Chapter 14

Disturbance, Pollution, and Climate Change

CHAPTER SUMMARY

Human activities around the globe have been linked to a diverse range of ecological changes in marine ecosystems. Many of these activities involve exploitation of biological and mineral resources, while other forms of interference are linked to industrial and agricultural discharges into river basins. Often, the changes in marine systems that result from these activities are catastrophic, while others are subtler and only become apparent after many years. In order to assess the relative ecological importance of human interference, it is necessary to understand how marine communities respond to ecological disturbance and environmental change. It is also imperative to appreciate the temporal and spatial scale at which these processes are relevant. Ascertaining whether ecological changes have occurred in response to disturbance or environmental changes requires a rigorous approach to experimental design and monitoring and a careful consideration of the issue of statistical power to detect change.

14.1 Introduction

Human activities are having an ever-increasing influence on global marine ecosystems. While the debate continues about the causes and significance of present global warming, few would now deny that the world's climate is changing more dramatically than at any time in the last century, and scientists are beginning to debate the proximity of the next ice age (King 2004, Weaver & Hillaire-Marcel 2004). Although the subtle effects of changes in weather patterns might not cause immediate alarm, significant events such as the break-up of a large area of the Antarctic ice shelf in 1995 made global media headlines (Vaughan & Doake 1996). With the exception of desert and polar regions, coastal areas are densely populated, particularly at the mouths of estuaries and

● Global climate is currently changing faster than at any time in the last 100 years and has already caused dramatic changes in polar ice cover.

sheltered bays that provide safe anchorages and convenient access inland. In contrast to the vast tracts of open oceanic waters, the coastal margins of the world's land masses are the focus of intensive human activities.

Human activities modify the marine environment both through the removal of biomass and habitats and via the addition of contaminants and physical structures. Marine biological resources are heavily exploited for consumption and economic gain, and habitats often altered incidentally. Rivers convey terrestrially derived material loaded with sewage, agricultural, and industrial pollutants onto the continental shelf. Understanding the ecological responses to these human activities requires an appreciation of both watershed and marine environmental processes. Marine traffic, oil and gas extraction, and dredging are all concentrated in shelf areas. Further offshore in the mid-ocean, direct human influences are limited to oceanic crossing by marine vessels and fishing activities. The decreasing influence of human activity with distance from the coast is related to the physical limitations imposed by the environment (wave height and depth) and the logistics of getting there. Nevertheless, these activities are expanding further offshore as coastal resources are depleted and technological developments enable exploitation of more extreme environments.

Understanding the ecological consequences of human activities in the coastal zone requires knowledge of land–ocean interactions that include watershed and marine environmental processes.

Extreme environments offshore in deeper water are made more accessible with technological advances that reduce costs, increase profitability, and increase safety.

In this chapter, the role of human activities in causing ecological disturbance is considered, and their significance is gauged against the scale and frequency of natural sources of disturbance. To understand the ecological importance of human activities in the marine environment, we need to be able to detect changes in measured ecological characteristics using appropriate observational or experimental techniques. An understanding of the scale at which ecological processes operate should underpin the ultimate selection of metrics chosen for study (e.g. diversity, abundance, biomass). We discuss the relevance of a selection of measures and the importance of appropriate experimental design. The wide range of human impacts on the marine environment dictates that we are selective in our coverage. Issues surrounding aquaculture and fisheries were addressed in previous chapters (Chapter 12 and 13).

14.2 Ecological Role of Disturbance

Earlier chapters have discussed the spatial and temporal scale at which various small and large-scale processes operate in the marine ecosystem (Chapter 1). These dynamic processes affect ecological processes and the structure of communities and habitats, such that they are in a continuous process of change. These natural fluctuations form the backdrop against which the relative importance of human activities should be assessed. Almost any human intervention in the marine environment

Most human activities in the marine environment cause some form of ecological disturbance. These occur against a background of natural disturbances that occur at a variety of spatial and temporal scales.

whether positive (e.g. habitat restoration) or negative (e.g. dumping of waste), leads to some measure of ecological disturbance. Pickett and White (1985) defined disturbance as 'any discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substratum availability, or the physical environment'.

14.2.1 Sources of disturbance

Disturbances act at different scales and frequencies. Changes in sea level, ocean temperature, and water circulation modify habitats and their associated fauna over large areas and usually over long timescales (>20 years). Natural phenomena such as cyclones and hurricanes have regional impacts on a seasonal basis and affect a wide range of marine habitats to different extents (Hall 1994). Periodic outbreaks of ecosystem engineering organisms such as starfishes and sea urchins can lead to periods of prolonged habitat modification (Chapter 7). Rising ocean temperatures may facilitate the proliferation of non-indigenous species, which can alter the existing community assemblage through the process of competition for space and other resources. Human agents of change vary from the direct effects of an oil spill, habitat damage by bottom fishing on the seabed, eutrophication of sea basins, discharge of toxic substances, to trampling across the seashore. Whatever the source of disturbance, it is a fundamental process that contributes to the maintenance of diversity in all ecosystems.

● Disturbance is one of the most important ecological processes in the maintenance of diversity by opening up resources for colonization by opportunistic species.

14.2.2 Scale of disturbance

There is an intimate link between the spatial and temporal scale at which environmental change occurs. At the smallest scales (μm) chemical reactions between sediment particles, flocculation of organic matter in the water column, viral and bacterial processes occur on a very short timescale. At the next scale up, minute-by-minute disturbances occur as the result of the feeding activities of macrofauna in sedimentary habitats and as a result of their movements upon or through the sediment, creating burrows or furrows in their wake. At slightly larger scales up to one metre, larger-scale bioturbating processes become important (e.g. faecal mound formation, burrow chamber excavation) and the feeding activities of megafauna such as birds, fishes, and crabs become important agents of disturbance that occur every tidal cycle. Thus the seabed is a mosaic of patches in different stages of alteration and recolonization (Grassle & Saunders 1973). Each modification on its own is relatively small compared to the total habitat resource, however the summed total of these effects is such that they have a significant role in determining the characteristics of that habitat (Fig. 14.1).

● Small-scale physical disturbances may seem relatively insignificant when considered on their own, but their additive effects may influence wider-scale community structure.

Fig. 14.1 The physical disturbance created by individual fauna such as estuarine fiddler crabs, may be small in scale (cm), but the additive effects of the disturbance created by the entire population within an estuary can lead to the complete reworking of the surface of these sediments in just one low tide (Photograph: M.J. Kaiser).

The additive effects of small-scale natural disturbances are well illustrated by the effects of the burrowing and feeding activities of the soldier crab (*Mictyris platycheles*) in south-eastern Tasmania. As soldier crabs forage across intertidal mudflats they create intensely disturbed areas of sediment in discrete patches interspersed with undisturbed areas. These small-scale disturbances have the most profound effect on the meiofaunal assemblage, such that nematodes, species richness, species diversity, and evenness were significantly reduced in disturbed as opposed to undisturbed areas, although total abundance was unaffected. Changes in community structure were subtle and resulted from an overall change in the balance of relative abundances of many species, rather than from changes in a few dominant species (Warwick et al. 1990). Hall et al. (1993) undertook a similar study in a Scottish sea loch. They hypothesized that the pit-digging activities of a large predatory crab (*Cancer pagurus*) would generate significant community changes in the soft-sediment assemblage. Yet despite a carefully designed experiment no ecological effects were apparent. They concluded that larger-scale natural disturbance processes such as wave disturbance, masked any subtle effects that pit digging by crabs might generate.

● Even the additive effects of multiple small-scale disturbances may be masked by much larger-scale environmental disturbances such as storm-induced wave perturbation.

14.2.3 Recovery rate

The scale of physical disturbances created by micro-, meio-, and macrofauna are relatively small and the recovery time of the habitat in response to those disturbances is rapid, occurring within seconds to weeks (Table 14.1). Recovery times in response to disturbance increase with scale. At smaller scales (<1 m) recolonization occurs through active movement and passive transport of adults into the disturbed area. At larger scales of disturbance typical patterns of recolonization occur. For example, physical disturbance of the seabed by towed bottom

Table 14.1 The relationship between the spatial scale of a disturbance and the time taken for recovery to occur. Recovery is defined as the point at which the biological community within the disturbed area is no longer significantly different from that in other similar and proximate areas that were not the subject of the disturbance (control areas).

Spatial scale	Temporal scale (frequency)	Processes
µm	milliseconds–seconds	chemical reactions, virus and bacterial driven processes
mm–cm	seconds–minutes	meiofaunal processes, macrofaunal sediment reworking, predation, herbivory, faecal production
0.1 m–1 m	minutes–days	bioturbation, diatom mat formation, megafaunal disturbances (e. g. feeding pit excarvation), precipitation of minerals (calcium carbonate mounds)
1 m–100 m	days–months	Recolonisation and redistribution of small microbiota, biomass and population fluctuations, sediments resuspension and settlement, bedload transport tidal scour and currents
100–10 000 m	months–years	Hurricane and strong events, iceberg scouring, submarine landslide events, siesmic activity, recolonisation of large macrobiota
>10 000 m	years–decades	volcanic and seismic activity, anoxic events, submarine landslides, global warming, El Niño events, coral bleaching, global warming, recolonization of large slow-growing macrobiota

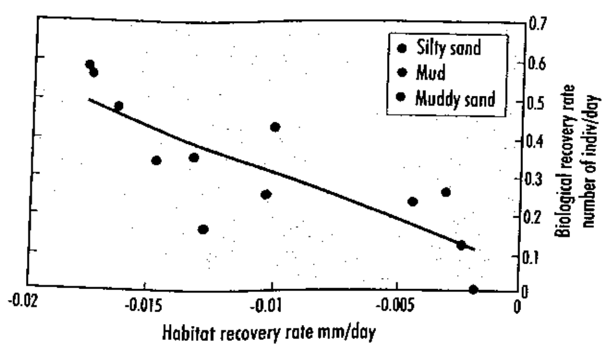


Fig. 14.2 The relationship between the rate at which sedimentary habitats recover (quantified as how quickly standard pits dug into the sediment filled in (decrease in depth of pit cm d^{-1})) and the rate of biological community recovery (measured in terms of the rate at which the difference between control and dug plots decreased in terms of the total number of individual organisms inhabiting the sediment). Muddy sand treatments filled in more slowly than any other treatment and consequently recolonization was much slower.

fishing gear disrupts the surface of the seabed and kills, damages, or digs out many of the infauna and epifauna in the path of the trawl. The immediate post-trawling effects are typified by a reduction in resident species number and abundance (Kaiser et al. 2002). However this is

● Recolonization after disturbance events follows a typical pattern, with short-term immigration of active and scavenging fauna followed by longer-term (months to years) recolonization through larval recruitment. The scavenging fauna disperse within a few days after the biota killed by the disturbance have been consumed.

