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A Review of Sediment Toxicity Bioassays Using the Amphipods and Polychaetes

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Abstract

Several bioassay methods have been developed since the US EPA/COE (United States Environmental Protection Agency/ Army Corps of Engineers) testing protocol was devised, involving a great variety of test species. The amphipods and the polychates are now beginning to be used routinely as standard bioassay organisms for assessing the toxicity of marine sediments for regulatory purposes. The present review has confirmed the potential of both the amphipods and the polychaetes for sediments toxicity bioassays.

Key Words: Marine pollution, heavy metal, reburial, emergence, LC_{50}, EC_{50}

Introduction

Marine pollution may be defined as:

'... the introduction by man, directly or indirectly, of substances or energy to the marine environment resulting in such deleterious effect as harm to living resources; hazards to human health; hindrance of marine activities including fishing; impairment of the quality for use of seawater; and reduction of amenities' (Clark, 1986).

Most marine pollution is caused by domestic wastes, industrial wastes, oil wastes, pesticides, insecticides, radioactive wastes and metals (Phillips and Rainbow, 1994). Cairns and Mount (1990) noted that over 9 million chemicals are listed in the Chemical Abstract Service's Registry of Chemicals, although only an estimated 76,000 are in daily use. Especially coastal and oceanic waters are increasingly affected by such pollutants (Bryan, 1984), one of the most important of which are metals (Phillips, 1980; Bat *et al.,* 1998-1999a). Nieboer and Richardson (1980) proposed the separation of such metals into three classes: A, borderline, and B, and this classification has been accepted by most authors (Depledge *et al*., 1994; Phillips and Rainbow, 1994; Phillips, 1995). Class A metal ions (*e.g.* all macro-nutrient metals such as Ca, Mg, K, Na) are essentially oxygen-seeking, while those Class B metal ions (*e.g.* Cu, Hg, Ag) seek out nitrogen or sulphur atoms; the Borderline metal ions (*e.g.* micro-nutrient metals such as Zn, Cd, Fe, Co, Ni) show intermediate behaviour (Nieboer and Richardson, 1980). Many metals are essential to organisms such that in their absence an organism can neither grow nor reproduce (Underwood, 1977). Major ions such as sodium, potassium, calcium and magnesium are essential to sustain life, whilst others are normally only present in trace amounts (<0.01% of the mass of the organism; Förstner and Wittmann, 1983). Essential life processes or molecules requiring metals include: (a) the respiratory pigment haemoglobin, found in vertebrates and many invertebrates and which contains iron; (b) the respiratory pigment of many molluscs and higher crustaceans, haemocyanin, which contains copper; (c) the respiratory pigment of tunicates which contains vanadium; (d) many enzymes contain zinc; and (e) vitamin B_{12} enzymes contain cobalt (Clark, 1986). Lists of essential metals vary from author to author but all include iron, magnesium, manganese, cobalt, zinc, copper (Viarengo, 1985) and Rainbow (1988) includes arsenic, chromium, molybdenum, nickel, selenium, tin and vanadium. All metals are taken up by aquatic organisms from solution and from food or particles (Rainbow, 1990; Rainbow and Phillips, 1993), and can be accumulated at high concentrations (Rainbow, 1988) when, whether essential or not, they may be potentially toxic to living organisms (Bryan, 1976b; Rainbow, 1985, 1993, 1995; Rainbow *et al*., 1990).

Sources of metals in the marine environment

Heavy metals found in sea water (Rainbow, 1993) are continuously released into the marine environment by both natural and artificial processes (Bryan, 1976a,b). The natural sources of metals in sea are reviewed by Turekian (1971) and categorised by Bryan (1976b) as follows: (a) Coastal supply, which includes input from rivers and from erosion due to wave action and glaciers; (b) deep sea supply, which includes metals released from particles or sediments by chemical processes; (c) supply which by-passes the near-shore environment and includes metals transported in the atmosphere as dust particles or as aerosols and also material which is produced by glacial erosion in polar regions and is transported by floating ice.

Anthropogenic sources of metals include: (a) atmospheric input from the burning of fossil fuels, the smelting and refining of metals, the use of leaded petrol in motor vehicles, fly ash from power stations and the use of seawater discharges cooling from operations at power stations. For some metals, inputs to the atmosphere as a result of human activities are greater than natural inputs and the sea acts as a sink for atmospheric contamination (Clark, 1986); (b) mining activities, such as tailings; (c) industrial processing of ores and the use of metal components, such as electroplating, pigments, electrical wiring, batteries, galvanising, fertilisers; (d) the release of sewage (Depledge *et al*., 1994), which was dumped at sea in considerable quantities by Britain and it has a high organic content with heavy metals (Clark, 1986); (e) contamination from ships in docks and harbours from the use of metals such as copper, tin and mercury in antifouling points and other metals such as lead, chromium and zinc in preservative paints (Bellinger and Benham, 1978; Young *et al*., 1979); (f) dredging spoil, particularly from industrialised estuaries may contain heavy metals and other contaminants which are then transferred to the dumping grounds (Clark, 1986).

Metals in Sediments

When introduced into the sea, organic and inorganic contaminants, particularly heavy metals, eventually accumulate in sediment (Luoma, 1983; Salomons *et al*., 1987; Tessier and Campbell, 1987) which become repositories or sinks (Waldichuk, 1985; Phillips, 1995). Sediments are also an ecologically important component of the aquatic environment and may play an important role in mediating the exchange of contaminants between particulate, dissolved and biological phases (Reynoldson and Day, 1993). Estuarine sediments are the major compartment in the coastal environment for heavy metals and other toxic materials by virtue of their small particle size (Davies-Colley *et al*., 1984) and contain variable concentrations of both essential and non-essential metals (Luoma and Bryan, 1978). Because of increasing industrial and recreational demands on coastal areas, especially estuarine environments, these systems have come under ever-increasing stress with resultant habitat deterioration and pollution leading to deleterious effects on benthic and pelagic communities, fisheries and eventually to human health through direct contact of organisms with the sediment or by resuspension of contaminated particles into the overlying water.

Estuaries are important habitats for wildlife and have historically been used as a source of food for transport and for disposing of waste material (McLusky, 1981). Many organisms live in or on estuarine sediments, including several economically important species and species involved in food chains terminating in shorebirds and fish of conservation significance (Adams *et al*., 1992). The protection of an estuarine or marine habitat from damage due to contaminants requires an understanding of both the sensitivity of invertebrates to contaminants and their ecological requirements. Toxicity tests are a convenient and appropriate way of accessing this sensitivity and also have the advantage of reflecting the bioavailable fraction of a contaminant, which can be very different from the total amount determined by chemical analysis (Hill *et al*., 1993).

Sediment Toxicity Tests

Historically, the evaluation of contaminant effects has emphasised surface waters rather than sediments (Ingersoll, 1995). For example, Standard Methods for the Examination of Water and Wastewater (1976) include a coverage of the general terminology and procedures for performing bioassays. Tentative procedures for undertaking amphipod bioassays appeared for the first time in the 14th edition (1976) although only freshwater amphipods (gammarids) were recommended. Marine polychaete annelids including *Neanthes arenaceodentata*, *N. succinea*, *N. virens*, *Capitella capitata* and *Ophryotrocha* sp. were also recommended for the characterisation of water toxicity. Sediment toxicity testing began in late 1970s (Burton, 1991), but the science of sediment toxicity is still very young (Burton and Scott, 1992; Ingersoll, 1995) and there were no standard methods for conducting sediment toxicity tests until the early 1990s (Burton and Scott, 1992). Even so, no completely standardised methodology has been published (Luoma and Ho, 1993), despite the advantages of these techniques for providing information on the ecological impact of contaminated sediment (Chapman and Long, 1983; Chapman, 1989; Long and Chapman, 1985; Bat and Raffaelli, 1998a; 1998b).

