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Experimental Studies of the Effects of Drilling Discharges [and Discussion]

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Experimental studies of the effects of drilling discharges

By M. J. Leaver[1], D. J. Murison[1], J. M. Davies[1] and D. Rafaelli[2]

[1] *Department of Agriculture and Fisheries for Scotland, Marine Laboratory, P.O. Box* 101,
*Victoria Road, Torry, Aberdeen AB*9 8*DB, U.K.*

[2] *Department of Zoology, University of Aberdeen, Tillydrone Avenue, Aberdeen AB*9 2*TN, U.K.*

The long-term effects of diesel and four different low-toxicity oil-based drilling-mud cuttings on the chemistry and benthic fauna of a marine sediment were compared.

Eighteen tanks (mesocosms) containing beach sediment were deployed in a control and five treatment, three replicate experimental set-up. The drill cuttings used had total oil concentrations ranging from 6 to 15% and were added to the tanks in quantities calculated to be representative of hydrocarbon levels measured in sediments 400–500 m from platforms in the North Sea.

Selected parameters monitored at varied intervals throughout the experiment were: redox profiles, sulphide concentrations, hydrocarbon concentration and meiofaunal abundance. Numbers of macrofaunal organisms evacuating the sediments in the seven days after treatment application were also recorded.

Redox measurements showed all sediments other than controls to be significantly reduced, but no differences were observed between treatments. Sediments became most reduced after 3 months and showed recovery thereafter, approaching control values after 15 months.

Sulphide levels peaked at about 3 months and declined thereafter, but were still elevated above control levels after 15 months. Highest concentrations were recorded in treatments with the highest oil content.

Mesobenthic meiofaunal abundance was significantly reduced in all cuttings treatments, but effects from individual treatments were indistinguishable. Epi/endobenthic copepod abundance increased markedly in all low-toxicity treatments, and particularly in those with lower oil content, but remained similar to control levels in diesel treatments throughout the experiment.

In the seven days after treatments, sediment evacuation rates by *Tellina* were highest in diesel and tended to reflect oil concentrations in low-toxicity treatments.

The results show that in equal oil concentrations diesel and low-toxicity oil-based drilling-mud cuttings have indistinguishable effects on sediment chemistry, but that, even after 15 months of weathering, diesel-based cuttings are demonstrably more toxic to benthic fauna.

Introduction

A number of North Sea surveys have demonstrated that sediment hydrocarbon levels around drilling platforms are elevated for distances of up to 3000 m (Grahl-Nielson *et al.* 1980; McIntosh *et al.* 1983; Addy *et al.* 1984; Massie *et al.* 1985). Effects on benthic fauna have been observed up to 1000 m from platforms, although the most deleterious effects occur within 500 m, and these have been correlated with sediment oil concentrations (Davies *et al.* 1984). Abnormally low redox potentials (*Eh*) in the surface layers of sediments close to platforms also appear to be linked with large oil-concentrations (Addy *et al.* 1984).

Elevated hydrocarbon levels in sediments close to platforms are primarily due to the discharge of oiled drill-cuttings which are separated from circulating oil-based muds during drilling operations. In an attempt to minimize environmental damage there has, in recent years, been a progressive switch from the use of diesel in oil-based muds to oils with lower acute toxicities. By 1984, only 5% of wells in the North Sea were drilled with diesel-based muds whereas 70% were drilled with low-toxicity oil-based muds. These low-toxicity muds may be over 200 times less toxic, in 96 h LC_{50} tests to the shrimp *Crangon* than diesel muds (Blackman *et al.* 1982). However, Davies *et al.* (1984) concluded that, up to 1983, there was insufficient evidence to distinguish between the ecological effects of diesel and low-toxicity oiled cuttings in the field. Addy *et al.* (1984) have pointed out that the high levels of hydrocarbons detected close to platforms are not necessarily toxic to the incumbent fauna, because the effects observed are consistent with changes due to organic enrichment. Similarly Dow (1984), using an experimental tank/sediment system, found no differences between diesel and first generation low-toxicity oiled cuttings in effects on benthic meiofauna, sediment physico-chemistry and microbial activity.

Here we describe the treatment of a model marine sediment, contained in onshore tanks, with diesel and four different low-toxicity oiled cuttings. The physico-chemistry and meiofauna of the sediments were monitored for a period of 15 months. The major objectives of the experiment were to distinguish between organic enrichment and toxicity effects and to determine whether relatively long-term exposures to different (i.e. diesel and low-toxicity) oiled cuttings produce different biological effects in the sediments.

MATERIALS AND METHODS

1. *Experimental tank system*

In April 1984 18 cylindrical glass fibre tanks 1.3 m in diameter and 0.8 m deep (figure 1) were filled to a depth of 20 cm with a natural marine sediment collected from the low-water springs zone at Aultbea, Loch Ewe, Scotland. The sediment was overlain with 40 cm of seawater, half of which was changed daily. Analysis of sediment particle size-distribution was done on five 2 cm diameter cores taken to 16 cm depth from the field site, as described by McIntyre & Murison (1973). Meiofaunal analyses were performed as described below.

In July 1984 equal masses (530 g) of oiled cuttings were spread evenly on the sediment surfaces to give a coating of about 1 mm. This gives an oil loading of 400 p.p.m. by mass in the top 2 cm, assuming an average cuttings oil content of 10%, and is representative of hydrocarbon concentrations detected at 500 m from several North Sea platforms (Grahl-Nielson *et al.* 1980; McIntosh *et al.* 1983; Addy *et al.* 1984; Massie *et al.* 1985). Diesel (D) and

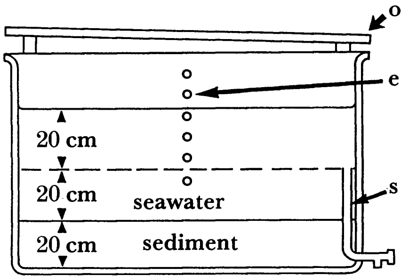

FIGURE 1. Section through a Loch Ewe sediment tank. o, Opaque lid; e, entry ports for seawater supply; s, standpipe.

four low-toxicity oiled cuttings (1, 2, 3, 4), obtained from North Sea drilling operations, were distributed randomly among five triplicate sets of tanks. A sixth undisturbed set acted as controls (C).

2. *Sampling programme*

Sediment redox profiles (*Eh*), sediment soluble sulphide concentrations, total hydrocarbons analysed by u.v.-fluorescence, *n*-alkanes by gas chromatography–flame ionization detector (GC–FID) and aromatics by gas chromatography–mass spectrometry (GC–MS) were determined before cuttings application and at 1 week and 1, 3, 5, 7, 10 and 15 months after application. Meiofauna was sampled on six occasions from two of the three tanks in each treatment over the 15 month experimental period. Care was taken to ensure that redox profiles and cores for sulphide, hydrocarbon and meiofaunal determinations were obtained from predesignated, undisturbed areas of sediment.

3. *Methods*

Redox profiles (up to six replicates) were generated by the method of Pearson & Stanley (1979) by using combination glass *Eh* electrodes (Russel pH Ltd., Type CMPT/RoD/SA/1.5 mm), by noting *Eh* at 1 cm intervals to a depth of 15 cm.

