

SECTION 6 - IMPACTS ON BENTHOS

6.1 Introduction

The importance of benthic communities in marine food webs leading to commercially exploitable yields of fish have been widely recognised. 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 of 80-90% of the material entering the consumers is re-mineralised and returned to the water column to support further primary production by the phytoplankton, leaving a small proportion incorporated into consumer biomass.

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 Figure 6.1 indicates that out of 100 g Carbon. m⁻².y⁻¹ produced at the sea surface as net primary production by the phytoplankton, only 0.3 g Carbon. m⁻².y⁻¹ appears as yield to man through the pelagic food web, and approximately 0.13 g Carbon. m⁻².y⁻¹ from demersal fish.

Despite the huge dissipation of materials which 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 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* 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 sea bed deposits have been widely studied in integrated investigations on the 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 organisms (Loosanoff, 1962; Sherk, 1971; Sherk *et al*. 1972, 1974; 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 (Morton, 1977; Conner & Simon, 1979; Johnston, 1981; Giesen *et al*, 1990; Onuf, 1994).

Comprehensive studies of the impact of dredging for marine aggregate 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) and Charlier & Charlier (1992). A recent review for the Minerals Management Service, U.S. 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 which 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 marine aggregate extraction 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 which inhabits the deposits in

the dredged area and surrounding deposits, latitude and the extent to which the community is naturally

adapted to high levels of sediment disturbance and suspended particulate load.

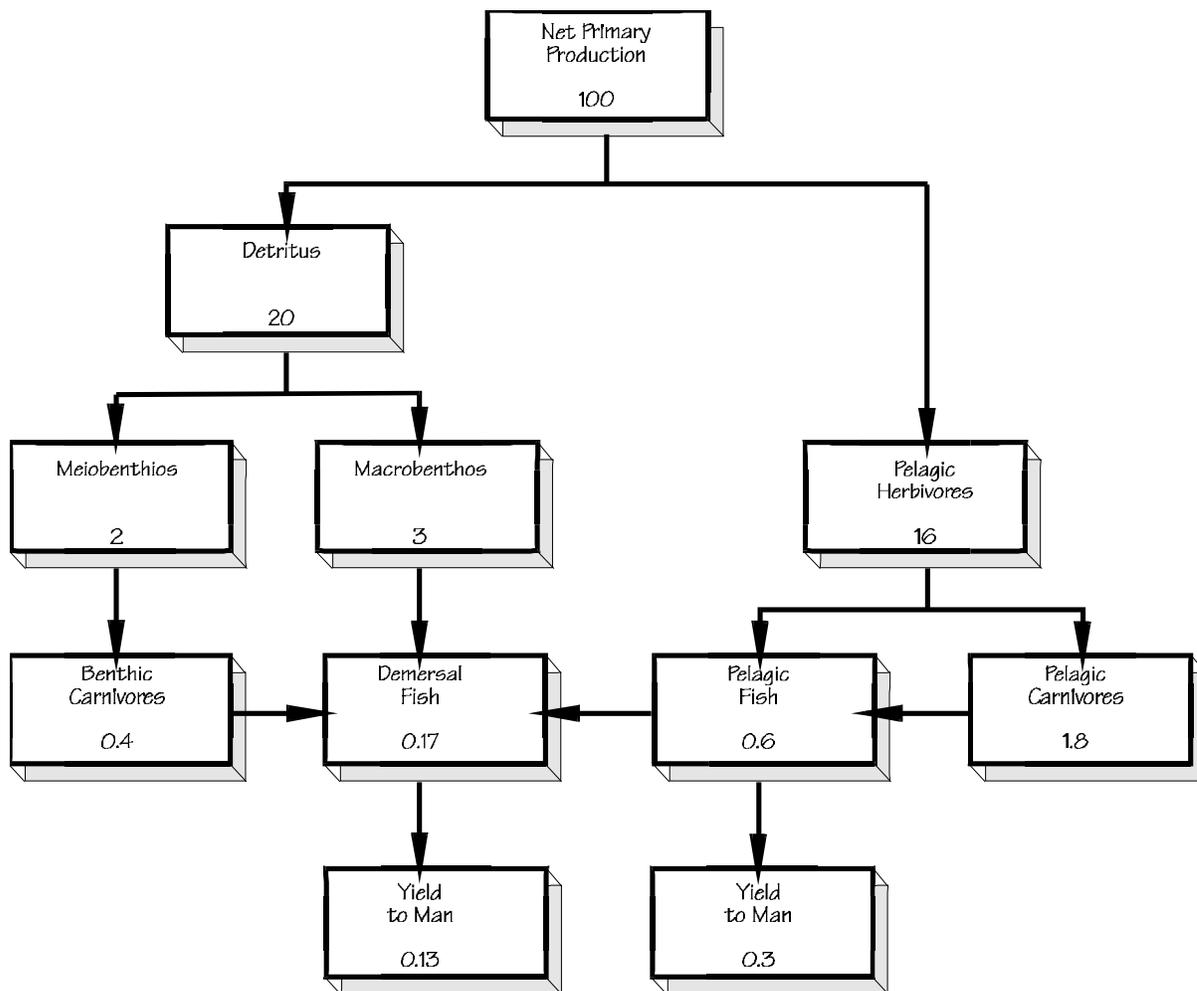


Figure 6.1 Simplified empirical Carbon flow diagram for the phytoplankton-based ecosystem of the shelf waters of the North Sea. Note that of the 100 grams Carbon per metre² per year (100gC.m⁻² yr⁻¹) of sea surface produced by the phytoplankton, the yield to man through pelagic food webs and pelagic fish is estimated to be 0.3gC.m⁻²yr⁻¹, whilst that through benthic food webs is 0.13gC.m⁻²yr⁻¹, or about 30% of the total exploitable fish yield to man (based on Steele, 1965)

In general, rapid rates of initial recolonisation have been reported for some coastal deposits where the organisms are mainly mobile "opportunistic" species which have a rapid rate of reproduction and growth. Such organisms may also be able to recolonise the deposits by migration of the adults (*see* McCall, 1976; Conner & Simon, 1979; Salomon, *et al*, 1982; Guillou & Hily, 1983; van der Veer *et al*, 1985; Pagliai *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 which 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 which characterise coastal ecosystems such as the Dutch Wadden Sea, van der Veer *et al* (1985) report that recolonisation takes 1-3 years in areas of strong currents but up to 5-10 years 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 which characterise 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 works may be extended in colder waters at high latitudes where communities typically comprise large slow-growing species which may take many years for recolonisation and growth.

In a Swedish fjord system, for example, a recovery which was indistinguishable from natural variations was established only after 8-years following closure of a pulp mill (Rosenberg, 1976), whilst de Groot (1979; *see also* Wright, 1977; Aschan, 1981) reports that recovery of communities within the Arctic Circle may take more than 12 years compared with estimates of approximately 3 years for deposits off the coast of the Netherlands. Similar extended time-scales for recolonisation 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 which 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 of time in the absence of the disturbance by dredging.

That is, the *actual* climax community which is attained may not be an appropriate measure of 'recovery' after a long period. Probably a more practical approach to the question of 'recovery' will be recognition of the establishment of a community which is capable of maintaining itself and in which, say, 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 (*see also* C-CORE, 1996).

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 importantly gives some insight into how long it might take for recovery in dredged areas and the surrounding deposits once dredging has ceased.

6.2 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 which colonise 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 characterised by populations which tend to have a high genetic variability which 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 colonisation 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 colonising species. "Opportunists" rely on a large investment in reproductive effort, rather than mobility, for success in colonising 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 which are able to take advantage of recently created empty habitats quickly and to colonise them with large populations. These mobile colonisers are often associated with frequently-disturbed habitats (*see* Osman, 1977). We distinguish these as "Mobile Opportunist" species (*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.

The following sections 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 activities in nearshore waters and estuarine systems.

