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Contaminant chemistry and toxicity of sediments in Sydney Harbour, Australia: spatial extent and chemistry–toxicity relationships

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ABSTRACT: The spatial distribution of chemical contamination and toxicity of surficial sediments in Sydney Harbour, Australia, was investigated in a 3-tiered, hierarchical approach. An initial chemical investigation throughout the entire estuary (Stage 1) indicated wide ranges and different spatial patterns in sediment chemical concentrations. Sediment quality guidelines (SQGs) were used as a preliminary estimate of possible toxicity in Stage 2 of the investigation. Assessment of chemical mixtures indicated that sediments in a small part (~2%) of the harbour had the highest probability of being toxic (~75%), whereas sediment in almost 25% of the port was estimated to have an intermediate (~50%) probability of being toxic. The SQG assessment in Stage 2 enabled careful stratification of the harbour into areas with different toxicity risks, reducing cost and time commitments in the final tier of assessment. The spatial survey carried out in Stage 3 involved concurrent chemical and ecotoxicological analyses. In this final stage, the degree of response in tests of amphipod survival in whole sediment samples, as well as in tests of microbial metabolism (Microtox©) and sea urchin egg fertilisation and embryo development in pore waters, generally increased with increasing chemical concentrations. However, amphipod response was lower than predicted due to relatively low sensitivity of the indigenous amphipod *Corophium colo*. Areas initially predicted, using SQGs, to be most at risk were highly toxic in the combined chemistry–toxicity investigation, while sediment from areas with the lowest predicted risk were least toxic, but still toxic in at least one ecotoxicological test. The results demonstrate that the empirical approach used for this study, which was originally developed in North America, produced plausible outcomes elsewhere and that observed toxicity, based on SQGs, matched predictions using different species but similar methodologies.

KEY WORDS: Sediment quality · Sediment toxicity · Sydney Harbour · Ecotoxicology · Sediment quality guidelines

INTRODUCTION

Sydney Harbour, on the central New South Wales (Australia) east coast, is approximately 30 km long and 3 km wide with an area of about 50 km^2 (Fig. 1). The harbour is a drowned valley estuary with a narrow, winding channel and irregular bathymetry. Embayments on the southern shores of the upper and central harbour are shallow (<7 m) and mantled with muddy sediments, whereas the lower estuary is moderately deep (~20 m) and characterised by sandy sediments.

Sydney Harbour catchment (~530 km^2) is intensely (86%) urbanised and industrialised and supports a population of almost 2 million people. Industrial and domestic wastes were discharged directly into Sydney Harbour until 1890, when ocean outfalls were commis-

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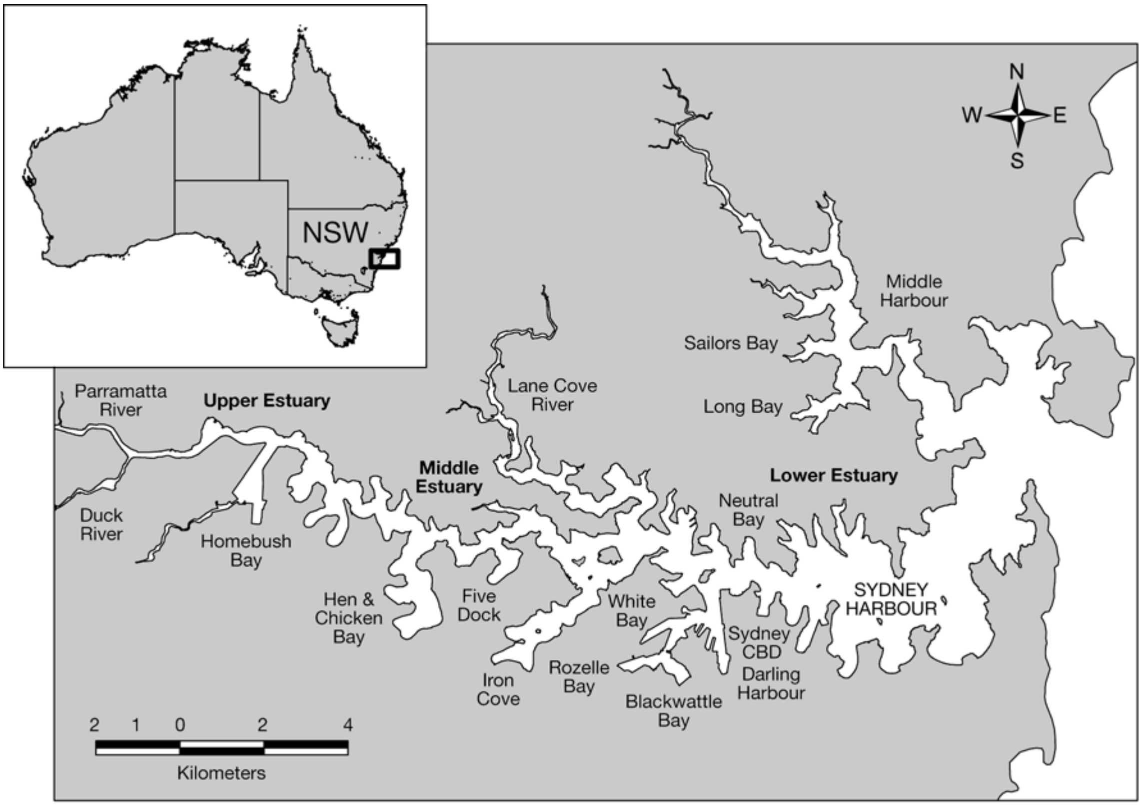

Fig. 1. Sydney Harbour and locations

sioned. Water quality began to improve with the advent of the Clean Waters Act and Regulations in 1976, which outlawed dumping in the harbour. Since then, the harbour has been slowly reverting mainly to a commuter, tourist and recreational base, while considerable industry, shipping and naval facilities are moving away.

The environmental status of Sydney Harbour was largely ignored until 15 yr ago, when a vigorous and sustained research effort was initiated. This work showed that extensive parts of the harbour were mantled with sediments containing high concentrations of a wide range of chemicals, and that these sediments were amongst the most contaminated of any capital port (Birch & Taylor 1999, 2000, McCready et al. 2000). The concentration of at least one contaminant exceeded pre-anthropogenic (background) values in all parts of the harbour, except in the estuary mouth area (Birch & Taylor 2002a,b). Consequently, any activity that disturbs bottom sediment will require additional environmental investigation according to guidelines set by the Australian and New Zealand Environment Conservation Council and the Agriculture and Resource Management Council of Australia and New Zealand (ANZECC/ARMCANZ 2000). The high concentrations of a large number of chemicals made it important to determine the toxicity of sediment in the harbour.

The spatial assessment of sediment quality in Sydney Harbour presented in this study is based on data gathered during an investigation that took place over 4 yr (1996 to 2000) and included several field studies. Detailed surficial sampling and analysis for a large number of chemicals were undertaken by Birch & Taylor (1999, 2000) and McCready et al. (2000). Their data was interpreted in Stage 1 of this study. Sediment quality guidelines (SQGs) were used as a preliminary test of possible toxicity by Birch & Taylor (2002a,b,c). The use of SQGs allowed harbour sediments to be prioritised according to the risk of adverse biological effects and enabled careful stratification of different areas in Stage 2 of this study, before committing to costly, concurrent chemical–ecotoxicological analyses in Stage 3. The final stage was a spatial study using previously published data to test the applicability of sediment quality guidelines for Australian conditions (McCready et al. 2004, 2005, 2006a,b,c).

The objectives of the current work were to demonstrate the benefits of the hierarchical approach for assessing contaminated sediment throughout a whole estuary, to determine spatial patterns of sediment toxicity in Sydney Harbour for the first time, and to test the suitability of sediment quality guidelines for the accurate prediction of presence or absence of toxicity for an urbanised/industrialised estuary containing wide ranges of contaminant types and concentrations. The combination of these 3 objectives is unique to the current work, and the results put Sydney Harbour into a global perspective chemically and toxicologically. The current work is also the first published spatial assessment of sediment toxicity for an Australian estuary.

MATERIALS AND METHODS

Stage 1: Chemical analyses of surficial sediment. The methodology for sampling and analyses has been described previously (Birch & Taylor 1999, 2000, McCready et al. 2000). Due to funding limitations, different numbers of samples were analysed for different classes of contaminants. Sampling was undertaken with a stainless steel box corer and only the most recent upper 2 to 3 cm layer was taken for analysis. A suite of 9 metals (Cd, Cr, Cu, Co, Fe, Mn, Ni, Pb and Zn) was analyzed in 875 sediment samples, and 140 samples were analysed for the presence of organochlorine pesticides (OCs) (DDT, DDD, DDE, chlordane, aldrin, heptachlor, dieldrin, heptachlor epoxide, lindane), hexachlorobenzene (HCB) and total polychlorinated biphenyls (PCBs, reported as Aroclors). A total of 16 priority pollutant polycyclic aromatic hydrocarbons (PAHs) (acenaphthene, acenaphthylene, anthracene, benz(*a*)anthracene, benzo[a]pyrene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*ghi*]perylene, chrysene, dibenz[*ah*]anthracene, fluoranthene, fluorene, indeno [1,2,3-*cd*]pyrene, naphthalene, phenanthrene and pyrene), as well as 2-methylnaphthalene, were determined in 124 samples. Metals were analysed using inductively coupled plasma optical emission spectrometry (ICP–OES), OCs by the 'one-step method' of Ahmad & Marolt (1986) and PAHs by gas chromatography–mass spectrometry (GC–MS).

