PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Some Conclusions Regarding Long-Term Biological Effects of Some Major Oil Spills [and Discussion]

J. H. Vandermeulen, H. M. Platt, J. M. Baker and A. J. Southward

Phil. Trans. R. Soc. Lond. B 1982 **297**, 335-351
doi: 10.1098/rstb.1982.0046

Phil. Trans. R. Soc. Lond. B **297**, 335–351 (1982)

at Britain

Some conclusions regarding long-term biological effects of some major oil spills

By J. H. Vandermeulen

Marine Ecology Laboratory, Bedford Institute of Oceanography, Dartmouth, Nova Scotia, Canada B2Y 4A2

Many of the world's major spills exhibit long-term consequences, associated mainly with lagoons, estuaries and marshes. This is due to the persistence of oil or petroleum fractions in these low-energy environments. The bioavailability of residual oil to infauna is influenced by several factors, such as solubility in water, feeding habit, weathering rate and sediment grain size. The time-courses for these long-term effects vary, but may run into decades for some community perturbations.

The effects are at all levels of organization, including cellular, organismic and the community. Although the number of documented long-term effects is small, they involve a wide range of biological processes: development, genetic, growth, feeding and assimilation, photosynthesis, recruitment and fecundity, and community stability.

It is important to note that the known effects are probably only representative of a much wider range of possible disorders that have occurred, but which have not been detected. This is due mainly to the selective nature of spill follow-up studies.

Long-term spill consequences are generally local phenomena and so far no single spill has, to our limited knowledge, significantly altered entire ecosystems or materially affected fisheries. The combination of several spills can, however, place considerable stress on an environment. Also, so far there is no indication of an increasing mutagenic or carcinogenic load in the marine environment due to biologically active petroleum fractions or to carcinogenic or mutagenic metabolites. There is, however, the possibility of local build-up of these compounds, as in 'hydrocarbon sink' areas, where such a burden may become a local problem.

Introduction

Not unexpectedly, all major spills have provided their own roster of effects, some major and others minor, but all have invariably been influenced by the behaviour of the spilled oil at the time. As well, with the exception of the considerable impact on marine birds, the bulk of the measured impact in each case has been on the adjoining coastlines.

That is not to say that there has not been any impact at sea, in the water column, because effects of various kinds have been documented. For example, there are the demonstrated effects on larval fish and eggs (the *Argo Merchant* (Grose & Mattson 1977; see also Wells 1982)) and the interesting observations on altered digestive enzyme patterns in zooplankton (Samain *et al.* 1981). However, with time the attention shifts away from the immediate short-term offshore effects and focuses on the more persistent long-term effects, primarily in low-energy coastal environments.

The reason, of course, lies in the persistence of hydrocarbons in these low-energy environments, while the open water column dilutes and self-cleanses readily. One purpose of this paper is to examine this persistence of residual oil, and to examine some of the factors that influence its bioavailability to infauna that live in these high-persistence environments.

At the same time there are aspects to the problem of oil in the marine environment that receive less frequent mention, possibly because they are not as obvious. These include potential long-term consequences surmised only from laboratory studies, as well as potential effects due to metabolites, formed either through chemical reactions or by biotransformation. While not obvious problems, they require separate mention because of the potential magnitude of their impact on the marine environment.

Oil persistence and fate

Oil behaviour at sea

The main characteristic of oil behaviour in the water column appears to be one of ready diffusion and brief persistence. It must be noted, however, that our understanding of oil distribution and movement through the water column, under spill conditions, comes in fact from very few field observations, bolstered to some extent by theoretical and laboratory measurements. Thus for the most part we are dealing with a slim data base, and with what are at best some educated guesses on hydrocarbon advection and diffusion.

Evaporation plays a major role in reducing and to some extent altering the chemical composition of surface oil slicks. Theoretical calculations suggest that in fact up to as much as 40% of the surface oil may become lost by this mechanism into the atmosphere during the first 24–48 h (MacKay & Leinonen 1977). This model ignored, however, several oceanographic features such as surface turbulence, and the estimate probably best applies to thin surface sheens on relatively calm surfaces, and least well to thick slicks or to 'mousse' under highly turbulent conditions.

Turbulence in the surface water layer is probably the other major factor acting to reduce the hydrocarbons at the water surface, in this case by entraining the surface oil into the water column underneath. This occurs in two ways: by direct dissolution of hydrocarbons into the water phase, and by breaking away of oil droplets from the overlying slick, which then advect and diffuse through the water column. This has been reported for several spills (e.g. *Arrow* (Forrester 1970), *Argo Merchant* (Cornillon in Grose & Mattson 1977) and *Kurdistan* (Vandermeulen *et al.* 1980)), although little is known of their subsequent behaviour or indeed of their composition. Gordon *et al.* (1976) examined experimentally this oil particle formation, and observed in their relatively quiescent mesocosm that a large part of the oil found in the water column did exist in particulate or microparticulate form (up to 98% in the size range 1–30 μm). A portion of this oil eventually did reach the bottom sediments in their experiments.

The distribution subsequently of oil in the water column is equally poorly understood. Observations from the *Argo Merchant* spill (Grose & Mattson 1977) and from the *Amoco Cadiz* (Marchand 1978) suggest that the bulk of the oil remains in the upper 10–20 m of the water column, and that levels return to some background level within weeks or, at worst, a couple of months. Even under blow-out conditions, e.g. *Ixtoc I*, the bulk of the oil in the water column appears to remain restricted to upper 20 m (Walter & Proni 1980). None the less, hydrocarbon concentrations can reach unexpectedly high levels, exceeding 100 ng g^{-1} in the *Amoco Cadiz* spill (Marchand 1978). These may last for several weeks.

As well, under particular local oceanographic conditions this stratified oil distribution in the water column is easily perturbed. For example, at the time of the *Amoco Cadiz* spill the water column became contaminated with hydrocarbons, down to 75 and 100 m, and with up to 152 ng g^{-1} concentrations. These various field observations show that the water column can become contaminated to a considerable depth.

In summary, in the water column we are dealing with a relatively short-lived hydrocarbon persistence, with a time course of days or at most weeks. Highest concentrations are associated with the air–water interface, with rapid decrease in concentrations in the surface waters. Under localized climatic or oceanographic conditions this stratified distribution profile may break down, with contamination occurring throughout the water column. These features are again short-lived, with concentrations in the nanogram per gram range persisting for days, and at worst 1–2 weeks (figure 1).

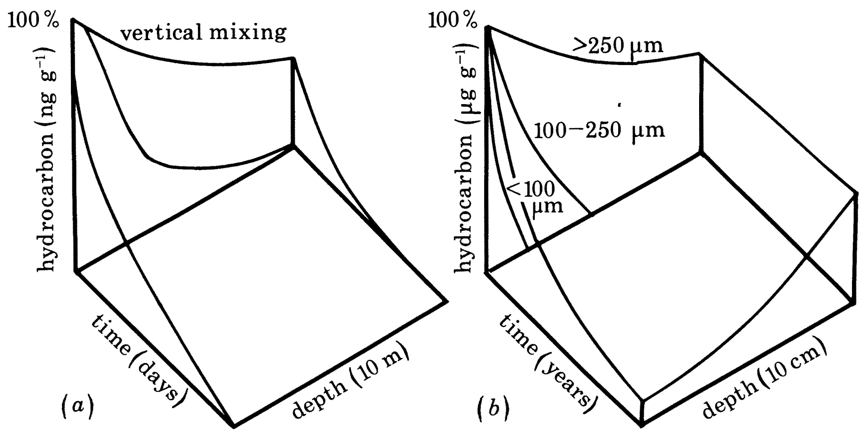

FIGURE 1. Scheme contrasting relative concentrations and residence times of petroleum in (*a*) offshore (water column) and (*b*) onshore (sediment) compartments of the marine environment. The sediment grain sizes are given in micrometres.