Soluble sulphide concentrations were determined by a method described in the Orion sulphide ion electrode, model 94-16, instruction manual (Orion Research Inc., Cambridge, Massachusetts, Form 94-16 IM/9730) and by Stanley *et al.* (1981) as modified by Dow (1984). Determinations were made on the top 5 cm of sediment (up to three replicate cores per tank) obtained with a truncated 5 ml syringe.

Sediment cores for hydrocarbon analyses, 2.2 cm in diameter and 2 cm deep, were obtained in duplicate for each tank by using a large truncated syringe and were immediately weighed into 500 ml screw top glass jars. Cores were also taken from each tank for dry mass analysis of the sediment.

Sediment cores and 15–20 g subsamples of fresh mixed cuttings were extracted into dichloromethane by the method of Massie *et al.* (1985). The volumes of the cuttings extracts were noted and 20 μl samples taken for further analysis before filtering and evaporating until all volatile solvent was removed. The remaining residues were weighed and then used as standard oils for u.v.-fluorescence analysis of the corresponding sediment samples.

Separation of sediment extracts and 20 μl cuttings extracts into the alkane and aromatic fractions was performed by high-performance liquid chromatography (P. R. Mackie, unpublished observations).

C_{15} to C_{33} *n*-alkanes, pristane and phytane were determined by capilliary GC–FID (Hewlett Packard 5880 with 2250 data system) as described by Mackie *et al.* (1978). Concentrations in original samples were determined from recovery of squalane standard with which the original extraction mixture was spiked.

Aromatic hydrocarbons were quantified by GC–MS (VG micromass 16F with VG 2250 data system) as described by Massie *et al.* (1985). Concentrations of aromatics in the original samples were determined from recovery of deuterated naphthalene, biphenyl, dibenzanthracene, anthracene and pyrene with which the original extracts were spiked.

u.v.-Fluorescence measurement of sediment oil concentrations was performed by the method of Massie *et al.* (1985) calibrating against oil extracted from the corresponding cuttings.

Meiofaunal samples were collected from 12 of the 18 tanks (i.e. 2 tanks per treatment) as duplicate 16 cm deep by 2.2 cm diameter cores by using transparent acrylic tubes. One core was divided into four horizontal sections of 4 cm each and the other retained intact. Samples were preserved in 5% neutral formalin with the addition of a small quantity of rose bengal stain until extraction of meiofauna by tapwater elutriation in a modified Boisseau apparatus. Enumeration of taxa was done on a static grid under a horizontal-scan stereo microscope.

Statistical analysis of meiofaunal abundance data was confined to the numerically dominant taxa, Nematoda and Harpacticoida (epi- and endobenthic). Transformed data were tested for homogeneity of cell mean-variances by Bartlett's test and overall differences between means were tested for significance by two-way analysis of variance.

RESULTS

Particle size-analysis of field site sediments indicated a moderately well-sorted fine sand (skewed to the right and leptocurtic) with median diameter 2.63 ϕ (0.162 mm).

Redox profiles taken immediately before cuttings addition, three months after setting up the sediment system, showed that all tanks had similar profiles to those later observed in control tanks throughout the experiment. Pretreatment sulphide levels were also similar to controls. Gravimetric analyses of fresh cuttings revealed oil contents as follows: diesel, 9%; low toxicity 1, 6%; low toxicity 2, 11%; low toxicity 3, 12%; low toxicity 4, 15%. 96 h LC_{50} tests of the muds to *Crangon* by R. A. A. Blackman (personal communication) were as follows: diesel, 55 p.p.m. by mass dry drilling mud; low toxicities 1, 2, 3 and 4, all greater than 10000 p.p.m. by mass dry drilling mud.

Redox profiles (which are plotted as means of up to six replicates) taken after cuttings additions, showed marked changes in the physico-chemical status of the sediment in all tanks to which cuttings were added. A rapid reduction in redox potential, reaching the least point in all treatments after three months (October), was followed by a slow recovery period. Fifteen months after the addition of cuttings, redox values at 1 cm below sediment surfaces were approaching those of control tanks (figure 2). Values at 9 cm showed a similar though less marked trend (figure 3) whereas at 15 cm there was little or no recovery. No differences could be discerned in redox profiles, at any time, between different treatments but all treatments were clearly different from controls which remained remarkably stable throughout the experiment (figure 4*a*–*c*).

Soluble sulphide in the top 5 cm of sediment of all cuttings treatments increased dramatically between one month and three months after cuttings addition. Thereafter sulphide levels fell in all treated tanks, but remained high when compared with those in controls. A slight increase occurred after 10–15 months in some cuttings treatments. The greatest sulphide concentrations appeared in the tanks with the highest oil loading (low toxicity 4), and the lowest in the diesel-cuttings treatment (excepting control tanks). The remaining treatments showed sulphide levels between those of diesel and low toxicity 4 and could not be separated (figure 5).

Total oil levels, determined from u.v.-fluorescence of sediment extracts, showed a steady decline in all cuttings treatments throughout the experiment (figure 6).

GC–MS data from all fresh cuttings and sediment cores 7 d after addition reveal very similar profiles, naphthalenes accounting for 65–75% of all polycyclic aromatic hydrocarbons (PAHs) analysed. By contrast, after 400 d naphthalenes were much reduced in all treatments and consequently anthracenes and phenanthrenes accounted for a larger percentage of total PAHs

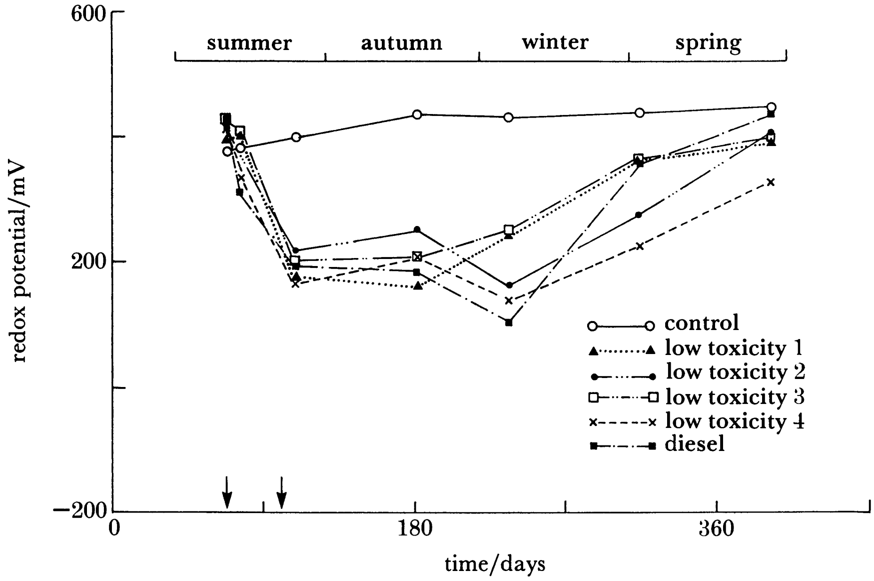

Figure 2. Redox potential of sediments at 1 cm depth over the period of the experiment. The arrows indicate times of treatments.

