PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

The Effects of Repeated small Oil Spillages and Chronic Discharges [and Discussion]

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The effects of repeated small oil spillages and chronic discharges

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The value of long-term biological studies of single spills, repeated small spills and chronic sources of pollution has long been recognized, but relatively few examples are to be found in the scientific literature.

A small number of case histories of long-term studies are described here that illustrate a range of biological effects resulting from chronic inputs in the form of repeated small spillages, refinery effluents and discharges from offshore platforms. These studies have taken place around refinery effluents and spill sites in Milford Haven between 1960 and the present, around a refinery effluent in Southampton Water from 1969 to the present and in two oilfields in the North Sea since 1973 and 1975 respectively.

Observed effects of chronic discharges in coastal waters range from localized and subtle to severe and long-lasting. In the producing offshore oilfields investigated so far, widespread severe effects have not yet been observed and such fields may present relatively few long-term biological problems. An important gap in knowledge that deserves attention is the biological effect in those fields where oil-based drilling muds have been used extensively, as preliminary evidence suggests that substantial quantities of oil can reach the seabed from this source.

While it is recognized that the total impact of any pollutant on marine systems may be impossible to measure, the factors that determine the form and size of biological effects may be induced from the above examples and other examples from the scientific literature. A summary of these factors is presented. It is concluded that the severity of effects depends largely on the type, nature and duration of the pollutant, the physical nature of the receiving environment, and its biological nature and sensitivity. Where pollutant loads are reduced or removed (naturally or by pollution control measures), biological damage is usually followed by recovery of the system to an apparently healthy condition. Subtle effects may, however, persist for considerable periods of time.

Introduction

Much public concern has been expressed about the effects of oil on marine life and this disquiet is reinforced by the well publicized and obviously damaging major tanker accidents. Of greater concern are the long-term chronic inputs around point sources, resulting from the production, handling and refining of oil. Some case histories of long-term studies of chronic inputs from refinery effluents, repeated spills and North Sea oilfields are summarized here. In no case has it been possible to look at effects in all components of marine systems; rather, one or more species or communities and the changes in them along spatial gradients and with time have been studied. From the findings of these studies, inferences have been drawn about the magnitude and persistence of effects.

Milford Haven, an oil port since 1960, has received chronic inputs of oil from spills and dispersant from their clean-up, and of oil and other components from refinery effluents for more than 20 years. Biological data for some habitats in the Haven are available from 1961 and have been added to by more recent studies. Chronic effects from a relatively small effluent from a modern refinery, discharged since 1960 into Littlewick Bay (a rocky bay), are described and contrasted with the effects of a large effluent from an older refinery discharged since 1953 to a saltmarsh creek system in Southampton Water.

In these long-term examples (20 years and more), information from a variety of sources has been used to assess changes. Problems of comparison and interpretation have cropped up because the earlier studies were not always planned to cover a variety of habitats or rigorously relate cause and effect. Over the years the monitoring programmes in these areas have been modified, improved and supplemented by additional studies to aid interpretation of observed changes. The recognition of mistakes and omissions from the earlier studies has proved invaluable in designing programmes that can obtain series of comparable data sets and better identify causal relations.

The recent expansion of oil exploration and production offshore in the North Sea has drawn attention to potential biological effects. Two examples of more recent monitoring schemes and their findings in North Sea oilfields since 1973 and 1975 respectively, are summarized and discussed.

Milford Haven after 20 years as an oil port

Milford Haven (figure 1) is a drowned river valley in the southwestern corner of Wales, which receives fresh water from the eastern and western Cleddau and several smaller streams such as the Pembroke River, Gann River and Sandyhaven Pill. The total freshwater input is about one hundredth of the tidal volume (tidal range 8.3 m on spring tides), and this results in high salinities through most of the Haven (33‰). Characteristic estuarine conditions are found only in the upper reaches of the Haven and around the mouths of rivers and, to a lesser extent, streams. The physical characteristics of the shores and water body are detailed in Nelson-Smith (1965).

The natural deep-water channel that extends into the Haven as far as Wear Point, in combination with shelter, makes it an ideal site for an oil port. Since the commissioning of the first refinery in 1960, three further refineries have been built (two in 1964 and one in 1973) as well as an oil-receiving terminal and pipeline (1961). The total annual capacity for refining is now *ca.* 28 Mt. All the refineries have extensive jetty systems, four well offshore and a fifth close inshore at Wear Point (figure 1). In addition to the refineries, a 2 MW oil-fired power station was opened in 1970. The power station and refineries all produce effluents consisting of drainage water, process water and cooling water, which contribute varying pollution loads to the Haven.

As well as being an oil port, the Haven, which lies within a National Park, has been designated an Area of Outstanding Natural Beauty. Biogeographically, it occupies a position at which the ranges of many Lusitanean and boreal species overlap in their distributions, and it possesses a very varied marine flora and fauna. Biological studies in the Haven date from the earliest stages of the recent oil industrialization and provide a description of some marine communities against which changes since that time can be assessed.

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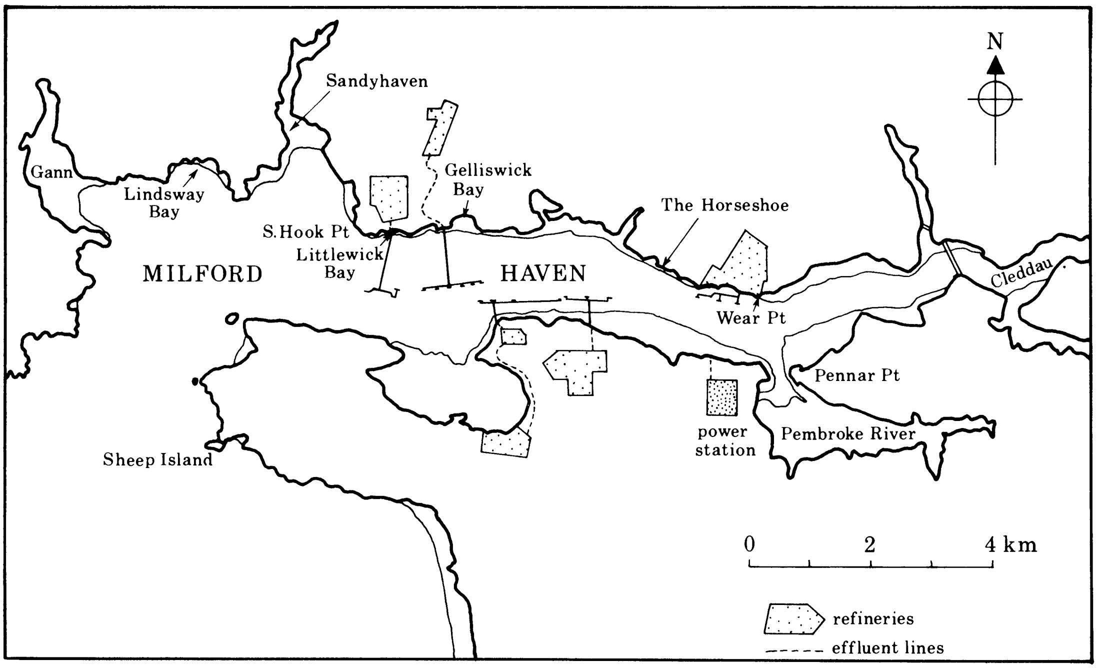

Figure 1. Milford Haven, showing the location of refineries and places named in the text.

Oil inputs to the Haven

Oil and other pollutants enter the waters of the Haven from a variety of sources. The most important are refinery effluents, oil spills, sewage and urban runoff. Comprehensive records of oil spills have been kept by the Milford Haven Conservancy Board (the port authority), and a summary since 1963 is given in table 1. Many of the recorded spills have been treated with dispersant chemicals, both at sea and on some rocky shores. A summary table of numbers of spills treated from 1977 to 1979 is given in table 2.

The quality of refinery effluents has changed since operations began in 1960. Before 1971 water authority specifications permitted the discharge of 50 mg l^{-1} oil in effluent, but in 1971 this was reduced to 25 mg l^{-1}. It is, however, difficult to compare effluent quality before and after 1971 because the early information is limited. Crapp (1970) estimated that in 1970 the three refineries then in operation were contributing some 270000 l of oil per year to the Haven. Abbiss *et al.* (unpublished data) have estimated current inputs from the effluents to be in the region of 200000 l per year, and the total input from all sources to be approximately 270000 l per year. Chronic inputs can thus be assumed to have peaked in the late 1960s when the 50 mg l^{-1} specification still applied and some 30–40 Mt of oil a year were processed, and the early 1970s when oil throughputs of the refineries were at a peak (43–59 Mt a year). A steady increase in input can be postulated up to that time with a slow decline since 1975 corresponding with a decline in tonnage handled. Several of the shores in the middle Haven have been subjected to repeated spills during its history as an oil port, and recent studies (Abbiss *et al.*, unpublished data) have found elevated levels of fresh and weathered oil in intertidal and subtidal sediments (5–480 μg g^{-1} total aliphatics). Similar levels have been found in other industrialized harbours.

