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An Ecologist's View of the Implications of the Observed Physiological and Biochemical Effects of Petroleum Compounds on Marine Organisms and Ecosystems [and Discussion]

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An ecologist's view of the implications of the observed physiological and biochemical effects of petroleum compounds on marine organisms and ecosystems

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The quantity of hydrocarbons in some seas and sediments approaches the concentrations at which oil can be lethal or cause sublethal effects to marine animals in the laboratory. Field studies of the biological consequences of oil spills show good agreement with the experimental data: intertidal and subtidal benthic communities are affected and can take a long time to recover, undergoing slow and subtle changes. The temporal changes seen after oil spills are comparable with the spatial changes observed around chronic discharges, essentially a simplification of the ecosystem with dominance of a few species. These changes cannot be expressed as a single index of diversity or of physiological stress. To understand the long-term consequences of oil pollution it is necessary to monitor the community as a whole, but well defined methods and objectives are required.

Introduction

Most of the 6000 Mt of oil that enters the sea each year (Connell & Miller 1981*a*; Farrington 1980) becomes degraded or evaporates into the atmosphere (Malins 1981). However, in some seas and bottom sediments the measured concentrations of hydrocarbons can be of the same order of magnitude as the experimentally determined concentrations that can kill or affect the life processes of fishes and invertebrates (Corner 1978). It follows that present levels of oil pollution of the sea could have harmful effects on natural communities. The results of laboratory experiments are often discounted by technologists as having little relevance to the real world, and some ecologists regard obscure sublethal effects as having little environmental importance (Lewis 1980). On the other hand, experimental work is usually conducted with hardy species that can survive and breed in the laboratory, and petroleum compounds may be expected to produce greater effects on the less hardy species found in natural communities (see discussion in Cole (1979*b*)). The extent to which the results of laboratory experiments hold good for natural communities can be judged from accounts of the biological consequences of accidental spills of oil, from the results of small-scale deliberate spills of oil in the field, and from the slight amount of published information on the effects of chronic discharges.

For reasons of space this review of field evidence is restricted to communities with which I am reasonably familiar, the seashore and benthic sediments, with most emphasis on long-term surveys. For the effects of oil on specialized communities, including salt marshes, mangrove swamps, coral reefs and seagrass beds, reference should be made to Wood & Johannes (1975) and to Dicks & Hartley (this symposium). For plankton communities the accompanying paper by Davenport (this symposium) gives full details. However, it should be noted that in contrast to experiments in the laboratory and with controlled ecosystems (Lacaze 1980), all accounts of the after-effects of accidental spills suggest little immediate damage to plankton communities or to spawning fish later (Lännergren 1978; Johnston 1978; Sherman & Busch 1978; Kühnhold 1979; Lindén *et al.* 1980). Many countries carry out routine plankton monitoring programmes, of which the most effective is probably the Continuous Plankton Recorder Survey of the North Sea and North Atlantic (Glover 1967). None of these surveys has yet shown trends that can be significantly correlated with oil pollution, though the effects of climate change can be seen (see, for example, Glover *et al.* 1974; Garrod & Colebrook 1978). The apparently harmless nature of spilled oil in the water column thus contrasts strongly with its toxicity to plankton organisms under experimental conditions and to intertidal and subtidal benthic communities.

Intertidal and subtidal benthos of sediments

Benthic infauna ecosystems are complex, and undisturbed stable communities can be very rich in species (Sanders 1968). A full understanding of the effects of oil is therefore possible only in regions where there is good knowledge of the taxonomy and distribution of the infauna, and an active research programme. These criteria are satisfied by the post-spill investigations carried out around Wood's Hole, in the Baltic and along the coast of Brittany. The following account is based on these North Atlantic studies, supplemented from certain other monitoring programmes.

