EFFECT OF POLLUTION ON MARINE ORGANISMS

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Papers reviewed herein are limited to the biological, chemical and physical effects of environmental disturbance, natural, and anthropogenic, on marine and estuarine plants and animals. The worldwide literature is reviewed including textbooks, conference proceedings, original research reports, and reviews published in 2003. A special feature is a tabular listing of the body burden of inorganic and organic contaminants in plants and animals. The effects of pollutants on individual species and communities are included. Changes at the cellular and subcellular level as caused by contaminants are reviewed.

GENERAL

The abstracts of the 12th International Symposium on Pollutant Responses in Marine Organisms were assembled and edited by Jones (2004). The general subjects of the presentations indicated the current research into the effects of marine pollution and included these topics: mechanisms of toxicity, bioaccumulation, reproduction and development, biotransformation, biomarkers, multisensors, oxidative stress, cytochrome 450, hormonal effects, and genotoxicity.

CONCENTRATIONS OF TOXICANTS IN MARINE ORGANISMS

The uptake of silver by a marine diatom increased significantly with increases in the concentration of nitrate, ammonium, and phosphates. The increase was believed to be the results of greater algal growth (Xu and Wang, 2004). The impact of dietary and water exposure on the accumulation of cadmium in the polychaete *Capitella capitata* was investigated by Selck and Forbes (2004). They found that fed worms relocated proteins from mitochondria to the cytosol in response to cadmium exposure. Cutaneous uptake of cadmium was related to the surface area of the worm. In an experimental field study using mussels, Raspor, et al. (2004) found that natural factors, such as food supply and maturation of gametes, contributed more to the metallothionein content than sublethal levels of cadmium.

The biomagnification of organic pollutants in Antarctic food webs was studied by Goerke, et al. (2004). The top predator, the southern elephant seal, biomagnified the organic compounds 30-160 fold relative to krill with the exception of hexachlorobenzene which was low in fish indicating metabolic elimination. Two species of polychaetes and one species of pelecypods were exposed Rust, et al.,(2004c) to difference sources of polycyclic aromatic hydrocarbons (PAH). Bioaccumulation of PAHs from soot and tire rubber was significant but not from a coal derived source. The influence of soot carbon in reducing PAH sorption was examined using four species each of polychaetes and pelecypods. The soot treatment resulted in lower biomagnification of PAH than the control with no soot present. The results suggest that more emphasis should focus on the species characteristics rather than sediment geochemistry (Rust, et al., 2004b). The bioaccumulation of PAHs and polychlorinated biphenyl (PCBs) was measured in the clam *Mya arenaria* under different environmental conditions by Lohmann, et al. (2004). Since different uptake rates were measured when exposed to water or sediment, it is important to consider these conditions when attempting to estimate bioaccumulation in benthic organisms.

The bioaccumulation and transfer of persistent organochlorine contaminants in Arctic food webs was reviewed by Borgå, et al. (2004). The Arctic ecosystem is somewhat unique in that food chains are long and involve a reduced number of species in comparison to other geographical regions. Dietary accumulation of organochlorine is an important route for exposure for zooplankton and fish. There have only been a few attempts to model trophic transfers of these contaminants in the Arctic environment to date. Hecht, et al., (2004) measured nonhylphenol accumulation in three species of amphipods over a 16 day period. Bioaccumulation was inversely related to the amount of organic carbon present. The accumulation factor ranged from 4.6 to 33.9 in these amphipods which indicated that they could constitute an important source of nonylphenol to fish. The ability of 11 species of benthic invertebrates (6 polychaetes, 3 clams, 2 crustaceans) to metabolize and bioaccumulate benzo[*a*]pyrene was evaluated. The amphipods and one species of clams had a low metabolic level for this compound; in contrast, *Nereis virens* had the greatest ability of these 11 species to metabolize organic compound (Rust, et al., 2004a).

The response to different biomarkers was measured by Pérez, et al., (2004) along an untreated pollution gradient in Spain. A clam and a polychaete were used as sentinel species in this study. Acetylcholinesterase was the most sensitive biomarker to both species especially the polychaete *Nereis diversicolor*.

The concentration of metals, other elements, and organics present in marine organisms is listed in Table I. Data on the analysis of marine organisms are given according to element or organic compound, group of organism, geographical locality, and reference.

ASSESSMENT TECHNIQUES

A new method was proposed to classify soft-bottom benthic assemblages according to environmental disturbance categories as part of the European Union Water Framework Directive (WFD) (Rosenberg et al., 2004). The Benthic Quality Index (BQI) is based on a tolerance value for each species, the relative abundance of the species, and the mean number of species at the sampling station. Comparisons with existing data indicated that the BQI appears to be a useful tool for assessing the quality of European coastal waters under the WFD. Borja et al. (2004) recommend that an integrative index of quality be used to assess ecological quality under the WFD. The simple index is based on individual scores for water, sediments, and biomonitoring species, which can be weighted according to relative value of each index in providing an integrated assessment of overall ecological status of coastal waters. Borja (2004) discussed the advantages and disadvantages of using the AMBI (AZTI Marine Biotic Index) in the WFD framework. The Bentix index, which is based on the AMBI, is also discussed and a framework is proposed for evaluating the potential applicability of each index to assess benthic environmental quality. The use of artificial substrates (bottle brushes) was shown to be a useful, low-cost biomonitoring tool for evaluation of meiofaunal communities in coastal marine systems (Mirto and Danovaro, 2004). Meiofaunal densities reached stable levels on the substrates within 6 days after deployment and substrate assemblages were comparable to natural meiofaunal communities in both control and polluted areas.

Thompson and Lowe (2004) described a benthic assessment method for polyhaline and mesohaline assemblages in San Francisco Bay that uses information on number of taxa, total abundance, oligochaete abundance, number of molluscan taxa, number of amphipod taxa, and abundances of *Capitella capitata* and *Streblospio benedicti*. Comparison of the assessment results with the mean effects range-median quotient (mERMq) values for sediments indicated that samples with mERMq above 0.74 were always characterized, according to the authors' criteria, as impacted. Riba et al. (2004a) assessed sediment toxicity in the Atlantic coast of Spain using five separate sediment toxicity tests and six lesion categories in fish and clams. A multivariate statistical analysis (factor analysis) was then used to calculate sediment quality guidelines for metals and organic substances. Sediment concentrations (mg/kg d.w.) below which adverse effects are expected to be low or minimal were: copper, 209; mercury, 0.54; and PCBs, 0.054. The benthic foraminferan *Rosalina leei* was used as an indicator organism for detecting effects of mercury on sensitive marine organisms (Saraswat et al., 2004). After an exposure period of 35–40 days, there was almost no growth in test organisms exposed to mercury concentrations >20 ng/L, whereas control organisms continued to grow for periods up to 60 days. Field and laboratory studies using the mollusk *Donax trunculus* indicated that this species is useful in monitoring pollution effects in sandy beaches (Moukrim et al., 2004). Biochemical parameters useful for detecting pollution effects were inhibition of acetylcholinesterase activity and increased gluthation S-transferase activity.

Perez and Wallace (2004) found that the prey capture rate (of *Artemia*) by the grass shrimp, *Palaemonetes pugio*, collected from a creek impacted by landfills, was only about half the capture rate for reference shrimp. Reduced prey capture also occurred in grass shrimp that were collected from a reference area and exposed to the impacted sediments and water for a period of 8 weeks, demonstrating that the response index could be induced in healthy animals. An early life stage toxicity test using the Ambon Damsel (*Pomacentrus amboinensis*) showed promise as a sensitive assessment of the effects of toxic substances on fishes of the Great Barrier Reef (Humphrey et al., 2004). For the 144-h static test, sensitive responses included percent hatch, length at hatching, and the presence of developmental abnormalities.

MARICULTURE EFFECTS

Effects of a land-based fish farm in the western Mediterranean were detected at distances up to 500 m from the outfall using stable isotope techniques (Vizzini and Mazzola, 2004). The effects of fish waste were primarily detected as enriched ^{15}N ratios at several ecosystem levels, while ^{13}C ratios showed less alteration from the waste material. Macleod et al. (2004) reported that sediment quality improved rapidly after removal of Atlantic salmon cages in Tasmania, but that benthic faunal community structure was significantly different from reference conditions at 36 months following cage removal. Cage fish farms in the Canary Islands have influenced the structure of intertidal algal communities near the facilities (Boyra et al., 2004). Significant changes in intertidal flora near the farms included dominance by macroalgae such as *Corallina elongata* (red) and *Caulerpa racemosa* (green). Hartstein and Rowden (2004) found that benthic macroinvertebrate assemblages under mussel farm sites located in depositional environments in New Zealand were significantly different from assemblages in nearby coastal areas, with increased abundances of polychaetes and lesser abundances of ophiuroids. Alternatively, a mussel farm located in an area of higher currents showed no differences between the assemblages under the site and in nearby areas. Evaluation of influent and effluent water quality of shrimp ponds in Mexico indicated that there was an overall depletion of dissolved nutrients in the water (ammonium, nitrate, and reactive phosphorus) and a net export of organic particulate matter (Ruiz-Fernández and Páez-Osuna, 2004). Water quality within the ponds was more dependent upon environmental factors, including influent water quality, than on mariculture management practices.

FIELD SURVEYS

Using analyses of polychaete taxocene characteristics, Belan (2004) found that the area of greatest pollutant stress in Vancouver Harbor (Canada) was Port Moody Arm, an area also displaying high concentrations of chlorinated organic substances and metals. The area of maximum benthic effects was characterized by low species richness, low polychaete biomass, and dominance by *Tharyx multifilis*. Je et al. (2004) applied various univariate and multivariate statistical methods to Vancouver harbor benthic community data and also found that Port Moody Arm had the most significantly affected assemblages. A pollution gradient was described, ranging from reference conditions in Howe Sound into the inner harbor. Demersal fish assemblages in Port Moody Arm were also different than those in other areas of the harbor and were dominated by English sole, *Pleuronectes vetulus* (Levings and Ong, 2004). English sole from Port Moody Arm were feeding opportunistically on the abundant polychaetes and had lower condition factors than sole from other areas of Vancouver Harbor or the reference site. A 10-year monitoring program of a sewage sludge disposal site off the Mediterranean coast of Israel documented a relatively stable impact zone over time, with a heavily impacted area of about 2.5 km^2 (Kress et al., 2004). Benthic conditions near the outfall were azoic in the fall, but improved to high abundances of pollution-tolerant and opportunistic polychaetes in the spring following dispersion of sediments during winter storm events. Although the impact zone varied seasonally and annually, there was no evidence of long-term increases in the overall accumulation of sewage solids. Soft bottom mollusk assemblages in Ceuta Harbor (North Africa) had considerably higher species richness and diversity than assemblages in other nearby harbors in the Gibraltar region (Guerra-Garcia and Garcia-Gómez, 2004). The enhanced mollusk community in Ceuta Harbor results from a unique configuration with two opposing entrances that enables improved flushing and sediment quality when compared with conventional harbor configurations (i.e., with one entrance). Shin et al. (2004) presented an updated baseline of macrobenthic conditions in Hong Kong harbor based on 120 sampling stations located throughout the harbor. Much of the harbor is dominated by a homogeneous assemblage of polychaetes, with some areas of stressed assemblages that are related to low circulation or organic enrichment. A 10-year monitoring program at McMurdo Station, Antarctica, revealed that benthic assemblages near a sewage outfall were only about one third as abundant as at nearby reference sites, although diversity, evenness, and taxa richness were as high or higher than at reference locations (Conlan et al., 2004). In the past, benthic assemblages at McMurdo Station have been very responsive to changes in sewage discharges, indication that recovery may be rapid following improved sewage treatment that was initiated in 2003.

