

Equilibrium Partitioning Approach to Define Sediment Quality Guideline of Some Metals in Chao Phraya Estuary, Thailand

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Abstract: This study aims to define numerical sediment quality guideline (SQG) based on equilibrium partitioning approach that represents availability of metal in estuarine sediment. The labile-phase metal was used in calculating the partition coefficient caused by remobilization of this fraction in the sediment-interstitial water system as a result of changing physicochemical condition in sediment. In addition, the metal bound with sulphide fraction was taken into account in calculating the SQG since this fraction is potentially available to interstitial water when oxidation process occurs in anoxic sediment. The SQG values for Chao Phraya estuarine sediment for cadmium, copper, lead and zinc were found in the range of 1.2–3.1, 7–105, 23–86 and 16–125 mg/kg dry weight in <63 μm fraction and CaCO_3 free basis, respectively. However, in order to support the calculated SQG value, toxicity test for metal using the same sediment should be further studied.

Key words: Sediment quality guideline, equilibrium partitioning approach, metal, Chao Phraya estuary.

Introduction

The purpose of this paper is to derivate the sediment quality guideline (SQG) for metals in estuarine sediment by using equilibrium partitioning (EqP) approach. This approach is a theoretically-based SQG that attempts to account for bioavailability of chemicals in sediment (McCauley et al., 2000; Burton, 2002). For metal, it is essential that only metal in labile fraction that is bioavailable metal must be considered in assessing the sediment toxicity. Since, it can be uptaken by benthic

organisms and can cause an adverse biological effect (Ankley et al., 1994).

The EqP approach is based on the several assumptions as follows: (1) the sediment toxicity is principally contorted by chemicals in interstitial water; (2) the sensitivity of both benthic organisms and pelagic organisms to chemicals is similar, thus the water quality criteria is used to define the acceptable effects concentration of chemicals in interstitial water; and (3) the partitioning of chemicals between sediment and intestinal water can be expressed as the partition

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coefficient at equilibrium (Shea, 1988; Di Toro et al., 1991). The general formula in calculating the SQG based on EqP approach is shown in Eq. 1.

$$\text{SQG} = K_D \times \text{WQC} \quad (1)$$

where SQG is sediment quality guideline (mg/kg dry wt.), K_D is partition coefficient (l/kg), and WQC is water quality criteria (mg/l).

The partition coefficient can be directly calculated from the metal concentration in sediment divide by the metal concentration in interstitial water. In practical, the sediment is extracted for analysis of the metal concentration. The difference in extraction methods (both extractants and experimental conditions) employed in extracting the metal in sediment will give difference in degree of metal concentration and thus lead to the difference in K_D value.

However, in this study it was decided that the metal concentration in sediment to be used for the K_D calculation should be the labile-phase metal because this phase can be remobilized into the sediment-interstitial water system by the changes in physicochemical properties of sediment such as redox potential, pH, salinity, etc. (Calmano et al., 1993; Filgueiras et al., 2002). The labile-phase metal mentioned here is the metal associated in non-residual fractions namely exchangeable cation, Fe-Mn oxides and organic matter (Salomons and Förstner, 1984; Chester et al., 1985; and Salomons, 1995).

In order to extract the labile-phase metal from sediment, the 25% (v/v) HOAc was tested in extracting the metal from non-residual fraction against three extractants namely 0.01 M CaCl_2 (Houba et al., 2000), 0.05 M EDTA (Fangueiro, 2002) and 1 M HCl (Snape et al., 2004). It was found that the 25% (v/v) HOAc extractant is the most effective amongst the three in extracting the metal from non-residual fraction. Moreover, Loring and Rantala (1992) reported that the 25% (v/v) HOAc will extract metals in easily exchangeable, amorphous of Fe-Mn oxides, carbonate and metal weakly held in organic matter fractions, without disturbing silicate lattices, resistant iron and manganese minerals, and organic compounds. Therefore, the 25% (v/v) HOAc extractant was used to extract the labile-phase metal from sediment.

In addition, the calculation of SQG value in this study does consider the metals bound with sulphide since it can be released from sediment to interstitial water and being available to benthic organisms, as shown in

Eq. (2). However, this formula assumed that the kinetic of metal sulphide oxidation is rapid and completely occurs during re-oxidation of anoxic sediment.

$$\text{SQG} = K_D \times \text{WQC} + M_{\text{AVS}} \quad (2)$$

where SQG is sediment quality guideline (mg/kg dry wt.), K_D is partition coefficient (l/kg), WQC is water quality criteria (mg/l), and M_{AVS} is metal bound with sulphide (mg/kg dry wt.).