Unfortunately little is known of the concomitant compositional changes in this water-accommodated oil. Evaporation at the surface results in the removal of some of the lower molecular mass hydrocarbons, thereby making fewer of these toxic components available to the water column (see, for example, Struhsaker 1977; Struhsaker *et al.* 1974). On the other hand, the simultaneous physical entrainment of entire oil droplets from the underside of the surface slick can counteract this evaporative loss of the lighter fractions.

Bacterial degradation may act on both the surface slicks and on the water-accommodated hydrocarbons, although again this contribution to compositional changes is poorly understood. Although most oceanic waters sampled to date contain hydrocarbon-utilizing microbes (see, for example Mulkins-Phillips & Stewart 1974; Atlas *et al.* 1978), it is suspected that very little microbial degradation in fact takes place either in the water column or at the surface (Van der Linden 1978, p. 171).

One aspect of hydrocarbon transformation at sea, gaining increasing interest and significance, is the photoxidative alteration of oil components. Photochemical breakdown products similar to those obtained in laboratory experimental studies have now been identified in 'mousse' from the *Amoco Cadiz* spill (Overton *et al.* 1979) and more recently in spilled oil from the *Ixtoc I* blowout (Overton *et al.* 1980). Very little is known, however, of the rates of photochemically mediated oil breakdown at sea, nor of the possible routes, persistence or even toxicity of the reaction products. Indeed, our present ability to detect these compounds in any meaningful way in environmental samples, including tissues, is virtually non-existent. Still, it appears that while this route of hydrocarbon breakdown has been overlooked in past spill studies, in fact photochemical breakdown products have no doubt been companions of each spill incident so far, and in some manner also form part of the hydrocarbon toxic load on the environmental biota.

Behaviour of oil in sediments

Onshore, and to some extent in benthic sediments, the same processes of evaporation, dissolution, photochemical degradation, etc., occur, but to differing degrees. For example, evaporation still plays a major role in altering surface oil, but it ceases to be an active process

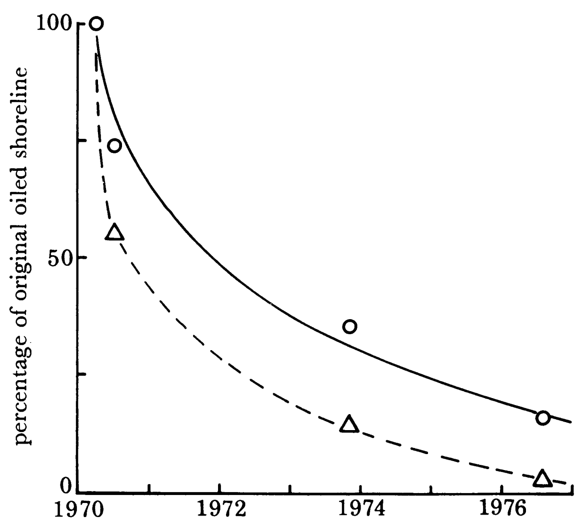

Figure 2. Erosion pattern of stranded *Arrow* Bunker C fuel oil on Chedabucto Bay shorelines, 1970–6. Curves are based on aerial and ground surveys in 1970, 1973 and 1976. The broken line refers to areas covered by heavy surface oiling only. The solid line describes the erosion pattern of total visible oil cover.

in buried oil. On the other hand, microbial degradation and dissolution become signifiant processes in terms of the long-term alteration and degradation of the stranded oil or its residue. Consequently the hydrocarbon persistence onshore differs markedly from that offshore, the residue persistence onshore being measured in years or even decades, and hydrocarbon concentrations often attaining high micrograms per gram levels (figure 1).

In high-energy and medium-energy coastal systems, self-cleaning of stranded oil is largely a function of coastal wave energy ('hydraulic action of breaking waves or swash and backwash' (Owens 1978)), and can proceed remarkably rapidly. A survey of surface oil erosion in Chedabucto Bay, Nova Scotia, over the 6 years after the *Arrow* break-up showed a steady although nonlinear erosion (figure 2). Half of the tar stranded along the midwater line on high-energy and medium-energy beaches was eroded by physical erosion in $1\frac{1}{2}$ to 2 years (Thomas 1973, 1978), and up to 75% of the oil was removed in the first 3 years after the spill (figure 2). Similar observations and time courses were obtained after the *Amoco Cadiz* spill for high-energy shorelines (see, for example, Vandermeulen *et al.* 1981). This pattern of rapid

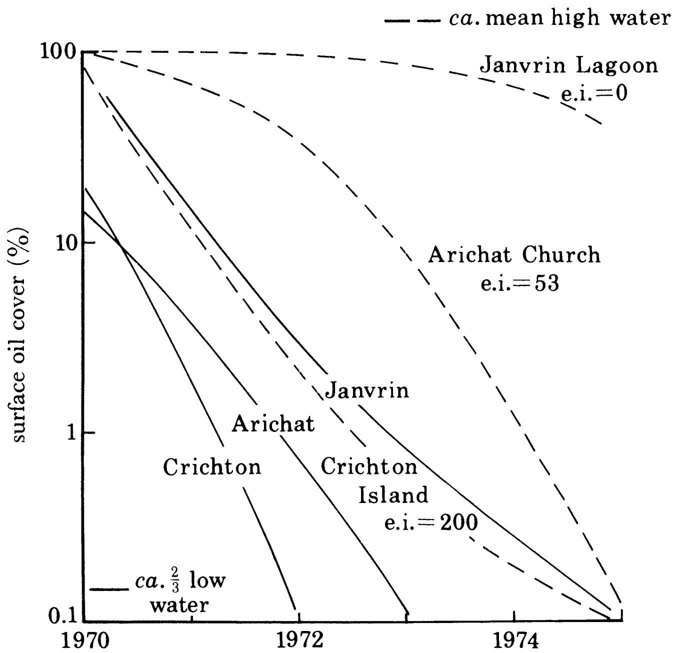

FIGURE 3. Rates of erosion of stranded Bunker C oil at two tidal levels under different wave-energy régimes, at representative stations in Chedabucto Bay, Nova Scotia (e.i. = exposure index, an index of shoreline wave exposure, based on wave energy, tidal action and shore topography). (After Thomas (1977).)

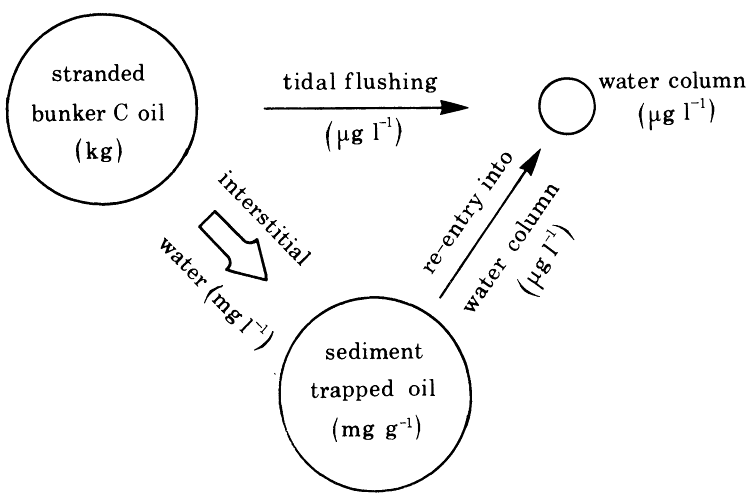

FIGURE 4. Summary of stranded Bunker C fuel oil re-entry routes for an oiled shoreline in Chedabucto Bay, Canada. (From Vandermeulen & Gordon (1976).)

self-cleaning for high-energy and medium-energy shorelines holds special significance for the recovery of the intertidal communities found here, in that for these communities the oil persistence is not a continuing problem or even a major factor.