Levings et al. (2004) documented a variety of adverse effects of acid mine drainage (AMD) on the estuarine ecosystem of Howe Sound, British Columbia, including reduced algal cover, lower abundances of chironomid larvae and amphipods, and significant toxicity of sediments and water to invertebrates. The AMD in Britannia Creek has resulted in a disruption of the estuarine food web, especially as evidenced by reductions of invertebrate food items for juvenile salmonids. Following diversion of the AMD to an offshore outfall in 2001, the intertidal communities at Brittania Beach have shown some improvement (Zis et al., 2004). However, taxa richness, abundance, and growth of *Fucus* remain below reference levels as a result of continuing releases of metals from soils and low levels of AMD in Britannia Creek. A shoreline discharge of iron mine tailings occurring over 16 years in northern Chile has resulted in a soft-bottom assemblage dominated by *Lumbrineris bifilaris* (polychaete) and *Diastylis tongoyensis* (cumacean) (Lancellotti and Stotz, 2004). Notably, the assemblages were not characterized by typical opportunistic species such as capitellid, spionid, or cirratulid polychaetes, which may be sensitive to the constant turbidity of inorganic materials associated with the discharge. Dredged material disposal in Cleveland Bay, Australia, resulted in relatively rapid changes (over 15 days) in the soft-bottom macroinvertebrate assemblages within the disposal site (Cruz-Motta and Collins, 2004). However, the measured effects of reduced total abundances and total taxa recovered quickly, and were not significantly different from areas outside the disposal site by 3 months after the disposal event. Chou et al. (2004c) documented impacts of increased sedimentation resulting from construction activities on the soft-bottom macroinvertebrate assemblages near islands south of Singapore. Effects of changes in sediment particle size and nutrient content included reduced abundances and taxa richness and a dominance of the polychaete families Eunicidae, Flabeligeridae, Nereidae, and Glyceridae. In Bahia Blanca estuary (Argentina), concentrations of organochlorine pesticides were higher in sediments from beds of the burrowing crab *Chasmagnathus granulata* than in sediments from outside the crab beds, indicating that the crab beds were serving as sinks for these substances (Menone et al., 2004). However, total concentrations of organochlorine pesticides were lower in the crab burrows than outside the burrows, suggesting that bioturbation and metabolism by the crabs are influencing sedimentary distribution of these chemicals. Venturini and Tommasi (2004) found that polychaete assemblages in fine-grained sediments with high PAH concentrations in Todos os Santos Bay (Brazil) were dominated by carnivores rather than by deposit feeders. These results may reflect a lower potential exposure of PAH in sediments to carnivores, resulting in functional modifications of the assemblage in response to PAH contamination. In the northern Red Sea near Eilat, Wielgus et al. (2004) found that the most significant factors associated with coral mortality were concentrations of total oxidized nitrogen (NO_2 + NO_3) and the presence of SCUBA divers, while sedimentation rate did not have a significant relationship with coral mortality. The critical threshold for total oxidized nitrogen concentrations was 0.4 μM, above which there were significantly lower abundances of stony corals and higher mortality of coral colonies. During a period of no ferry activity in the Parramatta River (Australia), the macroinvertabrate assemblages in an area subjected to boat-generated waves became similar to those of a no-wash zone (Bishop and Chapman, 2004). Following resumption of ferry service, the wash-zone assemblages moved towards their previous state, indicating the temporary nature of effects resulting from vessel waves. The hard-substrate assemblages of a polluted coastal lagoon (Sacca di Goro) near the Po River Delta displayed considerable change following establishment of a canal through the sand bar at the lagoon mouth (Marchini et al., 2004). With increased flushing, the assemblages were more marine-related, with dominance by bivalves such as *Mytilus galloprovincialis, Crassostrea gigas*, and *Musculista senhousia*. Experimental application of the green alga *Enteromorpha intestinalis* onto seagrass meadows in a coastal estuary

(Australia) resulted in significant declines in the densities of seagrass and associated benthic invertebrate fauna (Cummins et al., 2004). Under the algal cover, larger taxa and deeper burrowing species were nearly eliminated, but the epifaunal gastropod *Potamopyrgus antipodarium* became very abundant.

Eby and Crowder (2004) found that localized hypoxic events did not adversely affect the overall structure of fish assemblages in the Neuse River Estuary (North Carolina). During local hypoxic events encompassing up to 65 percent of the study area, fishes moved out of the localized areas into other, oxygenated parts of the estuary. In an organically-enriched embayment in Brazil, identification of intertidal meiofauna at the genus level for nematodes provided a better correlation with environmental variables than using major taxonomic groups such as nematodes, copepods, and polychaetes (Somerfield et al., 2004). Variations in the nematode assemblage were most closely related to BOD of the overlying water and percent gravel of the sediments. Uriarte and Villate (2004) described pollution gradients in the Bilbao estuary (Bay of Biscay) based on comparisons with mesozooplankton assemblages in a nearby unpolluted estuary. In the Bilbao estuary, mesozooplankton abundances were enhanced in the outer euhaline zone, but depressed in the inner estuary as evidenced by lower abundances of copepods, gastropod larvae, and cnidarians. Recent decreases in the discharge of untreated sewage into inshore waters of the Thanet coast (Kent, U.K.) have not resulted in corresponding decreases in the high accumulations of drift alga such as *Ulva lactuca* (Rogers et al., 2004). The continued accumulations of driftweed in this area may be the result of an increased photic zone following reductions in the discharge of suspended particulates and the generalized eutrophic conditions of the southern North Sea.

MARINE DEBRIS

Marine debris and entanglement surveys were conducted in Hawaii, Jordan, Oman. Turkey, Brazil, Australia, New Zealand and Hawaii. Abu-Hilal and Al-Najjar (2004) surveyed macroscopic (> 2 cm) debris and its sources on Jordanian beaches, accompanied by a review of the world literature. The authors used shoreline methods to conduct monthly debris transects on three beaches in the upper Gulf of Aqaba, identifying 101,000 items. Most debris was from land-based sources and was dominated by plastic. Offshore dive surveys revealed that tires and metals were the most frequent debris in the coral reefs, followed by metal objects. Beach goers were the largest source of Jordanian marine debris. Following a storm runoff event in Santa Monica Bay, California, nearshore small debris volume was larger than plankton volume (Lattin et al., 2004). Surface and mid-water plankton net trawling in March 2001, collected about 1 plastic item per cu m before a storm, but up to 19 per cu m following a storm and runoff from storm drains. Between September 2001 and February 2003, McDermid and McMullen (2004) surveyed storm berm sediment samples for small (1-15 mm) marine debris at nine remote Hawaiian archipelago beaches. A beach at Midway Island was most contaminated, and a leeward beach on Oahu the least. A large percentage (43%) of the plastics were 1-2.8 mm, a size easily ingested by planktivores and surface feeding sea birds. Plastic ingestion by Brazilian sea turtles was reported for the first time by Mascarenhas et al. (2004). One live rehabilitating sea turtle defecated 11 pieces of hard plastic and nine pieces of plastic bags while at the rehab location.

Balas et al. (2004) used artificial intelligence methods (fuzzy logic and neural network) to predict degrees of acceptability of liter on beaches of the Turkish Riviera. Use of artificial intelligence allowed incorporation of qualitative descriptors as well as quantitative monitoring methods to determine "how clean is clean?" Claereboudt (2004) collected nearly 4000 items weighing 59 kg in two surveys of debris on 11 northern Omani beaches; concentrations were lower in comparison with other locations around the world. The dominant material was foreign-related and local-origin plastic debris, accounting for 61% of the items; cigarette butts were abundant at two sites. The authors cite an unpublished study indicating that more than 70% of turtle hatchings that failed to reach the ocean were entangled in fishing gear. Marine debris declined about 86% between 1991 and 1999 at a long-term monitoring shoreline in South Australia (Edyvane et al., 2004); however, the trend was halted by a sharp increase in 2000. Litter was dominated by fishing-related debris including netting, baskets, bait boxes and liquor bottles.

It is estimated that 1478 sea lions die per year in Australian waters by entanglement, thus slowing recovery of depressed populations. Implementation in 2001 of Australian Commonwealth Policy of Fishery Bycatch included reducing injury to marine life caused by lost shark and lobster gear in the Kangaroo Island zone (Page et al., 2004). Comparison of pinniped entanglement surveys conducted at rookery sites before and after implementation indicated that the Policy has failed to reduce entanglement rates. Entanglement rates are greater for sea lions than for fur seals, possibly due to the greater foraging depths of fur seals.

PETROLEUM, OIL SPILL AND PAH EFFECTS

Zentos et al. (2004) conducted chemical and biological sampling for 8 months at 14 soft bottom stations ranging in depth from 2 to 32 m following the September 2000 Eurobulker 700 t crude oil spill in Evoikos Gulf, Greece. One month after the spill water column petroleum concentrations were normal, but clams remained contaminated after sediment concentrations had declined. At the deeper stations sediments and clams (*Venus verrucosa*) experienced declining hydrocrabon concentrations and rapid recovery of the benthic macrofauna. However, at shallow inshore sites, sediment and clam hydrocarbon contamination increased during the course of the study, and there were delayed impacts on the benthic biota. Populations of echinoderms and crustaceans were impacted the most. Azevedo et al. (2004) in a 1996 survey found that aliphatic and polycyclic aromatic hydrocarbon concentrations in mussels, *Perna perna*, inside the entrance to Guanabara Bay, Rio De Janeiro, Brazil, were comparable to concentrations elsewhere in lightly contaminated urban bay environments.

In a series of avoidance experiments in static aquaria, Ryder, et al. (2004) offered Australian sea stars, *Patiriella exigua*, sediments spiked with Bass Strait crude oil ranging in concentrations from 62 to 1580 µg/g dry weight. Sea stars were introduced to contaminated sediments, and adjacent clean sediments, at 1, 2, 4, 8, 16 and 32 days and observed for avoidance behavior as the oil weathered. Avoidance increased with sediment concentration and decreased with time (oil ageing). Animals moved off of oiled sediments onto clean sediment or the glass walls, but as the oil aged and concentrations dropped the sea stars returned, still preferring clean sediment.

Mohammed and Agard (2004) used a new fluorometric microplate assay for investigating effects of Trinidad natural seep oil on cytoskeleton integrity in mussels. Although mixed function oxidase proteins are generally absent in bivalve mollusks, the authors found that populations of the bivalve, *Corbicula caribea* and exposed naturally to nearly 1800 ppm of total petroleum hydrocarbons, had high NADPH-ferrihemoprotein reductase activity. The reductase was most active in stomach, not intestine and the authors suggested that this was a detoxifying mechanism allowing the clams to thrive near seeps.