The effects of oil are felt least on open coast intertidal sands where there is little macrofauna; in such beaches the meiofauna is killed or reduced by a spill, but can recover within a year (Giere 1979; Le Moal 1981). Sheltered sands show a high initial mortality, with reduction in biomass and number of species, although certain polychaetes, including *Arenicola* and some species of *Nephtys*, may survive and increase (Le Moal 1981 and personal observations; see also Gordon *et al.* 1978). As with finer subtidal sediments (see below), recolonization starts with opportunistic species, including capitellid and cirratulid polychaetes, and in the more sheltered beaches a characteristic 'pollution fauna' develops and persists for some years (Le Moal 1981). In Brittany, sheltered sandy beaches have retained oil from the *Amoco Cadiz* spill for more than 3 years, buried as much as 55 cm deep and only partly degraded (Long *et al.* 1981 and personal observations). Intertidal sediments in Nova Scotia retained oil for at least 6 years, and this was accompanied by elevated hydrocarbon levels in the tissues of clams, which had reduced growth rate and poor recruitment (Gilfillan & Vandermeulen 1978; Keizer *et al.* 1978; Thomas 1978). Oil from the *Torrey Canyon* persisted in beaches and subtidal sediment for at least 4–5 years (Southward & Southward 1978), even though large quantities of dispersants were used (Smith 1968): reports are still being received that dispersant treatment increases retention of oil by sands and finer sediments.

The animals of fine subtidal sediments are especially sensitive to fresh crude oil and fuel oils, and very high or even total mortalities have been reported (Blumer *et al.* 1971; Cabioch *et al.* 1980). The effects on burrowing filter-feeders in general, and on echinoderms, lamellibranch molluscs, ampeliscid amphipods and processid prawns in particular, are now well documented (Michael *et al.* 1975; Dauvin 1979; Sanders *et al.* 1980; Cabioch 1980; Elmgren *et al.* 1980; den Hartog & Jacobs 1980; Jacobs 1980; Glemarec & Hussenot 1981; Noel 1981). Recolonization of the denuded sediment begins first with opportunistic polychaetes (e.g. *Capitella* and *Mediomastus*), which reach great abundance in the absence of any other animals (references quoted above). Even within these fast-breeding species there may be a succession of single genotypes representing a range of tolerance of the oil (Grassle & Grassle 1974). Following this initial flush of one or two species, the biomass and species composition undergo a series of fluctuations of declining amplitude, extending for several years, until apparent stability is regained. In several studies where the effects of a spill have been followed in detail, at the worst oiled stations the fauna was still disturbed and the species composition abnormal at the time of reporting (from 1 to 5 years), as shown by indices of diversity, variation and similarity (see, for example, Sanders *et al.* 1980; Dauvin 1979; Le Moal 1981). The larger animals are slow to return, as are the ampeliscid amphipods, and this may be related to the length of time that the oil can persist in the sediments. Partly degraded oil can remain within subtidal sediments for over 5 years, and hydrocarbons can be detected in the tissues of resistant species and those recolonizing (Blumer *et al.* 1971; Keizer *et al.* 1978; Teal *et al.* 1978; Elmgren *et al.* 1980; Beslier *et al.* 1980).

Much less is known about the impact of oil on offshore and deeper water sediments. Little immediate effects on the fauna have been reported from spills and oil rig blowouts (Johnson 1979; Pratt 1978) but this may reflect lack of detailed knowledge of the particular ecosystem, since in some instances oil did reach the seabed. One long-term monitoring programme around an oil rig (Addy *et al.* 1978) has reported changes in the fauna that to some extent parallel those found in inshore sediments after acute oiling. Close to the rig a reduction in abundance of one polychaete was accompanied by an increase in the opportunistic cirratulid *Chaetozone*, apparently correlated with the total hydrocarbon content of the sediment (extractable organics, 100–400 $\mu g\ g^{-1}$). In contrast, Wiesenburg *et al.* (1981) claim that 20 years input of aromatics of low boiling-point to the Gulf of Mexico from an oil rig brine discharge has had no obvious harmful effect; Geyer (1981) thinks that thousands of years of natural oil seeps have had little effect on the fauna of the Gulf of Mexico. Neither of these accounts offers any biological evidence.