In low-energy systems such as lagoons, estuaries and marshes, or even in the spray-zone communities at the high-tide line of high-energy, fine-sediment environments, the situation becomes very different in terms of oil persistence (Thomas 1978) (figure 3). Wave energy ceases to be a factor in these low-energy environments, and instead microbial degradation and

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

dissolution become prominent. Since both of the latter processes affect different hydrocarbon fractions disproportionately (affecting primarily the lower molecular mass fractions), the persistence of the larger polynuclear aromatic hydrocarbons (PAHs) and their transformation compounds takes on a greater ecological significance in these environments (figure 4).

The potential of long-term oil persistence in low-energy coastal environments was recognized early by Blumer (Blumer & Sass 1972; Blumer *et al.* 1973), and has since been demonstrated

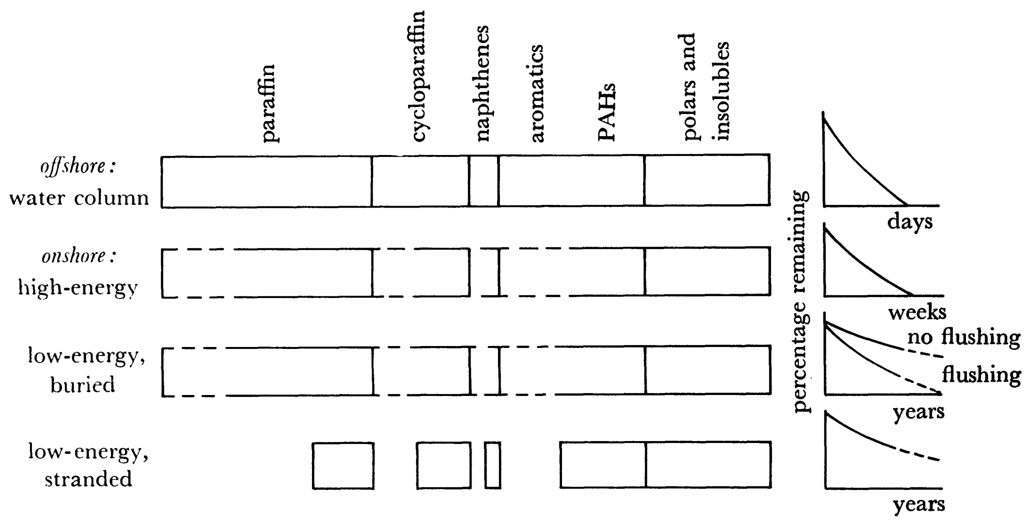

FIGURE 5. Scheme showing persistence of oil fractions as a function of location and environmental conditions. Based on experimental and field observations.

for most spills, e.g. *Florida* (Sanders 1978; Sanders *et al.* 1980; Teal *et al.* 1978), *Arrow* (Vandermeulen & Gordon 1976) and *Tamano* (Gilfillan *et al.* 1976). Indeed, the term 'hydrocarbon sink' is now commonly applied to these sort of low-energy, often fine-sediment, environments.

Originally, oil stranded along shorelines of these low-energy systems was perceived to remain immobile, susceptible only to periodic high-tide wave-action erosion, but it has since been found to be at least partially mobile (Vandermeulen & Gordon 1976). Surface tar from the *Arrow*, stranded along the upper tide mark of pebble–shingle beaches of Chedabucto Bay, Nova Scotia, was found to re-enter the marine environment under the combined action of tidal washing (dissolution) and interstitial beach pumping. Oil components from the stranded tar were washed down the beach face at each tidal incursion, only to enter into the interstitial pore water and become bound to the beach sediments. Only a minor portion was found to enter the water column either directly or subsequently from the beach sediments (figure 5).

This behaviour and re-apportioning of the stranded hydrocarbons is of special significance for associated intertidal and subtidal biota, since (*a*) they remain chronically vulnerable to low-level re-entry of hydrocarbons, either directly in the wash off the stranded tar, or indirectly from the oiled beach sediments, and (*b*) they potentially receive a steadily changing diet of hydrocarbons as result of long-term degradative processes taking place in the contaminant source. For example, the hydrocarbons bound within the beach sediments of Chedabucto Bay have lost most of the *n*-alkanes since 1970 (Vandermeulen *et al.* 1977). As a result, the clams in Janvrin Lagoon, Nova Scotia, are today no longer oiled by *Arrow* oil, but by an alkane-free derivative. Similar changes have been documented elsewhere. Measurements in the oiled Wild Harbor marsh, Massachusetts, have shown a similar loss in the lower aromatic fractions, with substituted naphthalenes and phenanthrenes decreasing in concentrations with time after the 1969 *Florida* spill and the 1974 spill in Windsor Cove (*Bouchard*) (Teal *et al.* 1978).

If we are to be absolutely thorough about this, however, then we must also take note of the absence of data on the formation of reaction products of either chemical transformation of the hydrocarbon residue in these sediments, or of the metabolite and other breakdown products resulting from microbial and more general biodegradation of the various fractions (see, for example, Gobson 1977). It is precisely because of the very long residence times in these low-energy 'sinks' that the potential formation of toxic reaction products becomes of significance in considering long-term impact on associated infauna and flora.

Bioavailability of oil: a case study

Following from the above it is interesting to compare oil persistence and its bioavailability in two coastal systems differing in self-cleaning, a high-energy rocky shoreline and a low-energy tidal river system, as obtained in current and continuing field studies (*Arrow*, *Amoco Cadiz*). In particular, I shall focus on a discussion of factors affecting oil bioavailability to the soft-shell clam, *Mya arenaria*, which inhabits and is representative of the biota associated with low-energy mudflats.

Hydrocarbon availability

The rocky shores of Ti Saozon (a small island off the coast of north Brittany near Roscoff) were heavily oiled after the *Amoco Cadiz* break-up. However, within days the very high concentrations of oil on this shoreline dropped dramatically (Chassé, personal communication), and within 2 months hydrocarbon levels in tide-pool water samples had decreased to near background levels (Vandermeulen *et al.* 1981); 9 months after the spill only traces of tar remained at the uppermost part of the rocky beach, mainly in and above the spray zone, with minor traces in the spray-zone tide pools.

Not unexpectedly, limpets taken from this site showed a parallel decrease in tissue hydrocarbon burden, corresponding to the observed decrease in environmentally available oil. It was noted, however, that the tissue hydrocarbon fluorescence pattern more closely resembled that of simultaneously collected tide-pool water samples than that of the parent stranded *Amoco Cadiz* oil. All samples contained traces of the smaller one-ring to four-ring aromatics, but only the parent oil contained the larger aromatics as determined by simultaneous excitation–emission fluorescence. These were present either in lesser amounts (water samples, figure 6) or only trace amounts (limpet tissues, figure 7) in the others.

The same sort of exercise carried out for the river mudflat – *Mya arenaria* system has shown the continued presence of *Amoco Cadiz*-derived hydrocarbons to now, associated mainly with the surface sediment layer. Preliminary investigations suggest that higher hydrocarbon concentrations are found in and near the vegetative berm of the river's edge, with asphalt-like pavement covering the sediments of the high tide line in some places in the upper reaches of the river.

Again, clams (*Mya arenaria*) taken from these sediments contained measurable hydrocarbon levels. However, in contrast to the *Patella* from the rocky shore, *Mya* tissue hydrocarbons did not drop to zero but continue, to this date, above background levels. In addition, the tissue hydrocarbon fluorescence spectra again resemble more closely those of water column samples rather than those of either stranded *Amoco Cadiz* oil or sediment-bound hydrocarbons.