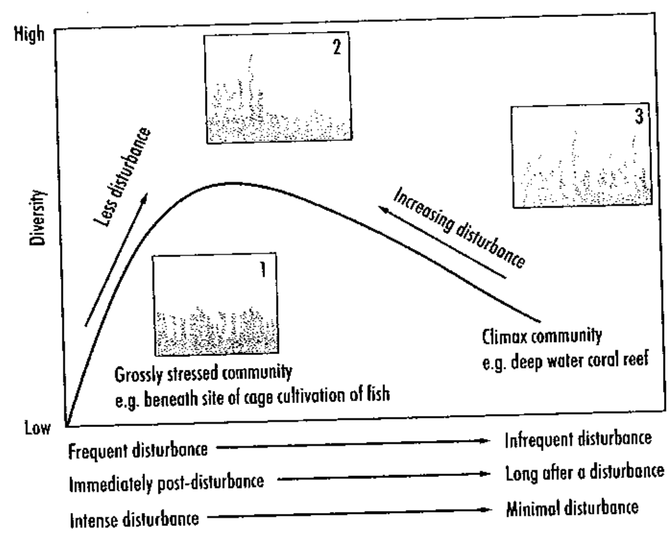
always accompanied by a short-term (2 to 3 days) influx of scavenging species, which are attracted to the carrion generated in the disturbed area (Ramsay et al. 1997). Thereafter, the recolonization rate is linked to the supply of larvae to the disturbed area and habitat stability. If disturbance involves disruption of the habitat, a process of habitat restoration is required before recolonization of the associated biota occurs in full (Fig. 14.2). For those habitats formed by living biota with slow growth rates and irregular recruitment of larvae recovery times are measured in years or even decades (Collie et al. 2000).

14.2.3 Intermediate disturbance hypothesis

The responses of species diversity and assemblage characteristics to disturbance can be described by an important ecological paradigm – the

Box 14.1 Intermediate disturbance hypothesis

Although not the first, Connell's (1978) application of the intermediate disturbance hypothesis (IDH) is perhaps the most quoted. He used the IDH to explain changes in species diversity under different scenarios of disturbance in two habitats of high diversity, tropical rainforests and coral reefs (Fig. 14.1). At low levels of disturbance, succession processes eventually lead to a climax community dominated by relatively few species of large biomass (3). When this state is disturbed, e.g. by a tree fall in a rain forest, or the damage of coral reef structures by a passing hurricane, space becomes available which permits colonization by opportunistic species (2). The modified assemblage now has a combination of climax and opportunistic species, thereby increasing diversity. As the severity or frequency of disturbance increases only a few opportunistic species persist resulting in an assemblage of low diversity (1).



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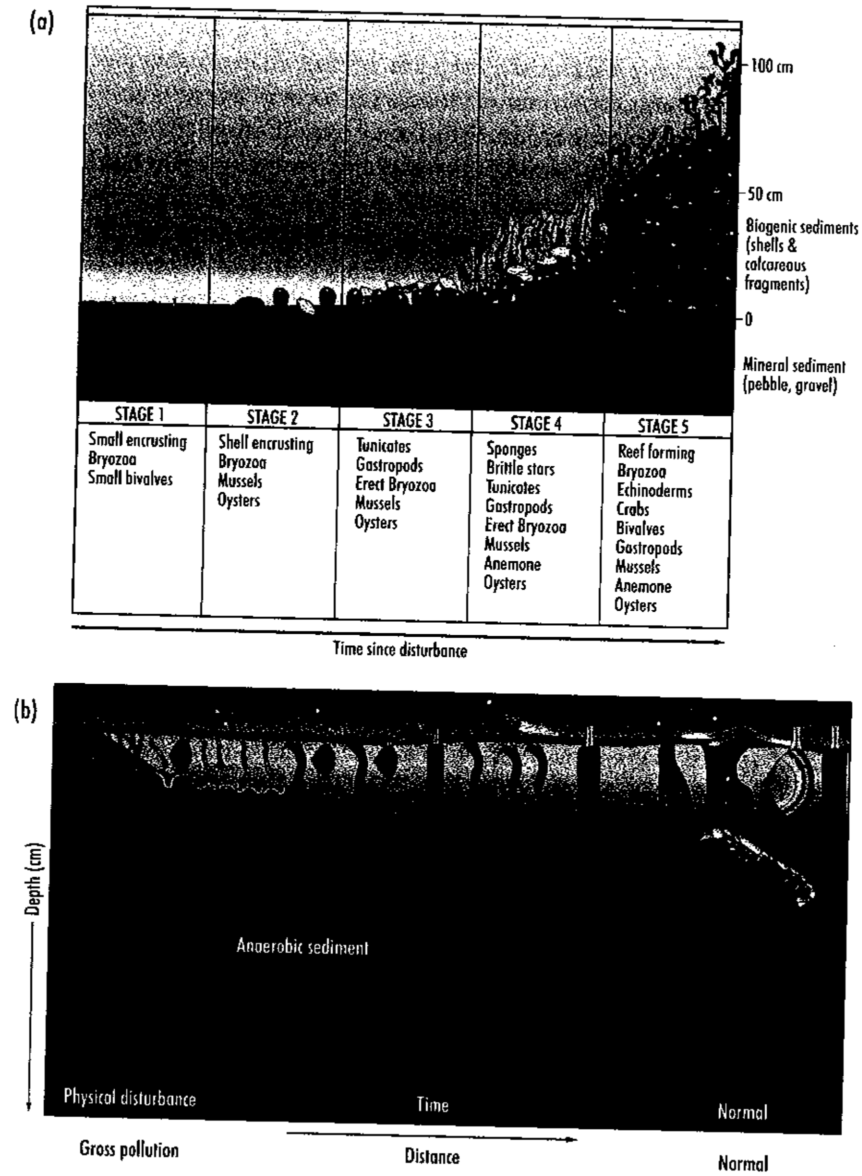

Fig. 14.3. The Pearson and Rosenberg (1978) model of the response of seabed communities to physical disturbance and a gradient of gross pollution (a + b). Towards the left of the diagrams the benthic communities experience the greatest intensity of stress and the community is characterized by low diversity and typically is dominated by only oligochaete and polychaete worms and mats of bacteria. (b) Only the top few mm of the sediment is oxygenated. As stress decreases to the right of the diagram, so the body size and longevity of the fauna increases (typified by large burrowing bivalves and echinoderms). The depth of oxygenated sediments gradually increases with decreasing stress levels and increasing bioturbatory activity. The same gradient occurs in response to organic enrichment usually along a gradient of distance from the source of input. The same model can be applied to the response of epifaunal communities (a) to dredging as in this example of macrofaunal succession on the seafloor of the Foveaux Strait, New Zealand (Cranfield et al. 2004). (Figure (a) reproduced from Journal of Sea Research, Vol. 52, 2004 p. 109–126 with permission from Elsevier.)

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intermediate disturbance hypothesis (Box 14.1). Rhoads (1974) and Pearson and Rosenberg's (1978) models of the changes that occur in macrofaunal community structure along a gradient of disturbance follow the principles of the intermediate disturbance hypothesis (Fig. 14.3). Their models of macrofaunal community responses to physical disturbance and organic enrichment are amply supported by direct observations. The Pearson and Rosenberg (1978) model is well illustrated by the situation that occurs beneath suspended fish farm cages in low energy environments such as fjords. The fallout of uneaten food pellets and faeces that descend to the seabed results in an organically enriched environment, which in extreme cases is suitable only for the growth of bacteria or a few species of small opportunistic taxa (oligochaetes and *Capitella* spp.) that are highly tolerant of low oxygen conditions (Chapter 13). Such a community has very low diversity. At increasing distances from the area beneath the cage the loading of organic enrichment decreases, dissolved oxygen levels increase, and environmental stress is reduced, such that the opportunistic species are gradually replaced by larger-body-sized biota such as echinoderms (sea urchins and brittlestars) and bivalves, many of which perform important **ecological functions** such as mixing sediments that enhance oxygenation and microbial production (Fig. 14.3).

● The intermediate disturbance hypothesis describes the response of community diversity across gradients of disturbance. Highest diversity occurs at intermediate levels of disturbance that result in assemblages with both opportunistic and climax species.

14.3 Measuring the Effects of Human Activities

In order to study the effects of human interference on the ecology of marine communities and their components, we need to be able quantify appropriate measures or metrics that respond in some way to the human activity of interest. These metrics must be amenable to quantification in a reliable and consistent manner. Processes that respond over decadal timescales require long-term strategies of sampling if these are to be quantified. The metrics to be measured may be **univariate responses** of single factors such as change in abundance or species richness; **distributional techniques** that measure the distribution of individuals or biomass among an assemblage of organisms; or **community responses** that measure the responses of more than one species, known as **multivariate responses**.