Fuller et al. (2004) evaluated the toxicity of mechanically dispersed Arabian medium crude oil, the dispersant Corexit 9500, and chemically dispersed Arabian oil, to larvae of two fish, juveniles of mysids, and luminescent bacteria (Microtox), under both continuous and declining exposure regimes at 25° C. Oil with dispersant was equal to or up to 20 times less toxic than oil alone. Continuous exposure was up to 20 times more toxic than declining exposure, such as expected to occur during dispersant operations. Un-weathered oil was more toxic than weathered. Hamoutene et al. (2004) exposed hemocytes of *Mytilus edulis* to oil, dispersants and chemically-disperse oil. Immune responses to oil were not altered by addition of Corexit 9527. Phagocytosis showed no response with diesel at 0.5 and 1.1 ppm but a slight decrease at 2 to 11 ppm. There was a non-significant decrease at 100 to 500 ppm Corexit 9527. There was no effect of diesel at below 8 ppm and Corexit below 200 ppm in 4 days exposure in vivo. The authors concluded that the dispersant Corexit 9527 was not likely affect mussel cellular functions if used in normal oil spill conditions.

Prudhoe Bay crude oil from the 1989 Exxon Valdez oil spill persisted for five years in sediments underlying mussels beds in Prince William Sound. In 1994, sediments at nine mussel beds were cleaned by first removing mussels and then relocating oily sediments into the wave zone where they were washed: the "cleaned" sediments and mussels were then replaced (Carls, et al., 2004). Monitoring through 1999 (5 years after cleaning) confirmed that, while this particular clean-up strategy resulted in a short-term reduction in sediment oil concentrations, there was no long-term term benefit.

Voparil et al. (2004) studied the ability of nine *Arenicola marina* digestive fluids, and an organic solvent, to extract 12 PAH's from Boston Harbor sediments and anthropogenic particles from Standard Reference Material tire tread, diesel soot, urban sources, coal dust and fly ash, all of which contaminate marine sediments near urban areas. Venturini and Tommasi (2004) used sediment physical/chemical characteristics, and NOAA sediment quality guidelines for PAHs to classify surface sediments at 28 sites representing PAH pollution gradients in Todos os Santos Bay, Brazil, then compared the results with polychaete community trophic structures from the same sites. Moderately silty sediments with high PAHs were dominated by a few species of carnivores instead of the predicted abundance of subsurface deposit feeders. Subsurface deposit feeding polychaetes dominated sites with the lowest PAH concentrations. In a detailed review of North Sea drill cuttings Breuer et al. (2004) noted that elevated hydrocarbons and elevated trace metals have remained relatively unchanged for decades and that following rig decommissioning, the elevated concentrations will persist until disturbed by physical reworking and bioturbation which may open new pathways for exposure, bioaccumulation and effects on marine organisms.

VESSEL COATINGS/ANTIFOULING PAINT

In 1999 an international science team under PICES surveyed 15 sites in Victoria and Vancouver harbors, British Columbia, Canada, to determine the status and trends of imposex in neogastropods (dogwinkles, *Nucella lima*, *Nucella emarginata* and *N. lamellosa*) and tributyltin (TBT) concentrations (Hogiiguchi, et al., 2004). Dogwinkles were locally extinct in Vancouver Harbor; concentrations of TBT were as high as 173 ng/g wet weight (ww) on foolish mussels (*Mytilus trosselus) and* 2230 ng/g in horse mussels (*Tresus capax*) confirming high exposure. Dogwinkles were present at all Victoria sites with high frequencies of imposex (to 100%) but at low severity (Relative Penis Length, RPL, 19 to 33.6, depending on species). Victoria dogwinkle butyltin concentrations ranged from about 2 to 22 ng/g ww. Compared to previous studies, it appeared neogastropods were recovering from imposex as measured by low RPS indices and declining imposex in dire whelk (*Searlesia dira*). Inoue et al. (2004) exposed adult pearl oysters (*Pinctada fucata martensii*) to TBT for one week and found reduced embryo development success from the highest maternal exposure (0.19 µg/L). A separate test fertilizing unexposed reproductive products directly in a range of TBT concentrations in water also indicated depressed embryo success at 0.191 µg/L and no development at all at 0.374 µg/L TBT. Using publish bioaccumulation factors, the authors estimated that the lowest-observed effect would be 0.0088 µg/L and no-observable effect at 0.004 µg/L.

TBT in sediments may also threaten sea grasses. Jensen, et al. (2004) subjected Odense Fjord sea grass, *Ruppia maritima*, to TBT polluted sediments and to TBT spiked control sediments. Spiked exposures were at 2.85 and 11.1 µg TBT Sn/kg and ambient polluted sediment at 28.3 µg TBT Sn/kg. After 3-4 weeks net photosynthetic activity was reduced 60% and relative growth rate reduced 8 to 25%. Intra-experimental differences occurred which may have been caused by matrix effects. Kubota et al. (2004) used an accelerator particle induce gamma-ray emission and Rutherford backscattering spectroscopy to measure butyl-associated tin within cells of a TBT-resistant marine bacteria extracted from sediments in a ship's hold. The Sn concentration was 4 ppb. Terlizzi et al. (2004) measure imposex incidence and severity in subtidal muricid snails (*Hexaplex trunculus*) collected in 2002 from 13 Italian Marine Protected Areas(MPA). Imposex was recorded at all sites and in 8 of the MPA's all females exhibited imposex. The authors concluded that management of MPA's against pollutants must be coupled with more region-wide management of pollution sources. Using ring bioassays, Straw and Ritchoff (2004) tested the ability of North Carolina mud snails (*Nassarius obsoletus*) from high and low imposex populations to respond to reproductive pheromones. Most snails from the high imposex sites did not respond to gender specific pheromones.

De Wolf et al. (2004) added intersex, defined here as the development of numerous male features in females, in periwinkles (*Littorina littorea*) to the list of abnormalities caused by TBT. Forty animals from each of five sites in the Netherlands were sampled and examined for intersex prevalence. Intersex prevalence declined from 2000 to 2002, consistent with declining TBT concentrations. TBT-induced intersex incidence was quantified in Roe's abalone (*Haliotis roei*) from five sites near Perth in Western Australia (Sloan and Gagnon 2002). Prevalence of this intersex ranged from 0 % at a remote reference site but 13 to 50% at four sites with nearby commercial or recreational boating activity. Karlsson and Eklund (2004) found that five of six new biocide-free anti-fouling paints were toxic to red macroalgae (*Ceramiam* spp.) and a harpacticoid copepod (*Nitocra spinipes*). The authors designed a test system useful for grading toxicity or safety of future paint products. The test painting the bottom of Petri dishes; the dishes were initially washed in seawater to reduce solvents and preservatives and then allowed to leach for 14 days after which the waters were used for toxicity tests. Only a silicon-based paint did not induce toxic effects. The authors recommended producers use standard toxicity procedures to help develop safe biocides

ENDOCRINE DISRUPTION EFFECTS

Bateman, et al. (2004) offered a standardized system for ranking intersex condition (oocytes in testes) in fish so that researchers can compare results on studies of endocrine disruption in fish. Testes from flounder (*Platichthys flesus*) collected from around the United Kingdom, 1998-2002, were examined histologically and oocyte development state and their distribution throughout the testes evaluated. The Oocyte Severity Index, was developed to account for both development stage and distribution throughout the testes. Kirby et al. (2004) published the first report of multi-year trends in the incidence of vitellogenin (VTG) activity in fish. They sampled male flounder (*Platichtys flesus*) plasma from six United Kingdom estuaries in 1999-2001 and compared the results with data from 1996 and 1997. Over the entire six year period VTG concentrations ranged from <0.2 µg/ml to > 100,000. Fish from one estuary (Alde) maintained concentrations below detection, whereas those from another maintained very high concentrations until the year 2000. A treatment plant in this river experienced 30-fold reduction in nonylphenol inputs. A third estuary site (Mersey) showed declining concentrations possibly correlated with initiation of secondary sewage treatment.

MASS MORTALITIES, ABNORMALITIES, HAZARDOUS ALGAL BLOOMS

Four major fish kills (2.5 million juvenile menhaden) in 2000 in upper Rehoboth Bay, Delaware, were initially linked to low dissolved oxygen but subsequent monitoring suggested the cause may have been toxicity from hydrogen sulfide which began to be produced in recently dredged depressions. Storm events mixed the layers, producing concentrations of hydrogen sulfide as high as 400 µM in surface waters (Luther et al., 2004). Tang and Au (2004) reviewed marine fish kill incidents caused by the red tide alga *Chattonella marina*, noting that gill ultra-structure damage preceded mortality. They conducted an exposure experiment to determine modes of action in gold lined sea bream (*Rhabdosargus sarba*). Fish were exposed to cultures of either the toxic *C. marina* or the non-toxic *Dunaliellla tertiolecta*. There was no fish mortality in the *Dunaliella* exposure, but mortality started in two hours in *C. marina* cultures. Tests indicated that the fish dying from *C. marina* poisoning were experiencing ionic regulatory disturbance leading to severe osmotic stress. Up to 65% of the *Macoma balthica* in the Gulf of Gdansk, Poland, had deformed shells in surveys conducted in 1997 and 1998 (Sokolowski et al., 2004); abnormal clams had higher concentrations of trace metals than normal ones. Additional surveys through 2002 produced clams with neoplasia and chromosomal abnormalities.

SEDIMENT TOXICITY

Currently there are no sediment quality guidelines for chromium but Berry et al. (2004) offer an approach, based on acid volatile sulfide (AVS) and then tested it using 10-day water and spiked sediment *Ampelisca abdita* bioassay. NOAA's ERL and ERM sediment quality guidelines predicted that most of the tested sediments should have been toxic. However, sediments containing measurable AVS and spiked with either Cr (III) or Cr (VI) had low pore water chromium concentrations and were not toxic, suggesting the guidelines were overly conservative. Sediments void of measurable AVS were toxic. The authors proposed an AVS-based logistics regression model for predicting toxicity and outlined some of its limitations.