These two sets of observations both point to the same bioavailability route of the stranded

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

hydrocarbons to these organisms, i.e. via the water rather than through the sediments. The manner in which this is accomplished differs somewhat, of course, for the two organisms (*Patella* sp. and *Mya arenaria*) involved – *Patella* by grazing on tidal-washed rock surfaces and *Mya arenaria* by sampling the water column above the sediment surface – but in each the end result

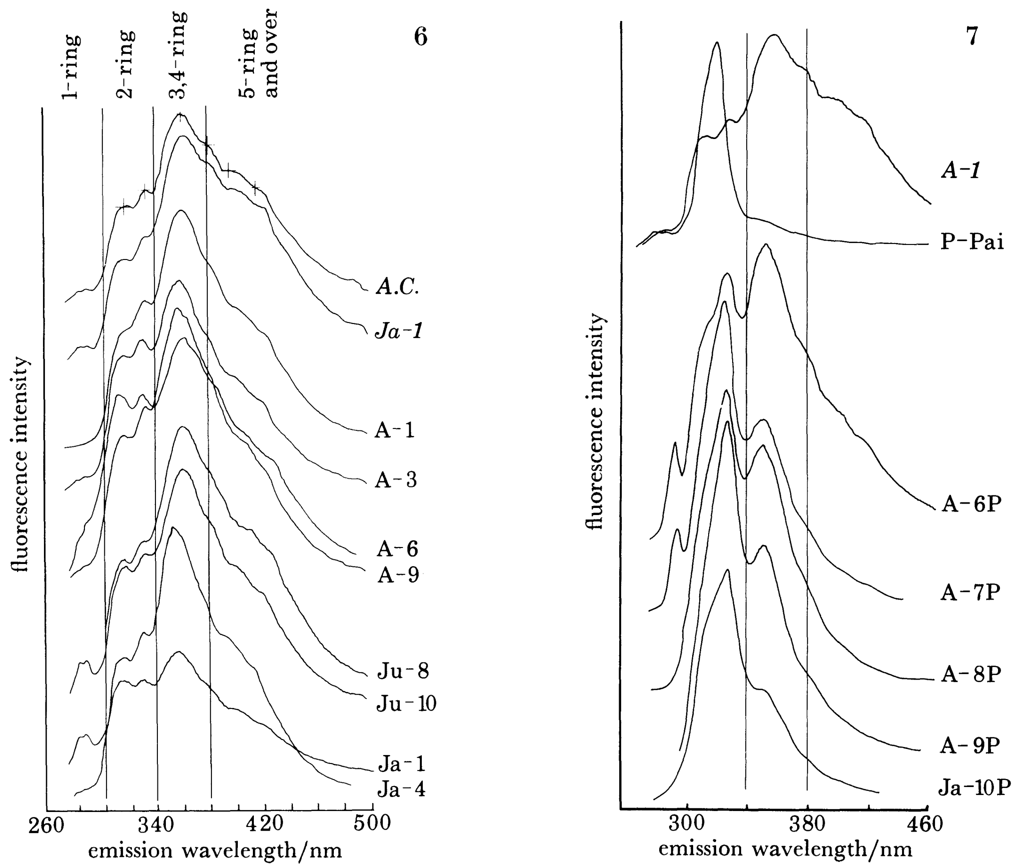

FIGURE 6. Fluorescence profiles (synchronous excitation–emission spectra) for tide-pool water samples taken from a high-energy rocky shore line after the *Amoco Cadiz* spill in April 1978. *A.C.* and *Ja-1* are *Amoco Cadiz* reference oil and January 1979 tar residue respectively. (A-1 to A-9, April 1978 water samples: Ju-8 and Ju-10, June 1978 samples; Ja-1 and Ja-4, January 1979 water samples. Indicated are the respective fluorescence regions for the different aromatic ring-compounds.)

FIGURE 7. Fluorescence profiles (synchronous excitation–emission spectra) for non-oiled limpets (P-Pai) from Paimpol, and for oiled limpets (A-6P to Ja-10) taken from oiled high-energy rocky shore after the *Amoco Cadiz* oil spill in April 1978. The profile for *Amoco Cadiz* reference oil sample is shown for comparison (*A-1*). (A-6P to A-9P, April 1978 limpets; Ja-10, January 1979 limpet.)

is the same. Only those fractions relatively soluble in seawater, i.e. the smaller aromatics, are eluted from the stranded oil and become available to the biota.

It is likely, however, that this simple model is mediated by various environmental or local factors. Earlier studies of an intertidal community contaminated with the oil from the *Arrow* showed higher contamination levels in *Mya arenaria* than in *Modiolus edulis* taken from the same study site, and, similarly, higher levels in *Spartina* than in a *Fucus* species (Vandermeulen & Gordon 1976). The differences there were ascribed to the burrowing or rooted habit of *Mya* and *Spartina* compared with those of *Modiolus* and *Fucus*, which live an attached existence above the sediments in the less contaminated water column.

Ecological and physical factors

Other factors that may affect the bioavailability of stranded oil components to *Mya* and other benthic organisms is their location relative to the height of the tidal cycle incursions plus their own feeding cycle.

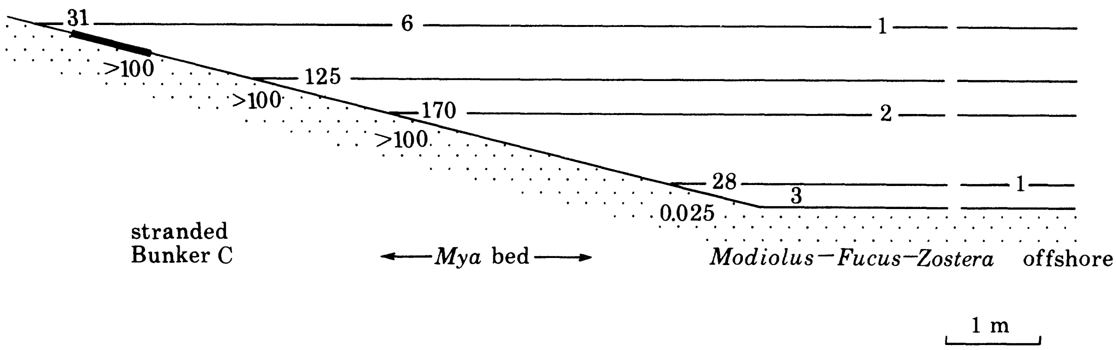

FIGURE 8. Petroleum hydrocarbon concentrations in tidal water and beach interstitial water for oiled shoreline, 5 years after *Arrow* Bunker C oiling. The heavy line at the top of the beach slope indicates the position of residual *Arrow* tar in 1975. Horizontal lines in the water column indicate four sampling points during the tidal cycle. Concentrations were determined spectrofluorimetrically: in water, they are expressed as micrograms per litre and in sediment as milligrams per litre. (For details see Vandermeulen & Gordon (1976).)

TABLE 1. HYDROCARBON CONCENTRATIONS IN WATER SAMPLES (100 ml) FROM TIDAL CHANNELS AND FROM SIPHON HOLES OF *MYA ARENARIA*

(April 1978, Aber Benoît Treglonou station. From Vandermeulen *et al.* (1981).)

sample description	hydrocarbon concentration† μg ml^{-1}
tidal channel:	
low tide	41.6
flooding tide – surface	7.6
subsurface	22.8
siphon holes:‡	
low tide	2453–52740
flooding tide (50 cm depth)	791.7

† Determined spectrofluorometrically. *Amoco Cadiz* oil characteristics confirmed with SEES scans.
‡ Siphon hole water collected from several clam holes and pooled for analysis.

Measurements of hydrocarbon concentrations in water samples taken from clam burrows over a tidal cycle showed (*a*) levels far greater than those found higher up in the tidal channels, and (*b*) a broad range of concentrations, with highest levels during the ebb phase when the burrows lie exposed (table 1). The very high levels in the clam burrows, especially during the ebb period, possibly represent residual hydrocarbons left behind during the receding ebbing tidal phase. Observations on tidal-water contamination over a beach contaminated with oil from the *Arrow* have shown an 'edge concentration', in which the highest hydrocarbon concentrations were found associated with the immediate edge of the advancing and receding water (figure 8).

