GOVERNMENT OF THE DISTRICT OF COLUMBIA
ENVIRONMENTAL REGULATION ADMINISTRATION

FINAL REPORT

THE DISTRICT OF COLUMBIA
SEDIMENT CORE ANALYSIS:

DETERMINATION OF THE VOLUME OF CONTAMINATED SEDIMENTS IN THE ANACOSTIA RIVER,
DISTRICT OF COLUMBIA

Grant No. 94g-95-WRMD01

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EXECUTIVE SUMMARY

The Chesapeake Bay Program has recently designated the Anacostia River as a Region of Concern (ROC) due to the high levels of sediment contaminants, sediment toxicity, and elevated concentrations of organic contaminants in resident fish species. Additionally, the U.S. EPA has indicated that the Anacostia River is chemically contaminated and in need of further study and possible remediation. As a result of this ROC designation, the District of Columbia developed a Regional Action Plan (RAP) that outlines a series of actions that will ultimately lead to the remediation of contaminated sediments in the tidal Anacostia River. Once contaminant sources are controlled and loads reduced, clean up of the sediments may be warranted and options for remediation include dredging, natural burial, biological remediation, and capping.

The RAP identifies specific information needed to develop sound, scientifically-based and financially feasible sediment remediation alternatives. One option that is considered is dredging of sediments that are determined to be "contaminated". This option requires an assessment of the level of contamination and the volume of sediment to be removed. Recent studies indicate that surface sediments are moderately to highly contaminated in specific areas of the tidal river. No information exists concerning the concentration of contaminants deeper in the sediments in many areas of the river. Additional information on the volume and mass of contaminated sediments is needed to determine the costs and benefits of removing sediments from areas within the Anacostia River as well as the impact of such dredging on the river..

To help answer some of these needs, the District of Columbia funded this project with the objectives to: 1) determine the volume of recently-deposited sediments in the lower Anacostia River, 2) determine the concentration, mass and volume of sediments with elevated levels of metal and organic contaminants, 3) estimate the burial rate of these contaminants, and 4) develop a preliminary remediation assessment and cost estimate for the contaminated areas of the lower tidal Anacostia River.

To accomplish these objectives a field sampling plan was designed and executed in the summer of 1995. The sampling plan involved extensive acoustic sub-bottom profiling and the coring of sediments in the lower tidal river (i.e., from the Sousa Bridge to Hains Point including the Washington Ship Channel). Eight 300 cm sediment gravity cores from the lower tidal Anacostia

River were obtained. The cores were analyzed for a suite of sediment contaminants including chlorinated and non-chlorinated hydrocarbons (e.g., polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs), chlordanes, and DDTs) and trace metals. The activities of ^{210}Pb and ^{137}Cs were also measured in various cores to determine sedimentation rates and areas of deposition within the tidal river.

Due to the dynamic nature of this system and the limited number of cores obtained, the results of the dating procedures and models were inconclusive. Mixing of the sediments by benthic organisms, methane emanations, and dredging, among other possible factors, may have been responsible for the lack of resolution in the results. It is also possible that high loadings of ^{210}Pb relative to the sediment inputs (i.e., urban focusing) could have resulted in erroneous dating calculations. In addition, not all sections were analyzed, and importantly the interval width of many sections were large (i.e., 10 to 50 cm) thereby limiting dating and contaminant resolution. Further sediment analysis, possibly with deeper cores, may allow the complexity of the system to be resolved.

Sediment concentrations of many trace metals were both horizontally and vertically variable throughout the area. Total sedimentary concentrations of cadmium (Cd), nickel (Ni), lead (Pb), and mercury (Hg) ranged from 0.16 to 6.3 µg Cd/g dw, 41.1 to 82.3 µg Ni/g dw, 23 to 1,026.0 µg Pb/g dw, and 0.07 to 10.8 µg Hg/g dw respectively. Both iron (Fe) and aluminum (Al) sediment concentrations were fairly constant with averages (± standard deviation) of 4.91 ± 0.50% and 8.26 ± 0.92% respectively. Generally, highest concentrations were found in the core just below the Washington Navy Yard (ANC-4), with lower concentrations in the sediments near Hains Point (ANC-1). However, in the deeper sections of the core obtained near the head of the Washington Ship Channel (WC-1), concentrations of Hg and Pb were extremely high and substantially enriched above background levels. Lowest sediment concentrations of all metals was observed in the bottom section ANC-3, located downstream of the Frederick Douglas Bridge (i.e., South Capitol Street Bridge).