Under the stable conditions which 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 which is already colonised 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 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 which occur at the two extremes of the environmental gradient and have different relative proportions of "Opportunistic" *r*-strategists and "Equilibrium" *K*-strategists.

Between these two extremes are communities whose species may be intermediate between those which occur at the extremes of the environmental gradient and have different relative proportions of *r*-strategists and *K*-strategists. The characteristics of *r*-selected and *K*-selected "Equilibrium Species" are summarised in Table 6.2 (based on McCall, 1976; Rees & Dare, 1993), although it should be emphasised that the distinction is to some extent an arbitrary one, and is blurred in habitats which 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 Figure 6.2a. 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 which operate at one end of the gradient and biologically-controlled interactions

under the more uniform environmental conditions which occur in deeper waters (see Sanders, 1969; Boesch & Rosenberg, 1981).

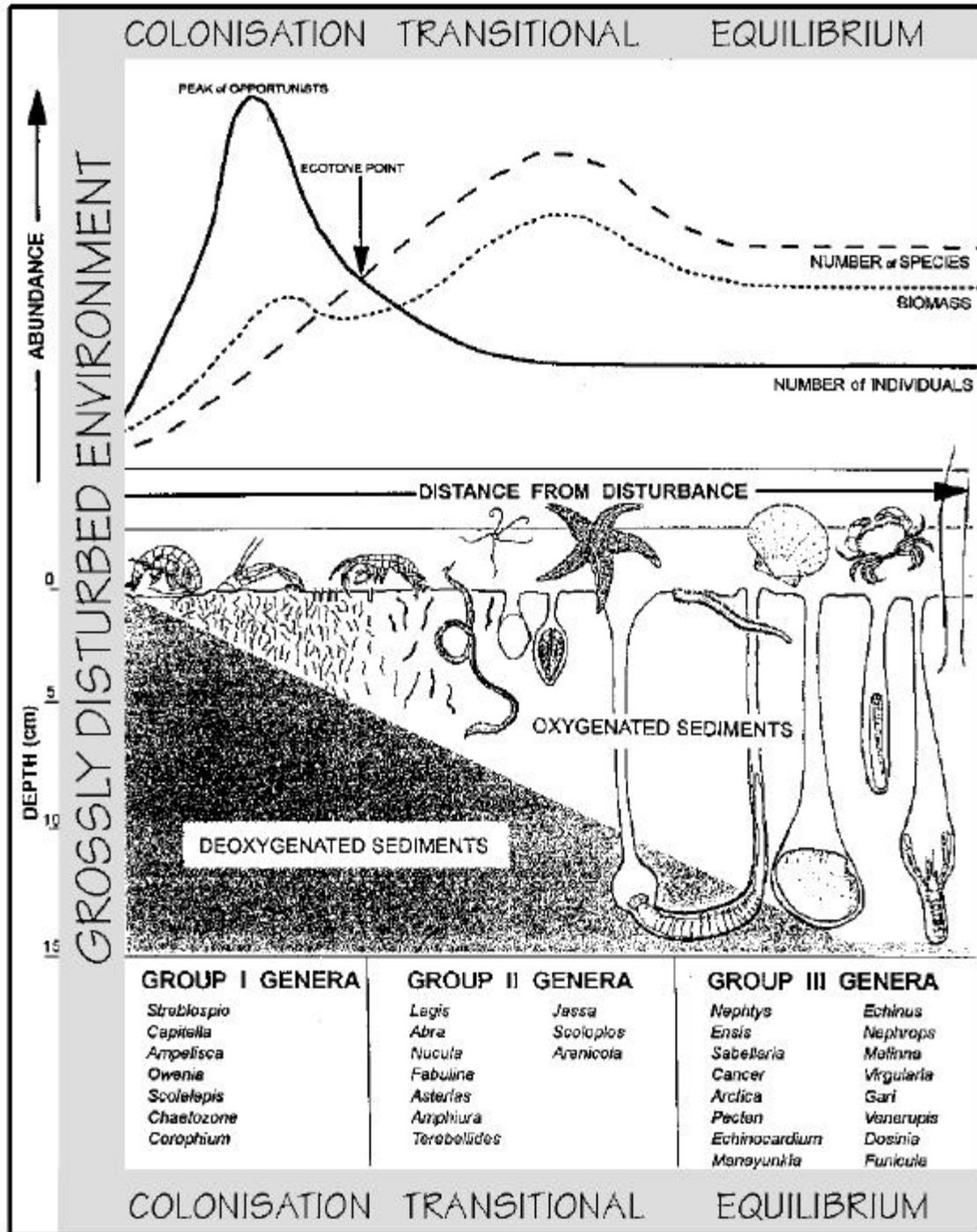


Figure 6.2a Pictorial diagram showing the ecological succession which characterises benthic communities through a gradient of environmental disturbance. Note that in highly disrupted environments (on the left side of the diagram) few organisms may be capable of survival. In polluted or semi-liquid muds the sediments are colonised 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, slow growing 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).

The large species which 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 (Figure 6.2a).

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) summarise 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 colonisation 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 produces consolidated silt deposits which then allow deposit-feeders such as the polychaete *Amphitrite*, the burrows of which in turn provide specialised 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 which 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 which 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 years (Wilson, 1971; Gruet, 1986). Disturbance of communities which are dominated by **K**-strategists may therefore take many years for recovery of their full community composition even though recolonisation 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 which characterise 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 6.2a are thus relatively smaller than their counterparts in deeper waters and comprise a "transitional community" which is confined to a restricted habitat in the surface oxygenated layer of sediment and comprises components which have many intermediate characteristics between typical **r**- and **K**-selected 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 species which 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 6.2a 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 "transition 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 6.2a.

EARLY COLONISING SPECIES <i>r</i> - selected	EQUILIBRIUM SPECIES <i>K</i> - selected
1. Mainly opportunistic species - (a) early reproduction - (b) many reproductions per year - (c) rapid growth - (d) early colonisers - (e) often catastrophic mortality 2. Small body size 3. Generally surface deposit feeders 4. Short life span: generally < 1 year 5. Population size variable, usually well below carrying capacity of environment and recolonised frequently 6. Brood protection with investment of energy into larval food provision (<i>lecithotrophic</i>)	1. Equilibrium species - (a) delayed reproduction - (b) few reproductions per year - (c) slow growth - (d) late colonisers - (e) low death rate 2. Large mobile animals 3. Deposit and suspension feeders 4. Long life span; several to many years 5. fairly constant in time; saturated community in equilibrium with carrying capacity of environment. No recolonisation necessary. 6. No brood protection; larvae widely distributed in the plankton
EXAMPLES	
<i>Streblospio benedicti</i> <i>Capitella capitata</i> <i>Owenia fusiformis</i> <i>Ampelisca abdita</i> <i>Scolelepis fuliginosus</i> <i>Chaetozone setosa</i> <i>Jassa marmorata</i>	<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</i> sp. <i>Amphiura filiformis</i> <i>Terebellides</i> sp. <i>Virgularia mirabilis</i> <i>Gari fervensis</i> <i>Tellina crassa</i> <i>Venerupis rhomboides</i> <i>Dosinia exoleta</i> <i>Scoloplos armiger</i> <i>Abra alba</i>

Table 6.2 Summary of the significant population characteristics of *r*- and *K*- selected species with examples of organisms commonly encountered within aggregate dredging areas of the United Kingdom waters (based on Pianka 1970; McCall 1976; Rees & Dare 1993; Holt et al 1995).

Finally at the extreme end of the physical gradient shown in Figure 6.2a, 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 *r*-strategists which are selected for small size, high fecundity and an ability to recolonise 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 6.2a) before these decline as organic pollution or high environmental disturbance eliminates even these rapid colonisers.

A useful tool which can be used to determine 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 which 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 6.2a.

A typical set of results taken from a recent survey of coastal communities in the eastern English Channel is shown in Figure 6.2b (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 additional contribution to the population.