Sediment toxicity may be defined as : ‘the ecological and biological changes that are caused by contaminated sediments’ or ‘an adverse response observed in a test organism exposed to a contaminated sediment’ (Luoma and Ho, 1993).

According to Chapman (1989), sediment bioassays can be used in two separate ways to develop sediment quality criteria: (a) sediment bioassay and chemical analyses can be conducted with sediments collected from contaminated and reference areas. The bioassay responses can be compared quantitatively to identify whether problems exist and the levels of contaminants in sediments can be related to the bioassay responses; (b) dose-response relationships can be developed in the laboratory by spiking sediments with individual and mixed contaminants and then carrying out bioassays on these sediments. A variety of test methods have been developed by the American Public Health Association (APHA), the American Society for Testing and Materials (ASTM), the U.S. Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers of Materials.

In developing a marine estuarine sediment bioassay protocol, a number of properties are desirable (DeWitt *et al*., 1989; Smith and Logan, 1993): (a) broad salinity tolerance; (b) high sensitivity to common sediment contaminants; (c) high survival rate under control conditions; (d) occupation of microhabitat(s) at or preferably, below the sediment-water interface to ensure maximum and consistent exposure to sediment contaminants; (e) low sensitivity to natural sediment variables, such as particle size and organic content, to allow a wide variety of sediment types to be tested; (f) broad geographic range to enhance the breadth of its application as a test species; (g) ease of collection, handling and maintenance in the laboratory; (h) ecological importance in estuarine systems; and (i) the ability to be cultured or year-round availability from the field. Ideally, a sediment toxicity test should also be rapid, simple and inexpensive (Luoma and Ho, 1993; Ingersoll, 1995; Bat *et al.,* 1998-1999b).

Only relatively few species have been extensively used for toxicity testing (Cairns and Mount, 1990) and there is no single biological response or test species that can meet all environmental and legislative requirements for effective toxicity testing (Widdows, 1993; Ingersoll, 1995; Rand *et al*., 1995). Nevertheless, benthic invertebrates have great potential for sediment toxicity tests (Reynoldson and Day, 1993), because they are intimately associated with sediments either through their burrowing activity or by ingestion of sediment particles (Luoma, 1983; Reynoldson, 1987; Bryan and Langston, 1992; Reynoldson and Day, 1993; Bat and Raffaelli, 1996; Bat, 1998).

Amphipods have proved especially useful and are commonly employed in sediment toxicity tests (Luoma and Ho, 1993; Bat *et al., 19*96) because of their high sensitivity (Swartz *et al*., 1982, 1985a, b) and because their population densities are known to decline along pollution gradients in the field (Bellan-Santini, 1980). One of the first bioassays for testing the toxicity of dredged material confirmed the high sensitivity of the infaunal amphipod *Paraphoxus epistomus* compared to other infaunal non-amphipod species *Protothaca staminea*, *Macoma inquinata*, *Glycinde picta* and Cumacea (Swartz *et al*., 1979).

Many amphipods, such as *Corophium salmonins*, *C. spinicorne* (ASTM, 1990), *Gammarus fasciatus, G. pulex, G. lacustris, Crangonyx gracillus,* and *Pentoperia hoyi* (Arthur, 1980; Burton, 1991) have been used or recommended for bioassays, sediments contaminated with heavy metals (Table 1). Other species, such as *Gammarus lacusta, G. duebeni, Echinogammarus pirloti, Stegocephaloides christianiensis, Hyperia galba, Hyale nilssoni, Talitrus saltator, Talorchestica deshayesii, Arcitalitrus dorrieni, Orchhestia cavimana* and particularly *O. mediterranea* have also been used extensively in the UK as coastal biomonitors of heavy metals (Rainbow and Moore, 1986; Moore and Rainbow, 1987; Rainbow *et al*., 1989; Weeks and Moore, 1991), but it is not appropriate to use them for sediment bioassays, because of their different (*e.g.* rocky) habitat.

Effects of the metals included the following: decreased survival, increased emergence from sediment, decreased burrowing or feeding activity and loss of ability to re-bury. Effects on uptake or depuration of metals were influenced by the presence of other metals, duration of exposure, metal concentrations, age (juvenile or adult), temperature and salinity. Many authors have investigated sublethal effects of exposing organisms to heavy metals, especially the effects on growth and the accumulation of metals in tissues. Some studies also showed that amphipods were the most sensitive taxon compared to crustaceans, mollusc and polychaetes. Because of the lack of a standard bioassay protocol, it would be unwise to compare the bioassay results from the different studies.

Polychaetes are also frequently employed in sediment toxicity tests (Luoma and Ho, 1993; Ingersoll, 1995; Table 2). Species used to date include *Cirriformia spirabrancha* (Milanovich *et al*., 1976), *Neanthes arenaceodentata* (Pesch and Morgan, 1978; Pesch, 1979; Pesch and Hoffman, 1983; Dillon *et al*., 1993), *Glycinde picta* (Swartz *et al*., 1979), *Crenodrilus serratus* (Reish, 1980), *Arenicola cristata* (Schoor and Newman, 1976; Rubinstein, 1979; Rubinstein *et al*., 1980; Walsh *et al*., 1986), *Nereis virens*, *Glycera dibranchiata* and *Nephtys caeca* (Olla *et al*., 1988), *Dynophilus gyrociliatus* (Åkesson, 1980; Long *et al*., 1990), *Ophryotrocha labronica*, *O. diadema* (Åkesson, 1980), *Streblospio benedicti* (Cheng *et al*., 1993) and *Hediste diversicolor* (Bat *et al*., 2001).

At present, organisations such as ASTM and the U.S. EPA are currently developing sediment bioassay protocols for selected species, including marine and estuarine amphipods and polychaetes (Ingersoll, 1995). In Europe, organisations such as the UK Ministry of Agriculture Fisheries and Food (MAFF), the Paris Commission (PARCOM), the Society of Environmental Toxicology and Chemistry (SETAC) and the Water Research Centre (WRc) are also developing test methods for selected species. In 1990 and 1992, consideration was given to the development of a whole sediment bioassay that could be used by MAFF for ship-board monitoring of sediment quality (Thain *et al*., 1994). A Paris Commission (PARCOM) sediment reworker ring-test for testing of chemicals used in the offshore oil industry using the polychaete worm *Nereis virens*, the bivalve *Abra alba* and the sea urchin *Echinocardium cordatum* was inconclusive, suggesting that none of these organisms might be suitable (Thain *et al*., 1994). *Nereis virens*, for example, was found to be a robust organism and generally insensitive to contaminants (Thain *et al*., 1994). These authors also suggested that the oyster embryo bioassay was not suitable for sediment testing but both the amphipod *Corophium volutator* and the polychaete *Arenicola marina* showed good potential for sediment quality monitoring programmes. A study supported by the European Commission and carried out by the Water Research Centre (WRc), Coastal and Marine Management (RIKZ, Netherlands), Institute for Inland Water Management (RIZA, Netherlands), Instituto Portugues de Investigacao (IPIMAR, Portugal), University of Utrecht (Netherlands) and University of Hamburg (Germany), also concluded that *Corophium volutator* had potential as a sediment bioassay organism, whereas the freshwater bivalve *Sphaerium corneum* and *Chronomus riparius* had too many disadvantages such as collection from field, transportation, laboratory maintenance and problems in culturing (Fleming *et al*., 1994). During the SETAC Workshop on sediment toxicity assessment, both *Corophium volutator* and *Arenicola marina* were recommended as test species for sediment bioassays (Hill *et al*., 1993; van den Hurk *et al*., 1992; Chapman, 1992; Chapman *et al*., 1992; Bat *et al.,* 1996; 1998; Bat and Raffaelli, 1998a; 1998b; 1999).