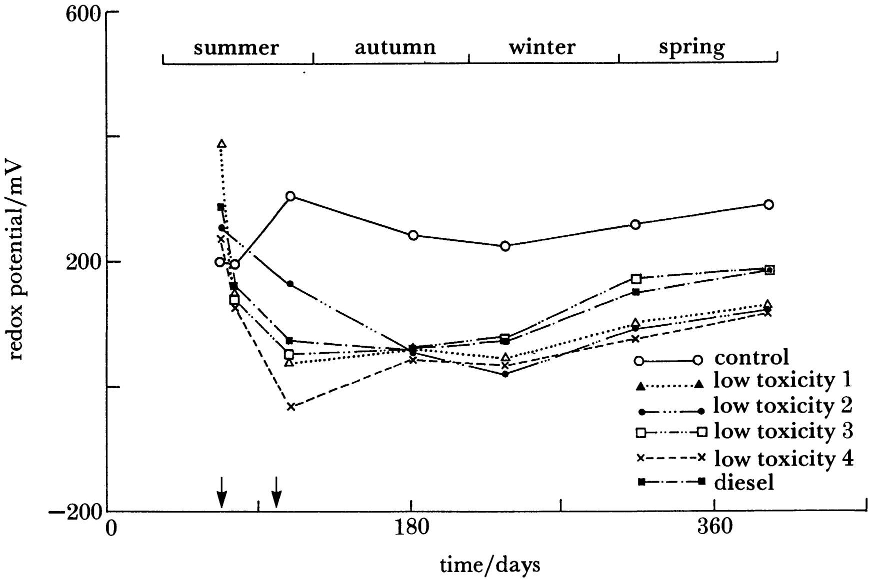

Figure 3. Redox potential of sediments at 9 cm depth over the period of the experiment. The arrows indicate times of treatments.

than previously. Diesel treatments showed the greatest total amounts of PAHs, in fresh cuttings, 7 d after addition and 400 d after addition, followed by low toxicities 4, 3, 2 and 1. Total PAHs were less after 400 d than after 7 d in all cuttings treatments (figure 7).

Alkane profiles generated from GC–FID data showed similar patterns in all cuttings treatments (figure 8*a*–*c*). Sediment cores taken from tanks 400 d after cuttings additions showed a general reduction in all *n*-alkanes when compared with fresh cuttings. This was obvious from examination of the relative amounts of essentially non-degradable pristane and phytane

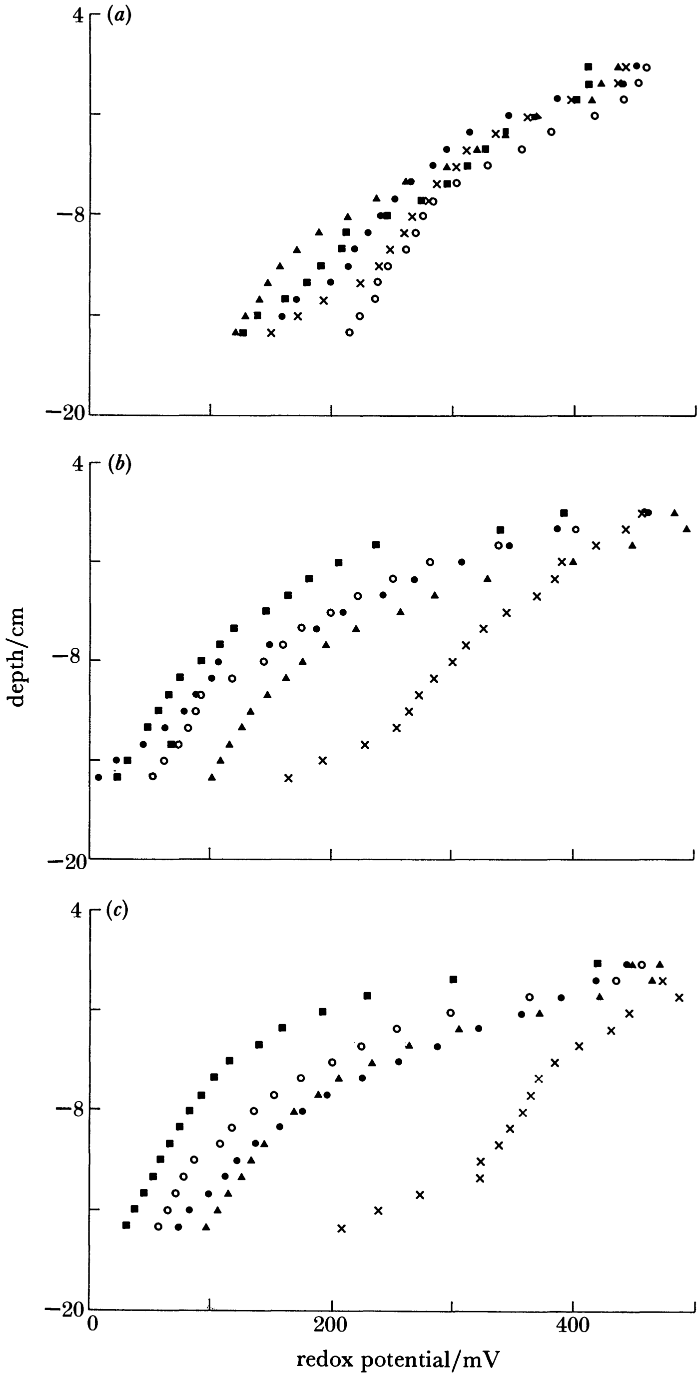

Figure 4. (*a*) Redox profiles of control tanks over period of experiment. (*b*) Redox profiles of low toxicity 4 tanks over the period of the experiment. (*c*) Redox profiles of diesel tanks over the period of the experiment. Symbols used: ×, predose; ●, 7 d; ■, 114 d; ○, 248 d; ▲, 470 d.

compared with C_{17} and C_{18} *n*-alkanes. The proportion of unresolved complex mixture (UCM) and more complex resolved aliphatic components also increased, particularly in low toxicity 4 treatments, which contained the greatest total level of aliphatics.

Laboratory examination of five field-site meiofaunal samples (day 0) indicated an abundant and fairly diverse community, numerically dominated by nematodes (84 %) with harpacticoid

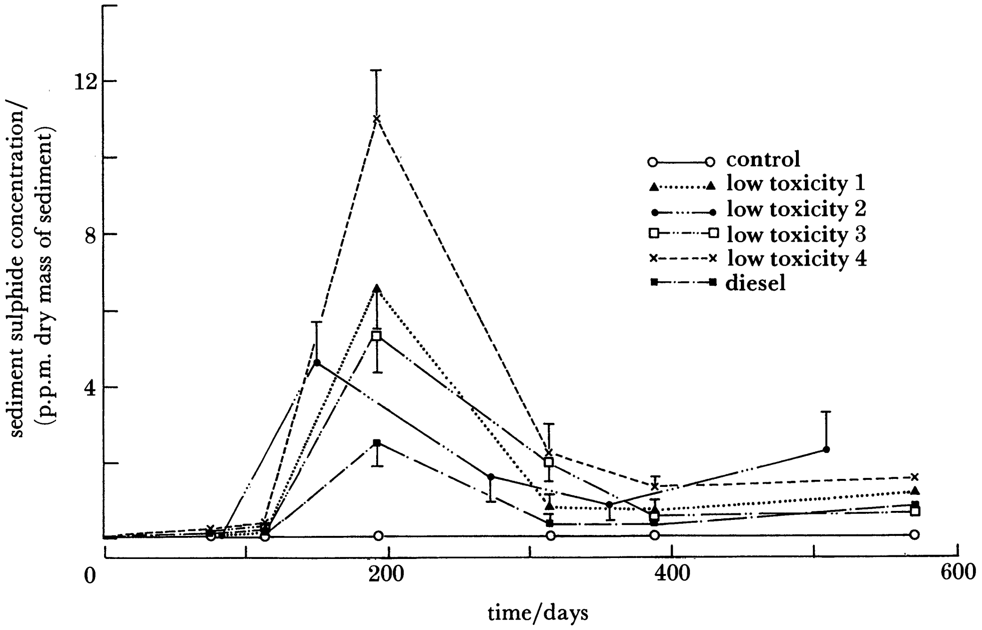

FIGURE 5. Sulphide concentrations of the tank sediments over the period of the experiment. The bars represent + and − standard error of the mean.