BIOMARKERS

Au (2004) reviewed14 pathological biomarkers as measurement parameters for marine pollution assessment. The utility, sensitivity, and specificity of biomarkers from external lesions to histology of various tissues in fish and bivalves were tabulated. Au and colleagues (2004) also studied the time course for ethoxyresorufin-*O*-deethylase (EROD) activity, a measure of cytochrome P4501A (CYP1A), plus adverse cytological impacts in fish exposed to dietary benzo[*a*]pyrene (B*a*P). Induced EROD activity in juvenile grouper (*Epinephelus areolatus*) indicated exposure only within a short time span (2 to 4 weeks). Cytological changes, as indicated by increased lipopigments, remained high during the entire four weeks of exposure, but then dropped off after termination of exposure. The ability of tributyltin (TBT) to inhibit EROD activity in hepatic microsomes of two fish species (red mullet, *Mulles barbatus*, and flounder, *Platichthys flesus*) was demonstrated by Morcillo et al.(2004). Conjugation of testosterone and B*a*P metabolism, but not glucuronidation of estradiol, was also inhibited by TBT. Kutlu and Susu (2004) demonstrated how EROD can be responsive to more than just organic contaminants. Using gammarid amphipods (*Gammarus pulex*), their experiments showed how EROD activity can be reduced by exposure to lead, thus demonstrating how overall toxicity cannot be fully interpreted on the basis of simple or singular observations since the underlying processes can be quite complex. Miller et al. (2004) encountered such interpretational issues when applying CYP1A and EROD activity levels in English sole (*Pleuronectes vetulus*) in a bioassessment survey of a harbor. Although fish from two sites within the harbor exhibited elevated enzyme activities in concordance with higher sediment PAH and PCB levels, fish collected outside the harbor at the presumed reference site also exhibited high enzyme activity which could not be explained by the sediment chemistry. Additional data from fish collected at these sites (Stehr et al. 2004) did indicate that sole also had higher prevalence of neoplastic and preneoplastic liver lesions, plus high levels of aromatic hydrocarbons metabolites in their bile. These conditions were not observed at the reference site.

Five of the 11 recommended methods for environmental assessment by the U. S. Geological Survey were applied to largemouth bass and sediments from two sites in Mobile Bay (Annavarapu et al., 2004). EROD activity was similar between sites. The yeast estrogen screen (YES) assay for estrogenic impacts was negative for both sites as well. A sediment assay for EROD activity did find differences, especially in the polar aromatic fractions of sediment extracts, which correlated well with total PAHs and seven high molecular weight PAHs. The YES assay was also applied to investigate the impacts of sewage treatment plants as sources of estrogenic mimics in two rivers (Peck et al. 2004). While estrogenic activity in surface waters collected 1 km up- and down-stream from outfalls in both rivers were below detection, effluents from both outfalls did exhibit similar estrogenic activity. Sediments both up-and down-stream, though, had an order of magnitude greater activity than did the effluents, indicating that sediments can be a sink and a secondary source of estrogenic chemicals. Tom et al. (2004) explored aspects of fish metallothionein transcription, especially options for the requisite normalization, as an environmental biomarker. Bonacci et al. (2004) reported on the optimization of a microtitre-plate assay of esterase activity in gills of the Antarctic scallop (*Adamussium colbecki*), potentially providing a useful monitoring tool for xenobiotics in remote areas, such as the Antarctic.

Following the observations of genotoxic compounds in water and seston in a harbor for oil-tankers, Carrozzino et al. (2004) investigated as to whether genotoxicity would be revealed by induction of DNA damage in sea anemone (*Actinia equina*), as indicated by the Comet assay. *Actinia* was found to be quite sensitive and did detect the presence of sublethal concentrations of genotoxins following a four hour exposure to seawater. In another environmental assessment of genotoxicity, eiders (*Somateria mollissima*) from sites in the heavily contaminated Baltic Sea were estimated to have greater levels of genetic damage, compared to populations from the Beaufort Sea, as indicated by cell to cell variation in blood DNA content (Matson et al. 2004). Roling et al. (2004) worked on finding new biomarker probes of pyrene exposure in adult male mummichog (*Fundulus heteroclitus*). They assessed alterations in liver mRNA expression by subtractive hybridization and differential display following 7-day lab exposures. While some genes demonstrating differential expression were found and confirmed with quantitative-PCR, many of these were not expressed in feral fish captured from a creosote-contaminated site, in comparison to a reference site. Similarly, Maples and Bain (2004) investigated differential gene expression in mummichog collected from a site contaminated with trivalent chromium and reported several different genes being either up- or down-regulated. Kim and Lee (2004) assessed the developmental sensitivity of grass shrimp (*Palaemonetes pugio*) to model genotoxic compounds. From their results, they suggest that late embryo stages, which were able to survive to hatching but displayed a large number of DNA strand breaks, may have higher DNA repair capability than earlier stages which failed to survive but did not have strand breaks.

The effects of fluctuating contaminant levels were studied in mussels (*Perna viridis*) in the lab under various dosing regimes with four model PAHs and four organochlorine pesticides (Siu et al. 2004). Genotoxic impacts, as indicated by micronuclei (MN), were found to be dose responsive and correlated with tissue residues; were greater in chronic exposures that received the same net dose as acute exposures; and, persisted for a period of weeks, even after cessation of exposure. The ability of mussels (*Mytilus galloprovincialis*) to tolerate natural fluctuations in environmental factors was tested by Petrović et al. (2004). Survival in air, as well as lysosomal membrane stability, lipid metabolism, and contaminant-induced peroxidation of cell membranes were all adversely impacted at contaminated urban and industrial Adriatic sites, versus reference sites. Observations of cholinergic antagonists upon the bradycardia induced by copper in limpets (*Patella caerulea*) suggest that gastropod ACh receptors do not fit the vertebrate nicotinic-muscarinic classification (Pugliese et al. 2004). Comparisons between semipermable membrane devices and oysters (*Crassostrea gigas*) to a range of water only (ie filtered at 0.1 µm) doses of a mixture of PAHs show that both approaches demonstrate concentration of PAHs (Huckins et al. 2004). However, the relative amounts accumulated are different. Oysters tended to accumulate PAHs with log $K_{OW} \leq 4.8$ to a lesser degree than SPMDs. PAH compounds with log $K_{OW} \geq 5.6$ were accumulated to a higher degree in oysters in the low dose exposure than the higher doses, presumably due to active feeding at low doses and toxicity-induced cessation of feeding at high doses (10 versus 100 µg/l). Hwang et al. (2004) demonstrated that lysosomal destabilization in oysters (*Crassostrea virginica*) was a transient condition, and its reversal correlated with decreasing tissue residues as transplanted, contaminated oysters depurated in clean water. Luengen et al. (2004)

compared responses in two species of mussels (*Mytilus californianus* and *M. trossulus/galloprovincialis*) using components of the cellular immune system as indicators of contaminant stress. Transplants at estuarine sites which accumulated higher tissue residues of organics and some trace metals did exhibit induced or elevated phagocytosis. Results also showed the coastal species, *M. californianus*, had a higher degree of induced contaminant stress endemic estuarine species. Certain tissue and cellular alterations in the gill epithelium, as well as in some visceral tissues (palps and intestinal epithelium), of mussels (*M. galloprovincialis*) were found to be in general concordance with the presumed pattern of pollution among the six stations studied (Domouhtsidou and Dimitriadis 2004).

Nonylphenol, at environmentally realistic concentrations, was found to cause oxidative stress to clams (*Tapes philippinarum*) by affecting the anti-oxidant enzyme, superoxide dismutase (SOD) but not catalase, as well as greatly impacting the clam's ability to reburrow following exposure (Matozzo et al. 2004). Conversely, Geracitano et al. (2004) found catalase but not SOD to increase in 4-day assays of nereids (*Laeonereis acuta*) from either polluted or unpolluted sites. Following 14 day exposures, they reported polychaetes from polluted sites were more susceptible to oxidative stress as indicated by SOD, catalase, and glutathione *S*-transferase (GST) activities.

Verslycke et al. (2004a) reviewed the potential use of mysids as monitoring organisms for endocrine disruption. They reported on acute lethality values for an estuarine mysid (*Neomysis integer*) for a set of suspected endocrine disrupters (Verslycke et al. 2004b). Mysids were found to be sensitive to all with methoprene and fenoxycarb as the most toxic. Experiments on the endocrine regulation of glycemia levels in crab (*Chasmagnathus granulata*), a proposed biomonitor of exposure and stress in crustacea, addressed functional questions of how heavy metals (cadmium and copper) may bring about hypoglycemia (Medesani et al. 2004). Results suggested that cadmium inhibits secretion of the hormone involved, the crustacean hyperglycemic hormone (CHH) and does not interfere with CHH receptors. Copper's main interaction appeared to be the same, but some level of interaction with CHH receptors, or possibly an inhibition of other carbohydrate pathway enzymes, could not be ruled out. The egg yolk precursor, vitellogenin, in fish has become a standard measurement endpoint for assessing estrogenic impacts of pollution. Scholz et al. (2004) demonstrated through experiments with a model species, medaka (*Oryzias latipes*), how in vivo gene expression (via detection of mRNA) or quantification of protein levels displayed similar sensitivity and might be sufficient for short term studies. For longer term studies and for detection of weak estrogenic impacts, immuno-chemical detection was recommended. Volz and Chandler (2004) reported a new assay for vitellin in crustaceans. Their technique was capable of detecting vitellin is as few as four copepods (*Amphiascus tenuiremis*).

TOXICITY

Machado, et al. (2004) examined the associations between acid-volitive sulfices, reactive Fe, and the distribution of Cu, Cd, Ni, Pb, and Zn in sediment samples from river and bay sites in the Iguacu River estuarine system in Brazil. Bejarano, et al. (2004) used a 14-day exposure to show that the copepod *Amphiascus tenuiremis* reproductive output of nauplii, copepodites and cultuch size was affected by sediments from Hilton Head Island and Okatee River sites in eastern U.S. when compared to copepods exposed to pristine sediments. An ecological risk assessment conducted wtih the sediments from the lagoon of Venice, Italy, found the chemical contaminants of concern to be Hg, Cd, As, Ni, and PCBs (Micheletti, et al., 2004).

A 28-day partial life-cycle test with the amphipod *Leptocheirus plumulosus* was developed by McGee, et al. (2004). Acute and chronic tests were conducted by two organizations using gradients of sediment contamination and the results indicated good agreement between acute and chronic tests and field measurements of contamination. Lee, et al. (2004) investigated the influences of spiked Zn and equilibrations time on the partitioning of Zn between pore water and estuarine sediments containg two levels of acid-volitile solids. They found than Zn concentrations in pore water decreased during the initial 30 days and amphipod mortality also decreased. Anderson, et al. (2004) assessed amphipod (*Eohaustorius estuarius*) survival in laboratory, *in situ* exposures, and homogenized sediment samples. They determined that the interaction of contaminants and variable physical parameters, such as salinity and temperature, may result in lower survival in the *in situ* exposures. The results of a reduced volume/reduced overlying water acute marine marine amphipod (*Ampelisca abdita*) sediment toxicity test indicated that this test is a reliable and precise measure of acute toxicity in marine sediments (Ferretti, et al., 2004).

Geffard, et al., 2004a, conducted oyster (*Crassostrea gigas*) embryo bioassays using decanted sediments and overlying water and found that abnormal larval development was caused primarily by direct contact with contaminated sediments. A benthic diatom (*Entomoneis cf. punctulata*) was used to develop a whole-sediment bioassay that examined esterase enzyme inhibition. The found this test suitable for sediment quality assessments in marine and estuarine systems (Adams and Stauber, 2004). Ho, et al. (2004) reported a procedure that effectively uses powdered coconut charcoil to sequester and remove toxicity from sediments contaminated with organic substances.