Stage 2: Determination of sediment quality and assessment of likelihood of toxicity. Sediment quality guidelines (SQGs) developed by Long et al. (1995) were used to assess the probability of toxicity, as these guidelines are the basis of interim national standards for Australia (ANZECC/ARMCANZ 2000, Simpson et al. 2005). These guidelines were developed by matching chemical and biological effects databases (Long et al. 1995) and consist of 2 guideline values for each of the 25 individual chemicals for which data are available, namely effects range low (ERL) and effects range median (ERM). The ERL denotes the value below which adverse biological effects are seldom observed and the ERM represents the value above which adverse biological effects are expected to occur frequently. The ERL is set at the lower 10th percentile and the ERM at the 50th percentile of the effects data distribution (Long et al. 1995). Concentrations between ERL and ERM indicate an intermediate, often irregular biological response. Individual chemicals were assessed for possible adverse biological effects by comparing concentrations at each site to the respective ERL and ERM values.

Contaminants rarely occur individually in natural systems, and the probability of sediment toxicity was determined for mixtures of contaminants using the mean ERM quotient (MERMQ) approach (Long & McDonald 1998, Long et al. 2000, 2006). In this approach, contaminant concentrations at each site are normalised to (divided by) respective ERM values, the quotients are summed and then divided by the total number of contaminants. The results of these computations were plotted using a geographic information system (MapInfo) to obtain distributions of MERMQ for each contaminant class (metals, OCs and PAHs), and MERMQs were combined for all contaminants to provide the probability of sediment toxicity (Long & MacDonald 1998, Long 2000, Long et al. 2000, 2006). Sediments were split into categories 1, 2, 3 and 4 (Long et al. 2000) for MERMQ values of <0.1, 0.1 to 0.5, 0.5 to 1.5 and >1.5, respectively. Based on the results of amphipod mortality tests, the categories in this ranking have a ~10% (low), ~25% (medium–low), ~50% (medium–high), and ~75% (high) probability of being toxic, respectively, in tests using marine amphipods in North American sediments (Long & McDonald 1998, Long et al. 2000).

Stage 3: Concurrent sediment chemistry and ecotoxicological investigations. Sampling, chemical analyses and ecotoxicological methodologies are described in detail in Spyrakis (2002) and McCready et al. (2004, 2005, 2006a,b,c) and are summarised briefly below. Methods were selected to be similar to and comparable with those used in North American estuarine monitoring programs (Long & Sloane 2005).

To determine the spatial extent of sediment toxicity in Sydney Harbour in Stage 3 of the study, the estuary was divided into areas (strata) based on MERMQ calculations, which allowed for careful stratification of the harbour before committing to more detailed and expensive, chemical–ecotoxicological studies in the third and final tier of investigation. The 5 bays (Homebush Bay, Hen and Chicken Bay, Five Dock, Iron Cove and Rozelle/Blackwattle Bays) exhibiting a strong contaminant gradient were divided into 12 strata characterised by different contaminant classes and sources (e.g. areas dominated by stormwater contamination or different types of industrial discharge). Samples were also selected from areas with intermediate contaminant concentrations (Duck River, Darling Harbour, Long Bay and Sailors Bay) (7 strata), and an additional 2 strata were chosen from areas with low contaminant concentrations (Lane Cove River and Middle Harbour Creek) to determine sediment toxicity in the least impacted locations of the harbour. Samples were collected randomly within each stratum.

Surficial (top 2 to 3 cm) sediment samples (n = 65) were analysed for 12 metals, 21 OCs, 24 PAHs and total organic carbon (TOC). PCBs were quantified as 7 Aroclor mixtures: Aroclors 1016, 1221, 1232, 1242, 1248, 1254 and 1260. Total PCB was the sum of the 7 Aroclors. A certified methodology, based on US EPA Method 8081 was used (USEPA 2000). However, PCB is not a widespread contaminant in Sydney Harbour, and all PCBs in harbour sediments were present as Aroclor 1254 only. Consequently, there are no concerns over summing Aroclors. Analyses were undertaken by a certified laboratory (Australian Laboratory Services, ALS), following US EPA procedures (USEPA 2000). Metals were extracted in nitric acid and analysed by inductively coupled plasma–atomic emission spectrometry or inductively coupled plasma–mass spectrometry. Organic compounds were extracted in 1:1 (v/v) dichloromethane:acetone and analysed using dual column gas chromatography with electron capture detection for organochlorines, and GC–MS for PAHs. TOC analysis was by high temperature evolution.

All samples were laboratory tested in a 10 d, whole-sediment amphipod survival test and a pore water sea urchin fertilisation test to assess both contaminant uptake routes, i.e. via the gills for the dissolved phase and by ingestion for the solid phase. Pore waters were extracted by standard procedures using pressurized squeezing with air (Carr & Chapman 1995). Most pore waters were also tested in sea urchin larval development (n = 61) and microbial bioluminescence Microtox© (n = 57) tests. Amphipod tests were conducted with *Corophium colo*, a euryhaline indigenous species, and the indigenous species *Heliocidaris tuberculata* was used in the sea urchin tests (Spyrakis 2002, McCready et al. 2004). Sample results were compared to negative controls with statistical t-tests using ToxCalc™ software (Version 5.0, Tidepool Scientific Software). There is no universal, single method, test or endpoint in ecotoxicological investigations. In field evaluations of sediment quality, a battery of tests is typically performed to form a weight of evidence with which to classify the samples. The 3 tests used in the current study provided information on 3 phases (partitions) of the sediment, i.e. the solid phase, pore water phase, and an organic solvent extract. The most widely used test of solid phase sediments is with amphipods, and the most frequently used endpoint is survival (Long and Sloane 2005). The most widely used tests for pore waters are on fertilization success of echinoderm gametes (Carr et al. 2003, Long et al. 2003a). Microtox© tests were undertaken based on North American experience, which demonstrated that some sediment that may not be toxic to invertebrates could stimulate a response when bacteria were exposed to organic toxicants in solvent extracts (Long et al. 1996, 1998, 2000). The 3 tests provided endpoints ranging from mortality to abnormal morphological development to fertilization success to metabolic activity, i.e. the tests were wide-ranging and not redundant or duplicative. Together, these tests give a broad perspective of adverse responses to mixtures of toxicants to which resident benthos may be exposed. The 3 tests were also chosen to ensure that the results were as comparable as possible to the North American baseline data.

Sediment chemistry and toxicity scoring matrix. Samples were ranked using a weight-of-evidence (WOE) matrix for sediment chemistry, toxic response and combined chemistry–toxicity, as described previously by Chapman et al. (2002), Long et al. (2003b), Adams et al. (2005), Long et al. (2005a,b), Long & Sloane (2005) and Simpson et al. (2005). The WOE approach tends to give equal weight to the chemistry and toxicity and/or benthic impairment data (Table 1).

Two previously used approaches were employed to assess sediment chemistry. Scores of 1 to 4 were given to the four MERMQ ranges of <0.1, 0.1 to 0.5. 0.5 to 1.5 and >1.5, respectively (Long & McDonald 1998, Long et al. 2000). The second scheme utilised the number of SQGs exceeded in each sample, i.e. for samples in which 1 or more ERLs but no ERMs were exceeded, a score of 1 was given, samples exceeding 1 to 5 ERMs scored 2, samples exceeding 6 to 10 ERMs scored 3 and samples exceeding >10 ERMs scored 4 (Long et al. 2000).

Scoring for sediment toxicity was achieved by using 2 approaches that have also previously been employed. The first approach considered toxic response as a percentage of the control, i.e. non-toxic (non-significant response, $p > 0.05$) scored 1, slightly toxic (significant response, $p < 0.05$) scored 2, moderately toxic (response significant and <80% of control affected $p < 0.05$) scored 3, and highly toxic (response significant and <50% of control $p < 0.05$) scored 4. The threshold of 80% of controls (called the minimum significant difference) was derived in power analysis of large ecotoxicological data sets from the 3 US coastlines for amphipods, sea urchins and other species in North America (Thursby et al. 1997, Carr & Biedenbach 1999, Phillips et al. 2001) and is considered to be an ecologically significant threshold that minimises false positives (Thursby et al. 1997, Carr & Biedenbach 1999, Long et al. 2001, Phillips et al. 2001). Scores for

Table 1. Sydney Harbour, Australia: Sediment chemistry and toxicity scoring matrix. MERMQ: mean effects range median quotient; SQGs: sediment quality guidelines. Chemistry and toxicity scores assigned as described in 'Materials and methods'; Category (a) based on MERMQ values, Category (b) is re-scaled toxic response as a percentage of control (adjusted total score), Category (c) is number of toxic and non-toxic responses in all tests

