Field Studies 5 (1982) 623–641

ENVIRONMENTAL EFFECTS (1969 TO 1981) OF A REFINERY EFFLUENT DISCHARGED INTO LITTLEWICK BAY, MILFORD HAVEN

By SARAN PETPIROON* and BRIAN DICKS
Field Studies Council Oil Pollution Research Unit, Orielton Field Centre, Pembroke, Dyfed, Wales, U.K.

ABSTRACT

Marine habitats within Milford Haven, Wales, have received a variety of contaminants from four refineries, a tank farm, and oil tanker activities since 1960, in addition to inputs of sewage, urban run-off, light industrial effluents and contaminants from a naval dockyard. Inputs from the oil industry have mainly been of crude oil, dispersant chemicals (used in the clean-up of spills) and refinery effluents. This paper summarises changes observed in rocky shore communities between 1969 and 1981 around a refinery effluent discharge in Littlewick Bay on the north shore of the Haven. Previous workers (Crapp, 1970; Baker 1976*a*) used a series of belt transect surveys of the intertidal zone in Littlewick Bay to define the form of effluent discharge effects and the distance from the discharge point to which effects could be observed. These were complemented by quantitative surveys of principal organisms affected by the discharge and laboratory experimental work (Dicks, 1976) to explain the nature of observed changes. The 1978 and 1981 survey work described here updates earlier findings and defines the form and extent of biological effects on rocky shore communities after 21 years of effluent discharge. The consequences of an improvement in effluent quality between 1971 and the present are discussed, in relation to biological findings which pre- and post-date effluent quality improvements.

The findings of recent studies were very similar to those of the earliest ones in this bay. Observed effects have been restricted throughout the survey period (1969–1981) to within about 200 m of the discharge point and have taken the form of reductions in the numbers of several shore species but notably grazing gastropods (*Littorina* and *Patella*) with corresponding increases in the abundance of fucoid algae. Changes in barnacle populations have also taken place. It has been inferred that the low salinity of the effluent, combined with its burden of oil and other contaminants, has produced these effects and that at least part of the observed change has resulted from effluent effects on larval stages rather than upon adults directly. The change in effluent quality (reduction of the maximum oil content from 50 ppm to 25 ppm) has not resulted in a corresponding reduction in the area of effluent effect.

INTRODUCTION

MUCH PUBLIC concern has been expressed about the effects of oil on marine life and this disquiet is reinforced by the well-publicised and obviously damaging major tanker accidents. As a result, the effects of oil spills have been much-studied since the wreck of the *Torrey Canyon* 1967 (Wardley-Smith, in press). The reasons for damage assessment are obvious, and include financial and amenity considerations as well as a need to protect and manage biological resources. Spills often produce severe local effects which may be relatively short-lived (e.g. on rocky shores (Baker, 1976*b*)) but in some cases persist for many years (e.g. in mangrove swamps, where reinstatement following death may take 20–40 years or longer to recover (Odum and Johannes, 1975)). Spills from tanker accidents contribute only a relatively small

*Present address:
Phuket Marine Biological Centre, P.O. Box 60, Phuket Isle, Thailand.

proportion of the total oil input to the sea (1–2% (National Academy of Sciences, 1975)). Of greater concern are the long-term chronic inputs around point sources which result from production, handling and refining of oil.

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 of refinery effluents for more than 20 years. Biological data for some habitats in the Haven are available from 1961 and have been complemented by more recent studies. An area that has received particular attention has been Littlewick Bay, a small stony bay flanked by rocky shores of Devonian Red Sandstone, which has received a refinery effluent discharge at the low-water mark since 1960. Changes in shore communities in relation to that discharge form the subject of studies reported here. Previous workers (Crapp, 1970; Baker, 1976*a*) used a belt transect survey method at various distances from the effluent discharge point, both eastward and westward, to define the nature and extent of effects of the discharge on rocky shore communities. These were complemented by quantitative studies of the principal organisms affected (Dicks, 1976; Baker, 1976*a*). All observed effects were restricted to within *ca.* 200 m of the discharge point. This paper describes the findings of a resurvey in 1978 of the original belt transects and a quantitative survey in 1981 of the principal organisms (barnacles and limpets). The findings are described and compared with the findings of the earlier studies, and the implications of an effluent clean-up programme (reduction in volume and oil content) between 1971 and the present are discussed in relation to biological data which pre- and post-date the effluent quality changes.

Milford Haven as an oil port

Milford Haven (Fig. 1) is a drowned river valley in the south-western corner of Wales, which receives fresh water from the Eastern and Western Cleddau Rivers and several smaller streams such as the Pembroke River, Gann River and Sandyhaven Pill. The total freshwater input is about 100 times less than the tidal volume (maximum tidal range 8.0 m on spring tides), and this results in high salinities throughout most of the Haven (33‰). Characteristic estuarine conditions are found only in the upper reaches, around the mouths of rivers and, to a lesser extent, streams. The physical characteristics of the shores and water body are detailed by Nelson-Smith (1965).

The natural deep-water channel which extends up the Haven as far as Wear Point, in combination with shelter from ocean swell, makes it an ideal site for an oil port. The first refinery was commissioned in 1960 with a capacity to refine 4.5 million tons of oil a year (m.t.y.), now increased to 8.7 m.t.y. In 1961 a crude receiving terminal and tank farm to handle 9.2 m.t.y. with a pipeline to a refinery near Swansea were completed. Two further refineries were commissioned in 1964 and are jointly in the process of building a catalytic cracking unit to provide a combined refining capacity of 14 m.t.y. The most recent (1973) addition to the Haven has been a 5 m.t.y. refinery which has also recently completed a catalytic cracking facility. All the refineries have extensive jetty systems, four well offshore and a fifth close inshore at Wear Point (Fig. 1). In addition to the refineries, a 2,000 Megawatt 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.