TABLE 1. OIL SPILL STATISTICS, 1963–80, MILFORD HAVEN:

(All figures provided by Milford

	1963	1964	1965	1966	1967	1968	1969
(1) tankers in passage and at moorings	5	—	—	2	1	2	3
(2) tankers at jetties and misc.	15	26	81	55	44	38	49
(3) jetties	8	8	2	15	5	12	6
(4) total no. of spills	28	34	83	72	50	52	58
(5) total spills/t	9.8	8.8	35.8	30.5	267.4†	13.6	16.5
(6) total oil tonnage/Mt	13.0	17.7	24.9	28.9	28.2	30.0	39.9
(7) total no. of ships involved	1236	1392	1985	2378	2680	2669	3226
(8) $10^5 \times$ percentage spilt	8	5	14	11	95	5	4
(9) no. of spills per megaton	2.1	1.9	3.3	2.5	1.8	1.7	1.4
(10) no. of spills per 100 ships	2.3	2.4	4.2	3.0	1.9	1.9	1.8

Includes all industry operations including ballasting and bunkering. Refinery data omitted.

† Includes a 250 t spill from the *Chrissi P. Goulandris*, which suffered extensive damage at sea and entered the Haven holed (15 m × 2 m).

‡ Includes 150 t from a single spill involving a product carrier that ran aground in the Haven.

Biological studies in the Haven

The longest series of data for the Haven has been for rocky shores (1961–79); Moyse & Nelson-Smith (1963) and Nelson-Smith (1967) looked at the abundance and distribution of about 60 common species of algae, animals and lichens on belt transects at 30 main sites within the Haven between June 1961 and October 1963. Although providing only semi-quantitative data for each species, the method provided a clear picture of vertical zonation patterns in relation to tidal height and also of change in shore communities with respect to increasing shelter from wave action within the Haven. Crapp (1970) used the same approach, modifying the method and abundance scales slightly to simplify field use and make the categories of abundance more relevant to observed differences in densities between shores. Crapp concluded that after 10 years of industrial activity, there had been no overall decline in the shore communities as a result of oil inputs to the Haven, and changes in animal distribution caused by oil could only be found at sites of spills or immediately around effluent discharges. The changes had taken the form (now documented for so many spills) of death of grazing gastropods followed by increases in abundance of algae and subsequent declines in other animals such as barnacles. Following the weathering and removal of the oil, communities have been found to recover over a variable period of years. Crapp did, however, identify long-term changes in barnacle populations and the gastropod *Monodonta lineata*, which he attributed to climatic change. These changes took the form of an increase in density and vertical range on the shore of *Semibalanus balanoides* over most of the Haven, increases in density of *Elminius modestus* on the lower shore, parallel decreases in *Chthamalus stellatus* on the lower shore, and a general decline in abundance and distribution of *M. lineata*.

Since 1970 the abundance scale–transect technique has been further developed to include a greater number of species (now more than 120, though 15 animal species are recorded only as present or absent) supplemented by counts in quadrats of key species known to be affected by oil pollution. Some caution must be used in comparing and interpreting the data, for reasons that include local microtopographical variation, seasonal variation in communities, worker variability and changes in methodology over the years. Only changes in abundance of two or more categories over large areas of a shore are now regarded as significant, except where continuous trends of small changes are found over long periods.

SPILL NUMBERS, VOLUMES, OIL THROUGHPUT AND TANKER NUMBERS

Haven Conservancy Board.)

1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	
3	2	2	3	2	—	1	—	1	—	1	(1)
46	36	46	32	32	23	21	20	32	32	23	(2)
6	11	8	11	9	2	7	8	9	8	6	(3)
55	49	56	46	43	25	29	28	44	40	30	(4)
14.9	161.0‡	17.8	2316.5§	14.5	57.2‖	63.9¶	10.1	10.8	13.1	4.9	(5)
41.3	43.1	45.7	53.1	49.2	44.7	43.0	38.5	40.8	41	38.8	(6)
3400	3490	3465	3659	4186	3331	3576	3567	3679	3997	3933	(7)
4	37	4	436	2	13	15	3	3	3	1	(8)
1.3	1.1	1.5	0.9	0.8	0.6	0.7	0.7	0.9	0.8	0.7	(9)
1.6	1.4	1.6	1.3	1.1	0.7	0.8	0.8	1.0	0.8	0.7	(10)

§ Includes 2300 t from a single spill involving a product carrier, the *Dona Marika*, which drifted ashore in the Haven.

‖ Includes 50 t from a single spill.

¶ Includes 50 t from a single spill.

Table 2. Spill statistics for Milford Haven, 1977–9 inclusive: numbers of spills treated with dispersant chemicals

year	total number of spills	number sprayed with dispersant	percentage sprayed with dispersant
1977	28	19	68
1978	44	37	84
1979	40	30	75

The most recent studies (Little, unpublished data) show a very similar overall picture to that found by Crapp. No widespread decline in the shore communities was evident in 1979 after almost 20 years of oil input. Considerable numbers of small variations between sites have been noted over the years and some species have shown more general changes in abundance or distribution within the Haven. The barnacle populations have continued to change since 1970, but the trends are somewhat different to those noted by Crapp. *Semibalanus balanoides* had declined in density by 1979 on most shores, *Elminius modestus* had increased in density and *Chthamalus stellatus* and *C. montagui* had extended their spatial distribution further into the Haven and increased in density at most sites. Similar changes have been noted for other shores in the British Isles, and must be considered unrelated to oil inputs to the Haven. *Monodonta lineata* had continued to decline and it is not known if this was a phenomen peculiar to the Haven or part of a more widespread occurrence.

Changes have also been noted in the distribution of fucoid algae within the Haven. *Fucus vesiculosus* var. *linearis* has spread into more sheltered areas of the Haven, with a corresponding decrease in abundance of *Ascophyllum nodosum* at the same sites. *F. serratus* now occurs less far up the shore, and *F. spiralis* and *Pelvetia canaliculata* have decreased in range and abundance. These changes are believed to have resulted from climatic change, and the hot summer of 1976, which affected many shores in the British Isles, may have been at least partly involved. Reduction in the extent of barnacles and limpets up the shore has also been noted and may well be attributed to the same cause. The absence of widespread effects from oil pollution in the Haven is further supported by the species richness in a range of sublittoral habitats (Addy 1976, 1979; Little & Hiscock, unpublished data).

It is interesting to note that after a large spill of petrol (*ca.* 2–3 kt) in Lindsway Bay from

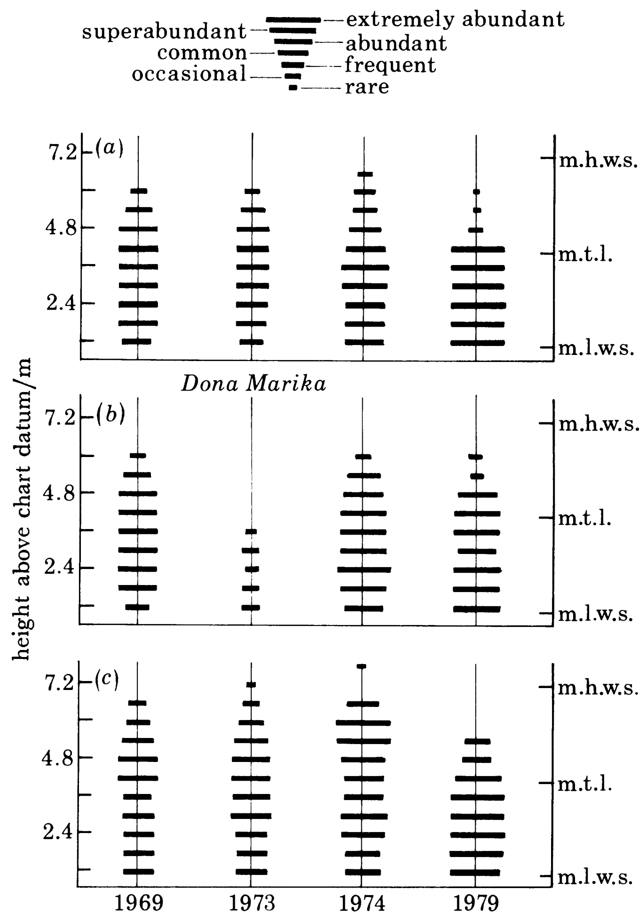

FIGURE 2. The abundance and distribution of *Patella vulgata* at (*a*) Musselwick East, (*b*) Watchouse Point and (*c*) South Hook Point, Milford Haven, in 1969, 1973, 1974 and 1979. The *Dona Marika* grounded in 1973 near Watchouse Point.

the *Dona Marika* (Blackman *et al.* 1973; Baker 1976*a*), which caused massive mortalities of limpets (figure 2) and barnacles over *ca.* 3 km of shoreline, recovery now appears to be complete (figure 2). Recently Abbiss *et al.* (unpublished data) reported an interesting pattern of hydrocarbon distribution in intertidal and subtidal sediments that shows no clear correlation of high levels (up to 480 μg g^{-1} total aliphatics) with sources of industry input such as jetties or effluent discharges (see figure 3). Rather, the levels of hydrocarbons show an excellent correlation with the proportion of mud (particles less than 63 μm in diameter) in the sediments (figure 4), the highest levels of which have been found in the inner reaches of the Haven. The sources of the oil in these sediments are postulated to be spills and well dispersed effluents carried to the inner Haven by tidal action over the years, together with contributions from shipping activity and leisure boating around Pembroke Dock and from urban runoff and sewage.

