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Dynamic Evaluation of Intertidal Wetland Sediment Quality in a Bay System

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ABSTRACT. Intertidal wetlands are fragile ecological systems vulnerable to changing environmental conditions. Using the Rapid Assessment Approach, intertidal wetland sediments were dynamically evaluated in the Deep Bay, South China in terms of monitored benthic macroinvertebrates. The observation and expectation ratio indicated variations of intertidal wetland sediment quality had taken place at certain ecologically sensitive areas, such as the Mai Po Marshes. Series of dynamic models were used to simulate the behavior of certain environmental variables (i.e. cadmium, copper, lead and zinc accumulation rates) that might have adverse effect on the growth of benthic macroinvertebrates, and hence interpreted the results of the ecological assessment. The changes of sediment quality were primarily due to anthropogenic activities and were further attributed largely to the industrial discharged heavy metals which were unevenly distributed and heavily accumulated at some sensitivity areas (i.e. Futian National Natural Reserve and Mai Po Marshes) with help of sediment deposition altered by long-term tidal flow actions in Deep Bay.

Keywords: heavy metals, intertidal wetland sediments, macroinvertebrates, models

1. Introduction

Estuaries and coastal seas are confronted by increasingly disruptive human activities related to socioeconomic development (Worm et al., 2006; Halpern et al., 2008). By the late 20th century, such human activities (including land reclamation, habitat destruction, resource over-exploitation and the introduction of contamination and disease) caused the loss of 65% of seagrasses and 48% of other submerged aquatic vegetation, depleting more than 90% of important species and destroying 67% of wetland habitats (Lotze et al., 2006). Estuarine wetlands are of considerable ecological importance (Mitsch and Wilson, 1996; Ji, 2008), and so it is necessary to accurately assess its sediment quality in order to reduce exploitation, improve water and sediment quality, and restore habitats of intertidal areas.

Futian National Natural Reserve (covers an area of 3 km^2) and Mai Po Marshes (extends for an area of 3.8 km^2) are the important intertidal wetlands attracting worldwide interests for wildlife conservation. They both include mangrove and mudflat, and locate at the eastern end of Deep Bay, which is situated between Shenzhen, a special economic zone of China, and Hong Kong (Figure 1). In spring and autumn each year, the intertidal wetlands of Deep Bay provide food to more than 60,000 birds of 270 species (Che and Cheung, 1998; Man et al., 2004), and so it is necessary to maintain the water and sediment quality of the intertidal wetlands at an acceptably high standard (the Government of the Hong Kong Special Administrative Region of the People's Republic of China (1997) has issued that limit levels for copper, lead, cadmium, mercury and arsenic in sediment of Mai Po Marshes were 130, 100, 0.65, 0.31 and 21 mg/kg, respectively). However, the rapid socioeconomic development of Shenzhen and Hong Kong led to an increasing amount of polluted water entering Deep Bay in the 1990s, particularly water contaminated by heavy metals from industrial effluent discharged into the Shenzhen River, Yuan Long Creek, and Tin Shui Wai Creek (Che and Cheung, 1998; Cheung and Wong, 2006). Consequently, the water and sediment quality at Futian National Natural Reserve and Mai Po Marshes progressively worsened (Che and Cheung, 1998; Duan et al., 2009) and their ecological functions were deteriorating. For example, the benthic infauna community, which was the major food source of the migratory birds, was characterized by low species richness and high dominance of a few pollution-tolerant opportunistic species (Blackmore, 1998; Cheung et al., 2003; Liang and Wong, 2003; Man et al., 2004). However, based on Shannon-Weaver species diversity index monitored by Cai et al. (2003), it was found that the degradation of species diversity in Mai Po Marshes was worse than that in its adjacent area, Futian National Natural Reserve. It

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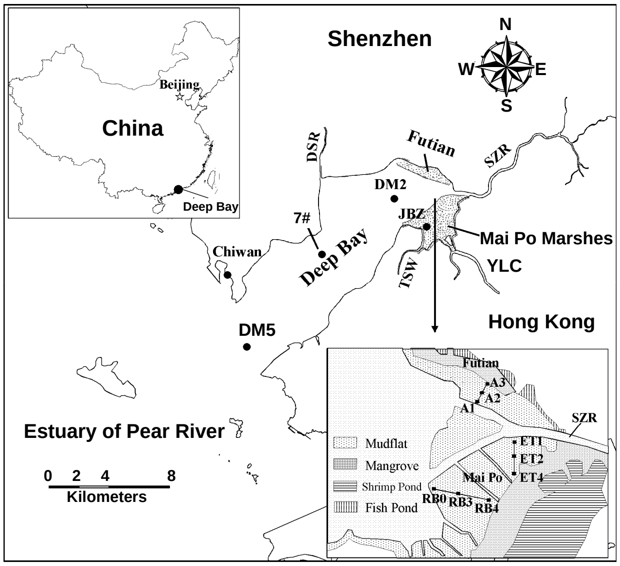

Figure 1. General location of Deep Bay, China, showing the sampling stations. (SZR, DSR, YLC and TSW indicate Shenzhen River, Dasha River, Yuen Long Creek and Tin Shui Wai Creek. Section A refers to A1, A2 and A3; Section ET refers to ET1, ET2 and ET4; and Section RB refers to RB0, RB3 and RB4).

was very interesting that neighbored areas (just separated by the Shenzhen River inlet) had quite different responses to the similar human interferences. However, until now, no literature has interpreted the interesting phenomenon.