These findings at first suggest a tremendous and chronic hydrocarbon insult to these bivalves, and no doubt this occurs in part. However, it is quite likely that the organisms avoid most of this contamination by (*a*) feeding only during the higher tide levels and (*b*) sampling only from above the sediment surface. Little is known of their possible avoidance of oiled waters, but this may play some role as well.

Another factor affecting more the benthic organisms of the upper tidal zone is their range relative to the tidal range. For the *Mya arenaria* in the Aber Benoît study, these are located near the top of the average tidal range, precisely where the water flow passing over them is nearing the flood stationary point, thus resulting in relatively longer exposure periods to the higher oil concentrations of the water's edge.

A further, subtler nuance on this theme has to do with the degree of exposure or cover of the stranded oil. Thus a comparison of water column oiling before and after a light burial of stranded *Arrow* oil with a fine gravel layer showed considerably reduced oil levels after the covering than before in the water column (Vandermeulen & Gordon 1976).

Taken together, these various observations represent a remarkable and somewhat bewildering interaction of chemical, behavioural and ecological factors affecting the bioavailability of stranded oil to benthic organisms in these coastal systems. Full understanding of long-term effects, however, also requires knowledge of individual physiological tolerances and peculiarities. Organisms differ in their tolerances and their ability to asimilate hydrocarbons. *Arenicola marina* seems to tolerate *Arrow* oil rather well (Gordon *et al.* 1978), but *Mya arenaria* fares less well with questioned hydrocarbon degradative ability (mixed function oxidase system) (Vandermeulen & Penrose 1978; Moore *et al.* 1980), and are susceptible to long-term physiological deterioration (see, for example, Gilfillan *et al.* 1976; Gilfillan & Vandermeulen 1978).

Long-term biological effects of some major marine oil spills

While each spill examined has been the subject of only limited long-term follow-up, in overview a broad range of long-term effects has been noted (see table 2). These range from abnormalities in development and recruitment disorders (e.g. (*Arrow*, *Florida*, *Tsesis*) to large-scale community perturbations (*Torrey Canyon*, *Florida*, *Tsesis*, *Amoco Cadiz*). Only one major spill, the *Argo Merchant*, has not yielded any long-term information.

It is worth noting that most of the known effects listed in table 2 are in fact accompanied by or are the result of parallel chronic oiling. The notable exception to this is the *Torrey Canyon* spill where, after the initial oiling and heavy mortalities, the recovery process proceeded on oil-free substrates cleaned by liberal use of dispersants. None the less, recovery to date has not reached pre-spill conditions, and the recovery process is still marked by fluctuations in densities and species composition typical of a perturbed recovery process (Southward & Southward 1978). The same interesting observations were made by Thomas (1978) in describing the recovery of some intertidal communities oiled after the *Arrow* spill, but self-cleaned very quickly. These observations emphasize the very important point that oil persistence is not a necessary requisite for long-term impact. Long-term effects at the community level persist and can be observed as well without chronic oiling.

The situation is far less understood, and is much less stable, for communities that face the combined effects of recovery instability and continuing chemotoxic effects of residual oiling. Obvious examples come from the *Florida* studies (Sanders 1978; Sanders *et al.* 1980), the

TABLE 2. SUMMARY OF KNOWN, LONG-TERM BIOLOGICAL EFFECTS OF SOME MAJOR MARINE OIL SPILLS

spills	effects	references
Torrey Canyon† 1967	continued community perturbations during recovery, 1967–1978	Southward & Southward (1978)
Florida† 1969	community abnormalities	Sanders (1978), Sanders *et al.* (1980), Cole (1978)
	genetic structure abnormalities in *Urosalpinx*, 1976	
	long-term inhibition of recruitment and low population densities in *Uca pugnax*: behavioural disorders	Krebs & Burns (1977)
Arrow 1970	intertidal species abnormalities, 1976	Thomas (1978)
	population abnormalities, *Mya*, 1976; C-flux depression in *Mya*, 1976; abnormal shell formation, *Mya*, 1976	Gilfillan & Vandermeulen (1978)
Searsport 1971	C-flux abnormalities in *Mya*, 1976	Gilfillan *et al.* (1976)
	suppressed recovery of *Mya* population, 1976	Mayo *et al.* (1978)
Zoe Colocotroni	partial recovery in red mangrove prop root community, 1979	Gilfillan *et al.* (1981)
Metula 1974	slow marsh recovery	Gundlach *et al.* (1982)
	alteration of total microbial ecology	Colwell *et al.* (1978)
Bouchard 1974	impaired reseeding and rhizome growth in salt marsh vegetation, reduced interstitial fauna, increased marsh erosion	Hampson & Moull (1978)
Argo Merchant 1974	no known long-term effects	
Tsesis 1977	reproductive effects leading to reduced *Pontoporeia* population, 1980	Elmgren *et al.* (1981)
	persistent disturbed community composition, 1980	Elmgren *et al.* (1981)
Amoco Cadiz 1978	benthic offshore sublittoral community perturbations, 1979	Cabioch *et al.* (1980)
	elevated mortalities in plaice, 1980	Friha & Conan (1981)

† Dispersants were used.

Bouchard spill (Hampson & Moull 1978), *Tampico Maru* (H. L. Sanders, personal communication), and the *Arrow* (Thomas 1978; Gilfillan & Vandermeulen 1978). Recovery of these communities at these sites is characterized by continued sharp fluctuations in densities and species, by the presence of opportunistic species, and by the slow recovery of the native species. An excellent example of this process has been provided by Elmgren *et al.* (1981) in connection with *Tsesis* spill. They note that 3 years after the spill the amphipod population, *Pontoporeia*, is still depressed whereas the clam, *Macoma*, apparently more resistant to the spilled oil, exists in inflated numbers, possibly as a result of unusually heavy recruitment during the time when

Pontoporeia was eliminated. This abnormal relation is thrown into further imbalance by the long lifespan of *Macoma* compared with that of the amphipod, which is thought to yield continued community disturbance for 5 or more years. The role of continued oiling throughout this period is totally unknown, but may be an integral part of the process.

It is too early to make intelligent guesses about such findings as the long-term genetic variability still found in Wild Harbor marsh *Littorina* (Cole 1978). However, the fact that this abnormality can be induced and can persist illustrates a long-term effect on a population at its most basic level, the gene pool.

Another comment concerns the kinds of long-term effect recorded, which in nearly all cases have involved either non-migratory organisms (*Mya*, *Urosalpinx*, etc.) or defined community structures (salt marsh, river mudflat, etc.), and for the most part in the intertidal zone. The reasons for this are obvious, of course – those of cost, accessibility of the study area, readily measurable effects – and most have focused on the low-energy environments, sites of greatest oil persistence, as the foregoing discussion has emphasized. This is not to say, however, that these are the only effects elicited, and that effects will be found only in the low-energy environments. These observed effects in many cases were singled out only because of the investigator's competence in that particular research area. It is regrettable in retrospect that the various spills were not examined by the same workers, for then all these spills would have been covered much more broadly, and today we would have a far better and more detailed understanding of the overall spill impact.

Potential problems

Not so obvious, but none the less real, are those consequences deriving from oil-induced effects known so far mainly from laboratory studies, and those due to metabolites. For example, little is known of oil impact on fish or fish stocks under field spill conditions. And, in all fairness it is likely that oil at sea has little significant impact on fish stocks (see, for example, Cole 1979). Indeed, it is improbable that we might measure or detect such impact (D. M. Ware, personal communication). On the other hand there is a growing data base originating from laboratory studies on the effects of oil and oil components of fish, on larval fish and on eggs (see, for example, Eldrige *et al.* 1977, 1978; Korn *et al.* 1979; Kuhnhold 1978; Kuhnhold *et al.* 1978; Moles *et al.* 1979; Rice *et al.* 1979; Struhsaker *et al.* 1974, 1977). Egg and larval stages appear to be particuarly sensitive in the fish life cycle, and abnormalities have been observed in the laboratory at hydrocarbon concentrations similar to those found under field spill or chronic pollution conditions. An effect of oil at sea on larval fish has been observed in one instance, the *Argo Merchant* spill (Longwell 1978).