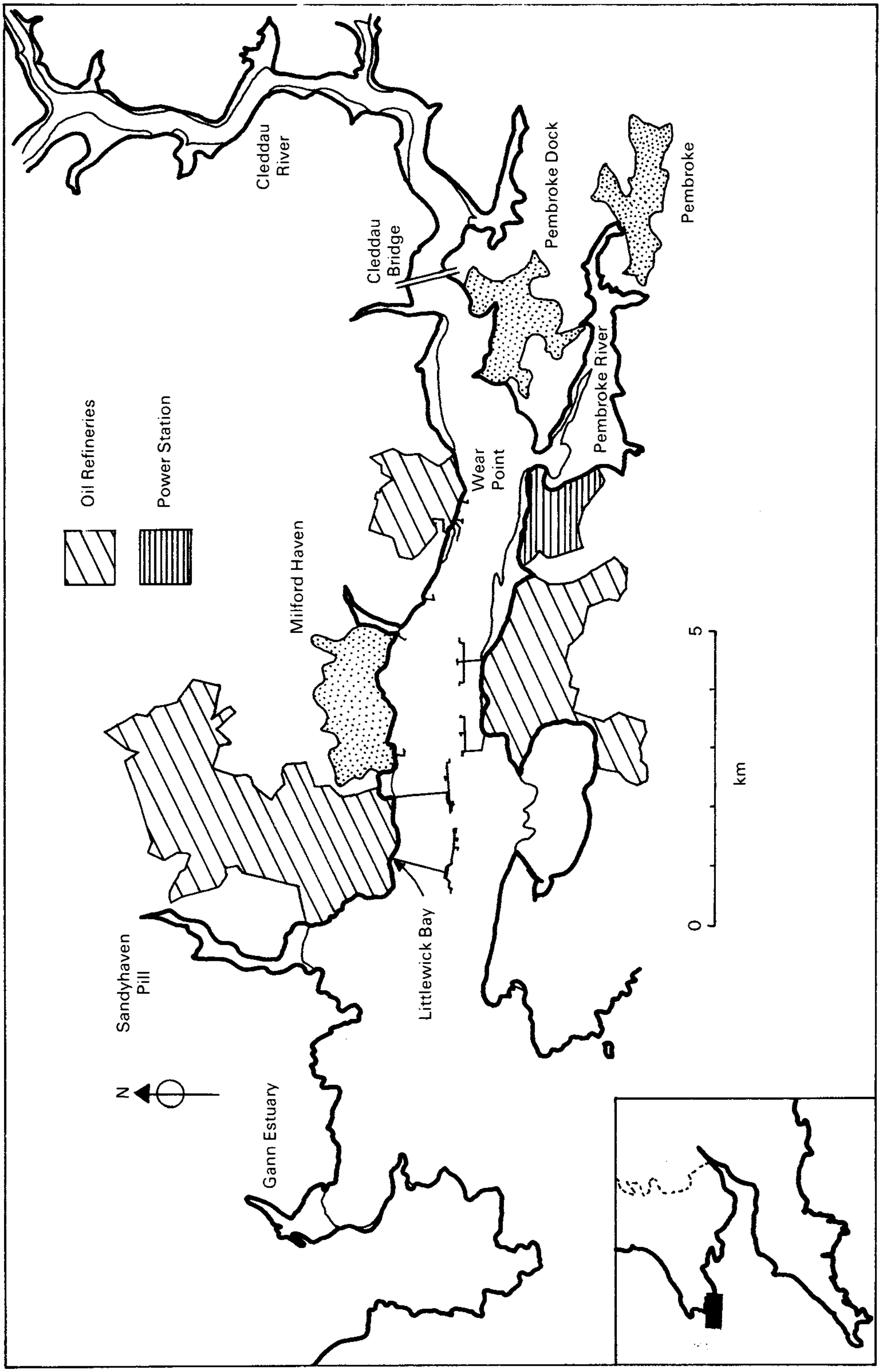

Fig. 1.
Map of Milford Haven showing the location of refineries and Littlewick Bay.

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 Lusitanian and Boreal species overlap in their distributions, and it possesses a very varied marine flora and fauna (Crothers, 1966; Jones and Williams, 1966).

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 and power station effluents, oil spills, sewage, light industrial effluents and urban run-off. Comprehensive records of oil spills have been kept by the Milford Haven Conservancy Board (the Port Authority). Many of the recorded spills have been treated with dispersant chemicals, both at sea and on some of the rocky shores.

The quality of refinery effluents has changed since operations began in 1960. Before 1971, Water Authority specifications permitted the discharge of 50 ppm oil in effluent, but in that year this was reduced to 25 ppm. Full current specifications are given in Table 1. It is, however, difficult to compare pre- and post-1971 effluent quality because the early information is limited. Crapp (1970) estimated that in that year the three refineries then in operation were contributing some 60,000 gallons of oil per year to the Haven. Dr T. Abbiss *et al.* (unpublished data) have estimated current inputs from the effluents to be in the region of 45,000 gallons per year, and the total input from all sources to be approximately 60,000 gallons per year. Chronic inputs can thus be assumed to have peaked in the late 1960s, when the 50 ppm specifications still applied and some 30–40 million tons of oil per year were processed, and the early 1970s when oil throughputs of the refineries were at a peak (43–59 million tons a year). A steady increase in input can be postulated up to that time with a slow decline since 1975 corresponding with the decline in tonnage handled.

Crude oil, however, is not the only contaminant in refinery effluents. In addition to a range of oils (depending on crude feedstock and refinery processes) they contain refinery products and other chemical constituents. They may be above ambient temperature, are often of lower salinity than the sea to which they discharge, and contain various suspended solids, all of which may affect the recipient marine systems. Many of these factors have been examined and summarised by Baker (1979).

The discharge to Littlewick Bay is from a relatively modern and mainly air-cooled refinery. The effluent arises from processes which include primary crude distillation, naphtha reforming, desulphurisation, sweetening and thermal cracking. The discharge volume is about 9,000 $m^3\ hr^{-1}$ (8.7 m.t.y.) and salinity of the discharge varies from 5 to 31‰ depending on stormwater flow. Ambient salinity of the Haven at the discharge point is about 33‰. A summary of changes in effluent oil content since 1963 is given in Table 2. The changes have resulted from, firstly, the amendment to Water Authority discharge regulations in 1971, noted above, and, secondly, from an "environmental awareness" campaign within the refinery aimed at improving operator performance and equipment efficiency. The earliest biological data for the bay (Crapp, 1970) pre-dates the effluent clean-up programme, and forms an interesting comparison with the findings of biological surveys in 1978 and 1981.

Table 1. *South West Wales River Authority quality conditions for refinery effluents in Milford Haven*

Temperature*	30°C
pH	5–9
Suspended solids*	50 ppm
4h permanganate (COD)*	25 ppm
Total oil*	25 ppm
Total phenols*	3 ppm
Ammoniacal nitrogen (as N)*	4–6 ppm
Cyanide (as HCN)*	0.1 ppm
Sulphide (as H_2S)*	1 ppm
Copper*	0.3 ppm

(From Baker, 1976*a*)
* Value not to be exceeded.

Table 2. *Effluent oil content (ppm) and volume (m^3 hr^{-1}) between 1963 and 1980 at Littlewick Bay, Milford Haven. Volumes are dry-weather flows*

Year	Oil Content	Volume
1963–1970	24–50+	360
1972	*c.* 25	360
1974	*c.* 25	360
1975	20–15	360
1976	15	360
1980*	10–15	252

Maximum storm flow: 2,000 m^3 hr^{-1}.
* Data from Lemlin (1980). Remainder from Baker (1979).

Methods

Changes in rocky shore communities adjacent to the effluent discharge point have been measured using a combination of two techniques—belt transects in relation to tidal height, which describe shore zonation patterns, and population estimates of dominant species using replicate quadrats.

Belt transects

The methods used for the survey of rocky shore communities followed those described by Moyse and Nelson-Smith (1963) as modified by Crapp (1970) and can briefly be summarised as follows:

1. A series of transect sites were selected at increasing distances on both sides of the discharge point (Fig. 2) in order to look for any gradient of pollution effect which the effluent might have had on the shore communities. The transects furthest from the effluent were regarded as reference sites. As far as possible each site was chosen for a similar type of substratum and slope with the upper part of the shore being bedrock without large rockpools or gullies.
2. At each selected site, a transect line was established by laying a measuring tape, approximately at right angles to the low-water line, up to the upper shore where the yellow lichens *Xanthoria* spp. and *Caloplaca* spp. were present. The uppermost

station was then marked with red paint and a compass bearing taken along the transect line to allow accurate relocation during future surveys.

3. Other authors (e.g. Crapp, 1971; Dalby *et al.,* 1978; Moyse and Nelson-Smith, 1963; Nelson-Smith, 1967; Syratt and Cowell, 1975) have shown that rocky shore animal and plant zonations can be clearly illustrated by sampling at 0.1 of the tidal range. Sample stations were accordingly established at 60 cm vertical intervals along the transect using a cross-staff technique devised by Nelson-Smith and described in Crapp (1970) (Fig. 3).
4. The checklist of 48 common littoral species compiled by Crapp was used, and the abundance of each of those species was assessed in a 2 m wide strip at each station. Abundance scales based on those devised by Crisp and Southward (1958), expanded by Ballantine (1961), Moyse and Nelson-Smith (1963) and subsequently modified by Crapp (1970) were used (Table 3). Methods employed in this study differed slightly from Crapp's technique. Crapp estimated abundance over a 10 m wide strip along the transect. Subsequent work by members of the Oil Pollution Research Unit has shown that estimation of a 2 m wide strip gives almost identical results for common species although rarities may be overlooked. Thus, for efficiency, a 2 m wide strip was used but if a species was "present" outside the sampling strip but within a 10 m strip, it was recorded as such on the data sheet.
5. Other species, not included in the checklist but found during the survey, were also recorded on the data sheet with their abundance estimated using an appropriate scale.