Several toxicity studies using *Corophium volutator* have been conducted since 1976 (Table 3). Eight of these (1-5, 22, 24, 27) administered toxicants via spiked sediment. Others used contaminated water with or without sediment (6-14, 26) but only two studies used also a choice experiment (1, 26). Four studies (15-16, 24, 26) measured the concentrations of heavy metals in animals and in sediment and laboratory bioassays with field samples were also conducted (17-21, 23, 25). Effects of toxicants included the following: decreased survival, reburial, increased emergence from sediment, immobilisation, and uptake of toxicants from seawater and/or sediment similar to that found for other amphipod species (Table 1). Several authors agree that a 10-day duration for a sediment bioassay is sufficient (Table 3: 18-21, 24).

For *Arenicola marina*, metal toxicity and sediment bioassay studies have mostly been done in the laboratory using radionucleids (Table 4: 1-5) and oils (9, 10, 17), respectively. Effects of toxicants on cast production of *Arenicola* have also been investigated (9-11, 14-16).

Clearly *Corophium volutator* and *Arenicola marina* have potential as test species for sediment bioassays in European waters. Not only do they respond to contaminated sediment, but they also fulfil many of criteria listed above (DeWitt *et al*., 1989; Smith and Logan, 1993; Bat and Raffaelli, 1998a). Because these organisms spend the majority of their life in the sediment, they are continuously exposed to contaminants and they ingest sediment (and contaminants) when feeding. They are usually available all the year round, often occur in high densities, tolerate a wide range of particle sizes and salinities and they have a broad geographic range. Both are important in food chains and probably play important roles in sediment community organisation.

There are clear advantages of the bioassays using both the amphipods and the polychaetes as a means of assessing sediment toxicity, and it is hoped that they will continue to be employed routinely in monitoring programmes for coastal waters.

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Table 1. Amphipod toxicology studies involving water and sediment exposures in laboratory and/or field bioassays

No	Species	Habitat[a]	Metal	Method[b]	Test duration	End point[c]	Temp. (°C)	Salinity (‰)	Results	Reference
1	*Allorchestes compressa*	SW	Cd, Zn	WAT, ST	96-120h	S	16.8-20.5	34.5	120h Cd LC_{50}= 0.2-4 ppm; 96h Zn LC_{50}= 0.58 ppm; this amphipod was more sensitive than crab, shrimp, mollusc and worm.	Ahsanullah, 1976
2	*Allorchestes compressa*	SW	Se	WAT, CF	96h	S	18	34.8-35.3	LC_{50}= 4.77 and 6.17 ppm from two different areas; juveniles were more sensitive than adults.	Ahsanullah and Palmer, 1980
3	*Allorchestes compressa*	SW	Cu	WAT, ST	96h	S	20	32±1	LC_{50} values for juveniles and adults were 0.11 and 0.50 ppm, respectively.	Ahsanullah and Florence,1984
4	*Allorchestes compressa*	SW	Zn, Cd, Cu	WAT, CF	96h	S	20.3±0.8	34.1±0.7	Cu was 1.6 times more toxic than Cd and 4 times more toxic than Zn; the toxicity of a combination of two and three metals is different from that of individual metals.	Ahsanullah *et al*., 1988
5	*Allorchestes compressa*	SW	Cd, Cr, Cu, Zn	WAT, CF	4wk	S, G, B	19±1	31±1	Cu was the most toxic metal, the second most toxic was Cd; the sublethal effects of the four metals appear to be in similar proportion to their lethal effects; Cd was accumulated without regulation until the amphipod died.	Ahsanullah and Williams, 1991
6	*Austrochiltonia subtenuis*	FW	Cd	WAT, ST	96h	S	15±1		96h LC_{50} = 0.04 ppm.	Thorp and Lake, 1974
7	*Chelura terebrans*	SW	Cd	WAT, ST	96h; 7 day	S	19.5	35	96h LC_{50} = 0.63 ppm and 7day LC_{50} = 0.2 ppm.	Hong and Reish, 1987
8	*Corophium insidiosum*	IN	Cd	WAT, ST	96h 7 day	S	19.5	35	96h LC_{50} = 1.27 ppm and 7day LC_{50} = 0.51 ppm.	Hong and Reish, 1987
9	*Corophium insidiosum*	IN	As, Cd, Cr, Cu, Pb, Hg, Zn	WAT, ST	96h - 20 days	S, A	19±1		96h LC_{50}s were 1.1, 0.68, 11, 0.6, >5, 0.02 and 1.9 ppm in order listed; the metal levels were <10, 23, 51.3, 3464, 832, 27.7 and 253 ppm dry wt in order listed.	Reish,1993
10	*Crangonyx pseudogracilis*	FW	Cd Cu, Cr, Pb, Hg, Mo, Ni, Sn, Zn	WAT, ST	48h 72h (only Ni) 96h	S	13		48h LC_{50} values were 34.6, 2.4, 2.2, 43.8, 0.47, 3618, 252, 72 and 121 ppm in order listed; 96h LC_{50}s were 1.7, 1.3, 0.42, 27.6, 0.001, 2623, 66 (72h), 50 and 19.8 ppm in order listed.	Martin and Holdich, 1986
11	*Elasmopus bampo*	C	Cd	WAT, ST	96h 7 day	S	19.5	35	96h LC_{50} = 0.57 ppm and 7day LC_{50} = 0.2 ppm.	Hong and Reish, 1987
12	*Elasmopus bampo*	C	As, Cd, Cr, Cu, Pb, Hg, Zn	WAT, ST	96h - 20 days	S, A	19±1		96h LC_{50}s were 2.75, 0.9, 3.4, 0.25, >10, 0.02, and 12.5 ppm in order listed; the metal levels were <0.01, 58.7, 11.5, 32, 1.2, <0.01 and 0.05 ppm dry wt in order listed.	Reish,1993
13	*Eohaustorius sencillus*	IN	Zn, Cd	SED, CF, CH	72h	S			Both Zn and EDTA decreased mortality in sediment containing Cd; when this amphipod was offered a choice between Cd-rich sediment and untreated sediment, 98% preferred the natural sediment.	Oakden et al., 1984a

Table 1. (Continue)