One site in the middle Haven has shown progressive changes in the distribution of organisms in recent years. In 1963 a rocky shore at The Horseshoe (figure 1) was dominated by limpets and barnacles, but with species of fucoid algae to be found over the whole shore. By 1970 a decline in algae and an increase in abundance of gastropods and barnacles had occurred. These changes are illustrated in figure 5. In 1978 increases in algal distribution and density (of *F. vesiculosus* in particular) had occurred over the middle of the shore, coupled with a decrease

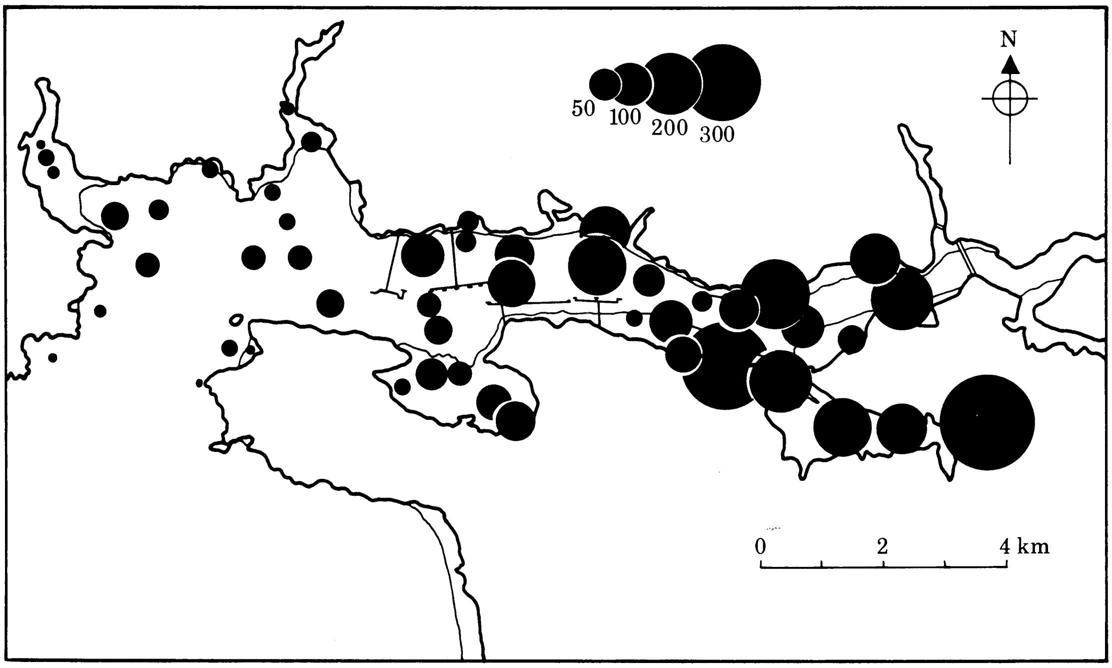

FIGURE 3. The distribution of hydrocarbons (total aliphatics) in subtidal and intertidal sediments, Milford Haven, in November 1979. The area of each circle is proportional to concentration (micrograms per gram). From Abbiss *et al.* (unpublished data).

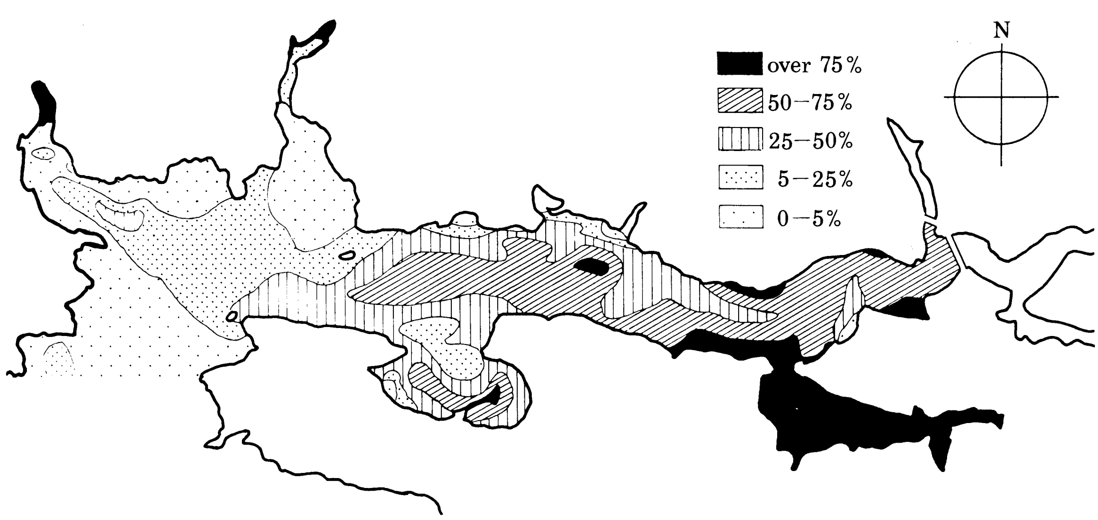

FIGURE 4. Percentage of 'fines' (particles less than 63 μm in diameter) in sediments in Milford Haven, November 1979. From Abbiss *et al.* (unpublished data).

in barnacle and gastropod density in the same area. By 1979 (figure 5) gastropods and barnacles were absent from the middle shore, and further increases in algal density had taken place. The changes are consistent with damage after oil spills, and indeed this site is immediately adjacent to an oil jetty from which eight spills were reported in the period 1977–9. While cause and effect may be implied, there is no hard proof.

Other shores in the Haven are known to have been oiled repeatedly without major changes in communities. For example, the east side of Gelliswick Bay below Hubberston Fort the shore was oiled on a total of seven separate occasions in 1977, 1978 and 1979 and the beach was cleaned once by dispersant spraying. Crapp (1970) reported the shore to be dominated by limpets and barnacles with light cover by fucoid seaweeds and this was still so in 1979 (Little, unpublished data). No significant changes to the shore between the two surveys were found in the data.

It is interesting to compare the changes noted above with those investigated experimentally by Baker (1971*a*, *b*, 1977) in saltmarsh communities. In the latter, effects of repeated oilings

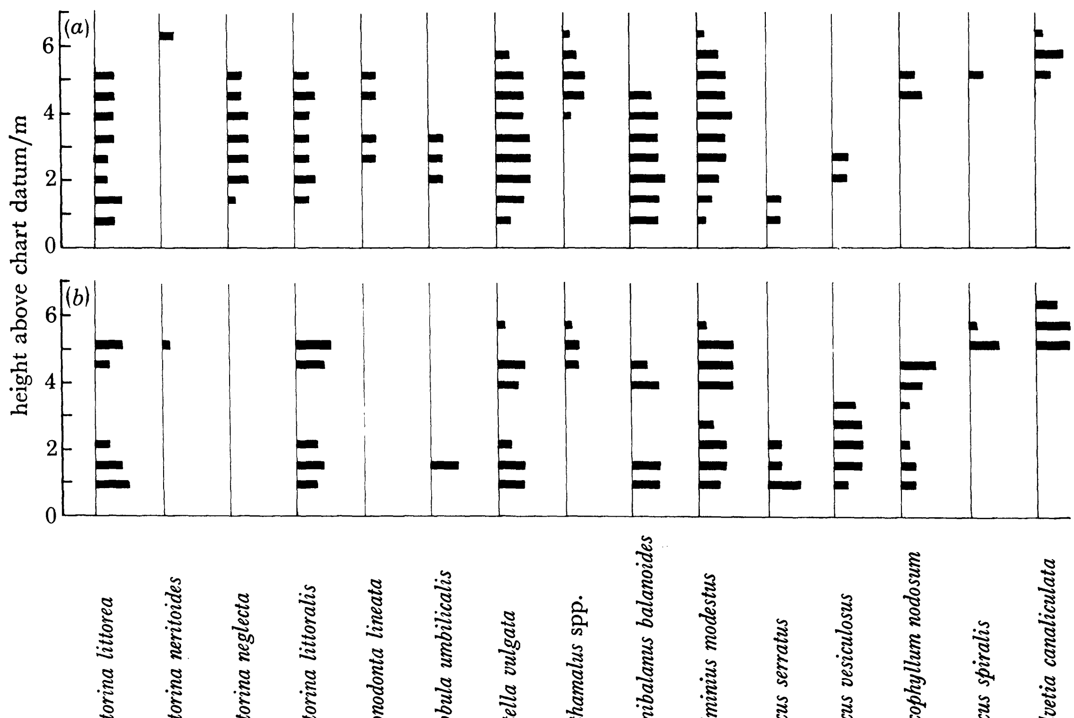

FIGURE 5. The abundance and vertical distribution of gastropods, barnacles and algae at the Horseshoe, Milford Haven, (*a*) 10 March 1970 and (*b*) 29 March 1979.

at monthly intervals up to a maximum of 12 months were severe for many marsh species and could persist for many years. The reasons for differences in effects of repeated spills in different habitats are many and include the degree to which pollutants may be retained and subsequently weathered and removed, and the sensitivity of organisms present. Stressed, physically controlled, high-energy habitats like exposed rocky shores appear least affected by chronic pollution; sheltered, sedimentary, low-energy habitats like saltmarshes and mangrove swamps appear relatively easily affected, and damage may persist for long periods.