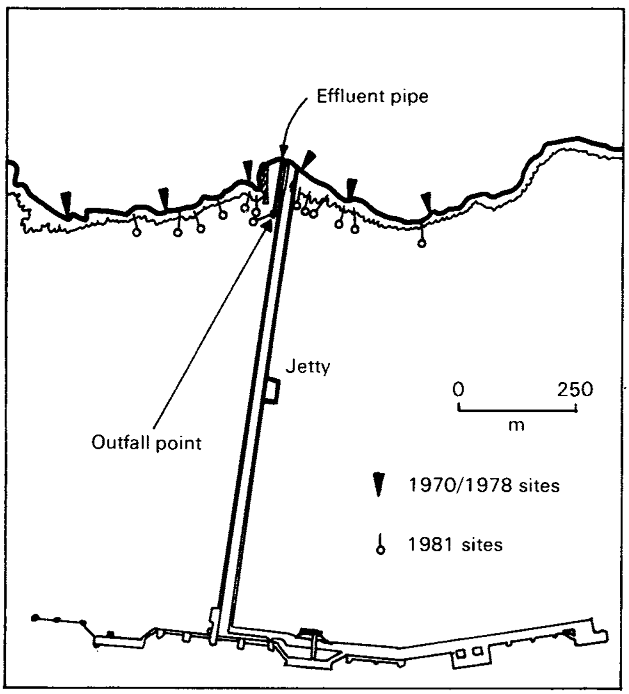

FIG. 2.
The location of transect and quadrat sites in 1970, 1978 and 1981, Littlewick Bay, Milford Haven. (Land at the top; Sea at the bottom.)

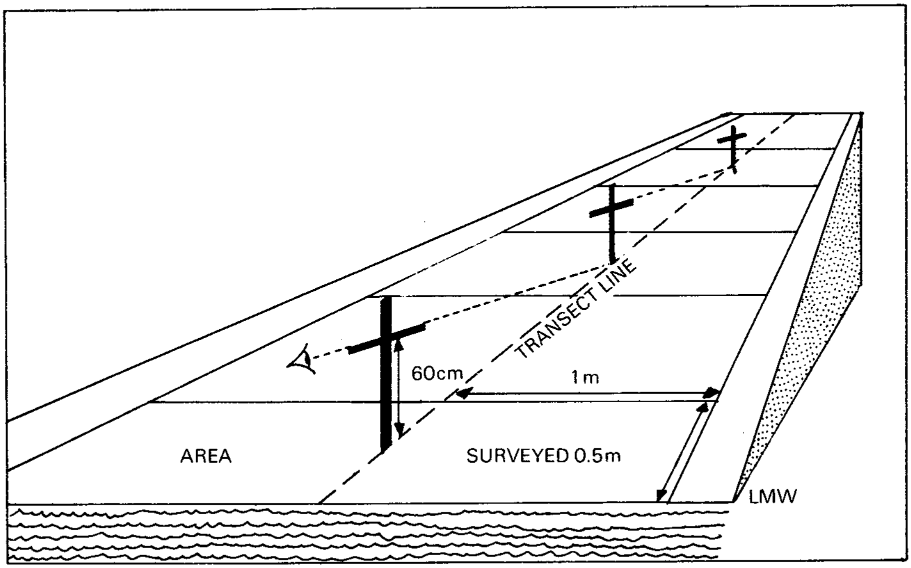

FIG. 3.
Schematic diagram showing the method of station location and sample areas along a 2 m belt transect. Sighting from station to station employs a cross-staff.

Quadrats

At the six transect sites surveyed by Crapp (1970) (Fig. 2), detailed measurements of density of *Patella vulgata* L., *Semibalanus balanoides* (L.), *Elminius modestus* Darwin and *Chthamalus** spp. were made in 1972 and 1974 (Baker, 1976*a*; Dicks, 1976). In 1981 these sites, and a further six, were re-surveyed (Fig. 2). A 0.25 m^2 quadrat was used for *P. vulgata* and a 0.0025 m^2 quadrat was used for barnacles. All measurements were taken on sloping bedrock faces of similar slope and aspect at mean tidal level. Quadrats were located randomly in a 10 m × 2 m horizontal strip using a tape measure and 0–99 random number tables. Limpet and barnacle densities were assessed in 20 replicate quadrats. In addition to densities, limpet size was measured across the longest shell axis using callipers. Sufficient random quadrats were examined to provide a minimum sample of 200 limpets.

RESULTS AND DISCUSSION

Transects

Early surveys of the discharge area in 1969 and 1970 are detailed in Crapp (1970). These were followed by experimental work and surveys in 1972, 1974, 1975 and 1978 (Baker, 1976*a;* Dicks, 1976; Petpiroon, 1982) and in 1981 as detailed below.

A 1958 photograph of Littlewick Bay in the 1968 Milford Haven Conservancy Board booklet shows that the shore was then dominated by limpets and barnacles. A slight gradient in exposure, as defined by the Ballantine (1961) exposure scale, was noted from east to west (Crapp, 1970) ranging from Grade 3 to Grade 5 (most shores here are Grade 4), with Littlewick Bay being slightly more sheltered than the shores either side.

The vertical distribution of the common species of shore animals and algae in 1978 on a typical shore in this area of the Haven is illustrated in Fig. 4.

*At the time of Crapp's (1970) survey the two species of *Chthamalus* were regarded as varieties of *C. stellatus* (Poli). We have retained this grouping for the purpose of continuity.

In 1970, the central transects (3 and 4, Fig. 2) had considerably reduced numbers of *Patella* spp., other gastropods and barnacles when compared with sites to the east and west, and elevated abundances of the algae *Fucus spiralis, F. vesiculosus* and *F. serratus*. These changes could not be explained by fluctuations in the degree of exposure. Hydrographic studies (Crapp, 1971; Dicks, 1976) 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 tides. It was concluded that the effects were caused by the effluent but that they were relatively minor, taking the form of changes in species composition and being restricted to within 200 m east and west of the discharge

Table 3. *Abundance scales used in scoring plants and animals along the reference transects (based on Crisp and Southward, 1958, modified by Crapp, 1970). Ex., extremely abundant; S, super-abundant; A, abundant; C, common; F, frequent; O, occasional; R, rare*

1. Angiosperms and lichens			
Ex	More than 80% cover	F	Large scattered patches
S	50–80% cover	O	Widely scattered patches, all small
A	20–50% cover	R	Only one or two patches
C	1–20% cover		
2. Algae			
Ex	More than 90% cover	F	Less than 5% cover, zone still apparent
S	60–90% cover	O	Scattered plants, zone indistinct
A	30–60% cover	R	Only one or two plants
C	5–30% cover		
3. Barnacles, *Littorina neritoides*, and small forms of *L.* "*saxatilis*"*			
Ex	More than 5 cm^{-2}	F	1–10 dm^{-2}, never more than 10 cm apart
S	3–5 cm^{-2}	O	1–100 m^{-2}, few within 10 cm of each other
A	1–3 cm^{-2}	R	Fewer than 1 m^{-2}
C	10–100 dm^{-2}		
4. Limpets (*Patella* spp.) and periwinkles (*Littorina* spp., but excluding *L. neritoides* and small forms of *L.* "*saxatilis*"*)			
Ex	More than 200 m^{-2}	F	1–10 m^{-2}
S	100–200 m^{-2}	O	1–10 m^{-2}
A	50–100 m^{-2}	R	Fewer than 1–10 m^{-2}
C	10–50 m^{-2}		
5. Gastropods (excluding *Patella* and *Littorina* spp.)			
Ex	More than 100 m^{-2}	F	Fewer than 1 m^{-2}, locally sometimes more
S	50–100 m^{-2}	O	Always fewer than 1 m^{-2}
A	10–50 m^{-2}	R	Fewer than 1–10 m^{-2}
C	1–10 m^{-2}, locally sometimes more		
6. *Mytilus edulis*			
Ex	More than 80% cover	F	Many scattered individuals and small patches
S	50–80% cover	O	Scattered individuals, no patches
A	20–50% cover	R	Fewer than 1 m^{-2}
C	Large patches but less than 20% cover		
7. *Pomatoceros triqueter*			
A	More than 50 tubes dm^{-2}	O	1–10 tubes m^{-2}
C	1–50 tubes dm^{-2}	R	Fewer than 1 tube m^{-2}
F	10–100 tubes m^{-2}		
8. "*Spirorbis*" spp.			
A	5 or more cm^{-2} on 50% of suitable surfaces	O	Fewer than 1 cm^{-2}
C	5 or more cm^{-2} on 5–50% of suitable surfaces	R	Fewer than 1 m^{-2}
F	1–5 cm^{-2} on 1–5% of suitable surfaces		

**Note:* At the time of Crapp's (1970) original survey the British rough winkles were regarded as belonging to a single, very variable species—*Littorina saxatilis*—and we have continued to use this grouping for the sake of continuity.