14.3.1 Univariate measures

When investigating the response of a single species to an environmental gradient or disturbance treatment in an experiment, it is typical to measure changes in abundance, biomass. or some other physiological, morphological, or behavioural response. However, measuring the response of a single species takes no account of the effect that changes in the abundance or body-size class distribution of that species might have on community structure. This is particularly important when the trophic status of taxa changes through their life history. This is typified by the relationship between cod (*Gadus morhua*) and whiting (*Merlangius merlangus*). Small cod are consumed by adult whiting, which in turn are eaten by adult cod. Single species that are the focus of long-term studies tend to be those that are easily identified and counted *in situ* (e.g. sea-birds, pinnipeds, limpets, barnacles), or show strong morphometric changes in response to a stress agent (e.g. development of male characteristics in female whelks). Often, such taxa are considered to be indicators of environmental change that may (or may not) be reflected in other parts of the ecosystem (Fig. 14.4).

● Measures of changes in body-size class describe the relative abundance of small and large individuals in a population. This is particularly important when trophic status changes with increasing body size e.g. the transition from juveniles considered as prey, to adults that become predators.

In most cases, the effects of human activities are not species-specific and have variable effects on different components of an assemblage of organisms. For example, release of a contaminant from a point source of discharge (e.g. chemical discharges from an outfall pipe) is likely to have greater cumulative effects on a range of sedentary fauna and less severe effects on mobile fauna that are transient within the area affected by the discharge. In the case of direct physical disturbance (e.g. by bottom

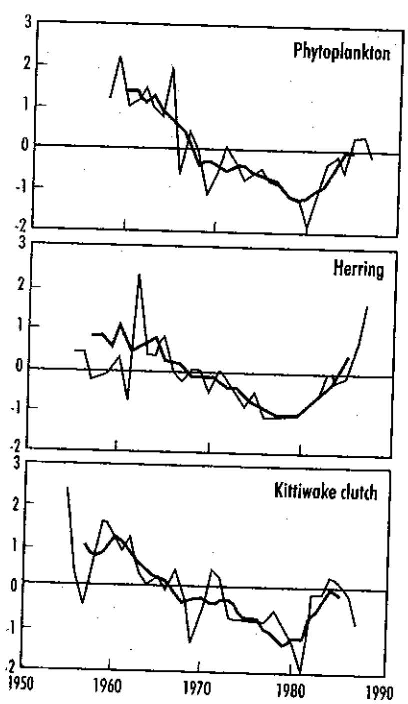
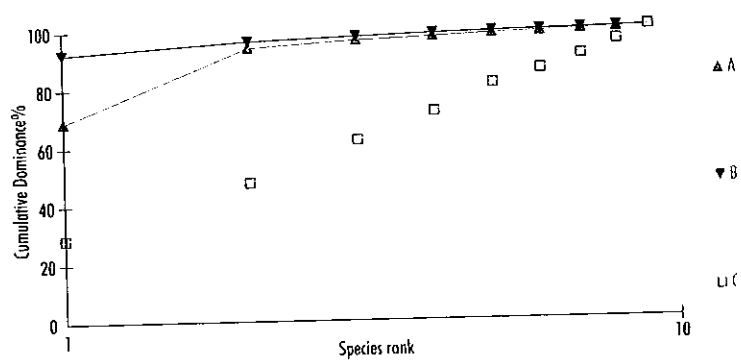


Fig. 14.4 Parallel long-term trends in phytoplankton biomass, herring abundance, and kittiwake clutch size driven by patterns in weather in the North Sea. The y axis is a relative scale showing the degree of change around a mean value (0 = no change from mean). (From Aebischer et al. 1990.)

Box 14.2 Univariate and distributional community characteristics

		Sample		
		A	B	C
Abra alba		150	2	1
Acanthocardia echinata		5	1	3
Acteon tornatilis		1	1	1
Chamelea gallina		1	3	6
Corbula gibba		1	1	4
Donax vittatus		55	5	2
Dosinia sp.		2	1	1
Ensis ensis		1	15	2
Fabulina fabula		2	350	1
Total number of species	S	**9**	**9**	**9**
Total number of individuals	N	**218**	**379**	**21**
Species richness (Margalef)	$d = (S-1)/\mathrm{Log}\,N$	**1.49**	**1.35**	**2.63**
Pielou's evenness J	$J' = H'/\mathrm{Log}\,S$	**0.4**	**0.18**	**0.9**

This example shows three samples of a benthic assemblage with only the abundance of the bivalve taxa recorded. The distribution of individuals within each of samples A, B, and C is very different. In samples A and B, *Abra alba* and *Fabulina fabula* are numerically dominant, while in sample C the individuals are more evenly distributed among the species. Below the community data are shown two univariate metrics of these samples, the total number of species, and the total number of individuals. These metrics are then used to calculate a diversity index (Margalef) or an index of evenness (Pielou) derived from the Shannon Wiener diversity index (H′) and the total number of species (S). Sample C has the highest species richness as the same number of species are distributed among a smaller number of individuals. Accordingly, sample C also has the highest index of evenness (measured on a scale between 0 and 1). The same community data can be displayed graphically in a *k*-dominance plot, which shows the cumulative percentage dominance of each species in the assemblage ranked according to dominance.



trawling or aggregate dredging) certain fauna may be more vulnerable than other constituents of the biological assemblage. The resultant community changes are often subtle and might not be revealed by the examination of the response of individual species. In these cases it is more usual to take multi-species samples that are representative of the biological assemblage. Within each sample, counts of multiple species can be collapsed into a single coefficient such as a diversity index. For example, some diversity indices give a measure of the extent to which a relatively small number of species account for a large proportion of the total number or biomass of individual organisms within a sample (Box 14.2, see also Chapter 1).

● Diversity indices summarize counts of multiple species within a sample in a single coefficient. Different indices are sensitive to either changes in rare or dominant species within the assemblage.

While diversity indices are a useful tool for detecting major community changes that occur as a result of gross disturbance or along steep environmental gradients, they can be insensitive if the changes are manifested as species replacements with similar levels of abundance. In addition, if diversity indices are to be used effectively it is essential that the scale at which community diversity metrics are measured is appropriate for the scale at which an agent of change may operate. A good example of this scale effect is the response of diversity metrics to bottom fishing disturbance. Fishing gears disturb large areas of the seabed. Even so, highly abundant small-body-sized animals (e.g. polychaete worms) are relatively resilient to physical disturbance, while less common larger fauna such as burrowing sea urchins and bivalves are highly vulnerable to disturbance. Typically, marine ecologists collect many small samples of the seabed to try and improve the chance of detecting responses to environmental change (14.3.5). However Kaiser (2003) found that it was not possible to detect the effects of fishing on community diversity when small-scale ($0.1\,m^2$) samples were collected. Only when larger-scale samples were considered, which sampled more effectively the less common and more vulnerable species, did the effects of fishing on diversity emerge (Fig. 14.5). This example underlines how sampling at the wrong scale can lead to erroneous conclusions about changes in community diversity!

● While collecting many small samples may increase statistical power, fewer larger samples reveal changes in rare species more effectively.

The use of diversity indices can fail to detect large changes or differences in the composition of biological communities. For example, two communities may have the same diversity index even though their species compositions are entirely different. Alternatively, the abundance of one species might decrease while another increases by an equivalent amount in response to environmental stress. These limitations have been overcome to an extent by the introduction of new measures such as taxonomic diversity and taxonomic distinctness (Somerfield et al. 1997) (Chapter 1). These indices are more sensitive to environmental and other disturbance gradients than traditional measures (Fig. 14.6). However, in order to calculate taxonomic distinctness it is necessary to have

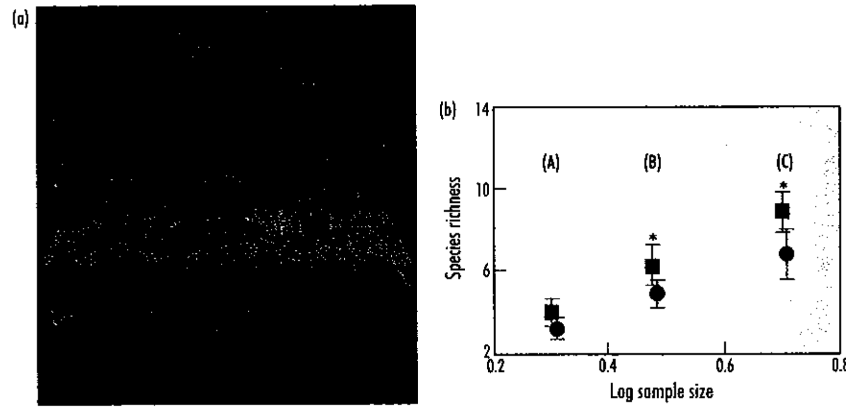


Fig. 14.5 The size of sample units collected (note: do not confuse with the number of samples collected) is critically important if rare species are those most vulnerable to a particular form of disturbance such as bottom fishing (a). If the size of samples collected is too small (A) then species richness will not be sampled with sufficient power to detect the effects of the disturbance. This is illustrated in an example of a study of the effects of sample size on the ability to detect the effects of fishing on benthic diversity (b). As the size of the samples collected increases (A, B and C) the effects of fishing on species richness (means ± 2 S.E.) suddenly become apparent as indicated by the significant differences (denoted by *) between fished and unfished areas.

● New diversity indices such as taxonomic distinctness are sensitive to changes in abundance or biomass of particularly species and have overcome some of the insensitivity of other indices.

a detailed knowledge of the taxonomy of most of the components of the assemblage, a feature that is lacking or variable for many areas of the world.