It is apparent that no general decline in the marine flora and fauna has occurred as a result of pollution. But why should observed effects of oil in the Haven be so localized considering the large inputs of oil to the system over a long period of time? Undoubtedly, numerous factors are involved (Dicks 1976*a*), but two of the most important are the efficient management of the port and its extensive tidal flushing. Residual water volumes at low water of spring tides have been calculated at 280×10^6 m^3 and the volume at high water of spring tides to be 600×10^6 m^3, a difference of 320×10^6 m^3 (Dicks 1976*a*). Water exchange of this order of magnitude ensures rapid and wide dispersal of oil slicks and effluents. Efficient port management has had several effects. Excluding larger spills of 50 t or more, which happen infrequently, only 5–35 t of oil are spilled from 13–59 Mt handled each year. Secondly, even small spills are usually rapidly sprayed with dispersant or, more recently, collected by using booms and skimmers, and this, together with rapid dilution and water exchange, has been responsible for reducing the number of shore oilings without necessarily overburdening the water column and subtidal areas with hydrocarbons. Thirdly, water authority regulations governing effluent discharges (discussed in the next section) have resulted in levels of chronic inputs with which the system appears to be able to cope. Several of the effluent streams are also discharged from jetties or, in one case, a headland, where dispersion characteristics are good, further reducing effects.

It is evident that careful management of effluent discharges combined with an understanding of the physical and biological nature of the receiving environment is vital to the minimization of damage.

Refinery effluents

Refinery effluents contain a wide range of oils, products and other chemical constituents, may be above ambient temperature, and contain suspended solids, all of which may variously affect marine systems into which they discharge. Many of these factors have been examined and summarized in Baker (1979). To illustrate in some detail the range of effects that can be found, two examples of chronic pollution from refinery effluent discharges are compared and contrasted below. The first, from a relatively modern and mainly air-cooled refinery, discharges to a small bay (Littlewick Bay) on the north side of Milford Haven (figure 1). The second, from an older and mainly water-cooled refinery, discharges to a saltmarsh creek system on the west side of Southampton Water at Fawley. There are substantial differences in processing and throughput of oil in the two refineries, summarized in table 3, and these considerably affect the quality and toxicity of the two effluent streams.

Both refineries have also changed the quality of their effluent streams over the last 10 years, one to meet new water authority oil-in-effluent specifications (Milford Haven) and the other to effect improvements in production of visible oil films in Southampton Water and to allow recovery of damaged intertidal saltmarshes. The improvements in effluent quality have mainly been by reducing oil content (as measured by infrared absorption spectrophotometry) and volume, and are summarized in table 4.

Apart from the obvious differences in flow rates, oil content and chemical constituents deriving from very different processes, they further differ in salinity. The Littlewick Bay effluent discharges at between 5 and 31‰ to an ambient salinity of *ca.* 33‰ and usually contains substantial amounts of fresh water, while the Fawley refinery, which uses mainly seawater for cooling, discharges at closer to ambient salinity (*ca.* 30‰).

Biological effects: Littlewick Bay

Early surveys in 1970 of the discharge area are detailed in Crapp (1970). These were followed by experimental work and surveys in 1972, 1974 and 1975 (Baker 1976*b*; Dicks 1976*b*) and more recently in 1981 as detailed below. Initially a series of transects was established along

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Table 3. Refinery processes at Fawley and Littlewick Bay, Milford Haven

(From Dicks (1981) and Lemlin (1980).)

processes	Fawley refinery	Littlewick Bay refinery, Milford Haven
primary crude distillation	+	+
naphtha reforming		+
desulphurization		+
sweetening		+
hydrofining	+	
catalytic reforming	+	
catalytic cracking	+	
thermal cracking	+	+
polymerization	+	
sulphur recovery	+	
solvents manufacture	+	
bitumen manufacture	+	
lubricating oil manufacture	+	
manufacture of: isobutylene, ethylene, butadiene, chlorobutyl/butyl rubber, methyl ethyl ketone, higher olefins, parammins (additives)	+	
present annual throughput capacity for crude oil/Mt	19.2	8.7

Table 4. Effluent oil content and volume between 1963 and 1980 at Littlewick Bay, Milford Haven, and Fawley

(Volumes are dry-weather flows.)

year	Littlewick Bay, Milford Haven: oil content / mg l^{-1}	Littlewick Bay, Milford Haven: volume / m^3 h^{-1}	Fawley‡: oil content / mg l^{-1}	Fawley‡: volume / 10^3 m^3 h^{-1}
1963–1970	24–50+	360	30+	28
1972	*ca.* 25	360	25	28
1974	*ca.* 25	360	14	23
1975	20–15	360	—	—
1976	15	360	10	21
1980†	10–15	252	10	21.5
maximum storm flow/(10^3 m^3 h^{-1})	2		4–6	

† Data from Lemlin (1980); remainder from Baker (1979).
‡ The figures given are averages for two separate discharges of oil, and are combined to provide the volume flow figures.

the shores up to 450 m westward and 390 m eastward from the location of the effluent discharge point (figure 6*a*). This allowed the identification of those species that showed gradients of change in relation to the effluent and also served to establish natural gradients along the shore in relation to exposure to wave action, e.g. a slight gradient in exposure as defined by the Ballantine (1961) scale was noted from east to west (Crapp 1971) ranging from grade 4 to grade 3, with Littlewick Bay being slightly more sheltered than the shores on either side. These studies were followed by more quantitative measures of key species at a greater number of sites (figure 6*b*), supported by hydrographic studies of dispersal of the effluent stream.

A 1958 photograph of Littlewick Bay in the 1968 Milford Haven Conservancy Board booklet shows that the shores were then dominated by limpets and barnacles. In 1971 the central transects (3 and 4, figure 6*a*) had considerably reduced numbers of *Patella vulgata*, other gastropods and barnacles compared with sites to the east and west, and elevated abundances

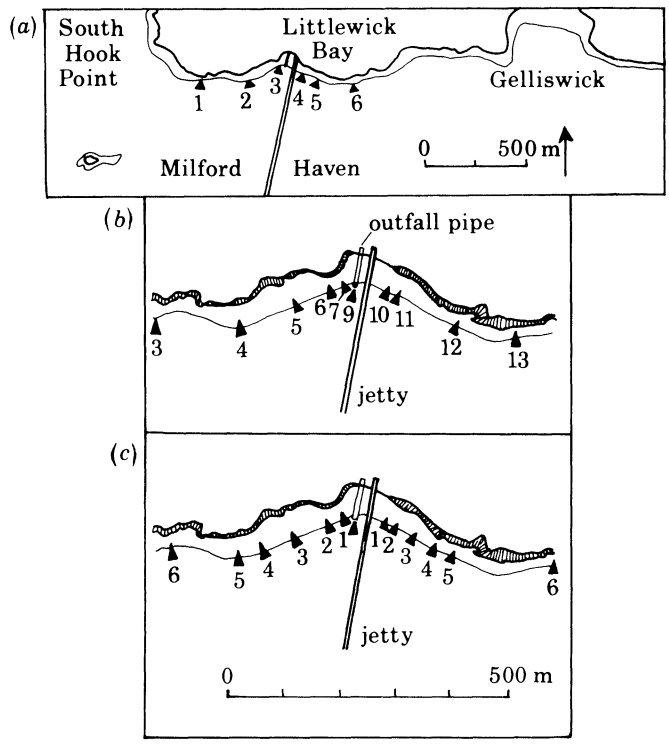

FIGURE 6. Littlewick Bay, Milford Haven: the locations of (*a*) transect sites in 1970 (Crapp 1970) and mid-shore sites at which limpet and barnacle densities were measured in (*b*) 1972 (Baker 1976*b*) and (*c*) 1981.

of *Fucus spiralis*, *F. vesiculosus* and *F. serratus*. These changes could not be explained by a change in the degree of exposure. Hydrographic studies (Crapp 1971; Dicks 1976*b*) indicated clearly that the effluent ponded in the shallow bay at slack water and ran east and west close inshore with the ebb and flood tide. It was concluded that the effects were caused by the effluent but that they were relatively minor changes in species composition and were restricted to within 200 m east and west of the discharge point. Resurvey of the transects in 1974 and 1977 showed the same pattern of effect with only minor unexplained differences from site to site.

More detailed measurements of the main animal species affected (*P. vulgata*, *Chthamalus* spp., *Elminius modestus* and *S. balanoides*) were carried out at a greater number of sites (figure 6*c*), which were selected to have similar aspects and slopes and were restricted to the middle shore only (mid-tide level).

