CHAPTER 1

Organic Contaminants in Sediments: Biological Processes

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INTRODUCTION

A wide variety of organic contaminants enter the environment through myriad sources, including inputs from industrial or municipal effluents, ocean dumping of wastes, terrestrial runoff, and atmospheric deposition. Many aspects of the physical and chemical processes affecting the transport and fate of contaminants within sediment are addressed in other chapters of this book on organic substances and sediments in water. This chapter focuses on the role of biological processes in sediments, either in the uptake of toxic chemicals by biota, the redistribution of sediments and sediment-bound contaminants by burrowing organisms, or microbial metabolism as a mechanism for remediating existing pollution. Biological processes are, of course, central to issues of environmental contamination. Our concern about toxic chemicals in the environment stems not from a mere academic interest in their chemistry, but rather a more anthropocentric concern for the adverse health and ecological effects of these compounds. In this volume, Pratt provides a pedagogical review of environmental toxicology, describing the current approaches to evaluating the effect of hazardous chemicals on ecological processes.[1]

Two major biological processes relevant to the toxic effects of sediment-associated contaminants are: bioaccumulation of contaminants from sediments (including the role of bioturbation in redistribution of contaminants to the bioactive zone), and biodegradation of toxic compounds within aquatic and aquifer sediments. In this chapter, we provide an overview of current understanding and major research directions being pursued in these areas. In this overview, the contributions of authors of subsequent chapters are highlighted.

BIOAVAILABILITY OF SEDIMENT-ASSOCIATED CONTAMINANTS

Many of the contaminants of ecological interest are compounds possessing characteristics that make them resistant to degradation and cause them to sorb strongly to particles. The movement of these particles to the sediments is the main route for the clearance of these persistent contaminants from the water column. These particles are subsequently transported to depositional areas in the particular water body by the existing current regime. Such movement results in the concentration of the contaminants primarily in the low-energy depositional regions of water bodies. These deposits can result in extremely high concentrations of contaminants in localized regions. Some of the more common areas for such concentration include harbors, which are generally areas of historic industrial contaminant discharge as well as areas of deposition. Further, by their construction or natural features, harbors are low-energy environments and often have very high rates of deposition. Such harbor areas have the highest levels of sediment-associated contaminants. In addition to their high levels of contamination, harbors are also areas requiring frequent dredging for navigational purposes. This dredging can result in the resuspension of buried contaminants.

In the context of dredging, the issue of contaminated sediments has been recognized for many years, and regulations have been established to prohibit the open-water disposal of contaminated materials dredged from harbors because of their potential impact on the biological community.[2] The significance of in-place or sediment-associated contaminants as sources of problems outside the dredging arena has been slower to be recognized. However, with the improvements of water quality through controls on discharges, these in-place contaminants are currently recognized as important contaminant sources equal to nonpoint runoff and atmospheric deposition.

In addition to harbor areas, natural depositional basins may also collect sufficient contaminant concentrations to result in alteration of benthic communities, as has been observed in the Great Lakes.[3] While most of the observed effects of contaminated sediments have been found in harbors and near shore areas, even the open-water depositional basins of the Great Lakes show sufficient contaminant concentrations to elicit effects on oligochaetes in laboratory bioassays.[4] From a regulatory perspective, it is often important to be able to determine the contribution of different sources of anthropogenic pollutants to bioaccumulation. In this volume, Sherblom and Eganhouse evaluate the use of source-specific marker compounds detected in biota to document and characterize the impact of municipal wastewaters on tissue burdens of specific contaminants.[5]

In addition to recognizing the impact of contaminated sediments on particular benthic species, sediment-associated contaminants have also been recognized as a potentially significant food chain contaminant source. This general recognition of the importance of contaminated sediments has highlighted the need to establish sediment quality criteria. These criteria must protect both the

indigenous benthic species that are directly exposed to sediment-associated contaminants and account for the accumulation of the contaminants through the food chain. Development of such criteria requires a fundamental understanding of the bioavailability of sediment-associated contaminants and the factors that control this bioavailability.