			Chemistry				**Toxicity**			**Combined scores**	
			MERMQ		SQGs			Response as percentage of control	Responses in all tests		
Stratum	Location	Sample no.	Value	Category (a)	No. exceeded	SQG category	Score	Category (b)	Category (c)	Category (a)+(b)	Category (a)+(c)
1	Duck River	60	0.4	2	2	2	7	2	2	4	4
		59	0.4	2	1	2	8	2	2	4	4
2	Parramatta River	58	0.3	2	0	2	11	3	4	5	6
3	Homebush Bay, W	61	1.3	3	8	3	11	3	3	6	6
4	Homebush Bay, E	62	0.6	3	4	2	8	2	3	5	6
		63	0.5	2	4	2	8	2	3	4	5
		38	0.7	3	5	2	11	3	4	6	7
		37	0.4	2	3	2	11	3	4	5	6
		34	1.5	4	7	3	11	3	4	7	8
		32	0.5	3	4	2	11	3	4	6	7
		6	0.3	2	3	2	13	4	3	6	5
		33	0.8	3	3	2	11	3	4	6	7
		7	5.0	4	4	2	9	3	4	7	8
		39	1.0	3	3	2	7	2	2	5	5
		36	0.5	3	3	2	10	3	3	6	6
		35	1.3	3	4	2	8	2	2	5	5
5	Hen and Chicken Bay, S	5	0.5	2	4	2	8	2	4	4	6
		8	0.5	3	4	2	12	3	4	6	7
		9	0.6	3	4	2	11	3	3	6	6
6	Hen and Chicken Bay, N	10	0.5	2	4	2	11	3	3	5	5
		15	0.4	2	4	2	12	3	4	5	6
		14	0.3	2	3	2	12	3	4	5	6
		13	0.4	2	3	2	12	3	4	5	6
		11	0.5	2	3	2	11	3	3	5	5
		12	0.4	2	3	2	12	3	4	5	6
7	Five Dock	4	1.3	3	3	2	12	3	3	6	6
8	Lane Cove River	55	0.2	2	0	2	8	2	3	4	5
		56	0.2	2	0	2	7	2	2	4	4
9	Iron Cove, SW	18	2.1	4	13	4	7	2	2	6	6
		17	2.1	4	13	4	6	2	2	6	6

Table 1. (continued)

			Chemistry					Toxicity		Combined scores	
			MERMQ		SQGs			Response as percentage of control	Responses in all tests		
Stratum	Location	Sample no.	Value	Category (a)	No. exceeded	SQG category	Score	Category (b)	Category (c)	Category (a)+(b)	Category (a)+(c)
		85	2.0	4	15	4	6	2	2	6	6
		86	1.7	4	15	4	8	2	2	6	6
		87	1.9	4	16	4	8	2	2	6	6
		88	1.6	4	12	4	6	2	2	6	6
10	Iron Cove, SE	54	0.9	3	8	3	9	3	3	6	6
		23	1.0	3	9	3	8	2	2	5	5
		22	1.5	3	13	4	7	2	2	5	5
		16	1.3	3	12	4	6	2	2	5	5
11	Iron Cove, N	21	1.1	3	9	3	6	2	2	5	5
		20	1.1	3	9	3	6	2	2	5	5
		19	1.1	3	11	4	6	2	2	5	5
12	Rozelle Bay, W	40	1.6	4	14	4	8	2	2	6	6
		24	2.1	4	13	4	9	3	2	7	6
		84	2.6	4	14	4	12	3	3	7	7
		83	2.3	4	14	4	11	3	3	7	7
		82	2.3	4	13	4	10	3	3	7	7
		81	1.9	4	13	4	10	3	3	7	7
		42	0.9	3	7	3	8	2	2	5	5
		41	1.3	3	12	4	11	3	3	6	6
13	Rozelle & Blackwattle Bays	25	0.6	3	4	2	6	2	2	5	5
		43	0.6	3	4	2	10	3	3	6	6
		26	0.4	2	3	2	8	2	2	4	4
14	White Bay	27	0.2	2	1	2	7	2	2	4	4
15	Blackwattle Bay	31	0.4	2	3	2	7	2	2	4	4
16	Cockle Bay	30	0.5	2	3	2	7	2	2	4	4
		29	0.3	2	1	2	7	2	2	4	4
17	Darling Harbour	28	0.2	2	1	2	6	2	2	4	4
18	Middle Harbour Creek	48	0.2	2	0	2	7	2	2	4	4
		47	0.2	2	0	2	9	3	3	5	5
		49	0.2	2	0	2	7	2	2	4	4
		57	0.3	2	0	2	11	3	4	5	6
19	Sailos Bay	50	0.9	3	10	3	7	2	2	5	5
20	Long Bay, N	44	1.0	3	11	3	9	3	3	6	6
		45	0.9	3	10	3	12	3	4	6	7
21	Long Bay, S	46	1.4	3	12	3	8	2	2	5	5

each test were added up and the overall toxicity score was then re-scaled to give a final score of 1 to 4 (Grapentine et al. 2002, Reynoldson et al. 2002, California State Water Resource Control Board 2006). The second approach for assessing overall toxicity considered the number of toxic and non-toxic responses in all tests, i.e. samples with non-toxic results in all tests or no more than one slightly toxic result scored 1, more than one slightly toxic result and no more than one moderately toxic result scored 2, more than one moderately toxic result and no more than one highly toxic result scored 3, and more than one highly toxic result scored 4. This scheme is similar to WOE approaches in other toxicological investigations (Chapman et al. 2002, Chapman & Anderson 2005, Simpson et al. 2005) (Table 1). An example of the results of these 2 chemistry and 2 toxicity assessment schemes is provided in Table 2 for 3 of the 21 strata.

Equal weight was given to the chemistry and toxicity data in the combined chemical–toxicological assessment, because the intent was to show spatial patterns and incidences of chemically mediated effects on benthic organisms as per recommendations of the California State Water Resource Control Board (2006). The combined assessment was made by summing the chemistry and toxicity scores and recalibrating the resulting total score into 4 categories (non-degraded, slightly degraded, highly degraded, and severely degraded). The mean of the summed scores was calculated for all samples within each stratum and placed into categories 1 to 4 in the final assessment of combined chemistry and toxicity.

RESULTS

Stage 1: Sediment chemistry

Studies undertaken in the first stage of assessment (Birch & Taylor 1999, 2000, McCready et al. 2000) showed that sediments of Sydney Harbour contained some of the highest reported concentrations of a wide range of contaminants (Table 3; data rearranged from McCready et al. 2006a). Contaminant concentrations increased markedly landward in

Table 2. Sydney Harbour, Australia: Example of detailed calculation of chemistry and toxicity scores for numbered samples from 3 selected strata. MERMQ: mean effects range median quotient, SQGs: sediment quality guidelines. Statistical tests: (ns) = not significant, (*) = slightly toxic (significant response, $p < 0.05$), (**) = moderately toxic (response significant and <80% of control), (***) = highly toxic (response significant and <50% of control). Chemistry and toxicity scores assigned as described in 'Materials and methods'. Data rearranged from McCready e al. (2006a)

	Iron Cove						Lane Cove			Rozelle Bay West						
	18	17	85	86	87	88	55	56	40	24	84	83	82	81	42	41
SEDIMENT CHEMISTRY																
MERMQ 25 Chemicals	2.07	2.12	1.99	1.68	1.94	1.63	0.18	0.21	1.57	2.1	2.59	2.27	2.29	1.9	0.88	1.25
Number of ERMs exceeded	13	13	15	15	16	12	0	0	14	13	14	14	13	13	7	12
TOXICITY (% control-adjusted response)																
Amphipod survival	97	106	101	96	97	103	107	109	81	82	65	53	78	54	89	80
Microtox© (pore water)	90	82	81	86	80	83	76	76	87	92	89	90	89	90	89	67
Sea urchin larval development	96	98	83	94	60	98	97	97	102	97	75	77	55	50	99	108
Sea urchin fertilisation	92	94	102	78	81	91	66	91	13	27	35	67	88	80	11	23
Statistical tests																
Amphipod survival	ns	ns	ns	ns	ns	ns	ns	ns	ns	*	**	**	**	**	ns	**
Microtox© (pore water)	*	*	*	*	*	*	**	**	*	*	*	*	*	*	*	**
Sea urchin larval development	*	ns	*	*	**	ns	ns	ns	ns	ns	**	**	**	**	ns	ns
Sea urchin fertilisation	*	*	ns	**	*	*	**	*	***	***	***	**	*	*	***	***
SCORES																
Chemistry																
MERMQ	4	4	4	4	4	4	2	2	4	4	4	4	4	4	3	3
Number of SQGs exceeded	4	4	4	4	4	4	2	2	4	4	4	4	4	4	3	4
Toxicity																
Response as percentage of control	2	2	2	2	2	2	2	2	2	3	3	3	3	3	2	3
Responses in all tests	2	2	2	2	2	2	2	3	2	2	3	3	3	3	2	3

the harbour, with high concentrations in the upper parts of embayments as well as in the western tributaries of Middle Harbour. Contaminant sources were related mainly to current and past stormwater discharge or historical dumping by industry (Birch & Taylor 2004). Stormwater-derived contaminants exhibited similar distributions for all 4 major contaminant classes (metals, OCs, PAHs and PCBs) and were highest in the upper reaches of embayments close to major stormwater inputs, especially on the southern shores of the middle estuary. The authors suggest that the concentrations were elevated due to proximity to the source, a mainly muddy substrate, and poor flushing by tides and currents. Discrete contaminant 'hot spots' were associated with discharge from specific industries located on the shores of the estuary.

Stage 2: Assessment of sediment quality using sediment quality guidelines

In the second stage of assessment, single chemicals and contaminant mixtures were determined in areas of the harbour exceeding SQGs (Birch & Taylor 2002a,b,c) to better prioritize these areas for more detailed chemical–toxicological investigations (Stage 3). Areas of Sydney Harbour with sediment exceeding ERM values for Cu, Pb and Zn, which were the most prevalent contaminants in the Harbour, represented approximately 2, 50, and 36% of the estuary, respectively. Sediment in all of the harbour, except a small area near the entrance, exceeded ERL values for at least one metal.

Organochlorine pesticides exceeded ERM values over extensive parts of Sydney Harbour, including the lower estuary, whereas sediments in only a small part of the port had PCB concentrations above the ERM value (Birch & Taylor 2002a,b,c). Based on guidelines, concentrations for at least one OC or PAH compound in sediments in almost all upper and middle parts of Sydney Harbour, including Middle Harbour, exceeded ERM values.