As with the trace metals, many organic contaminants varied substantially throughout this area. Total polycyclic aromatic hydrocarbons (PAHs) ranged from 0.11 to 49.3 µg/g dw with an overall average of 13.3 µg/g dw. The concentrations of total polychlorinated biphenyls (PCBs) exhibited

substantial differences between cores and in some cases with depth in each core. Overall, concentrations ranged from 3.8 to 3,810 ng/g dw with the lowest concentration found in the bottom section (280-300 cm) of ANC-3 and higher concentrations throughout the core near the Washington Navy Yard. Concentrations of total chlordanes and total DDTs ranged widely, as well; from 0.12 to 181 ng/g dw and 0.65 to 687 ng/g dw with lowest concentrations in the bottom sections of ANC-3 and ANC-1 (Hains Point) and highest concentrations in ANC-4.

This examination of the core data indicates substantial enrichment of many trace metals and organic contaminants over background concentrations. Also, the number of samples with contaminant concentrations exceeding guidelines associated with adverse biological effects (i.e., threshold effects levels and probable effects levels) indicate that the sediment bound contaminants within the Anacostia River and Washington Ship Channel pose a possible risk to aquatic organisms and human health via food chain bioaccumulation. In only the bottom section of core ANC-3 (280-300 cm), located near Greenleaf Point (below the Frederick Douglas Bridge) did the concentrations of many trace metals and all organic contaminants fall below background concentrations. Therefore, any remediation action must deal with the multiple contaminants throughout the river at depths up to and possibly exceeding 3 m.

The sub-bottom profiling from one transect indicates another possible sediment layer in the lower river at approximately 4 to 5 m below the sediment surface. It is possible that core samples at 3 m were taken just above a cleaner, less contaminated sediment. This idea is supported by the low concentration of trace metals and organic contaminants in the bottom section of core ANC-3. This core was taken off the channel closer to the northwest side of the Anacostia River where shoaling of the sediment layer may occur. Therefore, contaminated sediments may be only slightly deeper than 3 m in the river. A more precise determination of the depth of the contaminated sediment layer can be obtained only if deeper cores are taken within the system.

To illustrate the capacity of the lower river to accumulate and retain contaminants, a sediment inventory of both trace metals and organic contaminants was determined. The assumptions for this type of calculations include the limited number of cores (8 for the entire area,), and limited core depth (300 cm). Also, concentrations were interpolated between sediment layers that were not analyzed. However, this calculation does provide a first-order estimate of the mass of material

that has accumulated in the lower river over the past century or so as indicated by the sediment accumulation rates. The total and anthropogenic inventory of the various contaminants is substantial for this area. For organic contaminants such as total chlordane, DDTs, and PCBs, which do not occur naturally, the entire mass is anthropogenic and ranges from 0.23 metric tons (MT) for chlordane to 4.7 MT for total PCBs (1 MT = 1000 kg). For total PAHs, the amount of material is substantial (79 MT) and mostly anthropogenic (i.e., there are some naturally occurring aromatic hydrocarbons). There is also a substantial amount of trace metals within the sediments, from 6 MT for Hg to 2,280 MT for Zn. It is unclear why the Anacostia River contains large amounts of Zn given that it is not presently an industrialized area. Historical inputs from various federal facilities, land use changes and related stormwater inputs, as well as possible upstream sources all may have contributed to the level and amounts of sediment contaminants in the lower Anacostia River.

Sediment contamination in the lower Anacostia River is extensive and must be addressed if current impacts are to be eliminated. However, the benefits of any remediation activity (e.g., dredging, capping, bioremediation) will be temporary unless current sources of chemical contaminants are first controlled and loads reduced. Controlling certain nonpoint source contaminants, such as PAHs and trace metals from automobiles, may require extensive, costly and time-consuming efforts, such as storm water treatment, throughout the watershed. However, it may be possible to control certain contaminants, such as PCBs, with less difficulty and in a shorter time-frame, since they appear to come from some relatively localized sources. Any plan to reduce the load of multiple contaminants from this area must be a balance of chemical specific versus source specific control strategies.