Because of the importance of intertidal wetland sediments, proper evaluation of sediment quality has attracted a great attention in recent decades (Comte et al., 2010). Conventional assessments of sediment quality in wetland systems focus primarily on physical and chemical variables (DelValls and Chapman, 1998). However, this kind of approach is not comprehensive, nor sufficiently precise in identifying the interrelationship between sediment and its environment. Biological components of the organism living on sediment are more sensitive and responsive to changes in the environmental conditions (Diaz et al., 1993). Thus, Biological assessment, which can provide a direct measure of the state and functionality of an aquatic community of plants and animals, is widely applied nowadays (Archaimbault et al., 2010). Rapid Assessment Approach for Intertidal Wetland Sediments (RAITWS) (Ni et al., 2012), developed in recent years, is one of the biological methods. It is based on the principles of Rapid Biological Assessment (RBA) (Wright, 1995), which has been widely used to evaluate river biological quality because of its high efficiency (Wright et al., 1984; Chessman, 1995; Metzeling et al., 2003; Buss and Vitorino, 2010). However, since comprehensive field monitoring data for RBA is very difficult to obtain, it is too expensive to use RBA for intertidal wetlands affected by tidal dynamics (Ni et al., 2012). RAITWS overcomes the disadvantage and applies series of classical models, including hydrodynamics, sediment transport, bed deformation, and water quality models, to simulate the dynamic processes. Thus, RAITWS not only inherits the high efficiency of RBA, it is also able to reflect possible changes in environmental variables by incorporating a series of dynamic models, and so is particularly useful for assessment of intertidal wetland environmental quality in cases where ecological changes are taking place due to nearby coastal development (Ni et al., 2012).

In this paper, we used RAITWS to assess the impact of urbanization and ongoing industrial expansion on sediment quality at Futian National Natural Reserve and Mai Po Marshes. It would help us to understand the impact degree of anthropogenic activities under given natural environmental background. Furthermore, the RAITWS could also interpret why the neighbored areas (Futian National Natural Reserve and Mai Po Marshes) with similar natural environmental variables in the intertidal wetland sediments had different responses to the similar anthropogenic activities.

2. Methods

2.1. Rapid Assessment Approach for Intertidal Wetland Sediments (RAITWS)

RAITWS is a dynamic biological approach to assess the intertidal wetland sediment quality. It is based on the assumption that similar features and structures of biological communities occur at different undisturbed locations characterized by the same environmental parameters and having relatively similar types of habitat. According to the assumption, the anthropogenic activities can be separated from the natural variation, and the features of biological communities at test site without anthropogenic activities (called expectation) can be predicted. Then, sediment quality variation at test sites under anthropogenic pressure can be assessed by comparing the observed bio-community characteristics (called observation) with the reference data (the Observation to Expectation O/E ratio). In this paper, benthic macroinvertebrates are chosen as biological indicators because their relatively long life cycle, more stationary behavior, and occupation of a wider niche in the food web constitute a fairly complete biological accumulation process. And hence, the bio-communities comprise benthic macroinvertebrates.

The detailed procedure of this study is shown in Figure 2 and described as follows:

(1) Construction of reference groups (reference sites are classified into reference groups). Reference sites should satisfy the criteria for minimal disturbance by anthropogenic activities. Then, based on observed biological data, such as biomass and abundance of benthic macroinvertebrates, the reference sites can be classified into groups according to their uniformity by means of multivariate statistics.

(2) Selection of a subset of environmental variables. The environmental variables are the parameters which can indicate the benthic macroinvertebrates structure. In wetland, latitude, longitude, pH, salinity, water saturation, inundation time, tidal range, total organic carbon, and sediment size grading are always selected as the environmental variables.

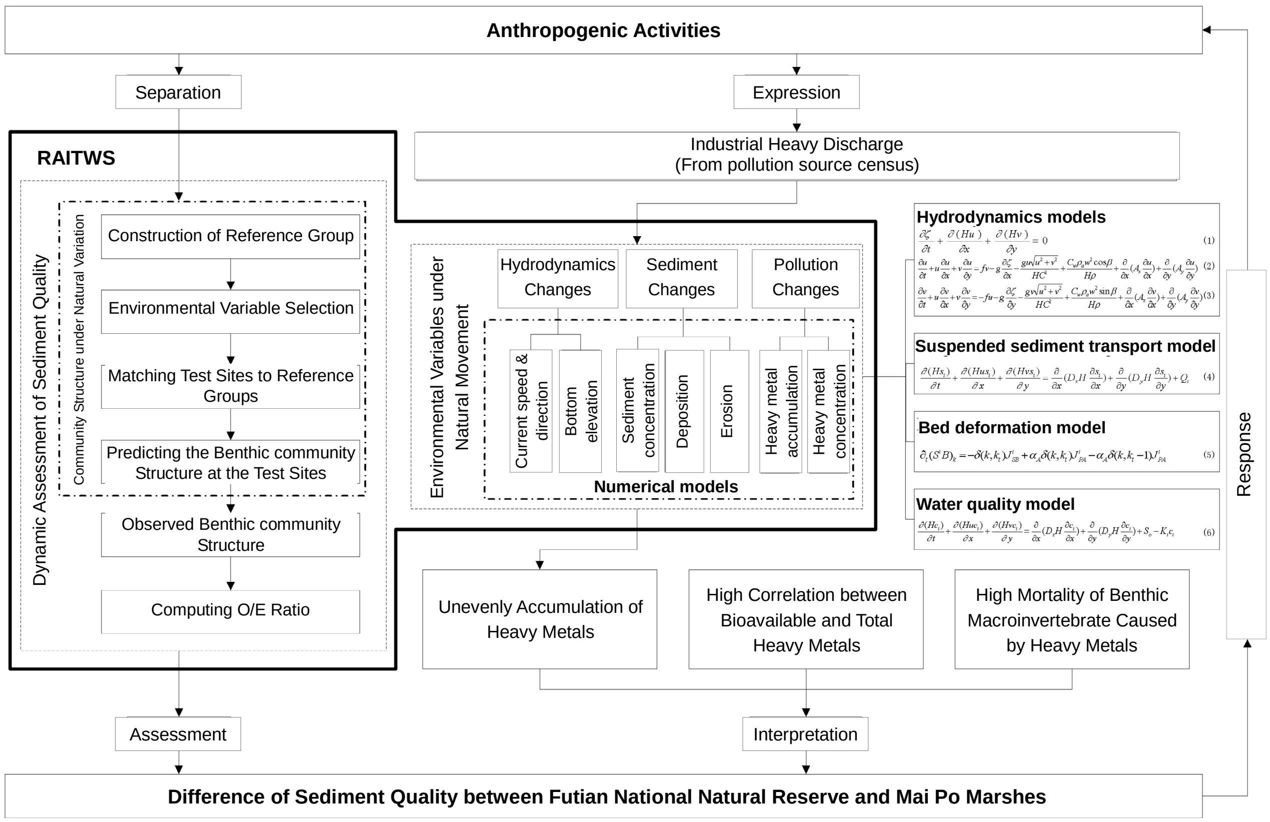

Figure 2. Flow diagram of the RAITWS approach in wetland of Deep Bay.