The MERMQ approach was used to determine the probability of sediment toxicity as undertaken in North America (Long et al. 2000) and Australia (ANZECC/ARMCANZ 2000, McCready et al. 2006a,b,c) and to classify Sydney Harbour into 'priority areas'. Sediments in the highest priority class (category 4) were located mainly in central estuary embayments (Iron Cove, Rozelle Bay) (Fig. 2) and category 3 sediments mantled the Parramatta River and embayments of the central estuary (Homebush Bay, Iron Cove, Five Dock, Rozelle and Blackwattle Bays). Category 2 sediments were located in the main channel of the central and lower harbour, Lane Cove and Middle Harbour, whereas the mouth of the harbour was mantled in category 1 sediment. Category 1, 2, 3 and 4 sediment corresponding to probable toxicities of approximately 10, 25, 50 and 75% (Long et al. 2000) comprised 19, 54, 25 and 2% of the harbour, respectively (Birch & Taylor 2002a,b,c).

Stage 3: Assessment of sediment quality by relating chemical and toxicological data

The final tier of assessment included synoptic sampling and analyses of chemical and ecotoxicological data and is examined in a spatial context for the first time in this paper. MERMQ values ranged from 5.0 for a sample with high DDT concentration in Homebush Bay to 0.2 in the least developed subcatchment in

Table 3. Sydney Harbour. Summary of sediment chemical data for the most prevalent chemicals in 4 classes (Stage 3, n = 65). Concentrations are expressed as dry weight and, mean values are rounded to 2 significant figures. SQGs: sediment quality guidelines, ERL: effects range low ERM: effects range-median (SQGs from Long et al. 1995)

Chemical	Concentration Minimum	Maximum	Mean	SQGs ERL	ERM	% samples amongst ranges of SQGs <ERL	ERL–ERM	>ERM
Metals (mg kg^{-1})								
Copper	20	701	210	34	270	2	75	23
Lead	78	1050	390	46.7	218	0	29	71
Zinc	75	8820	900	150	410	3	18	78
PAHs (µg kg^{-1})								
Benzo(*b*)fluoranthene	119	12800	3100	–	–	–	–	–
Fluoranthene	121	16200	4300	600	5100	11	60	29
Pyrene	161	23300	5100	665	2600	11	45	45
OCs (µg kg^{-1})								
Chlordane (cis- + trans-)	<0.5	44.2	2.8	–	–	–	–	–
Dieldrin	<0.5	102	7.2	–	–	–	–	–
Hexachlorobenzene	<0.5	1620	37	–	–	–	–	–
Total DDT	<0.5	5168	150	1.58	46.1	38	34	28
Total PCB (µg kg^{-1})	<5	514	40	22.7	180	83	11	6

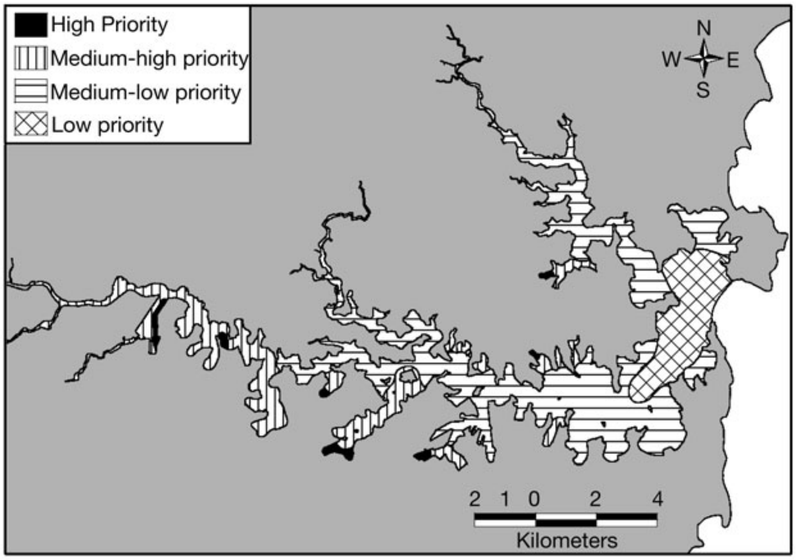

Fig. 2. Four priority categories for Mean effects range median quotients (MERMQ) based on earlier (Stage 2) chemical investigation. Low, Medium-low, medium-high and high priority categories related to MERMQ values of <0.1, 0.1–0.5, 0.5–1.0 and >1.5, respectively

upper Middle Harbour (Fig. 3). Most of eastern Homebush Bay (Stratum 4), south west Iron Cove (Stratum 9) and Rozelle Bay (Stratum 12) were included in the highest MERMQ (>1.5) category (i.e. score 4) (Fig. 4). Western Homebush Bay (Stratum 3), southern Hen and Chicken Bay (Stratum 5), northern and eastern Iron Cove (Strata 10 and 11), Five Dock (Stratum 7), Sailors Bay (Stratum 19), Long Bay (Strata 20 and 21) and central Rozelle Bay (Stratum 13) scored 3 (MERMQ 0.51 to 1.5). All samples in other strata scored 2 (MERMQ 0.1 to 0.5), including the strata in areas with the least anthropogenic influence, i.e. Lane Cove River (Stratum 8) and Middle Harbour Creek (Stratum 18). Assessment of sediment chemistry in the first and third tiers of investigation was spatially similar (Figs. 2 & 3).

In calculating quotients, the contribution of individual chemicals to the assessment is reduced through the averaging process and thus a sample containing a single chemical with a high concentration may result in a low computed mean ERM quotient and a low predicted probability of toxicity. Several areas in Sydney Harbour exhibit such characteristics. Sites with sediment concentrations of individual contaminants that exceeded ERMs by >5 times or where >10 chemicals exceeded ERMs have been identified on the figure depicting chemistry scores (Fig. 4). These locations were identified to determine whether

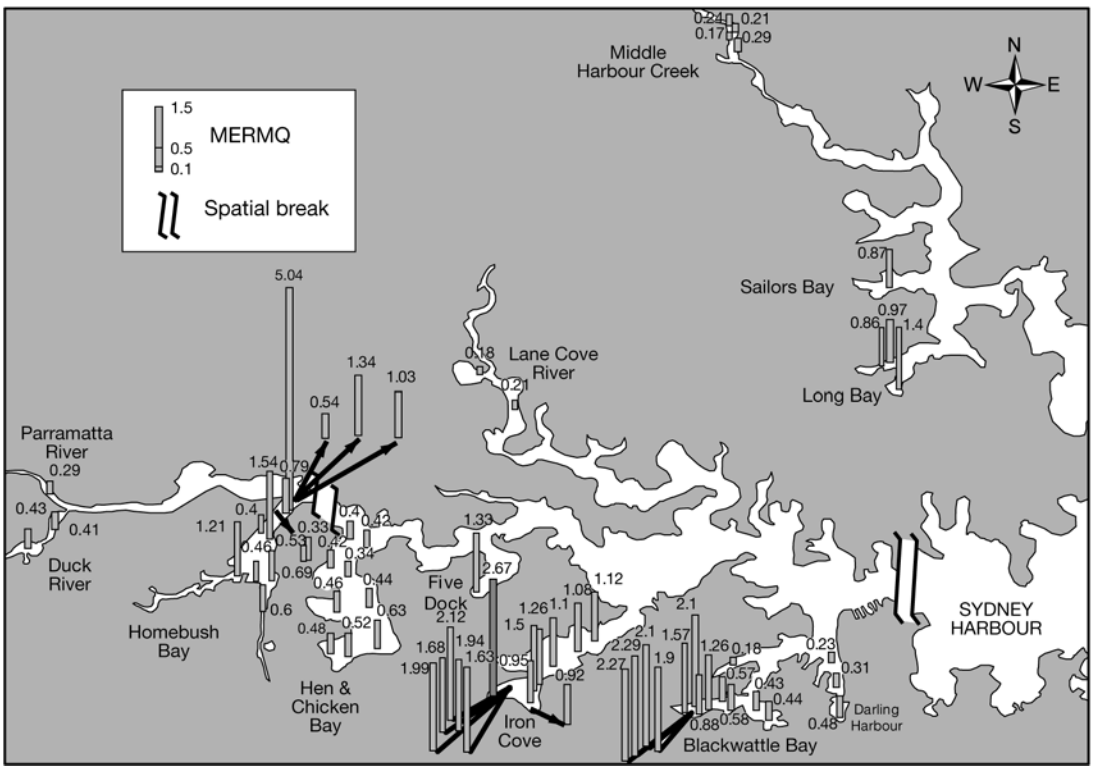

Fig. 3. Mean effects range median quotients (MERMQ) for the combined chemical–ecotoxicological investigation (Stage 3)