In 1972 a gradient of effects on limpets extended approximately 120 m west and 100 m east from the outfall (Baker 1976*b*). Densities were reduced close the effluent (figure 7), those remaining being much larger in size (figure 8). No juveniles were found within *ca*. 45 m of the

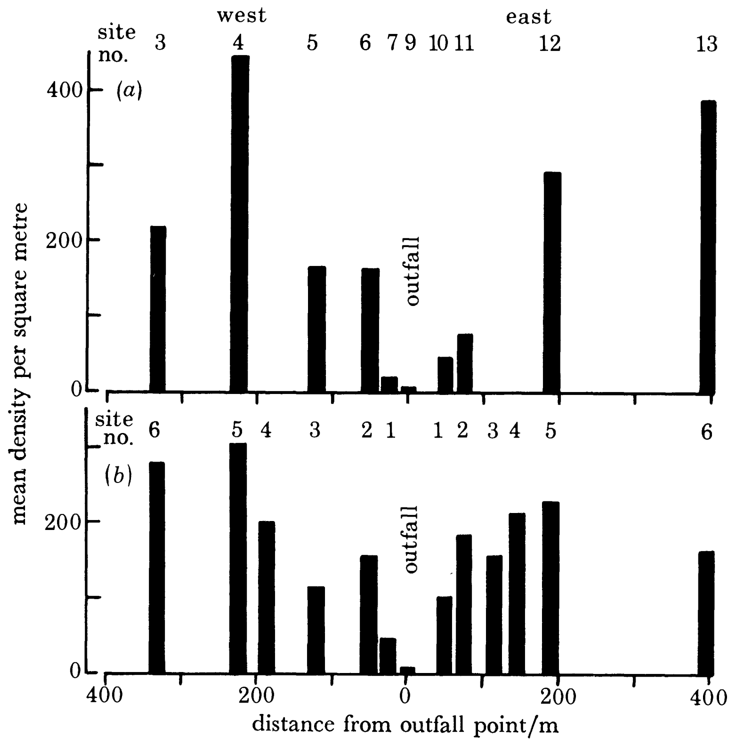

FIGURE 7. Mean density of *Patella vulgata* in the middle shore at Littlewick Bay, Milford Haven, (*a*) 1972 and (*b*) 1981.

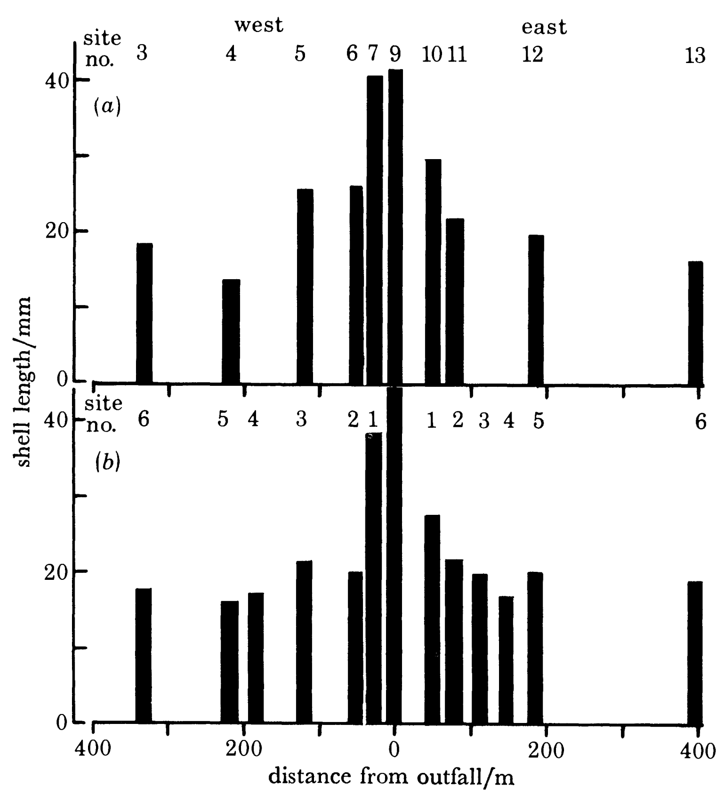

FIGURE 8. Mean shell length of *Patella vulgata* in the middle shore at Littlewick Bay, Milford Haven, (*a*) 1972 and (*b*) 1981.

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

outfall point. It was postulated that these effects stemmed from either the mortality of most juveniles, those surviving finding abundant food and growing rapidly, or no successful larval settlement had occurred, the limpets present predating the start of the effluent discharge. Both mechanisms may have been involved, with occasional 'clean' periods (e.g. during shutdown), allowing sporadic recruitment of juveniles. A similar distribution of limpets was found on the concrete of the pipe itself, with no limpets living immediately adjacent to the outfall point, but small numbers of very large limpets living between 10 and 40 m shorewards on the pipe.

Table 5. Densities of barnacles (adults and juveniles) in the Littlewick Bay mid-shore sites in 1974 and 1981

(Densities are given per 5 × 5 cm quadrat.)

site	1974	1981
5 west	41.0	45.5
4 west	33.3	23.1
3 west	32.5	19.2
2 west	22.7	10.5
1 west	9.5	3.3
discharge point	0	0
1 east	22.0	10.0
2 east	22.0	7.5
3 east	21.2	11.5
4 east	21.2	28.5
5 east	28.5	41.1

The same sites were resurveyed in 1981 and additional sites added to provide more comprehensive data (figure 6*c*). Both density (figure 7) and size distributions (figure 8) of limpets were similar to those found in 1972, but juveniles were found at the nearest sites both east and west, though none were found on the pipe itself.

Reduced densities of *S. balanoides* (the dominant species of barnacle in the middle shore at that time) were found in 1974 up to 40 m westwards from the outfall, but little effect was found to the east (Baker 1976*b*). On the concrete of the pipe no barnacles were found within 10 m of the discharge point, but low densities were found further shorewards. Inhibition of larval settlement was suspected of producing these very localized changes, and subsequent laboratory and field experiments supported this hypothesis (Dicks 1976*b*). Effluent was demonstrated to inhibit severely the activity of the larval stages of *S. balanoides*, partly as a result of its low salinity but also because of its content of oil and other constituents.

More substantial changes have been noted in barnacle distribution when comparing data from 1974 and 1981 (see table 5). Decreases in total barnacle density were evident at all central sites from 3 east to 3 west, presumably caused by the effluent, but there had also been changes in species composition similar to those found throughout the Haven. In 1974 the great majority of adults and many of the juveniles at all sites at Littlewick Bay were of *S. balanoides*; *E. modestus* and *Chthamalus* spp. formed less than 5% of the observed densities and were not included in the original counts. Photographic evidence confirms this finding, as do the data of Baker (1976*b*). In 1981 the density of *S. balanoides* had declined, although it was still the most common species at most sites (*ca.* 65% of the barnacles were this species). One exception to this pattern was at site 1 east, where 95% of the barnacles were *E. modestus*. Differences in susceptibility

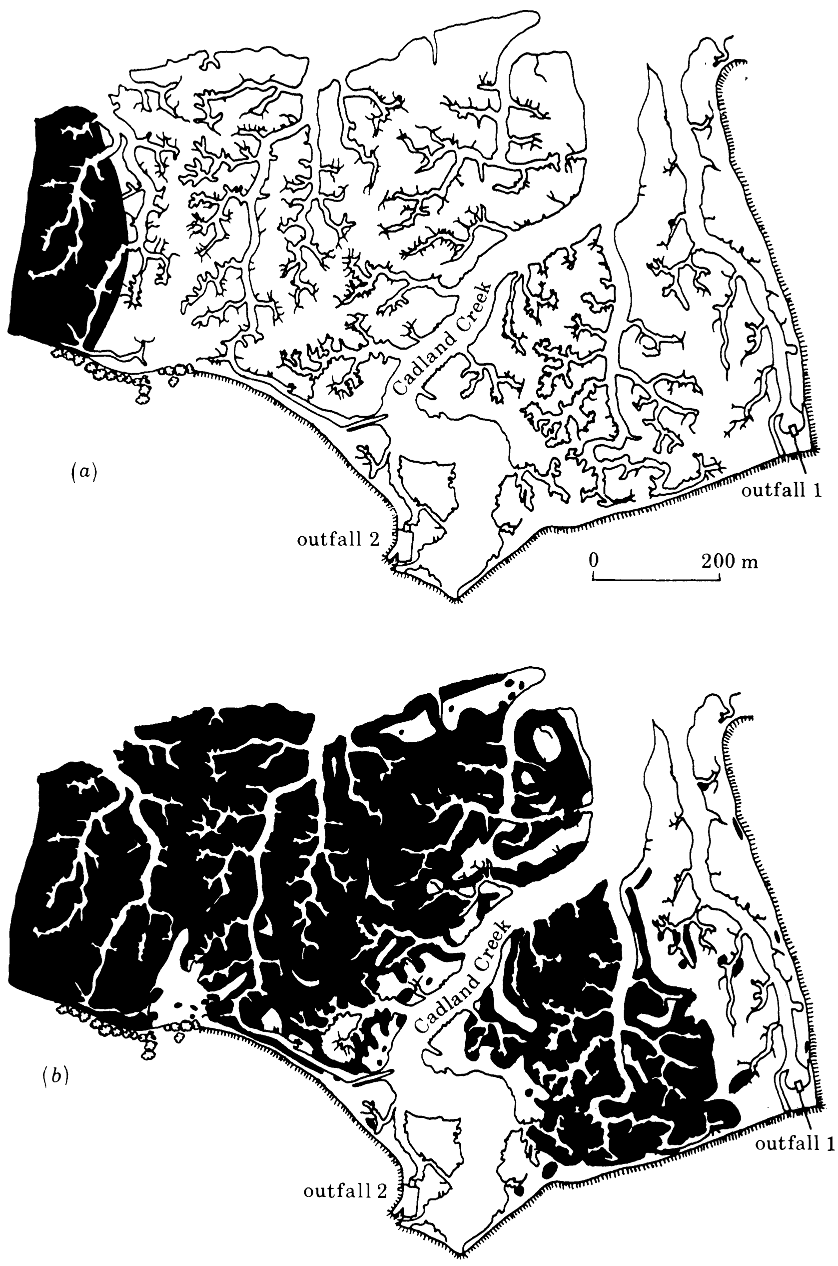

FIGURE 9 (*a*, *b*). For description see opposite.