(3) Matching test sites to reference groups. In this step, the assumption that similar features and structures of biological communities occur at different undisturbed locations characterized by the same environmental variables is used. Based on similarity of the selected subset of environmental variables, the probabilities of the test site falling into each reference group can be estimated from multiple discriminant analysis. In this paper, we want to assess intertidal wetland environmental quality over time. Therefore, the test sites are the same as the reference sites.

(4) Estimation of benthic fauna community structure at test sites. First, the occurrence probability of prescribed species to exist at the test site can be calculated by multiplying the percentage frequency of the prescribed species present in each reference group by the probability that the test site falls within the corresponding reference group. Second, the integrated occurrence probability of each prescribed species can be estimated by summing its occurrence probabilities over the entire set of reference groups. And then the estimated number of species at the test site (under the natural variation) can be predicted by summing the occurrence probability of all the prescribed species except the ones whose integrated occurrence probabilities are less than 50% (Moss et al., 1987). The estimated number of species at the test site is the expectation number of species under unpolluted environment.

(5) Computation of the observation and expectation ratio (O/E ratio). The O/E ratio is equal to the observed number divided by estimated number of species. The O/E ratio in the range from 0 to 1 can be used to evaluate the state of the benthos community structure, with 1 indicating that conditions in the observed community correspond to exactly those expected.

(6) Interpretation of sediment quality changes (variations of O/E ratio) by applying numerical model. The deviation of O/E ratio indicates whether the study area has been disturbed by human beings. However, it is difficult to explain the variation in O/E ratio between different sites because intertidal wetlands have highly sensitive environments affected by tidal dynamics and it is very difficult and expensive to measure the dynamic processes by monitoring program. Instead, series of dynamic numerical models can simulate the dynamic processes and characterize quantitatively certain environmental variables (e.g. depth of habitat, marine and hydrological conditions, physical and chemical properties of bed sediments, and heavy metal contamination). Moreover, the numerical models can also simulate the process how human issues are redistributed by natural movement. Thus, the variation in O/E ratio between different sites may be interpreted by the numerical models.

In practice, the intertidal wetland is assumed to be a shallow system, and so the numerical models could be two-dimensional horizontal. Thus, a typical set of depth-averaged governing equations (Ni et al., 2012) for the water-sediment-contaminant system (shown in Figure 2) are used. The mass

Table 1. Occurrence Frequency for Various Macroinvertebrates in Reference Groups (%) and the Integrated Occurrence Probability of these Species (%)

Species	Group 1	Group 2	Group 3	Integrated Occurrence Probability
Dendronereis Pinnaticirrus	33.3	100	100	77.8
Neanthes Glandicincta	33.3	40	100	44.4
Capitella Capitata	66.7	60	100	66.7
Sigambra Hanaokai	66.7	60	100	66.7
Potamilla Acuminata	33.3	100	100	77.8
Assiminea Brevicula	0	100	100	66.7
Nephthys Polybranchia	66.7	80	100	77.8
Marine Nematodes	0	100	100	66.7
Assiminea Sp.	0	100	100	66.7
Stenothyra Devalis	0	100	100	66.7
Bithynia Fuchsiana	0	80	100	55.6
Tharyx Sp.	33.3	40	100	44.4
Upogebia Major	66.7	0	0	22.2

and momentum of the flow hydrodynamics can be expressed as equations (1), (2) and (3) in Figure 2, where ζ is the tidal elevation above mean water level, H is the instantaneous total water depth, u and v are depth-averaged velocity components in the x and y directions, f is the Coriolis parameter, g is the gravitational acceleration, C is the Chézy coefficient related to bed roughness, C_w is the wind stress coefficient, ρ_a is the density of air, w is the wind speed, β is the angle between wind direction and the positive x direction, and A_x and A_y are the kinematic eddy viscosities in x and y directions. Suspended sediment transport can be described as equation (4), where s_i is the depth-averaged suspended sediment concentration of the *i-th* constituent, D_x and D_y are dispersion coefficients in the in x and y directions, and Q_i is the source/sink term for the *i-th* constituent. Bed deformation is governed as equation (5), where S^i is the mass sediment concentration of per total volume of bed layer k, B is the layer thickness, $J_{SB}{}^i$ is the net sediment mass flux, α_A is an armoring parameter (1 for armoring, 0 otherwise), $J_{PA}{}^i$ is the parent to armoring layer when the top or surface layer of the bed, k_t, acts to simulate armoring, and i is the *i-th* sediment constituent. Equation (6) can be used to describe two-dimensional horizontal water quality model, where c_i is the concentration of the *i-th* constituent, S_o is the source term; K_i is the decay coefficient of the *i-th* constituent.

In this paper, RAITWS is applied to assess the quality of sediments at Futian National Natural Reserve and Mai Po Marshes of Deep Bay, China (shown in Figure 1).