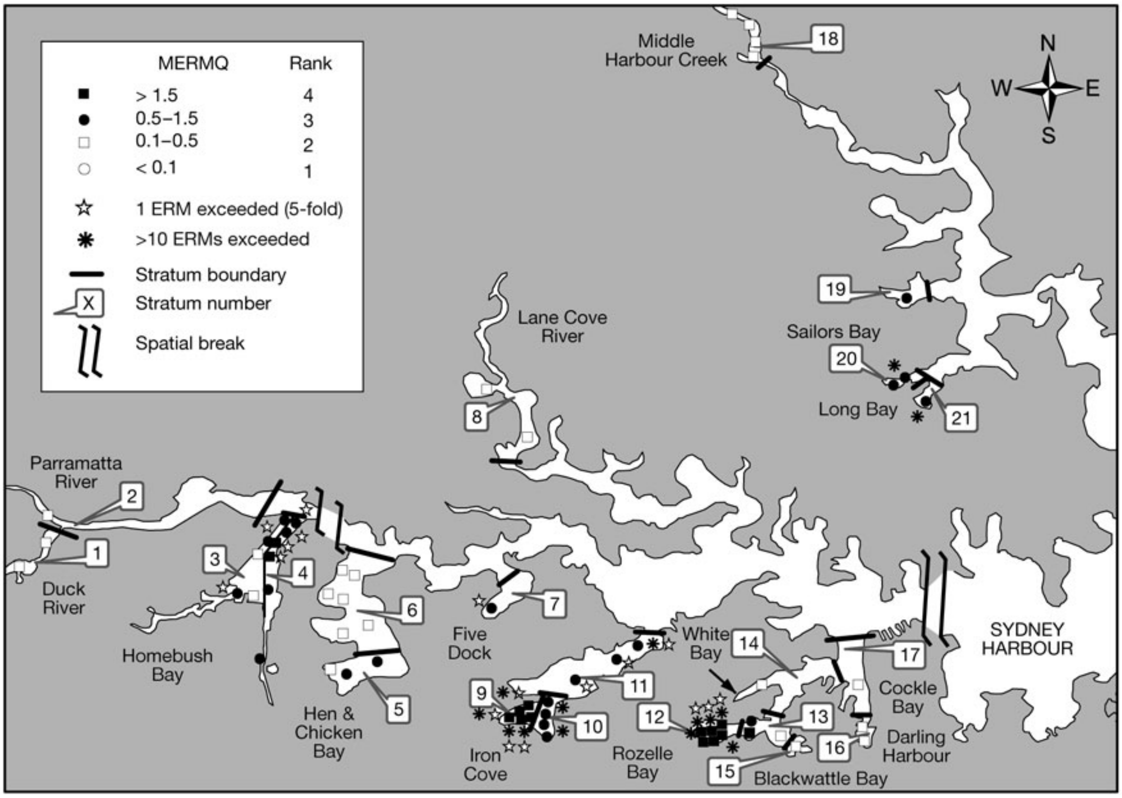

Fig. 4. Chemistry scores based on the mean effects range median quotient (MERMQ) approach. Scores of 1 to 4 were given to low, medium-low, medium-high and high categories, respectively

sediment in these areas exhibited toxicity in excess of predictions based on calculated MERMQ values.

Our assessments of the sediment chemistry data based on the numbers of SQGs exceeded produced results very similar to the outcomes of the MERMQ values. This observation is consistent with those reported for North American estuarine sediments (e.g. Long et al. 2000).

Sediment toxicity

In the current study, amphipod survival, Microtox© pore water, sea urchin larval development and sea urchin fertilisation tests showed sediments in Sydney Harbour to be toxic ($p < 0.05$) relative to negative controls in 17, 98, 59, and 98% of the cases, respectively. All samples were toxic in at least one test. In the sea urchin fertilization tests, 2% of the samples had a score of 1 (non-toxic), 31% had a score of 2 (slightly toxic), 18% had a score of 3 (moderately toxic), and 49% of samples were highly toxic with a score of 4.

In Sydney Harbour, the toxic response in the sea urchin fertilisation test was more extensive than that in the other tests (Fig. 5). Sediment in the majority of Parramatta River (Stratum 2), the southern embayments of the central harbour (Strata 3, 4, 5, 6 and 7) and Long Bay (Strata 20 and 21) was highly toxic in the sea urchin fertilisation test (significantly different from control ($p < 0.05$), and mean response <50% of control), whereas sediment in Iron Cove east (Stratum 10), Blackwattle Bay (Stratum 15) and Upper Middle Harbour (Stratum 18) was moderately toxic (significantly different from control ($p < 0.05$), and mean response 50 to 80% of control). Sediment in the remaining strata (Strata 3, 9, 11, 14, 16 and 17) were toxic (significantly different from control, and mean response ≥80% of control).

Only 2 sediment samples (in Homebush Bay east and Hen and Chicken Bay) in the present investigation were highly toxic in the amphipod survival test (Fig. 6). Sediment in limited parts of Homebush Bay east (Stratum 4), Five Dock (Stratum 7) and Rozelle Bay (Stratum 12) was moderately toxic in this test. Sediment from isolated sites in Rozelle and Long Bays was slightly toxic in the amphipod test and sediment in the remaining locations was non-toxic.

The Microtox© test of pore water produced less discriminative results (mean response 67 to 99% of controls) than the other 3 tests in the current study. Sediment from areas with high contaminant concentrations, especially metals, e.g. Iron Cove, Rozelle Bay

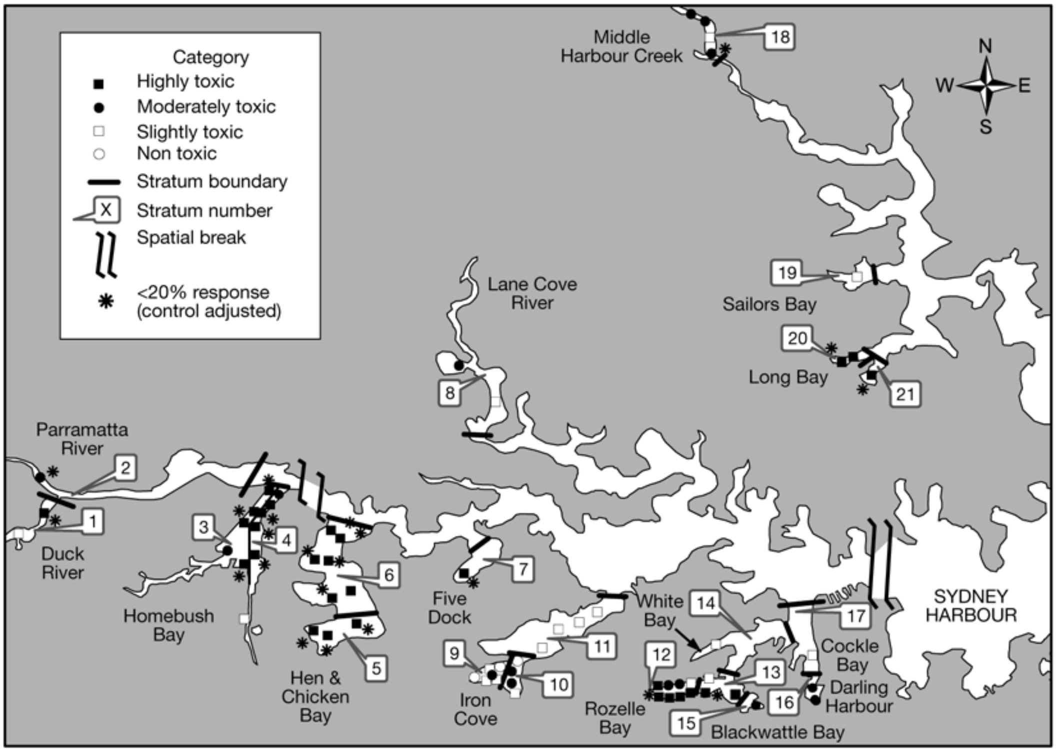

Fig. 5. Sea urchin fertilisation test. The scoring scheme considered toxic response as a percentage of control, i.e. non-toxic (non-significant response) scored 1, slightly toxic (significant response, $p < 0.05$) scored 2, moderately toxic (response significant and <80% of control) scored 3, and highly toxic (response significant and <50% of control) scored 4. Scores for each test were added together and the overall toxicity score was then re-scaled to give a final score of 1 to 4

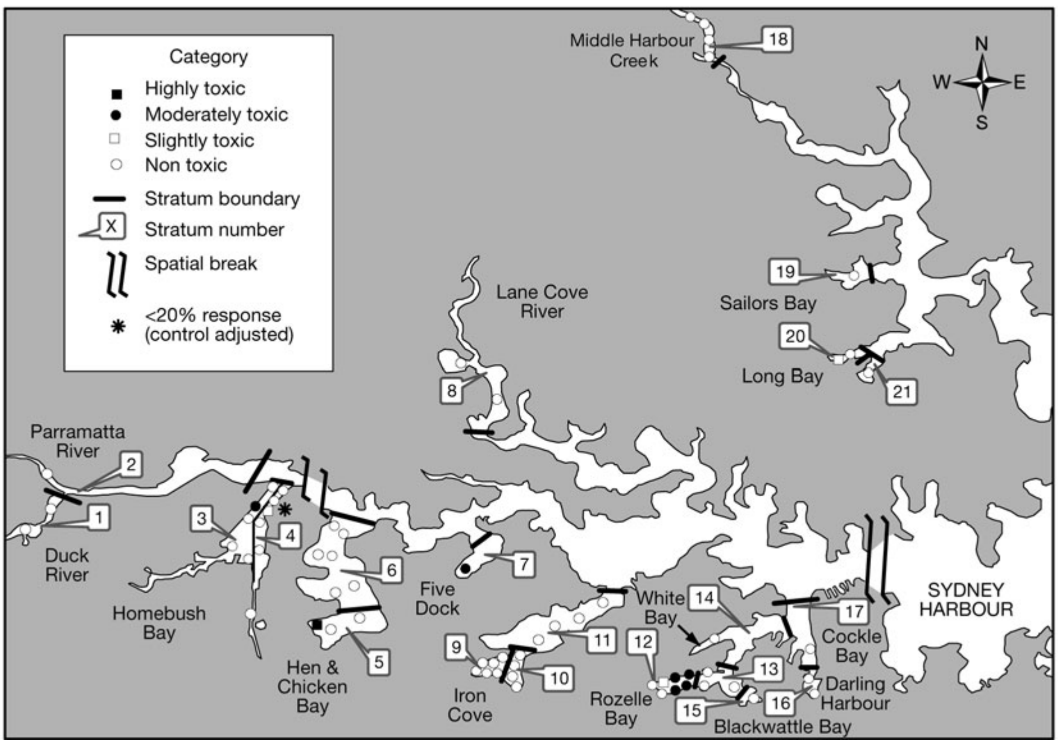

Fig. 6. Amphipod survival test. Scoring scheme as for Fig. 5

and Five Dock, showed high toxicity, while toxicity decreased for sediment samples with lower concentrations (e.g. White Bay, Cockle Bay). The sea urchin larval development test was less sensitive (~40% non-toxic responses) than the fertilisation test (~2% non-toxic responses), but spatial trends were similar.