Further along the gradient of sediment stability 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 & 4 there was a very large species variety of over 300 species 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 which is required for interpretation of the wider impact on community structure in the "Transition Zone".

These distinctions between the lifestyles and adaptive strategies of *r*- and *K*-selected species are of fundamental importance because they go some way towards accounting for the differences in the rate of recovery which 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 *r*-strategists or *K*-strategists and on the time which 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.

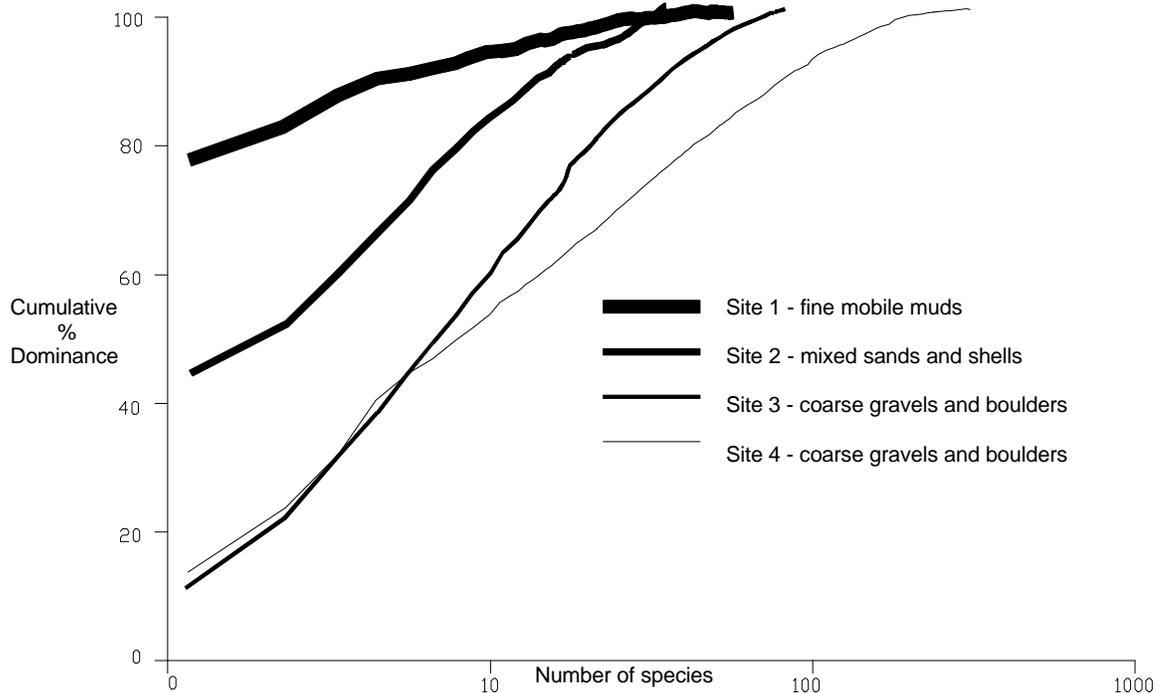


Figure 6.2b A set of typical **K**- dominance curves showing the proportional contribution of individual species to the overall community in fine mobile muds, in mixed sands with shells and in stable habitats comprising coarse gravels and boulders, located off the Kent coast (UK) at West Varne in 1996. The fine mobile muds are dominated by the “Opportunist” amphipod crustacean, *Ampelisca brevicornis*, whilst 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).

6.3 Ecological Succession & The Recolonisation 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 which recolonise deposits after the cessation of environmental disturbance.

McCall (1976) and Rees & Dare (1993) have recognised the occurrence of three main types of benthic components of marine communities based on the distinction between *r*-strategists and **K**-strategists. Group I species comprise those which colonise first after a community has been removed by disturbance. They comprise large populations of small sedentary tube-dwelling deposit-feeders which 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*-strategists.

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 colonisers, but the components attain a lower peak abundance than the smaller *r*-strategists and have a slower recruitment and growth rates.

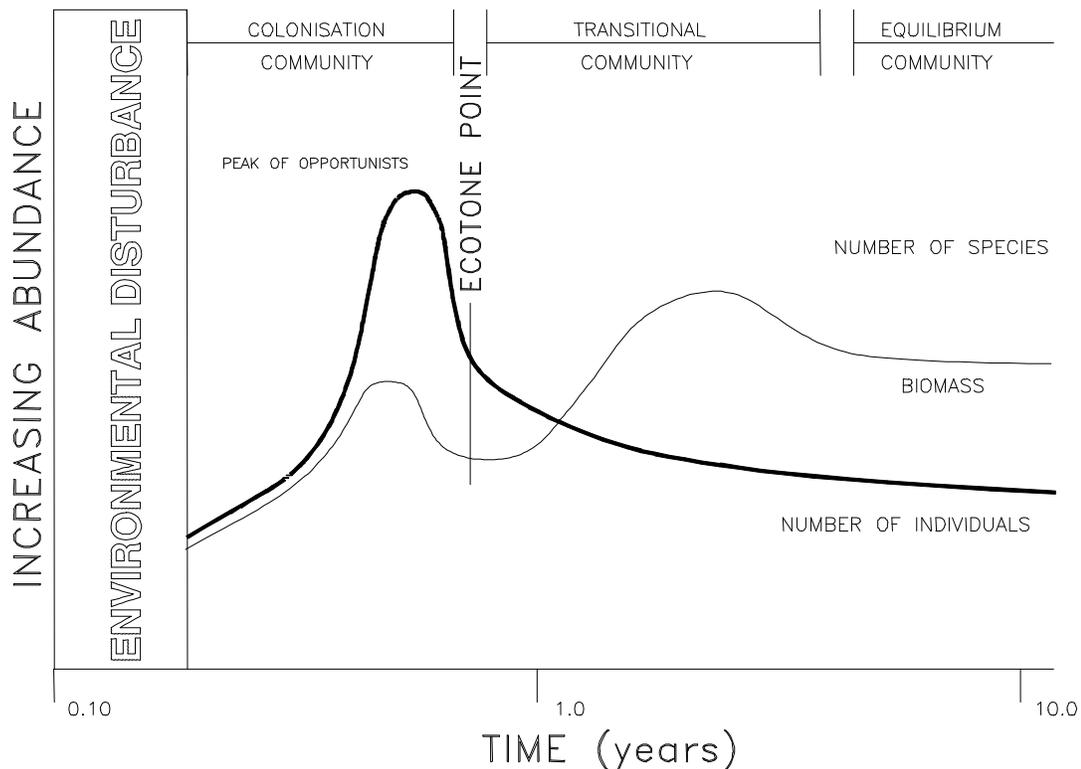


Figure 6.3 Schematic time series diagram showing a colonisation succession in a marine sediment following cessation of environmental disturbance. Initial colonisation is by “Opportunistic” species which reach a peak population density generally within 6 months of a new habitat becoming available for colonisation after the catastrophic mortality of the previous community. As the deposits are invaded by additional (larger) species, the population density of initial colonisers declines. This “Ecotone Point” marks the beginning of a “Transitional

Community” with high species diversity of a wide range of mixed **r**- and **K**- selected species. This period may last for 1-5 years 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 a final “Equilibrium Community” comprising larger, long-lived and slow growing species with complex biological interactions with one another (based on Pearson & Rosenberg, 1978)

Finally Group III species comprise larger slow-growing **K**-strategists 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 recolonisation process is shown in Figure 6.3. Inspection of Figure 6.2a shows that initially the sediments are almost devoid of benthic macrofauna.

The initial colonising species are few, but the number of individuals (population density) increase rapidly with time to a peak of (Group I) opportunist 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 species, population density and biomass is the same "Ecotone point" shown on the spatial gradient in Figure 6.3.