2.2. Sampling and Analysis

Baseline surveys of invertebrates (Peking University, 1995) were carried out during 1994 and 1995. In this survey, the abundance and biomass of benthic fauna on the mudflat were gauged four times each year (January, April, July and October). Similarly, regular samples were also collected from 1996 to 1999 in four seasons of each year. The sampling locations of these two survey programs were on mudflat without vegetation. The nine sampling sites were identified by GPS (Global Positioning System), six of which were located at Mai Po Marshes (ET1, ET2, ET4, RB0, RB3 and RB4), the three others at Futian National Natural Reserve (A1, A2 and A3) as shown in Figure 1.

The benthos samples were sampled by inserting a plastic pipe (with 10 cm diameter, 40 cm length) 20 cm beneath the mud surface (five cores were collected within 20 m^2 at each station), and then transferred to a bucket containing water, and the resulting slurry stirred manually. Benthos were collected by pouring the slurry through a 0.5 mm sieve and preserved in a 10% formalin solution stained with Rose Bengal. The procedure was repeated several times, until the bucket water was comparatively clear. The extracted benthos were contained in plastic bottles, and later separated, identified and counted at the laboratory. In determining the number of predominant species, the number of each species was first counted. In order to examine the similarities between different sites, those species common to the sites were identified. Any species with less than 10% overall occurrence frequency was excluded from the multivariate analysis, for reasons of accuracy. Then, the species with greater overall occurrence frequency (over 10%) were counted.

3. Results and Discussion

3.1. Estimation of Benthic Fauna Community Structure at Test Sites

Using K-means clustering from a computer software package, STATISTICA, baseline data (Peking University, 1995) on the abundance of the macroinvertebrates during 1994 and 1995 were used to divide the nine sampling sites into three reference groups as shown in Figure 3. Group 1 contained three sites (A1, A2 and A3) on the Shenzhen side of Deep Bay; Group 2 contained five sites (RB0, RB3, ET1, ET4, and ET2) on the Hong Kong side, and Group 3 contained a single site (RB4) also on the Hong Kong side of Deep Bay.

Given that the nine sites in the 1994 ~ 1995 reference groups and the nine test sites of 1996 ~ 1999 were identical, no significant difference was likely to have occurred in the natural environmental conditions of the two entities in only 4

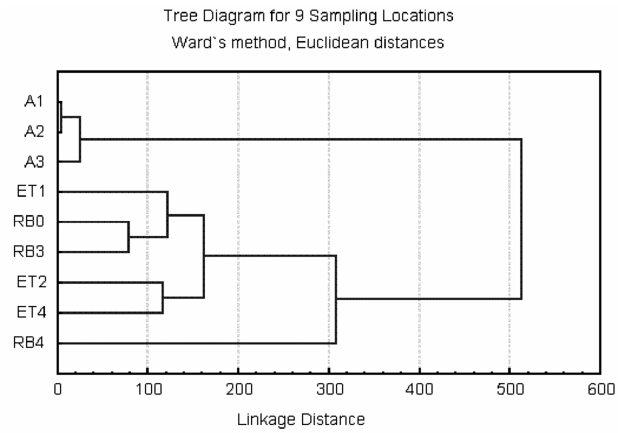

Figure 3. Clustering results from the nine reference sites.

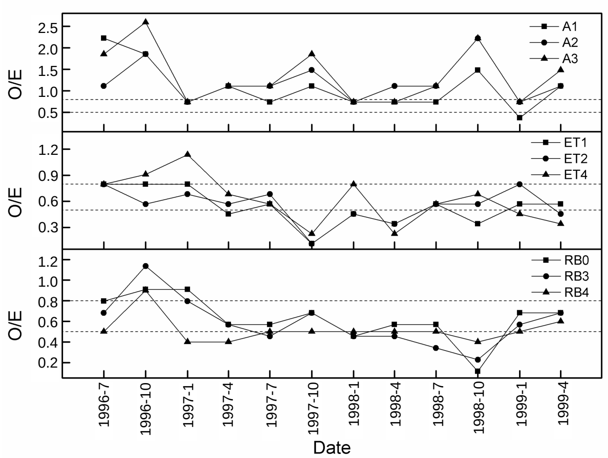

Figure 4. O/E ratios of benthic invertebrates at Sections A, ET and RB from July 1996 to April 1999.

years (Hong Kong Observatory, 2000). It was therefore reasonable to conclude that the 1996 ~ 1999 test sites would still fall into the same reference groups of 1994 ~ 1995. Consequently, it was unnecessary to have to select a subset of environmental variables in this case.

Table 1 listed the percentage of each species that occurred in each reference group, calculated according to the frequency of occurrence of the macroinvertebrates. For example, *Capitella capitata* occurred at 3 stations out of 5 stations in Group 2, and so the occurrence frequency of *Capitella capitata* was 60%. Moreover, the integrated occurrence probability of certain species was also listed in Table 1. The integrated occurrence probability of *Capitella capitata*, for instance, was 66.7% since it occurred at 6 stations out of all 9 reference stations. Ten species of benthic macroinvertebrate, *Dendronereis pinnaticirrus*, *Capitella capitata*, *Sigambra hanaokai*, *Potamilla acuminata*, *Assiminea brevicula*, *Nephthys polybranchia*, marine nematodes, *Assiminea* sp., *Stenothyra devalis*, and *Bithynia fuchsiana* were used as prediction species, each having > 50% integrated occurrence probability (Moss et al., 1987). The expected number of species at a test site was the sum of the occurrence frequencies of macroinvertebrates in the corresponding reference group. Note that the three species such as *Neanthes glandicincta*, *Tharyx* sp. and *Upogebia major* were excluded because the integrated occurrence probability of the each species was less than 50% (Moss et al., 1987), the number of expected macroinvertebrate species was 2.7 by summing of the occurrence frequencies of macroinvertebrates in Group 1. Similarly, using the data in Table 1, the numbers of expected macroinvertebrate species were 8.8 in Group 2, and 10 in Group 3.

