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Sediment Quality Assessment in the Gulf of Gdańsk (Baltic Sea) Using Complementary Lines of Evidence

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Abstract Sediments from Polish coastal environments were classified by a quality assessment approach that took into account trace metal and organic micropollutant concentrations, grain-size distribution, and organic carbon content. Generally, no benthic organisms were found at sites where sediments were classified as heavily polluted. However, areas characterized by a moderate contamination showed a variable composition of the benthic community and changing bioaccumulation patterns; therefore, no single species found in the Gulf of Gdańsk could be considered representative of the whole benthic environment. Although sediment monitoring must be considered a suitable tool to detect hot-spot pollution areas in coastal and inland waters, it should be complemented by bioaccumulation measurements to evaluate the actual risk posed by contaminants to benthic organisms. This "biological information" allows a better appreciation of the real benthic infaunal community exposure to chemicals and can usefully complement the existing sediment quality guidelines.

Keywords Benthic organisms · Bioaccumulation · Marine coastal environments · Probable effects level · Probable effect level quotient · Risk assessment · Sediment Quality Guidelines

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Contaminated sediments have been identified in freshwater, marine, and estuarine ecosystems throughout the United States and Europe (Pavoni and others 1988; Szpunar and others 1997; McGee and others 1999; HELCOM 2003; Thompson and Lowe 2004). However, despite regular sediment quality assessments by European Union (EU) member states, it is difficult to provide a reliable estimation of the overall amount of contaminated sediments. As sediments provide habitat for many organisms, contamination by persistent and bioaccumulable chemicals is of concern for benthic communities. Moreover, sediments influence the environmental fate of many chemical substances in aquatic ecosystems by acting as potential sinks and sources of substances that have entered the aquatic environment. Therefore, aquatic organisms may be exposed to chemical substances both through their immediate interactions with bed sediments and by pollutant release into the water column.

However, it is difficult to provide a reliable estimation of the risk posed by chemicals occurring in the sediment because their bioavailability depends on the existence of steady-state conditions among the sediment, the interstitial water, and the water column. It is commonly known that estuarine zones in Europe influenced by point sources of pollution, primarily from municipal and industrial sources, present the highest levels of sediment-associated contaminants: mainly where the tidal range and offshore currents are limited, such as in the Mediterranean and Baltic seas (NOAA 1990; Volpi Ghirardini and others 1999; Micheletti and others 2004).

Historically, the impact of pollutants on the sediment environment was first evaluated chemically. Chemical characterization provides information on the pollutants present and their concentrations but does not tell anything about their potential to cause biological damage. Long and Chapman (1985) proposed the Sediment Quality Triad (SQT), which combines chemical data with laboratory bioassays and field community characteristics to evaluate the impact of pollutants on benthic organisms (Chapman and others 1991; Chapman and others 1996). However, field data are difficult to interpret, as the composition of the benthic community at different sites could change because of many environmental factors (e.g., salinity, depth, grain size, currents, organic matter content) and not necessarily because of the contaminants under examination. This is particularly true for changeable environments such as brackish waters which are typically unstable environments over space and/or time. In these cases the quality of sediments can be successfully assessed using Sediment Quality Guidelines (SQGs) applied to chemistry measurements to indicate possible ecotoxicological risks associated with sediments (McCauley and others 2000). The SQGs are intended to be either protective of biological resources, or predictive of adverse effects, or both. Nowadays the use of SQGs (Long and others 2006) for evaluating the ecotoxicological significance of sediment-associated chemicals is an important part of the protection and management of freshwater, estuarine, and marine ecosystems. North American researchers and environmental regulators were the first to develop a variety of SQGs for controlling coastal sediment contamination and assessing its impact on benthic communities (MacDonald and others 1996; Long and others 1995, 1998; Hyland and others 1999; Long and others 2006). In Europe, research and regulation of contaminated sediments have been less coherent, with individual member states of the EU developing SQGs and monitoring strategies in a largely independent manner, searching for trigger values to indicate when sediment quality may be impacting the ecosystem (Ahlf and others 2002; Environment Agency 2002). The EU Water Framework Directive, in force since 2000, does not specifically address sediment management; however, it provides a good opportunity to implement strategies of this type for the quality of coastal areas.

