-selected equilibrium species depending on the degree of disturbance by waves and currents. These communities are held in a transitional state by natural environmental disturbance and are likely to recover within a period of 2–3 yr after cessation of dredging. Finally, the recovery curve for reef communities indicates that a period of 8–10 yr may be required for the long process of establishment and growth of the long-lived and slow-growing *K*-selected species characteristic of equilibrium communities.



The natural communities of gravel and sand deposits, however, contain varying proportions of slow-growing, *K*-selected equilibrium species, depending, among other factors, on the degree of disturbance by waves and the speed of tidal currents. In this case, the tail of the sigmoid recovery curve becomes more pronounced because the rarer components of the equilibrium community may take several years to recolonize the deposits, even after the main components of the community have become established. Where the deposits are sandy, periodic mortality of the long-lived components may result in major seasonal changes in community composition such as occurs in the North Sea on the Klaver Bank (van Moorsel 1994), and as has been reported for the sediments of Liverpool Bay by Eagle (1975). Under these conditions, the community will be held in a transitional state by natural environmental disturbance, and is likely to recover within 2–3 yr after cessation of dredging.

There is good evidence that disturbance of the deposits by man may result in a shift from the equilibrium community characteristic of undisturbed deposits towards the transitional community, which characterizes deposits in areas of natural environmental disturbance. Studies by de Groot (1984) suggest, for example, that the increasingly heavy bottom gear used by trawlers has been associated with a shift in community composition of the benthos of the North Sea, and this also applies to the benthos of the Wadden Sea.

As might be anticipated from the successional sequence shown in Figure 17, long-lived components such as molluscs and larger crustaceans in near-shore waters, such as the Wadden Sea, have decreased in numbers and diversity over the years and have been replaced by larger populations of rapidly growing polychaete species (Reise 1982, Riesen & Reise 1982, Reise & Schubert 1987).

Finally, the community recovery curve for reef communities indicates that a period of 8–10 yr may be required for the process of establishment and growth of the long-lived and slow-growing *K*-selected equilibrium species and for the development of the biological interactions that are familiar to those who have observed the immense diversity and complexity of life on undisturbed reef structures. This long process of establishment of an equilibrium community reflects partly the time required for colonization by rarer components of the community, but is also influenced by the nature and stability of the substratum following cessation of dredging, and the time required for complex stabilization processes involving both physical compaction and biological interactions. The relationship between biological community structure, sediment composition and seabed stability is considered in more detail below.

### Community composition and seabed stability

The influence of sediment composition in controlling the nature of communities of animals that live on the sea bed has been widely recognized since the pioneer studies of Petersen (1913), Thorson (1957) and Sanders (1958). Most recent evidence suggests, however, that the precise relationship between biological community composition and

specific properties of the sediments is poorly understood. In some estuaries and shallow water coastal embayments, fine grained and silty deposits clearly support an entirely distinct community compared with those from mobile sands or on stable substrata such as rocks and boulders.

On the other hand it is a matter of common observation that although very fine mobile muds may be dominated by opportunistic species such as the amphipod *Ampelisca brevicornis* or the polychaete *Lagis koreni*, the same silts can become consolidated into clays and then support long-lived and sedentary equilibrium species such as the boring piddock bivalves *Pholas dactylus* and *Barnea parva* as well as an epifauna of hydroids, ascidians and other species more characteristic of reefs. Clearly, the stability of the sediment, rather than particle size itself, is of importance in controlling community structure. In other instances it is clear that the deposits on the sea-bed undergo a complex process of consolidation or "armouring" that allows the establishment of communities that are more typical of rocks and reefs reflecting the complex relationships between the physical deposits and biological activities of the animals themselves.

The relationship between community composition and sediment type in deeper waters of the continental shelf is less well documented than that for estuaries and lagoons. Some early studies suggest that macrobenthic communities can be distinguished on a basis of sediment granulometry (Glémarec 1973, Buchanan et al. 1978, Flint 1981) but other studies have shown little correlation (Buchanan 1963, Day et al. 1971). Efforts to identify what physical properties are of greatest importance in controlling the structure of marine communities are often frustrated by the fact that most of the sediment variables obtained from conventional sorting methods are interdependent since they are expressed as a percentage of the total sample (see Weston 1988). A high percentage of silt, for example, is inversely related to the percentage of the other sediment components. Again, many of the physical properties of sediments are linked with other features such as depth of disturbance by wave action, strength and duration of currents, and may themselves be linked with complex biological interactions including the surface area available for microbial food components, and the presence of species that can exclude potential competitors.