3.2. Computation of O/E Ratio

The number of observed macroinvertebrate species in each sampling site during July 1996 to April 1999 was shown in Table 2. Based on the number of expected macroinvertebrate species at A1, A2 and A3 was 2.7, ET1, ET2, ET4, RB0 and RB3 was 8.8, and RB4 was 10.0, respectively, the O/E ratios in January, April, July and October could be respectively calculated as shown in Figure 4. Seven macroinvertebrate species occurred at ET1 in October 1996 and the expected macroinvertebrate species was 8.8, so the O/E ratio was 0.8 at ET1 in October 1996. The O/E ratios for the sites on the Shenzhen side (A1, A2 and A3) were relatively high (the majority > 0.8), and the number of macroinvertebrate corresponded closely to what was expected. Owing to implement of the strict management plan (the State Council of the People's Republic of China, 1994) by Chinese government from 1995, the observed number of species was, sometimes, much higher than the expected, implying that the sediment quality had been recovered on the Shenzhen side. Although O/E ratios < 0.6 were obtained for A1 in July 1997 and 1998 and January 1999, the summary of meteorological observations (Hong Kong Observatory, 2000) had shown that no significant difference was likely to have occurred in the natural environmental conditions of the site. One possible explanation for the exceptionally low O/E ratios at A1 was that the sampling site might be very close to the Shenzhen River where the ecological environment had been disrupted by industrial pollution (Shenzhen Municipal Water Service Bureau, 2000). The O/E ratios at the ET and RB sections on the Hong Kong side of Deep Bay were also displayed in Figure 4. In general, the O/E ratios were much lower (the majority < 0.8) than those on the Shenzhen side (the average O/E ratios at Futian National Natural Reserve and Mai Po Marshes were determined as 0.74 and 0.48, respectively), and indicated that the sediment quality of the Hong Kong wetlands deteriorated during 1996 to 1998. The worst degradation of sediment quality at ET appeared to have occurred in October 1997, as evidenced by the average O/E ratio decreasing to 0.1. For the RB section, the O/E ratios were lower than that at A and similar to those at ET. There was evidence of environmental degradation during 1997 ~ 1998, but perhaps some improvement in 1999. Carey (2001) monitored that there was a decrease of number of waterbirds during 1997 ~ 1998, but an 8.75% increase from 1998 to 1999. The degradation of sediment quality might lead to the decrease of number of waterbirds in Deep Bay. Figure 4 also showed the seasonal variation of O/E ratio. In general, the O/E ratios in wet seasons (April and July) were lower than those in dry seasons (October and January) except in October 1997, which was consistent with the freshwater inputs (Table 3). The exce-

Table 2. Number of Observed Macroinvertebrate Species in Sampling Sites

Time	Sampling Site								
	A1	A2	A3	ET1	ET2	ET4	RB0	RB3	RB4
July, 1996	6	3	5	7	7	7	7	6	5
October, 1996	5	5	7	7	5	8	8	10	9
January, 1997	2	2	2	7	6	10	8	7	4
April, 1997	3	3	3	4	5	6	5	5	4
July, 1997	2	3	3	5	6	5	5	4	5
October, 1997	3	4	5	1	1	2	6	6	5
January,1998	2	2	2	4	4	7	4	4	5
April, 1998	2	3	2	3	3	2	5	4	5
July, 1998	2	3	3	5	5	5	5	3	4
October, 1998	4	6	6	3	5	6	1	2	4
January,1999	1	2	2	5	7	4	6	5	5
April, 1999	3	3	4	5	4	3	6	6	6

Table 3. Flow Discharge, Suspended Sediment, and Heavy Metal Concentrations at River Inlets to Deep Bay*

Parameter	Shenzhen River	Dasha River	Yuen Long Creek	Tin Shui Wai Creek
Q_{wet} (m^3/s)	13.9	4.67	2.46	1.64
Q_{dry} (m^3/s)	1.4	0.47	0.34	0.23
SS_{wet} (g/L)	50	40	35	30
SS_{dry} (g/L)	30	10	10	10
Cd (μg/L)	0.5	0.2	0.25	0.25
Cu (μg/L)	15	12.2	32.8	12
Pb (μg/L)	5.4	3.2	7.7	8
Zn (μg/L)	50	76	140	40

* from Hong Kong Environmental Protection Department (2002) and Shenzhen Municipal Water Service Bureau (2000).

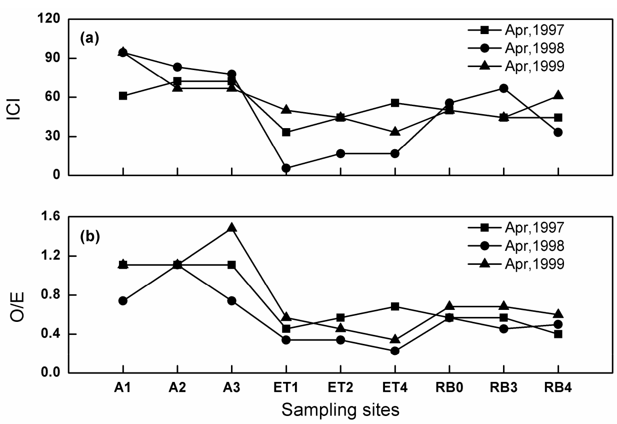

Figure 5. ICI (a) and O/E ratio (b) assessment results for Deep Bay (every April from 1997 to 1999).

ption might be caused by Shenzhen River Regulation Program (dredging the downstream of Shenzhen River in 1997) which could re-suspend and release the concentrated contaminants (such as heavy metals) in sediment, and then lead to worst sediment quality degradation in October 1997.