Scoring overall toxicity by summing the scores for individual tests and dividing the sum by the number of tests resulted in most samples from Homebush Bay, Hen and Chicken Bay, Rozelle Bay and Long Bay being allocated a score of 3, whereas the majority of the remaining samples scored 2 (Fig. 7). There were no samples in the first category (score 1) and only one sample scored 4 (Homebush Bay) (Table 1). Scoring overall toxicity using the second approach, which considered the number of toxic and non-toxic responses in all tests (similar to Chapman et al. 2002, Chapman & Anderson 2005), resulted in more samples (15) being assigned to the highest category (score 4), while distributions were spatially similar and no samples scored 1 (Table 1).

DISCUSSION

Spatial relationships between sediment chemistry and toxicity in Sydney Harbour

Areas with high sediment concentrations of multiple chemical contaminants showed the highest toxic responses in both amphipod and sea urchin fertilisation tests. Amphipod mortality steadily increased with increasing MERMQs, as reported in North America (Long et al. 2000, 2006, Long & Sloane 2005). Iron Cove west (Stratum 9) and east (Stratum 10) were exceptions to the trend of increasing toxicity with increasing chemical concentration. Average MERMQs for these 2 strata were 1.9 and 1.2, respectively, yet only 1 site in south-west and 2 sites in east Iron Cove were moderately toxic in the sea urchin fertilisation test. One site in the west was non-toxic and the remaining sites were all only slightly toxic in this test. All sites in both east and west Iron Cove were non-toxic in the amphipod test. High MERMQs in sediments from south Iron Cove were driven by high PAH quotients (mean 2.6). Recent investigations have revealed that PAH compounds in these sediments are pyrogenic in nature and that this type of material may be poorly bioavailable (McCready et al. 2000, Golding et al. 2007).

Sediment in south Hen and Chicken Bay (Strata 5 and 6) showed the highest Cu concentrations (600 to 700 µg g^{-1} and 2 to 3 times the ERM value) in Sydney Harbour (Irvine & Birch 1998, Birch & Taylor 1999) along with moderate Zn concentrations (700 to 900 µg g^{-1}; 1.5 to 2 times the ERM value). Toxicity response in the sea urchin fertilisation and larval development tests in this bay was the highest in the harbour. All samples in the bay were highly toxic in the fertilisation test and most sites were highly toxic in the larval development test. However, only one sample was moderately toxic in the amphipod test and the remaining samples were non-toxic. This bay was an example of observed toxic responses being greater than predicted by the MERMQs. It is common to observe a greater toxic response in pore water tests than in solid phase tests (Long et al. 1996, Stronkhorst et al. 2001, Bay et al. 2007) because the gametes used for these tests have not developed the mechanisms to metabolize and detoxify toxicants to which they are exposed, and toxicants dissolved in pore waters are ultimately bioavailable. There is empirical evidence that urchin gametes are more sensitive to the presence of trace metals than to organic compounds dissolved in the water. Metals are highly enriched in Port Jackson sediments, and in this case sedimentary Cu is greatly elevated in Hen and Chicken Bay.

Sediment in Five Dock (Stratum 7) contained high concentrations of Zn (8800 µg g^{-1}) and moderate amounts of Cu (180 µg g^{-1}) and Pb (600 µg g^{-1}), however, low concentrations of other contaminants resulted in a MERMQ of 1.3. Mean control-adjusted responses in the sea urchin fertilisation and amphipod survival tests were 13% fertilisation and 62% survival, respectively. Although the Zn concentration at this site was 22 times the ERM value (410 µg g^{-1}), the sediment responded at a toxicity level (total toxicity score of 3) corresponding to the MERMQ value (category 3) for combined chemical mixtures. A site in Homebush Bay had DDE concentrations 4 times higher than the ERM value, but mean ERM quotients were <1.5 in most of the bay and the mean toxicity score was 2. These responses indicate that high concentrations of single chemicals may not be accurate predictors of toxicity, and that the MERMQ approach may not always accurately characterize the status of contaminant mixtures in sediment. A similar evaluation of potential toxicity and bioavailability has been noted for Cr in sediments associated with chromite ore in North America (Becker et al. 2006).

Sampling undertaken adjacent to one of the least developed subcatchments of Lane Cove had the lowest average MERMQ (0.2) and the sediments were non-toxic in the amphipod survival and the larval development tests, but toxic to moderately toxic in the sea urchin fertilisation test. The other low-impact areas sampled, i.e. in Middle Harbour Creek (average MERMQ 0.2), were non-toxic in the amphipod survival test for all 4 sites, but sea urchin fertilisation and larval development tests varied from highly to moderately toxic. Sediment with the lowest contaminant con-

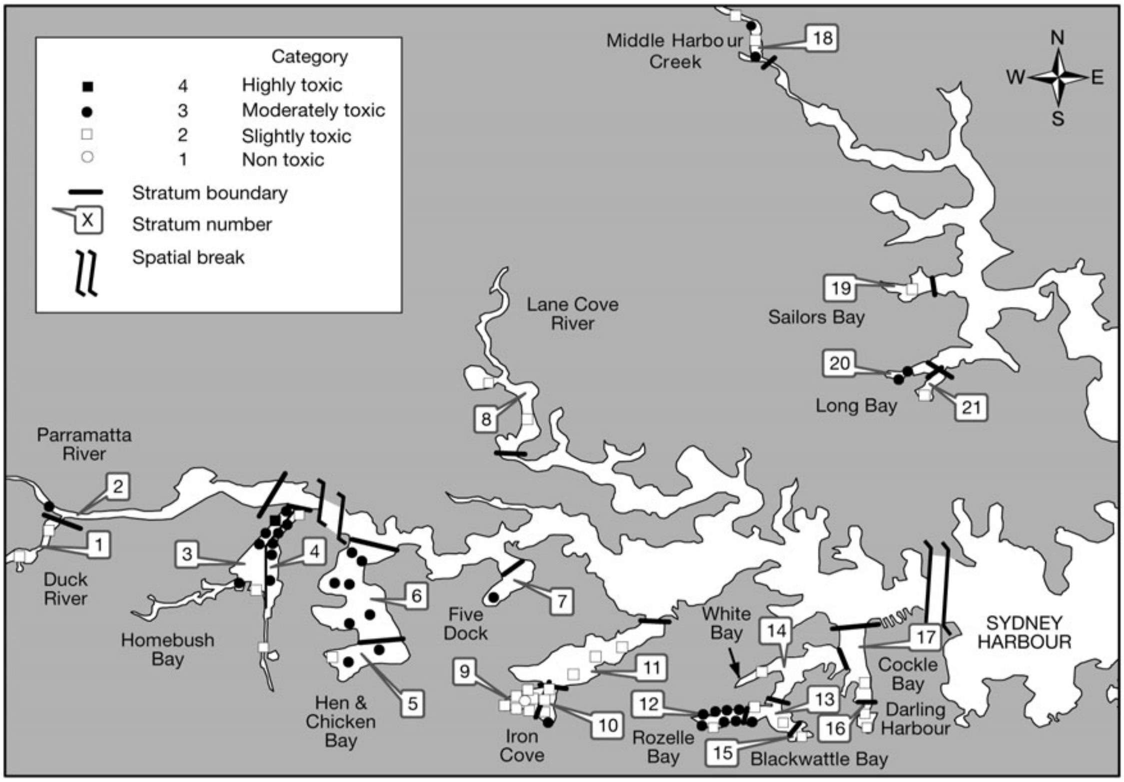

Fig. 7. Overall toxicity scores for 4 tests based on a scheme which considered toxic response as a percentage of control, i.e. non-toxic (non-significant response) scored 1, toxic (significant response, $p < 0.05$) scored 2, highly toxic (response significant and <80% of control) scored 3, and severely toxic (response significant and <50% of control) scored 4. Scores for each test were added together and the overall toxicity score was then re-scaled to give a final score of 1 to 4

centrations in the harbour (MERMQ 0.3) had toxic responses in the sea urchin tests, and samples with slightly elevated PAH and metal concentrations showed a higher toxic response, which was also predicted by the SQGs. In North America, tests performed with echinoderm embryos are almost always more sensitive than those conducted with adult amphipods (Long et al. 1996, Long & Sloane 2005).