In general, the sulphide is usually important in biogeochemical process of estuarine sediment due to its supply from seawater in sulphate form. The sulphate in seawater is classified as a major ion, which is found at 7.7% (w/w) of total dissolved ions or 0.0282 mol/kg of seawater (Libes, 1992). Under anoxic sediment, the sulphide will be generated *via* sulphate reduction process. Therefore, most of anoxic estuarine sediment contains relatively high amounts of sulphide (Elderfield and Hepworth, 1975). It is well known that sulphide plays an important role in controlling speciation of metal in sediment-interstitial water system (Lee et al., 2000).

In anoxic estuarine sediment, sulphide is primarily bound with metals and also formed insoluble metal sulphide, which results in a loss of dissolved metals from interstitial water. However, re-oxidation processes in sediment may occur, if anoxic sediment is exposed and oxidized (Lin and Morse, 1991; Petersen et al., 1997) through bioturbation, tides, storms, bottom trawling, dredging activities, etc. These processes can induce sediment re-suspension resulting in oxidation of sediment, which lead to release of dissolved metals into interstitial water. The study of Cooper and Morse (1996) has reported that up to 90% of metals bound with sulphide can be oxidized within one day of exposure to oxic condition. Consequently, the re-oxidation processes cause the transformation of insoluble metal sulphide to dissolved metals in interstitial water.

Methods

Sampling Sites and Samples Collection

The sampling site was located at the Chao Phraya river mouth, Samut Prakarn province, Thailand (Figure 1). Five sediment samples were collected on March 9, 2011 by using push corer made from Plexiglas® (acrylic) with an inner diameter of 6.4 cm and length of 70 cm. For each sediment core sample, a top 30 cm of sediment and a 30 cm water column over the sediment surface was collected. The top-end of the corer was immediately sealed with plastic bag. A Superlene Nylon 6 (polyamide) plug was inserted into the bottom-end

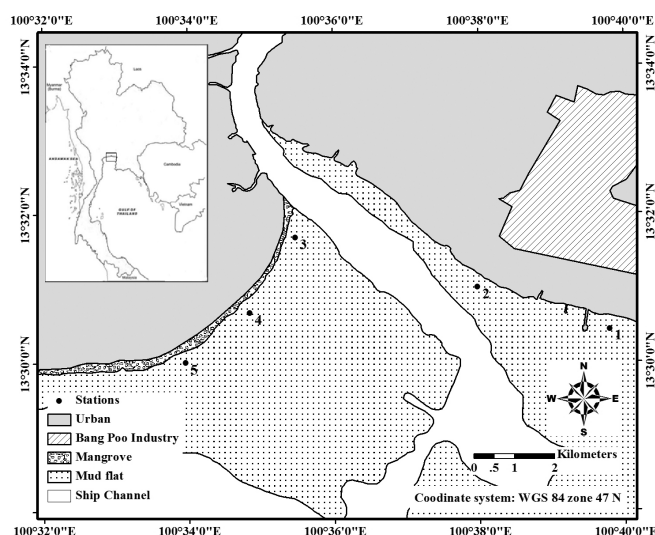


Figure 1: Sampling site at the Chao Phraya river mouth, Samut Prakarn province.

of the core to hold the sediment. The sediment core samples were stored in an ice box. The cores were held up-right in a vertical position after sampling and during transportation back to the laboratory. In the laboratory, these sediment cores were stored in the dark at 4 °C in refrigerator.

Subsampling and Interstitial Water Extraction of Sediment from Sediment Core Samples