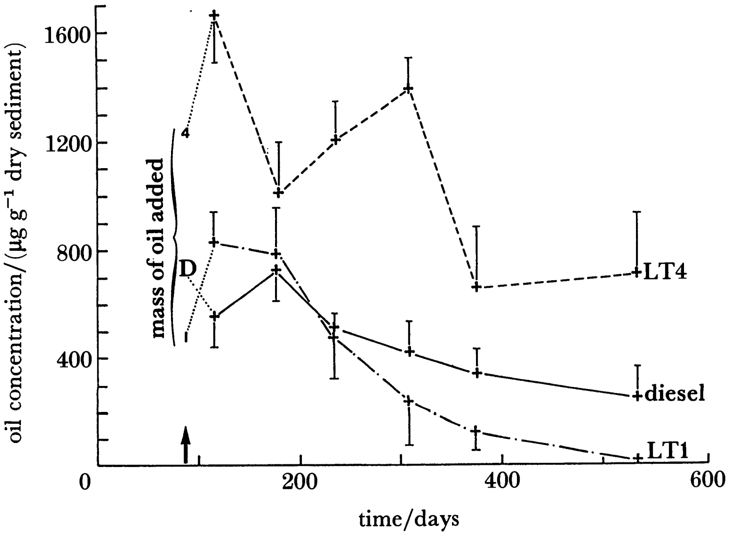

FIGURE 6. Total oil concentrations in the tank sediments over the period of the experiment. The bars represent + and − standard error of the mean. The arrow indicates the time of cuttings addition. LT1 and LT4 stand for low toxicity 1 and low toxicity 4.

copepods (epi-, endo- and mesobenthic), turbellarians, gastrotrichs and ciliates well represented (figure 9). Smaller numbers of other taxa including tardigrades, oligochaetes, archiannelids, ostracods, halacarids and temporary meiofaunal stages of polychaetes and bivalves were also recorded. Pretreatment meiofaunal abundance was recorded in the 12 tanks after a 9 week acclimation period (day 63) and showed a similarly diverse community with mean abundance reduced by about 25% (figure 9). Nematodes remained dominant (83%) and for this reason were used as a convenient index of tank meiofaunal abundance after addition of cuttings.

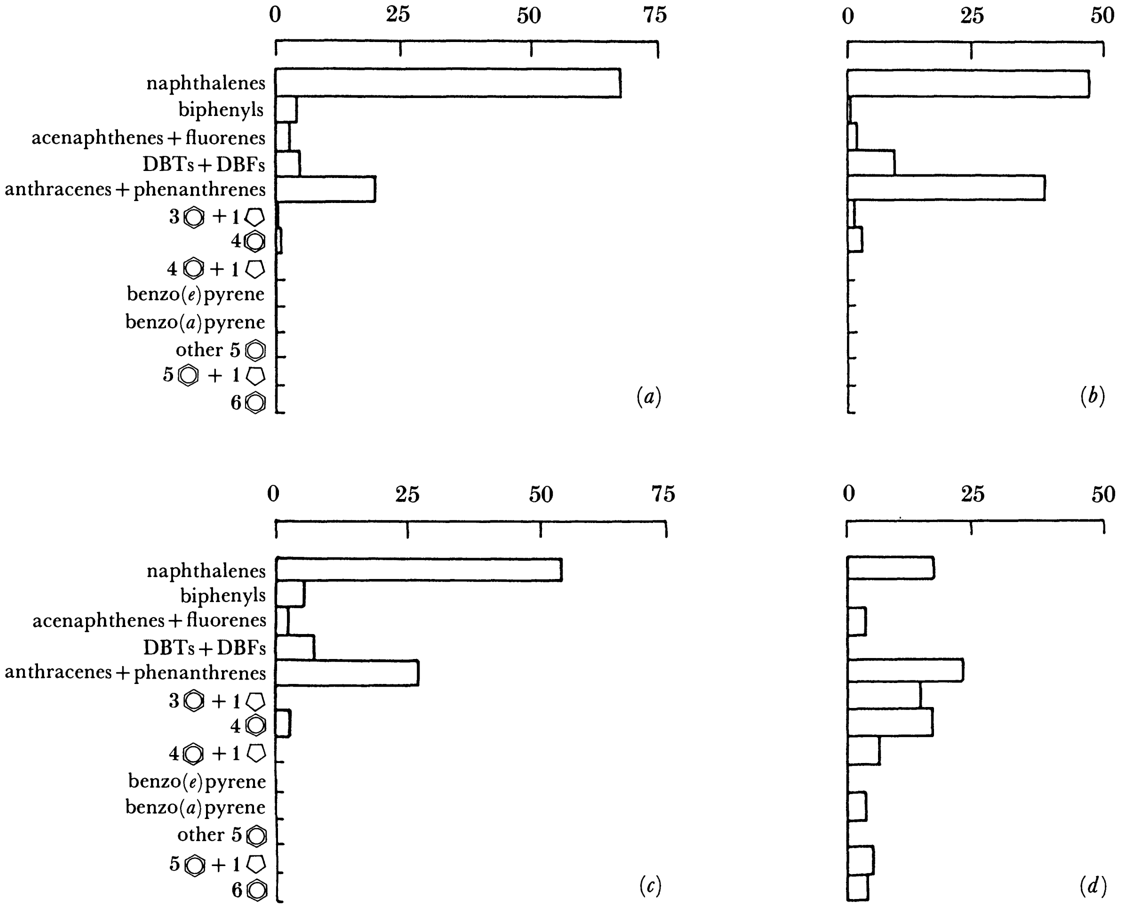

FIGURE 7. Percentage composition of polycyclic aromatics in diesel (*a*, *b*) and low toxicity 1 tanks (*c*, *d*) at 7 d (*a*, *c*) and 400 d (*b*, *d*) after cuttings addition. Total aromatics content (in $\mu g\ g^{-1}$ dry mass): (*a*), 6.75; (*b*) 2.14; (*c*) 0.90; (*d*) 0.53.

A substantial decline in nematode abundance was observed in all cuttings treatments in the first month after application (figure 10). The steep decline continued until numbers had fallen in all tanks to approximately one quarter of pretreatment levels, three months after application. This decline continued at a lower rate in diesel and low toxicity 4 treatments during the remainder of the experiment. By contrast, the other low toxicity treatments, particularly low toxicities 1 and 3, showed considerable fluctuations in nematode numbers with the mean abundance in the former exceeding that of controls after 11 months. Compared with treatments, controls maintained a significantly greater, though declining, mean abundance throughout the experiment. Although this decline may be associated with ecological factors inherent in tank isolation experiments, it is probable that the effect reflects the natural abundance cycle of meiofauna in this region (figure 10, adapted from McIntyre & Murison (1973)).