14.3.2 Distributional measures

These techniques summarize a set of species counts from a single sample as a curve or histogram. For example, *k*-dominance curves rank species in decreasing order of abundance, the steeper the initial part of the curve the greater the contribution of one or a few species to the overall abundance (Box 14.2). Counts of each taxon are converted to percentage abundance relative to the total number of individuals in the sample. The cumulative percentages are then plotted against species rank (Lambshead et al. 1983). A similar plot can be generated for biomass and superimposed on the abundance plot to give ABC (abundance-biomass-comparison) curves as defined by Warwick (1986). In a relatively pristine or low energy environment, a community typically will be dominated by large-bodied individuals (e.g. sponges, soft corals, large fishes) and the biomass curve would lie above the abundance curve (relatively few individuals contribute to a relatively high proportion of the biomass, giving a steep biomass curve). At increasing levels of environmental or physical stress the large-body-sized biota are replaced by high abundances of very

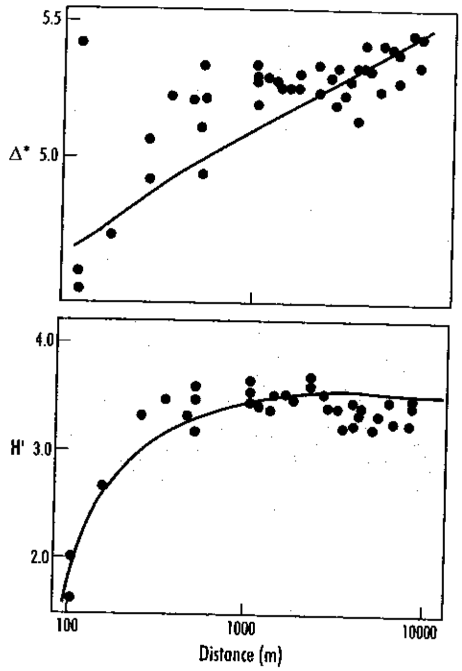


Fig. 14.6 The example shown is calculated from species abundance data for an oilfield in the North Sea. Various diversity indices were calculated and plotted against distance from the centre of the oilfield to a distance of 10 km. Taxonomic distinctness (Δ^*) was found to be the most sensitive univariate measure of community structure, increasing linearly as samples were collected further away from the centre of hydrocarbon contamination while the Shannon-Wiener diversity index (H') was insensitive beyond 100 m from the centre of the oilfield (Somerfield et al. 1997).

small biota (many individuals contribute a relatively small proportion of the total biomass but a relatively high proportion of total abundance, giving a steep abundance curve) and the biomass and abundance curves eventually cross over such that the abundance curve lies above the biomass curve.

● Distributional techniques such as ABC curves provide a useful insight into changes in biomass distribution relative to the abundance of biota in response to disturbance.

14.3.3 Multivariate techniques

These techniques compare two or more samples based on the extent to which these samples contain the same or different species at varying levels of abundance. Multivariate analytical techniques incorporate both species identity and abundance or biomass. A comparison between pairs of samples will yield a similarity coefficient that indicates the degree (percentage) of similarity between each pair of samples. When there are more than two pairs of samples, pair-wise comparisons are made between each possible combination of samples. The resulting similarity coefficients are used as the basis to classify or cluster mutually similar

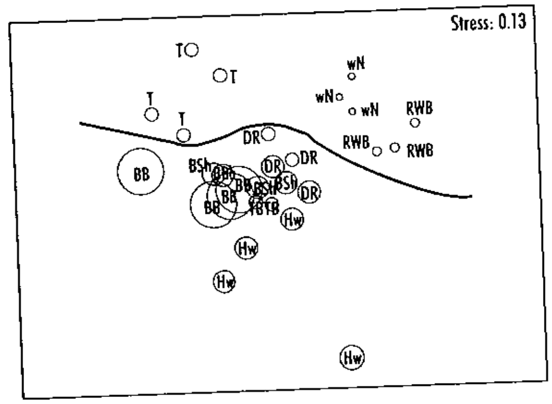


Fig. 14.7 An MDS ordination plot of fish and benthic communities sampled from sandbanks in the Irish Sea using a bottom fishing trawl. Each point represents one standardized trawl sample, and the codes relate to locations around the coast of Wales. Interpretation of the plot is quite simple. The closer together the samples appear in the plot the greater the degree of similarity between them in terms of species composition and abundance. Not surprisingly, the samples collected from the same location tend to be more similar to each other than samples collected from elsewhere. In addition, a key environmental parameter (carbonate content of the sediment) can be overlaid on the plot. This shows that locations shown above the solid line of the plot have a relatively low carbonate content (small bubbles), while those below the line have a relatively high carbonate content (large bubbles). (From Kaiser et al. 2004).

● Multivariate techniques such as multidimensional scaling ordination (MDS) can be used to visualize the literal similarity between multiple samples in relation to experimental manipulations or environmental gradients. The closer together two sample points appear in the ordination plot, the more similar their assemblage composition; the further apart they appear in the plot, the more dissimilar their constituents.

groups of samples. This can be taken on a further stage to yield an ordination plot (Fig. 14.7), which maps the samples in either two or three dimensions such that the distances between pairs of samples reflect their relative dissimilarity of species composition (Clarke & Warwick 1994). In a similar manner, the same samples can be analysed in terms of their environmental characteristics (depth, salinity, bottom current speed, etc.) and another ordination plot generated. The similarity matrices on which the species and environmental data plots are generated can then be compared to investigate the extent to which environmental similarities among samples reflect patterns in community data (Fig. 14.7). These analyses are complex but have been made accessible to non-specialists through the development of software packages such as PRIMER (Plymouth Routines in Multivariate Ecological Research).

14.3.4 Detecting change

As marine ecologists, we are interested in detecting ecological changes in response to human activities in the marine environment. This might be the change in biomass of plankton or fish abundance over several decades or might be the change in community characteristics in areas where aggregate extraction occurs. To stand any chance of success we need to ask the correct question at the outset. This is a problem that besets many of the long-term data sets that continue to be maintained today. While they are an invaluable source of data, many were never designed to answer some of today's ecological challenges. Long-term data sets often track environmental factors such as seawater temperature, which is reflected in the response of ecological metrics such as phytoplankton biomass (Fig. 14.4). However, many other variables are also correlated to these fluctuations and may be either synchronous or out of phase. Long-term ecological changes are strongly driven by environmental fluctuations and it is difficult to disentangle the relative importance of the influence of multiple environmental factors. For example, to ascertain the non-lethal effects of chemical contaminants on fish larval survival, it would be desirable to conduct a series of controlled laboratory experiments that manipulated only this variable while environmental conditions were held constant. Adding additional variables into the experimental design is possible but much more demanding on resources. Similarly, the ecological consequences of activities such as dumping of waste, channel dredging, aggregate extraction, and bottom fishing require an experimental approach to determine the degree of change attributed to these activities and the rate at which restoration of community attributes occurs (if at all).

● Long-term data sets are an invaluable tool to elucidate long-term trends and responses in biological metrics. However, many of these data sets are unable to answer conclusively some of today's pressing environmental questions due to their original design and sampling limitations.

Long-term data sets of fish abundance were set up specifically to measure fluctuations in spawning stock biomass (Chapter 12) and provide key guidance when setting conservation priorities for fish populations. However, as fish become less abundant with over-exploitation, it becomes more difficult to sample the population with accuracy (i.e. there simply are not enough fish to sample, or they are too elusive). Knowledge of the statistical power of surveys to detect trends is essential, since the consequences of not detecting a real trend can be profound (Box 14.3). In other words, a population may be heading for extinction but we might not detect this critical decline because the survey design does not have sufficient statistical power. This situation is particularly acute for species that are highly vulnerable to fishing effects or that are uncommon within an assemblage under normal circumstances. Maxwell and Jennings (unpublished data), examined these effects for North Sea fish survey data and found that the power of the survey to detect abundance decreases on timescales of <10 years was low. Hence, there is a real danger that some species could become extinct many years before we become aware of their demise.

● Vulnerable species could become extinct many years before we realize their demise if survey design is inadequate to detect declines in their populations.

Box 14.3 The importance of statistical power

Statistical power is not about flexing your analytical prowess. Every experimental study has some level of error whether from deficiencies in sampling or due to natural variability. Statistical tests are associated with a probability value, which is often set at 0.05. What this means is that there is a 1 in 20 chance that the result obtained occurred by chance alone. Hence, there is always a danger that our apparently significant effect is in fact false and occurred by pure chance. This is known as a **Type I** error. This is particularly likely to occur when undertaking multiple pair-wise tests, the more tests that we perform the greater the chance of getting an erroneous result by pure chance (e.g. testing for differences in each environmental parameter measured at each of two sites). The chance of making a Type I error is reduced by correcting or lowering the level at which significance is deemed to have been achieved (e.g. from 0.05 to 0.01).

When significant differences occur between two treatments, but we fail to find this difference, a **Type II** error has occurred. Non-significant outcomes are equally as important and just as informative as significant effects. Consider the example of the study of the effects of a shellfish fishery on benthic communities, the outcome of which may decide whether fisheries managers decide to close, or permit harvesting, and possibly whether the fishermen continue to make a living or not. When no effect of the fishery is reported it is important to examine whether the statistical power of the experiment was sufficient to detect the effects of interest (see Hall et al. 1993 for an elegant example). This underlines the value of **preliminary sampling** that can be used to calculate the amount of replication required to detect a certain level of effect (e.g. a 10% change in abundance of a particular species or community metric) before the main experimental manipulation is performed. When reading scientific papers that report non-significant results, always examine the error bars surrounding the mean or median values that are reported. If these are large compared to the mean or median (i.e. they have a high **error : mean ratio**) then treat their declared 'non-significance' with caution.

14.3.5 The experimental approach

The ability to design and interpret appropriately designed experiments is one of the fundamental tools of the ecologist. There are a number of crucial steps that must be considered to avoid common pitfalls made by both students and professionals alike. First, what is the question you are trying to answer? Let us say we are interested in knowing the effects of bivalve shellfish harvesting on the associated benthic fauna in a sandflat habitat. So the question is 'What changes does shellfish harvesting cause in the associated benthic community?', or to put it as a null hypothesis 'Shellfish harvesting causes no change in associated benthic community structure'. Thus we have identified that we need to manipulate a benthic assemblage containing an appropriate target species (e.g. clams, cockles, mussels) by subjecting it to a typical harvesting methodology (e.g. hand raking, suction dredging). This is the experimental treatment. The manipulated treatment will need to be compared with a control that is not subjected to the treatment.

● Good experimental design is dependent upon asking the correct question at the outset.