SQGs have been derived using both mechanistic and empirical approaches including equilibrium partitioning, the threshold effects level (TEL), the probable effects level (PEL), the apparent effects threshold (AET), "consesus-based" evaluation, and linear regression modeling (LRM) (for a complete review see Wenning and others 2005 and references therein). Over the past few years, criticism has been expressed about the use of SQGs in quality assessment because they were considered not protective enough of indirect effects through bioaccumulation (Wenning and others 2005); thus there is a need to assess the potential for sediment-associated contaminants to bioaccumulate and to evaluate any direct or indirect potential effects associated with bioaccumulation.

In the present paper these issues are discussed in the context of a case study carried out in the Gulf of Gdańsk (Poland), a typical example of a brackish water system (Kowalczuk and others 2005; Zettler and others 2007), characterized by steep latitudinal and vertical salinity gradients, where chemical monitoring of sediments and infaunal organisms was carried out.

Materials and Methods

Sampling and Chemical Analyses

Surface (0- to 10-cm) sediment samples were collected from 35 sites in the coastal zone of the Gulf of Gdańsk (Fig. 1) and in an area 180 km south of the Gulf of Gdańsk along the Brda River, an effluent of the Vistula river (sites 50, 51–52; not shown), in 2003 using an Eckman grab sampler. Loss on ignition (%LOI), DDT, PAH, PCBs, BT (butyltin compounds), As, Cd, Cu, Ni, Pb, Zn, and Cr concentrations reported here are those already discussed by Falandysz and others (2006), to which the reader is referred for the relevant analytical details. For this study, all samples were analyzed for their content of fine sediments (fraction < 63 μm) and for the level of Hg. Mercury was determined directly on solid samples (0.1 g dry sediment) using an automated Hg analyzer (AMA 254; Altech) according to U.S. EPA method 7473; CAS 7439-97-6 (http://www.epa.gov/waste/hazard/testmethods/sw846/pdfs/7473.pdf). Chinese certified geological reference material (GWB-stream sediment) was used to check the accuracy and precision of Hg measurements. Mean Hg concentration was 0.098 ± 0.019 μg g^{-1}, comparing favorably with the certified value of 0.100 ± 0.02 μg g^{-1}.

An aliquot of the collected sediments was initally sieved at 2 mm, followed by crushing and sieving at 250 μm to retain the infaunal organisms. When benthic organisms were present, the composition of the benthic assemblages was qualitatively described and soft tissues from at least six individuals of each species were pooled and immediately frozen. For analytical determinations, tissue samples were freeze-dried and analyzed for DDTs, PCBs, and BTs as described by Galassi and others (2008). According to the recovery efficiency the total analytical variability is assumed to be about 25%.