Chronic pollution from oil refineries produces well marked changes in the benthos of inshore sediments. Campaigns to clean up effluents and reduce their hydrocarbon content have led to significant recolonization, though not necessarily to full recovery (Reish 1965, 1971; Wharfe 1975; Leppäkoski & Lindström 1978; Dicks & Iball 1981). In contrast, the reconstruction of outfalls may merely shift pollution effects from one part of a bay to another (Bellan 1979). Some natural oil seeps are claimed to have little effect on the biomass and diversity of the nearby benthos (Spies *et al.* 1978), but in this example opportunistic cirratulids and oligochaetes are commoner closer to the oil sources, and instability is indicated. Some further understanding of the effects of chronic oil pollution on sediments and benthic infauna is provided by controlled ecosystem experiments. Continuous addition of oil to experimental enclosures leads to accumulation of hydrocarbons in the sediments (Grassle *et al.* 1981). There are significant changes in species abundance and diversity, as well as behaviour, at hydrocarbon levels between 120 and 620 $ng\ g^{-1}$ (Vanderhorst *et al.* 1978; Busdosh *et al.* 1980; Grassle *et al.* 1981). The macrofauna and total meiofauna decrease, though there may be increases in ciliates and foraminiferans. Elimination of the larger species and proliferation of some of the smaller produces a simpler community, an effect that is shown by other pollutants and is observed to a lesser extent in untreated enclosures held over a long time (Menzel 1977). The effect of oil is to force the change in community structure at a faster rate.

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Rocky intertidal and sublittoral

Rocky shores are usually more accessible to investigation than other marine habitats, and superficially they seem the easiest thing to study. However, caution is necessary in attributing a particular zonation or community structure or change in community to a single cause; abiotic factors such as temperature, desiccation and wave action operate very strongly on rocky shores (Sanders 1968; Lewis 1976), and modify the influences of competition and predation (Paine 1966; Connell 1972, 1975; Menge & Sutherland 1976). However, as a result of previous researches it has been comparatively easy to describe, if not interpret, the effects of oil on rocky shore communities of northwest Europe (Nelson-Smith 1972, 1978, 1979; Baker 1978; Giere 1978; Southward & Southward 1978). On wave-washed (high-energy) shores a single spill is transient, and the oil is usually removed from the midtidal region (eulittoral zone) within 2 years. In the supralittoral fringe and the splash zone above it may persist for 5–10 years. The biological effects of a spill on such shores are those of a severe, short disturbance, followed by a long period (5–10 years or more) of restoration.

The sequence of events takes much the same course whether the oil is a fresh crude, a distilled product, or weathered oil treated heavily with dispersants, and similar events have been reported after a 'red tide' (Cross & Southgate 1980; Myers *et al.* 1980). The initial impact kills most of the animals and some of the algae. In the absence of the grazing herbivores the algae recolonize first, and there is a 'flush' of ephemeral species, green or brown, according to season and latitude, followed by or accompanied by heavy settlement of the fucoids. Also in the absence of grazing, the infralittoral fringe algae, including the laminarians, may extend higher up the shore. The growth of algae is followed by recolonization of the grazing herbivores in larger numbers than before, but at first with fewer species, then the ecosystem proceeds in a series of slow fluctuations of different dominants ('damped oscillations') before diversity and equilibrium are restored (see Southward & Southward (1978) for earlier references; see also Floc'h & Diouris (1980) and Glemarec (1981)). Enough is known to provide a basis for conceptual models, and full mathematical models have been proposed to explain general community interactions on rocky shores (Seip 1980, 1982). The major changes are so easy to recognize that detailed pre-spill surveys may not be needed to follow the main events on shores where the fauna and flora are adequately known (Cowell & Syratt 1979). However, without detailed previous counts it is difficult to tell when a community has returned to normal. The slow return of animals leads to exuberant growth of algae in the first 2 and 3 years, giving a false impression of health and 'recovery' to superficial observers. As a result there have been many misleading statements in the 'grey' literature of pollution about 'rapid recovery' of rocky shores. There is evidence from photographs, not yet published, that even 10 years may not be long enough for the full recovery of shores in Cornwall. This is partly because many of the mature communities exist as a mosaic of patches with asynchronous succession, each patch having resulted from local disturbance due to heavy predation, changing abiotic factors or chance. The effect of an overall disturbance, such as that caused by oil, is to impose a uniform pattern of development on the whole shore. As a result diversity is reduced, as is stockholding capacity for the higher trophic levels and the long-lived grazers. The length of time needed for full recovery therefore depends not only on the rate of recruitment of the long-lived species but also on how long it takes to break the overall pattern imposed by the original disturbance.