Prior to this, the community is characterised 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 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 6.3 indicates that colonisation 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**-strategists for the limited space available results in the elimination of some of the transitional colonisers (*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 colonisers, 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 6.3 (*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 which 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' in Figure 6.3. In the stable environmental conditions of deeper waters, the replacement of the initial colonisers in the transitional community following complex biological interactions between the **K**-strategists may take several years.

6.4 The Impact Of Dredging On Biological Resources

6.4.1 Sensitivity To Disturbance

The impact of disturbance by the drag head during marine aggregate dredging has been reviewed in Section 6.2. 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 which 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. There is little information on the depths of deposition of material derived from the screening process and hopper overflow from aggregates dredging, other than that predicted by modelling techniques (*see, for example, HR Wallingford, 1993*). Such field information that is available suggests that deposition depths are very small beyond 150 metres from the track of a dredger (*see* Gajewski & Uscinowicz, 1993). Further work is required.

A good deal of the apparent "recolonisation" 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, 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; Fr chet te *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 seawater, 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 south-west coast of England (Howell & Shelton, 1970).

In general, however, most recent studies of filter-feeders which 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 recolonisation).

Sampling is conventionally carried out by means of a grab which 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 more recent 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, however, complicated 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. The Hamon Grab (Plate 6.4.1) takes a scoop out of the seabed deposits, rather than relying on the closure of opposing jaws (*see* Holme & McIntyre, 1984).

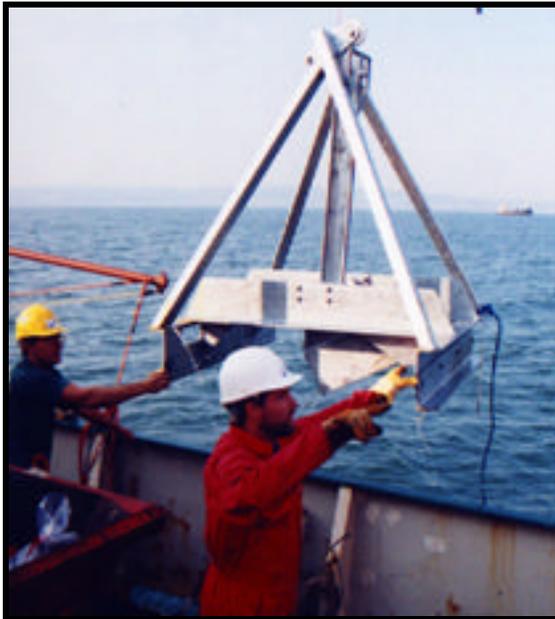


Plate 6.4.1 Recovery of Hamon Grab during quantitative investigation of benthic fauna (Coastline Surveys Ltd)

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. We have developed a modified Hamon Grab (Plate 6.4.1) with varying bucket geometry of square and cylindrical form for improved sampling

efficiency. These buckets are formed of stainless steel to reduce contamination when conducting pollutant investigations.

Studies such as Kenny & Rees (1994) and Newell & Seiderer (1996c) emphasise 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 6.4.1 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).

Further inspection of Figure 6.4.1 shows that the main population of the reef-building tubeworm, *Sabellaria spinulosa* ("called "Ross" by fishermen) occurs in the north-western part of the survey area, and corresponds with a localised patch of coarse stones and cobbles which give sufficient stability to support a rich reef community. This species may be predated upon by the pink shrimp (*Pandalus*) (*see* Warren, 1973) and is potentially important as a feeding ground for a variety of demersal fish species.

In contrast, the populations of the comb-worm (*Lagis* = *Pectinaria koreni*) occur in mobile muddy sands in the south-west 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 which requires conservation.

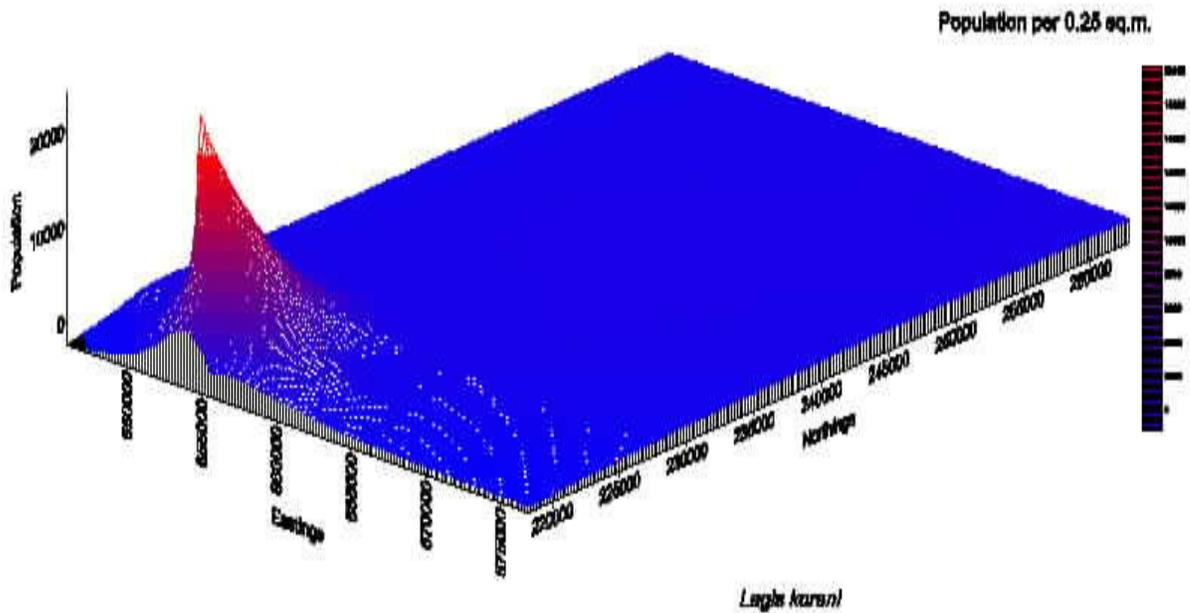
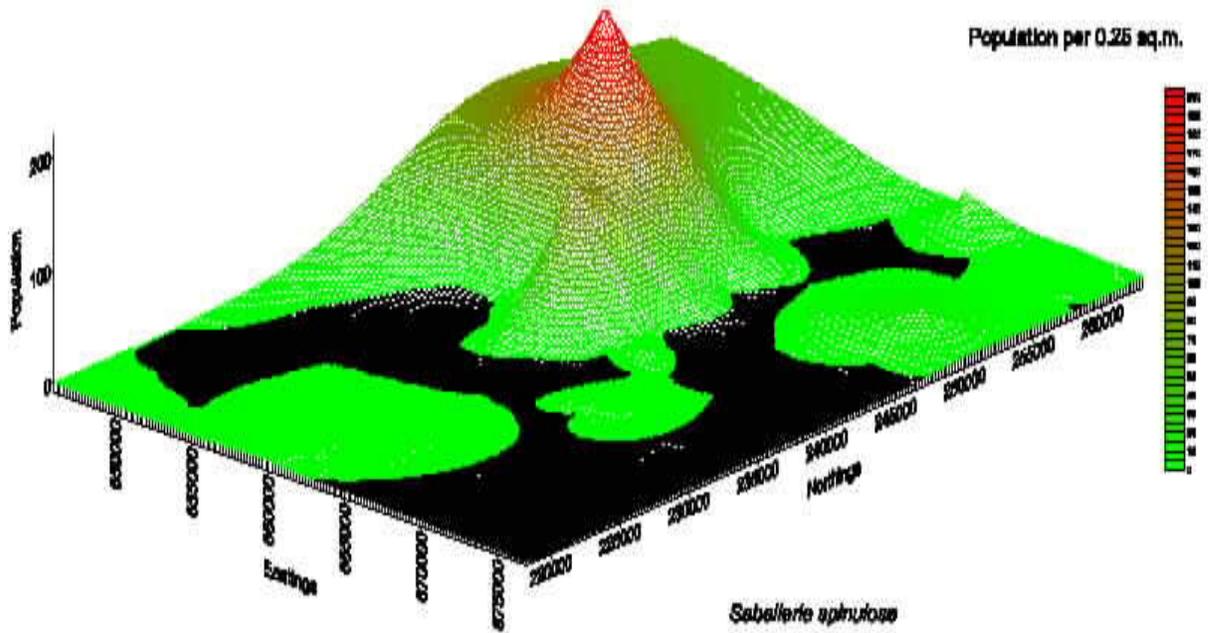