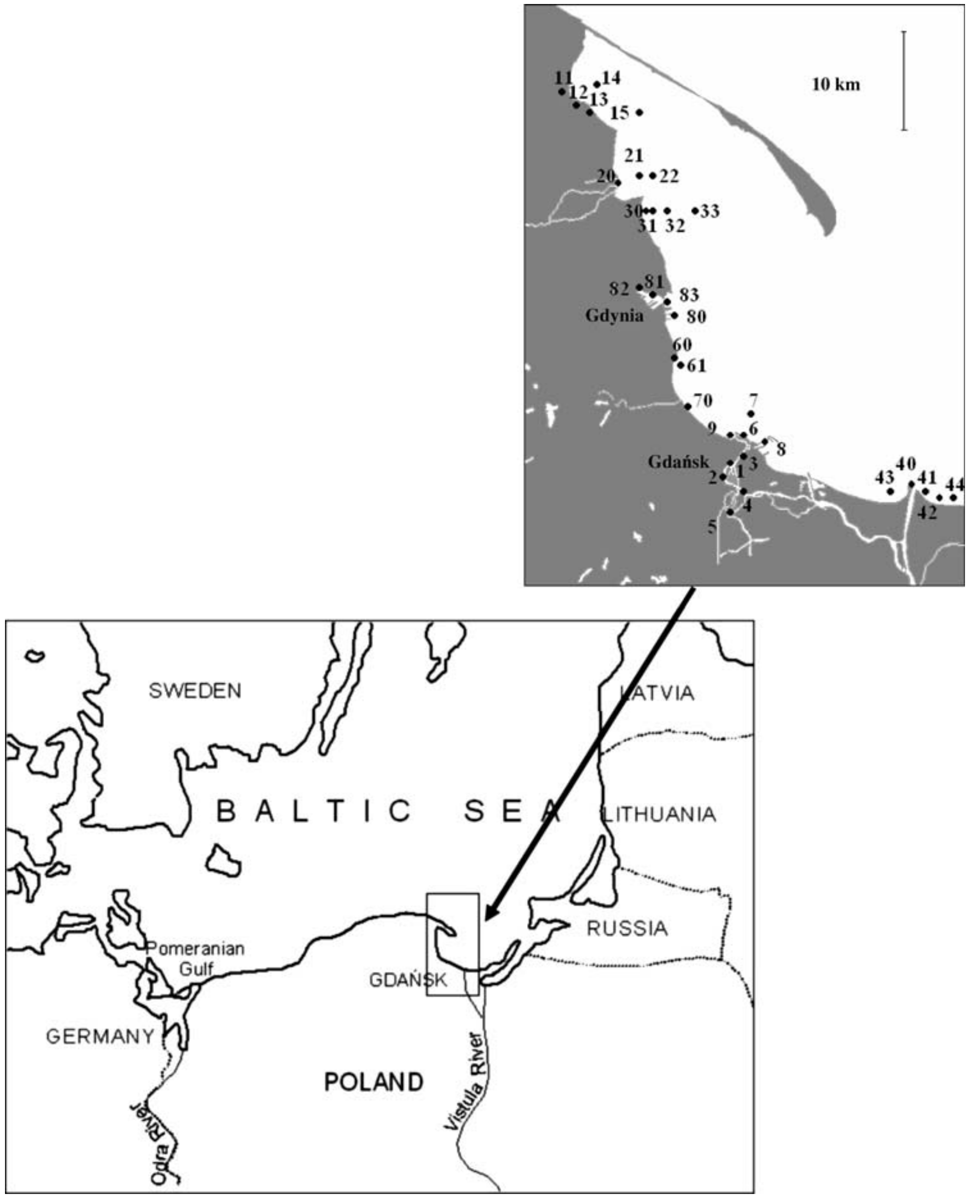

Fig. 1 Location of the sampling stations in the Gulf of Gdańsk

Statistical Analysis

The overall differences and similarties among sampling sites were evaluated using Q-mode cluster analysis. Raw data were centered and reduced, similarities between the various objects (sites) were then estimated with Euclidean distances, and the final clusters were determined with the "weighted pair group average" method (Davis 1986). Cluster analysis was restricted to those sites for which the full array of chemical analysis was available. The variables used in cluster analysis were DDTs, PAH, organic matter, fine fraction, As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn (see Table 1). PCB and TB were not considered because they were below detection limits at many sites. An arbitrary value of 4 ng/g Hg (the lowest measured for all sites) was used for reference sites to include this peculiar and important element in the overall analysis.

Results and Discussion

Occurrence of Benthic Organisms in the Gulf of Gdańsk

The highest number of species was at site 70, where ten species were found. At site 61, eight species were sampled and at sites 15, 20 and 80, seven species were collected. At the other sites, when present, a lower number of species was found (Table 1). No living organisms were present in the inner channels of the city of Gdańsk (sites 1 to 6;

Table 1 Number of benthic species, organic matter, and fraction <63 μm of collected sediments: pollutant concentrations (expressed as dry weight), probable effect levels (PELs), and PEL quotient (PEL-Q) at the different sites are reported