3.3. Comparison of RAITWS and ICI

The Invertebrate Community Index (ICI) due to Hicks (1997) is widely used nowadays to assess wetland environmental quality, in particular for evaluating the effect of nearby urbanization on permanent freshwater inundation of wetlands. ICI approach is based on the premise that the community of plants and animals living in a wetland will reflect the health of a wetland. When a wetland is damaged, the diversity of animals and plants often decreases and the composition of species changes. Typically, the proportion of organisms that are intolerant to human disturbances will decrease while the proportion of individuals or species that are more tolerant to the disturbance will increase. In comparison to a minimally disturbed site, a plowed wetland located in a cornfield may have fewer plant and animal species. It also may be dominated by organisms that can tolerate poor environmental conditions. In ICI approach, the ecological condition of the main invertebrate communities is analyzed using a multivariate method and then characteristic of the community described by a variable (Roy et al., 2003). Applied in the Eastern Corn Belt Plains ecoregion (Norton et al., 2000), ICI has been proved a valueble means to assess the ecological status of wetlands (Spieles and Mitsch, 2000; Collins, 2008). For Deep Bay, six variables are considered: total dry biomass of all macroinvertebrate species, total number of species, Community Taxonomic Similarity Index (Fang, 1999), proportion of dominant species, proportion of Oligochaetes, and proportion of *Capitella capitata*. As an integrated parameter, ICI assesses the biological integrity of wetland by aggregating all the variables (Fang, 1999). More details about the calculation of ICI can be found in Hicks (1997) and Fang (1999). In this paper, ICI is used as an indirect evidence to show the availability of RAITWS.

Figure 5 presented respectively the O/E ratios from

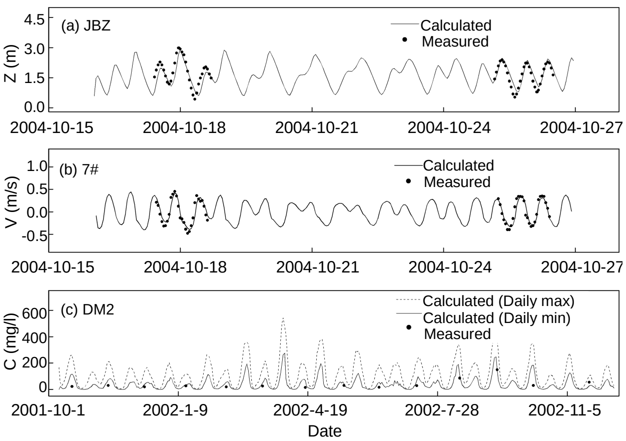

Figure 6. Comparison of calculated and measured tidal elevation (Z), velocity (V) and concentration of suspended solid (C) (the locations of points, JBZ, 7# and DM2, are shown in Fig. 1; Daily max and Daily min indicate daily maximum and minimum suspended sediment (SS) concentrations, predicted by EFDC).

RAITWS and ICI obtained for each site in April 1997, 1998, and 1999. The general variation trends were similar, both indicating better sediment quality at Section A than Sections ET and RB. Although Site A1 experienced an environmental impact in April 1997, it had recovered in April 1998 and 1999. The sediment quality at Section ET had worsened from April 1997 to April 1998, but bettered during 1998 ~ 1999. Section RB remained affected throughout the period of interest.

Although both RAITWS and ICI can give satisfactory results, the RAITWS permits quick judgment as to whether the environmental quality is degrading since it originates from the Rapid Biological Assessment (RBA) method (Adams et al., 1992) and thus inherits the major advantages of RBA. RAITWS also considers the dynamic behavior of the various environmental variables included in the assessment. Moreover, RAITWS is considerably cheaper (Ni et al., 2012). Given its speed and effectiveness as a technique for assessing the intertidal wetlands, RAITWS should be more suitable for implementation in developing countries, where field data and analysis resources are scarce.

3.4. Interpretation of Sediment Quality Changes

Based on the estimated O/E ratios shown in Figure 4, the sediment quality of wetland at Futian National Natural Reserve and Mai Po Marshes worsened from 1996 to 1999. According to the summary of meteorological observations (Hong Kong Observatory, 2000), it was found that no significant difference was likely to have occurred in the natural environmental conditions of the two entities in only 4 years. Moreover, Fang (1999) showed that clay, silt and sand fractions on the Shenzhen side were 51, 44 and 5%, respectively, while the corresponding observed results were 48, 50 and 2% on the Hong Kong side. The fractions of total organic carbon in sediment at the Section A, ET and RB were 2.62, 2.82 and 2.80%, respectively, indicating that there was no apparent correlation between the low O/E ratio and sediment size grading or total organic carbon (Fang, 1999). Hence, natural environmental conditions might not lead to the degradation of sediment quality. However, huge socioeconomic changes had taken place in the Deep Bay area in 1990s. The variation of O/E values might primarily reflect the disturbance of human activities. It was nevertheless impossible to identify any single human activity in the area that had a predominant impact on the quality of the wetland sediment. Instead, it was sensible to identify the major environmental variables that influenced the quality of sediment in Deep Bay. It was common knowledge that high concentrations of heavy metals, polychlorinated biphenyls, polycyclic aromatic hydrocarbon and other organic contaminants had adverse affects on macroinvertebrates (Long et al., 1995; Chapman and Wang, 2001; Balthis et al., 2010). However, based on the investigation by Liu et al. (2010), heavy metal pollution, rather than organic contaminants, was the primary pollution at the mouth of Shenzhen River (near Mai Po Marshes). Moreover, according to a study by the Laboratory of Environmental Toxicology of the University of Hong Kong (2003), heavy metals such as cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn) damaged macroinvertebrate populations and decreased the species number in Mai Po Marshes. For example, note that background values of Cd, Cu, Pb, and Zn in Shenzhen soil were 0.086, 11.1, 40.9 and 78.7 mg/kg, respectively, it was found that the heavy metal concentrations in sediments were high (0.418 mg/kg for Cd, 92 mg/kg for Cu, 60.4 mg/kg for Pb, 582 mg/kg for Zn) at the mouth of Shenzhen River in 1996. The concentrated heavy metal in sediment could be released and washed to the Mai Po Marshes (Shenzhen Municipal Water Service Bureau, 2000), due to Shenzhen River Regulation Program in 1997 (dredging the downstream of Shenzhen River). That could lead to the worst environmental degradation at ET in October 1997 (Wong, 2004). Furthermore, Figure 4 also indicated that the ecological degradation under anthropogenic activities in Mai Po Marshes was worse than that in its adjacent area, Futian National Natural Reserve. It was very interesting that neighbored areas (just separated by the Shenzhen River inlet) had quite different responses to the similar human interferences (heavy metal discharge). With this in mind, a dynamic numerical model, the Environmental Fluid Dynamics Computer Code (EFDC) (Hamrick, 1992), was used to simulate the accumulation rates of Cd, Cu, Pb and Zn in Deep Bay, and hence explained why the O/E ratios were lower at Sections ET and RB than at Section A.