No	Species	Habitat[a]	Metal	Method[b]	Test duration	End point[c]	Temp. (°C)	Salinity (‰)	Results	Reference
14	*Eohaustorius estuarius*	IN	Cd	WAT, SED	4 days	S		30	The amphipods held in the laboratory exhibited an increased sensitivity (lowered LC_{50}) to Cd; 4-day LC_{50}s were 41.9, 36.1 and 14.5 ppm (in water) for animals held in the laboratory for 11, 17 and 121 days, respectively.	Meador,1993
15	*Gammarus pseudolimnaeus*	FW	Pb	WAT, CF	96h-28 days	S, A	15		Pb was toxic to amphipods and caused more than 50% mortality at concentrations of 136 ppb and above after 96h; 28-day LC_{50} = 28.4 ppb and 96h LC_{50} = 124 ppb; Pb levels in animals increased with increased Pb levels in the water after 28 days.	Spehar *et al*., 1978
16	*Grandidierella japonica*	IN	Cd	WAT, ST	96h 7 day	S	19.5	35	96h LC_{50} = 1.17 ppm and 7day LC_{50} = 0.5 ppm.	Hong and Reish, 1987
17	*Hyallella azteca*	FW	Pb	WAT, ST	12-120h	S			Free Pb concentration reflects Pb's biochemical activity better than total Pb; the highest mortality rates are associated with the highest free Pb concentrations.	Freedman *et al*., 1980
18	*Orchestia gammarellus*	SW (supra littoral)	Zn, Cu	WAT, ST	21 days	U, A	10±1	33	This species showed net accumulation of dissolved Zn and Cu at all exposures between 20 and 1000 ppb Zn and 13 and 1000 ppb Cu in seawater; ^{65}Zn uptake rate was 0.430 ppm Zn d^{-1}; there was no significant excretion of labelled zinc detected in the urine of amphipods exposed to labelled zinc in solution.	Weeks and Rainbow, 1991
19	*Orchestia gammarellus*	SW (supra littoral)	Cu, Zn	WAT, ST	21 days	U	10	33	This species accumulated Cu and Zn from a range of Cu- and Zn-enriched algal foods; accumulation of Cu from food was a more important route than the accumulation of Cu from solution.	Weeks and Rainbow, 1993
20	*Orchestia gammarellus*	SW (supra littoral)	Zn, Cd	WAT, ST	4 days	U	10	vary	Zn uptake rate increased linearly with increased total dissolved labelled Zn concentrations; at 33‰ NaCl free Zn ion concentrations would have been 63% of the total Zn at each exposure; the presence of EDTA reduced the mean uptake rate of each metal; between salinities of 36.5‰ and 25‰ there was correlation between free ion concentrations of both metals and metal uptake rates; Cd uptake rates were higher in lower salinity.	Rainbow *et al*., 1993
21	*Orchestia mediterranea*	SW (littoral)	Zn, Cu	WAT, ST	21 days	U, A	10±1	33	This species showed net accumulation of dissolved Zn and Cu at all exposures between 20 and 1000 ppb Zn and 13 and 1000 ppb Cu in seawater; ^{65}Zn uptake rate was 0.408 ppm Zn d^{-1}; this species was able to obtain sufficient metabolic Cu from solution.	Weeks and Rainbow, 1991
22	*Orchestia mediterranea*	SW (littoral)	Cu, Zn	WAT, ST	21 days	U	10	33	This species accumulated Cu and Zn from a range of Cu- and Zn-enriched algal foods; this species was unable to meet its Cu requirements from a food source, but was able to achieve all its Cu requirements from solution.	Weeks and Rainbow, 1993

Table 1. (Continue)

No	*Species*	Habitat[a]	Metal	Method[b]	Test duration	End point[c]	Temp. (°C)	Salinity (‰)	Results	Reference
23	*Pontoporeia affinis*	C	Cd	WAT, SED, CF	up to 460 days	S, G, A	2-6	6.8-7.3	After 105d there was no significant difference in survival of amphipods exposed to 6.3 and 31 ppb; mortality became significant at 100 ppb; Cd accumulation was consistently greater in animals than in sediment; there was an increased accumulation of Cd in the sediment when animals were present.	Sundelin, 1983
24	*Rhepoxynius abronius*	IN	Zn, Cd	SED, CF, CH	72h	S			Lethal concentration of Cd was increased when Zn was present; preferred sediment with complexed vs. non-complexed Cd.	Oakden *et al*., 1984a
25	*Rhepoxynius abronius* *Rhepoxynius fatigans*	IN	Zn, Cd	SED, CF, CH	72h	B			These amphipods avoided sediments containing high concentrations of these metals; burrowed into sediment containing low concentrations of two metals.	Oakden *et al*., 1984b
26	*Rhepoxynius abronius*	IN	Cd	WAT, SED, ST	4 days 10 days	S, E, R	15	25	There was an inverse relationship between Cd levels in sediment and both survival and reburial; 10-day LC_{50} for survival and EC_{50} for reburial were 6.9 and 6.5 ppm (in sediment), respectively; amphipods emerged from sediment containing 8.09 and 9.34 ppm Cd, at 16.2 ppm emergence was most rapid during the first 4-6 days and then declined; 4-day LC_{50} for survival was 1.61 ppm (in seawater) and EC_{50} for reburial was 0.55 ppm in seawater.	Swartz *et al*., 1985a
27	*Rhepoxynius abronius*	IN	Cd	SED	10 days	S, E, R	15	25	Survival and reburial decreased with increasing Cd concentrations in sediment, emergence rate decreased rapidly after 6 days at 16 ppm in sediment.	Swartz *et al*., 1985b
28	*Rhepoxynius abronius*	IN	Cd	SED, ST	10 days	S, E, R	15	25	LC_{50} values ranged from 9.44 to 11.45 ppm; EC_{50} (emergence) values ranged from 9.12 to 11.06 ppm; EC_{50} (reburial) values ranged from 7.66 to 10.39 ppm ; this amphipod was recommended for comparison of sediment toxicity tests.	Mearns *et al*., 1986
29	*Rhepoxynius abronius*	IN	Cd	WAT, ST	96h	S	19.5	35	96h LC_{50} = 0.24 ppm.	Hong and Reish, 1987
30	*Rhepoxynius abronius*	IN	Cd	WAT, SED, ST, CF	96h	S, R	15	25	Cd toxicity to this species appears to be due to Cd dissolved in interstitial water; survival and reburial decreased with increasing dissolved and total sediment Cd concentration.	Kemp and Swartz, 1988
31	*Echinogammarus olivii*	SW	Cu, Zn, Pb	WAT, ST	96h	S	15	17	96h Cu LC_{50}= 0.21-0.28 ppm; 96h Zn LC_{50}= 1-1.57 ppm; 96h Pb LC_{50}= 0.58-0.67 ppm;	Bat *et al.,* 1999
32	*Gammarus pulex pulex*	FW	Cu, Zn, Pb	WAT, ST	96h	S	15, 20, 25		The LC_{50} values of Cu, Zn and Pb ranged from 0.028 to 0.080, 5.2 to 12.1 and 11.2 to 23.2 mg/l, respectively. The results indicated that Cu was more toxic to the species followed by Zn and Pb.	Bat *et al.,* 2000

[a] IN= infaunal, SW= seawater, FW= freshwater, C= cultured animals
[b] WAT= water, SED= sediment, ST= static system, CF= continuous-flow system, CH= choice experiment
[c] S= survival, G= growth, E= emergence, R= reburial, B= burrowing, A= accumulation, U= uptake