Ovigerous crabs (*Chasmagnathus granulatus*) were exposed to Cu, Zn, and Pb by Lavolpe, et al. (2004) during embryonic development and observed eye, body pigmentary, and body morphological abnormalities in hatched larvae. Wu and Chen (2004) found 24-, 48-, 72-, and 96-hr LC50s for white shrimp (*Litopenaeus vannamei*) exposed to Cd and Zn, and examined the effects of these metals on oxygen consumption and on the distribution of these elements in the gills. Riba, et al. (2004b) exposed clams (*Ruditapes philippinarum*) to Zn-, Cd-, Pb-, and Cu-contaminated sediments to measure changes in sublethal and acute toxicities at altered pH or salinities levels. A metallothionein-like protein from killifish (*Heterandria formosa*) exposed to Cd was separated by Xie and Klerks (2004).

Virgilio and Abbiati (2004) used the polychaete *Hediste diversicolor* collected from three locations and did allozyme electrophoresis at siz loci to demonstrate that under Cu exposure, specimens with genotypes $ALD^{100/100}$ and $PGI^{102/102}$ had lower mortalities than other genotypes. Multispecies algal bioassays fror assessing cu toxicity with three marine (*Micromonas pusilla, Phaeodactylum tricornutum,* and *Heterocapsa miei*) and three freshwater (Microsys*tis aeruginosa, Pseudokirchneriella subcapitata*, and *Trachelomonas* sp.) microalge was developed by Franklin, et al. (2004). They determined that single species bioassasys may over- and underestimate metal toxicity in natural waters.

Chlorpyrifos exposure between 8 and 16 mg/L caused physicological damage to the endogenous rhythms of the Manila clam *Ruditapes phillippinarum* (Kim, et al., 2004). Mézin and Hale (2003) examined the effects of dissolved humic acids on the acute toxicities of chlorphrifos and DDT by using the freshwater crustacean, *Ceriodaphnia dubia* and the salt water mysid *Americamysis bahia.* They found that humic acid had no effect on the mortality of *A. bahia* to either pesticide at 20 ppt salinity, but greatly reduced mortality of *C. dubia* for both pesticides in freshwater. Chandler, et al. (2004) used the copepod *Amphiascus tenuiremis* in life cycle and 90-hr acute exposures to the insecticide fipronil and found 94 percent reproductive failure at 9.42 µg/L and a 96-hr LC50 of 6.8 µg/L.

Boudreau, et al., (2004) concluded that morphological abnormalities in mummichogs (*Fundulus heteroclitus*) were not sensitive indicators of exposure to estrogenic or antiestrogenic waterborne endrocrine disrupting substances at environmentally relevant concentrations. They exposed embryonic, larval, and juvenile stages of killifish to an estrogenic agonist (17*a*-ethynylestradiol) or antagonist (ZM198.154) and examined the incidence of skeletal abnormalities. Robinson, et al. (2004) used 28- and 6-month exposures of sand gobies (*Pomatoschistus minutus*) to 4-*tert*-octylphenol to demonstrate that such exposure resulted in elevated vitellogin mRNA expression, elevated plasma alkali-labile phosphate concentrations, and inhibited development of male nuiptial coloration and sperm duct glands. Embryos and preflexion larvae of the red sea bream (*Pagrus major*) were used by Kikkawa, et al. (2004) to demostrate the toxicity of carbon dioxide and acidification.

Vijayavel, et al. (2004) showed increased lipid peroxidation activity in the crab *Scylla errata* exposed to naphthalne, and decreased enzymatic (catalase, glutathione peroxidase, superoxide dismutase) and non-enzymatic antioxidants (vitamins C, E, and glutathione) for hepatopancreas, haemolymph and oxary. Zeolite column chromatograhy was demonstrated by Burgess, et al. (2004) to be effective at removing a wide range of ammonia concentrations under experimental marine conditions.

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Table 1—Chemical residues reported in marine organisms

Chemical	**Type of Organism**	**Concentration, µg/g dry wt**	**Locality**	**Reference**
Aluminum	Algae	5-4,167	Antarctica	Runcie and Riddle, 2004
		67-688	Antarctica	Ahn, et al., 2004
	Amphipod	103-435	Italy	Ugolini, et al., 2004
		277-502	Corsica	Ugolini, et al., 2004
	Reptiles	0.3-31.1[a]	Spain	Torrent, et al., 2004
		0.3-8.3[a]	Cannary Is.	Torrent, et al., 2004
	Birds	32.5-90.5	Canadia	Mallory, et al., 2004
		0-13.4	Beaufort Sea	Franson, et al., 2004
Antimony	Pelecypod	<0.005-0.27	Red Sea	de Mora, et al.,2004
	Fish	<0.001-0.002	Red Sea	de Mora, et al., 2004

		0.001-0.03	Caspian Sea	Agusa, et al., 2004
	Reptiles	0.009-0.18	China	Lam, et al., 2004
	Mammal	<0.01-0.02	Brazil	Kunito, et al., 2004
Arsenic	Agae	4-13	Anartica	Runcie and Riddle, 2004
	Polychaetes	36.5-1,054	Italy	Fattorini, et al., 2004
	Pelecypods	11.1-156	Red Sea	de Mora, et al., 2004
		0.7-25[a]	Spain	Saavedra, et al., 2004
		0.57-3.02[a]	China	Liang, et al., 2004
		1.5-2.8[a]	Norway	Airas, et al., 2004
		0.5-1.4[a]	Australia	Lincoln-Smith and Cooper, 2004
		10-68[a]	Mediterranean	Andral, et all, 2004
	Gastropods	5.17-53.8[a]	China	Liang, et al., 2004
	Cephalopods	2.1-8.9[a]	China	Fang, et al., 2004
	Fish	0.8-22.4	Red Sea	de Mora, et al., 2004
		12.6-212[a]	Mediterranean	Storelle and Marcotrigiano, 2004
		0-7.6[a]	China	Fang, et al., 2004
		8.2-290	Alaska	Meador, et al., 2004
		0.8-0.9[a]	Many sites	Foran, et al., 2004
		1.5-45.6	California	Meador, et al., 2004
	Reptiles	0.08-150[a]	Canary Is.	Torrent, et al., 2004

		1.3-19.6	China	Lam, et al., 2004
		0.08-122[a]	Spain	Torrent, et al., 2004
	Birds	1.8-104	Canada,Arctic	Mallory, et al., 2004
		<0.1-21	Japan	Fujihara, et al., 2004
		0.-0.74	Beaufort Sea	Franson, et al., 2004
		0.8-4.2[a]	Canada,Arctic	Braune and Simon, 2004
	Bird eggs	<0.5[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	0.74-2.4	Brazil	Kunito, et al., 2004
		0-0.4	Baja CA	De Luna and Rosales-Hoz, 2004
Barium	Fish	0.01-5.4	Caspian Sea	Agusa, et al., 2004
	Reptiles	0.7-3.8	China	Lam, et al., 2004
	Birds	0.11-0.7[a]	Alaska	Wilson, et al., 2004
		0.0.17	Beaufort Sea	Franson, et al,, 2004
	Mammals	<0.001-0.15	Brazil	Kunito, et al., 2004
Bismuth	Fish	<0.001-0.14	Caspian Sea	Agusa, et al., 2004
Cadmium	Algae	1.4-13.4	Antarctica	Runcie and Riddle, 2004
		1.4.5	Antarctica	Ahn, et al., 2004
	Sea Grass	0.2-1.6	Italy	Ancora, et. al., 2004
	Polychaetes	0.5-3.0	Adriatic Sea	Bocchetti, et al., 2004

	0.13-1.6	Naples, Ialy	Bocchetto. et al., 2004
Barnacles	7.15-12.8	Australia	Da Silva, et al., 2004
	4-16	Baltic Sea	Rainbow, et al., 2004
Amphipods	0.82-388	Italy	Ugolini, et al., 2004
	1.3-3.3	Corsica	Ugolini, et al., 2004
Decapods	0.4-14.9	Gulf of CA	Morales-Hernández, et al., 2004
	0.4-4	Gulf of CA	Ruelas-Inzunza and Páez-Osuna, 2004
Pelecypods	0.5-264	France	Bustamante and Miramand, 2004
	3-8	Baltic Sea	Rainbow, et al., 2004
	1.17-21.9	Red Sea	de Mora, et al., 2004
	5.6-28.5[c]	Baja CA	Segovia-Zavela, et al, 2004
	0.03-0.12[a]	Spain	Saavedra, et al., 2004
	0.1-6.1[a]	China	Liang, et al., 2004
	0.05-0.2	Norway	Airas, et al., 2004
	<1-2.6	Venezuela	LaBrecque, et al., 2004
	4.74-7.93	New Brunswick	Chou, et al., 2004b
	0.1-.0	Morocco	Banaoui, et al., 200
	0.5-6[c]	Mediterranean	Andral, et al., 2004
	4-52	Greece	Domouhtsidou,et

			al.,2004
	0.2-1.1[a]	France	Geffard, et al., 2004b
Gastropods	35-100	Israel	Siboni, et al., 2004
	0.2-22.3[a]	China	Liang, et al., 2004
	0.15-0.27	Antarctica	Ahn, et al., 2004
Cephalopods	0.036-1.6[a]	China	Fang, et al., 2004
Echinoderms	0.1-2.5	Belgium	Danis, et al., 2004
Fish	<0.001-0.23	Caspian Sea	Agusa, et al., 2004
	<0.001-195	Red Sea	de Mora, et al., 2004
	0.26-2.05	Portugal	Vinagre, et al., 2004
	0.001-0.1[a]	China	Fang, et al., 2004
	0.02-0.01	Baja CA	Bolton, et al., 2004
	0.02-5	Washington	Bolton, et al., 2004
	0.005[a]	various	Foran, et al., 2004
Reptiles	0.01-61[a]	Canary Is.	Torrent, et al., 2004
	0.01-61.1[a]	Spain	Torrent, et al., 2004
	0-175.5	China	Lam, et al., 2004
Birds	0.02-0.97[a]	Alaska	Wilson, et al., 2004
	0.1-23.3	Canada,Arctic	Mallory, et al;, 2004
	12-22[a]	Japan	Ikemoto, et al., 2004
	0-0.08	Beaufort Sea	Franson, et al., 2004
	5.6-10	Canada,Arctic	Braune and Simon, 2004

	Bird eggs	<0.08[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	0.09-69.6	Greenland	Dietz, et al., 2004
		0.004-56	Brazil	Kunito, et al., 2004
		0.003-4.25[a]	Japan	Yang, et al., 2004a
Cesium	Fish	<0.01-0.07	Caspian Sea	Agusa, et al., 2004
	Reptiles	0.4-0,46	China	Lam, et al., 2004
	Mammals	0.02-0.16	Brazil	Kunito, et al., 2004
Chromium	Algae	0.3-13.1	Antarctic	Runcie and Riddle, 2004
	Fungi	0-22.6[b]	India	Vala, et al., 2004
	Polychaetes	0.6-0.9	Adriatic Sea	Bocchetti, et al., 2004
		0.5-1.3	Naples, Italy	Bocchetti, et al., 2004
	Amphipods	1-19.7	Italy	Ugolini, et al., 2004
		14-182	Corsica	Ugolini, et. al., 2004
	Decapods	0.4-5.6	Gulf of CA	Morales-Hernández, et al., 2004
	Pelecypods	<0.01-3.5	Red Sea	de Mora, et al., 2004
		0.14-6.2[a]	China	Liang, et al., 2004
		<1-6.2	Venezuela	LaBrecque, et al., 2004
		0.1-1.0[a]	Australia	Lincoln-Smith and Cooper, 2004
		0.4-7.5[b]	Mediterranean	Andral, et al., 2004