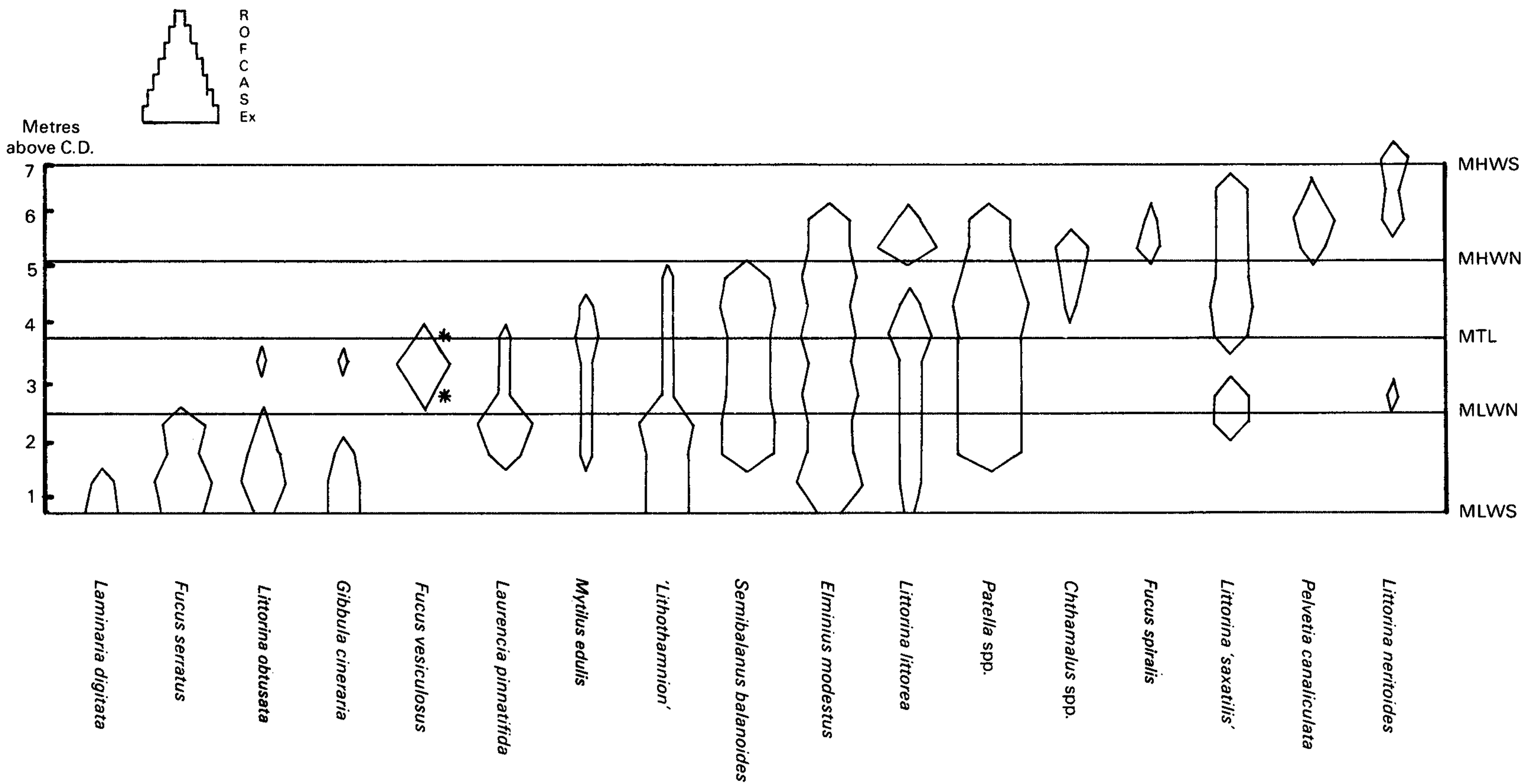

FIG. 4.

The vertical distribution and abundance of common shore species at Bullwell Bay, Milford Haven, in 1979.
* denotes the presence of the exposed shore form of *F. vesiculosus (F. vesiculosus* var. *limearis)*

point. Resurvey of the original Crapp transects in 1974 (Baker, 1976*a*) showed the same pattern of effect and revealed only minor (unexplained) differences from site to site. Unfortunately, the transects were not permanently marked by Crapp and relocation could only be approximate using the distances from the discharge point and selecting appropriate rock faces. In May 1978 the sites were again resurveyed (Petpiroon, 1982). The results for distribution of *Patella* spp., *Chthamalus* spp., *E. modestus, S. balanoides, F. serratus, F. vesiculosus, F. spiralis* and *Ascophyllum nodosum* are shown along with the results of original surveys by Crapp in Figs. 5–9.

The limpets and barnacles, usually found in substantial numbers on this type of shore, were considerably reduced in abundance on transects 3 and 4. The seaweeds *F. serratus, F. vesiculosus* and *F. spiralis* were all particularly abundant in the same transects and *A. nodosum* was found on transect 3.

The results of the 1978 study confirm and extend Crapp's findings of 1970. There is still a marked difference in the composition of shore biota between the transects nearest to the outfall (transects 3 and 4) and those further away from it (Figs. 5–9). Transects 3 and 4 still support abundant growth of fucoid algae but many animal species are absent from these sites. Those present are reduced in density compared to other transects away from the discharge point. The exclusion of the top shells (*Monodonta lineata* and *Gibbula umbilicalis*), the dog whelk (*Nucella lapillus*) and the barnacles (*Chthamalus* spp.) from transects 3 and 4 suggests that these species are less tolerant of the water conditions near the outfall. Similarly, the reduction in the density of limpet, *Patella* spp., and flat periwinkle, *Littorina obtusata,* populations near the outfall indicates that these species are also affected by the effluent discharged.

The abundance of *Fucus* spp. on transect 3 and 4 is very likely to be the consequence of reduced grazing pressure from molluscs (limpets, in particular) at these sites. The marginally greater shelter provided by the bay could be another factor favouring fucoid establishment, but it is known from photographic evidence that these algae were not there in great abundance before industrialisation began.

From Fig. 5 it can be seen that the occurrence of *E. modestus* is only marginally reduced on the central transects as compared with the other barnacles. It appears that *E. modestus* is more tolerant of effluent effects in Littlewick Bay than *S. balanoides* or *Chthamalus* spp. The latter two "species" show equally low abundance near the outfall site. In addition to the direct effluent effects, *S. balanoides* and *Chthamalus* spp. have been reported to do less well under the cover of a fucoid canopy (Lewis, 1964), and thus the low density of these barnacles may be indirectly affected by fucoid abundance on transects 3 and 4.

Minor changes in the density and vertical distribution of fucoids, limpets and barnacles between the 1970 and 1978 surveys are evident. These changes probably reflect more widespread alterations in shore communities of the Haven (Dicks and Hartley, 1982; Little, unpublished data) rather than additional effluent effects. The hot summer of 1976 was implicated in producing at least some of the changes, particularly in vertical distribution of organisms.

In addition to the illustrated distributions, other gastropods (littorinids, *Nucella lapillus*, and the top shells, *Gibbula umbilicalis* and *Monodonta lineata*) were also considerably reduced in density or absent from transects 3 and 4 adjacent to the effluent discharge point (Table 4).