Table 2. Polychaete toxicology studies involving water and sediment exposures in laboratory and/or field bioassays

No	Species	Habitat[a]	Metal	Method[b]	Test duration	End point[c]	Temp. (°C)	Salinity (‰)	Results	Reference
1	*Capitella capitata*	C	Cu, Zn	WAT	16 days or more	REP			Variable concentrations of Cu and Zn caused fatal abnormalities in the first or second generation of larvae.	Reish *et al*., 1974
2	*Capitella capitata*	IN	Hg	WAT, ST	0.25h- 2days	S	10		The worms are shown to be fairly resistant to high concentrations of inorganic Hg; LT_{50} increases with decreasing Hg concentration.	Warren,1976
3	*Capitella capitata*	C	Cd, Cr, Cu, Pb, Hg, Zn	WAT	96h 28day	S			96h LC_{50}s were 7.5, 5, 0.2, 6.8, <0.1 and 3.5 ppm for adults and 0.22, 8, 0.18, 1.2, 0.014 and 1.7 ppm for trochophore larvae in order listed; 28-day LC_{50}s were 0.7, 0.28, 0.2, 1, 0.1 and 1.25 ppm for adults in order listed.	Reish *et al*., 1976
4	*Capitella capitata*	C	Ca, Mg, Al, Na, Co, Cu, Fe, Pb, Mn, Rb, Ag, Zn Sr,Ni,K,Cd	WAT, SED, ST, detritus	90 days	G, A	20±1		Nutritional quality of the food source influenced metal uptake; metal accumulation in the animals was significantly increased when fed detritus containing metal levels significantly elevated above natural levels.	Windom *et al*., 1982
5	*Cirriformia spirabrancha*	IN	Cu	WAT, SED	5-34 days	S, U	10	29	In Cu concentrations at or below 0.08 ppm the worms survived for at least 21 days; dissolved yellow organics were shown to have no effect on the rate of Cu uptake by the worms in seawater.	Milanovich *et al*., 1976
6	*Ctenodrilus serratus*	C	Cd, Cr, Cu, Pb, Hg, Zn	WAT	96h 21 days	S, REP			Hg and Cu were the most toxic to this polychaete.	Reish and Carr, 1978
7	*Glycera dibranchiata*	IN	Cd	SED	7 days 14 days 21 days 28 days	B, U, feeding	15	20-25	After 28 d, Cd body burdens were lower in this species (120 ppm) than in *Nereis virens*, but higher than in *Nephtys caeca*; this was the same for burrowing behaviour; after 28, Cd-exposed and unexposed *G. dibranchiata* presented with live *Euzonus mucronata* showed no significant differences in feeding.	Olla *et al*., 1988
8	*Hermione hystrix*	IN	Zn	WAT, SED, CF	several days to two moths (or more)	A	20±2		Worms accumulated ^{65}Zn from sediments; the presence of worms in the sediment caused the release of ^{65}Zn to overlying water.	Renfro, 1973
9	*Melinna palmata*	IN	Cu	WAT, SED					This species consistently contains a high Cu concentration; Cu may reduce the palatability of the tissues and is accumulated by the organism as a chemical defence against predation.	Gibbs *et al*., 1981
10	*Namanereis merukensis*	IN	Hg, Cu, Pb	WAT, ST	96h	S	room ?	35.5-36.7	96h LC50 values were 0.041, 0.55 and 3.75 ppm for Hg, Cu and Pb, respectively.	Varshney and Abidi, 1988

Table 2. (Continue)

No	Species	Habitat[a]	Metal	Method[b]	Test duration	End point[c]	Temp. (°C)	Salinity (‰)	Results	Reference
11	*Neanthes arenaceodentata*	C	Cd, Cr, Cu, Pb, Hg, Zn	WAT	96h 28day	S			96h LC_{50}s were 12, >1, 0.3, >10, 0.022 and 1.8 ppm for adults and 12.5, >1, 0.3, >7.5, 0.1 and 0.9 ppm for juveniles in order listed; 28-day LC_{50}s were 3, 0.55, 0.25, 3.2, 0.017 and 1.4 ppm for adults and 3, 0.7, 0.14, 2.5, 0.09 and 0.9 for juveniles in order listed.	Reish *et al*., 1976
12	*Neanthes arenaceodentata*	C	Cu	WAT, SED, CF	28 days	S	17±1	31±1	28-day LC_{50} was lower for worms exposed without sediment than those with sediment, 0.044 and 0.10 ppm Cu in seawater, respectively.	Pesch and Morgan, 1978
13	*Neanthes arenaceodentata*	C	Cu	WAT, SED, CF	85 days	S, A	18±1	32±1	TL_{50} was 7.8 days without sediment, 36.5 days with sediment, 54.5 days with mixture and 50 days with mud.	Pesch, 1979
14	*Neanthes arenaceodentata*	C	Ag	WAT, SED, CF	96h 10 days 28 days	S, B	20±1	30±2	28-day LC_{50} for the participating laboratories were 165±52 ppb; the ratio of the highest LC_{50} value was 2.23; 96h and 10-day LC_{50} values were 233 and 206 ppb, respectively; most of the live worms were able to burrow.	Pesch and Hoffman, 1983
15	*Neanthes arenaceodentata*	C	Zn, Cd	WAT	36h-6wk	A, U	4 21		Uptake occurs from free ionic pool of metal and EDTA and EDTA-metal complexes are largely excluded; in unfed worms the metals accumulate linearly with time at a rate which decreases when temperature is reduced; beginning of exposure ligands appear to bind Cd in preference to Zn but after 50h the worms selectively accumulate Zn by a process requiring metabolic energy.	Mason *et al*., 1988
16	*Neanthes arenaceodentata*	IN	Cd	WAT, SED, ST, CF	96h 28 days	S, G	20	30	96h- LC_{50} was 5.2 ppm; 0% survival at 6.5 ppm and 100 % survival at 3.8 ppm; grain size of sediment had no significant effect on survival and growth; direct transfer from 30 ‰ seawater to salinities ≤15‰ had a highly significant and adverse effect on survival and growth.	Dillon *et al*., 1993
17	*Neanthes vaali*	IN	Cd, Zn	WAT, ST	96-168h	S	18.5-18.7	32.7-34.2	168h Cd LC_{50}= 6.4 ppm; 96h Zn LC_{50}= 5.5pm.	Ahsanullah, 1976
18	*Nephthys hombergi*	IN	Cu, Zn	WAT, SED	96h	S, U			96h Cu LC_{50}= 0.7 and 0.25 ppm tolerant and non-tolerant animals, respectively; metal levels 18 and 2120 ppm Cu normal and contaminated areas, respectively, and 305 and 483 ppm Zn normal and contaminated areas, respectively.	Bryan,1976a

Table 2. (Continue)