Mesobenthic (interstitial) copepods were eliminated in all treated tanks within 100 d of additions, and controls exhibited a general decline in abundance over much of the experimental period (figure 11). By contrast epi- and endobenthic forms showed a more varied response (figure 12). All low-toxicity treatments exhibited erratic fluctuations reflecting substantially

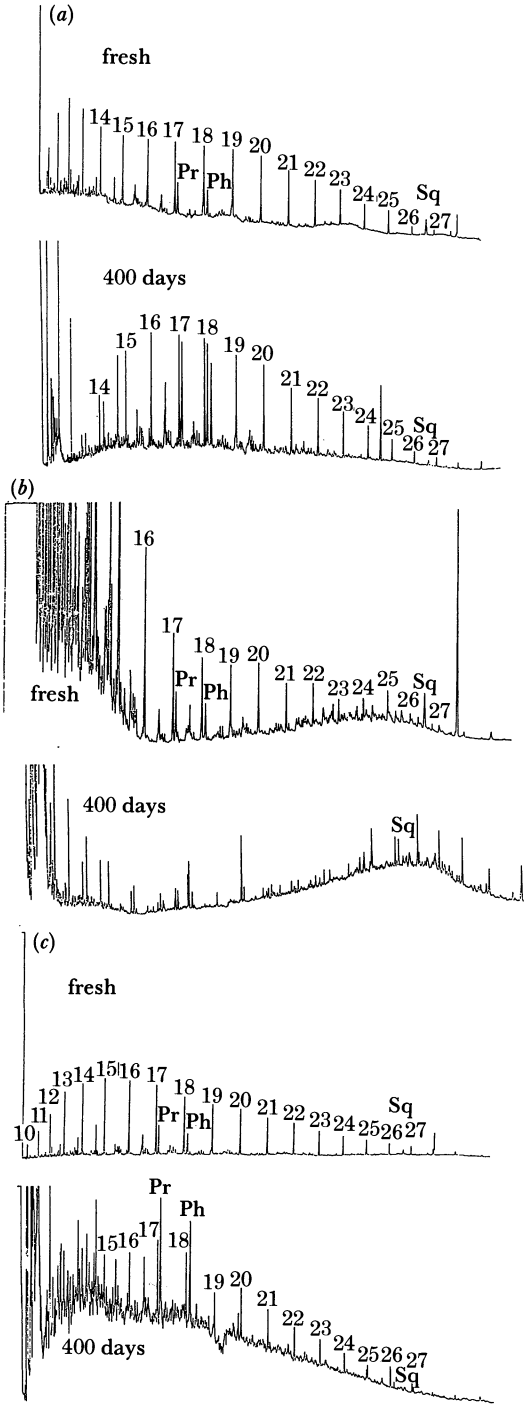

FIGURE 8. (*a*). GC–FID trace of aliphatics from fresh diesel-cuttings and sediment 400 d after cuttings addition. Squalane (Sq) in fresh-cuttings = 0.22 mg g^{-1}; squalane in sediment = 55.2 ng g^{-1} dry mass. (*b*) GC–FID trace of aliphatics from fresh low toxicity 1 cuttings and sediment 400 d after cuttings addition. Squalane (Sq) in fresh cuttings = 0.17 mg g^{-1}; squalane in sediment = 61.6 ng g^{-1} dry mass. (*c*) GC–FID trace of aliphatics from fresh low toxicity 4 cuttings and sediment 400 d after cuttings addition. Squalane (Sq) in fresh cuttings = 17.9 mg g^{-1}; squalane in sediment = 57.1 ng g^{-1} dry mass.

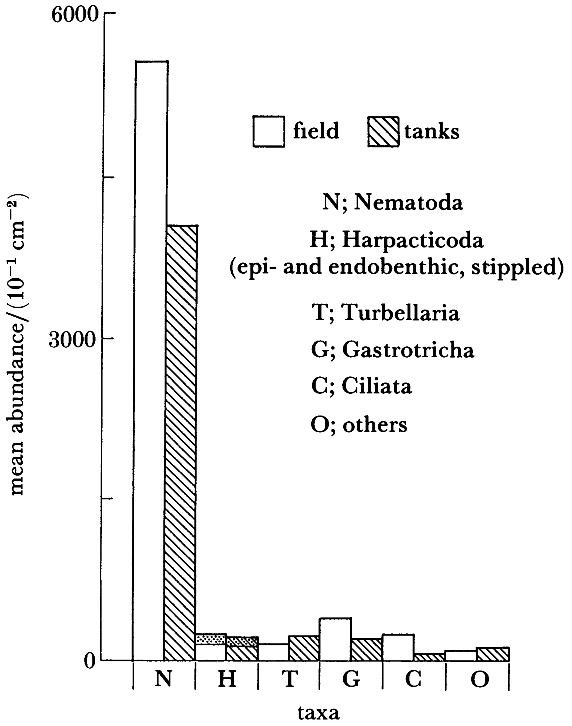

FIGURE 9. Comparison of mean abundances of field meiofauna with pretreatment tank meiofauna.

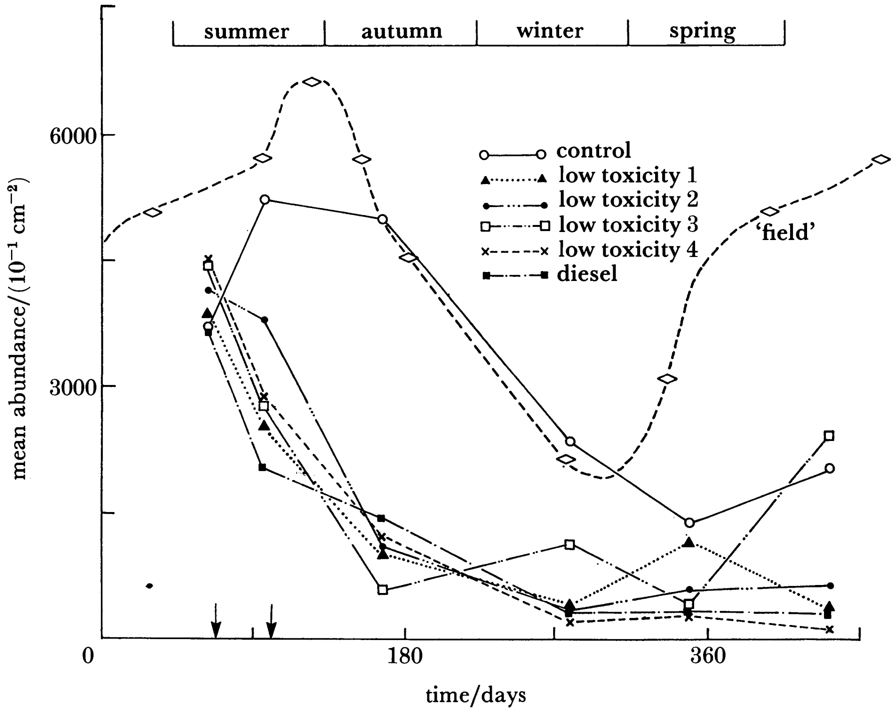

FIGURE 10. Nematode abundance in control and treated tanks over the period of the experiment. Also shown is the local field abundance cycle. Arrows indicate times of treatments.