of the species of barnacle to effluent are not known and suggest the need for caution in interpreting these changes.

As noted earlier, effluent quality improved in the early 1970s to meet new water authority regulations. Since then a refinery 'environmental awareness' campaign has resulted in steady improvement of the operation of effluent treatment equipment and a consequent reduction in oil content (see table 4). These changes have not resulted in any corresponding biological change between 1970 and the present. However, since some of the observed effects have been attributed to the salinity of the effluent as well as to its chemical content, change in oil specification would not remove them.

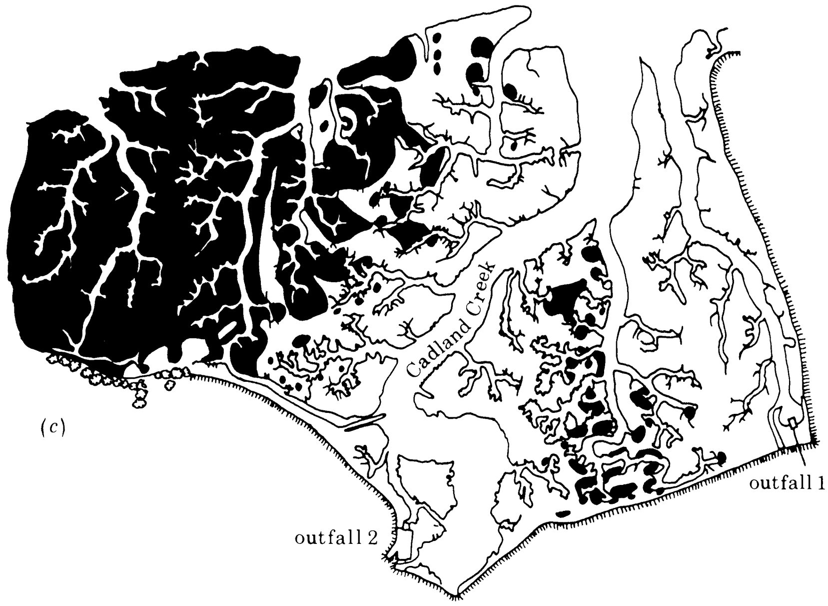

FIGURE 9. The distribution (shaded areas) of vegetation on the Fawley marsh in 1970 (*a*), of *Salicornia* spp. in 1978 (*b*) and *Spartina anglica* in 1980 (*c*).

Biological effects: Fawley refinery

In 1951 and 1953 two effluent streams from a refinery–petrochemicals complex began discharging to the dendritic creek system of the Fawley marshes (a Spartinetum) at the high-water mark. The locations of the discharges and the marsh creek system are shown in figure 9. As well as oil and other chemicals normally discharged in the effluent, oil from other sources occasionally enters the creek system. These sources include:

(*a*) accidental spillage within the refinery that might produce unusually high oil contents in the effluent;

(*b*) oil spillages from the refinery jetty installations and tankers;

(*c*) oil spillages from other installations in the area;

(*d*) oil spillage from the considerable ship traffic using Southampton Water.

In contrast to Littlewick Bay the effluent discharges produced extensive damage to the saltmarshes between 1953 and 1970, devegetating an area of marsh *ca.* 1000 m by 600 m (figure 9). A programme of effluent quality improvement reduced oil content and volume (summarized in table 4) and has resulted in a dramatic recovery of the marsh; this recolonization process has been monitored by surveys twice yearly since 1972. A simple vegetation mapping technique has been used to measure the spread of the five main marsh plant species over previously denuded areas. Animal communities of the marsh and the seabed around the discharge point to Southampton Water have also been studied by replicate sampling with cores and grabs respectively. Full details of the findings to 1980 can be found in Baker (1971 *a*, *b*), Dicks (1976 *c*, 1977, 1981), and this paper updates those findings with results from the most recent surveys of 1980 and 1981.

Throughout the monitoring period (1972 to the present) the most rapid recolonization by plants has been by those species that have seeded well, notably *Salicornia* spp. (figure 9) and *Aster tripolium*. *Spartina anglica*, the formerly dominant marsh species, reinvaded vegetatively and

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

slowly at first but later (1977–80) seeded well and spread more rapidly (figure 9). Major improvements in the vegetation seemed to be related to reductions in the formation of surface oil films following a reduction in oil content. Several series of transplantation experiments with *S. anglica*, in combination with the results of vegetation surveys, defined areas of persistent effluent effect and also confirmed that these were the result of effluent characteristics rather than accumulated pollution loads in marsh sediments. Aliphatic hydrocarbon levels were investigated for sediments at 14 sites over the saltmarsh in 1979 and 1980, and most showed high concentrations (range 340–17750 μg g^{-1} total aliphatics; see figure 10). Hydrocarbons were mostly old and weathered, but recent inputs were also found.

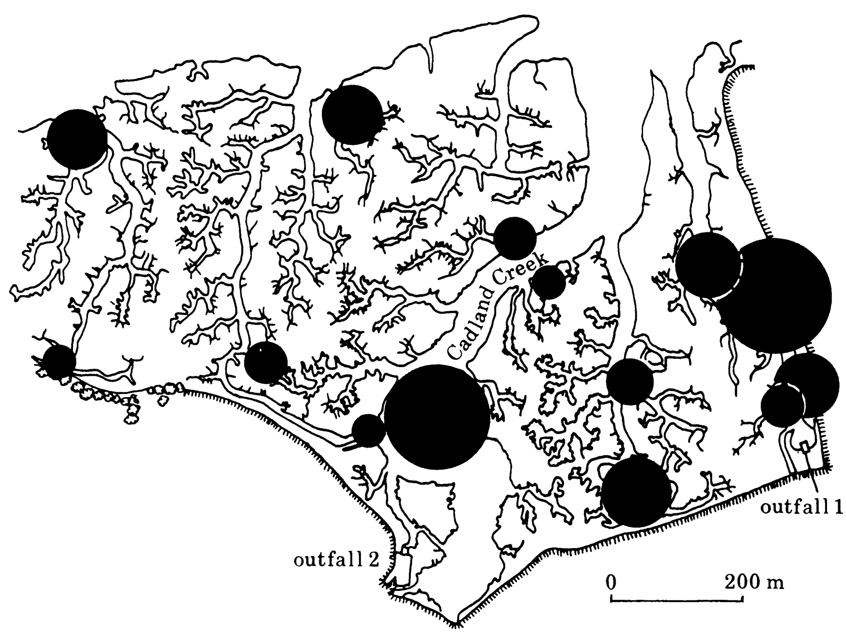

FIGURE 10. Distribution of hydrocarbons (total aliphatics, micrograms per gram) in the sediments of Fawley marsh, November 1980.

Animal communities of the marsh have also been investigated since 1979 and were found to be impoverished over most of the area previously denuded of vegetation. The most abundant organisms over the survey area were opportunistic oligochaetes (Enchytraeidae and Tubificidae) and they showed similar patterns of distribution around the effluents to those of the plant species in 1980. Sediments heavily contaminated with weathered oil (up to 5875 μg g^{-1} total aliphatics) supported apparently healthy communities of marsh plants and oligochaetes. Oligochaetes were also found in small numbers where 17750 μg g^{-1} total aliphatics was recorded.

Studies of the seabed communities around the mouth of Cadland Creek defined natural gradients in community type in relation to sediment type, superimposed on which were gradients of effluent effect. Only *Capitella capitata* and *Nereis diversicolor* survived in the mouth of the creek and a gradient in numbers of *Nereis diversicolor* away from the creek mouth was recorded (figure 11). Sediments around the mouth of the creek also showed high levels of pollution by hydrocarbons (up to 6450 μg g^{-1} total aliphatics) with gradients of concentration away from that point. Correlations between gradients in hydrocarbon concentration and organism density were good, and a comparison of the data with earlier data from wider areas

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

of Southampton Water (1975 and 1976; Levell, unpublished data) suggests that little change in size or type of effect on the seabed from this effluent has occurred during the last 5 years.