After storing in the dark at 4°C for 21 days until equilibrated, the top 20-cm sediment in the core sample was subsampled into four layers of 5 cm thickness inside a nitrogen glove box to avoid oxidation of anoxic sediment (Bufflap and Allen, 1995). Each sediment subsample was measured for its redox potential (Eh) with a Platinum Combination Redox Electrode (Thermo Orion 290A), and was divided into two portions. The first portion of sediment (approximate 10 g) was kept in a sealed plastic bag and stored at 4 °C for analyzing acid volatile sulphide (AVS) and simultaneously extracted metals (SEMs). The remaining wet sediment portion was placed into a 50-ml polyethylene centrifuge bottle for extracting of interstitial water, which was centrifuged at 3500 relative centrifugal forces (RCF) (approx. 3900 rounds per minutes) for 30 minutes at room temperature. The extracted interstitial water was in-line filtered using the polypropylene syringe through the 0.45 µm, acid-cleaned Nuclepore® filter membrane under the oxygen-free atmosphere. The filtered interstitial water was immediately adjusted pH to <2 with concentrated sub-boiling distilled HNO₃ before storing for further analysis. After centrifugation, the

sediment samples (solid phase) were freeze-dried until dryness (approx. 24 hours) before storing in labelled plastic bag for further analysis.

Laboratory Analysis

Freeze-dried sediment was analyzed for their characteristics (i.e., grain size composition, organic matter, and calcium carbonates) and was extracted for the labile-phase metal. To ensure that a portion taken for analysis represents the sediment sample, each dried sediment sample was homogenized by using coning and quartering technique (Gerlach et al., 1990).

Analysis of grain size composition of sediment was performed by sieve-pipette method after pre-treatment sample to remove organic matter and calcium carbonate (Beuselinck et al., 1998). Determination of organic carbon content in sediment was done using the Walkley-Black method (Loring and Rantala, 1992). Calcium carbonate content in sediments was analyzed by the acid-base titration method as described by Sompongchaiyakul (1989).

The labile-phase metal in sediment was extracted by 25% (v/v) HOAc (Loring and Rantala, 1992). Sample of the freeze-dried sediment (0.8 g) was extracted for six hours with 10 ml of 25% (v/v) HOAc. After six hours, the solution and sediment was separated by centrifuging at 3500 RCF for 30 minutes at room temperature. The clear supernatant was poured and made up the volume to 10 ml with Milli-Q water. The concentration of labile-phase metal was determined by graphite furnace atomic absorption spectrophotometry (GFAAS) for Cd, Cu and Pb and by flame furnace atomic absorption spectrometry (FAAS) for Zn.

The interstitial water sample was pre-concentrated by using the Cobalt-APDC co-precipitation method (modified from Boyle and Edmond, 1977). The interstitial water (5 ml) volume was made up to 40 ml with Milli-Q water. High-purity NH₃OH solution was added to adjust pH ~3. The co-reagent 2.00 ml and 2% APDC reagent 2 ml were added and shaken for one minute and then left at room temperature for 30 minutes. After 30 minutes, sample was centrifuged at 3500 RCF for 30 minutes at room temperature. The precipitated colloid on bottom of centrifuge tube was washed with 30 ml of Milli-Q water and centrifuged at 3500 RCF for 30 minutes at room temperature again. The concentrated nitric acid 65% Suprapur® for trace analysis (Merck) was added 300 µl to re-dissolve a precipitate colloids and made up the volume to 5 ml

by Milli-Q water. The pre-concentrated interstitial water sample was kept for 24 hours before determination of Cd, Cu, Pb and Zn by GFAAS.

The wet sediment was extracted to determinate the AVS and SEMs by using the cold-acid purge-and-trap method (Allen et al., 1991). Approximately 10 g of wet sediment was extracted with 1 M HCl and purged with nitrogen gas for one hour. The sulphide which was trapped in 0.5 M NaOH solution (AVS) was determined by colorimetric method. After releasing AVS, the acidified sediment sample was filtered through GF/C filter. The metal solubilized from the sediment in acid condition (SEMs) was determined by GFAAS for Cd, Cu and Pb; and by FAAS for Zn.

Results and Discussion

After subsampling of sediment core samples, redox potential (Eh) was measured immediately. Moisture was taken from the difference of the weight before and after

freeze dried. The characteristics of sediment include grain size composition, organic compound content and calcium carbonate content which are reported in Table 1.

In general, the concentration of metal in sediment will depend on characteristics of sediment. For example, Förstner and Wittmann (1981) reported that the metal concentration tends to decrease with increasing grain size in sediment. Another study, Chester and Voutsinou (1981), reported that calcium carbonate content plays an important role of the “dilution” affecting the metal concentration especially in marine sediment. Therefore, in order to reduce the sediment characteristics effect in reporting the metal concentration in sediment and help in better inter-comparison of results, both fine grain sediment fraction (<63 μm) and calcium carbonate content was recommended as a primary reference for normalization of metal in estuarine sediment (Chester and Voutsinou, 1981; Loring, 1991; Szava-Kovats, 2008).