Figure 6.4.1 Schematic diagram of a survey area off Suffolk (southern North Sea) showing distribution of the comb worm (*Lagis koreni*) in fine deposits of the survey area, and that of the colonial “Ross” worm (*Sabellaria spinulosa*) in areas where coarse boulders provide a stable environment for the development of reef-forming species. Population density in numbers of individuals per 0.25m² sample obtained using a Hamon Grab (Newell & Seiderer, 1997b)

6.4.2 Impacts To Diversity & 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 recolonisation by passive transport of adult organisms (*see* Hall, 1994) and the intrinsic rates of reproduction, recolonisation and growth of the communities which normally inhabit 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 sand and gravel deposits in the Southern North Sea are summarised in Table 6.4.2. This shows that both maintenance and marine aggregate dredging can be expected to result in a 30-70% reduction of species diversity, a 40-95% reduction in the number of individuals, and a similar reduction in the biomass of benthic communities in the dredged area.

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, New York	Shallow Lagoon Mud	26	79	63-79	Kaplan <i>et al</i> , 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
Klaverbank, 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

Table 6.4.2 Table showing the impact of dredging on benthic community composition from various habitats.

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 spillways 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 which are deposited beyond a few hundred metres of the dredger trail. 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 Figure 6.4.2a. 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 occurrence of a similar phenomenon applicable to marine aggregate dredging

operations, was identified in Hitchcock & Dearnaley,

1995, and is outlined in Section 4.4.

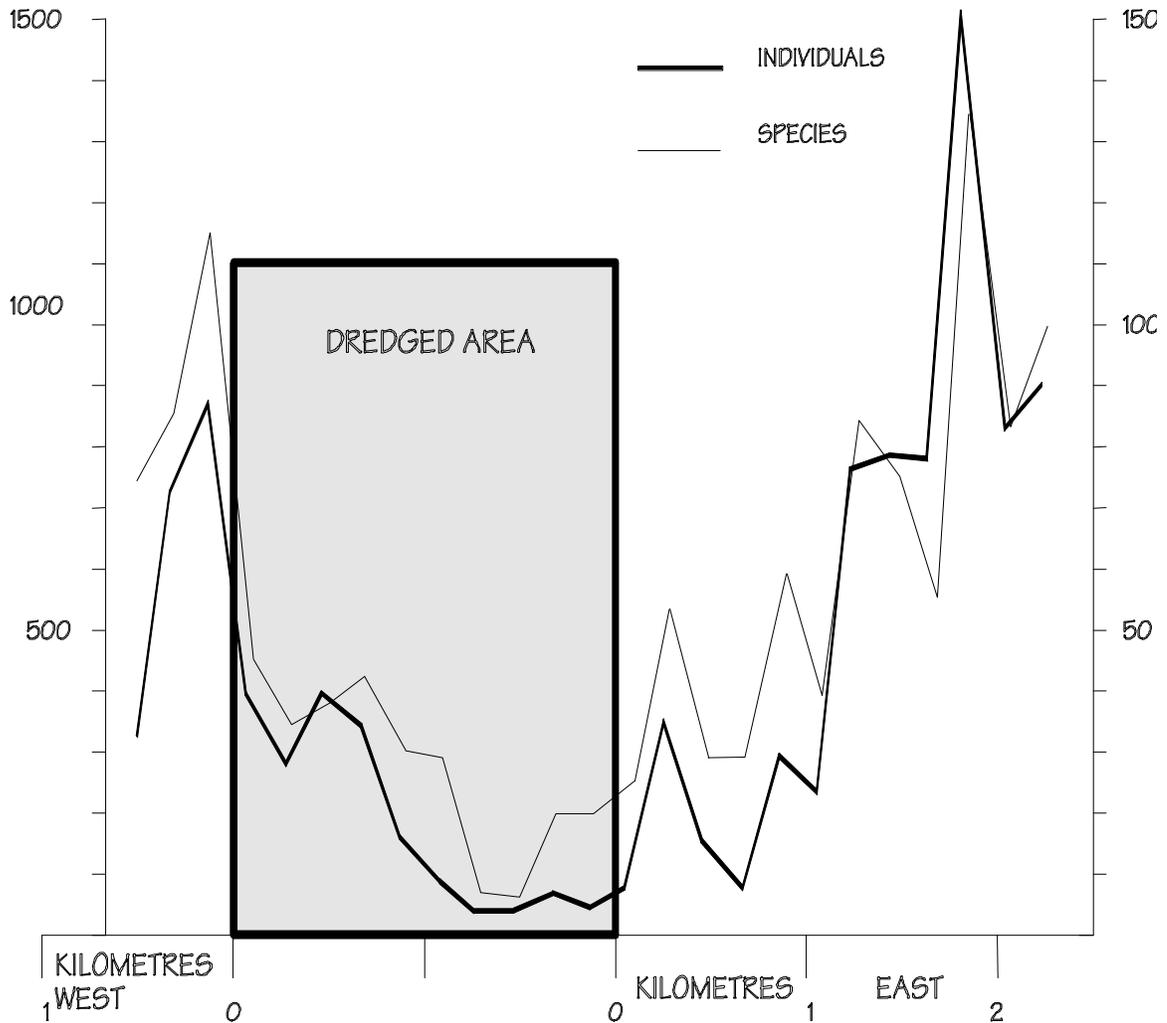


Figure 6.4.2a Diagram showing the number of individuals and species of benthos recorded in July 1982 on a transect crossing a dredged area on a sub-littoral sandbank in Moreton Bay, Australia. Note that species variety and population density increased rapidly outside the immediate boundaries of the dredged area (based on Poiner & Kennedy, 1982)

Other than this study, there is surprisingly little detailed information on the precise boundaries of biological impact surrounding areas which 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 which are apparently unaffected by dredging. The concept of whether a community has 'recovered' cannot therefore be taken to imply that marine community composition is necessarily stable over long time periods and that communities in dredged areas should therefore be expected to return to their pre-dredged species composition. By way of an example specific to potential aggregate extraction, variations in the population density and species composition of the large bivalve population recorded between 1988 & 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 Figure 6.4.2b. 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.

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 two years after cessation of dredging.

A short period of aggregate extraction was carried out in the study area on the Klaverbank 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 which evidently