In summary, the distribution patterns for most shore species found in 1978 were very similar to those reported by Crapp (1970). Effects of the effluent stream

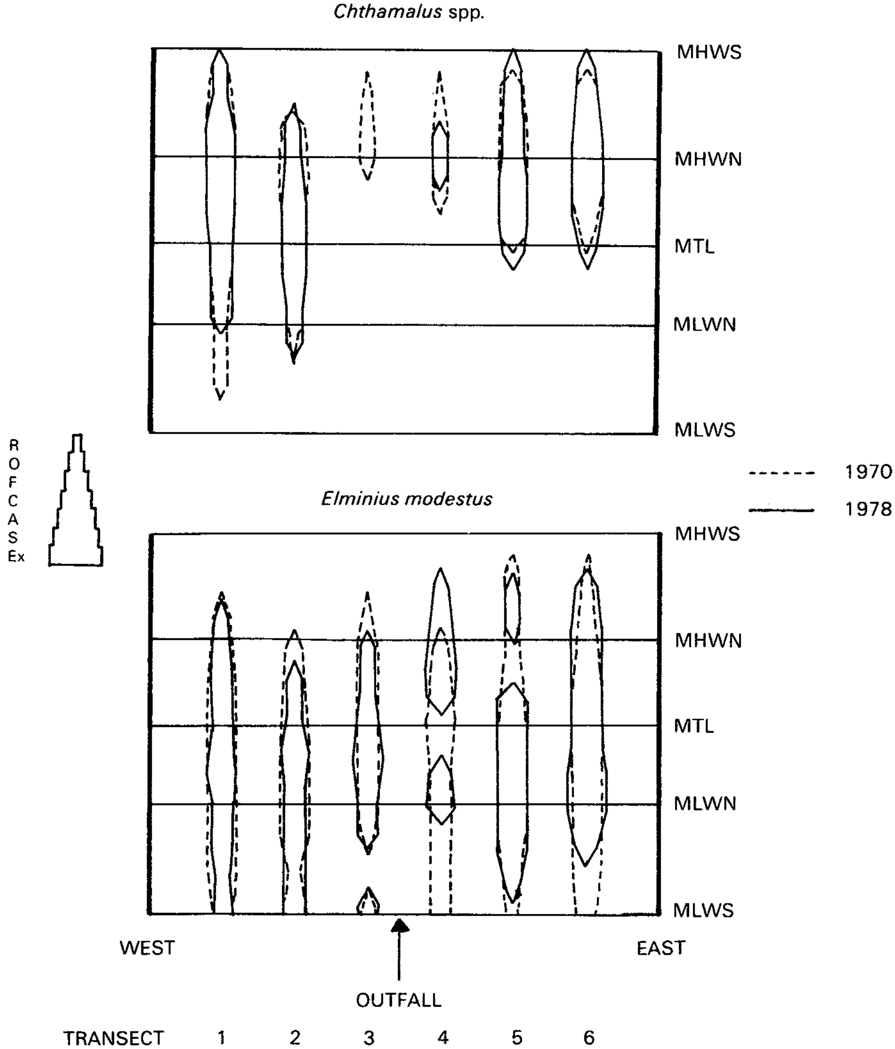

FIG. 5.
The distribution of *Chthamalus* spp. and *Elminius modestus* on the Littlewick Bay transects, Milford Haven, 1970 and 1978.

remain restricted to the shores immediately east and west of the discharge point. Changes in abundance and vertical distribution of species such as *Fucus* spp., *Patella* spp. and *S. balanoides* occur on most transects and reflect wider changes in the Haven rather than effluent effects.

Quadrat studies

In 1972, the *Patella* population showed a size/density gradient which extended approximately 120 m west and 100 m east of the outfall (Baker, 1976*a*). Densities were reduced close to the discharge, but those individuals present were of a large size. No juveniles were found within about 45 m of the outfall point. It was postulated that these effects could have arisen in one of two ways. The effluent might

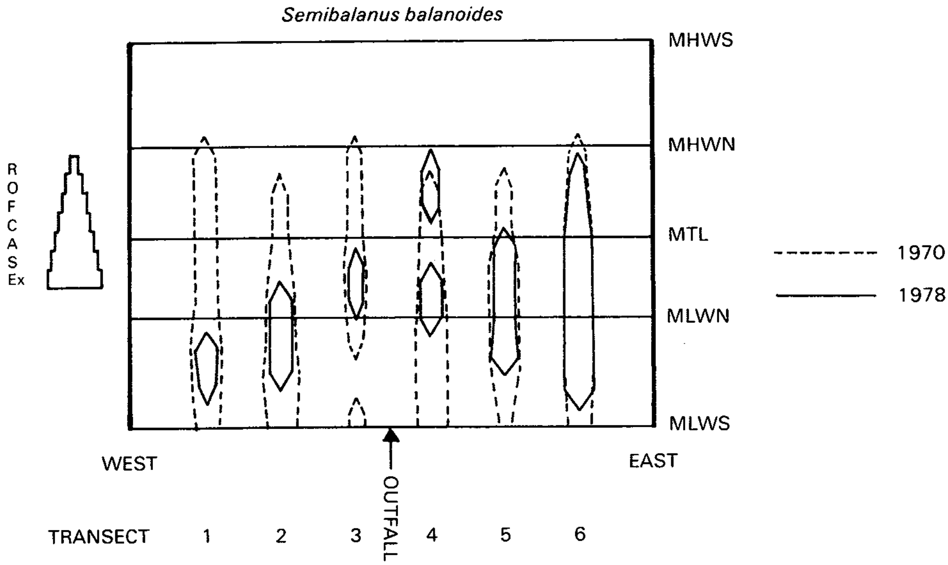

FIG. 6.
The distribution of *Semibalanus balanoides* on the Littlewick Bay transects, Milford Haven, 1970 and 1978.

have killed the majority of larvae/juveniles attempting to colonise the area; those surviving would have found abundant food and would have been able to grow rapidly. Alternatively, there might have been no successful larval settlement in recent years; the limpets then present were, perhaps, very old and pre-dated the 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. No limpets were seen immediately adjacent to the outfall point, but small numbers of very large individuals were living between 10 and 40 m shorewards on the pipe.

The same sites were resurveyed in 1981 and additional ones added to provide more comprehensive data (Fig. 2). Both mean density (Fig. 10) and mean shell size (Fig. 11) of limpets were similar to that found in 1972, but larval settlement had recently occurred. Some juveniles (<10 mm shell length) were found at the nearest sites both east and west, though none was found on the pipe itself.

The distribution of barnacles was studied in 1974 (Dicks, 1976). Reduced densities of *S. balanoides* (the dominant species of barnacle in the middle shore at that time) were found up to 40 m westwards from the outfall, but little effect was found to the east. 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 in producing these very localised changes and subsequent laboratory and field experiments supported this hypothesis (Dicks, 1976). Effluent was demonstrated to severely inhibit activity of the larval stages of *S. balanoides,* partly as a result of its low salinity, but also because of its burden of oil and other constituents.

Subsequently, some changes have been noted in barnacle distribution (see Table 5). By 1981 decreases in total barnacle density were evident at all central sites from 3