Throughout the monitoring period, outfall no. 1 has shown consistently greater effects on the marsh vegetation than outfall no. 2. This has been attributed partly to its higher oil content (the outfall receives a greater load of dirty water streams than no. 2) and partly to a sour-water stream (which contains sulphur compounds and phenols). Earlier findings clearly suggest that

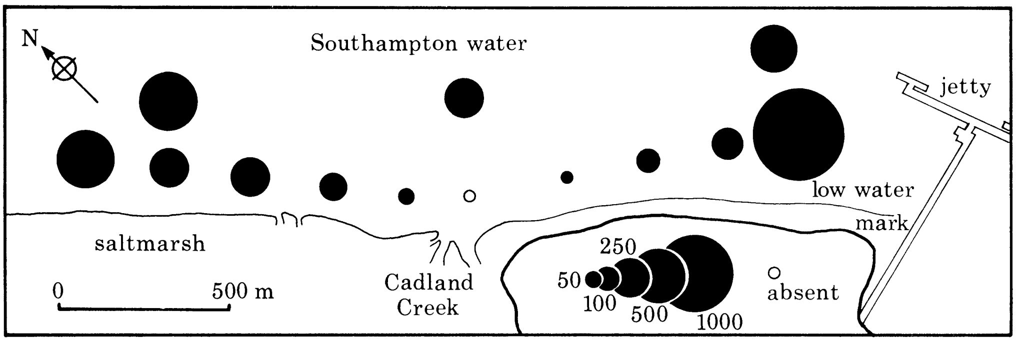

FIGURE 11. The density of *Nereis diversicolor* (numbers per square metre) at sites on the seabed around the mouth of Cadland Creek, Fawley, May 1980.

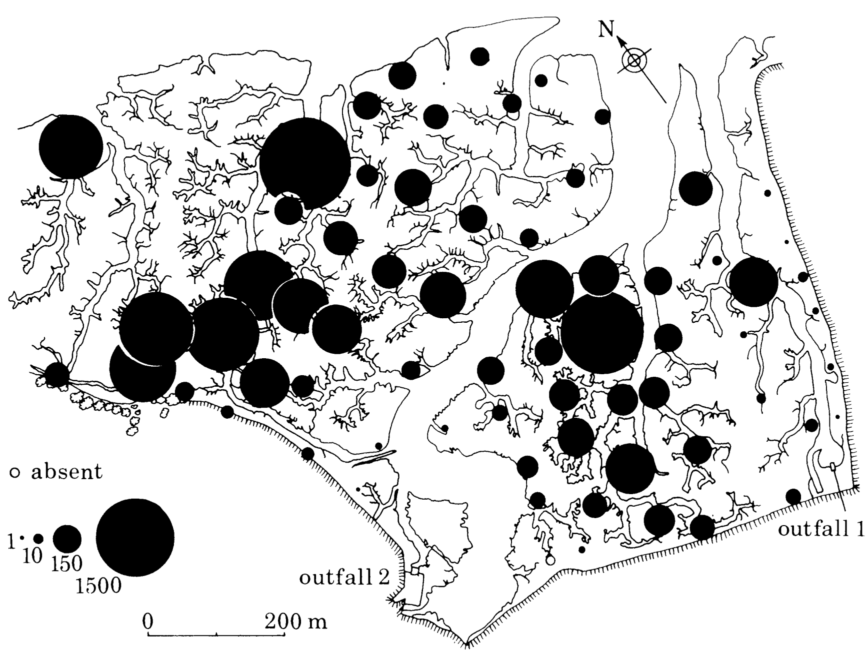

FIGURE 12. The distribution of oligochaetes (numbers per 10 cm diameter core), Fawley marsh, November 1980.

for the marsh vegetation, a reduction in oil content has been the major factor in promoting recovery. Thus, to reduce further the oil content of outfall no. 1 and encourage rapid regrowth, a dual-media filtration unit is being installed, to operate from early 1982.

Although vegetation surveys have defined recovery and areas of continued effluent effect precisely, the results of effluent improvements are not usually seen in the plants for up to a year, the time taken for them to complete a full seeding–growth cycle. The oligochaetes found in the marsh sediments have recently been sampled by single cores at 69 locations (figure 12) and in November 1980 showed patterns of distribution that correlated well with vegetation distribution, showing areas of effect around both effluent discharges. In particular they occurred in the muds very close to the discharge point of outfall no. 1, albeit in low numbers, and showed numerical gradients in density away from the creek carrying the discharge. Outfall no. 1 was closed for reconstruction and repair early in 1981, and only 6 weeks later oligochaete numbers had increased at six of the seven sites closest to the outfall. Mean density at the seven sites in November 1980 was 9.4 oligochaetes per core and in February 1981 was 19.4 oligochaetes per core.

Taxonomic problems with the Oligochaeta (Brinkhurst 1980) have so far precluded the identification of differences in sensitivity to the effluents that may occur between species, but used in these monitoring studies as a single taxon (Oligochaeta indet.) they appear to be sensitive to changes in effluent quality and to respond rapidly to change. The responses and usefulness of oligochaetes in effluent monitoring studies is being further investigated, and both these studies and studies elsewhere (for examples see Brinkhurst 1980) have shown some families and species to be useful in monitoring organic pollution.

Comparison of observed effects in Littlewick Bay and at Fawley

The Littlewick Bay and Fawley effluents differ widely in their effects. These differences are mainly related to three factors:

(*a*) the size and composition of the effluent;

(*b*) the rate of immediate dispersion and dilution;

(*c*) the biological nature of the receiving environment and susceptibility of organisms.

In the late 1960s and early 1970s both the Littlewick Bay and Fawley effluents were of similar oil content but differed greatly in volume and content of other chemical components. The effluent in Littlewick Bay was small, relatively rapidly dispersed by large tidal movements, and there was little chance for oil to accumulate on the well drained rocky shores of the bay. Biological consequences were therefore small, and effects appeared restricted to larval stages of limpets and barnacles. The Fawley effluent was large and, although rapidly dispersed in Southampton Water during most stages of the tide, was ponded for relatively long periods of time in the saltmarshes during the double high tide peculiar to Southampton Water. Thus, at each high spring tide the plants of the marsh were subjected to submersion in effluent. Under these conditions the marsh acted like a giant separator basin, and oil rapidly formed surface films which adhered to leaves and stems, resulting in widespread damage. The process was more analogous to repeated spills than a continuous discharge. Substantial quantities of oil also became stranded on the sediments and incorporated to the mud.

Clean-up of the Milford effluent resulted in reducing oil content by *ca.* 50%, but little biological improvement resulted. This was not unexpected because at least some of the observed effects resulted from the low salinity of the effluent, a factor not under the control of the refinery. At Fawley the same order of magnitude improvement, i.e. a 50% reduction of oil content, resulted in marked decreases in oil film formation followed by rapid and dramatic reinvasion of damaged areas by the plants. Further improvements promoted further recovery. Saltmarsh plants have thus proved to be sensitive indicators of change in effluent quality. In areas at the furthest extent of effluent damage, where recolonization has been occurring for 10 years, the marsh now appears healthy with a full complement of the plants and animals to be found in nearby undamaged marsh areas. Nevertheless, there are still obvious differences in species composition compared with nearby areas of Spartinetum. Such subtle effects may persist for long periods of time.

Subtidally, improvements in effluent quality do not appear to have had such obvious effect, although the data sets are not as complete as for the saltmarsh. The reasons are not clear, but in these areas continuous input of effluent occurs, rather than the sporadic high-tide oilings of the marsh. The marsh effects also appear to result mainly from free oil; subtidally, other effluent components not removed by improved oil removal systems may be involved. Nevertheless, gradients in animal distribution did correlate well with hydrocarbon content (both weathered and fresh) of the sediments.

The above data suggest that subtidally the distribution of animals could be predicted from oil contents of the sediments, but this is not so in the saltmarsh. There, correlation between animal and plant distribution was good, but correlation between both and the hydrocarbon content of sediments was poor. Additionally, very high concentrations of oil in sediments in some areas did not inhibit saltmarsh plant growth (up to 5875 $\mu g\ g^{-1}$ total aliphatics) or, apparently, the survival of *Capitella capitata* and *Nereis diversicolor* (up to 6450 $\mu g\ g^{-1}$ total aliphatics) and oligochaetes (found in large numbers where levels up to 4570 $\mu g\ g^{-1}$ total aliphatics and in small numbers up to 17750 $\mu g\ g^{-1}$ total aliphatics).

The relations between hydrocarbon concentrations in sediments and effects on organisms are not fully understood at present, and prediction of biological effects from hydrocarbon data should be approached with caution.