Combined sediment chemistry and toxicity scores

The highest combined chemistry and toxicity score using MERMQ and percentage toxic response approaches was observed for sediment in Rozelle Bay (Stratum 9), followed by Long Bay (Stratum 20), Homebush Bay east (Stratum 4) and Hen and Chicken Bay south (Stratum 5) (scores of >3) (Fig. 8). Sediment in Iron Cove west (Stratum 9) had combined chemistry and toxicity scores of 3. Samples in Five Dock (Stratum 7), Iron Cove east and north (Strata 10 and 11), as well as Hen and Chicken Bay north (Stratum 6) had combined scores of >2 (moderately degraded). Remaining areas were slightly degraded, but no strata had combined scores of <2 (non-degraded) (Fig. 8).

Original prioritisation compared to final combined chemistry–toxicity assessment

The spatial trends in the original prioritisation of the harbour, based on the MERMQs produced in the second tier of investigation, closely predicted outcomes obtained in the final combined chemical–ecotoxicological study. Homebush Bay, Long Bay and Rozelle Bay were in the highest category (i.e. most degraded) in both assessments. Other category 4 areas (high priority areas, Fig. 2) in the earlier MERMQ study, i.e. Neutral Bay and the main channel upstream of Hen and Chicken Bay, were not included in the third tier of investigation due to inaccessibility. South Iron Cove was classified in the highest category in the earlier assessment, but scored 3 in the final assessment, probably due to poorly bioavailable, pyrogenic PAHs (Gustafsson et al. 1997, McCready et al. 2000, Golding et al. 2007). Hen and Chicken Bay south was included in category 3 (medium–high priority) in the earlier assessment based on MERMQs and in the highly degraded category in the final appraisal, due to the fact that sediment toxicity was greater than predicted. Five Dock, north Iron Cove, Blackwattle Bay and Hen and Chicken Bay north were classified in category 3 in

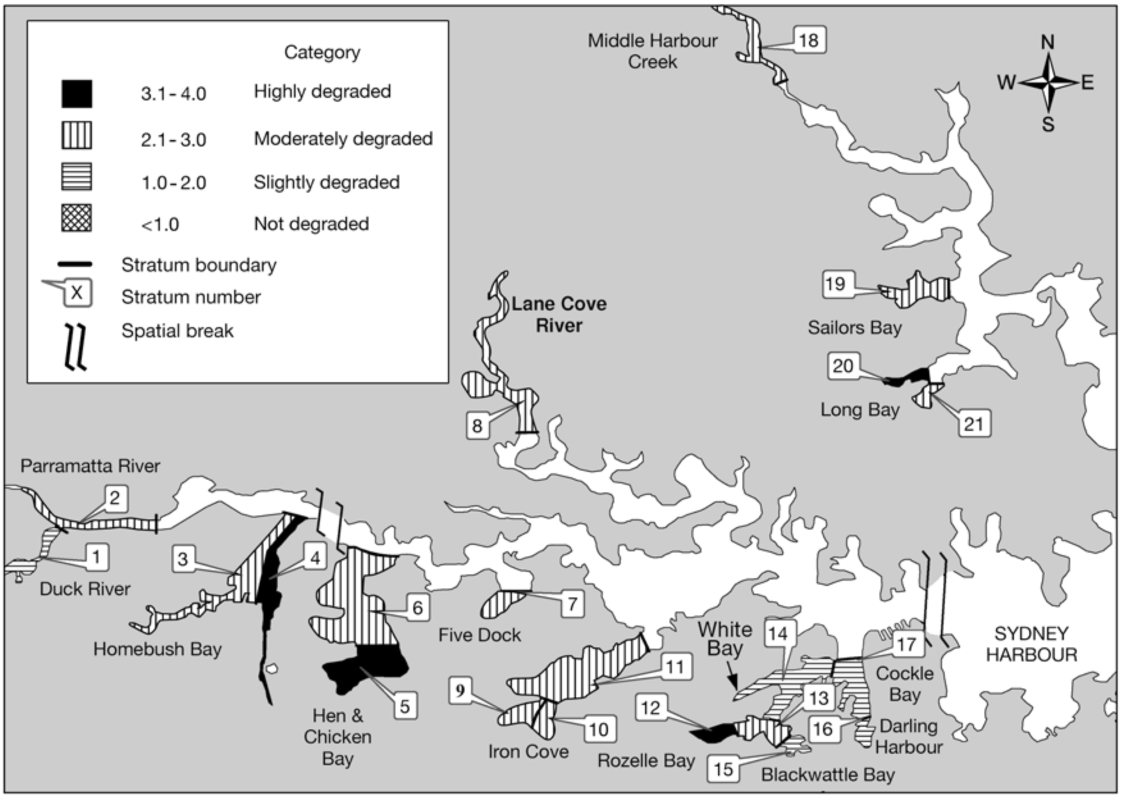

Fig. 8. Combined chemistry and toxicity categories by stratum were created by summing the chemistry and toxicity scores and recalibrating the total score into 4 categories. The mean of summed scores was calculated for all samples within each stratum and placed into categories 1 to 4 (not degraded, slightly degraded, moderately degraded, and highly degraded) in the final assessment

both assessments (medium–high priority based on MERMQs and moderately degraded), while White Bay, Cockle Bay and Darling Harbour were included in category 2 in both assessments (medium–low priority based on MERMQs and slightly degraded). The earlier MERMQ assessment did not predict any of the sediments in the investigated areas to be non-toxic, which was verified in the final chemical–ecotoxicological study. The current study showed that the ability of the MERMQ approach to predict sediment toxicity was good, which provides confidence in its use as a reliable screening technique.

The hierarchical approach

The hierarchal approach allowed the final tier of investigation to focus on high-priority areas (only 8.2 km^2 or 16%) of the harbour. The GIS-referenced database provided valuable information on concentration gradients and types of chemicals present in the sediments, which assisted in stratifying the estuary into priority areas and thus reduced the costs associated with the ecotoxicological investigation. The areas assigned to highly toxic, moderately toxic and slightly toxic categories comprised 17, 52 and 31% of the investigated region, respectively. These results compared well with the original assessment using MERMQs during Stage 2, which subdivided the same area into 15, 54 and 31% for high, medium–high and medium–low priority classes, respectively.

Global perspective

Only 16% of the Port Jackson estuary was investigated in the present study, which makes it difficult to compare the present results with comprehensive North American data sets for sediment toxicity. However, the majority of the highly urbanised areas of the harbour were included in the current work, and if it is assumed that the remaining regions are non-toxic, then highly toxic, moderately and slightly toxic areas mantle 2.7, 8.3 and 5% of the total area of the estuary. In a survey of 1543 samples using a variety of toxicological tools and covering 7300 km^2 in 25 estuaries from the 3 North American coasts, 7% of the combined area was toxic (Long 2000), which is not dissimilar to the 11% toxic plus moderately toxic proportion of Sydney Harbour in the current study. In a separate study of 1068 estuarine samples, also from the 3 US coasts (Long et al. 1998), concentrations of at least one chemical exceeded an ERM value in 27% of the samples, at least one ERL value, but no ERM values were exceeded in 42% of the samples, and chemical concentrations in 31% of the samples did not equal or exceed any ERLs. This can be directly compared to the whole of Sydney Harbour, where chemical concentrations in the sediment in 8% of the estuary did not exceed any ERL values, 46% of the port had sediment exceeding at least one ERM, and sediment in 11% of the area exceeded more than 6 ERMs (Birch and Taylor 2002b).

CONCLUSIONS

In this first spatial investigation of sediment toxicity in Sydney Harbour, patterns of toxicity and chemistry were generally similar, although not entirely consistent. Sediment in areas with the lowest MERMQs showed the lowest toxic responses and sediments in regions with the highest MERMQs were most toxic. No sediment in any area of the harbour was non-toxic in all 4 tests, a result that was predicted from the MERMQs (minimum of 0.2) calculated in the first tier of investigation. However, sediment in Iron Cove contained high concentrations of metals and PAHs, but the toxicity response was less than expected, possibly due to the presence of less bioavailable, pyrogenic PAHs. MERMQs for combined contaminants were low (0.4 to 0.5) in sediments mantling Hen and Chicken Bay; however, toxic response was higher than expected in these copper-rich sediments. These results demonstrate that the empirical approach to assessment of sediment quality developed in North America produced acceptable outcomes in a different part of the world with very different conditions.

The toxic response of the indigenous amphipod species *Corophium colo* used in the current study in whole sediment tests was lower than predicted from SQGs. However, combining these results with that of the more sensitive sea urchin tests for pore water resulted in total toxicity scores that were close to those predicted by the chemistry scores. Sediment toxicity predicted using MERMQs in the second tier of investigation was similar to results of the final tier using combined chemical–ecotoxicological data. Sediments in all parts of Sydney Harbour were toxic in at least one test, as was predicted in the first and second tiers of the investigation using MERMQs. These observations of toxicity, based on the SQGs, matched predictions of toxicity as well as can be expected, and were comparable to the observations made in North America using different species but similar sampling and testing methods.

A hierarchical approach allowed areas of interest in the harbour to be identified, thereby reducing the expensive chemical–ecotoxicological investigation to <20% of the harbour area. The approach used in the current study is consistent with the recently adopted Australian sediment quality guidelines, in which SQGs are used to screen sites and toxicity data are used in the final assessment of risk (ANZECC/ ARMCANZ 2000). Such an approach has not been applied on an estuary-wide basis in Australia previously. The Australian sediment guidelines do not include other lines of evidence, such as benthic community or bioaccumulation data.