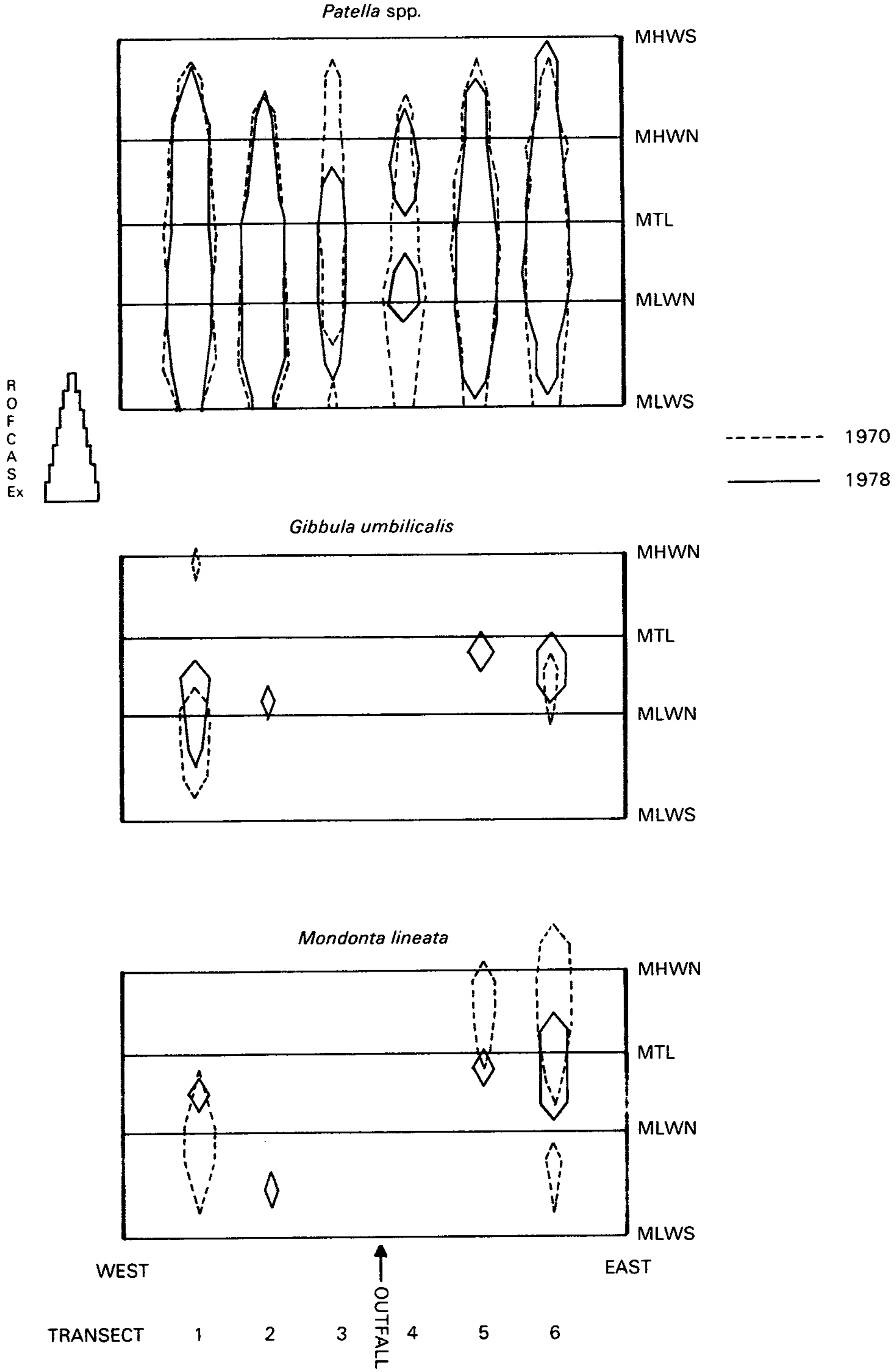

FIG. 7.
The distribution of *Patella* spp., *Gibbula umbilicalis* and *Monodonta lineata* on the Littlewick Bay transects, Milford Haven, 1970 and 1978.

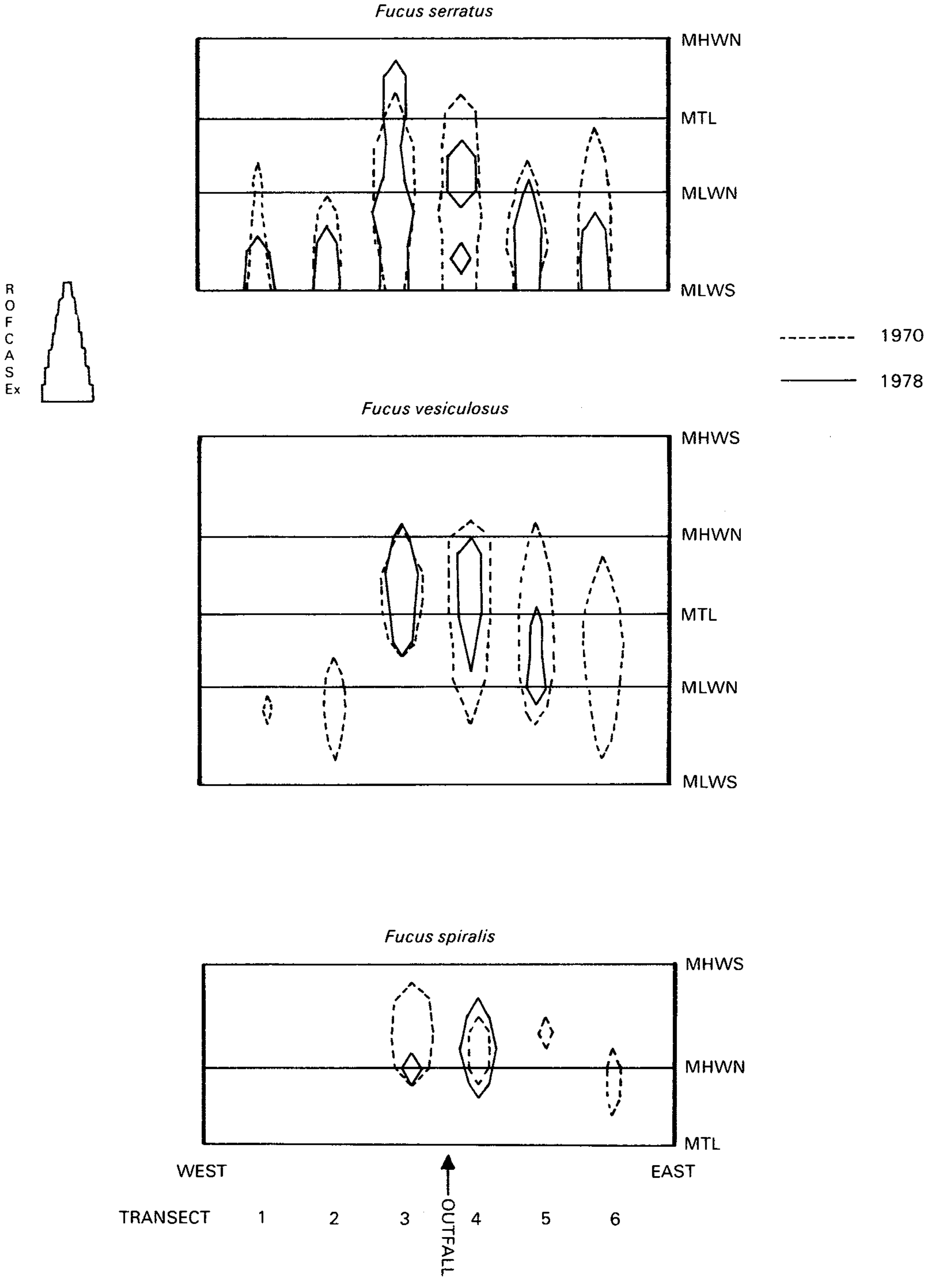

FIG. 8.
The distribution of *Fucus serratus, F. vesiculosus* and *F. spiralis* on the Littlewick Bay transects, Milford Haven, 1970 and 1978.

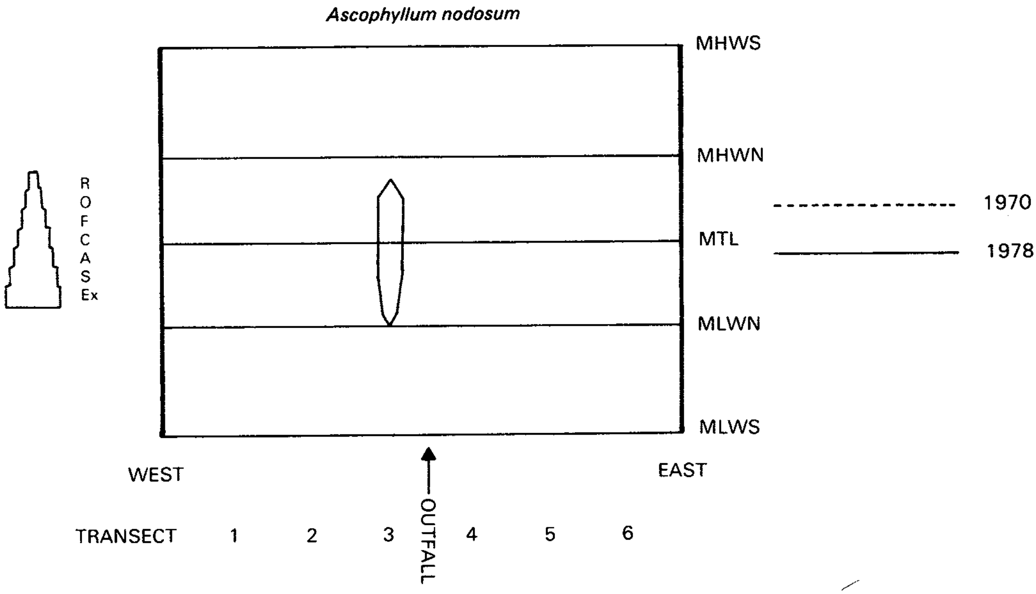

FIG. 9.
The distribution of *Ascophyllum nodosum* on the Littlewick Bay transects, Milford Haven, 1970 and 1978.

east to 3 west, possibly caused by the effluent but perhaps resulting from natural variability. There have also been changes in species composition of barnacle communities elsewhere in the Haven (Dicks and Hartley, 1982; Little, unpublished data) and these changes are reflected in Littlewick Bay. Up to 1974 the great majority of adults and many of the juveniles at all sites in Littlewick Bay were of *S. balanoides*. *E. modestus* and *Chthamalus* spp. accounted for less than 5% of the total density and were not included in the original counts. Photographic evidence confirms this statement, as do the data of Baker (1976*a*). In 1981 *S. balanoides* was still the commonest species at most sites but only accounted for about 65% of the barnacles present. At site 1 east 95% of the barnacles were *E. modestus*. Some of this change in species composition may have been as a result of the 1976 hot summer (as noted earlier) but the differential susceptibilities of the various species to effluent are not precisely known. There is need for caution in interpreting these changes.