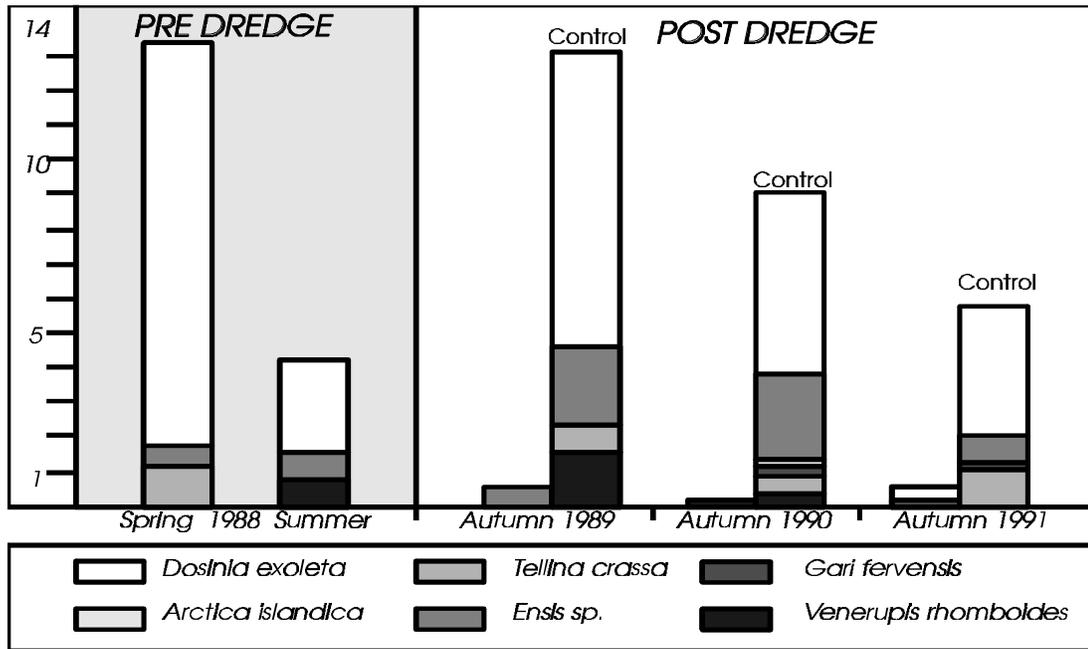


Figure 6.4.2b Diagram summarising the changes in population density and species composition of large bivalves on the Klaverbank in the Southern North Sea between 1988 and 1991. Pre-dredging values in 1988 show major seasonal changes in density and species composition. After dredging in the summer of 1989, large differences in population density and species composition emerged between dredges and control areas, and these differences persisted for at least two years (after van Moorsel, 1994)

6.5 The Process Of Recolonisation & 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, whether these are natural or induced by man, therefore use relatively sophisticated analytical techniques which 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 remit 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 & Green (1993), Clarke & Warwick (1994, and references cited therein).

Probably the most widely-used methods are detrended correspondence analysis (DECORANA), an ordination technique which 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 which characterise 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 which 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 to date of the impact of aggregate dredging on community composition and on the process of recolonisation 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 eight months in the first instance, although this was increased to two years in an extended study of the recolonisation process (Kenny & Rees, 1996).

Dredging occurred in April 1992 over a period of 4 days, during which the TSHD "Sand Harrier" removed a total of 52,000 tonnes of mixed aggregate from an area measuring 500-metres by 270-metres, an estimated 70% of the surface deposits down to an average depth of 0.3-metres 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 eight month period between March and December 1992.

The results from their study are summarised in Figure 6.5a. 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 recolonisation occurs even over this relatively short time interval.

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 individuals per m², compared with a relatively uniform invertebrate population density of 3,300 individuals per m² in the reference site. Again, the population density showed a significant increase in the 7-months after dredging had ceased.

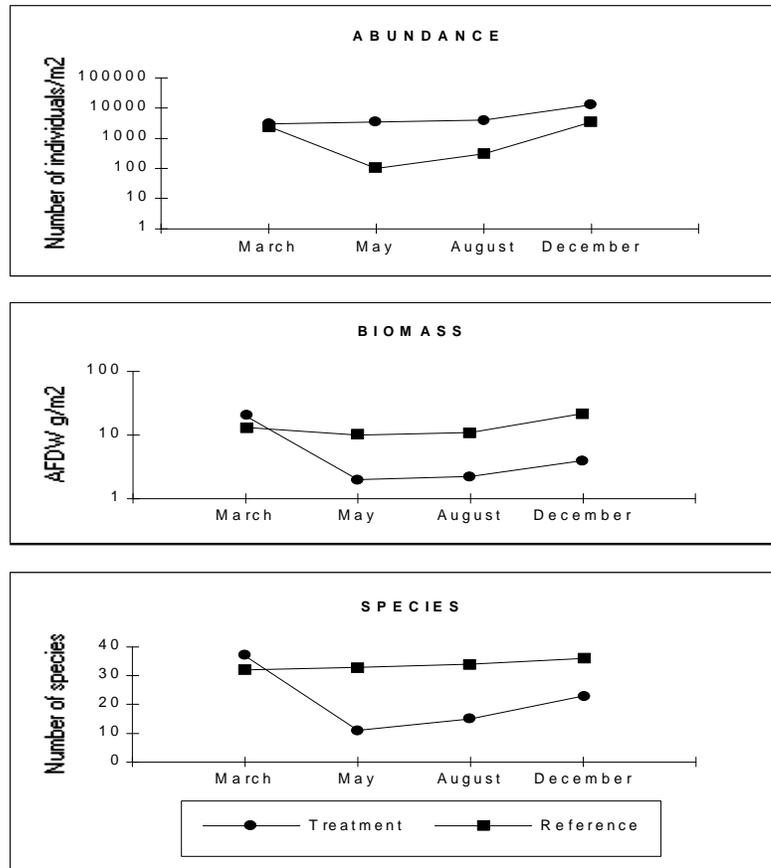


Figure 6.5a Graphs showing the mean values for the abundance of individuals (No. per m²) from five Hamon Grab samples each of 0.25m² taken in a dredged site and at a reference site. Dredging occurred in April, and samples were taken in the pre-dredged deposits in March 1992 and through to December 1992. Values for biomass are expressed as grams ash-free dry weight per m² from the five Hamon Grab samples. The average number of species in each of the five Hamon Grab samples is also shown. 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 recolonisation, 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 and Rees, 1994)

Inspection of Figure 6.5a shows that the high biomass of 23 grams (Ash-free dry weight) per m² was reduced to only 1 gram (AFDW) per m² 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 (number of individuals). This implies that recolonisation was initially by small individuals which then grew relatively slowly during the 7-months after dredging had ceased.

Figure 6.5b 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 Norfolk, and in the seven months after dredging (after Kenny & Rees, 1994). Their multivariate analysis of community structure prior to dredging and during in the months following dredging shows a number of important features of the

recolonisation process which highlight the general

principles of succession outlined in Figure 6.2.

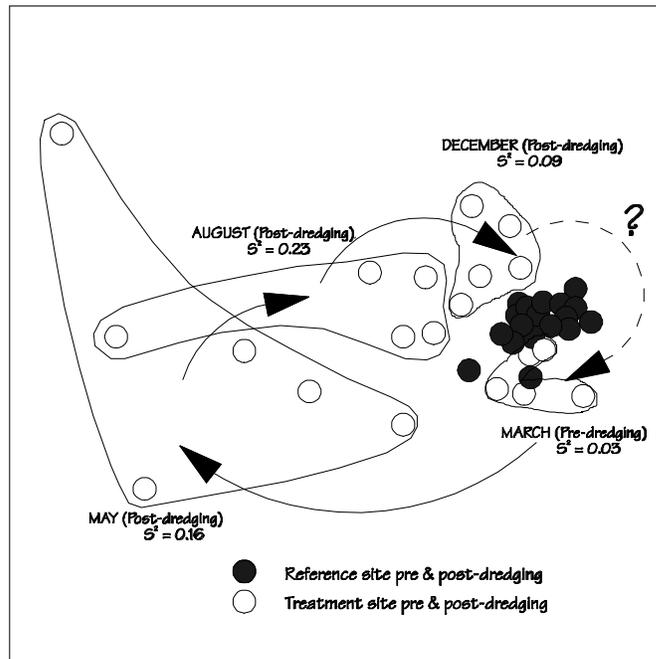


Figure 6.5b Two dimensional multi-dimensional scaling (MDS) ordination for the benthic communities in a survey area off Norfolk, UK, in March 1992 prior to dredging, and in May, August and December 1992 following trial dredging. Note that dredging of the experimental area resulted in an initial impact 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 monitoring period (after Kenny & Rees, 1994)

The first point which 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 distances between samples (circles) or sample groups (lines) indicate their biological dissimilarity.