It would therefore seem provident to re-examine the dogma that offshore effects are either ephemeral or non-existent, and instead consider useful studies that may be designed for the offshore in the event of a future major spill. It is agreed that there are awesome difficulties associated both with sampling and with separating out natural from oil-induced fluctuations. But is should be realized that our present sampling methods are simply not good enough to allow a categorical no-impact judgement. For example, any impact on larval fish is missed completely by present-day sampling techniques, since injured or dead larval fish are almost immediately lost from the water column either by predation or simple disintegration (S. D. Rice, personal communication).

Relevant here also are the observations of Samain *et al.* (1980) who monitored changing trypsin–amylase activity ratios in zooplankton populations of the *Amoco Cadiz* oiled waters off north Brittany. Their technique, first described for metal-induced enzyme changes in *Artemia salina* (Alayse-Danet *et al.* 1979), when applied to the broader oceanic plankton community showed remarkable correlations between the oiling incidence and the enzyme levels in these organisms. Although again not long-term *per se*, these findings do bear on possible broad physiological changes that can be elicited in water column populations. And they do hint at other such sweeping changes in populations that may well occur in the field, but without our being aware of their occurrence.

The second unknown is the biochemical fate of the petroleum hydrocarbons. Many undergo various biotransformations and are then instantly lost from detection, due mainly to our inability to follow these compounds effectively once they undergo even the simplest chemical modification. To our unprepared eye it appears then that the parent compound is lost from the sample or organism, and this is often interpreted as its being removed from our concern. But if we are to take a holistic view of chemical contamination, then our concern must extend to these secondary products, and we must wonder what pathways the secondary and even tertiary descendants follow, in sediments and in tissues.

This feature of spills is a valid concern that has not been part of spill follow-up studies to date, in part because of analytical limitations. Recent laboratory studies are now providing evidence that some of these biotransformation products may become directly associated with important intracellular macromolecule pools, including nucleic acids (Varanasi & Gmur 1980, 1981).

However, it should be noted that to date there exists no sign of an increasing mutagenic or carcinogenic load in the marine environment due to petroleum hydrocarbon metabolites *per se*. Any threats of that kind are more likely to come from other toxicants, such as from chlorinated organics. Hydrocarbon metabolites are, none the less, an inevitable product of spilled oil, and in the localized areas of high concentrations, as in the 'sink' areas, these may be of some long-term, albeit not understood, consequence.

Summary and conclusion

In summary, many of the world's major spills studied so far exhibit long-term effects on one kind or another. Most of these effects are associated mainly with the so-called low-energy environments, such as lagoons, estuaries and salt marshes. The time-courses for these long-term effects vary, but easily run into decades for some community perturbations.

It is significant to note that, although the persistence of oil is generally held to be requisite for the persistence of biological effects, in fact this is not always so. Continued disturbances in community structure demonstrated in two spill case studies (the *Torrey Canyon* and the *Arrow*) continue to this day on oil-free surfaces.

The observed and documented effects are probably only representative of many other possible disorders that have occurred and still occur on the sites of these spills. For the most part the effects that have been observed reflect the research area and capability of the investigator. Thus a listing of known long-term biological effects (table 2) is more correctly a listing of selected biological effects. That is to say, we have looked at only a few of the total possible effects. In this respect we would be much further ahead if the same group of investigators involved in long-term oil-spill studies had investigated all spills together. This also argues for closer integration of the various spill-study efforts.

The single most important conclusion to be drawn from this overview, however, is that a very broad range of effects can be elicited and that these persist, in some instances, for over a decade and possibly longer. They are found at all levels of organization – cellular, organismic and community – and involve a wide range of biological processes – development, genetic, growth, feeding and assimilation, photosynthesis, recruitment and fecundity, and community stability. These admittedly are all local effects, found here and there around the rim of the world's oceans. Also, so far no spill has significantly altered entire ecosystems or materially affected fisheries, to our knowledge. In addition, oiled systems do recover to some state approaching pre-spill conditions. At the same time, these effects are evidence of the ability of man to induce change in surprisingly large portions of the marine environment (the *Amoco Cadiz* affected about 360 km of Brittany shoreline, with impact in benthic sediments down to 50 m (Conan *et al.* 1981; Beslier *et al.* 1980)) and point out that, although the marine environment is huge and resilient, it is also vulnerable.

Special thanks are due to Dr P. G. Wells for the many hours of discussion of this subject and for his critical reading of the manuscript. Dr D. C. Malins and Dr J. Teal both read and commented on the final drafts. I express my appreciation to both.

References

Alayse-Danet, A. M., Charlou, J. L., Jezequel, M. & Samain, J. F. 1979 Modèle de détection rapide des effets sublétaux des polluants: modification des taux d'amylase et de trypsine d'*Artemia salina* contaminées par le cuivre ou le zinc. *Mar. Biol.* **51**, 41–46.

Atlas, R. M., Horowitz, A. & Busdosh, M. 1978 Prudhoe crude oil in arctic marine ice, water and sediment ecosystems: degradation and interactions with microbial and benthic communities. *J. Fish Res. Bd Can.* **35**, 585–590.

Beslier, A., Birrien, J. L., Cabioch, L., Larsonneur, C. & LeBorgne, L. 1980 La pollution des Baies de Morlaix et de Lannion par les hydrocarbures de l'*Amoco Cadiz*: Repartition sur les fonds et evolution. *Helgoländer wiss. Meeresunters.* **33**, 209–224.

Blumer, M. & Sass, J. 1972 Oil pollution: persistence and degradation of spilled fuel oil. *Science, N.Y.* **176**, 1120–1122.

Blumer, M., Ehrhardt, M. & Jones, J. H. 1973 The environmental fate of stranded crude oil. *Deep Sea Res.* **20**, 239–259.

Cabioch, L., Dauvin, J. C., Mora Bermudez, J. & Rodriguez Babio, C. 1980 Effets de la marée noire de l'Amoco Cadiz sur le benthos sublittoral du nord de la Bretagne. *Helgoländer Meeresunters.* **33**, 192–208.

Cole, H. A. 1979 Pollution of the sea and its effects. *Proc. R. Soc. Lond.* B **205**, 17–30.

Cole, T. J. 1978 Preliminary ecological-genetic comparison between unperturbed and oil-impacted *Urosalpinx cinerea* (Prosobranchia: Gastropoda) populations: Nobska Point (Woods Hole) and Wild Harbour (West Falmouth), Massachusetts. *J. Fish. Res. Bd Can.* **35**, 624–629.

Colwell, R. R., Mills, A. L., Walker, J. D., Garcia-Tello, P. & Campos-P., V. 1978 Microbial ecology studies of the *Metula* spill in the Straits of Magellan. *J. Fish. Res. Bd Can.* **35**, 573–580.

Conan, G., Laubier, L., Marchand, M. & d'Ozouville, L. 1981 Introduction. In *Amoco Cadiz: fates and effects of the oil spill. Proceedings of the International Symposium, Brest, 19–22 November 1979*, pp. 1–5. Paris: Centre National Pour l'Exploration des Océans.

Conover, R. J. 1971 Some relations between zooplankton and Bunker C oil in Chedabucto Bay following the wreck of the tanker *Arrow. J. Fish. Res. Bd Can.* **28**, 1327–1330.

Eldridge, M. B., Echeverria, T. & Korn, S. 1978 Fate of 14-C benzene in Pacific herring (*Clupea harengus pallasi*) eggs larvae. *J. Fish. Res. Bd Can.* **35**, 861–865.