		0.15-0.75[a]	Spain	Saavedra, et al., 2004
	Gastropods	0.08-0.78[a]	China	Liang, et al., 2004
	Fish	0.2-0.95	Caspian Sea	Agusa, et al., 2004
		<0.01-0.08	Red Sea	de Mora, et al. 2004
		0-0.7	Brit.Columbia	Bolton, et al., 2004
		0-0.9	Washington	Bolton, et al., 2004
	Reptiles	0-2.7	China	Lam, et al., 2004
	Birds	<0.5[a]	Canada,Arctic	Braune and Simon 2004
	Bird eggs	<0.5[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	0.2-3.9	Brazil	Kunito, et al., 2004
Cobalt	Pelecypods	0.15-12.9	Red Sea	de Mora, 2004
	Fish	<0.001-0.09	Caspian Sea	Agusa, et al., 2004
		<0.005-0.42	Red Sea	de Mora, et al., 2004
		0.5-0.8[a]	Various	Foran, et al., 2004
	Reptiles	0.5-11.4	China	Lam, et al., 2004
	Birds	0.3	Canada,Arctic	Mallory, et al., 2004
	Mammals	0.009-0.054	Brazil	Kunito, et al., 2004
Copper	Algae	1.2-49	Antarctic	Runcie and Riddle, 2004
		6.5-9.6	Antarctic	Ahn, et al., 2004
	Spermatophyte leaves	3-250[b]	New Jersey	Windham, et al., 2004

Spermatophyte stems	3-175[b]	New Jersey	Windham, et al., 2004
Polychaetes	19.4-30.3	Adriatic Sea	Bocchetti, et al., 2004
	7.2-27.2	Naples, Italy	Bocchetti, et al., 2004
	100-180[b]	Australia	King, et al., 2004
Barnacles	30-70	Baltic Sea	Rainbow, et al., 2004
Amphipods	47-63.4	Italy	Ugolini, et al., 2004
	32.6-54.7	Corsica	Ugolini, et al., 2004
Decapods	27-930	Gulf of CA	Morales-Hernández, et al., 2004
	20-450	Gulf of CA	Ruelas-Inzuna and Páez-Osuna, 2004
Pelecypods	1-507	France	Bustamante and Miramand, 2004
	6-12	Baltic Sea	Rainbow, et al., 2004
	3.1-276	Red Sea	de Mora, 2004
	11-152	Venezuela	LaBrecque, et al., 2004
	5.3-10.5	New Brunswick	Chou, et al., 2004b
	15-80[a]	Australia	Lincoln-Smith and Cooper, 2004
	5-72	Morocco	Banaoui, et al., 2004
	3.3-9.2[b]	Mediterranean	Andral, et al., 2004
	4-35	Greece	Domouhtsidou,et al.,2004

	6-21	Hong Kong	Chelazzi, et al., 2004
	1-8[a]	France	Geffard, et al., 2004
Gastropods	65-115	Antarctic	Ahn, et al., 2004
Cephalopods	2.4-21.1[a]	China	Fang, et al., 2004
Echinoderms	0.4-19.7	Belgium	Danis, et al., 2004
Fish	0.23-276	Red Sea	de Mora, et al., 2004
	0.62-3.9	Caspian Sea	Agusa, et al., 2004
	1.4-58.8	Portugal	Vinagre, et al., 2004
	0.1-1.95[a]	China	Fang, et al., 2004
	1.4-1.6	Brit.Columbia	Bolton, et al., 2004
	1.2-46	Washington	Bolton, et al., 2004
	0.8-0.9[a]	Various	Foran, et al., 2004
Reptiles	0.1-65.6[a]	Spain	Torrent, et al., 2004
	1-133	China	Lam, et al., 2004
Birds	9-225.3	Canada,Arctic	Mallory, et al., 2004
	3.6-5.1[a]	Japan	Ikemoto, et al., 2004
	0.04-2.56	Beaufort Sea	Franson, et al., 2004
	4.6-7[a]	Canada,Arctic	Braune and Simon, 2004
Bird eggs	0.7-1.8	Canada,Arctic	Braune and Simon, 2004
Mammals	14.3-1,970	Brazil	Kunito, et al., 2004
	3.7-23[a]	Japan	Ikemoto, et al., 2004

		1.35-60.9[a]	Japan	Yang, et al., 2004a
Gallium	Fish	<0.001-0.37	Caspian Sea	Agusa, et al., 2004
	Mammals	<0.001-0.027	Brazil	Kunito, et al., 2004
Indium	Fish	0.001-0.036	Caspian Sea	Agusa, et al., 2004
Iron	Algae	31-6,716	Antarctic	Runcie and Riddle, 2004
		172-1,730	Antarctic	Ahn, et al., 2004
	Sea Grasses	0-7,000	Italy	Ancora, et al., 2004
	Polychaetes	1,360-2,770	Adriatic Sea	Bocchetti, et al., 2004
		1,400-3,900	Naples,Italy	Bocchetti, et al., 2004
	Amphipods	233-540	Italy	Ugolini, et al., 2004
		231-481	Corsica	Ugolini, et al., 2004
	Decapods	9-512	Gulf of CA	Morales-Hernández, et al., 2004
		60-180	Gulf of CA	Ruelas-Inzunza and Páez-Osuna, 2004
	Pelecypods	90-517	Red Sea	de Mora, et al., 2004
		31-847	New Brunswick	Chou, et al., 2004b
	Gastropods	2.5-5.7	Antarctic	Ahn, et al., 2004
	Fish	2.4-2,866	Red Sea	de Mora, et al., 2004
	Reptiles	0.06-2,180	Canary Is.	Torrent, et al., 2004
		0.06-2,180	Spain	Torrent, et al., 2004
	Birds	239.5-2,270	Canada,Arctic	Mallory, et al., 2004

		320-460[a]	Canada,Arctic	Braune and Simon, 2004
	Bird eggs	31-45[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	250-1,470	Brazil	Kunito, et al., 2004
Lead	Algae	0.14-0.44	Antarctica	Ahn, et al., 2004
		0.9-21.5	Antarctic	Runcie and Riddle, 2004
	Sea Grasses	0.15-22	Italy	Ancora, et al., 2004
	Polychaetes	0.85-1.53	Adriatic Sea	Bocchetti, et al., 2004
		0.88-1.98	Naples,Italy	Bocchetti, et al., 2004
	Barnacles	15-78	Baltic Sea	Rainbow, et al., 2004
	Amphipods	0.5-3.5	Italy	Ugolini, et al., 2004
		1.3-3.3	Corsica	Ugolini, et al., 2004
	Decapods	0.7-5.5	Gulf of CA	Morales-Hernández, et al., 2004
		0.2-1	Gulf of CA	Ruelas-Inzunz and Páez-Osuna, 2004
	Pelecypods	<0.01-3.9	Red Sea	de Mora, et al., 2004
		0.05-0.4[a]	Spain	Saavedra, et al., 2004
		0.13-0.5[a]	China	Liang, et al., 2004
		0.3-1.8[a]	Norway	Airas, et al., 2004
		5-12	Baltic Sea	Rainbow, et al., 2004

	<1.5-4.9	Venezuela	LaBrecque, et al., 2004
	0.1-0.8[a]	Australia	Lincoln-Smith and Cooper, 2004
	0.5-27	Morocco	Banaoui, et al., 2004
	0.6-5.4[a]	Mediterranean	Andral. et al., 2004
	22-250	Greece	Domouhtsidou,et al.,2004
Gastropods	0.12-0.24[a]	China	Liang, et al., 2004
Cephalopods	0,02-0.06[a]	China	Fang, et al., 2004
Echinoderms	3.8-68	Belgium	Ancora, et al., 2004
Fish	<0.01-0.55	Red Sea	de Mora, et al., 2004
	2.16-8.9	Portugal	Vinagre, et al., 2004
	0.01-0.14[a]	China	Fang, et al., 2004
	0-0.23	Baja CA	Bolton, et al., 2004
	0.4-1.3	Washington	Bolton, et al., 2004
	0.04[a]	Various	Foran, et al., 2004
	<0.0001-0.55	Caspian Sea	Agusa, et al., 2004
Reptiles	0.02-33.1[a]	Canary Is.	Torrent, et al., 2004
	0.082-0.83	China	Lam, et al., 2004
	0.02-4/9	Spain	LaBrecque, et al., 2004
Birds	0.1-0.64	Alaska	Wilson, et al., 2004
	0.9-7.5	Canada,Arctic	Mallory, et al., 2004
	0.01-0.68	Beaufort Sea	Franson, et al., 2004

	Mammals	0.007-5.12	Brazil	Kunito, et al., 2004
		15-55	Baja CA	De Luna and Rosales-Hoz, 2004
Manganese	Algae	2.8-154	Antarctic	Runcie and Riddle, 2004
		450-560	Antarctic	Ahn, et al., 2004
	Sea Grasses	0-200	Italy	Ancora, et al., 2004
	Polychaetes	3.2-4	Adriatic Sea	Bocchetti, et al., 2004
		3.3-11.7	Naples, Italy	Bocchetti, et al., 2004
	Barnacles	50-200	Baltic Sea	Rainbow, et al., 2004
	Decapods	0.9-23.7	Gulf of CA	Morales-Hernández, et al., 2004
		0.5-11	Gulf of CA	Ruelas-Inzunza and Páez-Osuna, 2004
	Pelecypods	2.8-1,100	Red Sea	de Mora, et al., 2004
		15-105	Baltic Sea	Rainbow, et al., 2004
		19.7-24.3	New Brunswick	Chou, et al., 2004b
		0.6-3.4[a]	Australia	Lincoln-Smith and Cooper, 2004
	Gastropods	450-560	Antarctic	Ahn, et al., 2004
	Fish	0.21-2.7	Caspian Sea	Agusa, et al., 2004
		0.06-4.8	Red Sea	de Mora, et al., 2004
	Fish otoliths	0.19-35	Ireland	Brophy, et al., 2004

	Reptiles	0.2-28.8	China	Lam, et al., 2004
	Birds	1.4-19.6	Canada,Arctic	Mallory, et al., 2004
		0.02-2.09	Beaufort Sea	Franson, et al., 2004
		20043.2-3.8[a]	Canada,Arctic	Braune andSimon, 2004
	Bird eggs	<0.5[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	0.28-20.2	Brazil	Kunito, et al., 2004
		0.8-4.7[a]	Japan	Lang, et al., 2004
Manganese	Algae	1,496-16,942	Antarctic	Runcie and Riddle, 2004
	Birds	698-1,228	Canada,Arctic	Mallory, et al., 2004
		281-355	Beaufort Sea	Franson, et al., 2004
Mercury	Sea grass	0.015-0.09	Italy	Ancora, et al., 2004
	Amphipods	0.14-0.24	Italy	Ugolini, et al., 2004
		0.12-0.16	Corsica	Ugolini, et al., 2004
	Decapods	0.06-0.62	Gulf of CA	Ruelas-Inzunsa, et al., 2004
	Crustaceans	0.34-2.2	Canary Is.	Branco, et al., 2004
		0.96-2.1	Azores	Branco, et al., 2004
	Pelecypods	0.035-0.32	Red Sea	de Mora, et al., 2004
		0.02-0.09[a]	Spain	Saavedra, et al., 2004
		0.01-0.05[a]	China	Liang, et al., 2004