In order to understand bioavailability, a definition and a measure must be available so that comparisons can be made. Bioavailability of sediment-associated contaminants can be defined as "the fraction of the total contaminant in the interstitial water and on the sediment particles that is available for bioaccumulation,"[6] where *bioaccumulation* is the accumulation of a contaminant via all routes available to the organism. Chemical measures of contaminant concentration in sediment do not always reflect the bioavailable fraction of the sediment-associated contaminants;[7,8] therefore, a simple measurement of sediment residue is insufficient to describe the contaminant concentration to which biota are truly exposed. Two approaches have been described in the literature to describe the bioavailability of sediment-associated contaminants: (1) a comparison of the contaminant concentration in the organism and the sediment at steady state,[9-13] and (2) measurement of the uptake clearance of sediment-associated contaminants.[7,14-19] In spite of having two biological approaches available, no chemical approach is available to define the bioavailable fraction of sediment-bound residues. Thus, better approaches for describing the bioavailable fraction of sediments are needed.

The main factors that appear to affect bioavailability can be divided into two major areas: factors that alter the partitioning of contaminants to sediments, and biological factors that alter the exposure and accumulation of the contaminants.[20-23] In all cases, the physicochemical characteristics of the organic contaminant, primarily the compound's water solubility and hydrophobicity, are the major factors for determining its bioavailability. A complicating factor in any physicochemical assessment of bioavailability is the role of natural "dissolved" (or colloidal) organic matter (DOM) in sorbing hydrophobic organic contaminants and metals.[24] In general, organic contaminants sorbed to DOM are unavailable for uptake by biota.[14,25-27] However, predictions of the fraction of truly dissolved (i.e., bioavailable) pollutant in bulk or interstitial water is hampered by poorly understood variability in the contaminant binding capacity of DOM from different natural waters.[28-29] In this volume, Evans demonstrates that the DOM present in water from one Canadian lake has little effect on bioaccumulation of a polychlorinated biphenyl (PCB).[30] In contrast, Kukkonen et al. (also in this volume) examine a range of natural waters and demonstrate that the chemical and physical properties of DOM influence the extent of binding of polycyclic aromatic hydrocarbon (PAH) versus PCBs, but that the reduction in bioavailability remains correlated with physicochemical association of the contaminant with the natural organic macromolecule.[31] Södergren (also this volume) proposes an innovative monitoring tool to empirically estimate the bioavailable fraction of a contaminant in a natural system; the approach is based on the use of solvent-filled

dialysis bags to mimic bioaccumulation of organic contaminants by lipid pools of aquatic organisms.[32]

Accurate estimation of physicochemical partitioning of contaminants is not, by itself, sufficient to predict accumulation of environmental contaminants. Bioavailability of the sediment-associated contaminants needs to be understood as a complex interaction between physical, chemical, and biological characteristics that change with site, compound, and species of interest, thus making it difficult to understand and predict the bioavailability without detailed, site-specific information. Lee (this volume) illustrates this point by demonstrating the complicating influences of feeding behavior, burrowing characteristics, contact time of pollutants with sediment, and nonequilibrium sorption processes on bioaccumulation of organic contaminants in benthic invertebrates.[19]

An important, but poorly understood, aspect of contaminant accumulation from sediment is the relative role of water versus sediment particles as the source of exposure. The uptake of contaminants by sediment-associated organisms has generally been reported as primarily via the interstitial water; however, more recent studies suggest that ingestion of contaminated particles may play as large a role—or even a larger role—in bioaccumulation of sediment-associated contaminants.[7,33] Establishing which of the routes of accumulation are most important for these contaminants—both for specific benthos and for organisms strongly coupled to the benthos through food chain transport—will dictate the approach required for establishing sediment quality criteria. There are currently several approaches for the establishment of sediment quality criteria, including, but not limited to, sediment quality triad, apparent effects threshold, screening level concentration, equilibrium partitioning approach, and spiked sediment bioassays. In terms of effects on the benthos, several of the approaches give very similar estimates of the level of sediment-associated contaminants producing effects.[34] In spite of the similarity of the results from the above comparison, the science is not sufficiently established that one method for establishing criteria can be selected over another. Given the weaknesses in our current understanding, the use and comparison of several of the available methods may be the best approach for making decisions about the extent and significance of sediment contamination.

Even with several methods available for estimating an effects level for specific contaminants, there are limitations to the types of situations that can be addressed. There is no method that can establish the appropriate level of contaminants where the exposure occurs in mixtures, the usual situation in sediment exposures. Further, most of the bioassays currently in use for contaminated sediments address acute and, in some cases, subacute end points. There is, in general, an absence of bioassays available for measuring chronic effects. Finally, none of the criteria approaches consider the effects of food chain transfer and its impact on higher trophic level species.