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LITERATURE CITED

Adams WJ, Green A, Ahlf W, Brown SS and others (2005) Using sediment assessment tools and a weight-of-evidence approach. In: Wenning RJ, Batley GE, Ingersoll CG, Moore DW (eds) Use of sediment quality guidelines and related tools for assessment of contaminated sediment. Society of Environmental Toxicology and Chemistry, Pensacola

Ahmad N, Marolt RS (1986) One-step extraction and cleanup procedure for determination of p,p′-DDT, p,p′-DDD, and p,p′-DDE in fish. J Assoc Off Anal Chem 69(4):581–586

ANZECC/ARMCANZ (2000) Australian and New Zealand guidelines for fresh and marine water quality. Volume 1, Section 3.5 – Sediment quality guidelines. Australian and New Zealand Environment and Conservation Council, and Agriculture and Resource Management Council of Australia and New Zealand, Canberra

Bay SD, Greenstein D, Young D (2007) Evaluation of methods for measuring sediment toxicity in California bays and estuaries. Technical Report 503. Southern California Coastal Water Research Project, Costa Mesa, CA

Becker DS, Long ER, Proctor DM, Ginn TC (2006) Evaluation of potential toxicity and bioavailability of chromium in sediments associated with chromite ore processing residue. Environ Toxicol Chem 25:2576–2583

Birch GF, Taylor SE (1999) Source of heavy metals in sediments of Port Jackson estuary, Australia. Sci Total Environ 227:123–138

Birch GF, Taylor SE (2000) The distribution and possible sources of organochlorine residues in sediments of a large urban estuary, Port Jackson, Sydney. Aust J Earth Sci 47:749–756

Birch G, Taylor S (2002a) Possible biological significance of contaminated sediments in Port Jackson, Sydney, Australia. Environ Monit Assess 77:179–190

Birch GF, Taylor SE (2002b) Application of sediment quality guidelines in the assessment of contaminated surficial sediments in Port Jackson (Sydney Harbour), Australia. Environ Manage 29:860–870

Birch GF, Taylor SE (2002c). Assessment of possible sediment toxicity of contaminated sediments in Port Jackson estuary, Sydney, Australia. Hydrobiologia 472:19–27

Birch GF, Taylor SE (2004) Sydney Harbour and catchment: contaminant status of Sydney Harbour sediments: a handbook for the public and professionals. Geological Society of Australia, Environmental, Engineering and Hydrogeology Specialist Group, Sydney

California State Water Resource Control Board (2006) Development of sediment quality objectives for enclosed bays and estuaries. Informational document. State Water Resources Control Board, Division of Water Quality, Sacramento, CA

Carr RS, Chapman DC (1995) Comparison of methods for conducting marine and estuarine sediment pore water toxicity tests — extraction, storage, and handling techniques. Arch Environ Contam Toxicol 28:69–77

Carr RS, Biedenbach JM (1999) Use of power analysis to develop detectable significance criteria for sea urchin toxicity tests. Aquat Ecosyst Health Manage 2:413–418

Carr RS, Long ER, Mondon JA, Montagna PA, Roscigno PF (2003) Porewater toxicity tests in sediment quality triad studies. In: Carr RS, Nipper M (eds) Porewater toxicity testing: biological, chemical, and ecological considerations. SETAC, Pensacola, FL, p 201–228

Chapman PM, Anderson J (2005) A decision-making framework for sediment contamination. Integr Environ Assess Manag 1:163–173

Chapman PM, McDonald BG, Lawrence GS (2002) Weight of evidence issues and frameworks for sediment quality (and other) assessment. Hum Ecol Risk Assess 8: 1489–1516

Golding CJ, Gobas FAPC, Birch GF (2007) Characterization of polycyclic aromatic hydrocarbon bioavailability in estuarine sediments using thin-film extraction. Environ Toxicol Chem 26:829–836

Grapentine L, Anderson J, Boyd D (2002) A decision making framework for sediment assessment developed for the Great Lakes. Hum Ecol Risk Assess 8:1641–1655

Gustafsson O, Haghseta F, Chan C, Macfarlane J, Gschwend PM (1996) Quantification of the dilute sedimentary soot phase: Implications for PAH speciation and bioavailability. Environ Sci Technol 31(1):203–209

Irvine I, Birch GF (1998) Distribution of heavy metals in surficial sediments of Port Jackson, Sydney, New South Wales. Aust J Earth Sci 45(2):297–304

Long ER (2000) Spatial extent of sediment toxicity in U. S. estuaries and marine bays. Environ Monit Assess 64: 391–407

Long ER, MacDonald DD (1998) Recommended uses of empirically derived sediment quality guidelines for marine and estuarine ecosystems. Hum Ecol Risk Assess 4:1019–1039

Long ER, Sloane GM (2005) Development and use of assessment techniques for coastal sediments. In: Bortone SA (ed) Estuarine indicators. CRC Press, Boca Raton, FL

Long ER, MacDonald DD, Smith SL, Calder FD (1995) Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ Manage 19:81–97

Long ER, Robertson A, Wolfe DA, Hameedi J, Sloane GM (1996) Estimates of the spatial extent of sediment toxicity in major US estuaries. Environ Sci Technol 30:3585–3592

Long ER, Field LJ, MacDonald DD (1998) Predicting toxicity in marine sediments with numerical sediment quality guidelines. Environ Toxicol Chem 17:714–727

Long ER, MacDonald DD, Severn CG, Hong CB (2000) Classifying the probabilities of acute toxicity in marine sediments with empirically-derived sediment quality guidelines. Environ Toxicol Chem 19:2598–2601

Long ER, Hong CB, Severn CG (2001) Relationships between acute sediment toxicity in laboratory tests and abundance and diversity of benthic infauna in marine sediments: a review. Environ Toxicol Chem 20, 1: 46–60

Long ER, Carr RS, Montagna PA (2003a) Porewater toxicity tests: value as a component of sediment quality triad assessments. In: Carr RS, Nipper M (eds) Porewater toxicity testing: biological, chemical, and ecological considerations. SETAC, Pensacola, FL, p 163–200

Long ER, Dutch M, Aasen S, Welch K, Hameedi MJ (2003b) Chemical contamination, acute toxicity in laboratory tests, and benthic impacts in sediments of Puget Sound. A summary of results of the joint 1997–1999 Ecology/NOAA survey. Washington State Dept of Ecology Publ No 03-03-048. Dept of Ecology, Olympia, WA

Long ER, Dutch M, Aasen S, Welch K, Hameedi MJ (2005a) Spatial extent of degraded sediment quality in Puget Sound (Washington State, USA) based upon measures of the Sediment Quality Triad. Environ Monitor Assess 111 (1–3):173–222

Long ER, Winger PV, Maruya KA, Otero L, Seal T (2005b) Chemical contamination and toxicity in freshwater sediments of Miami-Dade County canals. Technical Report, Florida Department of Environmental Protection. Tallahassee, FL

Long ER, Ingersoll CG, MacDonald DD (2006) Calculation and uses of mean sediment quality guideline quotients: A critical review. Environ Sci Technol 40:1726–1736

McCready S, Slee D, Birch GF, Taylor SE (2000) The distribution of polycyclic aromatic hydrocarbons in surficial sediments of Sydney Harbour, Australia. Mar Pollut Bull 40: 999–1006

McCready S, Spyrakis G, Greely CR, Birch GF, Long EL (2004) Toxicity of surficial sediments from Sydney Harbour and vicinity, Australia. Environ Monit Assess 96: 53–83

McCready S, Greely CR, Hyne RV, Birch GF, Long ER (2005) Sensitivity of an indigenous amphipod, *Corophium* sp. to chemical contaminants in laboratory toxicity tests conducted with field collected sediment from Sydney Harbour, Australia and vicinity. Environ Toxicol Chem 24: 2545–2552

McCready S, Birch GF, Long ER (2006a) Metallic and organic contaminants in sediments of Sydney Harbour and vicinity — a chemical dataset for evaluating sediment quality guidelines. Environ Int 32:455–465

McCready S, Birch GF, Long ER, Spyrakis G, Greely CR (2006b) An evaluation of Australian sediment quality guidelines. Arch Environ Contam Toxicol 50:306–315

McCready S, Birch GF, Long EL, Spyrakis G, Greely CR (2006c) Predictive abilities of numerical sediment quality guidelines in Sydney Harbour, Australia, and vicinity. Environ Int 32(5):638–649

Phillips BM, Hunt JW, Anderson BS, Puckett HM, Fairey R, Wilson CJ, Tjeerdema R (2001) Statistical significance of sediment toxicity test results: threshold values derived by the detectable significance approach. Environ Toxicol Chem 20:371–373

Reynoldson TB, Thompson SP, Milani D (2002) Integrating multiple toxicological endpoints in a decision-making framework for contaminated sediments. Hum Ecol Risk Assess 8:1569–1584

Simpson SL, Batley GE, Chariton AA, Stauber JL and others (2005) Handbook for sediment quality assessment. Centre for Environmental Contaminant Research, CSIRO, Sydney

Spyrakis G (2002) Assessment of the predictive ability of sediment quality guidelines using sea urchin toxicity tests. MSc Thesis, University of Sydney

Stronkhorst J, Schipper CA, Honkoop J, van Essen K (2001) Disposal of dredged material in Dutch coastal waters. Technical Report RIKZ/2001.030. Ministry of Transport, Public Works, and Water Management, The Hague.

Thursby GB, Heltshe J, Scott KJ (1997) Revised approach to toxicity test acceptability criteria using a statistical performance assessment. Environ Toxicol Chem 16(6): 1322–1329

USEPA (2000) Test methods for evaluating solid wastes, physical/chemical methods (SW-846). CD ROM ver 2.0. US Environmental Protection Agency and US Department of Commerce National Technical Information Service

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