Partly for this reason, most recent studies have concluded that the complexity of soft-bottom communities defies any simple paradigm relating to a single factor, and that there should be a shift towards understanding relationships between the distribution of organisms in terms of a dynamic relationship between the sediments and their hydrodynamic environment. According to this view, complex shear forces at the sediment-water interface are considered to play a dominant role in controlling food availability, settlement of larvae, microbial food availability, pore water flow and other environmental features that affect the benthic organisms that inhabit marine deposits. It is therefore considered unlikely that any one factor alone, or even a combination of single granulometric properties, can account for the distribution of animals in most sedimentary habitats (for review, see Snelgrove & Butman 1994).

Despite this emerging view that sediment granulometry itself is unlikely to control the composition and distribution of biological communities on the sea bed, concern has been expressed that dredging for marine aggregates can result in significant changes in sediment composition. Studies off Dieppe, France have shown, for example, a large increase in the proportion of fine sand in deposits that have been intensively worked for marine aggregates (see Desprez 1992, ICES 1992, 1993). Again, the infill of pits and grooves from dredging for marine aggregates is commonly dominated by the

fine deposits which are capable of mobilization by shear stress induced by waves and tidal currents (Dickson & Lee 1972, Shelton & Rolfe 1972, Millner et al. 1977).

If sediment composition were of importance in controlling biological community composition, such changes following dredging could potentially prevent subsequent recolonization by communities that were similar to those that occurred in the deposits prior to dredging (see Windom 1976) and could by implication affect the nature and abundance of food organisms for commercial fish stocks.

We have analyzed the relationship between biological community composition and the sediment granulometry in undredged coastal deposits in the English Channel and southern North Sea and find that both biological communities and the sediments fall into relatively distinct groups or communities when analyzed by multivariate techniques (Newell & Seiderer 1997d). However, there is little evidence of any correspondence between the distribution of different sediment types and biological communities in the survey areas. Analysis of the Spearman rank correlation between the similarity of biological communities and any one, or a combination of, particle size indices show that granulometric properties of the sediments are likely to account for a maximum of 45% of the variability of the biological component, leaving approximately 55% determined by other environmental factors.

The conclusion to be drawn from these results is that they support recent views that biological community composition is not controlled by any one, or a combination of simple granulometric properties of the sediments such as particle size distribution. It is considered more likely that biological community composition is controlled by an array of environmental variables, many of them reflecting an interaction between particle mobility at the sediment-water interface and complex associations of chemical and biological factors operating over long periods.

Such interactions are not easily measured or analyzed, but the results clearly suggest that restoration of sediment composition after completion of dredging for marine aggregates is not, within broad limits, a prerequisite for the establishment of marine communities that are comparable with those that occurred in the deposits prior to dredging. What is possibly of more importance in controlling the time course of recovery of an equilibrium community characteristic of undisturbed deposits is the process of compaction and stabilization. This will reflect changes in sediment composition, but is also in equilibrium with seabed disturbance from tidal currents and wave action, both of which show spatial variations and interactions with water depth. The processes associated with compaction and stability of seabed deposits may therefore largely control the establishment of long-lived components of equilibrium communities and account for the dominance of opportunistic species in the initial stages of colonization of recently sedimented material in unconsolidated deposits after the cessation of dredging.

## Conclusions

At the outset of this review, we assessed the importance of the benthic community to fisheries production and outlined our intention of providing an ecological framework within which the impact of dredging can be understood. We have shown that systems

models for shelf waters such as the North Sea suggest that the flow of materials from primary production by the phytoplankton passes partly through planktonic grazers, but that 20–50% sinks to the sea-bed either from dead and decaying phytoplankton cells, or as faecal material derived from the feeding activities of the grazing zooplankton (Steele 1974, Joiris et al. 1982, Newell et al. 1988). Such material then passes into the benthic food web, whose production in turn forms an important food resource for demersal fish.