Five different scenarios were evaluated for the remediation of contaminated sediments in the lower tidal river. These range from complete removal of all contaminated sediments to more limited dredging in particularly contaminated areas. This preliminary cost analysis of sediment remediation options suggests that comprehensive remediation of contaminated sediment in the tidal Anacostia River may not be feasible due to the magnitude and extent of the chemical contamination. The cost for the comprehensive remediation option is between $100 and $800 million dollars. However, once the sources of chemical contaminants are sufficiently controlled, a

more detailed study of limited remediation alternatives could be conducted. The limited remediation options entail a combination of dredging and capping in the two most highly contaminated areas of the lower river. The cost for the limited remediation option is between $20 and $100 million dollars.

It is difficult to predict to what extent any remediation action will contribute toward meeting the goal of eliminating chemical contaminant impacts on human and ecological health. Mathematical models that estimate the ecological benefits of reducing the concentration (or availability) of sediment contaminants can be employed to help choose between various remediation alternatives. Such models could be developed and used as part of a more detailed study on the feasibility of the remediation of contaminated sediments.

This study has provided a detailed first look at the vertical extent of sediment contamination in the lower Anacostia River. The results indicate that significant contamination of sediments exits to depths of at least 3 meters, and extends across most of the lower tidal Anacostia River. The large volume of contaminated sediments makes dredging of all sediments an unrealistic remediation option due to its high cost. Options involving limited dredging and or capping, coupled with source reduction are more feasible. This increased understanding of conditions in Anacostia sediments will be a valuable guide for planning future assessments and other management actions.

ACKNOWLEDGMENTS

INTRODUCTION

Sediment contamination problems have been documented in an increasing number of areas in the country (Lyman et al., 1987; NAS, 1989; NOAA, 1991) and Chesapeake Bay (Eskin et al., 1996). Sediments are a major reservoir for anthropogenic contaminants due to the particle-reactive behavior and low water solubility of many pollutants (Young et al., 1985; Olsen et al., 1982). The U.S. EPA recently evaluated more than 21,000 sampling stations nationwide as part of the National Sediment Inventory (U.S. EPA, 1994f; see also NOAA, 1994). Samples or concentrations were compared to several sets of criteria, including EPA's proposed sediment quality criteria (SQCs), NOAA's effects range median and low values (ER-M/ER-L), Environment Canada's probable and threshold effects levels (PEL/TEL), and the apparent effect threshold values (AETs). Other comparisons to theoretical bioaccumulation potential to human health endpoints, metal to sediment sulfide ratios, and fish tissue contaminant levels were also included. Twenty-six percent of these sites evaluated by the EPA (U.S. EPA, 1994f) were categorized as having a high probably of adverse effects (i.e., Tier 1), while 49% were listed as having intermediate probability of adverse effects (i.e., Tier 2). However, it must be noted that most data are from non-random sampling programs (i.e., programs that focused on finding high levels) and these percentages are therefore biased high. The chemicals most often responsible in the Tier 1 category were PCBs, mercury (Hg), total DDT, and to a lesser extent copper (Cu), nickel (Ni), and lead (Pb).

Contaminated sediments can have a direct effect on aquatic life, on recreation and an indirect effect on human health via the accumulation of many contaminants through the food chain (i.e., fish). Current food chain models have shown linkages between the level of specific contaminants in the sediments and concentrations in fish within the overlying waters. However, in using the biota-sediment accumulation factor (BSAF), Berman (1990) showed no clear relationships were evident between organic carbon-normalized sediment concentrations and lipid-normalized residues in biota (benthic organisms and fish). The use of the BSAFs assumes simple partitioning and thermodynamic equilibrium between contaminants in the sediments and fish, and many aquatic systems are not in equilibrium in both space and time. Also, BSAF calculations ignore various transfers and contaminant pathways that can take between the sediment and fish. More dynamic

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steady-state or time varying models have been useful in describing the movement of contaminants in the food chain (see Thomann et al., 1992). The net result is that sediments are an important source of contaminants to both benthic and potentially pelagic organisms. For example, since 1989, the District of Columbia's Commissioner of Public Health has issued an advisory concerning human consumption of channel catfish and other species in the Potomac and Anacostia Rivers as a result of high polychlorinated biphenyls (PCBs) and chlordane concentrations.