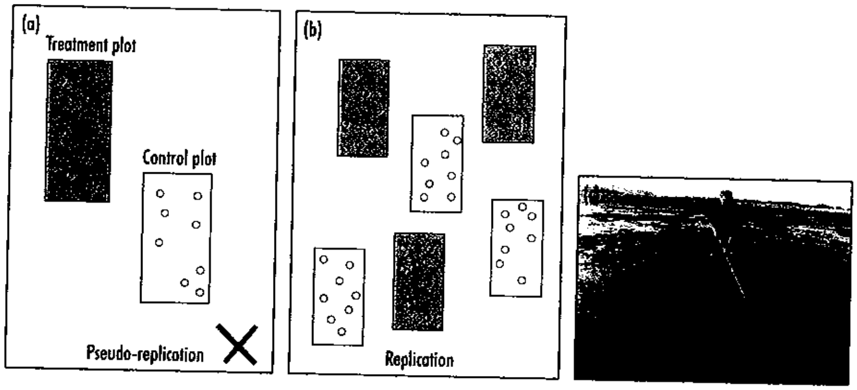


Fig. 14.8 (a) The wrong way to do it! One treatment plot and one control plot each sampled eight times. The eight samples are not replicates of the treatment, they are simple multiple samples of a plot. Multiple samples are taken to ensure that a representative selection of the biological assemblage is sampled at the correct spatial scale. (b) The right way to do it! Multiple replicated treatment and control plots. Each plot is a replicate in an investigation to determine, for example, the effects of shellfish harvesting on benthic biota (c).

An appropriate site needs to be selected, ideally this should be relatively uniform in nature, with no obvious additional sources of variation (e.g. riverine discharge at one end) or these will need to be taken into account in the eventual experimental design. The chosen site will also have a representative fauna, clearly there is no point investigating the effects of shellfish harvesting at a site with no shellfish. The treatment (the act of harvesting) needs to replicate as closely as possible the intensity and scale of activity that is normally conducted (Fig. 14.8), otherwise there is a danger that the results will be open to the accusation that they are not truly representative of the real activities they purport to represent. Other factors to consider might be the seasonality, intensity, and frequency of the normal harvesting activities.

- If an experiment is designed to answer questions about the effects of a particular human activity, it is important that the scale and techniques used to create the treatment effect are representative of the activity of interest.

Having identified a likely site for the experiment it is important to ensure that there are no strong biological gradients by taking preliminary samples in a random grid pattern across the proposed study site. A strong biological gradient (i.e. much higher densities of shellfish at one end of the site) will complicate the process of ascertaining the effects of the shellfish harvesting treatment but can be accounted for by using a Latin squares approach that replicates the control and treatment plots across the environmental gradient. However, time and budget constraints do not always permit the luxury of preliminary sampling, hence it is important to build additional treatment and control replicates into the design to ensure that any anomalous replicates do not jeopardize the validity of the experiment. It is much better to sample an excessive number of replicates, as it may not be possible to return and collect additional samples at a later stage once the experimental manipulation has been completed.

- Preliminary sampling often identifies environmental or biological gradients or anomalies that may complicate later analysis, and enables the calculation of the sampling effort necessary to detect a given level of change in a variable.

14.3.6 Pitfalls to avoid

This brings us onto a classic pitfall: the difference between a sample and a replicate. In Fig. 14.8a a treatment and control plot are shown. Eight samples have been collected from each plot, from which it is possible to calculate means and errors and to generate statistical tests for significant differences between the two plots in measured variables (abundance, biomass). But this tells us nothing about the effects of shellfishing on the benthos. It can only tell us whether one plot is different (or not) compared with the other. The samples within the treatment and control plots are *not* replicates. The common pitfall is to treat the samples within the plots as if they were replicates of the treatment and control. This is known as pseudoreplication. A properly replicated design is shown in Fig. 14.8b. Clearly more sampling effort is used, but the treatment and control plots are replicated (2 treatments × 3 replicate plots × 8 samples). So what can one do if time and budget dictate that this is too many samples? It is important to preserve or increase replication at the level that addresses best the initial question. In this example, if we had to reduce our sampling effort by half, it would be better to take only 4 samples from each of 3 control and 3 treatment plots rather than taking 6 samples from each of 2 control and 2 treatment plots. The greater the level of replication, the greater the statistical power to detect differences among treatments (Box 14.3).

What can we do if it is not feasible to replicate the treatment? Abandon the experiment is one alternative, but not very helpful if a scientific study is required for policy advice. Typically such an example might occur if the cost of creating the treatment is prohibitive, as in the study of the ecological effects of aggregate extraction undertaken by Kenny and Rees (1996). Using ships to generate experiments can be very expensive, typically $US25 000 per day and clearly places a constraint on what can be achieved on a fixed budget. Kenny and Rees (1996) overcame this problem by using a Before After Control Impact (BACI) design in which both a designated control and treatment plot are studied prior to any experimental disturbance (Underwood 1991). Once the similarity of these plots was confirmed the experiment proceeded. The design is still pseudoreplicated (there is only one treatment and one control plot) but we have more confidence that the difference detected immediately after aggregate extraction is indeed attributed to the effects associated with dredging (as opposed to some other random factor). However, an examination of this study reveals the weakness of such designs as the authors were unable to account for changes recorded some 24 months later and were only able to speculate that these were site-specific changes caused by local physical forcing due to wave action (Kenny & Rees 1996).

In some cases, a Before, After, Control, Impact experimental design is the only viable option, but such a design has far weaker powers of inference than a fully replicated experiment with equal numbers of control and treatment replicates.

BACI designs are appropriate for the examination of short-term responses to human interference, but are highly unsatisfactory for detection of long-term effects. This basic design could be improved by introducing multiple replicate control plots that can be studied for the effects of random factors that might also lead to changes in community structure. For example, if four control plots were studied, three might exhibit similar changes over time while one of these might experience a chance settlement of larvae of a particular species that did not occur at the other sites. This design tells us that we can be more confident that the difference between the treatment and control sites that we studied are real effects and not attributed to a chance occurrence.

14.4 Agents of Change

14.4.1 Riverine input and land use

The composition and quantity of riverine discharge into the shelf seas will affect habitat composition and biology in the immediate area of discharge. Globally, this is an important process given that 70% of the sediment input to the sea is from riverine sources, although this varies from region to region (Milliman 1991 and see Chapter 7). Dam construction, fluvial management regimes, and the effects of changing patterns in precipitation linked to global climate change will all affect the rate and quantity of riverine inputs in the marine environment. The amount of sediment discharged is related to drainage basin size; the surface area of a small drainage basin provides lower storage capacity for sediment than a larger basin. Thus smaller rivers discharge a proportionately greater load of sediment per unit of water discharged than much larger systems.

● Some 70% of sediment input into the sea is derived from rivers, hence changes in land-use and rainfall are likely to have a significant impact on marine communities in coastal regions.

Some of the world's largest rivers are located in South America where they are known to play an important role in structuring the topography and sedimentary conditions of the adjacent seabed (Amazon and Papuan continental shelves). Close to the river mouth the water column is heavily laden with sediment and results in a distinct layering of sediment reflecting discrete periods of deposition of riverine sediment and erosion of the seabed. Consequently the fauna is highly impoverished in this environment and tends to be dominated by bacteria (Rhoads et al. 1985). Field studies demonstrate that the body size, abundance, and depth distribution of benthic fauna are at their lowest across the continental shelf coincident with peaks of riverine discharge and maximum trade-wind stress. During periods of low riverine discharge and minimal wind stress, macrofauna abundance was highest and bacterial biomass increased by a factor of two (Aller & Stupakoff 1996).

The quality and quantity of sediment transported by river systems is affected by the land-use practices that occur in the river catchment.

Deforestation and other agricultural practices have led to substantial soil erosion such that it is estimated to be 3.7 times greater than 2500 years ago, prior to the period when forests were cleared for agricultural purposes. Widespread cultivation of land in northern China is considered to have caused the tenfold increase in sediment discharge from the Yellow River. The increasing demand for greater areas of agricultural land to support subsistence farming in developing countries seems likely to increase the sediment discharge from regionally important rivers (Hall 2002).

● Deforestation for agricultural purposes has been associated with a 3.7 times increase in soil erosion.

14.4.2 Eutrophication

High nutrient loads in river discharge are associated with phytoplankton blooms that generate mainly diatomaceous phytodetritus. This material descends to the seabed where it fuels high rates of microbial respiration and depletion of oxygen (anoxia) at the sediment–water interface (Fig. 14.9). Such events tend to coincide with periods of calm weather when inshore waters undergo stratification. The Baltic Sea, the Adriatic, and the Gulf of Mexico are well-documented examples of locations where such

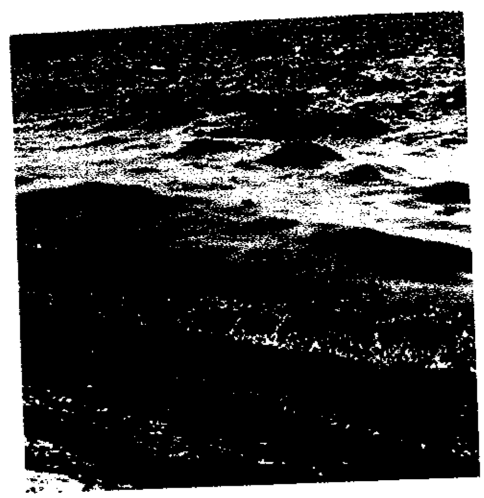


Fig. 14.9 Although this looks like the aftermath of an oil pollution incident washing ashore, it is in fact the product of a phytoplankton bloom die-off off the coast of Cape Cod (NE United States), lovingly known as 'munge' by local people (the word 'munge' or 'mung' means to mash into a mess). The genus responsible for the bloom is probably *Phaeocystis*. (Photograph: D. A. Kaiser.)

events occur on a regular basis (Turner & Rabalais 1994; Chapter 7). Systems that are particularly susceptible to eutrophication would include those with restricted tidal inundation, poor water exchange or those susceptible to stratification. Typically such systems include fjordic areas, enclosed bays, and areas with relatively deep water.