Table 1: Sedimentological and geochemical characteristics of core sediments collected from the Chao Phraya River mouth on 9 March 2011

Stations	Layers (cm)	Eh (mV)	Moisture (%)	Size composition (%)			OC (%)	CaCO ₃ (%)
				Sand (>63 μm)	Silt (2-63 μm)	Clay (<2 μm)		
1	0 – 5	-96.0	43	27	49	24	0.70	1.12
	5 – 10	-108.0	32	56	27	17	0.40	1.29
	10 – 15	-130.8	32	42	40	18	0.53	1.60
	15 – 20	-151.4	31	41	37	22	0.76	2.06
2	0 – 5	-141.6	34	27	53	20	0.43	1.91
	5 – 10	-188.9	37	35	44	21	0.82	1.57
	10 – 15	-241.6	37	35	48	17	0.83	1.32
	15 – 20	-255.4	31	24	51	25	1.01	1.34
3	0 – 5	-19.9	50	26	52	22	0.95	1.73
	5 – 10	-88.0	45	28	68	4	1.05	2.20
	10 – 15	-76.6	42	33	53	14	0.89	8.61
	15 – 20	-96.0	43	27	50	23	1.03	3.47
4	0 – 5	-94.0	31	3	71	26	0.36	3.23
	5 – 10	-117.3	32	8	69	23	0.58	5.25
	10 – 15	-131.0	38	2	70	28	0.78	4.33
	15 – 20	-189.3	33	2	66	32	0.97	4.39
5	0 – 5	-89.5	73	67	12	21	1.66	3.70
	5 – 10	-79.2	60	56	20	24	1.27	7.76
	10 – 15	-130.1	55	29	43	28	1.23	3.62
	15 – 20	-130.3	56	23	57	20	1.44	3.28

Calculation of Partition Coefficient for Metals

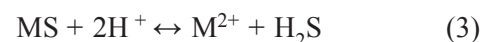
The K_D for metal was calculated by dividing the metal concentration in sediment, extracted by using 25% (v/v) HOAc and also calculated on the basis of size normalization (<63 μm) and carbonate-free basis, with the metal concentrations in the interstitial water. The metals concentration in sediment and in interstitial water; and the K_D are reported in Table 2.

From Table 2, it is found that the K_D value of each sediment core in depth is different. In general, the variation of the K_D seems to be related to the natural variability in sediment characteristics because these factors exert strong influence on chemicals concentration both in sediment and interstitial water. Thus, some data group of K_D may have orders of magnitude differences, even for a single metal (US-EPA, 2005).

Calculation of Metals Bound with Sulphide

One of the important factors in calculating the SQG for metal in this study is metal bound with sulphide.

The M_{AVS} value was calculated from reaction of metal sulphide from the analysis of AVS and SEMs (Eq. 3), as modified from Allen et al. (1993).



where MS is metal sulphide; 2H^+ is purged gas; M^{2+} is metal in SEMs fraction; and H_2S is hydrogen sulphide (AVS).

From the AVS and SEMs analysis, the AVS is operationally defined as sulphide that forms into hydrogen sulphide (H_2S) under the acid condition of mixing sediment with 1 M HCl (acidification step). Whereas, the SEM is operationally defined as metal that bind mole to mole with sulphide (includes FeS and MS). The metal in SEMs fraction is dissolved in acidification step (M^{2+}) (Allen et al., 1991). The concentration of AVS and SEMs (after normalized with fine grain size of sediment and CaCO_3 free basis) in sediment subsamples are reported in Table 3.

Table 2: Concentration of metals in sediment and interstitial water; and the partition coefficient of metals