No	Species	Habitat[a]	Metal	Method[b]	Test duration	End point[c]	Temp. (°C)	Salinity (‰)	Results	Reference
19	*Nephtys caeca*	IN	Cd	SED	7 days 14 days 21 days 28 days	B, U	15	20-25	After 28 d, Cd body burdens were lowest in this species (39 ppm) compared to *Glycera dibranchiata* and *Nereis virens*; burrowing by Cd-exposed *N. caeca* was significantly slower at in 14 and 28 d than in those other polychaetes.	Olla *et al*., 1988
20	*Nereis diversicolor*	IN	Cu	WAT, SED	7 day 37 day	S, U	13		Tolerance to the toxic effects of Cu is very different in two populations of the same species.	Bryan and Hummerstone, 1971
21	*Nereis diversicolor*	IN	Zn, Cd	WAT, SED	96 h 816 h	S, U	13	0.35-17.5 17.5	Zn is regulated by the worm, whereas Cd is not; in laboratory, increasing concentrations in solution the rate of absorption of Cd increases more rapidly than that of Zn; in the field, concentrations of Zn in the worms vary less than those of Cd and populations from high-Zn sediments are better at regulating Zn than normal populations and these worms more resistant to Zn than normal worms.	Bryan and Hummerstone, 1973a
22	*Nereis diversicolor*	IN	Mn	WAT, SED	1 wk 2 wk	S, U	13	1.6-20	With decreasing salinity, the concentration factor increases; cleaning process (gut contents) removed about 70% of Mn absorbed from the two higher concentrations.	Bryan and Hummerstone, 1973b
23	*Nereis diversicolor*	IN	Zn	WAT, SED, CF	days - two moths (or more)	A	20±2		Worms can accumulate ^{65}Zn from sediments; the presence of worms in the sediment causes the release of ^{65}Zn to overlying water.	Renfro, 1973
24	*Nereis diversicolor*	IN	Cu, Zn	WAT, SED	96h	S, U			96h Cu LC_{50}= 2.3 and 0.54 ppm tolerant and non-tolerant animals, respectively; metal levels 22 and 1140 ppm Cu normal and contaminated areas, respectively.	Bryan,1976a
25	*Nereis diversicolor*	IN	Fe	SED, C	10-88 days	U, A	15±1		Bioavailabilty of ^{55}Fe was shown to depend on its concentration in sediment and not on sediment type; trends in uptake were uniform, but accumulation of ^{55}Fe appeared to be complete after 25 to 35 days.	Jennings and Fowler, 1980
26	*Nereis virens*	IN	Cu, Zn, Cd, Pb	SED, ST	30 days	A	10±0.5		Cu and Zn concentrations in worms exposed to the sediments showed no significant changes from initial values; it was suggested that this species might be useful for testing for Cd and Pb bioavailability.	Ray *et al*., 1981
27	*Nereis virens*	IN	Cd	WAT, SED, ST	30 days	A	10±1		Cd levels in worms increased with increasing Cd levels in sediment; smaller worms accumulated higher amaunts of Cd (per unit wt) and at a greater rate than larger ones; uptake rate of Cd by worms was related to the Cd concentrations in water which in turn was related to the Cd concentrations in sediment.	Ray and McLeese, 1983

Table 2. (Continue)

No	Species	Habitata	Metal	Methodb	Test duration	End pointc	Temp. (°C)	Salinity (‰)	Results	Reference
28	*Nereis virens*	IN	Cd	SED	7 days 14 days 21 days 28 days	B, U	15	20-25	After 28 d, uptake was highest in this species (319 ppm) compared to *Glycera dibranchiata* and *Nephtys caeca*.	Olla *et al*., 1988
29	*Nereis virens*	IN	Cd	WAT	24h, 48h, 96h	S	20	20	24h, 48h and 96h Cd LC_{50}s were 25, 25 and 11 ppm, respectively.	Eisler, 1971
30	*Ophryotrocha diadema*	C	Cd, Cr, Cu, Pb, Hg, Zn	WAT	96h 21 days	S, REP			Hg and Cu were the most toxic to this species.	Reish and Carr, 1978
31	*Ophryotrocha labronica*	SW	Zn, Cu, Hg, Cd, Fe, Pb	WAT, ST		S, G	20		The order of toxicity is Hg Cu Zn Cd Fe Pb; a significant suppression of growth rate in Cu solutions containing 0.1 and 0.05 ppm Cu; no significant growth suppression was obtained in 0.1 ppm Zn or 10 ppm Pb.	Brown and Ahsanullah, 1971
32	*Ophryotrocha labronica*	SW	Cu	WAT, ST	9day 3wk 5wk	S, G, REP	21-22		Larvae showed an improved tolerance to 1 and 5 ppm after acclimatization in 0.025 ppm Cu, adults acclimated for 3wk in 0.1 ppm showed no difference from control.	Saliba and Ahsanullah, 1973
33	*Phyllodoce maculata*	IN	Cu	WAT, ST	21days	S, A	10		The rate of uptake may be the lethal factor, rather than the amount of Cu accumulated.	McLusky and Phillips,1975
34	*Hediste diversicolor*	IN	Zn, Pb	WAT, SED, ST	10 days 28 days	S	20		Mortality has increased with increasing concentrasions of zinc and lead. Zn was more toxic to the species than Pb. Small worms are more sensitive to Zn and Pb than bigger worms.	Bat *et al.,* 2001

[a] IN= infaunal, SW= seawater, FW= freshwater, C= cultured animals
[b] WAT= water, SED= sediment, ST= static system, CF= continuous-flow system
[c] S= survival, G= growth, E= emergence, R= reburial, B= burrowing, A= accumulation, U= uptake

Table 3. *Corophium volutator* toxicology studies involving water and sediment exposures in laboratory and/or field bioassays

No	Toxicant and/or study area	Method[a]	Test duration	End point[b]	Temp. (°C)	Salinity (‰)	Results	Reference
1	Hg	SED, ST, L, AD	21h	B, CH and non-CH	10		In control and sediment containing 0.001, 0.1 and 10 ppm Hg, most individuals burrowed, no individual burrowed in the 1000 ppm Hg in sediment; amphipods avoided sediment containing 0.001 ppm Hg (the lowest concentrations tested) when they were offered untreated sediment as a choice.	Erdem and Meadows, 1980
2	Hg	SED, ST, L, AD	6h - 33 days	S, U	10		The percentage mortality of amphipods increased with time and with Hg concentration; all animals in sediment treated with 1000 ppm were dead by 6h, very few *Corophium* in 10 ppm or less died after day 7; living and dead animals accumulated large amounts of Hg; accumulation is greater in dead than in living animals.	Meadows and Erdem, 1982
3	Cd, sewage sludge	SED, ST, L	up to 35 days	A	10		^{109}Cd associated with sewage sludge is taken and accumulated; *Corophium* was unable to regulate the body concentration of Cd.	Caparis and Rainbow, 1994
4	^{241}Am, ^{238}Pu	SED, WAT, CF, L	4 days 14 days	U	14±2		Uptake of both radionuclides (from sediment) by *Corophium* was 10 times and 50 times greater than uptake by the clam *Scrobicularia plana* and the lugworm *Arenicola marina*; similarly *Corophium* accumulated 10 times more Am and 14 times more Pu (from seawater) than *S. plana* and 78 times more Am and 180 times more Pu than *A. marina*.	Miramand *et al*., 1982
5	Sediment bioassays in European waters (comparison studies)	SED, ST, AD, L	10 days	S	15±1	31±2	*Corophium* was recommended for use in sediment toxicity tests in European waters or estuaries.	Fleming *et al*., 1994
6	Cu, Zn	WAT	96h 168h	S, U		50% sea water	96h LC_{50} were 66 ppm for Cu; 168h LC_{50} s 50 and 32 ppm Cu for tolerant and non-tolerant animals, respectively; Cu levels in tolerant and non-tolerant animals were 499 and 96 ppm, respectively; Zn levels in tolerant and non-tolerant animals were 254 and 130 ppm, respectively.	Bryan, 1976a,b
7	^{244}Cm	WAT, ST, L	11 days	U	14±1		*Corophium* accumulated more Cm than *Arenicola marina, Cerastoderma edule, Nereis diversicolor* and *Scrobicularia plana,* reaching concentration factors above 700 after 11 d.	Miramand *et al*., 1987
8	Cr	WAT, ST, L	up to 384h	S	5±0.5 10±0.5 15±0.5	5-40	Toxicity of Cr increased as temperature increased and salinity decreased.	Bryant *et al*., 1984
9	As	WAT, ST, L	up to 384h	S	5±0.5 10±0.5 15±0.5	5-35	LT_{50} decreased with increasing As concentration for all combinations of temperature and salinity; *Corophium* was more sensitive to As than *Macoma balthica*; 96h LC_{50}values for *Corophium* ranged from 6 to 60 ppm depending on temperature.	Bryant *et al*., 1985a
10	Ni, Zn	WAT, ST, L	up to 384h	S	5±0.5 10±0.5 15±0.5	5-35	Maximum survival at low temperature and high salinity levels for both metals; 96h LC_{50}s for Ni and Zn ranged from 5 to 54 ppm and 1 to 16 ppm, respectively.	Bryant *et al*., 1985b