An alternate approach to evaluating the significance of sediment exposures

is the measurement of biological markers in macroinvertebrates and fish collected from contaminated areas. In this approach, it is the biochemical or physiological expression of contaminant exposure and effect (e.g., genotoxic damage, induction of detoxication systems, or histological lesions) that is assayed, rather than the level of individual contaminants in either the sediment or the organism.[35] Biomarkers have the advantage that the complexities of physicochemical partitioning, bioaccumulation, and pharmacodynamics are bypassed, and the end points of most direct concern to animal health are probed explicitly. Long and Buchman have compared and found reasonable agreement between the results of biomarker-based biomonitoring of San Francisco Bay with several more traditional end points such as toxicity tests and concentrations of chemicals in sediments.[36] While biomarker responses measured in organisms from sites with contaminated sediments offer the promise of providing an integrated assessment of the biological significance of exposure, the approach is still under development and requires more research before it can be used as a reliable indicator of sediment quality.

In the above discussion, the concentrations of contaminants in the sediment were assumed to be constant. However, both physical and biological processes can affect the amount of a chemical in the sediment. One of the major factors (in addition to the biological and partitioning factors considered above) that will modify exposure is alteration of the sediment concentration through removal of contaminated sediments from the bioactive layer. The influx of fresh sediment will bury and, to some extent, dilute the contaminated sediment in the bioactive layer, particularly if remedial changes have reduced the concentration on incoming particles. This burial can, however, be modified through bioturbation by benthos that feed in a conveyor-belt fashion. Such bioturbation processes will result in the transport of contaminant-laden particles to the top of the bioactive zone.[37,38] These bioturbing processes extend the duration of exposure of benthic organisms to incoming particle-associated contaminants. Such actions, in conjunction with changes in the depositional regimes, may actually result in the reappearance of contaminants into the bioactive layers of sediments.

A productive approach to integrating the range of physical, chemical, and biological processes that can affect bioaccumulation from sediments involves mathematical modeling of multiply-interacting processes. Mackay et al. (this volume) illustrate the useful insights that can be obtained with "environmental video games" based on a mass balance model of pollutant fate, transport, and bioaccumulation.[39] Young et al. (also this volume) provide an example of field observations of DDE and PCB accumulation from marine sediments that are encouragingly consistent with the fugacity models.[40]

The principal objective of this portion of the symposium is to highlight and illustrate the complex interactions that dictate the total exposure of the biota to sediment-associated contaminants and to indicate gaps in our understanding that limit capabilities to predict the results of such exposures. Even with the number of papers on the interaction of biota with sediment-associated

contaminants increasing rapidly, this field is very young and of such complexity that a complete understanding will require many years of effort. As a result, complete understanding of this field will not occur rapidly and interim approaches will be required for assessing and controlling contaminated sediments.

BIODEGRADATION OF SEDIMENT-ASSOCIATED CONTAMINANTS

While our focus on contaminants in sediments may be motivated by our concern about their toxic effects on biological systems, it is one of the symmetries of nature that biological processes also act to alleviate these concerns by catalyzing the degradation of contaminants. While the degradative capabilities of microorganisms have long been recognized, they have more recently become the focal point for environmental initiatives aimed at restoring contaminated areas. Attempts to remove or isolate contamination through physical and chemical techniques are expensive and often ineffective. It has become clear that in situ bioremediation offers the best practical hope for removing contaminants in natural systems.

Although many organic contaminants that associate with sediments have proven to be highly resistant to aerobic biodegradation, new approaches and understandings are promising increased potential for the biodegradation of many of these contaminants. The purpose of this section is not to extensively review biodegradation of sediment-associated contaminants, but rather to highlight productive and promising research areas associated with biodegradation and to place the papers in this section in the context of current research. Increasing attention to anaerobic degradation, particularly anaerobic dechlorinations,[41,42] and cometabolism has led to a greater appreciation of the degradability of organic compounds by mechanisms not commonly examined. Also, a wider range of compounds have also proven to be degradable by aerobic mechanisms (e.g., PCBs). Often the expression of these degradative capabilities in nature is limited, but genetic engineering offers the potential for increasing the expression of degradative genes and for providing tools for the examination of biodegradation in sediments. Knight and Colwell (in this volume) provide an overview of both the theory and practice of molecular genetics and recombinant DNA technology.[43]

The search for understanding of the degradation of sediment-associated organic compounds has been pursued in many types of sedimentary environments. Exciting research on degradation of PCBs in aquatic sediments has been complemented by new ideas and approaches being developed in examinations of aquifer sediments and in unsaturated terrestrial sediments. The most significant work on biodegradation through anaerobic dechlorination has focused on the fate of PCBs in aquatic sediments, most of which are anaerobic. On the other hand, many of the interesting new findings related to both traditional aerobic degradation and to cometabolic mechanisms come from

work on unsaturated terrestrial sediments or on oxygenated aquifer sediment and groundwater.