In the colder regions of the North Atlantic the midtidal zone is without dominant grazers

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such as *Patella* and the top-shells, and changes after an oil spill are less conspicuous (Michael 1977; Thomas 1978). There are also less obvious changes after oil contamination of sheltered (low-energy) shores in the North Atlantic and Baltic. In such places, however, oil persists on the rocks for several years, and there may be additional damage due to release of oil trapped in nearby soft sediments and marshes. Resettlement may take place, but juveniles are subsequently lost or growth is reduced. Especially in colder climates, the recovery of sheltered shores is therefore more protracted than on high-energy shores, though the changes are not so spectacular (Thomas 1978, 1981; Notini 1980; Gundelach *et al.* 1981; personal observations).

Rocky shores of the eastern Pacific carry rich communities that are not yet fully understood, ecologically or even taxonomically. Animal associations of wave-swept rocks in the Oregonian faunal province exist as an ever-changing mosaic of patches undergoing succession of species, with turnover times of 8–35 years (Pain & Levin 1981). Exclusion of or change in population of the acmaeid limpets has little influence on local algal growth and succession, though it may enhance recruitment of barnacles (Sousa 1979; Kitting 1980). In these communities it appears that only very large gross mortalities would change the overall population structure, and it is therefore not surprising that a moderate spill of fuel oil on a rocky shore of Washington State apparently had little general effect (Clark *et al.* 1978). The sea-urchin, *Stronglyocentrotus purpuratus*, was affected during the first 10 months and there was also a long-term decline in abundance of barnacles, mussels and a sea-anemone, and hydrocarbon residues could be detected in the mussels for 5 years. A spill of Bunker C oil in San Francisco Bay killed some barnacles, limpets and crabs, but the changes observed were reported to be within the limits of natural fluctuations, and recovery was apparently achieved in 5 years (Chan 1977).

In contrast to these experiences, a spill of diesel fuel in northern Mexico (Californian transition zone faunal province) had long-lasting effects comparable with those observed after spills in the North Atlantic. There were immediate mortalities of molluscs and echinoderms (North *et al.* 1965). In the absence of grazers, especially of the sea-urchins and abalones, the giant kelp, *Macrocystis*, flourished, and formed an extensive kelp bed, passing through two complete successional cycles. Recolonization by animals began with barnacles and other species with planktonic larvae, but even 7 years after the spill the populations of sea-urchins and abalones were judged to be smaller than previously (North 1973). It is a fact that diesel oil can be extremely toxic, as noted in Hong Kong, where limpet populations took several years to recover from a small spill (Stirling 1977).

Much less is known about the effects of chronic oil pollution on rocky shores. The classic survey of the development of Milford Haven as an oil port shows the overall effect of the constant small spills to be slight (Baker 1976; Nelson-Smith 1972). However, there have been local changes comparable with those seen after acute spills or due to chronic general pollution (Smyth 1968): typically this amounts to increases in algal cover and decreases in grazing herbivores, including *Patella* (Crapp 1971; Nelson-Smith 1979), indicating a shift in the population balance towards a community more typical of sheltered shores (Baker 1976). The local changes around Milford Haven contrast with the apparently slight effects of much heavier oiling from natural oil seeps around Santa Barbara, California (Straughan 1981, and this symposium).