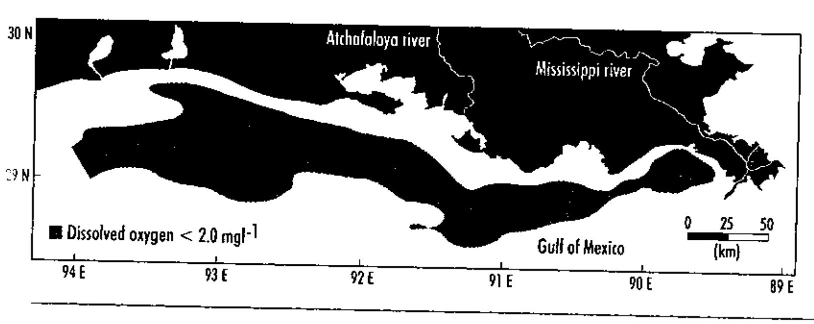
While periodic anoxic events occur naturally, their extent and severity appears to be related to human influences. Nitrogen loading doubled in the River Mississippi between the 1900s and 1980s, and it is thought that the increasing scale of anoxic events in the Gulf of Mexico is linked to eutrophication (Turner & Rabalais 1994; Box 14.4). Evidence for this hypothesis arises from studies of stable isotope signatures and organic tracers in sediment cores, which indicate that nutrient levels started to increase in the 1950s and finally levelled off in the 1980s, coinciding with a threefold increase in the use of chemical fertilizers by the agriculture industry in the latter half of the twentieth century (Eadie et al. 1994). The sources of nitrogen that flow into the Mississippi would appear to be diffuse and can originate as much as 1000 km upstream, thus it is almost impossible to pinpoint a single major source of contamination. Arresting the activities of an entire industry will take considerable scientific proof that requires the elimination of other possible causative agents (e.g. climate change). For example, hydrocarbon fuel consumption appears to be the primary source of atmospheric inputs of nitrogen, which have increased by 50 to 200% over the last 50 years (Paerl 1995).

● High … enclosed … prone to … stratification … associated … of coastal …

● Sources of nitrogen … river systems are diffuse … occur many hundreds of kilometres from the sea, but evidence suggests that agricultural run-off is the primary culprit.

Box 14.4 Anoxic events in the Gulf of Mexico

Weather conditions, large-scale physical oceanographic processes, biological productivity, the amount of nutrient discharge, and other factors affect oxygen depletion in coastal waters. The largest zone of oxygen-depleted waters in the entire western Atlantic Ocean, occurs in the northern Gulf of Mexico on the Louisiana continental shelf adjacent to the outflows of the Mississippi and Atchafalaya Rivers. In the last decade the extent of bottom water hypoxia (16 000 to 18 000 km^2) has been greater than twice the surface area of the Chesapeake Bay, rivalling extensive hypoxic/anoxic regions of the Baltic and Black Seas. Depending upon environmental conditions, hypoxia occurs from late February to early October, but is most widespread, persistent and severe in June, July, and August. See also Justić et al. 2005. Figure reproduced with permission of Nacy Rabalais.



14.4.3 Water use

Clearly the inputs derived from riverine discharge have an important influence on shelf sea ecosystem processes. Consequently, any activity that alters riverine discharge is likely to have ecological consequences for near-shore ecosystems. Major dam schemes are responsible for some striking changes in marine ecosystems. For example, in the Adriatic Sea, the output of the River Po and adjacent river systems has been lowered by 12% in the recent years. The reduction in nutrient inputs has been associated with a decrease in primary production in the local shelf water mass (Alongi 1998). The structure of many river delta regions is maintained by the supply of suspended riverine sediment. Accretion of the Nile delta was reversed after the construction of the barrage on the River Nile in 1868, and further exacerbated with the construction of the Aswan dam. The reduction in supply of riverine sediment to the coastline led to coastal erosion rates of between 5 and 240 m per year. The associated reduction in nutrients transported out to sea was linked with a decline in landings from fisheries for both pelagic and demersal fisheries (Box 14.5). In later years, fishery production increased as the reductions in nutrients supplied through the Nile discharge were replaced through coastal urbanization and the concomitant increase in discharge of sewage into the River Nile (Nixon 2004).

● Reductions in riverine discharge as a result of water abstraction and dam construction affect coastal erosion, primary production, and commercial fisheries.

14.4.4 Hydrocarbon exploitation

The extraction of oil and gas from the sea inevitably releases hydrocarbons into the marine environment as a result of drilling activities, and in the past due to the use of contaminated drilling mud (Olsgard & Gray 1995). The ecological effects of the addition of fine sediment to the seabed beneath drilling platforms and the contaminants within drilling mud are associated with a localized reduction in species diversity. These effects are ameliorated with increasing distance from the drilling platform. In the case of gas platforms in the Gulf of Mexico, community changes are confined to within 100 m of the drilling platform (Montagna & Harper 1996). However, in the North Sea, the cumulative effects of using drilling mud over periods of up to nine years, led to ecological changes at distances of up to 6 km from the drilling platform. However, the more recent use of water-based drilling mud has considerably reduced adverse ecological effects on seabed communities (Olsgard & Gray 1995). The pollutant effects of drilling activities are far less dramatic than the impacts of accidents that occur while carrying oil in bulk across the oceans.

● Understanding the negative ecological effects of the contaminants in drilling mud has prompted the oil and gas industry to seek more environmentally friendly alternatives.

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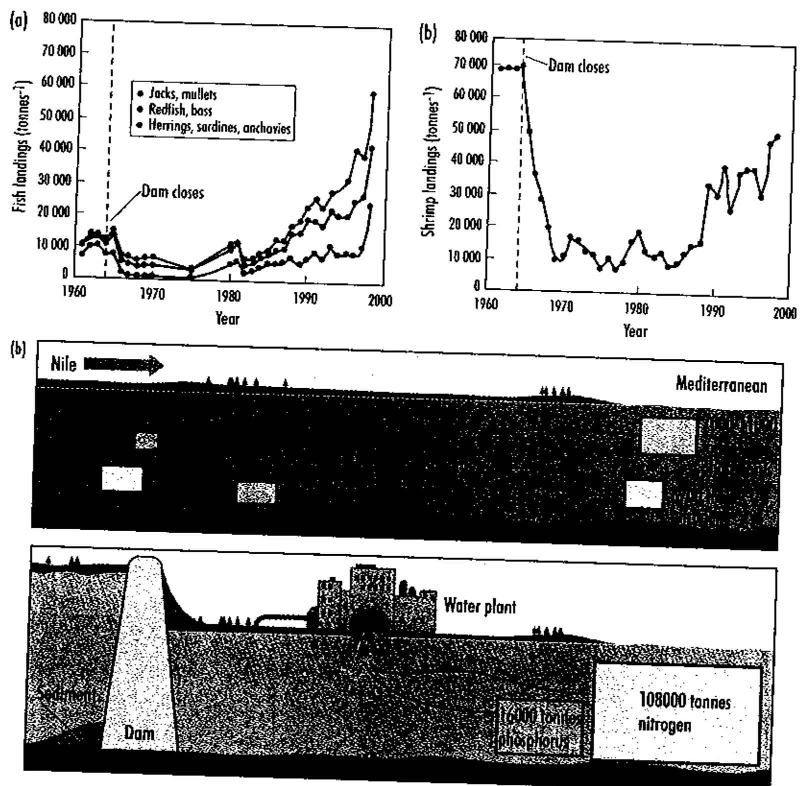
Box 14.5 Effects of damming the River Nile

Catches of commercially important fish species (a) fell dramatically after closure of the Aswan High Dam in 1964. Both demersal (jacks, mullets, bass, and redfish) and pelagic (herring, sardine, and anchovy) fish landings remained low until 1980 when they rose steadily. Shrimp landings showed similar trends but have never recovered to levels experienced prior to closure of the Aswan high dam. The declines in fisheries are in part thought to relate to the fall in phosphorus and nitrogen released into the Mediterranean Sea from the River Nile. (b) However, as populations along the river expanded it seems highly probable that inputs from sewage have offset the loss of nutrient input as a result of closure of the Aswan High Dam. Reproduced with permission from American Scientist 92: 156–165.

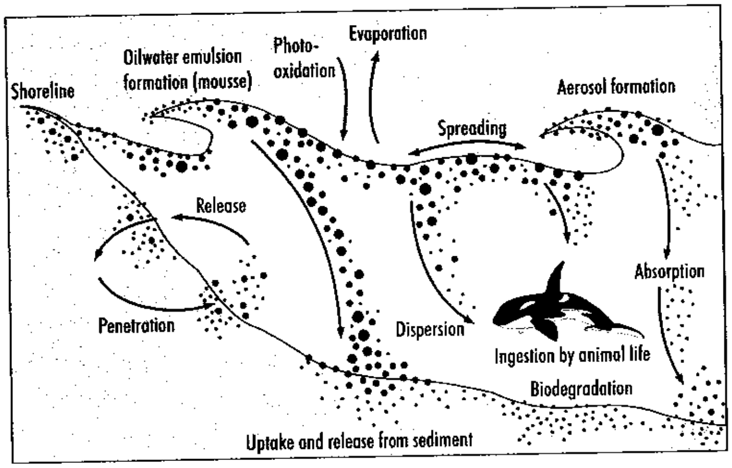


High profile examples of oil spillage incidents include the grounding of the MV *Exxon Valdez* off Canada and the MV *Braer* off the Shetland Islands or the destruction of oil pipelines and refineries during the Gulf War. The grounding of the MV *Braer* differed from most oil spill incidents in that it coincided with extremely severe storm force winds that were maintained for a number of weeks. The resulting sea conditions meant that the oil was highly dispersed by natural wave action (Box 14.6). Technology for dealing with these incidents and our understanding of the ecology of the biological recovery processes that occur after such incidents is much improved. Indeed, it is often ecologically better to leave natural ecological processes to degrade oil spills rather than to intervene with the use of chemical dispersants. The use of dispersants is often driven more by aesthetic demands to preserve the appeal of tourist beaches rather than on any scientific grounds. In the early 1970s the oil dispersants used to clean beaches after the MV *Torrey Canyon* oil spill were more harmful to organisms than the oil itself (Southward & Southward 1978). Nowadays, dispersants are much less toxic and bioremediation of oiled contaminated habitats is enhanced by the use of bacterial digestion to speed the breakdown of the oil.