Stations	Layers (cm)	C_s				C_{IW}				K_D			
		Cd	Cu	Pb	Zn	Cd	Cu	Pb	Zn	Cd	Cu	Pb	Zn
1	0 – 5	55	11,260	4,832	98,506	0.122	2.51	0.67	42.57	451	4,487	7,202	2,314
	5 – 10	72	21,692	9,784	156,009	0.155	2.51	0.55	87.45	464	8,637	17,698	1,784
	10 – 15	58	17,884	8,338	116,350	0.176	1.64	0.48	76.65	327	10,916	17,365	1,518
	15 – 20	73	23,443	9,479	106,057	0.163	0.75	0.40	49.23	446	31,092	23,758	2,155
2	0 – 5	44	4,788	2,655	48,268	0.186	2.29	0.86	75.48	237	2,091	3,104	639
	5 – 10	70	7,485	3,875	71,745	0.134	0.93	1.28	37.76	524	8,091	3,035	1,900
	10 – 15	82	8,337	3,823	77,806	0.174	0.94	1.53	62.85	469	8,895	2,498	1,238
	15 – 20	69	6,791	3,980	65,116	0.133	1.95	0.58	87.73	518	3,489	6,900	742
3	0 – 5	61	3,606	3,265	34,245	0.180	3.17	1.30	65.15	340	1,136	2,508	526
	5 – 10	71	3,585	4,337	30,351	0.151	0.78	0.63	68.12	468	4,611	6,831	446
	10 – 15	69	4,035	5,573	33,291	0.225	1.38	1.29	88.93	308	2,929	4,323	374
	15 – 20	76	3,149	5,659	27,481	0.126	2.46	1.02	52.66	601	1,280	5,527	522
4	0 – 5	68	4,413	8,273	119,154	0.194	1.53	1.90	63.47	351	2,885	4,343	1,877
	5 – 10	79	3,471	9,406	66,935	0.099	0.68	0.41	68.26	797	5,095	22,953	981
	10 – 15	66	2,953	4,694	30,612	0.126	0.93	1.27	50.07	526	3,158	3,706	611
	15 – 20	60	3,260	5,138	27,851	0.113	3.42	2.24	51.59	529	952	2,292	540
5	0 – 5	68	2,884	3,379	14,412	0.168	3.36	0.83	33.28	404	860	4,053	433
	5 – 10	72	1,930	4,993	14,968	0.141	1.11	0.60	30.82	511	1,747	8,378	486
	10 – 15	96	1,751	4,993	8,871	0.337	1.20	0.69	50.51	284	1,463	7,278	176
	15 – 20	42	1,725	5,245	10,292	0.107	4.12	1.13	82.13	396	419	4,654	125

C_s : Metals concentrations in sediments ($\mu\text{g}/\text{kg}$ dry wt. in < 63 μm fraction and free CaCO_3 basis)

C_{IW} : Dissolved metals concentrations in interstitial water ($\mu\text{g}/\text{l}$)

K_D : Partition coefficient of metals (l/kg)

Table 3: Concentration of acid volatile sulphide, simultaneously extracted metals and the metals bound with sulphide values

Stations	Layers (cm)	AVS	AVS/5	SEMs				M_{AVS}			
				Cd	Cu	Pb	Zn	Cd	Cu	Pb	Zn
1	0 – 5	0.67	0.13	0.0007	0.51	0.12	3.82	0.08	8.5	24.9	8.7
	5 – 10	0.66	0.13	0.0007	0.47	0.19	4.69	0.08	8.4	27.2	8.6
	10 – 15	1.35	0.27	0.0006	0.30	0.13	2.96	0.07	17.2	27.8	17.7
	15 – 20	1.32	0.26	0.0005	0.35	0.14	2.72	0.06	16.7	28.2	17.2
2	0 – 5	0.87	0.17	0.0003	0.07	0.04	1.05	0.04	11.0	7.9	11.3
	5 – 10	1.50	0.30	0.0007	0.23	0.08	2.55	0.08	19.0	16.8	19.6
	10 – 15	1.24	0.25	0.0009	0.35	0.08	2.58	0.10	15.7	16.2	16.2
	15 – 20	1.42	0.28	0.0009	0.40	0.08	3.49	0.10	18.1	17.3	18.6
3	0 – 5	0.04	0.01	0.0007	0.49	0.10	1.56	0.08	0.5	1.8	0.6
	5 – 10	1.74	0.35	0.0008	0.27	0.08	1.52	0.09	22.1	17.0	22.8
	10 – 15	1.14	0.23	0.0008	0.21	0.11	1.88	0.09	14.6	22.3	15.0
	15 – 20	1.51	0.30	0.0008	0.12	0.07	1.82	0.09	19.2	15.1	19.8
4	0 – 5	0.09	0.02	0.0010	0.26	0.15	6.20	0.11	1.1	3.5	1.1
	5 – 10	0.67	0.13	0.0009	0.45	0.11	3.73	0.10	8.5	23.3	8.8
	10 – 15	0.69	0.14	0.0007	0.29	0.09	1.92	0.08	8.8	18.2	9.1
	15 – 20	0.95	0.19	0.0008	0.25	0.10	1.57	0.09	12.0	21.4	12.4
5	0 – 5	0.03	0.01	0.0011	0.33	0.10	1.43	0.12	0.4	1.1	0.4
	5 – 10	2.21	0.44	0.0013	0.19	0.07	1.27	0.15	28.1	14.1	28.9
	10 – 15	0.53	0.11	0.0006	0.21	0.08	1.12	0.07	6.7	15.7	6.9
	15 – 20	1.08	0.22	0.0006	0.41	0.08	1.21	0.07	13.7	17.2	14.1