Site no.	No. of species	Organic matter (%)	Fraction <63 μm (%)	Total DDT (PEL = 51.7; ng g^{-1})	Total PAH (PEL = 16.8; μg g^{-1})	Total PCB (PEL = 189; ng g^{-1})	BTs (μg g^{-1})	As (PEL = 41.6; μg g^{-1})	Cd (PEL = 4.21; μg g^{-1})	Cu (PEL = 108; μg g^{-1})	Ni (PEL = 42.8; μg g^{-1})	Pb (PEL = 112; μg g^{-1})	Zn (PEL = 271; μg g^{-1})	Cr (PEL = 160; μg g^{-1})	Hg (PEL = 700; ng g^{-1})	PEL-Q
1	0	3.8	24.2	37.1	n.m	420	30	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	448	n.c.
2	0	10	19.5	28.4	52	230	7.2	8.40	2.4	180	20	190	550	63	169	1.10
3	0	7.5	28.2	6.9	12	21	4.1	7.00	0.75	67	26	130	240	64	198	0.48
4	0	8.5	35.3	14.1	5.9	52	2.8	7.30	1.8	120	19	150	590	81	465	0.70
5	0	4.9	27.5	14.5	5.7	<0.5	0.64	4.70	1.4	30	12	48	260	38	306	0.34
6	0	5.1	23.7	24.1	18	4	0.53	3.00	2.1	34	9.3	61	270	37	441	0.46
7	0	2.7	28.7	3	0.73	<0.5	0.18	4.30	0.37	13	9.5	21	61	29	45	0.12
8	1	0.4	6.6	0.77	0.2	<0.5	<0.05	0.92	0.08	1.7	2.3	7	13	11	6	0.03
9	8	0.5	4.5	0.95	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	9	n.c.
11	0	11.5	24.7	1.2	2.3	<0.5	<0.05	3.60	0.32	7.3	5.7	19	41	19	37	0.09
12	6	0.5	15.8	0.65	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	4	n.c.
13	4	2.2	22.0	0.96	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	5	n.c.
14	0	7.5	27.6	0.9	2.3	<0.5	0.063	3.00	0.26	5.8	5.6	14	366	15	71	0.20
15	7	3.9	23.8	0.11	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	9	n.c.
20	7	1.8	18.4	0.11	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	9	n.c.
21	1	1.5	8.4	2.8	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	6	n.c.
30	2	1.7	18.5	0.33	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	12	n.c.
31	0	0.5	19.4	0.13	170	<0.5	<0.05	0.85	0.11	1.9	1.7	5	14	6.9	4	0.94
32	6	0.4	18.8	0.7	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	4	n.c.
33	1	0.6	16.4	<0.05	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	12	n.c.
40	6	0.3	27.3	0.89	0.45	<0.5	<0.05	0.78	0.05	1	1.3	4.1	17	2.3	4	0.02
41	3	1.4	21.2	5.4	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	15	n.c.
42	2	0.7	18.2	1.8	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	13	n.c.
43	6	0.3	27.7	<0.05	0.22	<0.5	<0.05	1.7	0.06	1.4	1.9	5.1	17	4	5	0.02
44	6	0.3	21.0	0.34	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	5	n.c.
50	0.	15.6	12.0	16.6	8.8	<0.5	0.082	3.2	1.1	59	17	99	520	55	341	0.52
51/52	0	6.2	24.4	6.4	9.4	15	0.10	2.8	1.1	77	27	54	110	130	3404	0.82
60	3	0.7	24.9	0.36	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	13	n.c.
61	8	0.5	31.0	1.4	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	4	n.c.
70	10	1.7	29	0.15	n.m	<0.5	<0.05	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	n.m.	9	n.c.
80	7	0.4	11.1	0.43	0.25	<0.5	<0.05	1.3	0.12	14	3.2	21	29	11	23	0.06
81	0	1.6	18.5	5.7	1.3	<0.5	0.35	2.2	0.35	14	4.8	19	77	30	75	0.12
82	0	2.6	21.3	22.1	2.1	42	0.55	3.2	0.69	26	7.3	29	130	26	97	0.22
83	0	2.7	25.4	11.5	2.1	<0.5	0.63	3.7	0.66	23	7.7	36	140	29	169	0.20

Note: *n.m.* not measured, *n.c.* not calculated. No bottom fauna was found at the sites highlighted in gray. Underlined values indicate concentrations higher than PEL

Fig. 1), in the port of Gdynia (sites 81 to 83), at sites 31, 50, and 51–52, and in Puck Bay (site 11), where sediments were probably hypoxic since they are affected by municipal sewage (Galassi and others 2008). Hypoxic and anoxic situations can be critical for the survival of coastal benthic fauna as observed, e.g., in Scandinavian fjords, in the North sea, in Japan, on all three coasts of the United States, and in the Gulf of Mexico (Diaz and Rosenberg 1995; Rosenberg and others 2001).