Sediment contamination can also have an effect on the use of navigational waters requiring periodic dredging. Once a sediment is contaminated, restrictions on the handling and disposal may raise the cost of dredging to levels that are prohibitive. The development of remediation techniques for contaminated sediments is relatively recent. The range of action includes no action or natural burial, sediment capping, in-situ treatment, dredging with off-site treatment and replacement, and dredging with disposal. Dredging and disposal is the traditional method of removing/treating contaminated sediments but, because of the potential for further environmental damage and the cost of disposal, other alternatives are being considered. The no-action method allows for natural sedimentation, potentially augmented with added clean sediment (i.e., capping), to mix and bury with the in-place contaminated sediments. Although the "no action" method is attractive in that actual dredging is not used and sediment disturbance is reduced, its overall cost may be misleading. For this method to work effectively, action(s) need to be taken to reduce the source of metals or organic compounds to the area. In-place treatments have focused on ways to solidify the sediments or immobilize the contaminants within the sediment matrix. Costs can be high and only a few trial methods are currently being tested (U.S. EPA, 1993). All of these methods entail substantial cost to reduce sources and remove or clean contaminated sediments. It is imperative therefore to have an accurate assessment of the extent, volume, and mass of sediment contaminants in a given area so that appropriate management actions can be taken.

Objectives

The specific objectives of this project are: 1) to determine the volume of recently-deposited sediments in the lower Anacostia River, 2) to determine the concentration, mass and volume of sediments with elevated levels of metal and organic contaminants, and 3) to estimate the burial

rate of these contaminants. Lastly, a preliminary remediation assessment and cost estimate was made for the contaminated areas of the lower tidal Anacostia River.

Background

There are few studies concerning the sources and distribution of potentially toxic inorganic and organic chemicals in the Washington, D.C. area (Figure 1). Friebele et al. (1988) presented an extensive literature survey on the distribution and temporal trends of toxic chemicals in the rivers and streams of the Potomac River Basin, including the waters of the District of Columbia. While this report contained historical information concerning toxic organic and inorganic material in water, little information was reported concerning sediment contamination. In a recent study by Velinsky et al. (1992), surface sediment concentrations of trace metals (e.g., Pb, Cd, and Zn) were found to be elevated in the lower Anacostia River compared to background levels as determined by normalization to a crustal element (e.g., Fe). High concentrations of organic contaminants were also observed in this area, especially near the South Capitol Street Bridge. Pinkney et al. (1993) also reported substantial concentrations of sediment bound trace metals and organic contaminants in the tidal Anacostia River, with highest concentrations below the Pennsylvania Avenue Bridge. The major present-day source of these contaminants to the tidal river appears to be from urban runoff, via combined and storm sewer runoff. Mass balance modeling of the tidal area of the river from Bladensburg Marina to Hains Point supports this conclusion and suggests that upstream sources, direct atmospheric deposition, and point sources are small compared to urban storm water runoff (Velinsky et al., 1993; Velinsky, unpublished data) and possibly combined sewer overflows.

The studies mentioned above focused on the distribution of contaminants in the surface sediments of the river (top 5 cm or so). One investigated the distribution of metals deeper in the sediments (Brush et al., 1982; Martin et al., 1981). This study investigated sediment accumulation rates and trace metal distributions in the entire Potomac estuary. In this study only one core was taken within the District of Columbia and was located in the Washington Ship Channel (Site 17; Figure 1). Analysis using ^{210}Pb indicated constant deposition with a sediment