● The use of dispersants on oil spills can have more severe ecological consequences than if the oil were permitted to biodegrade. However, social and economic issues often dictate that unsightly oil needs to be removed from tourist hotspots, and thereby override ecological considerations.

Box 14.6 Impact of oil on the marine environment

Crude oils and petroleum products are complex substances, typically most of the volatile fractions evaporate into the atmosphere shortly after they have been released into the environment. The heavier fractions are then exposed to a variety of physical, chemical, and biological processes according to the prevailing weather conditions. For example aerosols and emulsions are formed as a result of wave action, while exposure to ultraviolet light leads to photo-oxidation at the surface of the sea. Persistent rough seas contributed to emulsification of the oil in the MV *Braer* incident and helped the rapid dispersal of the oil.



14.4.4 Mining geological resources

Advances in marine technology have made the mineral resources of the seabed accessible in a wide range of different marine environments. These range from relatively shallow coastal waters where sand and aggregate (gravel) are extracted, to deep-sea mining of manganese nodules on the abyssal plain, to diamond dredging off the coast of Namibia. In each of these examples, sediments are removed directly from the seabed (usually via a dredge head) and the desired products extracted on board the vessel at the surface. Not surprisingly, these activities represent an intense localized disturbance to the seabed and completely alter the biological and geological composition of the habitat.

● Most aggregate extraction is permitted under licenses granted for defined areas of the seabed. Improvements in satellite navigation systems have enabled more accurate compliance with these licenses by aggregate companies, but also enable stricter enforcement through the use of 'black box' recorders that log vessel movements.

The demand for raw materials to supply housing and construction projects is unlikely to decline in the foreseeable future, and Charlier and Charlier (1992) predicted that production would reach 200 million tonnes for the United Kingdom alone in the next century. Extraction of marine aggregate in the United Kingdom has increased from 7 million tonnes per annum in 1969 to a relatively constant figure of approximately 22 million tonnes per annum between 1989 and 2003. Dredging of aggregates and the disturbance of the seabed generated by towed bottom fishing gears has some similarities in terms of their ecological effects on benthic biota and habitats. However, while fishing activities are widespread, aggregate dredging is confined to restricted areas that are strictly controlled through licensing schemes. However, dredging is a much more intensive activity, and removes the habitat to depths of up to 1 m. Hence, recovery of the habitat is a key process in determining the rate of recovery of the biota associated with that habitat (Dernie et al. 2003, Fig. 14.2).

● Aggregate extraction causes intense, but localized, disturbance of the seabed that can lead to long-term habitat alteration lasting in excess of several years.

Kenny and Rees (1996) used a commercial gravel dredger to remove 50 000 tonnes of aggregate off the east coast of England. Two years later, the sediment composition in the area from which the aggregate had been removed was still dominated by finer particles than in a comparable control area, and was much more prone to resuspension as a result of wave action. This is perhaps not surprising given that habitat recovery is likely to occur as a result of the active transport of gravel material into the site, a process that requires highly energetic processes (violent storm activity and strong seabed currents). Although species diversity was restored after two years, the biomass of many of the longer-lived species remained greatly reduced (e.g. the horse mussel, *Modiolus modiolus*). The natural disturbance regime at individual sites will determine the rate of habitat and biological recovery that is likely to occur as a result of dredge disturbance (Hall 1994).

Fears of shortages of metal resources in the early 1970s sparked interest in the idea of mining manganese nodules in the deep sea. Manganese nodules contain metals such as iron, manganese, copper, nickel, cobalt, zinc, and silver. Dredging for manganese nodules involves the use of remotely operated seabed vehicles that crawl along the seabed scooping up surface deposits, and thereby create a furrow in the seabed surface. Experimental studies have demonstrated (not surprisingly) that the dredging process reduces both habitat and species diversity within the dredge track, and produces a sediment plume. Sedimentation at these depths occurs at the rate of a few millimetres every 1000 years. As a result, dredge tracks will persist for many years creating long-term changes in deep seabed topography, while filter-feeding organisms may be adversely affected by the unnaturally large pulse of sediment that settles out of the plume (Theil & Schriever 1990).

● Early excitement regarding the feasibility of deep-sea mineral extraction subsided with a reduction in the stock market value of commodities such as copper and cobalt.

Mining minerals at extreme depths is an expensive business and only likely to become economically viable when the combined nickel, copper, and cobalt content of the manganese nodule exceeds 2.5% at an abundance of $>10\,\mathrm{kg\,m^{-2}}$. Only a few regions in the Pacific Ocean and from the Central Indian Basin in the Indian Ocean meet these criteria. Economic-grade manganese nodules are generally found in the middle of the ocean in water depths exceeding 4500 m. Manganese nodules would have to be lifted at the rate of 3 million tonnes per year for 20 years at an individual mine site covering an area $>6000\,\mathrm{km^2}$ to make such a proposition economically feasible. The lower costs associated with the terrestrial mining of these metals mean that mining in the deep sea is likely to remain on hold for the foreseeable future (Glasby 2000).

14.4.5 Contaminants

In addition to concern regarding the effects of elevated nutrient inputs into coastal waters, there is considerable concern over the long-term and often subtle effects of persistent contaminants in the marine environment. Many of these contaminants, such as radionuclides and organic substances, are derived from industrial and heat generation sources. Polychlorinated biphenyls (PCBs) are particularly persistent and accumulate through the food chain with negative effects expressed in top predators. For example, PCB contamination of prey has been linked to hatching failure for a number of avian predators, while contaminants such as mercury occur in higher concentration in the feathers of seabirds sampled in the latter half of the twentieth century compared with those sampled from museum collections (Arcos et al. 2002). While the effects of such contaminants on apex predators have been well documented, the effects on organisms at lower trophic levels is less well studied. Although invertebrates such as amphipods are used in lethal toxicity tests, the incidence of contaminant mortality associated with bioengineering organisms (Chapter 7) that affect habitat structure remains unknown. The effects of contaminants may be subtle in that they do not necessarily cause direct mortality but may have negative population effects via their influence on recruitment processes and larval viability. For example, Atlantic croaker (*Micropogonias undulatus*) larvae exposed to the chemical Aroclor 1254 have significantly reduced growth rates and are less capable of evading predator stimuli (McCarthy et al. 2003).

Abnormalities in the growth and reproduction of a number of different molluscs became apparent in the mid 1980s. Symptoms included reduced growth, shell deformation, and declining population abundance in specific localities. In particular, dogwhelk (*Nucella lapillus*) populations had become locally extinct in the United Kingdom and elsewhere, usually in localities close to harbours and ports. The most striking symptom associated with these declines was the occurrence of imposex. Imposex is manifested in female gastropods by the development of a *vas deferens* and in some cases a penis that lowers reproductive success. Not all molluscs exhibited the same symptoms; populations of *Littorina littorea* had lower egg production, while oysters developed thickened and deformed shells (Matthiessen et al. 1995). The causative agent was identified as **tributyl tin** (TBT) based anti-fouling paint that was used on the hulls of boats. Legislation has since banned the use of this paint on certain categories of vessels and has been associated with a recovery in mollusc populations in countries where its use is restricted.

● Contaminants [illegible] sub lethal [illegible] performance [illegible] survival rate of [illegible]. Larval stages are [illegible] vulnerable to these [illegible] are easily overlooked.

● See Evans et al. 2000 for the controversy surrounding potential replacements for TBT.

14.5 Climate Change

So far we have examined the effects of human activities that have relatively restricted spatial effects on communities or particular groups of animals. However the potentially significant consequences of global climate change for marine systems and activities affect all regions of the planet at both small and large scales (Hall 2002). Those changes likely to occur in response to global warming include sea level rise, water-column warming, precipitation, wind speed and patterns, water column circulation, and the frequency and intensity of storms. We give only a few examples of environmental changes that might affect marine ecosystems.

14.5.1 Temperature effects

Global warming is likely to affect biological processes and biodiversity in the oceans. Rising temperatures may result in a reduction in the equator to pole temperature gradient. The frequency of El Niño-like conditions in the tropical Pacific will probably increase, with the eastern tropical Pacific warming to a greater extent than the western tropical Pacific (IPCC 2000). In conjunction with hydrographic conditions, temperature plays a significant role in determining the biogeographic distribution of species and we can undoubtedly expect to see changes in species distributions with climate change (Chapter 1). For example, a northward extension of warmer conditions may enable the merger of the currently geographically disjunct distributions of warm temperate species such as the commercially important clam *Mercenaria mercenaria* and the eastern oyster (*Crassostrea virginica*). The present northern limit of these species is Cape Cod on the eastern seaboard of the United States, although isolated populations are found further north in the warmer waters of the Gulf of St Lawrence. These temperature changes are also likely to extend the spread of some non-native species that have been introduced through ship's ballast water or other practices such as aquaculture. Many introduced species are considered to be pests as they predate commercially valuable shellfish, for example the green crab (*Carcinus maenas*) that has become established on the western coast of the United States and in Australia.

● Increasing sea temperatures will extend the poleward distribution of low latitude species and may enhance the geographic spread of non-native invasive species.

We have already seen in Chapters 4, 6, and 7 that semi-enclosed water bodies are prone to anoxic events as a result of stratification of the water column and oxygen depletion at the seabed. The frequency of such events may increase as warmer temperatures lower the oxygen carrying capacity of water while the biological oxygen demand of the biota will increase. Lower oxygen carrying capacity, coupled with eutrophication will increase the likelihood of such events, which may have more frequent adverse effects on coastal fisheries. Temperature effects on fisheries are of particular concern since water temperature is a strong predictor of fish distributions, the timing of spawning, growth rate, and survival of larvae (Heath 1992; Lehody et al. 1997). In the coastal waters of the USA, for example, a reduction in the spatial extent of oxygen rich cool water may cause concomitant changes in the distribution of striped bass (*Morone saxatilis*) (Coutant 1990). Further increases in global sea temperature are likely to cause new combinations of species mixes both in fish and benthic assemblages.