It has been estimated from empirical models developed for the North Sea that as much as 30% of total fisheries yield to man is derived from benthic resources (see Fig. 1, p. 129). Production by the benthos is therefore important, not only as a resource in itself, but as a key food resource for demersal fish stocks. It becomes an increasingly important component of the marine food web in near-shore waters where primary production by larger macrophytes and seagrasses living on the sea-bed largely replaces that from the phytoplankton in the water column (for review, see Mann 1982).

From this it is clear that reclamation of large areas of coastal wetlands, coastal embayments or estuaries can have a potentially important effect on the supply of materials and energy to marine food webs, and that even in plankton-based deeper water ecosystems such as the North Sea, fish yields based on benthic production are sufficiently large to warrant proper conservation of benthic resources. Our review has concentrated, therefore, on the nature of benthic communities, their susceptibility to disturbance by dredging and land reclamation works, and on the evidence that is available for the recovery times required for the re-establishment of community structure following dredging or spoils disposal.

Our review of the literature shows that the communities of near-shore habitats are characterized by large populations of a relatively restricted variety of species that are well-adapted to exploit space that has become newly available by episodic catastrophic mortality. Such species are generally small, often mobile, and are selected for maximum rate of population increase, with high fecundity, dense settlement, rapid growth and rather a short life cycle. Such species have been designated "*r*-strategists" (see MacArthur & Wilson 1967, Pianka 1970) and have been referred to in our review as opportunists. Their population characteristics allow a rapid recovery of the initial community structure in deposits that are naturally subjected to high levels of environmental disturbance. It is, not surprising, therefore, to find that there are frequent reports in the literature of community recovery times that range from a few weeks to several months for disturbed deposits such as semi-liquid muds in tidal fresh waters, estuaries, lagoons and dredged channels (see Table 5, p. 162 and Fig. 17, p. 164).

In deeper waters, or where the substratum is sufficiently stable to allow the long-term survival of benthic organisms, the habitat tends to be crowded. Under these conditions, organisms have an "equilibrium strategy" and are selected for maximum competitive ability in an environment in which space for colonization and subsequent growth is limiting. Such species have been designated "*K*-strategists" and devote a larger proportion of their resources to non-reproductive processes such as growth, predator avoidance, and investment in larger adults (MacArthur & Wilson 1967, Gadgil & Bossert 1970). Because the *K*-selected equilibrium species live longer, they tend to have wider limits of physiological tolerance, which allows them to survive those variations in environmental conditions that occur in their habitat. Many have active site selection phases that include chemical recognition of the presence of adults of the same species, a strategy which ensures that environmental conditions have been within

the limits of tolerance for long enough to allow survival of other members of the same species (for review, see Newell 1979).

Such *K*-selected equilibrium species develop complex biological associations with other long-lived components of the community, and may alter the environment in such a way as to both allow the presence of many other species that would not otherwise occur, and also inhibit other potential competitors for space. Biological interactions between the components of equilibrium communities that are characteristic of stable substrata thus lead to the development of complex communities that may take many years, or decades, to re-establish following destruction. It is therefore not surprising to find that as one moves along a gradient of increasing sediment stability from muds through sands to gravels and reefs, there is a corresponding increase in the times reported for recovery of community structure (Table 5, p. 162).

Knowledge of the components that comprise the benthic community on the sea bed, whether these are *r*-selected opportunistic species or *K*-selected equilibrium species, thus gives important information not only on key resources that may require protection, but on the likely rate of recovery following dredging. Inspection of the schematic colonization succession shown in Figure 17 (p. 164) suggests that a recovery time of 6–8 months is characteristic for many estuarine muds whereas sands and gravels may take from 2–3 yr depending on the proportion of sand and the local disturbance by waves and currents. As the deposits become coarser, estimates of 5–10 yr are probably realistic for the development of the complex biological associations between the slow-growing components of equilibrium communities characteristic of reef structures.

Our review suggests that processes associated with compaction and stabilization of seabed deposits may largely control the time-course of recovery of these long-lived components of equilibrium communities and account for the dominance of opportunistic species in the initial stages of colonization of recently sedimented material in unconsolidated deposits following the cessation of dredging.

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