Although barnacles, limpets and other mid-shore animals were absent from the end of the discharge pipe in all surveys, some algae have colonised. *F. serratus* and *F. spiralis* were both present but have not been observed to reproduce adjacent to the outfall, although specimens further shorewards on the pipe and on adjacent rock have formed fruiting bodies.

CONCLUSIONS

The findings of belt transects and quadrat studies have been in close agreement over the ten years of survey, and it is evident that only relatively small changes in the total effect of the effluent have occurred.

Table 4. *Summary of changes in abundance of common shore species on the Littlewick Bay transects between the two surveys: 1970 (Crapp) vs. 1978 (Petpiroon)*

Species	Transects 1	2	3	4	5	6
Algae						
Pelvetia canaliculata	⊖	⊖	+	–	–	–
Fucus vesiculosus	⊖	⊖	–	––	––	⊖
F. spiralis			––	+	⊖	⊖
F. serratus	nc	–	nc	––	–	––
Ascophyllum nodosum			(++)			
Laminaria digitata	nc	–	––	⊖	––	–
Enteromorpha spp.	++	+	nc	nc	+	–
Ulva spp.			–	–	⊕	⊕
Catenella caespitosa		⊖	–	nc	nc	
Gigartina stellata	nc	–	–	nc	nc	nc
Chondrus crispus	⊕		nc			⊕
Laurencia pinnatifida	–	⊖		⊖		
Lithothamnion spp.	–	–	––	⊖	–	⊕
Lomentaria articulata	+	–	nc	⊖	–	⊕
Palmaria palmata	–	nc	+	⊖	–	–
Porphyra umbilicalis	+	⊕			⊕	⊕
Animals						
Chthamalus spp.	–	–	⊖	–	nc	nc
Elminius modestus	nc	nc	nc	nc	–	nc
Semibalanus balanoides	––	––	––	––	––	–
Balanus crenatus	⊕	nc	nc	nc	+	+
Patella spp.	nc	nc	–	–	nc	nc
Gibbula umbilicalis	–	⊕			⊕	+
Monodonta lineata	––	⊕			–	–
Littorina littorea		⊖			+	–
L. obtusata	+	++	––	nc	++	––
L. neritoides	–	–		–	––	+
L. neglecta	–	–			––	++
L. "saxatilis"	++	++	+	+	++	++
Actinia equina	––	––	⊖	––	––	––
Spirorbis spirorbis			⊖	⊖		⊖
Pomatoceros triqueter	⊕	nc	nc	nc	+	+
Mytilus edulis	⊕				⊕	+
Nucella lapillus	⊖					nc

\+ = small increase
++ = large increase
– = small decrease
–– = large decrease
nc = present, no change. Absent, no change is left blank.
○ = change from absent or to absent

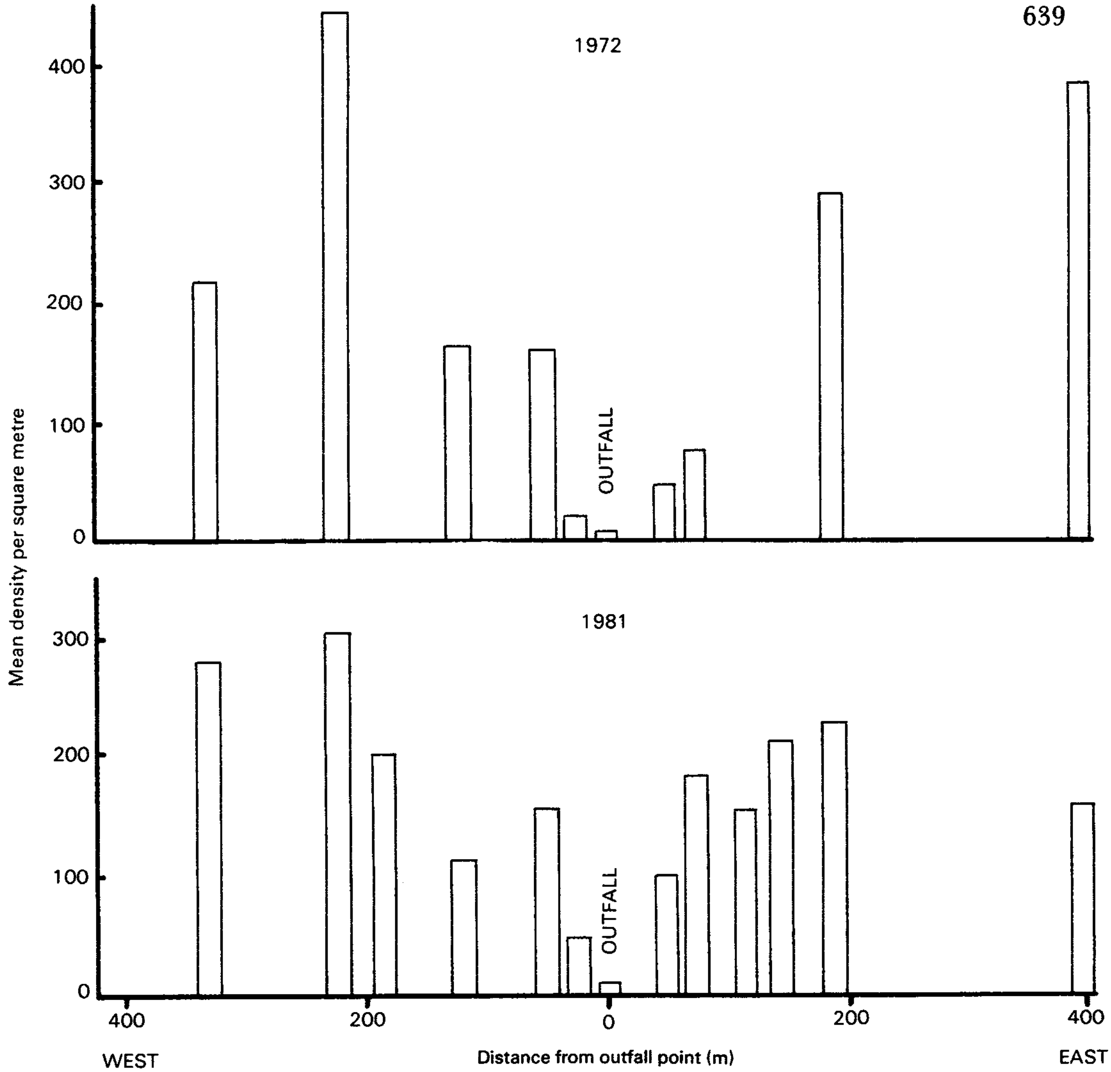

FIG. 10.
Mean density of *Patella* spp. in the middle shore at Littlewick Bay, Milford Haven, in 1972 (from Baker, 1976*a*) and 1981.