Coral bleaching is an increasing global phenomenon (Harvell et al. 1999) that occurs when the symbiotic zooxanthellae are expelled, algal pigmentation is lost, or both scenarios occur, giving the coral tissues a 'bleached' appearance (Chapter 10). These events have been linked with exceptionally high seawater temperatures, elevated doses of ultraviolet radiation, infection by pathogens, and the alteration of local salinity regimes. The degradation of coral reefs has important economic consequences for local communities that often depend on reef related eco-tourism and associated fisheries as an important source of income.

● Coral bleaching has severe implications for artisanal fisheries and eco-tourism.

14.5.2 Change in rainfall

The effects of climate change on rainfall may be important given the influence of river and land run-off in estuarine and coastal systems

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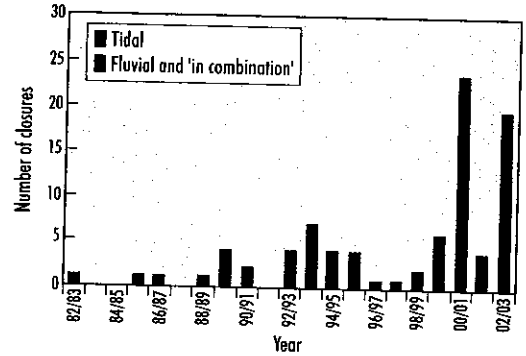

Fig. 14.10 The increasing frequency of closures of the R Thames barrier in recent yea one indication of the increasi effect of elevated rainfall and greater incidence of tidal sur and sea level change (King 20(

(Chapters 4 and 7). The effects of increasing rainfall can also combine with tidal surges resulting in a greater intensity of coastal and flood plain inundation (Fig. 14.10). Rainfall is a prominent seasonal feature in areas of the tropics especially where monsoon rains occur. Global predictions of changes in rainfall suggest little change in southeast Asia and South America, where riverine influences are especially important, but potential small increases (5 to 20%) for the Indian subcontinent, East Asia, Canada, and Northern Europe and potentially large increases (>20%) for Northern Asia and the Sahara (IPCC 2000). Even if these predictions are accurate, it is difficult to ascertain their consequences given the changing use of water abstraction with industrial growth in many of the countries affected. Changes in rainfall will also have an effect on buoyancy driven water currents that occur in the vicinity of estuaries (Chapter 7). Increasing the rate and volume of water discharged from rivers may alter the position of density gradient fronts, and may drive the ensuing coastal sediment plume further along the coastline (see ROFIs, Chapter 7). Seabed communities, fisheries, and marine mammals associated with these systems would change their distribution accordingly.

14.5.3 Water circulation

Intensification of alongshore wind stress on the ocean surface may already have increased coastal **upwelling intensity** (Bakun 1990). As many of the most productive marine regions in the world occur in regions of intense upwelling, the potential for climate change to affect fisheries seems highly likely (Bakun 1990). Recent anomalous changes in the upwelling of cool nutrient-rich water off the coast of the western United States led to an extensive anoxic event resulting from unusually high productivity. Extensive areas of the seabed from the continental shelf edge to within 700 m of the surf zone were severely oxygen depleted, resulting in mass kills of commercially important fish and

Box 14.7 Non-native oysters in the Chesapeake Bay

This text is paraphrased from a testimony given to the 108th US Congress (First Session), Subcommittee on Fisheries Conservation, Wildlife and Oceans, Committee on resources, United States House of Representative in 2003 by Professors J.L. Anderson and R. Whitlatch, both experts in oyster ecology and the ecology of Chesapeake Bay.

Good morning Mr Chairman and members of the Subcommittee. Thank you for this opportunity to speak to you about the proposed introduction of the non-native oyster *Crassostrea ariakensis*.

The [native] oyster stock in the Chesapeake Bay has declined dramatically. Harvest is now about one percent of what it was at the end of the 19th century. Fishing pressure and habitat degradation resulting from agricultural, industrial and residential pollution, deforestation, and oyster reef destruction have contributed to the decline. In recent de cades, however, the diseases MSX and Dermo have been identified as the core reasons for further decline. It should be noted that MSX is caused by a parasite that was introduced to the East Coast from Asia. Fisheries management efforts and various restoration programs have not been successful in restoring the oyster stock to date. The loss of the oyster has been devastating to the oyster industry and its dependent communities. Those that remain in the Chesapeake oyster-processing sector now rely on oysters that are brought in from the Gulf of Mexico region and other areas for their economic survival. Furthermore, the loss of oysters has contributed to declines in water quality and clarity.

The introduction of the non-native Suminoe oyster, or *Crassostrea ariakensis*, from Asia has been proposed as a solution to these difficult problems. Indications are that it may grow well in the Chesapeake Bay and it is known to be resistant to MSX and Dermo. Despite the positive results of introductions of some oyster species, some extremely negative consequences have been observed as well. A major risk of introducing a non-native oyster comes from pathogens, such as MSX, or the introduction of other animals or plants that may be attached to oysters. While in Australia and New Zealand, introduced non-native oysters have displaced native oysters.

Aquaculture of sterile non-native oysters, represents an appropriate interim step that possesses least risk (in terms of the available options) to the Chesapeake Bay and its dependent communities. However, limits and controls on aquaculture practices must be implemented to minimize the risk of introducing pathogens or reproductive non-native oysters during this transitional phase. This approach may provide limited benefit to parts of the oyster industry and it provides decision makers with the added information required to make future decisions. Moreover, this option allows more time for innovative, science-based efforts to restore native oyster populations. On the other hand, the option of not allowing any introduction, fails to address fishing industry concerns and will not result in improved understanding of the ramifications of non-native introductions. It may also increase the risk of rogue or uncontrolled introductions. The option of the direct introduction of reproductive non-native oysters, is not advised given the limited knowledge base on *C. ariakensis* and the potential for irreversible consequences of introducing a reproductive non-native oyster into the Chesapeake Bay. It is unlikely that there exists any 'quick fix' to the Chesapeake oyster situation.

● Changes in wind stress may intensify coastal upwelling and have been linked already to anomalously high production and mass kills of commercially important fisheries.

shellfish species (Grantham et al. 2004). Changes in wind patterns may alter wind-induced currents that are important for larval transport, which means that larvae may no longer be conveyed to habitats suitable for further development (Heath 1992).

14.6 Interaction of Multiple Factors

We have provided a sample of the influence of human activities on the marine environment. We have dealt with each of these factors separately for simplicity, but multiple factors may operate at the same time or in sequence. While one factor on its own may not have serious consequences for an organism or community, the synergistic action of several factors may prove catastrophic. For example, organisms under one form of environmental stress (e.g. rising mean sea temperatures) may be less able to cope with the physiological demands of adapting to increasing frequencies of freshwater discharge that might occur with changing rainfall patterns, and their resilience may be further weakened by exposure to industrial contaminants. The effects of multiple and sequential human interference are well illustrated by a testimony given to the United States Congress to encourage it to consider the consequences of introducing a non-native oyster into Chesapeake Bay. Declines in native oyster stocks in Chesapeake Bay are well documented and have been linked to reduced water quality and over-fishing, but attempts to reinvigorate the oyster stocks have repeatedly failed. One solution appears to be to introduce a non-native species that may be more tolerant of present day environmental conditions and parasitic fauna that were accidentally introduced into Chesapeake Bay as a result of past aquaculture introductions or discharge of ship's ballast water. The testimony given in Box 14.7 highlights the complex considerations that face managers of natural resources that have to weigh the ecological risks and potential benefits of such introductions.

CHAPTER SUMMARY

- Natural agents of disturbance to marine communities and systems occur across a full range of spatial and temporal scales.
- The response of community metrics (diversity) to disturbance, can be predicted from ecological paradigms such as the intermediate disturbance hypothesis.
- The relative impact of human activities on marine ecosystems needs to be assessed against a background of natural environmental fluctuations that occur at a variety of temporal scales. Our ability to detect the effects of human intervention critically depends on an appropriate analytical or experimental approach.
- Coastal shelf environments are subjected to the most intensive human activities that include fishing, aquaculture, mineral and hydrocarbon extraction, shipping activities, tourism and discharges of effluents and pollutants.
- Eutrophication that has resulted from elevated inputs of organic matter and nutrients from agricultural run-off have caused wide-spread bloom of toxic microalgae and anoxic events that have resulted in mass mortalities of marine biota.

- Persistent contaminants such as PCBs have subtle effects at the population level by affecting survivorship of larvae and juveniles. The full impact of these contaminants is a subject of speculation.
- The congested inhabitation of coastal margins means that much of the world's population are vulnerable to changes in sea level linked to global climate warming.
- Increases in global rainfall will elevate freshwater and sediment discharge into coastal waters thereby affecting density-driven currents in ROFIs and will change the distribution of biological communities.

FURTHER READING

Clarke & Warwick (1994) provide a readable and sympathetic (to the non-mathematical) guide to the detection of ecological changes in marine (and other) communities. Evans et al. (2000) provides a synthesis of the debate surrounding replacement of tributyl tin with suitable less environmentally damaging alternatives. Hall (2002) provides a review of agents of change in coastal systems in the present and also examines possible future impacts.

- Clarke, K. & Warwick, R. 1994. *Change in Marine Communities: an Approach to Statistical Analysis and Interpretation*. Natural Environmental Research Council, Plymouth Marine Laboratory, Plymouth.
- Evans, S. M., Birchenough, A. C. & Brancato, M.S. 2000. The TBT ban: out of the frying pan into the fire? *Marine Pollution Bulletin* 40: 204–11.
- Hall, S. J. 2002. The continental shelf benthic ecosystem: current status, agents for change and future prospects. *Environmental Conservation* 29: 350–74.