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Discussion

The impact of oil on marine life is an emotive and controversial topic, but for many years it was widely believed that oil was relatively harmless to marine life.

As experimental evidence accumulated for toxic effects, there was much talk of the 'real world'; petroleum technologists thought there was an ample margin of safety in dilution factors, and reports of field damage were discounted. A columnist for the *New Yorker* magazine (Wertenbaker 1973) has put on record the way in which a group of scientists at the Wood's Hole Oceanographic Institution were 'bitterly attacked – their abilities, their reputations, and their characters as well as their conclusions – for their reports on the West Falmouth Spill' (see Mackin 1973; cf. Blumer *et al.* 1970, 1971; Sanders 1974, 1978). There were also attempts to belittle the effects of the *Torrey Canyon* spillage; it was extensively circulated that the oiled and dispersant-treated shores can recover in 2 years (Cole 1971; Steinhart & Steinhart 1972; Mackin 1973; Wardley Smith 1976) and other scientists were misled (Mellanby 1972; Hyland & Schneider 1976). The myth of rapid and complete recovery of oiled rocky shores is still appearing in the literature (Mackay 1978; Wyman 1978; Burke & Golob 1979). Other attempts to minimize the ecological effects of oil include the suggestion that 'small spills seldom result in measurable adverse effects, and if measurable, they are slight and short-term' (Leek *et al.* 1981). This sort of argument supposes that there is a threshold concentration below which the presence of oil is not important. It is true that some experiments show a nonlinear response, but the tailing-off of lethal or sublethal effects is obtained at such low concentrations that physical factors, especially surface phenomena, will lead to a non-uniform distribution of oil. Another argument is sometimes used to excuse not searching for the effects of pollution: that natural changes are so large, we need not be concerned about the much smaller effects of man's activities. The fallacy here is that man-made effects tend to be persistent, additive or cumulative, and hence there is no obvious reason why they cannot be detected over and above seasonal trends and other natural fluctuations (Glover 1977). Nevertheless, this argument continues in various forms (see, for example, Cushing 1979; Lewis 1980); and it is perfectly true that existing monitoring of trends in plankton and fish populations shows extensive natural fluctuation, and the consequences of overfishing, but nothing that can be significantly correlated with pollution (Cole 1979*a*; Sherman *et al.* 1981). It may be that a response has not yet begun to show or that our monitoring is not sensitive enough.

However, there is no doubt from the evidence reviewed here that oil is toxic to marine organisms; on this point there is good agreement between laboratory experiments and the field observations on intertidal communities and subtidal benthos. There is also no doubt that at least part of the results of laboratory investigations into long-term sublethal effects are borne out by evidence from the field: where petroleum hydrocarbons are still present, growth is reduced and recruitment fails. It is more difficult to apportion the blame for the field effects between the direct action of oil on cell membranes and indirect responses of the metabolism or the nervous system to the gradual accumulation of hydrocarbons in the tissues. Much of the impact of an oil spill can be explained by the effects of the hydrocarbons on cell membranes (Marsland 1933; Sollman 1949; Mullins 1954) with subsequent alteration in van der Waals forces, swelling of the membrane, and increases in permeability (Seeman 1972; Heldal *et al.* 1978; Haydon *et al.* 1980). Empirical evidence shows an inverse relation between solubility in water and narcotic or toxic effects of the aliphatic and aromatic series (Hutchinson *et al.*

1979; but see also Crisp *et al.* 1967; Le Roux 1976). There is a commonly held view that the initial impact of a spill is due to the low boiling-point fraction, including the lighter aromatics (see references in Hutchinson *et al.* (1979), but cf. Baker (1971)). This may be so for short periods, when the first effects would be narcosis or a change in behaviour; lethal effects would be secondary, and associated with predation or failure to hold onto the substratum or maintain burrowing, so that vulnerability to abiotic factors such as waves and tides was increased. However, for periods beyond a few hours, the less soluble alkanes and the higher aromatics such as anthracene and pyrene will be more important toxic agents. Changes in membrane permeability can explain many of the sublethal effects. This is obvious when ion balance is altered, but changes in behaviour probably also result from interference with the chemical senses after alteration of membrane properties (Atema & Stein 1974; Miller 1980). Many of the physiological and biochemical changes may be a reaction at the cell level to membrane permeability changes, and a product of the increased work needed to maintain cell processes (Sanders *et al.* 1980). Reproductive failure due to regression of the gonads could stem from the same basic cause. The effects of oil on filter feeders in particular, and their delayed recovery in natural communities (see above) might also be explained by membrane effects, especially where ciliated epithelium is involved in the filtration process.