North Sea fields

Oil industry activity in the North Sea is considerable: there are currently 28 producing fields, 17 fields under development and 13 fields likely to be developed in the near future. Oil production from the area is expected to continue into the next century. How much of a cause for concern are offshore oil installations? Read & Blackman (1980) have recently put the oily water discharges likely to result from North Sea oilfield operations in the perspective of existing inputs to coastal waters and concluded that the offshore discharges constitute only minor additions. However, it must be remembered that the development and operation of offshore oilfields gives rise to a host of potential stresses to the environment in addition to the input of petrogenic hydrocarbons. These stresses have recently been reviewed by Dicks (1982), and include the physical presence of installations, discharge of drill fluids and cuttings, sediment disturbance through anchoring and pipeline trenching, inputs of sewage and garbage and the discharge of water containing various toxic chemicals and biological inhibitors. It is difficult to assess the relative importance of each potential stress and to separate the effects of one from those of the others.

There are many difficulties associated with the use of the vagile and motile organisms of the sea surface and water column in the assessment of pollution effects (Dicks 1982). For this reason benthic organisms are often the subject of pollution monitoring studies, either at the individual level (Bayne *et al.* 1979) or at the community level (Elmgren & Cederwall 1979; Govaere *et al.* 1980). Benthic organisms are effectively sessile and act as integrators of the effects of the various kinds and levels of pollutants and disturbance with time. In view of this and the need to separate natural change from pollution effect, the oilfield monitoring studies carried out by the Oil Pollution Research Unit have relied on quantitative investigations of the macrofauna. The sampling has been carried out at a number of stations situated at various distances from the installations. Effects reported from our two longest running studies (since 1975 and 1973 respectively) range from no detectable effect in the Forties oilfield after 3 years of production (Hartley 1979) to changes in faunal community composition and structure over an area of some 9.5 km^2 in another North Sea oilfield (Addy *et al.* 1978).

Wolfson *et al.* (1979) have reported a limited area of effect (*ca.* 100 m in radius) around an oil production platform off California. The effects were largely caused by sediment modification resulting from the accumulation of drill cuttings and the shells of *Mytilus* spp. and other fouling organisms from the platform legs. It is likely that similar areas of effect occur around every offshore production platform and where water-based drilling fluids have been used are not cause for concern. However, there is recent evidence that where oil-based drill fluids have been used, the discharge of cuttings without effective treatment to remove hydrocarbons can result in substantial elevations of the hydrocarbon levels in benthic sediments (Grahl-Nielsen *et al.* 1980; Law & Blackman 1981). Grahl-Nielsen *et al.* (1980) found hydrocarbons attributable to the discharge of contaminated cuttings over an area of 5 km^2 around the Statfjord 'A' platform in the Norwegian sector of the North Sea. The total hydrocarbon levels in sediments at the six sampling stations closest to the platform (within 500 m) averaged 433 $\mu g\ g^{-1}$. By extrapolation from their data the volume of oil discharged in association with drill cuttings during the development of the field can be estimated to be about 2300 t (i.e. 24% of the wet mass of cuttings). Law & Blackman (1981) reported that cuttings from wells drilled with the use of oil-based fluids discharged in the U.K. sector of the North Sea contained 6–14% wet mass diesel oil and that most of the hydrocarbons associated with cuttings were carried down to the seabed. They compared sediment hydrocarbon levels in the Beryl field, developed with the extensive use of oil-based fluids, with the Auk and Fulmar fields, where water-based fluids were used. Their results indicate that the level of hydrocarbon contamination was much greater around the Beryl field and that oil attributed to drill fluids and cuttings was present to at least 18.5 km from the Beryl platform. Law & Blackman (1980) suggested that, extrapolating from the results of the 1977 Ekofisk survey (Addy *et al.* 1978), the benthic fauna around the Beryl platform would be adversely affected to a distance of at least 2 km.

The above studies suggest that the projected volumes of oil discharged from U.K. offshore oil installations may be considerably greater than the estimates of Read & Blackman (1980), (who considered only oil in production and displacement water) as several oilfields in the U.K. sector of the North Sea have been developed mainly with the use of oil-based drill fluids, e.g. Beryl, Montrose and Thistle (Law & Blackman 1981). The investigations of Grahl-Nielsen *et al.* (1980) and Law & Blackman (1981) were concerned with the chemical assessment of hydrocarbon pollution resulting from the discharge of oil-based drill fluids and cuttings. So far an assessment of the biological impact of these discharges does not appear to have been undertaken. Such an assessment is essential to establish whether these discharges are a cause for concern and current discharge regulations are adequate. It is worth noting that diesel oil (no. 2 fuel oil) has been shown to have serious and persistent effects on benthic fauna (see, for example, Sanders *et al.* 1980).

In conclusion, the few available data from long-running studies around North Sea oil platforms suggest that fields developed with the use of water-based drill fluids do not present serious threats to the benthic fauna. By inference this conclusion may be applied to other

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

components of the marine ecosystem since the benthic fauna is very largely dependent on the quality of the overlying water. In contrast there are no biological data available to assess the effects of the elevated sediment hydrocarbon levels observed around some North Sea fields where the development has involved the use and discharge of oil-based drill fluids. This is clearly a serious gap in our knowledge and a priority for future field studies.

Conclusions

Three factors have been consistently shown to control the severity of biological effects in the examples of chronic inputs given above: the volume and chemical nature of the pollutant, the physical nature of the receiving environment and its biological nature and composition. Observed effects reported here range from localized and subtle to widespread and persistent. In Milford Haven, a combination of good management and extensive tidal flushing has resulted in little long-term impact from a massive oil-industry development. Only subtle localized effects can be found at sites subjected to small spills and around a refinery effluent discharging to Littlewick Bay. In the latter case some of the effects resulted from salinity change rather than pollutant load, and reduction of effluent oil content has not reduced the area of effect. At Fawley, where pollution loads have been substantially reduced over the last 10 years, damage to saltmarshes has been followed by recovery, although a return to a Spartinetum may take many years. Poor correlation between hydrocarbon loads in sediments and observed distributions of organisms on the marsh suggest the use of caution in interpreting sediment hydrocarbon data, and a need for further study. Offshore, in the relatively small number of North Sea oilfields investigated so far, widespread severe biological effects have not been observed. In one North Sea field (Addy *et al.* 1978) macrobenthic communities have shown changes around platforms, notably in the centre of the field. In the Forties field, no effects of development have yet been recorded. These differences appear related to length of time in production, amounts of oil produced, the degree of construction activity in the field and the biological and physical nature of the surrounding environment. These fields may present relatively few long-term biological problems. Recent evidence suggests that in oilfields where oil-based drilling muds have been used extensively, substantial quantities of oil may reach the seabed and are likely to affect seabed communities. No biological data are currently available and this is a priority for future investigation.

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Discussion

Dale Straughan (*University of Southern California*, *Los Angeles*, *U.S.A.*). I should like to discuss the Ekofisk work. It is a while since I have read reports but I believe it was a well designed study. On the basis of this I have recently suggested that many of the studies that indicated no change are due to poor compiling. The authors now say that Ekofisk is a special case. Would they care to extrapolate from the Ekofisk situation?

J. P. Hartley. The Ekofisk field certainly is a special case. The method of oil export used until 1975 is uncommon in the North Sea and resulted in the periodic discharge of oily displacement water near the seabed. Similarly the high numerical dominance of the benthic fauna by a single species over a period of several years is unusual. Such communities tend to be unstable and more usually found in shallow water where environmental disturbance, for example by storms, can result in defaunation of sediments rendering them suitable for monopolization by opportunistic species. Stephen (1923) reported a pure *Ditrupa* (Polychaeta: Serpulidae) community, west of the Shetlands, and we have found this community in the Celtic Sea where the population density of *Ditrupa* was *ca.* $5600/m^2$. The reasons for the establishment of such communities in shelf depths are unclear but there is the possibility that small changes in environmental conditions may result in faunal succession and the establishment of a more usual community structure. Since the Ekofisk field is an unrepresentative example of the level and kind of industrial activity and benthic fauna found in the North Sea, I believe it unwise to extrapolate too far to other situations.

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R. B. Clark (*Department of Zoology*, *University of Newcastle*, *U.K.*). The apparent lack of impact of chronic inputs and frequent small oil spillages in Milford Haven contrasts with the prolonged impact of the pollution in Buzzards Bay where petroleum hydrocarbons became trapped in sediments and released over a prolonged period, so constituting a chronic pollution. Would Dr Dicks please comment?

B. Dicks. The contrast results primarily from two factors: the type of oil spilled in Buzzards Bay and its long-term retention in sediments. The Buzzards Bay spill was of toxic no. 2 fuel oil (diesel), which deeply penetrated anaerobic sediments in a low physical energy habitat and was subsequently slowly released more or less unchanged in toxicity. Chronic inputs to Milford Haven are many and varied, but have been mainly of effluents, dispersants or crude oils (lower in toxicity than diesel) to a highly dispersive, well oxygenated system where dilution and degradation are relatively rapid. So far no equivalent situation to that in Buzzards Bay has arisen in the Haven, the only large light and toxic product spill (*ca.* 2300 t of 4 star petrol from the *Dona Marika* in 1973) having occurred in an exposed sandy bay where hydrocarbons were not retained in sediments for long periods. However, some of the more sheltered areas of the Haven could have similar problems to those in Buzzards Bay.