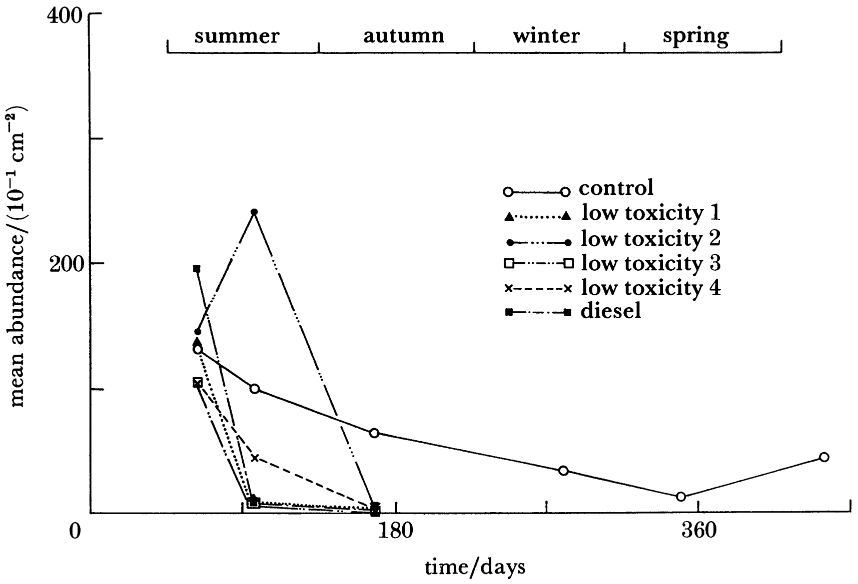

FIGURE 11. Interstitial (mesobenthic) copepod abundances in control and treated tanks over the duration of the experiment.

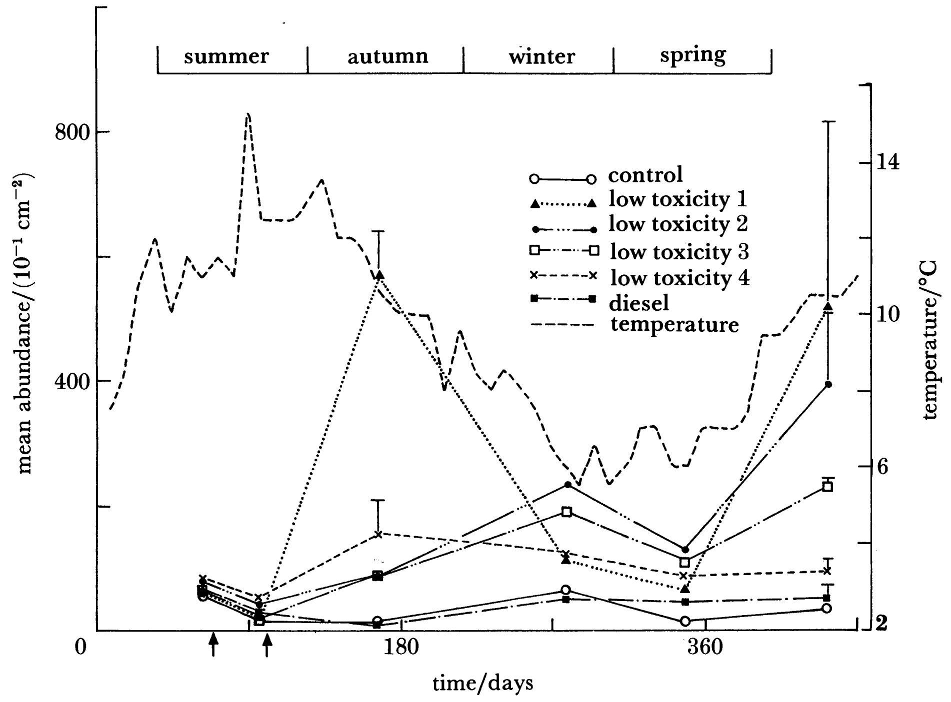

FIGURE 12. Epi- and endobenthic copepod abundances in control and treated tanks over the period of the experiment. Arrows indicate times of treatments.

enhanced populations at certain times during the experiment. Peak abundance (576 individuals per 10 cm^2) was recorded in low toxicity 1 in October, three months after cuttings application, with evidence of a similar effect the following June when a mean density of 517 individuals per 10 cm^2 was recorded in this treatment. Numbers of epi- and endobenthic copepods in control and diesel-cuttings tanks remained relatively small and constant throughout the experiment (13–70 individuals per 10 cm^2).

Although highly significant meiofaunal effects were indicated from statistical analyses no clear comparisons between cuttings could be shown because of complex time and treatment interactions.

The macrofauna of the sediments has not yet been analysed because the tanks are still being monitored, but counts of *Tellina* coming to the surface in the seven days immediately after cuttings addition (figure 13) indicated that diesel-treated tanks had the greatest rates of evacuation, followed by low toxicity 4 and then the other cuttings treatments. No *Tellina* were observed exposed on the sediment surfaces of control tanks during this period.

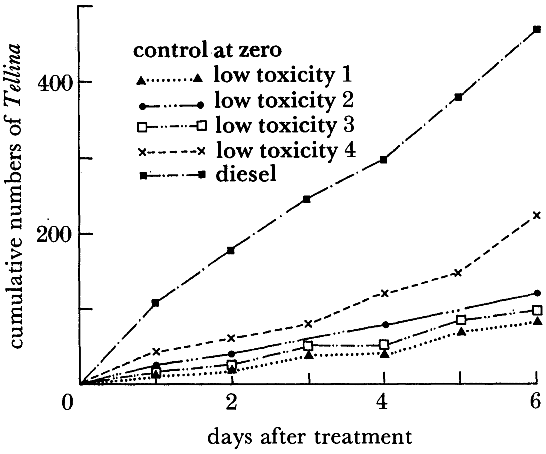

FIGURE 13. *Tellina* mortality in the days immediately after cuttings addition.

DISCUSSION

The tank system appears, from consideration of sediment physico-chemistry and meiofaunal populations, representative of the original field situation. Redox profiles taken immediately before cuttings addition (day 65, i.e. 65 d after collection of sediment) showed oxidized sediments in all tanks, similar to profiles recorded from the sediment collection area. Pretreatment tank sulphide levels, as in natural sediments of this type, were universally low (less than 10 p.p.b.† by mass dry drilling mud). Meiofaunal numbers, although 25% lower than field populations, showed unaltered high diversity, with nematodes accounting for 83% of the population in the tanks compared with 84% in field samples after 63 d.

The rapidity of the change in redox profiles immediately after cuttings addition argues against organic enrichment as the main agent. Dow (1984) noted an equally swift response in sediments experimentally exposed to oiled cuttings in winter when microbial activity was at a minimum. The effects were more likely to have been caused by smothering of the sediment and severe limitation of water and gas exchange because, for equal masses of cuttings and

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† In this paper one billion is used to represent 10^9.

despite differing oil levels, the magnitude and rate of change of redox in all cuttings treated tanks was essentially the same.