Other effects of prolonged sublethal concentrations of oil, such as those that result in abnormal young after exposure of the adult during gonad maturation (Kühnhold *et al.* 1979), are more likely to be related to accumulation of hydrocarbons in the tissues (GESAMP 1977; Lee 1977; Corner 1978; Connel & Miller 1981*a*, *b*), with additional complications related to the induction of lesions and genetic changes (see Sindermann, this symposium; Beardmore *et al.* 1980).

From the ecological point of view these differences, and the apparent differences between acute and sublethal effects, are not so important. The overall effects of oil are the same as any other disturbance to the community: after a single exposure to oil some or all of the organisms die, and replacement occurs through a succession of opportunist species until diversity and stability are regained. Under chronic pollution the community is destabilized by mortality of the more sensitive species or by elimination of species that are rendered unfit without actually being killed; the place of such species is taken by opportunists, which are not necessarily the most resistant to oil under laboratory conditions (Gray 1980*b*). In other words, a community that persists under chronic pollution is comparable with that found during the recovery phase after an acute pollution. The sensitivity of a community to chronic pollution, and the time needed to recover from acute pollution, depend on the complexity of the original population. Communities already living under some stress, whether environmental or otherwise, may show a smaller initial impact, and may return to 'normal' faster than less stressed communities, but much will depend on the complexity of the biological interactions.

If oil does have deleterious effects on benthic and intertidal communities, are we to continue laboratory experiments and enclosed ecosystem experiments as well as monitor natural communities? The experimental approach has shown convincingly that petroleum compounds are toxic or have sublethal effects on the behaviour, biochemistry and physiology of the test organisms. It appears that what is needed now is an extension of work into mutagenic effects and into selection of genotypes, to cover not only the possible effects of oil but all forms of industrial and urban input of pollutants into the sea. Such research should be regarded as basic, leaving applied scientists to work out how to reduce input of pollutants. For the latter purpose

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it is useful to have available test organisms for monitoring effluent quality, though this should not be regarded as a substitute for improved chemical analysis. Cole (1979*b*) has suggested reliance on economically important species for bioassay, and deprecates the search for more sensitive species. Economically useful test organisms already exist in the mussel and the oyster, the latter most likely to be available in a purer genetic line. Burns & Smith (1981) recommend the simple and elegant monitoring system that employs the mussel's capacity for accumulating hydrocarbons in its lipids to a much higher concentration than in the water; the lipids are then examined by gas chromatography and the hydrocarbons identified. If we could recognize low-level pollutants in this way we could calculate probable solubilities in water by physico-chemical methods, and predict their effect on other organisms by comparison with existing data. Such a programme would provide information more rapidly than time-consuming measurements of metabolism and growth, which may have no greater sensitivity than more commonly used ecological techniques (Gray 1980*b*). Biochemical monitoring, such as measurement of tissue oxidase activity, is comparatively rapid (Lee *et al.* 1980), but when significant increases have been reported there are usually accompanying changes in community structure and detectable levels of hydrocarbons in the environment (Sanders *et al.* 1980). It would seem that the main advantage of physiological and biochemical monitoring methods is not the detection of lower levels of pollutants but the use of quantitative techniques that rely on sophisticated analytical instruments rather than trained biologists. Such monitoring gives no indication of change in species composition or community structure, except when the organism being analysed disappears, and it would thus be difficult to assess the degree of damage to the ecosystem or predict future trends if pollution were to continue or were abated.