AVS: Acid volatile sulphide (mmols-S²⁻/kg dry wt.)

SEMs: Simultaneously extracted metals (mmols/kg dry wt. in <63 μ m fraction and free CaCO₃)

M_{AVS} : Metals bound with sulphide (mg/kg dry wt. in <63 μ m fraction and free CaCO₃)

The calculation of M_{AVS} is based on the assumption as follows: (1) oxidation kinetic of M_{AVS} in anoxic sediment occurs rapidly and completely, and (2) competitive ability of each metal to binding with sulphide is equal. Therefore, amount of sulphide mole that bond with each metal, was calculated from the total molar of AVS divided by the number of metals bound with sulphide which is five. Since, five metals (Cd²⁺, Cu²⁺, Ni²⁺, Pb²⁺ and Zn²⁺) are common metals that bound with sulphide in the sediment, these can be found in SEMs fraction (Di Toro et al., 1992; Allen, 1993). Based on the above principle, the M_{AVS} for Cd, Cu, Pb and Zn is reported in Table 3.

Numerical Sediment Quality Guideline

In this study, the SQG values were calculated following the Eq. 2 by using the K_D (Table 2) and the M_{AVS} (Table 3). The adopted WQC was from the Thai coastal water quality standards for Cd (5 μ g/l), Cu (8 μ g/l), Pb (8.5

μ g/l) and Zn (50 μ g/l), published in the Thai Royal Government Gazette Vol. 124, Part 11, dated February 1, B.E.2550 (Thai Royal Government Gazette, 2007).

The SQG values for Cd, Cu, Pb and Zn are illustrated in Figure 2. The SQG values of Cd, Cu, Pb and Zn were found in a range of 1.2–4.1, 7–265, 23–230 and 16–125 mg/kg dry wt. in <63 μ m fraction and CaCO₃ free basis, respectively. However, some of the SQG values were clustered to the outlier and the far-outlier of data group by using box-plot model (Figure 2). In order to report the SQG values that represented most of the data group, the outlier and far-outlier value of data will not be brought into consideration. Consequently, the SQG values for Cd, Cu, Pb and Zn for the Chao Praya river mouth sediment after removal of the outlier and far-outlier values were in a range of 1.2–3.1, 7–105, 23–86 and 16–125 mg/kg dry wt. in <63 μ m fraction and CaCO₃ free basis, respectively.