Table 3. (Continue)

No	Toxicant and/or study area	Method[a]	Test duration	End point[b]	Temp. (°C)	Salinity (‰)	Results	Reference
11	Cr, As, Z, Ni	WAT, ST, L	up to 384h	S	5-15	5-35	Maximal toxicity occured in highest temperatures and lowest salinities; for As no effect of salinity was observed.	McLusky and Bryant, 1985
12	Cr, As, Z, Ni	WAT, ST, L	up to 384h	S	5-15	5-35(40) in 5 ‰ incre-ments	In general, metal toxicity increases as salinity decreases and as temperature increases; 96h LC_{50} values indicate a rank order of metal toxicity of Zn> Cr> Ni> As.	collective studies in review by McLusky *et al*., 1986
13	Cd, Pb	WAT, ST, L	96h	A	15.5±0.5	25	*Corophium* accumulated Cd and Pb from contaminated seawater; animals exposed for 96h to the non-essential metals, a power function $Y=aX^b$ is generally suitable to describe the relation between the levels of metals in organism (Y) and seawater (X); for *Corophium* a=93.66 and b=0.65 for Cd, a=1639.97 and b=0.62 for Pb.	Amiard *et al*., 1987
14	Cu	WAT, ST, L, AD	14 days	S, A	18±1	25	*Corophium* appeared to be a net accumulator of Cu, and Cu exposure resulted in a lowered reproductive success rate; mortality was higher at low oxygen saturations (below 30%).	Eriksson and Weeks, 1994
15	Hg in Elbe estuary,Germany						Concentrations in tissues were 0.05 - 0.10 ppm.	Zauke, 1977
16	Cu, Zn, Mn, Fe, Ca, Mg, Pb, Ni, Co, Cd in Dulas Bay and Menai Strait (N.Wales)	AD					Only Cu occurs in higher levels (259µg/g) from Dulas Bay; Ca, Mg and Pb are higher in the Menai Strait sediments but the levels in animals are similar for both area; Ni, Co and Cd have not been detected in both sediments and animals; Cu could be excreted directly in an insoluble form.	Icely and Nott, 1980
17	Sediment bioassay in Halifax Harbour	SED, WAT, L	96h 12-19 days	S, R	15±2		96h LC_{50} values for Cd ranged from 10.1 ppm to 22.7 ppm in seawater; *Corophium* was less sensitive to Halifax Harbour sediments than *Rhepoxynius abronius*; 47.3% of *Corophium* burrowed in sediment at the end of the bioassay.	Tay *et al*., 1992
18	Sediment bioassays in North Sea	SED, ST, L	10 days	S, I			*Corophium* was recommended for use in sediment toxicity tests in European waters, especially in UK.	Chapman *et al*., 1992
19	Sediment bioassays in North Sea (comparison studies)	SED, ST, L	10 days	S, I, E	14±1		Survival of *Corophium* was significantly reduced in more sediment samples than any of the other species tested (*Rhepoxynius abronius*, *Bathyporeia sarsi*); first observations of emergence and immobilisation were recorded on the third days of exposure, at the end of the test 25% of *Corophium* were immobilised.	van den Hurk *et al*., 1992
20	Sediment bioassays in UK (estuaries)	SED, ST, AD,ship-board testing	10 days	S			None of sediments tested were highly toxic to *Corophium*, mortalities above 50% were not observed; suggested that *Corophium* was suitable for deployment in sediment quality monitoring programmes, particularly in estuarine areas.	Thain *et al*., 1994

Table 3. (Continue)

No	Toxicant and/or study area	Methoda	Test duration	End pointb	Temp. (°C)	Salinity (‰)	Results	Reference
21	Crude oil in sediment from England	SED, ST, AD, L	10 days	S	13±1	33	Mortality of *Corophium* was significantly elevated at the most contaminated site; in the control sediment mortality was 12%; suggesting that *Corophium* can be used in laboratory bioassays.	Roddie *et al*., 1994
22	Cu, Zn, Cd	SED, ST, AD, L	10 days	S, E, B	11°C± 1	32	LC_{50} indicate that Cd was much more toxic than Cu or Zn, being 14, 37, 32 µg g^{-1}, respectively and a similar trend was seen for the EC_{50}s. The emergence from sediment differed greatly between concentrations of 30 to 57, 26 to 59 and 9.18 to 28.27 µg g^{-1} of Cu, Zn and Cd, respectively.	Bat and Raffaelli, 1998a
23	Organically enriched sediment	SED	10 days 28 days	S, E, B			*Corophium* can survive in organically enriched sediment if they have no alternative, suggesting that *Corophium* is relatively tolerant of organically enriched sediment.	Bat and Raffaelli, 1998b
24	permethrin	SED, L	28	S	15°C± 1	32	28-day LC_{50} was 67 ng g^{-1} ranging from 55 to 82 ng g^{-1}	Bat and Raffaelli, 1996
25	Sediment from Sotiel and Gibraleon in Spain	SED	10	S, E, B	11±1° C	30	Only 20% of the amphipods survived at the end of the 10-day exposure to the Gibraleon sediments. All live animals were able to rebury successfully. No *Corophium* had burrowed in the Sotiel sediment.	Bat et el., 1996
26	Cu, Zn, Cd	WAT, SED, ST	3, 6, 24, 48, 72 and 96 h	A, CH	11±1° C	32	BCF were inversely related to seawater with Cu, Zn and Cd, with the lowest exposure concentration having the highest BCF. In the non-choice experiment *Corophium* survival declined with increasing sediment metal levels as did burrowing activity. When Cd and Zn were present together *Corophium* mortality was less than with Cd alone.	Bat *et al*., 1998
27	Cu, Zn, Cd	WAT, SED, ST	4 days 10 days	U	11±1° C	32	Metals were determined in *Corophium* tissues in individuals with gut contents and in individuals with contents excluded by three different protocols.	Bat and Raffaelli, 1999