Cluster analyses (Fig. 2) showed a clear separation between sites where no fauna was present and sites (8, 40, 43, and 80) where at least one species was found; these sites are far from direct input of effluents (Fig. 1) (Galassi and others 2008).

The analysis clearly separated site 31 (the only one with gross contamination with PAHs) and site 51–52, which was strongly contaminated with Hg (Table 1). All the other sites formed subgroups between one rather big cluster, reflecting both contamination patterns and changes in organic matter content.

Quality Assessment with Numerical Guidelines

Sediment Quality Guidelines (SQGs) are widely recognized as useful tools for assessing the quality of sediments of marine and freshwater environments (Shea 1988; Burton 2002). They consist of numerical values that help to interpret the impact on the benthic community of chemicals present in sediments, and were developed by considering the impairment caused by inorganic and organic micropollutants on benthic organisms (Long and others 1995, 2006; MacDonald and others 1996, 2004).

The concentrations of contaminants in sediments and the corresponding probable effects levels (PELs) proposed by MacDonald and others (1996) are listed in Table 1. The

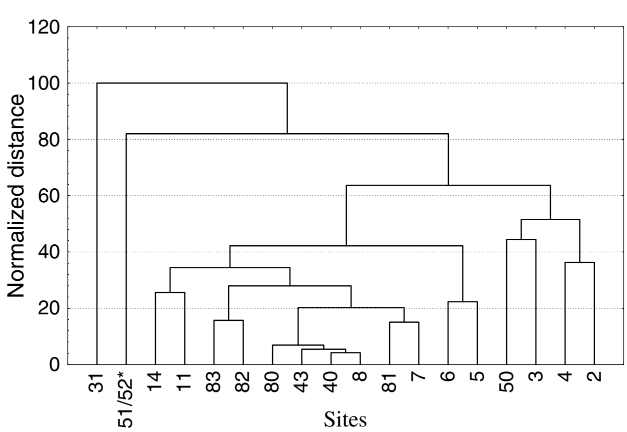

Fig. 2 Results of Q-mode cluster analysis for selected sites in the Gulf of Gdansk. Distances on the vertical axis are expressed as normalized Euclidean distances: (linkage distance/maximum distance) × 100

PEL for BTs was not available. No benthic organisms were found at sites where even a single PEL was exceeded, but at sites 7 and 11 bottom fauna was not found even though no PEL was exceeded. Probably the absence of organisms might be ascribed to some other causes, such as coarse pollution of urban origin, or other physical or biological sources of disturbance, such as the lack of oxygen or the presence of some predator. At sites where trace metal concentrations were determined (Table 1), the probable effect level quotient (PEL-Q), which is an estimate of the combined effects of all pollutants occurring in a given sediment, was calculated according to Fairey and others (2001). Combining the PEL-Q values with the presence/absence of biota at each sampling site, it appears that no benthic organisms were present at sites 2–7, 11, 14, or 81–83 (Table 1), with PEL-Q values > 0.06.

A study carried out by Hyland and others (1999) at 231 subtidal stations in southeastern U.S. estuaries revealed that the adverse effects of contaminants impaired 73% to 75% of the benthic communities when PEL-Q values exceeded 0.096. MacDonald and others (2004) observed reduced total abundance of benthic organisms for PEL-Q values ranging from 0.05 to 0.33. Therefore the PEL-Q threshold found in the present research, even if empirically derived, is within the range of values calculated using more refined statistical approaches (Hyland and others 1999; MacDonald and others 2004).