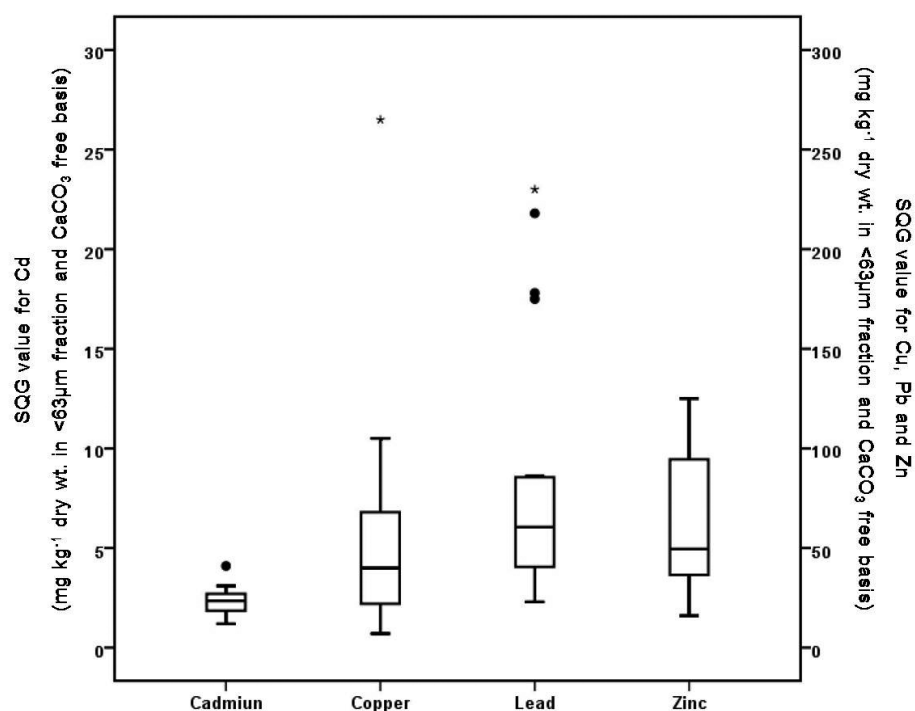


Figure 2: Sediment quality guideline for Cd, Cu, Pb and Zn for Chao Praya river mouth sediment showing a box-plot comparison of median and range among metals (the horizontal line in the box means median (or 50th percentile), the upper and lower lines outside the box refer to 75th and 25th percentiles (so-called interquartile range; IQR), filled circle and star refer to outlier (1.5 IQR) and far-outlier (3 IQR) of data).

From the results of the SQG values, in order to estimate adverse biological effects for benthic organism, it is necessary to compare the maximum SQG value with toxicity level on the benthic organism. To date, the data of sediment toxicity on benthic organism is not established in Thailand. Thus, in this study, the toxicity data of amphipod (*Melita plumulosa*) reported by King et al. (2006) is chosen to compare with the maximum SQG value (Table 4). It was found that the maximum SQG values were about 10 to 50-fold lower than the levels that cause toxic to *M. plumulosa*. This may be due to two reasons. Firstly, sediment toxicity testing for metals were likely to be overestimated due to metal strongly bound with sediments. This process leads to a loss of metal toxicity (Simpson et al., 2004).

Secondly, the WQC values being used in the calculation of SQG value may not reflect toxicity on benthic organism. Moreover, different country established different WQC. For example, recommended water quality criteria of chronic toxic for seawater in USA for Cd, Cu, Pb and Zn is 8.8, 3.1, 8.1 and 81 µg/L, respectively (US-EPA, 2009), while in Thailand, the water quality criteria for Cd, Cu, Pb and Zn is 5, 8, 8.5 and 50 µg/L, respectively. The WQC value that does

not reflect toxicity on benthic organism or the difference of WQC value can lead to under or over estimation of sediment toxicity. This may be a disadvantage of the EqP approach in calculating the SQG value.

Conclusions

One of the important tools in assessing the contaminated sediment that affects benthic organisms is the SQG. The difference of SQG worldwide depends on methodological approaches and physicochemical properties of sediment. Thus, the SQG value should be established specifically for local sediment. The SQG calculation for metal should be considered only the labile-phase metal in sediment because it will relate with sediment toxicity. The EqP approach is a theoretically-based SQG that attempts to account for bioavailability of chemicals in sediment. Therefore, the EqP approach is believed to be the most appropriate approach in predicting potentiality for adverse effects on benthic organisms. However, the SQG should be studied coupled with sediment toxicity test for local sediment, since benthic organisms in different area may adapt to tolerant to different toxicity level.

Table 4: Sensitivity of amphipod (*Melita plumulosa*) to contaminant-spiked sediments (modified from King et al. (2006))

<i>Life stage</i>	<i>Metals</i>	<i>Effect</i>	<i>Concentration of metal in sediment (mg/kg dry wt.)</i>
Adult	Cd	Lethal concentration fifty	> 260
		No observable effect concentration	260
	Cu	Lethal concentration fifty	1,310
		No observable effect concentration	520
	Pb	Lethal concentration fifty	> 3,560
		No observable effect concentration	3,560
	Zn	Lethal concentration fifty	> 9,040
		No observable effect concentration	2,290
	Cd	Lethal concentration fifty	1,630
		No observable effect concentration	620
Juvenile (no feeding)	Cu	Lethal concentration fifty	790
		No observable effect concentration	460
	Pb	Lethal concentration fifty	1,980
		No observable effect concentration	580
	Zn	Lethal concentration fifty	1,790
		No observable effect concentration	< 2,290

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