Flow in Deep Bay was essentially tidal driven. Freshwater inputs to Deep Bay included discharges from the Shenzhen River, Dasha River, Yuen Long Creek, and Tin Shui Wai Creek. Table 3 listed the average discharges entering Deep Bay from these rivers during the wet and dry seasons (Peking University, 1995), which were taken as input data to the numerical model. Concentrations of heavy metals and input suspended sediment (SS) at the river mouths were also provided in Table 3 by Hong Kong Environmental Protection Department (2002) and Shenzhen Municipal Water Service Bureau (2000). Open boundary conditions included the tidal elevation at

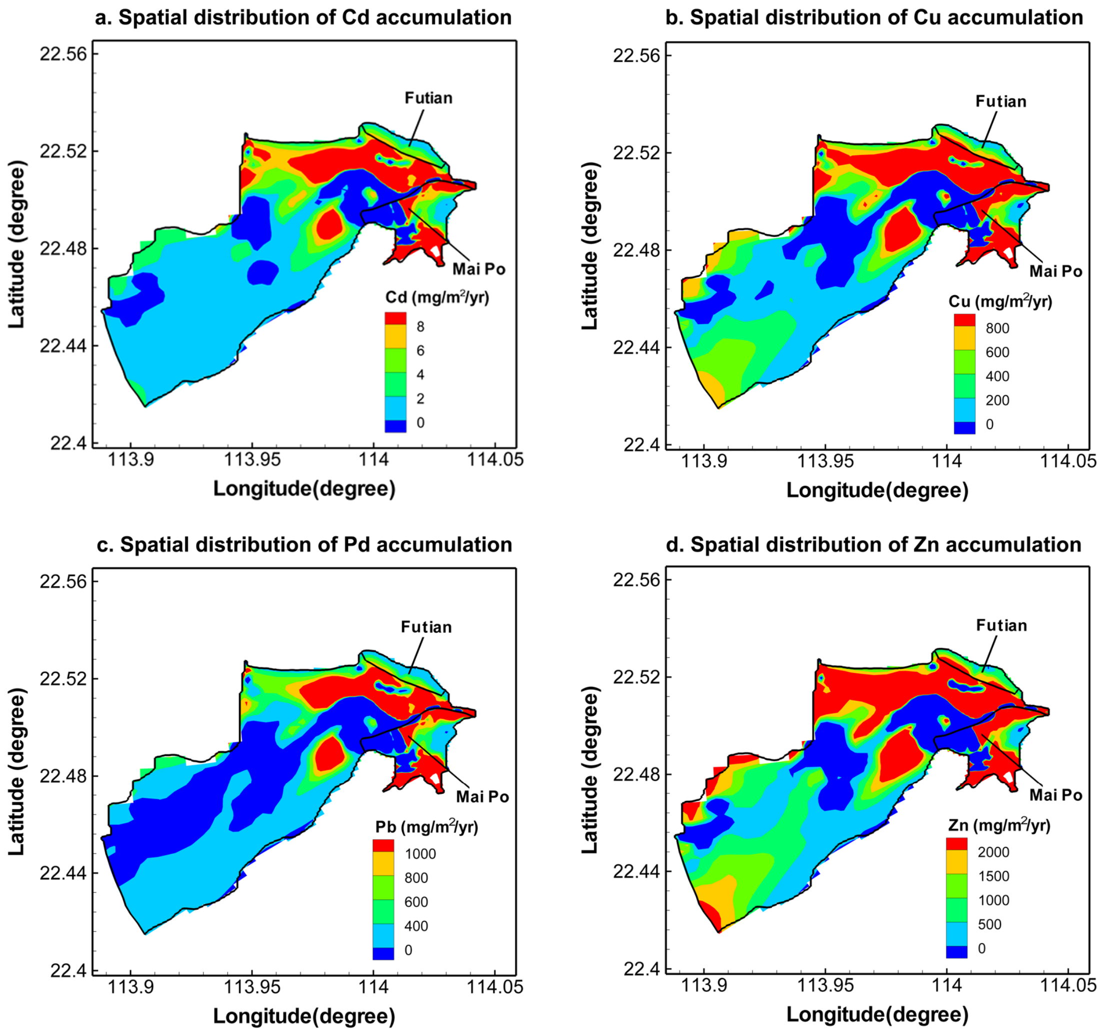

Figure 7. Spatial distribution of heavy metals accumulation on the bed of Deep Bay.

Chiwan station (National Marine Data and Information Service, 2001) which was used to prescribe the forcing condition at the entrance to Deep Bay. Field data on SS and heavy metal concentrations at Station DM5 (Hong Kong Environmental Protection Department, 2002) were also prescribed at the open boundary. The sediment density was specified as 2650 kg/m^3. The default values of the parameters in the EFDC model (Hamrick, 1992; Ji et al., 2002) were used.