The experimental area was again sampled in May 1992, one month after completion of dredging. At this stage it can be seen from Figure 6.5b 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. Second, 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 Figure 6.5b (see also Warwick & Clarke, 1993). This increased variance would be expected when some samples were taken from the dredged furrows themselves whilst 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 colonisation of the gravel deposits off the Norfolk coast was accomplished within the following 7-month period. Inspection of Figure 6.5b 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 recolonised 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 colonised the dredged area in the following 7-months.

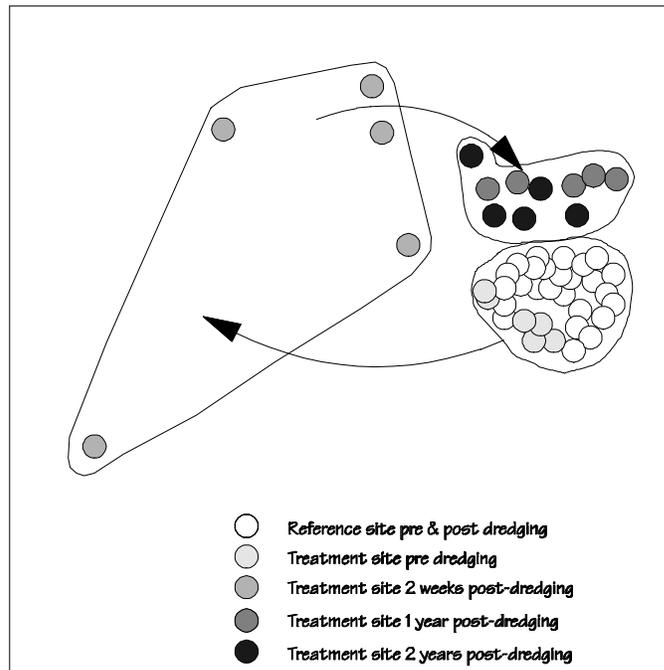


Figure 6.5c Two dimensional multidimensional scaling (MDS) ordination for the benthic communities in a survey area off Norfolk, UK, in March 1992, and for the following two years post-dredging. Note that despite the increasing similarity of the community in the dredged area to those in the surrounding sediments over the two year post-dredging period, recovery had not been fully accomplished even after two years (after Kenny & Rees, 1996)

The study was then extended to include data for a two-year 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 two-year 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 the work of Kenny and Rees that the community composition in their study dredged area was not restored even two years after dredging. Inspection of Figure 6.5c 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 two weeks after dredging is shown, as well as data collected in May 1993, one year post-dredging and in May 1994, two years after dredging.

It is apparent from Figure 6.5c that despite the significant recolonisation which 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 two years. Whether this reflects residual differences in the nature of the deposits following dredging, or the long time period required for establishment of the rarer components of the original community is not yet known.

The results which have been reviewed above thus show that the process of recolonisation involves two distinct phases; first recolonisation of species composition and population density by settlement of small individuals as larvae and juveniles; second a period of growth during which the biomass approaches that in the undisturbed deposits.

Inspection of Figure 6.5c 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 which

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 6.3).

6.6 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 6.2 and 6.3.

That is, communities which inhabit fine semi-liquid and disturbed muds comprise "opportunistic" *r*-selected species which have a high rate of recolonisation and which can reach high population densities within weeks or months of a catastrophic mortality. Conversely, communities which inhabit less disturbed deposits of deeper waters or coarse substrata have complex associations and are characterised by large slow-growing species which are selected for maximum competitive advantage in a habitat where space is already crowded. These large, slow-growing *K*-selected species recolonise only slowly following disturbance and may take several (or many) years for recovery of full species composition and biomass.

Table 6.6a 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 summarised in Table 6.6a 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* sp., *Ilyodrilus*, *Coelotanypus* sp. and *Procladius* to migrate through the surrounding deposits and to recolonise disturbed muds as adults. A similar recolonisation of disturbed deposits in dredged channels may also account for the relatively fast recolonisation of some muds and sands in nearshore waters, especially those where tidal currents may transport juveniles into the dredged area (*see* Hall, 1994).

Inspection of the recolonisation rates reported in the literature and summarised in Table 6.6a suggest that a period of 2-4 years 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-years in coarser deposits, including coral reef areas.

Interestingly, the data for areas in Tampa Bay, Florida, which had been dredged for oyster shell suggest that a period of as much as 10-years 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 colonising species which enable a faster recovery than might occur solely by larval settlement and growth (*see also* van Moorsel, 1993; 1994).

There are unfortunately insufficient data available to suggest guidelines on what proportion of the original community should be left intact to optimise recovery of equilibrium communities on the seabed. However, by the very nature of the sometimes patchy distribution of aggregate resources, dredging constraints such as water depth and contaminants such as clay pockets, it is becoming common practice for certain areas of seabed to be 'zoned off' and protected from dredging either throughout the life of the Licence, or on a rotational basis as extraction proceeds across a Licence Area. Such 'zoning' may have multiple mitigation benefits in not only providing potential recolonisation stocks but also providing recognised areas for other sea bed users. In the case of gold and diamond placer mining, the orebody is often quite patchy and in this case the least impact and fastest recovery has been achieved by intensive mining in patches, after which the extraction vessel moves well away to another high-grade patch leaving the mined area to recover undisturbed (D.V. Ellis, 1998, *pers.comm.*)

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 summarised in Table 6.6b which shows that even 16-years 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 recolonisation by small individuals which were in the process of growth towards the original biomass levels of the undisturbed deposits, a process which would clearly take several further years.

Locality	Habitat Type	Recovery Time	Source
James River, Virginia, USA	Freshwater semi-liquid muds	+/- 3 weeks	Diaz, 1994
Coos Bay, Oregon, USA	Disturbed muds	4 weeks	McCauley <i>et al</i> , 1977
Gulf of Cagliari, Sardinia	Channel muds	6 months	Pagliari <i>et al</i> , 1985
Mobile Bay, Alabama, USA	Channel muds	6 months	Clarke <i>et al</i> , 1990
Chesapeake Bay, USA	Muds and sands	18 months	Pfitzenmeyer, 1970
Goose Creak, Long Island, New York	Lagoon muds	> 11 months	Kaplan <i>et al</i> , 1975
Klaverbank, (Dutch Sector) North Sea	Sands and gravels	1-2 years (excl. bivalves)	van Moorsel, 1994
Dieppe, France	Sands and gravels	> 2 years	Desprez, 1992
Lowestoft, Norfolk, UK	Gravels	> 2 years	Kenny and Rees, 1994, 1996
Dutch Sector, North Sea coastal waters	Sands	3 years	de Groot, 1979, 1986
Tampa Bay, Florida	Oyster shell (complete defaunation)	> 4 years	US Army Corps of Engineers, 1974
Tampa Bay, Florida	Oyster shell (incomplete defaunation)	6 - 12 months	Conner and Simon, 1979
Boca Ciega Bay, Florida	Shells and sands	10 years	Taylor and Salomon, 1968
Beaufort Sea	Sands and gravels	12 years	Wright, 1977
Florida coastal waters	Coral reefs	> 7 years	Courtenay <i>et al</i> , 1972
Hawaii coastal waters	Coral reefs	> 5 years	Maragos, 1979

Table 6.6a Summary of rates of recovery of the benthic fauna following dredging in various habitats. Note that highly disturbed sediments in tidal freshwaters and estuaries which 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 which are dominated by long-lived components with complex biological interactions controlling community structure. Longevity and slow growth is also associated with slow recolonisation rates in sub-arctic seas. Examples have been arranged along a gradient from disturbed muds of freshwater-tidal estuarine conditions to stable reef assemblages.

In the tidal channels, both the rate of infill and recolonisation was related to the speed of currents. Here a partial recovery of 57% of the species and 67% of the biomass was recorded after 3-years in a tidal channel at Paesensrede, with greater recovery and in 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 recolonisation rates for the benthic community of estuarine muds, sands, gravels and reef areas have been superimposed onto a generalised colonisation succession in Figure 6.6, and allows some predictions to be made on the rates of recovery of deposits following dredging. The fine muds which characterise coastal embayments, estuaries and

lagoons are likely to be colonised by large populations of a relatively restricted variety of "Opportunistic" *r*-selected species which are capable of rapid colonisation within months of space being made available for colonisation and growth.