	0.008-0.025[a]	Norway	Airas, et al., 2004
	0.24-0.4	New Brunswick	Chou, et al., 2004b
	0.5-2.6	Morocco	Banaoui, et al., 2004
	0.05-0.34[b]	Mediterranean	Andral, et al., 2004
Gastropods	0.01-0.06[a]	China	Liang, et al., 2004
Cephalopods	0.4-1.4	Canary Is.	Branco, et al., 2004
	0.4-4.6	Azores	Branco, et al., 2004
	0.009-0.047[a]	China	Fang, et al., 2004
Fish	<0.05-3.5	Caspian Sea	Agusa, et al., 2004
	0.29-4.65	Red Sea	de Mora, et al., 2004
	0.16-1.8[a]	Canary Is.	Branco, et al., 2004
	0.16-1.2[a]	Azores	Branco, et al., 2004
	0.04-0.84[a]	Texas	Sager, 2004
	0.002-0.03[a]	China	Fang, et al., 2004
	0.07-3.4[a]	Florida	Adams, 2004
	0.03-0.04a	Various	Foran, et al., 2004
Reptiles	0.01-0.47[a]	Canary Is.	Torrent, et al., 2004
	0.001-0.47[a]	Spain	Torrent, et al., 2004
	0.0047-0.78	China	Lam, et al., 2004
Birds	0.68-2.1[a]	Canada, Arctic	Braune and Simon, 2004
	0.1-3.6	Canada, Arctic	Mallory, et al., 2004
	0.07-0.31[a]	Alaska	Wilson, et al., 2004

		36-94[a]	Japan	Ikemoto, et al., 2004
	Bird eggs	0.13-0.35[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	20040.58-59[a]	Japan	Ikemoto, et al., 2004
		0.3-3.16[a]	Japan	Yang, et al., 2004a
		1.1-290	Brazil	Kunito, et al., 2004
		0.26-9.9	Greenland	Dietz, et al., 2004
Molybdenium	Fish	<0.001-0.086	Caspian Sea	Agusa, et al., 2004
	Reptiles	0.02-1.2	China	Lam, et al., 2004
	Birds	0.03-0.78	Beaufort Sea	Franson, et al., 2004
		<0.5-0.8[a]	Canada, Arctic	Braune and Simon, 2004
	Bird eggs	<0.5[a]	Canada, Arctic	Braune and Simon, 2004
	Mammals	0.036-7.34	Brazil	Kunito, et al., 2004
Nickel	Algae	1-15.4	Antarctic	Runcie and Riddle, 2004
	Polychaetes	2.3-2.45	Adriatic Sea	Bocchetti, et al., 2004
		1.13-2.6	Naples, Italy	Bocchetti, et al., 2004
	Barnacles	10-75	Baltic Sea	Rainbow, et al., 2004
	Decapods	0.4-11.2	Gulf of CA	Morales-Hernández, et al., 2004
	Pelecypods	0.07-35.8	Red Sea	de Mora, et al., 2004

		0.3-2.5[a]	Spain	Saavedra, et al., 2004
		6-30.7	Venezuela	LaBrecque, et al., 2004
		0.03-0.14[a]	Australia	Lincoln-Smith and Cooper, 2004
		0.8-10.5[b]	Mediterranean	Andral, et al., 2004
	Fish	<0.01-0.11	Red Sea	de Mora, et al., 2004
		0.5-2.3	Br. Columbia	Bolton, et al., 2004
		0.4-25	Washington	Bolton, et al., 2004
	Reptiles	0.01-48.1[a]	Spain	Torrent, et al., 2004
		0.01-48.1[a]	Canary Is.	Torrent, et al., 2004
		0.15-1.07	China	Lam, et al., 2004
Rubidium	Fish	0.53-4.2	Caspian Sea	Agusa, et al., 2004
	Mammals	7.93-7.7	Brazil	Kunito, et al., 2004
Selenium	Mysids	1.5-5.4[b]	California	Schlekat, et al., 2004
	Pelecypods	1.0-12.8	Red Sea	de Mora, et al., 2004
		4.5-24[b]	California	Schlekat, et al., 2004
	Fish	2.1-22.6	Red Sea	de Mora, et al., 2004
	Reptiles	0.7-25.6	China	Lam, et al., 2004b
	Birds	18.5-93	Beaufort Sea	Franson, et al., 2004
		2.3-16[a]	Canada, Arctic	Baune and Simon, 2004
		1.2-14.7[a]	Alaska	Wilson, et al., 2004
		1.5-63.7	Canada, Arctic	Mallory, et al., 2004

		11-30[a]	Japan	Ikemoto, et al., 2004
	Bird eggs	0.6-1.1[a]	Canada, Arctic	Baune and Simon, 2004
	Mammals	3-190	Brazil	Kunito, et al., 2004
		1-25[a]	Japan	Ikemoto, et al., 2004
		0.3-5.3[a]	Japan	Yang, et al., 2004a
		0-4.5	Baja CA	De Luna and Rosales-Hos, 2004
Silver	Pelecypods	0.16-4.7[c]	Baja CA	Segovia-Zavala, et al., 2004
		0.02-0.35[a]	France	Geffard, et al., 2004b
		0.01-3.2	Red Sea	de Mora, et al., 2004
		0.02-0.15[a]	Spain	Saavedra, et al., 2004
	Fish	<0.001-0.06	Caspian Sea	Agusa, et al., 2004
		<0.001-0.7	Red Sea	de Mora, et al., 2004
	Birds	0.04-0.08[a]	Japan	Ikemoto, et al., 2004
	Mammals	0.12-1.3[a]	Japan	Ikemoto, et al., 2004
		0.01-1.7[a]	Japan	Yang, et al., 2004b
Strontium	Fish	0.44-12	Caspian Sea	Agusa, et al., 2004
		10[a]	Various	Foran, et al., 2004
	Reptiles	1-114	China	Lam, et al., 2004
	Birds	<1.5[a]	Canada,Arctic	Baune and Simon, 2004

	Bird eggs	1.6-3.3[a]	Canada,Arctic	Baune and Simon, 2004
	Mammals	0.13-1.58	Brazil	Kunito, et al., 2004
Tin	Pelecypods	0.02-0.4[a]	China	Liang, et al., 2004
		0.05-0.18[a]	Australia	Lincoln-Smith and Cooper, 2004
	Gastropods	0.02-0.09[a]	China	Liang, et al., 2004
	Fish	<0.001-0.015	Caspian Sea	Agusa, et al., 2004
		0.01[a]	Various	Foran, et al., 2004
	Birds	<0.5-0.5[a]	Canada,Arctic	Braune and Simon, 2004
	Bird eggs	<0.5[a]	Canada,Arctic	Braune and Simon, 2004
Thallium	Fish	<0.001-0.03	Caspian Sea	Agusa, et al., 2004
	Reptiles	0.001-0.02	China	Lam, et al., 2004
Vanadium	Pelecypods	0.28-7.3	Red Sea	de Mora, et al., 2004
		2-13.2	Venezuela	LaBrecque, et al., 2004
	Fish	<0.001-0.23	Caspian Sea	Agusa, et al., 2004
		<0.01-2.67	Red Sea	de Mora, et al., 2004
	Reptiles	0.28-1.4	China	Lam, et al., 2004b
	Birds	0.03-0.07[a]	France	Kammerer, et al., 2004
	Mammals	0.16-1.58	Brazil	Kunito, et al., 2004
		0-0.3[a]	Japan	Yang, et al., 2004a

Uranium	Pelecypods	0.01-0.5	Red Sea	de Mora, et al., 2004
	Fish	<0.01-0.5	Red Sea	de Mora, et al., 2004
Zinc	Algae	2.6-131	Antarctic	Runcie and Riddle, 2004
		23-37	Antarctic	Ahn, et al., 2004
		10-118	Ireland	Stengel, et al., 2004
	Sea grass	15-225	Italy	Ancora, et al., 2004
	Polychaetes	39.6-63.9	Adriatic Sea	Bocchetti, et al., 2004
		40.5-70.3	Naples,Italy	Bocchetti, et al., 2004
		250-300[b]	Australia	King, et al., 2004
	Barnacles	4,000-12,000	Baltic Sea	Rainbow, et al., 2004
	Amphipods	112-242	Italy	Ugolini, et al., 2004
		187-236	Corsica	Ugolini, et al., 2004
	Decapods	24-533	Gulf of CA	Morales-Hernández, et al., 2004
		10-100	Gulf of CA	Ruelas-Inzunza and Páez-Osuna, 2004
	Pelecypods	39-7,278	France	Mallory, et al., 2004
		60-200	Baltic Sea	Rainbow, et al., 2004
		69-4,290	Red Sea	de Mora, et al., 2004
		0.6-25[a]	Spain	Saavedra, et al., 2004
		20-900[b]	Australia	King, et al., 2004
		15-50[a]	Norway	Airas, et al., 2004

	55-266	Venezuela	LaBrecque, et al.,2004
	81.9-111	New Brunswick	Chou, et al., 2004b
	300-1,100	Australia	Lincoln-Smith and Cooper, 2004
	80-500	Morocco	Banaoui, et al., 2004
	130-200[b]	Mediterrean	Andral, et al., 2004
	24-48[a]	France	Geffard, et al., 2004b
	80-370	Greece	Domouhtsidou, et al., 2004
Gastropods	60-80	Antarctic	Ahn, et al., 2004
Cephalopods	7.5-20.1[a]	China	Fang, et al., 2004
Echinoderms	5.8-106	Belgium	Damis. et al., 2094
Fish	3.6-42.2[a]	China	Fang, et al., 2004
	11.6-68.9	Caspian Sea	Agusa, et al., 2004
	5.8-2,400	Red Sea	de Mora, et al., 2004
	15.7-175.9	Portugal	Vinagre, et al., 2004
Reptiles	0.05-216[a]	Spain	Torrent, et al., 2004
	105-255	China	Lam, et al., 2004
Birds	37.2-259	Canada,Arctic	Mallory, et al., 2004
	56-69[a]	Japan	Ikemoto, et al., 2004
	18.2-39	Beaufort Sea	Franson, et al., 2004
	34-48[a]	Canada,Arctic	Braune and Simon, 2004