The calibrated model was verified against independent measurements of water free surface motions, velocities and suspended matters. Figure 6 showed comparison of calculated and measured results on tidal elevation, velocity and suspended solid. The heavy metal accumulation rate could be calculated by the EFDC model (Hamrick, 1992). Figure 7 showed the simulated spatial distribution of annual accumulated heavy metals in mudflats in Deep Bay under the average discharges from all rivers. The modeled accumulation rates for cadmium, copper, lead and zinc in the mudflat sediment of Mai Po Marshes were about 9.6, 1.48, 1.57 and 3.79 g/m^2/yr, respecttively. Man et al. (2004) found that in mudflat sediments at Mai Po Marshes cadmium, lead and zinc in sediments ranged 0.5 ~ 0.8, 7.3 ~ 69.1 and 39.5 ~ 192.0 μg/g, respectively. Fang (1999) reported that the concentration of copper in the intertidal wetlands of Mai Po Marshes varied from 85 to 121 μg/g. Furthermore, Wong and Li (1990) found that the sedimentation rate of the inner Deep Bay was 14.9 mm/yr, and McChesney (1997) observed that the sedimentation rate of Mai Po Marshes was ranged from 25 to 30 mm/yr. Based on the above information, the accumulation rates of heavy metals could be estimated approximately by multiplying metal concentration and sedimentation rate (Peking University, 1995). The resulting accumulation rates from the literatures were 7.4 mg/m^2/yr for cadmium, 1.06 g/m^2/yr for copper, 0.7 g/m^2/yr for lead and 1.5 g/m^2/yr for zinc in the mudflat sediment of Mai Po Marshes. It seemed that the EFCD model simulation gave satisfactory but slightly higher corresponding average accumulation rates. The discrepancies might be from neglectting effects of grain size distribution (Larrose et al., 2010), degradation (Van Den Berg et al., 1999), bioavailability, volatilization, photolysis (Hu et al., 2007) and even flood event

(Coynel et al., 2007) in the complex interaction process. The idealized numerical modeling should incorporate biological processes if data allows.

Figure 7 indicated that most of the heavy metals accumulated at the eastern end of Deep Bay, in the vicinity of the river inlets. Given that the Shenzhen River, Yuan Long Creek and Tin Shui Wai Creek were the major sources of heavy metals, it was as expected that the rates of accumulated heavy metals in Mai Po Marshes (listed above) were more than in Futian National Natural Reserve (3.6 mg/ m^2/yr for Cd, 0.45 g/m^2/yr for Cu, 0.46 g/m^2/yr for Pb, 1.33 g/m^2/yr for Zn). Differences in accumulation rates were due to the higher current speeds during rising and ebbing tides at Futian National Natural Reserve, which caused higher sediment re-suspension values and lower heavy metal accumulation rates.

However, it was also reported that the speciation of the metals and their physico-chemical forms were very important when the effects of metals in the environment was considered (Jonnalagadda and Rao, 1993; Maret et al., 2003; De Jonge et al., 2008; Iwasaki et al., 2009). Only the bioavailable form could be used by macroinvertebrate, and the bioavailable metals affected the structure and production of benthic macroinvertebrate communities (De Lange et al., 2004). Fang (1999) found that the bioavailable metals in intertidal area of Deep Bay had a high correlation with the total metals. Peking University (1995) and Fang (1999) revealed that the bioavailable fractions of Cu, Pb, Zn and Cd were more than 55, 55, 45 and 50%, respectively. Moreover, it was found that the benthic macroinvertebrate had ingested heavy metals and significantly high heavy metals accumulation in benthic macroinvertebrate had been observed in Mai Po Marshes (Lam and Lam, 2004; Lai et al., 2005; Cheung and Wong, 2006). It was also reported that the heavy metals in sediment of Mai Po led to higher mortality of benthic macroinvertebrate in the ecotoxicological study of Mai Po sediments (Liang, 2007; Kwok et al., 2010). Furthermore, the Laboratory of Environmental Toxicology of the University of Hong Kong (2003) reported that heavy metals such as cadmium, copper, lead and zinc damaged macroinvertebrate populations and leaded to the decrease of the number of macroinvertebrate species in Mai Po Marshes. Thus, the EFDC simulations of the accumulation rates of heavy metals in the sediment helped to provide an explanation of the differences in macroinvertebrates communities between the different intertidal wetland sites. The accumulation rates of Cd, Cu, Pb, and Zn on the bed of Mai Po Marshes were respectively 2.7, 3.3, 3.4, and 2.8 times higher than for Futian National Natural Reserve, implying that the heavy metal accumulation rates at Mai Po Marshes were about 3 times more than at its adjacent Futian National Natural Reserve. Although benthic macroinvertebrates might be affected by some extreme events (Gamito et al., 2010), considering the long life cycle of benthic macroinvertebrates, it was likely that their responses to changes in the ambient environmental conditions would be chronic rather than instantaneous. Therefore, the long-term impacts of accumulated heavy metals on benthic macroinvertebrates provided a credible reason for the differences in O/E ratios obtained between the Shenzhen and Hong Kong sites.

4. Conclusions

Rapid Assessment of Intertidal Wetland Sediments (RAITWS) has been applied to assess the environmental impacts on sediment quality at sites located in Futian National Natural Reserve and Mai Po Marshes, along the coast of Deep Bay, China. Field data on sediments and macroinvertebrates obtained in the 1990s were utilized in the RAITWS, from which the average O/E ratios at Futian National Natural Reserve and Mai Po Marshes were determined as 0.74 and 0.48. By interpreting the O/E ratios, it was evident that the intertidal wetland in Futian National Natural Reserve remained environmentally stable, whereas the wetland in Mai Po Marshes deteriorated gradually from 1996 to 1999. The findings of RAITWS were in broad agreement with the ICI method, thus providing partial verification of RAITWS for the Deep Bay intertidal wetlands. RAITWS was considerably cheaper (in manpower and instrumentation requirements). Further study based on pollution source investigation and numerical simulations indicated that heavy metal, which might affect the growth of benthic macroinvertebrates, was the major pollution contributor and unevenly distributed in the study areas. The heavy metal accumulation rates at Mai Po Marshes were about 3 times more than at its adjacent Futian National Natural Reserve. This significant difference could lead to divergence between observed and expected reference bio-community structures.

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