[a] WAT= water, SED= sediment, ST= static system, CF= continuous-flow system, AD= adult animals, L= laboratory
[b] S= survival, E= emergence, B= burrowing, A= accumulation, U= uptake, CH= choice experiment, I= immobilisation

Table 4. *Arenicola marina* toxicology studies involving water and sediment exposures in laboratory and/or field bioassays

No	Toxicant and/or study area	Method[a]	Test duration	End point[b]	Temp. (°C)	Salinity (‰)	Results	Reference
1	^{137}Cs, ^{60}Co	WAT, SED, ST, L	3 wk 1 to 2 months (depuration)	U, DEP	14-16		Worms can reduce both radionucleids from their body with increased time in seawater, with and without sediment; at the beginning elimination of Cs decreased rapidly and then slowly, elimination of Co was faster in water with sediment than in water only; before depuration Cs was concentrated in digestive tube (57%), after depuration Cs was found in skin and muscles; before and after depuration Co was concentrated in digestive tube and blood.	Amiard-Triquet, 1974a
2	^{137}Cs	WAT, SED, ST, L	8 to 11 days	U	14-16	vary	There was an inverse relationship between salinity of seawater and Cs levels in worms; similarly for K in seawater and Cs in worms; Cs was concentrated in digestive tube in contaminated normal seawater, Cs of the worms was higher in artificial seawater containing 50% less K than those in normal seawater.	Amiard-Triquet, 1974b
3	^{57}Co, ^{137}Cs, ^{141}Ce	WAT, SED, ST, L	1 to 2 wk up to 40 days	U	3±1 5±1 15±1 17±1	40% seawater	Co in seawater was accumulated in kidney, Cs in seawater was accumulated in soft tissues (homogen), Ce in seawater was accumulated in external skin and digestive system; there was an inverse relationship between organic content of sediment and both Co and Cs levels in worms.	Amiard-Triquet, 1975
4	^{241}Am, ^{238}Pu	SED, WAT, CF, L	6 days 14 days	U	14±2		*Arenicola* preferentially accumulated Am rather than Pu; *Arenicola* accumulated less Am and Pu than the clam *Scrobicularia plana* and the amphipod *Corophium volutator* both from seawater and sediment.	Miramand *et al*., 1982
5	^{244}Cm	WAT, ST, L	11 days	U	14±1		Cm uptake by *Arenicola* and *Nereis diversicolor* was similar but lower than that found for *Corophium volutator, Cerastoderma edule* and *Scrobicularia plana*.	Miramand *et al*., 1987
6	Cd, Cu, Pb, Zn, Mg in the coast of Wales			A			Cd was present in lowest concentrations both in worms and sediments; Zn, Mn and Pb all decreased in concentration with increase in body weight, but Cd and Cu were not related to body weight; for all metals there were significant positive correlations between the metal levels in worms and the metal levels in sediment.	Packer *et al*., 1980
7	Cd, Cu, Ni, Zn in Loughor Estuary, S.Wales			A			*Arenicola* casts contained slightly higher average metal concentrations than those sediment.	Brown, 1986
8	Cd, Pb, Zn	WAT, SED		A			Levels of all metals in casts were less than those in sediment; dominant uptake of Cd was via the dissolved phase; Cd levels in *Arenicola* was 27± 18.5 ppm.	Loring and Prosi, 1986
9	Oiled sediment	WAT, SED, ST, CF,L	3-7 days	S, CA, E, U			High concentrations of oil in seawater and in sediment forced worms to surface or to stop ingesting sediment; oil concentrations in casts were lower than those in unworked sediment.	Prouse and Gordon, 1976
10	Oiled sediment	SED, CF, L	5 days	B, S, CA, U	3.7-16	30	Some worms surfaced in oil concentrations 153 ppm and some worms died when concentrations reached 275 ppm; in all experiments only 14% of *Arenicola* died; worms burrowed into sediment within minutes; cast activity reduced at higher concentrations; oil levels in casts were lower than those in unworked sediment.	Gordon *et al*., 1978

Table 4. (Continue)

No	Toxicant and/or study area	Methoda	Test duration	End pointb	Temp. (°C)	Salinity (‰)	Results	Reference
11	Kuwait oil and BP1100X dispersant in Sandyhaven Pill	Field	several months	S, CA			Kuwait oil and BP1100X and 1:1 and 5:1 mixtures of both all reduced population density of worms; pollutants also reduced feeding activity of up to 75% of the worms when natural populations of *Arenicola* were sprayed with Kuwait crude oil at a rate of 0.2 L/m^{-2} ; there was a rapid decline in cast production following the spills over next month, a gradual increase in feeding activity occured reaching a constant level at about 50-75% of the original worm density.	Levell, 1976
12	Aroclor 1254	SED, WAT, ST, L	5 days	U	room temp.		Sediment containing 1 ppm A1254, worms accumulated 0.24±0.08 ppm A1254; the addition of clean sand did not effect the rate of uptake.	Courtney and Langston, 1978
13	Hydrocarbon ^{14}C-1-naphthalene	WAT, SED, L	up to 24h	U, A, DEP			Uptake was rapid in all tissues, the most important site for accumulation being the stomach wall and the oesophagenal glands; the loss of the hydrocarbon from the tissues was rapid.	Lyes, 1979
14	Sediment with diesel-based drilling mud and TBT from Maplin Sands, Coast of Essex	SED	6 months	CA			All treatments except 0.1 mg TBT kg^{-1} impaired the casting activity of *Arenicola*; this technique was found to be useful.	Matthiessen and Thain, 1989
15	Sediment bioassays in UK (estuaries)	SED, ST, AD,ship-board testing	10 days	S, CA			No mortalities were observed in control sediment; contaminated natural sediments effected feeding behaviour; *Arenicola* bioassays were found easy to deploy for ship-board monitoring.	Thain *et al*., 1994
16	Cu, Zn, Cd	WAT, SED, ST, L	4 days	U	9 ± 1°C	32	No lugworms survived at the end of the exposure to concentrations of 20 µg g^{-1} Cu, 60 µg g^{-1} Zn and 35 µg g^{-1} Cd in sediment. Mortality of lugworms increased with increasing copper, zinc and cadmium sediment concentrations, this becoming more significant at higher concentrations.	Bat, 1998
17	Cu, Zn, Cd	SED, ST, L	10 days	S, E, CA, A	9 ± 1°C	32	LC_{50} analyses show that Cu was more toxic to lugworms than either Zn or Cd, the LC_{50}s being 20, 50 and 35 µg g^{-1} Cu, Zn and Cd, respectively. Lugworms were able to burrow in sediment containing 14 µg Cu g^{-1} , 52 µg Zn g^{-1}, 25 µg Cd g^{-1} or less. At higher concentrations of the metals the size of the casts produced declined sharply. Tissue metal concentrations increased with increasing copper, zinc and cadmium sediment concentrations.	Bat and Raffaelli, 1998a

[a] WAT= water, SED= sediment, ST= static system, CF= continuous-flow system, L= laboratory
[b] S= survival, E= emergence, B= burrowing, A= accumulation, U= uptake, DEP= depuration, CA= casting activity

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