Traditionally, many types of contaminants that have been considered to be resistant to biodegradation—ranging from relatively simple chlorinated ethenes (such as trichloroethylene), to much larger, more complex chlorinated compounds (such as PCBs), and large, complex nonchlorinated organic compounds (such as the PAHs). Different classes of these compounds are likely to be found in different sediment settings, and this influences the type of degradative processes that are important in the fate of these compounds. The higher-molecular-weight, nonpolar classes are relatively insoluble in water. Thus, they can be found in aquatic sediments which are likely to be anaerobic, in unsaturated soils where they are likely to be exposed to aerobic conditions, and in groundwater sediments where the conditions may either be aerobic or anaerobic. The lower-molecular-weight, polar compounds are likely only to be found in the terrestrial systems, such as the surface soils, aquifer sediment, and groundwater environments. These compounds would probably not become associated with aquatic sediments due to their relatively high aqueous solubility. However, they may be found in detectable quantities in aquifer sediments due to limited water flow through these sediments. The examination of these varied environments is leading to new approaches to biodegradation, including the linking of anaerobic-aerobic systems for degradation of chlorinated compounds.[44]

There are both broad similarities in the degradation of compounds in terrestrial and aquatic sediments and significant differences. One similarity is the issue of the bioavailability of the high-molecular-weight, nonpolar compounds. In aquatic sediments, unsaturated terrestrial sediments, and aquifer sediments, these compounds will tend to be strongly sorbed to particles. In this state they are likely to more resistant to biological degradation simply due to the inability of the bacteria to access this material. Due to the similarities among the sediment systems, comparisons of processes and rates of bacterial activity in aquatic and aquifer sediments can be useful in obtaining insights into microbial function in these environments.[45] A significant difference between the aquatic sediments and aquifer sediments is the presence of macroinvertebrates and other fauna in the aquatic sediments. Bioturbation and other effects can increase biodegradation rates of organic contaminants on sediments.[46,47] Also, the presence of exotic natural compounds, such as halophenols, may stimulate populations able to degrade similar organic contaminants, thus serving as a source of bacteria for biodegradation of a much broader array of contaminants.[48]

Anaerobic Dechlorinations

Intense research on the reductive dechlorinations of PCBs was prompted by the observation of dechlorination of PCBs in aquatic sediments.[42,49,50] Analysis of other sediments by Lake et al. (this volume) has expanded the

range of sediments in which dechlorinations of PCBs have been observed.[51] Laboratory demonstrations of dechlorination of PCBs soon followed.[52] Questen et al. have found that *meta* and *para* chlorines were removed during dechlorination and that dechlorination was slower and less effective with increasing chlorination.[44] They also found that sediment samples from different sites contained microorganisms with different specificities for PCB dechlorination.

Chlorinated ethylenes (CEs) are also degraded via anaerobic dechlorination. Bouwer et al. have observed anaerobic degradation of perchloroethylene (PCE), trichloroethylene (TCE), and dichloroethylene (DCE) by dechlorination mechanisms and concluded that methanogenesis was the principal biochemical reaction responsible for transforming TCE and PCE.[53,54] Fathepure et al. have demonstrated that a pure methanogenic culture (*Methanosarcina* sp. DCM) degraded TCE and PCE and proposed that PCE was reduced by the same electron carrier that would normally reduce carbon dioxide to methane.[55,56] Freedman and Gossett have shown that numerous chlorinated ethenes may be dechlorinated by a mixed methanogenic culture.[57] Although anaerobic dechlorination is a slow process, it is the only known mechanism for PCE degradation. Unfortunately, PCE or TCE dechlorination often yields vinyl chloride, which is considered to be one of the most carcinogenic compounds.[58]