Area	Habitat	Time Interval Since Dredging	% Recovery	
			Species	Biomass
Terschelling sand	Tidal flat	16 years	0	0
Oosterbierum	Tidal watershed	4 years	85	39
Paesensrede	Tidal channel	3 years	57	67
Holwerderbalg	Tidal channel	2 years	64	100
Kikkertgat	Tidal channel	1 year	88	116

Table 6.6b Table showing the percentage recovery recorded in a variety of habitats in the estuarine Dutch Wadden Sea following dredging up to 15-years previously (based on van der Veer et al, 1985)

Because such deposits are subject to regular disturbance under natural conditions prior to dredging, the ecological succession recovers to the colonisation phase shown in Figure 6.6a, but does not proceed to the development of *K*-selected slow-growing species within the community. Recovery of the "normal" community in disturbed deposits such as muds can therefore be achieved within months of cessation of dredging, or disposal of spoils.

The natural communities of gravel and sand deposits, however, contain varying proportions of slow-growing, *K*-selected equilibrium species, depending amongst 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 recolonise 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 a period of 2-3 years 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 characterises 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 6.6, long-lived components such as molluscs and larger crustaceans in nearshore 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 (Riesen & Reise, 1982; Reise, 1982; Reise & Schubert, 1987).

Finally, the community recovery curve for reef communities indicates that a period of 8-10 years may be required for the long process of establishment and growth of the long-lived and slow-growing *K*-selected species and for the development of the biological interactions which are familiar to those who have observed the immense diversity and complexity of life on undisturbed reef structures.

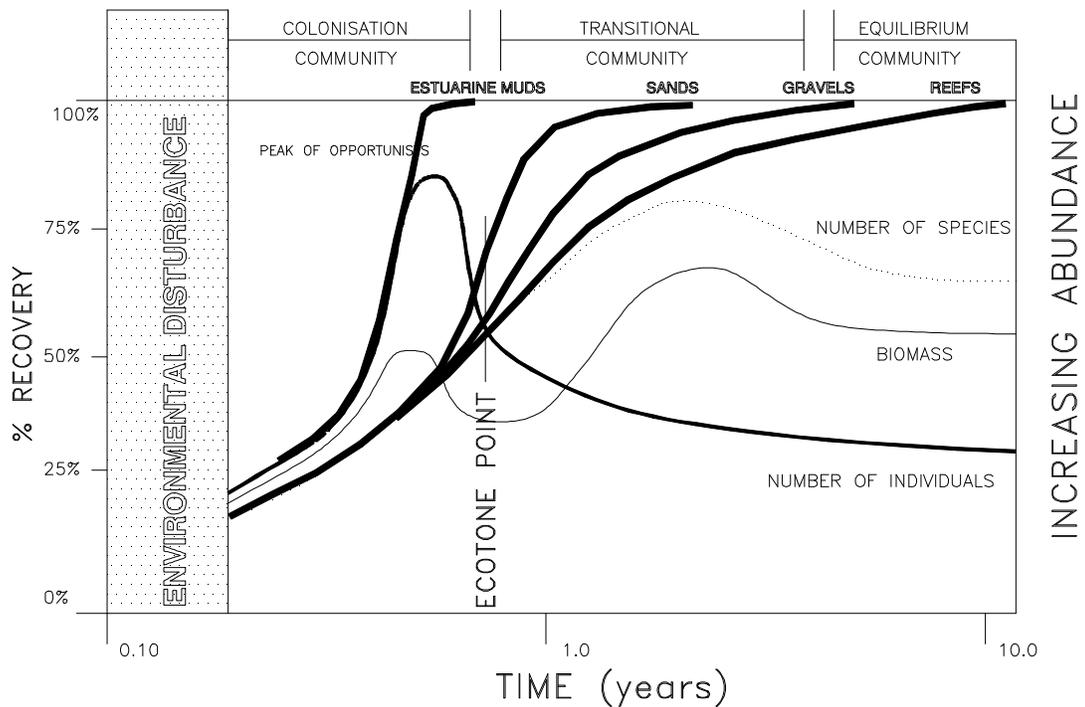


Figure 6.6 Schematic diagram showing the likely recolonisation rates for the benthic community of estuarine muds, sands, and reef areas. The curves for recovery have been superimposed onto a generalised colonisation succession and allows some predictions to be made on the rates of recovery of deposits following dredging. Note that the fine muds which characterise coastal embayments, estuaries and lagoons are likely to be recolonised by a relatively restricted variety of “Opportunistic” *r*- selected species within months of space being made available for recolonisation and growth. Because such deposits are subject to regular disturbance, the succession recovers to the colonisation phase, but does not proceed to the development of long-lived slow growing *K*- selected species. The natural communities of sands and gravels, 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 years after cessation of dredging. Finally, the recovery curve for reef communities indicates that a period of 8-10 years 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.

6.7 Community Composition and Seabed Stability

The influence of sediment composition in controlling the composition of communities of animals which live on the sea bed has been widely recognised 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 seabed undergo a complex process of consolidation or "armouring" which allows the establishment of communities which are more typical of rocks and reefs and which reflects 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 which 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 which affect the benthic organisms which 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 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 which 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 mobilisation 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 recolonisation by communities which were similar to those which 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 analysed 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 analysed 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 only 30% of the variability of the biological component, leaving approximately 70% 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 time 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 which are comparable with those which 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 stabilisation. 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 colonisation of recently sedimented material in unconsolidated deposits after the cessation of dredging.

6.8 Conclusions

At the beginning of this section, 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 seabed 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* Figure 6.1). 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 nearshore waters where primary

production by larger macrophytes and seagrasses living on the seabed 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 therefore concentrated on the nature of benthic communities, their susceptibility to disturbance by dredging and land reclamation works, and on the evidence which 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 nearshore habitats are characterised by large populations of a relatively restricted variety of species which are well-adapted to exploit space which has become newly available by episodic catastrophic

mortality. Such species are generally small, often mobile ("Opportunistic") 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 their population characteristics allow a rapid recovery of the initial community structure in deposits which are naturally subjected to high levels of environmental disturbance. It is therefore not surprising to find that there are frequent reports in the literature of community recovery times which range from a few weeks to several months for disturbed deposits such as semi-liquid muds in tidal freshwaters, estuaries, lagoons and dredged channels (see Table 6.6a & Figure 6.6).

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 colonisation 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 species live longer, they tend to have wider limits of physiological tolerance which allows them to survive those variations in environmental conditions which occur in their habitat.

Many have active site selection phases which 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 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 which would not otherwise occur, and also inhibit other potential competitors for space.

Biological interactions between the components of "Equilibrium Communities" which are characteristic of stable substrata thus lead to the development of complex communities which 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 which are reported for recovery of community structure (Table 6.6a).

Knowledge of the components which comprise the benthic community on the sea bed, whether these are *r*-selected or *K*-selected, thus gives important information not only on key resources which may require protection, but on the likely rate of recovery following dredging. Inspection of the schematic colonisation succession shown in Figure 6.6 suggests that a recovery time of 6-8 months is characteristic for many estuarine muds whilst sands and gravels may take from 2-3 years depending on the proportion of sand and the local disturbance by waves and currents. As the deposits become coarser, estimates of 5-10 years are probably realistic for the development of the complex biological associations between the slow-growing components of equilibrium communities characteristic of reef structures.

Our information presented herein suggests that processes associated with compaction and stabilisation 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 colonisation of recently sedimented and unconsolidated material following cessation of dredging. Further work to identify the rheological and structural properties of the deposited sediments is required to further consolidate these observations.