Table 5. *Densities of barnacles (adults and juveniles) in the Littlewick Bay mid-shore sites in 1974 (Dicks, 1976) and 1981. Densities are given per 5 × 5 cm quadrat* (mean of 20 quadrats)

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

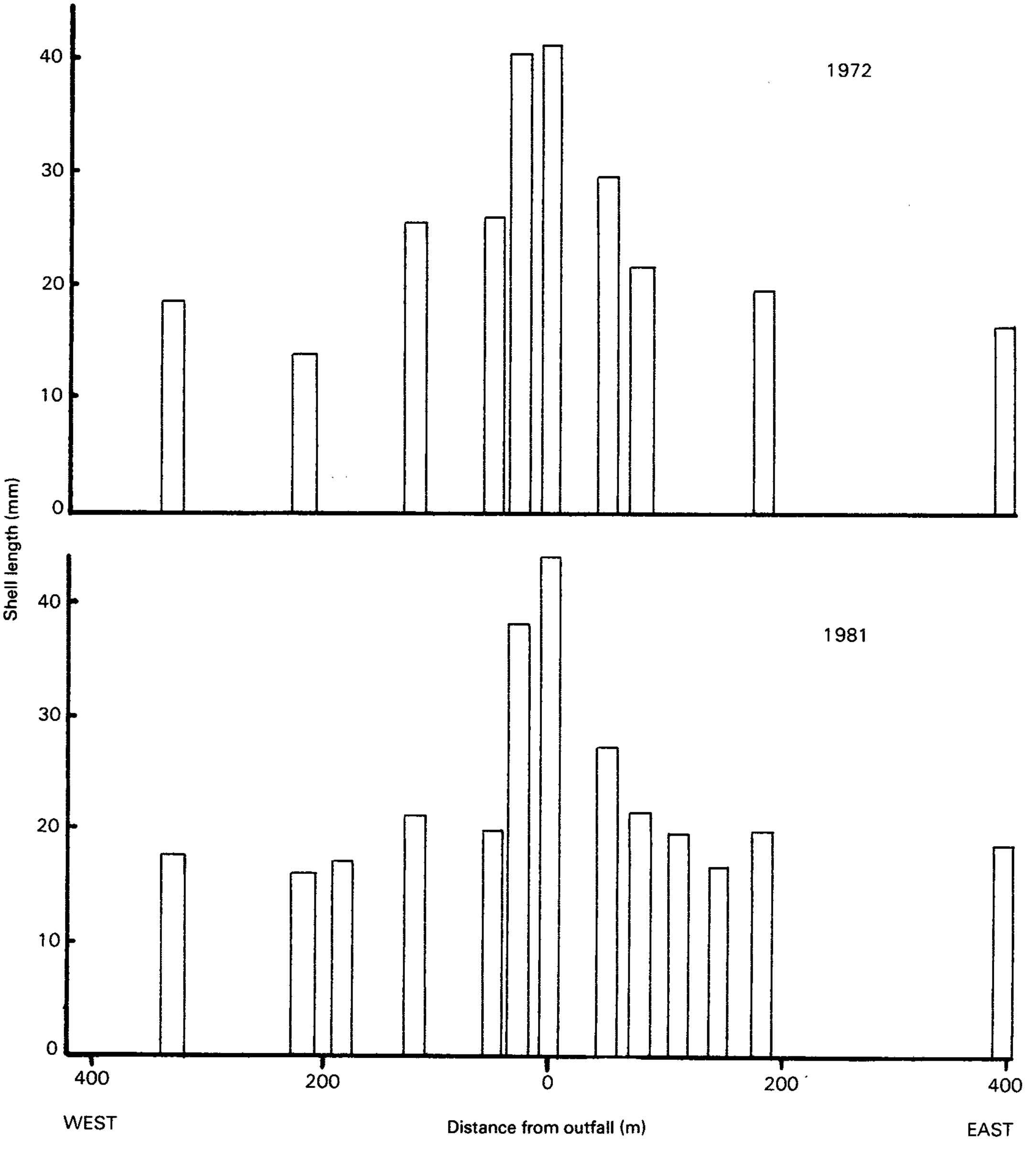

FIG. 11.
Mean shell length (mm) of *Patella* spp. in the middle shore at Littlewick Bay in 1972 (from Baker, 1976*a*) and 1981.

Effluent quality improvements in the early 1970s (to meet new Water Authority regulations) and a refinery "environmental awareness" campaign have resulted in steady improvement of the operation of effluent treatment equipment and a consequent reduction in oil content. These changes have not resulted in any corresponding reduction in areas of biological effect between 1970 and the present. However, since some of the observed effects are attributable to the reduced salinity of the effluent (rather than to its chemical content), changes in oil specification would not be expected to alter them.

It is likely that the low salinity of this particular effluent and the fact that it discharges into a small bay, slightly more sheltered than adjacent shores, are the key factors in producing the observed effects. The low salinity causes the effluent to float after discharge and it tends to pond as a surface layer in Littlewick Bay. The rise and fall of the tides result in short periods of contact with all levels of the shore during each tide, the eastern shores on flood tides and western shores on ebb tides. The effects of the effluent discharge cannot be considered as significant damage although they are biologically interesting. Small volume, relatively clean, effluents from modern, mainly air-cooled refineries discharged to a high-energy habitat where dispersion is good rarely result in significant biological damage (Baker, 1979; Dicks and Hartley, 1982).

References

BAKER, J. M. (1976*a*). Investigation of refinery effluent effects through field surveys. In: *Marine Ecology and Oil Pollution*. Ed. Baker, J. M. Applied Science Publishers, Barking. pp. 201–226.

BAKER, J. M. (1976*b*). Ecological changes in Milford Haven during its history as an oil port. In: *Marine Ecology and Oil Pollution*. Ed. Baker, J. M. Applied Science Publishers, Barking. pp. 55–66.

BAKER, J. M. (1979). The environmental impact of refinery effluents. *CONCAWE* Report No. *5/79*.

BALLANTINE, W. J. (1961). A biologically defined exposure scale for the comparative description of rocky shores. *Field Studies* **1** (3), 1–19.

CRAPP, G. B. (1970). *The Biological Effects of Marine Oil Pollution and Shore Cleaning*. Ph.D. Thesis, University of Wales, Swansea.

CRAPP, G. B. (1971). Monitoring the rocky shore. In: *The Ecological Effects of Oil Pollution on Littoral Communities*. Ed. Cowell, E. B. Applied Science Publishers, Barking. pp. 102–113.

CRISP, D. J., and SOUTHWARD, A. J. (1958). The distribution of intertidal organisms along the coasts of the English Channel. *Journal of the Marine Biological Association, U.K.* 37, 157–208.

CROTHERS, J. H. (1966). Dale Fort Marine Fauna. *Field Studies*. Supplement to Vol. 2. 169 pp.

DALBY, D. H., COWELL, E. B., SYRATT, W. J., and CROTHERS, J. H. (1978). An exposure scale for marine shores in Western Norway. *Journal of the Marine Biological Association, U.K.* 58, 975–996.

DICKS, B. (1976). The importance of behavioural patterns in toxicity testing and ecological prediction. In: *Marine Ecology and Oil Pollution*. Ed. Baker, J. M. Applied Science Publishers, Barking. pp. 303–320.

DICKS, B., and HARTLEY, J. P (1982). The effects of repeated small oil spillages and chronic discharges. *Philosophical transactions of the Royal Society of London*. Series B, 297, 285–307.

JONES, W. E., and WILLIAMS, R. (1966). The seaweeds of Dale. *Field Studies* **2**, 303–330.

LEMLIN, J. S. (1980). The value of ecological monitoring in the management of petroleum industry discharges: experience in Esso Petroleum Company U.K. refineries. In: *Proceedings of Conference on the Environmental Impact of Man's Use of Water*. I.A.W.P.R., Brighton, November 1980.

LEWIS, J. R. (1964). *The Ecology of Rocky Shores*. English Universities Press, London. 323 pp.

MOYSE, J., and NELSON-SMITH, A. (1963). Zonation of animals and plants on rocky shores around Dale. *Field Studies* **1** (5), 1–32.

National Academy of Sciences (1975). *Petroleum in the Marine Environment*. N.A.S., Washington, D.C.

NELSON-SMITH, A. (1965). Marine biology of Milford Haven: the physical environment. *Field Studies* **2**, 155–188.

NELSON-SMITH, A. (1967). Marine biology of Milford Haven: the distribution of littoral plants and animals. *Field Studies* **2**, 435–477.

ODUM, W. E. and JOHANNES, R. E. (1975). The responses of mangroves to man-induced environmental stress. In: *Tropical Marine Pollution*. Eds. Ferguson-Woods, E. J. and Johannes, R. E., Elsevier pp. 52–62.

PETPIROON, S. (1982). *Biological Effects of Coastal Oil Refinery Effluents*. Ph.D. Thesis, University of Wales, Swansea.

SYRATT, W. T., and COWELL, E. B. (1975). *The littoral ecology of the area around Monstad Refinery, Fensfjorden*. Baseline survey and related investigations 1971–1978. British Petroleum, London.

WARDLEY-SMITH, J., Ed. (in press). *The Control of Oil Pollution*. 2nd Ed. Graham & Trotman, U.K. 251 pp.