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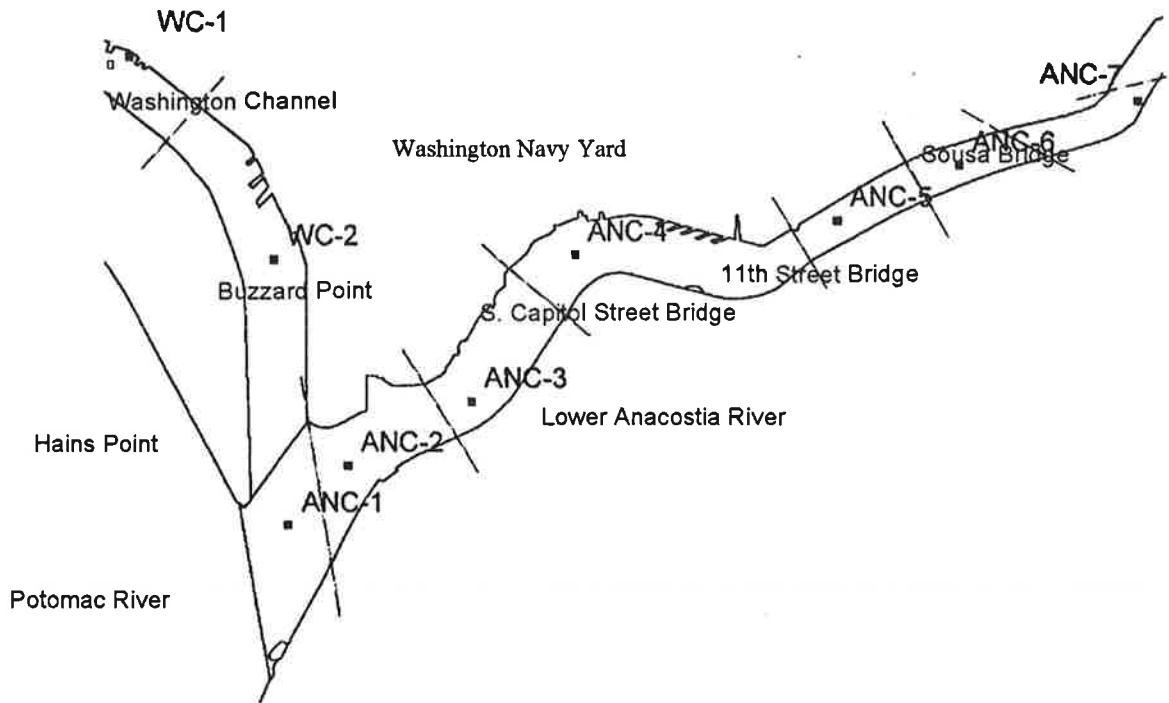
Figure 1 Map of the lower Anacostia River and Washington Ship Channel showing the approximate locations of the sediment core sampling and the division of the river and channel into sections for the remediation analysis based on the locations of the cores. WC-2 and Core 17 from Brush et al (1980) are at the same location.

accumulation rate of 0.9 g/cm^2-yr (1.6 cm/yr) from 1878 to1978, while pollen analysis yielded rates that were approximately half of what was determined using ^{210}Pb dating. Each type of analysis has its limitations and these are fully discussed in Brush et al. (1982). Trace metals such as Cd, Cu, and Pb exhibited different distributions with depth in the sediment (Figure 2). Cd concentrations started to increase at approximately 90 cm and peaked at 60 cm with a concentration of approximately 4 μg/g (Note: all concentrations on a dry weight basis). Concentrations then deceased slightly and remained constant to the surface (2 μg/g). Concentrations of Cu in the core showed increasing concentrations from 120 cm to approximately 60 cm and remained near constant (80 to 100 μg/g) to the surface (i.e., recent sediments). Lead concentrations were variable at depth and constant from approximately 60 cm to the surface with concentrations of approximately 200 μg/g. These distributions suggest that the source(s) of these metals to the Washington Ship Channel were not similar and the constant levels in the surface sections indicate that inputs have not started to decrease by 1982. Also, it appears that background concentrations of Cu and Pb were not reached within the 120 cm length core, whereas concentrations of Cd below approximately 100 cm did reached background levels.

In the Anacostia River, Scatena (1987) obtained 16 sediment cores in the early 1980s for visual description and pollen analysis. He found many cores with discontinuous sand and leaf lenses in both the vertical and horizontal directions. Gas bubbles, a result of organic matter decomposition (e.g., methanogenesis), were prevalent in many cores with densities of approximately 1.6 bubbles/cm^2. In these cores peaks in ragweed pollen were related to specific hydrologic and agricultural changes within the basin. Sediment mass balance calculations indicated that approximately 85% of the sediment that enters the lower river is trapped and buried, with the mode of sediment input varying between suspension settling and bedload transport. Mixing was thought to be via physical processes (non-tidal) but some cores showed some bioturbation in the surface sections. Recent sedimentation rates varied between 0.9 to 1.6 cm/yr. Unfortunately, no chemical analyses were done on these cores.