Eldridge, M. B., Echeverria, T. & Whipple, J. A. 1977 Energetics of Pacific herring (*Clupea harengus pallasi*) embryos and larvae exposed to low concentrations of benzene, a monoaromatic component of crude oil. *Trans. Am. Fish. Soc.* **106**, 452–461.

Elmgren, R., Hansson, S., Larsson, U. & Sundelin, B. 1981 The *Tsesis* oil spill; acute and long-term impact on the benthic ecosystem. I.C.E.S. Contribution C.M. 1981/E, p. 24.

Elmgren, R., Vargo, G. A., Grassle, J. F., Grassle, J. P., Heinle, D. R., Langlois, G. & Vargo, S. L. 1980 Tropic interactions in experimental marine ecosystems perturbed by oil. In *Microcosms in ecological research* (ed. J. P. Giesy, Jr) (U.S. Department of Energy Symposium Series no. 52), pp. 779–800.

PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Forrester, W. D. 1971 Distribution of suspended oil particles following the grounding of the tanker Arrow. *J. mar. Res.* **29**, 151–170.

Friha, M. & Conan, G. 1981 Long term impact of hydrocarbon pollution from the *Amoco Cadiz* on the mortality of plaice (*Pleuronectes platessa*) in the Aber Benoit estuary. I.C.E.S. Contribution C.M. 1981/E, p. 55.

Gibson, D. T. 1977 Biodegradation of aromatic petroleum hydrocarbons. In *Fate and effects of petroleum hydrocarbons in marine ecosystems and organisms*. (ed. D. A. Wolfe), pp. 36–46. New York: Pergamon Press.

Gilfillan, E. S. & Vandermeulen, J. H. 1978 Alterations in growth and physiology of soft-shell clams, *Mya arenaria*, chronically oiled with Bunker C from Chedabucto Bay, Nova Scotia, 1970–1976. *J. Fish. Res. Bd Can.* **35**, 630–636.

Gilfillan, E. S., Mayo, D., Hanson, S., Donovan, D. & Jiang, L. C. 1976 Reduction in carbon flux in *Mya arenaria* caused by a spell of No. 6 fuel oil. *Mar. Biol.* **37**, 115–123.

Gilfillan, E. S., Page, D. S., Gerber, R. P., Hansen, S., Cooley, J. & Hotham, J. W. 1981 Fate of the *Zoe Colocontroni* oil spill and its effects on infaunal communities associated with mangroves. In *1981 Oil Spill Conference, Atlanta, Georgia, 2–5 March 1981*, pp. 353–360. Washington, D.C.: American Petroleum Institute.

Gordon, D. C., Jr, Dale, J. & Keizer, P. D. 1978 Importance of sediment working by the deposit-feeding polychaete *Arenicola marina* on the weathering rate of sediment-bound oil. *J. Fish. Res. Bd Can.* **35**, 591–603.

Gordon, D. C., Jr, Keizer, P. D., Hardstaff, W. R. & Aldous, D. G. 1976 Fate of crude oil spilled on seawater contained in outdoor tanks. *Environ. Sci. Technol.* **10**, 580–585.

Grose, P. L. & Mattson, J. S. 1977 *The Argo Merchant oil spill. A preliminary scientific report.* (133 pages plus appendices and plates.) U.S. Department of Commerce.

Gundlach, E. R., Domeracki, D. D. & Thebeau, L. C. 1982 Persistence of Metula oil in the Strait of Magellan six and one-half years after the incident. *Oil Petrochem. Pollut.* (In the press.)

Hampson, G. R. & Moull, E. T. 1978 No. 2 fuel oil spill in Bourne, Massachusetts. Immediate assessment of the effects on marine invertebrates and a 3-year study of growth and recovery of a salt marsh. *J. Fish. Res. Bd Can.* **35**, 731–744.

Korn, S., Moles, D. A. & Rice, S. D. 1979 Effects of temperature on the median tolerance limit of pink salmon and shrimp exposed to toluene, naphthalene and Cook Inlet crude oil. *Bull. environ. Contam. Toxicol.* **25**, 521–525.

Krebs, C. T. & Burns, K. A. 1977 Long term effects of an oil spill on populations of the salt-marsh crab *Uca pugnax*. *Science, N.Y.* **197**, 484–487.

Kuhnold, W. W. 1978 Effects of the water soluble fraction of a Venezuelan heavy fuel oil (no. 6) on cod eggs and larvae. In *In the wake of the* Argo Merchant. *Proceedings of a Symposium, January 1978*, pp. 126–130. Kingston: University of Rhode Island Center for Ocean Management Studies.

Kuhnhold, W. W., Everich, D., Stegeman, J. J., Lake, J. & Wole, R. E. 1978 Effects of low levels of hydrocarbons on embryonic, larval and adult winter flounder (*Pseudopleuronectes americanus*). In *Proceedings of a Conference on Assessment Ecological Impacts of Oil Spills, June 1978*, Keystone, Colorado: American Institute of Biological Sciences.

Longwell, A. C. 1978 Field and laboratory measurements of stress responses at the chromosome and cell levels in planktonic fish eggs and the oil problem. In *In the wake of the Argo Merchant. Proceedings of a Symposium, January 1978*, pp. 116–125. Kingston: University of Rhode Island, Center for Ocean Management Studies.

MacKay, D. & Leinonen, P. J. 1977 *Mathematical model of the behaviour of oil spills on water with natural and chemical dispersion.* Economic and Technical Review Report EPS-3-EC-77-19, Environmental Protection Service, Fisheries and Environment Canada.

Marchand, M. 1978 Estimation par spectrofluorométrie des concentrations d'hydrocarbures dans l'eau de mer en Manche Occidentiale a la suite du naufrage de l'Amoco Cadiz, du 30 mars au 18 avril 1978. In *Amoco Cadiz Premières observations sur la pollution par le hydrocarbures, Brest, France, 7 June 1978* (Actes de colloques no. 6), pp. 27–38. Brest: C.N.E.X.O.

Moles, A., Rice, S. D. & Horn, S. 1979 Sensitivity of Alaskan freshwater and anadromous fishes to Prudhoe Bay crude oil and benzene. *Trans. Am. Fish Soc.* **108**, 408–414.

Moore, M. N., Livingstone, D. R., Donkin, P., Bayne, B. L., Widdows, J. & Lowe, D. M. 1980 Mixed function oxygenases and xenobiotic detoxication/toxication systems in bivalve molluscs. *Helgoländer Meeresunters.* **33**, 278–291.

Mulkins-Phillips, G. J. & Stewart, J. E. 1974 Distribution of hydrocarbon-utilizing bacteria in Northwestern Atlantic waters and coastal sediments. *Can. J. Microbiol.* **20**, 955–962.

Overton, E. G., Patel, J. R. & Laseter, J. L. 1979 Chemical characterization of mousse and selected environmental samples from the *Amoco Cadiz* oil spill. In *1979 Oil Spill Conference, Los Angeles* (A.P.I. Publication no. 4308), pp. 169–174. E.P.A./A.P.I./U.S.C.G.

Overton, E. G., Laseter, J. L., Mascarella, W., Raschke, C., Nuiry, I. & Farrington, J. W. 1980 Photo-chemical oxidation of Ixtoc-I oil. In *Preliminary results from the September 1979 Researcher/Pierce Ixtoc-I cruise* (*Proc. Symp., Key Biscayen, Florida, 9–10 June, 1980*), pp. 341–386. N.O.A.A./RD/MP3 Publication.

Owens, E. H. 1978 Mechanical dispersal of oil stranded in the littoral zone. *J. Fish. Res. Bd Can.* **35**, 563–572.

Rice, S. D., Moles, A., Taylor, T. L. & Karinen, J. F. 1979 Sensitivity of 39 Alaskan marine species to Cook Inlet crude oil and no. 2 fuel oil. In *Proceedings, 1979 Oil Spill Conferences, Los Angeles*, pp. 503–507. Washington, D.C.: American Petroleum Institute.