However, the changes in sediment soluble sulphide levels did appear to be associated with organic enrichment. Soluble sulphide in marine sediments is overwhelmingly the product of bacterial sulphate reduction, the activity of which reflects the amount and biological lability of organic matter available for decomposition (Goldhaber & Kaplan 1975). Sulphide levels in cuttings-treated tanks reached a maximum three months after application, but rose only very slightly in the first two months, before increasing rapidly between two and three months. Levels then dropped rapidly to a constant level of about ten times that of controls. This pattern is consistent with the observations of Westrich & Berner (1984) that organic material can be divided into two classes: easily degradable material capable of supporting high levels of sulphate reduction, and more refractory material able to support only small levels of sulphate reduction. The tank with the greatest sulphide levels, low toxicity 4, also contained the greatest total oil and *n*-alkane concentrations. Diesel-oiled cuttings-treated tanks generated the smallest sulphide concentrations and contained the smallest alkane levels but not the smallest total oil levels. *n*-Alkanes are known to be more readily degraded than aromatics (Delaune *et al.* 1980) and thus appeared to act as substrates for rapid growth of the sulphate reducers once conditions were appropriate.

The nematode abundance fell rapidly in all cuttings-treated tanks immediately after application. This was most likely to have been a response to the change in the physico-chemical status of the sediments because the decrease in these organisms paralleled the change in redox profiles more closely than the other parameters measured. Numbers continued to fall at a reduced rate in diesel and low toxicity 4 tanks but fluctuated in other treated tanks, sometimes to levels greater than those in controls. In isolated experimental systems, such as the tanks described here, physico-chemical recovery may not be associated with the corresponding degree of biological recovery because of the constraints on recruitment. Thus the increasing fluctuations in nematode numbers observed in the low toxicity tanks may be indicative of potential for recovery, true recovery being inhibited by lack of available recolonizers, and may represent single species at peak reproductive times.

Numbers of epi- and endobenthic copepods fluctuated erratically in all treatments except controls and diesel, which remained relatively constant throughout the experiment. The greatest fluctuations occurred in the low toxicity 1 treatment (that with the smallest oil concentration). Similarly elevated densities of benthic harpacticoid copepods have been observed in organically enriched field and experimental environments such as in an accumulating sewage sludge dumping ground (Moore & Pearson 1986); in oil sediments near a North Sea platform (Moore *et al.*, this symposium); in fuel-oil treated MERL mesocosms (Grassle *et al.* 1981); and in organically enriched (powdered *Ascophyllum nodosum*) mesocosms (Gee *et al.* 1985). Physically disturbed sediments have also been shown to induce an opportunistic response from an epibenthic copepod species (Alongi 1985). By contrast, the opportunistic response of these copepods does not seem to have occurred at all in the diesel treatment or to such an extent in the low toxicity 4 treatment. The lack of a response in the diesel treatment may be due to the relatively high concentrations of PAH (as shown by the GC–MS data) which are primarily responsible for the toxic effects of diesel (Anderson *et al.* 1974). Similarly, the reduced response in the low toxicity 4 treatment may also be due to the high PAH levels incurred by the large oil concentration in this treatment. Investigations into the effects of natural crude-oil seepage in the Santa Barbara Channel have indicated that the meiofaunal community may act as an important intermediate in the transfer of energy from microbial hydrocarbon degrading complexes to higher trophic levels (Montagna & Spies 1985). Dow (1984) noted increased numbers of aerobic heterotrophic bacteria in tanks dosed with comparable levels of diesel and low-toxicity oiled cuttings, but did not see an increase in glucose or amino acid utilization. Naphthalene mineralization rates were greater, showing that the larger numbers of bacteria were using the oil as a food source. The greatest copepod abundances occurred in the low toxicity 1 treatment, the treatment with the least PAH levels and the least total oil levels. The low abundance and suppressed seasonal response of epi- and endobenthic copepods in the control treatment compared with the normal field situation may be due to a potential food limitation imposed by the artificial water-exchange system. These results suggest a ‘playoff’ between organic enrichment and toxicity in the effects of oiled cuttings on epi- and endobenthic copepods.

The change in the ratio of nematodes to copepods under different pollution loadings has received much attention recently (see, for example, Boucher 1985; Moore & Pearson 1986; Sandulli 1987). The technique was originally suggested for monitoring pollution on coarse-particle (bathing) beaches, but has frequently been extended to silty sublittoral sediments. The copepod fauna of sublittoral muds is quite unlike that of sandy beaches and the effects of organic loading on sediment chemistry and the nematode fauna are also likely to be substantially different. Not too surprisingly, the responses of nematodes and copepods to pollution differ markedly in the two habitats. In common with other sublittoral pollution studies, the nematode/epibenthic copepod ratio declined in cuttings treatments; nematodes decreased and epibenthic copepods were either unaffected or increased (table 1). As expected, the ratio of nematodes to mesopsammic (interstitial) copepods increased under stress, these copepods being more vulnerable to smothering effects of the cuttings. There were no discernable between-treatment differences in the behaviour of these ratios.

TABLE 1. RATIOS OF NEMATODES TO MESOPSAMMIC COPEPODS IN CUTTINGS TREATED AND CONTROL TANKS OVER THE DURATION OF THE EXPERIMENT

treatment time		control	low toxicity 1	low toxicity 2	low toxicity 3	low toxicity 4	diesel
day 63	E	67.6	72.0	53.6	77.5	54.2	52.6
	M	28.6	28.6	29.0	43.2	43.8	18.8
day 96	E	377.1	141.2	82.9	139.5	52.9	79.4
	M	53.9	363.1	15.8	348.4	67.7	229.3
day 167	E	388.2	1.8	12.3	6.6	7.6	143.0
	M	81.4	—	—	—	—	—
day 279	E	36.8	3.0	1.5	5.8	1.8	6.6
	M	73.6	—	—	—	—	—
day 349	E	87.8	16.9	4.5	3.3	3.3	7.2
	M	140.4	—	—	—	—	—
day 433	E	56.9	0.7	1.6	10.3	1.3	5.7
	M	48.8	—	—	—	—	—

field ratio: epi- and endobenthic (E) 70.5, mesopsammic (M) 40.1

The diesel-oiled cuttings appeared more toxic to the macrofauna in the tanks during the seven days after addition. However, no data are available on macrofaunal populations immediately before addition, although the original field-site has been sampled. It will be interesting to examine the macrofaunal populations when the experiment has run its course.

CONCLUSIONS

The results presented here would seem to support the contention that diesel-oiled cuttings differ from low toxicity cuttings in their long-term impact. The higher short-term acute toxicity of diesel was apparent from counts of macrofauna evacuating sediments. Greater long-term toxicity may be a factor in the inability of diesel treatments to support large populations of epi- and endobenthic copepods despite having similar total oil levels to low toxicity treatments which supported elevated copepod abundances.

Effects on the physico-chemistry of the sediments appeared to be twofold: an initial smothering effect dependent more on the total quantity of cuttings than their oil contents, and an organic enrichment effect dependent on the biological lability and quantity of oil. Organic enrichment, although contributing to high toxic-sulphide levels, seems to be a relatively short-term effect compared with the long-term toxicity of certain hydrocarbons. On balance, therefore, it is likely that the discharge of low-toxicity oiled cuttings during drilling operations will have a less toxic long- and short-term impact than the discharge of equivalent amounts of diesel-oiled cuttings.