In some river systems, point sources of contaminants have been eliminated or lowered while the relative importance of nonpoint sources (e.g., urban runoff) has increased. Even if strict
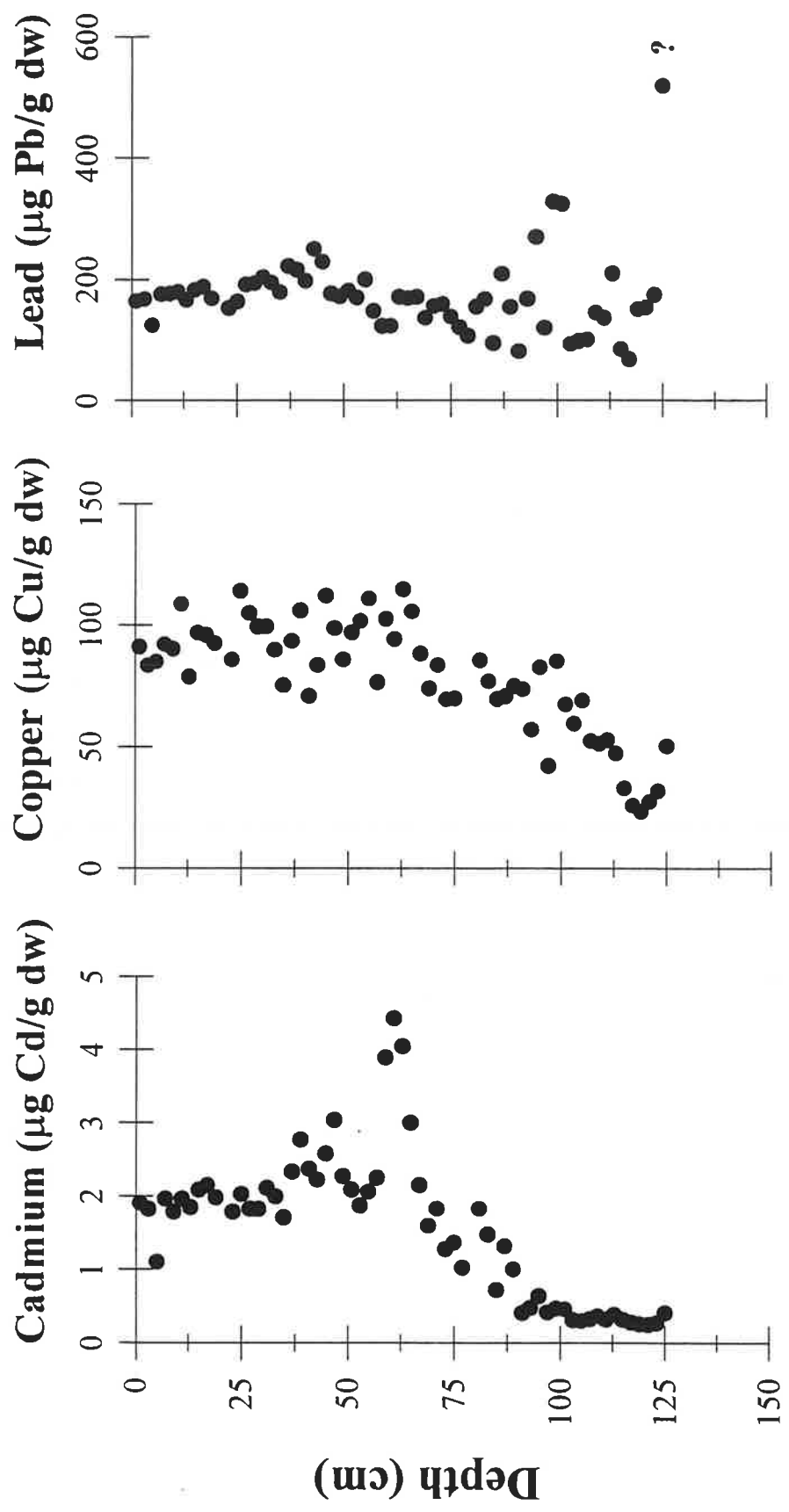
Figure 2. Distribution of Selected Trace Metals at Site 17 in the Washington Ship Channel (from Martin et al., 1981).

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source reductions are implemented for point sources and urban runoff, the sediments within a system can still be a source of contaminants to the overlying water column and effect the benthic community and resident fish population (Velinsky and Cummins, 1994; Schlekat et al., 1994). Flegal and Sanudo-Wihelmy (1993), for example, showed that the elevated concentrations of dissolved trace metals in the waters of San Diego Bay are a result of a sedimentary flux of dissolved trace metals to the overlying waters, since all wastewater discharges were terminated in the mid-1960s. Their results suggest that sediments with high concentrations of metals can act as a long term source of metals (and possible other contaminants) to a system even if point and nonpoint sources are reduced. Other benthic processes, such as tidal and storm resuspension, can also help maintain the availability of sediment contaminants to fish and other biological communities of the river.