However, the vulnerability of benthic communities to different contaminants might vary from one geographical area to another, depending on the exact community composition. Moreover, a number of confounding factors such as depth, salinity, sediment texture, amount of organic matter, slopes, predation, and other biological interactions can be responsible for the different composition of the benthic communities (Hyland and others 2003). In the case of brackish/salty systems such as the Gulf of Gdańsk, it is very probable that an important confounding factor could be represented by the temporal and spatial changes of salinity. In this area horizontal and vertical salinity gradients are particularly pronounced and can affect both the composition of the benthic community and the species abundance (Kot-Wasik and others 2003; Zettler and others 2007). Macrobenthic communities of brackish environments often suffer abrupt changes in number of species and individuals as a response to abiotic stress (Zaret 1982).

Bioaccumulation of Organic Contaminants in Benthic Organisms

Because aquatic animals bioaccumulate persistent pollutants according to their physiological status, lipid content, age, metabolism, and trophic position, differences are expected among species with regard to their accumulation capability for chemicals.

Bioavailability of contaminants depends on the ability to be released from a sediment particle and transported across a biological membrane, where it may accumulate, with potentially toxic effects. Body burden is influenced by exposure history, the physical and chemical properties of the contaminant itself, and its degree of aging, as well as by sediment characteristics such as organic carbon content and composition and grain size distribution (Batley and others 2002). With SQGs the question of bioavailabilty is not overtly addressed because only total contaminant concentrations are typically used in the supporting data. At the moment even the use of normalization to organic carbon content of sediments seems not to improve the predictive ability of the SQGs (Batley and others 2002).

Bioaccumulation of contaminants by benthic organisms is actually considered a complementary line of evidence to SQGs to better understand the environmental risks associated with sediment pollution. To go deeper into this aspect we examined our results in two ways. First, the concentrations of TBTs, DDTs, and PCBs determined in different benthic species (Table 2), at the sites having at least three species in common (Galassi and others 2008), were compared not in terms of absolute values but as their relative abundances (ratios) in each species. If sediment characteristics would not influence the availability of these contaminants in the Gulf of Gdańsk, the relative abundances of pollutants in each species would be determined only by the characteristics of the species, that is, the ratio between the concentrations should remain constant at the different sampling sites. Our results revealed that *Crangon crangon* and *Platichthys flesus* maintained rather constant BT/DDT, BT/PCB, and DDT/PCB ratios. The same was not observed for the mussel *Mytilus trossulus* (Table 2), whose BT/DDT and BT/PCB ratios were extremely variable from site to site. This would reflect the different habitat of *Mytilus trossulus*, which lives attached to hard substrates and has a closer association with the water column and fresh organic material rather than with the bottom sediments.

Besides the substantial habitat differences between sites, which can affect, for example, the occurrence and distribution of *Mytilus* spp. depending on the availability of rocks or human artifacts, there is a vertical, horizontal, and depth gradient that characterizes the sandy shore (Chan and Caley 2003) and influences both species and pollutant occurrence and bioavailability. Correlations between concentrations of DDT and those of PCBs in sediments were not observed for any of the organisms sampled in this study. Bioaccumulated concentrations actually differed markedly for similar concentrations in sediments (Table 2). The bioavailability of hydrophobic chemicals is generally related to sediment organic carbon content and sediment size (Heather and others 1996; Micheletti and others 2004), but our results show that organisms sampled at two sites with very similar sediment compositions (i.e., sites 32 and 44) reached different concentration factors (CFs) for DDTs (Table 2). This finding would suggest that the organic matter content and the fine fraction of sediments are not the

Table 2 Accumulation and relative abundances (ratios) of BT, DDTs, and PCB for selected species and sites (see text for details)

Site No.	Species	BT (ng g^{-1} ww)	DDTs (ng g^{-1} ww)	PCB (ng g^{-1} ww)	BT/DDT	BT/PCB	DDT/PCB	CF of DDT
9	*Mytilus trossulus*	<20	0.95	4.28			0.2	1.0
	Crangon crangon	205	1.58	3.05	130	67	0.5	1.7
	Platichthys flesus	106	2.91	7.19	36	15	0.4	3.1
32	*Mytilus trossulus*	133	0.14	0.4	950	333	0.4	0.2
	Crangon crangon	168	1.63	4.56	103	37	0.4	2.3
	Platichthys flesus	60	3.85	15.29	16	4	0.3	5.5
44	*Mytilus trossulus*	13	1.09	3.35	12	4	0.3	3.2
	Crangon crangon	144	1.00	2.28	144	63	0.4	2.9
	Platichthys flesus	95	5.48	6.84	17	14	0.8	16.1
60	*Mytilus trossulus*	13	2.06	3.14	6	4	0.7	5.7
	Crangon crangon	212	3.21	2.92	66	73	1.1	8.9
	Platichthys flesus	61	8.96	17.09	7	4	0.5	24.9
70	*Mytilus trossulus*	34	0.75	3.41	45	10	0.2	5.0
	Crangon crangon	221	1.61	2.26	137	98	0.7	10.7
	Platichthys flesus	114	3.39	10.04	34	11	0.3	22.6