If there is no substitute for ecological monitoring, what are we to make of the evident disagreement among biologists about the value of continuous monitoring programmes and environmental impact statements? With regard to pre-pollution surveys, Hedgpeth (1973*a*) has criticized many for being inadequate and inept; in contrast, Cole (1979*c*) believes that too much emphasis has been given to transient and minor unimportant aspects, and praises the critical path approach adopted in the U.K. for radioactive discharges. Ghiselin (1975) can see no reason why baseline surveys and environmental impact assessments should not also be contributions to knowledge, especially as there are still many regions where not enough is known about the biota and their interactions.

Many surveys are made to satisfy legislative requirements only, whether carried out by government servants, independent consultants or industrial staff: some are padded out more than others with irrelevant detail to justify the expense, and few are refereed before being circulated (Rosenberg *et al.* 1981). Even the best monitoring leads to discussion about objectives and results. There is a tendency among some ecologists to try to reduce their findings to a single index or mathematical expression, apparently in the naïve belief that this will make their conclusions more acceptable to lawyers and engineers (Kaesler *et al.* 1978). This attitude has been criticized by Hedgpeth (1973*b*), who has also commented unfavourably on a parallel trend towards reliance on models as a substitute for field work (Hedgpeth 1977). Nevertheless, more papers are appearing devoid of biological data but replete with transformed statistics and mathematical formulae, from which it is impossible to retrieve any of the original facts: surely it is possible to simplify raw data without obfuscation?

In theory, the various diversity indices used by ecologists ought to convey more meaningful information than some of the other statistical parameters. Unfortunately they are all affected by sample size and considerably changed by the addition or subtraction of even one species (Peet 1975), and are thus truly comparable only when used by one worker or a team using the same sampling methods over a given geographical region or faunal province. The studies after the West Falmouth spill (Sanders 1978; Sanders *et al.* 1980) have shown the value of measurements of fidelity, discrepancy and variation in comparing the degree of disturbance to the ecosystem at stations with somewhat differing faunas. The log-normal distribution can also give a clear picture of benthic population instability induced by pollution and subsequent recovery, and is comparatively easy to use (Gray & Mirza 1979; Gray 1980*a*). However, none of these methods is a complete substitute for proper comparisons of species and structure, and all require the same detailed sampling and analysis of the samples. Sampling frequency and sample size are the basic problems of any monitoring programme. A complete study of primary producers really entails 28 samples a year (Gray 1980*b*), and monthly samples may not be enough to follow changes in a community that shows large seasonal fluctuations in species and abundance. Macroplankton and young fish monitoring suffers from lack of information if samples are taken less often than once a week (personal observation). An alternative approach is possible where the animal population contains a good proportion of species which live longer than 1 or 2 years. Under stable and relatively undisturbed conditions it is then feasible to reduce the sampling frequency to once a year, choosing a period when seasonal fluctuations are least. This method has worked well for infaunal benthos off Northumberland (Buchanan *et al.* 1978) and has disclosed medium and long-term natural trends in intertidal barnacles (Southward 1967; Southward *et al.* 1975). However, annual sampling does not allow a full assessment of the role of abiotic factors that operate on the recruitment of juveniles, and which may be an important element in controlling adult population density (Lewis 1977; Cushing 1981).

As already noted, the effect of oil, and many other forms of pollution, is to undermine community structure and destabilize the ecosystem, driving it in the direction of monoculture. The synergistic effects shown between crude oil and dispersants suggest that other forms of pollution, such as by heavy metals, would also increase the toxicity of oil (Malins & Collier 1981). Unstable ecosystems can show temporary increases in biomass of a few organisms, which, if regularly harvested, might increase the total yield of organic production taken from the sea, e.g. seaweeds, or 'trash' fish for reduction to meal. At present there appears to be a public consensus that oil produced from the seabed, or carried over the sea, is a preferable source of power to nuclear energy. However, until there is a public consensus to maximize all output from the sea, including the production of organic matter, there is an ethical obligation to interfere as little as possible with existing natural communities. In other words we have to prevent oil reaching the ecosystem, or mitigate the effects. There are surprising ecological benefits from even the most modest clean-up campaigns: it might be better to channel more of the funding for applied research in this direction.