REFERENCES

Addy, J. M., Hartley, J. P. & Tibbetts, P. J. C. 1984 Ecological effects of low toxicity oil-based mud drilling in the Beatrice oilfield. *Mar. Pollut. Bull.* **15**, 429–436.

Alongi, D. M. 1985 Effect of physical disturbance on population dynamics and trophic interactions among microbes and meiofauna. *J. mar. Res.* **43**, 351–364.

Anderson, J. W., Neff, J. M., Cox, B. A., Tatem, H. E. & Hightower, G. M. 1974 Characteristics of dispersions and water-soluble extracts of crude and refined oils and their toxicity to estuarine crustaceans and fish. *Mar. Biol.* **27**, 75–85.

Blackman, R. A. A., Fileman, T. W. & Law, R. J. 1982 Oil-based drill-muds in the North Sea – the use of alternative base-oils. ICES CM: 1982/E: 13. (Unpublished manuscript.)

Boucher, G. 1985 Long term monitoring of meiofaunal densities after the *Amoco Cadiz* oil spill. *Mar. Pollut. Bull.* **16**, 328–333.

Davies, J. M., Addy, J. M., Blackman, R. A., Blanchard, J. R., Ferbrache, J. E., Moore, D. C., Somerville, H. J., Whitehead, A. & Wilkinson, T. 1984 Environmental effects of the use of oil-based drilling muds in the North Sea. *Mar. Pollut. Bull.* **15**, 363–370.

Delaune, R. D., Hambrick, G. A. & Patrick, W. H. 1980 Degradation of hydrocarbons in oxidised and reduced sediments. *Mar. Pollut. Bull.* **11**, 103–106.

Dow, F. K. 1984 Studies on the environmental effects of production water and drill cuttings from North Sea oil installations. Ph.D. thesis, University of Aberdeen.

Gee, J. M., Warwick, R. M., Schaanning, M., Berge, J. A. & Ambrose, W. G. 1985 Effects of organic enrichment on meiofaunal abundance and community structure in sublittoral soft sediments. *J. exp. mar. Biol. Ecol.* **91**, 247–262.

Goldhaber, M. B. & Kaplan, I. R. 1975 Controls and consequences of sulphate reduction in recent marine sediments. *Soil Sci.* **119**, 42–55.

Grahl-Nielson, O., Sunby, S., Westrheim, K. & Wilhelmsen, S. 1980 Petroleum hydrocarbons in sediment resulting from drilling discharges from a production platform in the North Sea. In *Proceedings of the Symposium on Research on Environmental Fate and Effects of Drilling Fluids and Cuttings, Lake Buena Vista, Florida, 21–24 January* 1980, vol. 1, pp. 541–561. Washington, D.C.: U.S. Government Printing Office.

Grassle, J. F., Elmgren, R. & Grassle, J. P. 1981 Response of benthic communities in MERL experimental ecosystems to low-level chronic additions of No. 2 fuel oil. *Mar. environ. Res.* **4**, 279–297.

Mackie, P. R., Hardy, R. & Whittle, K. J. 1978 Preliminary assessment of the presence of oil in the ecosystem at Ekofisk after the blowout, 22–30 April 1977. *J. Fish. Res. Bd Can.* **35**, 544–551.

Massie, L. C., Ward, A. P., Davies, J. M. & Mackie, P. R. 1985 The effects of oil exploration and production in the northern North Sea. Part 1. The levels of hydrocarbons in water and sediments in selected areas, 1978–1981. *Mar. environ. Res.* **15**, 165–213.

McIntosh, A. D., Massie, L. C. & Mackie, P. R. 1983 A survey of hydrocarbon levels and some biodegradation rates in water and sediments around North Sea oil platforms, 1981, 1982. ICES CM: 1983/E: 42. (Unpublished manuscript.)

McIntyre, A. D. & Murison, D. J. 1973 The meiofauna of a flatfish nursery ground. *J. mar. biol. Ass. U.K.* **53**, 93–118.

Montagna, P. A. & Spies, R. B. 1985 Meiofauna and chlorophyll associated with *Beggiatoa* mats of a natural submarine oil seep. *Mar. environ. Res.* **16**, 231–242.

Moore, C. G. & Pearson, T. H. 1986 Response of a marine benthic copepod assemblage to organic enrichment. In *Proceedings of the Second International Conference on Copepoda, Ottawa, Canada, 13–17 August 1984.* (ed. G. Schriever, H. K. Schminke & C.-t. Shih), National Museum of Natural Sciences, Ottawa. *Syllogeus* no. 58, 369–373.

Pearson, T. H. & Stanley, S. O. 1979 Comparative measurement of the redox potential of marine sediments as a rapid means of assessing the effect of organic pollution. *Mar. Biol.* **53**, 371–379.

Sandulli, R. 1987 Preliminary report of a laboratory experiment on the effects of organic pollution on meiofauna. *Acta XVII Congresso S.I.B.M. Ferrara, 1985* in *Nova Thalassia.* (In the press.)

Stanley, S. O., Leftley, J. W., Lightfoot, A., Robertson, N., Stanley, I. M. & Vance, I. 1981 The Loch Eil project: sediment chemistry, sedimentation and the chemistry of the overlying water in Loch Eil. *J. exp. mar. Biol. Ecol.* **55**, 299–313.

Westrich, J. T. & Berner, R. A. 1984 The role of sedimentary organic matter in bacterial sulphate reduction: the G model tested. *Limnol. Oceanogr.* **29**, 236–249.

Discussion

W. A. Hamilton (*Department of Microbiology, University of Aberdeen, U.K.*). Dr Davies puts considerable stress on redox as a component of his test system. Would he care to comment on the fact that whereas he was recording values of around −200 mV in the Beryl field, in his tanks all values were above zero?

J. M. Davies. I suggest that whereas the gross environment might be of positive redox, there would none the less be microenvironments that would be of lower redox and that these would be the site of sulphide production.

W. A. Hamilton. I agree with Dr Davies's suggestion regarding microenvironments and sulphide production, but this limitation of probe measurements would apply equally to all systems studied and therefore the point still holds that the tank model system did not truly mirror the conditions under cutting piles on the seabed.

J. M. Davies. The experiment was not designed to simulate the conditions under the cuttings pile in the immediate vicinity of the platform but rather the conditions 400–500 m from the platform where oil concentrations in the top 2 cm of the sediment are of the order of 500 p.p.m. by mass dry sediment rather than 150000 p.p.m. by mass dry sediment in the cuttings pile.

R. A. A. Blackman (*MAFF Fisheries Laboratory, Burnham-on-Crouch, U.K.*). Could Dr Davies say what role *Tellina* played in the economy of the tanks? Was it a competitor for food, making resources available through faeces and pseudo-faeces, reworking surface sediments, etc.? (It was assumed that the graph displayed showed only the mortality rates for *Tellina* surfacing in the tanks, but that some remained alive in the sediments.)

J. M. Davies. *Tellina* are filter feeders, predominantly filtering plankton from the water column, although they have been shown to remove sediment detritus in the immediate vicinity of their siphons.

The total numbers of *Tellina* remaining in the tanks will not be determined until the end of the experiment when the tanks are emptied.