Samain, J. F., Moal, J., Coum, A., LeCoz, J. R. & Daniel, J. Y. 1980 Effects of the *Amoco Cadiz* oil spill on zooplankton: A new possibility of ecophysiological survey. *Helgoländer wiss. Meeresunters.* **33**, 225–235.

Sanders, H. L. 1978 *Florida* oil spill impact on the Buzzards Bay benthic fauna: West Falmouth. *J. Fish. Res. Bd Can.* **35**, 717–730.

Sanders, H. L., Grassle, J. F., Hampson, G. R., Morse, L. S., Garner-Price, S. & Jones, C. C. 1980 Anatomy of an oil spill: long-term effects from the grounding of the barge *Florida* off West Falmouth, Massachusetts. *J. mar. Res.* **38**, 265–380.

Southward, A. J. & Southward, E. C. 1978 Recolonization of rocky shores in Cornwall after use of toxic dispersants to clean up the *Torrey Canyon* spill. *J. Fish. Res. Bd Can.* **35**, 682–706.

Struhsaker, J. W. 1977 Effects of benzene (a toxic component of petroleum) on spawning Pacific herring, *Clupea harengus pallasi*. *Fish. Bull. Mass.* **75**, 43–49.

Struhsaker, J. W., Eldridge, M. B. & Echeverria, T. 1974 Effects of benzene (a water-soluble component of crude oil) on eggs and larvae of Pacific herring and northern anchovy. In *Pollution and physiology of marine organisms* (ed. J. F. Vernberg & W. B. Vernberg), pp. 253–284. New York: Academic Press.

Teal, J. M., Burns, K. & Farrington, J. 1978 Analyses of aromatic hydrocarbons in intertidal sediments resulting from two spills of No. 2 fuel oil in Buzzards Bay, Massachusetts. *J. Fish. Res. Bd Can.* **35**, 510–520.

Thomas, M. L. H. 1973 Effects of Bunker C oil on intertidal and lagoonal biota in Chedabucto Bay, Nova Scotia. *J. Fish. Res. Bd Can.* **30**, 83–90.

Thomas, M. L. H. 1978 Comparison of oiled and unoiled intertidal communities in Chedabucto Bay, Nova Scotia. *J. Fish. Res. Bd Can.* **35**, 707–716.

Van Der Linden, A. C. 1978 Degradation of oil in the marine environment. In *Developments in biodegradation of hydrocarbons* (ed. R. J. Watkinson), vol. 1, pp. 165–200. London: Applied Science Publishers.

Vandermeulen, J. H. & Gordon, D. C., Jr 1976 Re-entry of five year old stranded Bunker C fuel oil from a low-energy beach into the water, sediments, and biota of Chedabucto Bay, Nova Scotia. *J. Fish. Res. Bd Can.* **33**, 2002–2010.

Vandermeulen, H. J. & Penrose, W. R. 1978 Absence of aryl hydrocarbon hydroxylase (AHH) in three marine bivalves. *J. Fish. Res. Bd Can.* **35**, 643–647.

Vandermeulen, J. H., Amero, B. & Ahern, T. P. 1980 Physical weathering of Kurdistan oil: droplet formation and effect on shore-ice melting. In *Scientific studies during the Kurdistan Tanker incident* (ed. J. H. Vandermeulen), *pp.* 105–119. Bedford Institute of Oceanography.

Vandermeulen, J. H., Keizer, P. D. & Penrose, W. R. 1977 Persistence of non-alkane components of Bunker C oil in beach sediments of Chedabucto Bay, and lack of their metabolism by molluscs. In *Proc. 1977 Oil Spill Conference, New Orleans*, pp. 469–473. Washington, D.C.: American Petroleum Institute.

Vandermeulen, J. H., Long, B. F. N. & Ahern, T. P. 1981 Bioavailability of stranded *Amoco Cadiz* oil as a function of environmental self-cleaning April 1978 – January 1979. In *Amoco Cadiz: fates and effects of the oil spill* (*Proc. Int. Symp., November 1979, Brest*), pp. 585–598. Brest: Centre National pour l'Exploitation des Océans.

Varanasi, U. & Gmur, D. J. 1980 Metabolic activation and covalent binding of benzo(a) pyrene to deoxyribonucleic acid catalyzed by liver enzymes of marine fish. *Biochem. Pharmac.* **29**, 753–761.

Varanasi, U. & Gmur, D. J. 1981 Hydrocarbons and metabolites in English sole (*Parophrys vetulus*) exposed simultaneously to ^{3}H-benzo(a)pyrene and ^{14}C-naphthalene in oil-contaminated sediments. *Aquat. Toxicol.* **1**, 49–67.

Walter, D. J. & Proni, J. R. 1980 Acoustic observations of subsurface scattering during a cruise at the *Ixtoc-I* blowout in the Bay of Campeche, Gulf of Mexico. In *Preliminary results from the September 1979 Researcher/Pierce Ixtoc-I cruise* (*Proc. Symp., Key Biscayne, Florida, 9–10 June*, 1980), pp. 525–544. N.O.A.A./RD/MP3 Publication.

Wells, P. G. 1982 Zooplankton. In *Oil and dispersants in Canadian seas: research appraisal and recommendations* (ed. J. B. Sprague, J. H. Vandermeulen & P. G. Wells). Environment Canada, E.P.S. Special Report Series. (In the press.)

Discussion

H. M. Platt (*British Museum (National History), U.K.*). In figure 7, fluorescence spectra of control specimens of *Patella* showed a distinct peak in the region associated with two-ringed aromatic hydrocarbons. Since polynuclear aromatic hydrocarbons are not thought to be synthesized by living organisms, could you suggest what chemical species might be responsible for this background noise and where it might come from?

J. H. Vandermeulen. No, I cannot. There are various compounds and groups of compounds, such as certain pigments, that will fluoresce in this range.

We have demonstrated this problem previously in work with *Mya arenaria*, where there exists a fair amount of native fluorescing material intrinsic to the organisms (see Vandermeulen & Gordon 1976). This interference from native fluorescing materials is one of the major problems in analysing environmental samples.

J. M. Baker (*Field Studies Council, Orielton Field Centre, Pembroke, U.K.*). One of Dr Vandermeulen's tables showed differences in Bunker 'C' oil levels in *Zostera* and *Fucus*. Could he describe how the samples were treated, as this has some bearing on whether one concludes that *Zostera* is translocating oil from its roots, or whether oil is merely adhering more to the *Zostera* surface because it has a more lipophilic cuticle than *Fucus*?

J. H. Vandermeulen. All samples were rinsed with redistilled acetone to remove petroleum hydrocarbons that might be adsorbed onto the plant of leaf surfaces, precisely to avoid the sort of external contamination referred to. In this way we hope that we are describing only the internal hydrocarbon content of these marine plants. We are satisfied that in fact this is what we have.

A. J. Southward (*Marine Biological Association, Plymouth, U.K.*). The lack of tissue hydrocarbons in *Modiolus demissus* from Chedabucto Bay is surprising in view of the fact that this species often lives buried in mud, which would be expected to retain some of the oil. *Mytilus* in Buzzards Bay were badly affected by the *Florida* spill. Does this difference tell us something about the relative behaviour and effects of Bunker C oil compared with no. 2 fuel oil?

J. H. Vandermeulen. I think that the difference lies in the ecological habit of the clams, and not in the oils. The mussels sampled from the study site in Chedabucto Bay were not buried within the mud, but were taken from rock and stone surfaces embedded in the shore mud face at the lower limit of the tidal range. They were in fact epibenthic, along with the *Fucus* samples, and they would be exposed to water-borne hydrocarbons rather than to hydrocarbons from within the beach and the beach sediments.