	Bird eggs	12-15[a]	Canada,Arctic	Braune and Simon, 2004
	Mammals	22.4-43.6	Greenland	Dietz, et al., 2004
		111-575	Brazil	Kunito, et al., 2004
		22-82[a]	Japan	Ikemoto, et al., 2004
		2.3-46[a]	China	Yang, et al., 2004a
		20-60	Baja CA	De Luna and Rosales-Hoz, 2004
∑Butytins	Amphipods	0.002-0.46[a]	Japan	Takeuchi, et al., 2004
	Decapods	0.007[a]	California	Kannan, et al., 2004
	Pelecypods	0.01[a]	California	Kannan, et al., 2004
	Gastropods	0.001-0.002[a]	California	Kannan, et al., 2004
	Echiuroids	0.0065[a]	California	Kannan, et al., 2004
	Echinoderms	0.008[a]	California	Kannan, et al., 2004
	Mammals	0.004-0.05[a]	California	Kannan, et al, 2004
∑Chlordanes	Pelecypods	0.002[a]	California	Kannan, et al., 2004
	Gastropods	0.00003[a]	California	Kannan, et al., 2004
	Decapods	0.00003[a]	California	Kannan, et al., 2004
	Echinoderms	0.00003a	California	Kannan, et al., 2004
	Fish	<0.00001-0.25[a]	Sicily	Stefanella, et al., 2004
	Reptiles	0.04-0.25[c]	No.Carolina	Keller, et al., 2004b
		0.23-0.26[a]	No.Carolina	Keller, et al., 2004a
	Birds	0.006-0.009[a]	Greenland	Vorkamp, et al., 2004

	Bird eggs	0.0028-0.009	Greenland	Vorkamp, et al., 2004
	Mammals	0.04-12.4[c]	Gulf of Mexico	Struntz, et al., 2004
		0.026-0.5[a]	California	Kannan, et al., 2004
		0-260[c]	Gulf St.Lawrence	Metcalfe, et al., 2004
∑DDT	Decapods	0.0003[a]	California	Kannan, et al., 2004
	Pelecypods	0.012[a]	California	Kannan, et al., 2004
		0.0055	Black Sea	Kurt and Ozkoc, 2004
		<0.0001-0.15[a]	China	Yang, et al., 2004
	Gastropods	0.0008-0.004[a]	California	Kannan, et al., 2004
	Echinoderms	0.002[a]	California	Kannan, et al., 2004
	Fish	0.019[a]	Br. Columbia	Bolton, et al., 2004
		0.00009-0.03[c]	Pakistan	Munshi, et al., 2004
		0.002-4.73[a]	Sicily	Stefanelli, et al., 2004
		0.6-1.1[c]	Adriatic Sea	Storelli, et al., 2004a
		0.14-4.7[a]	Italy	Storelli, et al., 2004b
		0.019[a]	Br.Columbia	Bolton, et al., 2004
		0.05-0.23[a]	Washington	Bolton, et al., 2004
	Reptiles	0.14-0.58[c]	No.Carolina	Keller, et al., 2004b
	Birds	0.01-0.15[a]	Greenland	Vorkamp, et al., 2004
	Bird eggs	0.034-0.19[a]	Greenland	Vorkamp, et al., 2004
	Mammals	1,122-3,418[c]	Gulf St.Lawrence	Metcalfe, et al., 2004

		0.3-8.6[a]	California	Kannan, et al., 2004
		2.3-5[a]	Greenland	Dietz, et al., 2004
		0.2-5.5[c]	Australia	Evans, et al., 2004
		0.3-37.8[c]	Gulf Mexico	Struntz, et al., 2004
		0.15-1.7[c]	Baja Calif.	Valdez-Márquez, et al., 2004
		0.058-180[c]	Brazil	Kajiwara, et. al., 2004
Dieldrin	Decapods	0.00012[a]	California	Kannan, et al., 2004
	Pelecypods	0.001[a]	California	Kannan, et al., 2004
		0.0001-0.0008	Black Sea	Kurt and Ozkoc, 2004
	Gastropods	0.00004-0.0001[a]	California	Kannan, et al., 2004
	Echinoderms	0.00012[a]	California	Kannan, et al., 2004
	Fish	0-0.0055[c]	Pakistan	Munshi, et al., 2004
		0.0000.1-0.017[a]	Sicily	Stefanelli, et al., 2004
		0	Taiwan	Yuan, et al., 2004
		0.03	Gulf Mexico	Norena-Barrosa, et al. 2004
	Reptiles	0-0.035[c]	No.Carolina	Keller, et al., 2004b
		0.047-0.073[a]	No.Carolina	Keller, et al., 2004a
	Mammals	347-363[c]	Gulf St.Lawrence	Metcalfe, et al., 2004
		0.018-1.8[c]	Gulf Mexico	Struntz, et al., 2004
		0.0006-0.39[c]	Brazil	Kajiwara, et al., 2004

Heptachlor	Pelecypods	0-0.0016	Black Sea	Kurt and Ozkoc, 2004
	Fish	0	Taiwan	Yuan, et al., 2004
	Reptiles	0-0.1[c]	No.Carolina	Keller, et al., 2004b
	Mammals	0-2.1[c]	Gulf St.Lawrence	Metcalfe, et al., 2004
		<0.0002[c]	Gulf Mexico	Struntz, et al., 2004
∑hexachlorobenzene	Pelecypods	0-0.0003	Black Sea	Kuet and Ozkoc, 2004
	Fish	0-0.001[c]	Pakistan	Munshi, et al., 2004
		<0.0000-0.013[a]	Sicily	Stefanelli, et al., 2004
		0.009-0.0177[c]	Adriatic Sea	Storelli, et al., 2004a
		0.0006	Br.Columbia	Bolton, et al., 2004
		0.0007-0.03[a]	Washington	Bolton, et al., 2004
		0.05	Gulf Mexico	Noreña-Barroso, et al., 2004
	Reptiles	0-0.0027[c]	No.Carolina	Keller, et al., 2004b
	Birds	0.007-0.05[a]	Greenland	Vorkamp, et al., 2004
	Bird eggs	0.01-0.05[a]	Greenland	Vorkamp, et al., 204
	Bird blood	0.01-0.0.024[a]	Norway	Bustnes, et al., 2004
	Mammals	0.0014-0.4[c]	Brazil	Kajiwara, et al., 2004
		<0.0002-0.09[c]	Gulf Mexico	Struntz, et al., 2004
		0.6-1.0[a]	Greenland	Dietz, et al., 2004
∑hydrochlorohexane	Decapods	0.001[a]	California	Kannan, et al., 2004
	Pelecypods	0.001[a]	California	Kannan, et al., 2004

		<0.00008-0.006[a]	China	Yang, et al., 2004a
	Gastropods	0.0001-0.0006[a]	California	Kannan, et al., 2004
	Echinoderms	0.0007[a]	California	Kannan, et al., 2004
	Fish	0.00001-0.003[a]	Pakistan	Munshi, et al., 2004
		<0.00005-0.007[a]	Sicily	Stefanelli, et al., 2004
		0.003-0.008[c]	Adriatic Sea	Storelli, et al., 2004a
	Birds	0.0013-0.013[a]	Greenland	Vorkamp, et al., 2004
	Bird eggs	0.005-0.017[a]	Greenland	Vorkamp, et al., 2004
	Mammals	<0.001-0.55[c]	Brazil	Kajiwara, et al., 2004
		0.1-1.0[c]	Baja Calif.	Valdez-Márquez, et al., 2004
		0-0.3[c]	Australia	Evans, et al., 2004
		0.15-0.2[a]	Greenland	Dietz, et al., 2004
		0.003-0.08[a]	California	Kannan, et al., 2004
		70-108[c]	Gulf St.Lawrence	Metcalfe, et al., 2004
Mirex	Fish	0.00001-0.03[a]	Sicily	Stefanelli, et al., 2004
		0.07	Gulf Mexico	Noreña-Barroso, et al., 2004
	Reptiles	0.03-0.035a	No.Carolina	Keller, et al., 2004a
		0-0.044[c]	No.Carolina	Keller, et al., 2004b
	Mammals	6.4-10[c]	Gulf St.Lawrence	Metcalfe, et al., 2004

		0.046-1.2[c]	Gulf Mexico	Strantz, et al., 2004
∑Organochlorines	Decapods	0.0003-0.023[a]	Belgium	Voorspoels, et al., 2004
	Echinoderms	0.003-0.01[a]	Belgium	Voorspoels, et al., 2004
	Fish	0.0018-0.4[a]	Belgium	Voorspoels, et al., 2004
∑PCBs	Coral	0.006-0.037	Red Sea	El Nemr, et al., 2004
	Decapods	0.0003[a]	California	Kannan, et al., 2004
		0.0015-0.28[a]	Belgium	Voorspoels, et al., 2004
	Pelecypods	0.016[a]	California	Kannan, et al., 2004
		0	Black Sea	Kurt and Ozkoc, 2004
	Gastropods	0.0003[a]	California	Kannan, et al., 2004
	Echinoderms	0.0003[a]	California	Kannan, et al., 2004
		0.35-1.7[c]	Belgium	Voorspoels, et al., 2004
	Fish	0.023-3.2[a]	Belgium	Voorspoels, et al., 2004
		0.00001-0.06[c]	Pakistan	Munshi, et al., 2004
		0.007-4.7[a]	Sicily	Stefanelli, et al., 2004
		0.9-1.4[c]	Adriatic Sea	Storelli, et al., 2004a
		0.2[a]	Br.Columbia	Bolton, et al., 2004

		0.22-3.8[a]	Washington	Bolton, et al., 2004
		0.14-3.2[a]	Italy	Storelli, et al., 2004b
		0.1-1.6	Gulf Mexico	Noreña-Barrosa, et al., 2004
	Reptiles	0.6-2.8[c]	No.Carolina	Keller, et al., 2004b
		8.6-8.9[a]	No.Carolina	Keller, et al., 2004a
	Birds	0.015-0.23[a]	Greenland	Vorkamp, et al., 2004
	Bird eggs	0.078-0.33[a]	Greeland	Vorkamp, et al., 2004
	Mammals	1.3-79[c]	Brazil	Kajiwara, et al., 2004
		0.005-0.16[c]	Baja CA	Valdez-Márquiz, et al., 2004
		1.3-77.6[c]	Gulf Mexico	Strantz, et al., 2004
		0.3-1.9[c]	Australia	Evans, et al., 2004
		18.1[c]	Washington	Ross, et al., 2004
		1.1-2.5[c]	Br.Columbia	Ross, et al., 2004
		0.07-8.7[a]	California	Kannan, et al., 2004
		0.23-33.73[c]	Taiwan	Chou, et al., 2004a
PAHs	Polychaetes	0.008-4.16	Brazil	Venturini and Tommasi, 2004
	Pelecypods	0.002-12.1	Alaska	Boehm, et al., 2004
		0.05-0.42	Brazil	Azevedo, et al., 2004
		0.1-4.5	Australia	Lincoln-Smith and Cooper, 2004

	Fish	0.76	Gulf Mexico	Noreña-Barroso, et al., 2004
		0.0002-6.0[a,b]	Ireland	Jonsson, et al., 2004
Tributytin	Pelecypod	0.058-0.83	India	Bhosle, et al., 2004
		0.01-0.79	Portugal	Barroso, et al., 2004

a

[a]Wet weight

[b]Experimental study

[c]Lipid weight