ww wet weight

According to recovery efficiency the total analytical variability is assumed to be about 25%. Concentration factors (CFs) for DDT are calculated as the ratio of concentrations in organisms vs. sediments only variables responsible for the bioaccumulation extent in these benthic organisms. Another possibility is that hydrological processes can prevent chemical equilibrium between sediments and biota from being achieved.

General Considerations for Future Monitoring in the Gulf of Gdańsk

Because no single species found in the Gulf can be considered representative of the whole benthic environment, it is probable that effective coastal monitoring cannot be carried out by simply making use of a single sentinel species. In fact, a misleading picture of the spatial distribution of pollution could be drawn when comparing a pollutant accumulated in mussels or in other sentinel species collected at sites separated by hundreds of kilometers, because small-scale differences are often much greater than those found on a regional scale. The results of previous monitoring studies using *Mytilus* spp. revealed that the Gulf of Gdańsk is much more polluted than other Baltic shores (Potrykus and others 2003). Organochlorine compound (OC) levels in *Mytilus* spp. collected from the Gulf of Gdańsk in 2003 ranged between 0.09 and 6.94 ng g^{-1} wet weight (ww) for pp'-DDE and from 0.40 to 11.61 ng g^{-1} ww for PCB. TBT contamination ranged between <20 and 133 ng g^{-1} ww (Galassi and others 2008). Since the lowest values for OC compounds and TBT measured in benthic organisms in the Gulf of Gdańsk are comparable to, or even lower than, concentrations measured outside the Gulf, in the southwestern part of the Baltic Sea (Albalat and others 2002; Potrykus and others 2003), a global judgment on the quality of this environment based on only one species should be avoided. Point pollution sources are responsible for the highest contaminant levels in sediments and organisms, and seem to affect only very limited coastal areas. Under such conditions, which are very common in coastal environments subjected to the influence of river inflow and urban wastewater discharges, it is practically impossible to find a single sentinel organism that can be considered representative of the whole benthic environment.

Conclusion

SQGs can give useful estimations of the potential hazard for aquatic fauna, but require additional evaluation of the actual ecological risk posed by different contaminants in complex environmental settings. The study of contaminant bioaccumulation by benthic organisms (biomonitoring) is generally seen as a complementary line of information to SQGs. However, none of the species examined in this study was suitable, alone, as a bioindicator to assess the risk of bioaccumulation of DDTs, PCBs, and BTs in the Gulf of Gdańsk, because the interspecific differences which were observed in the extent of bioaccumulation could not be explained on the basis of the lipid content and/or trophic position. A value of 0.06 for the PEL-Q was established as the threshold above which a "benthic desert" is found in the Gulf of Gdańsk. This value is at the lower end of PEL-Q ranges previously reported to have an adverse effect on benthic community composition in other geographical areas (e.g., Hyland and others 1999; MacDonald and others 2004). However, as advocated by the EU following the discussions around the implementation of the Water Framework Directive (2000/60/EC) (Brils 2008), sediment chronic toxicity based on representative species of the benthic community typical of the geographical area under consideration should be undertaken to evaluate whether the proposed threshold is protective enough for long-term exposures and adverse effects deriving from biomagnification through the trophic web.

Finally, better communication is needed about existing experience in EU countries with specific methods for how to derive SQGs and to use them in risk assessment or in pass/fail quality assessment steps. More insight is also needed into the benefits and disadvantages of these frameworks, thus enabling a better EU-wide discussion on the perspectives of a sediment management system based on SQGs.

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