Dechlorinations of other compounds, such as chloroaromatic compounds, pesticides, and herbicides, have also been a recent focus of study,[41] and much of that work indicates that dehalogenation does not occur with sulfate as an elector acceptor.[59] Dehalogenation may be inhibited in groundwater aquifer sediments by the presence of sulfate,[60] and similar results have been reported for aquatic sediments.[61] Suflita et al. (this volume) examine both the conditions promoting anaerobic biotransformation of herbicides (finding dehalogenation under methanogenic conditions) and the pathways for the degradation.[62] The relationship between sediment/water properties and the reductive dechlorination of various other compounds is addressed by Hale et al. (also in this volume), who have related reductive dechlorination of dichlorophenols to characteristics such as pH, nitrate and sulfate concentrations, and redox potential.[63]

Cometabolism

The terms *cometabolism, fortuitous metabolism,* and *cooxidation* are often used to describe the same process, and much debate is centered on the use of these terms. In aerobic degradation, the bacteria obtain carbon and energy from oxidation of the contaminants, whereas with cometabolism this benefit to the bacteria does not occur. Dalton and Stirling define cometabolism as having four features:[64]

1. A cometabolite does not support growth.
2. The products accumulate stoichiometrically.

3. Transformation is associated with increased oxygen uptake.
4. Cometabolism entails adventitious utilization of existing enzyme systems.

Most cometabolic metabolism has been associated with the activity of various mono- and dioxygenase enzymes. Some of the primary groups of organisms associated with cometabolic metabolism are methane oxidizers, toluene degraders, and ammonia oxidizers. Mono- and dioxygenase enzymes in other types of bacteria have also been implicated in the cometabolic degradation.

The key to cometabolism is the broad substrate specificity of the oxygenase enzymes. Perhaps the best-studied enzyme involved in cometabolism is methane monooxygenase (MMO). The interest in MMO was largely prompted by the work of Wilson and Wilson on TCE degradation.[65] Since then, all CEs except PCE have been reported to be degraded by methanotrophic mixed cultures.[66-70] This MMO enzyme catalyzes the transformation of methane to methanol in the presence of nicotinamide-adenine-dinucleotide (NADH) and oxygen. MMO has been found to have a broad substrate range, which in addition to methane and the CEs includes benzene, bicyclohexyl, butadiene, chlorophenol, cresol, ethylbenzene, phenol, ethene, propene, pyridine, styrene, toluene, and xylene,[71] as well as carbon monoxide[72,73] and ammonia.[74] The details of the pathways and kinetics, as well as the effect of groundwater chemistry on CE degradation, have been examined in a number of studies using pure cultures.[75-77] Palumbo et al. (in this volume) describe the inhibitory effect of ammonia as well as the effect of other groundwater components on TCE degradation by the methanotrophs.[78]

The utility of methanotrophs for in situ degradation of CEs in aquifer sediments has been examined in simulated and actual aquifer systems. Moore et al. spiked a methane-stimulated aquifer with *trans*-1,2-DCE (1 mg/L) and found that 80% of the *trans*-1,2-DCE was degraded after 200 hours.[79] In field studies, Semprini et al. found that *trans*-DCE was degraded to a greater degree than either *cis*-1,2-DCE or TCE but was not degraded as fast as vinyl chloride.[80] Laboratory studies by Grbic-Galic et al. (in this volume) indicate that formate may increase TCE degradation in aquifer sediments by providing an alterative source of reducing power that is not a competitive inhibitor of TCE degradation.[81]

Mono- and dioxygenases from other types of bacteria have also been implicated in oxidation of organic contaminants. The ammonia monooxygenase (AMO) enzyme from nitrifying bacteria, *Nitrosomonas europaea,* has also been shown to have the ability to degrade numerous contaminants including aliphatic, cyclic, and halogenated hydrocarbons.[82-84] AMO is structurally similar to MMO,[85] and the mechanisms of degradation may be similar too because the AMO is capable of oxidizing methane and most other MMO substrates. Similarly, the toluene dioxygenase from *Pseudomonas putida* is able to oxidize TCE and related compounds.[86-89]

Aerobic Degradation

Perhaps the best examples of aerobic mechanisms for the degradation of recalcitrant compounds is the current work on PCBs and PAHs. Degradation of the PCBs with few chlorines was described in the 1970s and early 1980s.[90-93] Since then the range of congeners that have been demonstrated to be degraded has expanded and now includes congeners up to hexachlorobiphenyls.[94,95] In a hazard assessment of PCBs, Hooper et al. describe the many problems remaining in the understanding of PCB degradation.[96]