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Discussion

Baroness White (*House of Lords, London, U.K.*). I speak as a legislator, not a scientist. We have been classed by Dr Southward, together with engineers, as simple people. We have to be. As Chairman of the House of Lords Select Committee on the European Communities, I am very much aware of the interaction between science and technology, on the one hand, and legislation on the other, not least in the very detailed E.E.C. Directives and Regulations that emanate from Brussels. The group of Directives on water quality and fresh water and marine pollution is but one example.

We need sound scientific advice, but it must be at a practical level. We cannot wait for scientific perfection. We need to deploy public resources of money and trained manpower economically, but in a way that commands scientific respect. So there must be mutual understanding.

A. J. Southward. I had no intention in grouping legislators and engineers together, other than they were the other persons most likely to be interested in the results of biological monitoring. I was criticizing those biologists who attempt to reduce their results to a single index, but who are adding to the confusion rather than simplifying, since a single biological index, unlike an engineering modulus, for example, cannot convey the information needed. The E.E.C. water quality Directives are a good example of too much reliance on apparently quantitative indices rather than on commonsense scientific opinions.

S. L. Vader (*University of Tromsö, Norway*). I should like to hear more about the effect of oil on membranes and who has carried out this work. In our laboratory we have studied the effect of oil on eggs, sperm and the fertilization process, and we have not found any acute effect such as we find with several of the newer oil dispersants.

A. J. Southward. I apologize for too brief a reference to the work on squid giant axons (Haydon *et al.* 1980), in which it was found that aliphatic compounds (not crude oil) caused changes in transmission and capacity consistent with the theory that hydrocarbons act by causing swelling of the membrane. Possibly the squid axon is more suitable for this sort of experiment than egg membranes, being more robust.

J. S. Gray (*University of Oslo, Department of Marine Biology and Limnology, Oslo, Norway*). Dr Southward has argued convincingly for ecological monitoring of effects of oil spills. Yet in my experience a lot of effort is wasted on attempting to study all possible species. In any given community 60–70% of the species are rare, represented by one or two individuals, and we do not know why they are rare. It is inordinately time-consuming to sort and identify these species and they are rarely used in analyses (e.g. most multivariate methods remove rare species). Most effort is given to quantifying the three or four commonest species that predominate in number or biomass. However, I believe that subtle changes in community structure after pollution incidents affect primarily a group of 10–12 neither rare nor common species. Rather than advocating ecological studies alone, I should like to see a suite of techniques, biochemical, physiological and genetic, applied to these ecologically sensitive species as the ideal way to study long-term effects of pollutants.

D. J. Crisp, F.R.S. (*Marine Science Laboratories, Menai Bridge, Gwynedd, U.K.*). A major problem in comparing benthic communities is that we cannot see them *in situ*, so that the information acquired by survey consists of random samples of animals grouped into taxa. Diversity indices, though now recognized as being of very limited ecological application, are the only way of combining such impoverished data into a single figure. The great advantage of studies of rocky shores is that the investigator can see the communities, their relation to topography and to each other, and he can come back year after year to survey the same spot, with clear implications as Dr Southward showed in his photographs. No one reports diversity indices of shore animals since we can use much more informative approaches. It is not surprising that the effects of oil pollution are more difficult to demonstrate from infaunal benthos sampling, cost a great deal more, and that negative results are less convincing.

A. J. Southward. I agree with Professor Crisp that sampling infaunal benthos is much more complicated than observing rocky shores. However, the number of diversity and other indices proposed by different researchers amply illustrates their dissatisfaction with any one index. We must avoid the position where a single index might be selected and made a legislative requirement of pollution monitoring. In this connection, the suggestions made by Professor Gray, for combined monitoring and experimental tests on a suite of species, would seem to offer a solution.