In addition to their role in cometabolic degradation of numerous substrates, mono- and dioxygenase enzymes have also been implicated in the aerobic degradation of PAH by Cerniglia (in this volume).[97] A *Mycobacterium* species capable of growth on phenanthrene has been isolated from estuarine sediment.[98] PAH degradation has been examined in marine sediments, and the rates of degradation have been related to preexposure to related compounds.[99,100]

The importance of acclimation of bacterial to contaminants in both aerobic and anaerobic environments is receiving increasing attention.[101-103] PAH oxidation in marine sediments has been shown to be positively related to oxygen content and to acclimation to high concentrations of PAH for 1 to 2 weeks.[102] In anaerobic lake sediments, the existence of a lag period and subsequent acclimation for degradation of halobenzoates has been noted[104] and characterized.[103] The acclimation effect has also been noted for other compounds in water samples.[105] Schermerhorn and Ventullo (in this volume) have devised a method for examination of acclimation of periphyton communities to organic chemicals.[106] They found that exposure to linear alkylbenzene sulfonate decreased lag times and increased rate constants for degradation.

Genetic Engineering and Molecular Techniques

Potential applications of genetic engineering to degradation of sediment-associated contaminants include changing the regulatory control of degradative genes and providing tools for examination of degradation. The regulatory control of degradative enzymes can be changed to enable production of degradative enzymes without the presence of an inducer (which may compete with the contaminant for sites on the enzyme) or to shift the expression of the enzyme to a nongrowth stage of the bacterial cell growth cycle. Genetic engineering can be also used to provide tools for the examination of the degradative process through the use of "reporter" constructs.

One example of using genetic engineering to alter regulatory control of biodegradative enzymes comes from work with the control of oxygenase expression for degradation of TCE. Typically, degradation of contaminants at low concentrations is difficult due a variety of factors, including, in the case of oxygenase enzymes, competitive inhibition by the enzyme's preferred substrate, which needs to be present to promote production of the enzyme. A

change in regulatory control of the enzyme expression can reduce this problem. The toluene-degrading (C1C2BADE) genes from *Pseudomonas putida* were introduced into *E. coli* JM109.[107] The construct degraded TCE at a constant rate, whereas *Pseudomonas putida* degraded TCE at a faster rate initially but at a slower rate at the end of the experiment. Winter et al. cloned the toluene monooxygenase genes into *E. coli* and were able to degrade TCE from 16 ppm to 2 ppb, a level that it is difficult to achieve with nonengineered bacteria.[108]

The use of "reporter" constructs and other molecular techniques may enable researchers to better understand the function of degradative bacteria in sediments. Detection of catabolic genotypes in sediments can give indications of the capacity of the microbial community for degradation of organic contaminants. Tracing of microbial populations capable of chlorinated biphenyl degradation has been demonstrated both in aquatic systems and in terrestrial sediments,[109-111] and the use of DNA probes for detection of metal resistance in soils and aquatic environments has been demonstrated.[112,113] Molecular probes hybridizing with 16S RNA have been developed for the detection of methylotrophic organisms.[114] Sayler et al. (in this volume) describe the use of bioluminescence technology for measuring activity of degradative genes in sediments and also review the information on the occurrence of specific degradative genes in subsurface communities.[115] As pointed out by Knight and Colwell,[43] molecular technology is a rapidly evolving frontier that offers great promise in remediation of environmental contamination.

SUMMARY

This volume will examine key concepts and processes related to biological processes affecting organic contaminants in sediments. We begin with integrative overviews of the physicochemical, biological, and toxicological processes that need to be understood in order to evaluate the fate and effects of hazardous chemicals in sediment-water systems, as well as an overview of new molecular techniques offering great promise for enhancing our capabilities to remediate environmental contamination. Innovative approaches to assessing higher-level ecological responses to contaminants are being developed, and modeling of sediment-water interactions provide valuable insights into the distribution of contaminants on a macroscopic scale; however, it is clear that we have an incomplete understanding of key biological processes controlling the dose of contaminant to which organisms are exposed. The concepts and processes associated with uptake and accumulation of sediment-associated contaminants are divided into two sections focusing on aspects of bioavailability of contaminants, and observations of bioaccumulation of contaminants in natural sediments.

Part IV of this volume is devoted to discussion of biological processes that offer the promise of ameliorating existing problems through biological degra-

dation of sediment-associated contaminants. Chapters are divided into sections focusing on processes of anaerobic dechlorinations and cometabolism of contaminants. The final chapter illustrates an application of molecular techniques in bacteria to better understand the functioning of degradative bacteria in sediments.

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