The U.S. EPA's Chesapeake Bay Program has recently designated the Anacostia River as a Region of Concern (ROC) due to the high levels of sediment contaminants, sediment toxicity, and elevated concentrations of organic contaminants in resident fish species (Velinsky and Cummins, 1994; Velinsky et al., 1994a,b; Wade et al., 1994; Schlekat et al., 1994, U.S. EPA, 1993). Additionally, the U.S. EPA indicated that the Anacostia River is potentially contaminated and in need of further study (U.S. EPA, 1996). As a result of this ROC designation, the District of Columbia developed a Regional Action Plan (RAP) prioritizing methods and alternatives to help remediate the sediments and clean up this area (DCRA, 1996). Some examples of these alternatives for contaminated sediments, dependent on the specific area, include source reduction, dredging, natural burial, and capping. To develop sound, scientifically-based and financially responsible remediation alternatives the plan identifies some specific information needs concerning the sediments. One option for sediment remediation that is considered is the dredging of sediments that are determined to be "contaminated". This option is requires an assessment of the level of contamination and the volume of sediment to be removed. Recent studies (Velinsky et al., 1992; Pinkney et al., 1993) indicate that surface sediments are moderate to highly contaminated in specific areas of the tidal river. No information exists concerning the concentration of contaminants with depth in the sediments in many areas of the river. The volume

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and mass of dredged sediments, the impact of dredging on the river, and sediment disposal methods are some of the issues that need to be considered to determine the cost-benefit ratios of removing sediments from areas within the Anacostia River.

STUDY AREA AND FIELD SAMPLING

Study Area

The District of Columbia (DC) lies along the fall line at the boundary between the Atlantic Coastal Plain and the Piedmont Plateau, and is at the head of navigation of the Potomac estuary (Figure 1). The western and northern sections of the DC area are part of the Piedmont, which is underlain by deformed meta-sedimentary and meta-igneous rocks. From the mid-section of the city to the south is the Coastal Plain which contains unmetamorphosed fluvial and marine sediments (Reed and Obermeier, 1989).

Presently, there are three major rivers or streams in the DC area: the Potomac and Anacostia rivers, and Rock Creek, which drains into the Potomac River just south of Georgetown (Figure 1). Average yearly flows for the Potomac River at Chain Bridge, Anacostia River, and Rock Creek are 1.03×10^{10}, 1.16×10^8, and 5.5×10^7 m^3 yr^{-1}, respectively. The drainage areas of the Anacostia River (310 km^2) and Rock Creek (160 km^2) are small compared to the Potomac River (e.g., at Chain Bridge, 29600 km^2), and both of the smaller water bodies drain predominantly urban environments.

During the past 200 years, the DC river scape has been altered by sedimentation, dredging, and filling (Williams, 1989). The Tidal Basin (surface area of 0.4 km^2), with an average depth of approximately 2 m, receives inputs from the Potomac River as well as storm water runoff and atmospheric deposition. The Washington Ship Channel is connected to the Tidal Basin in the north section via a floodgate and to the Anacostia River at the southern end (Figure 1). The center of this channel has been dredged in the past with bottom depths ranging from <1 m to approximately 8 m. The Washington Ship Channel is bordered by a park on the western side and residential/commercial development on the eastern side.

The flow of the Anacostia River is controlled by streamflow of the Northeast and Northwest

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branches, which join at Bladensburg (MD). The tidal waters in the lower Anacostia River, south of Bladensburg, have a long residence time (e.g., 35 days) due to the large volume relative to the runoff of the watershed. Here the river resembles a lake (Scatena, 1987) and allows suspended sediments to settle out of the water within the tidal portion of the river. Sedimentation rates are reported to be 3.2 g cm^{-2} yr^{-1} or 1.9 cm yr^{-1} on a dry-sediment basis (Scatena, 1987). While the center channel of the Anacostia River has been dredged in the past, depths outside the channel generally range from 0.5 to 5 m.

There is evidence for historical contamination of the sediments in the study area due to past shipping and boating uses and inputs via combined and storm sewer runoff (Velinsky et al., 1992). Approximately 30 storm and six combined sewers discharge into the lower Anacostia River (i.e., south of the Kingman Lake area to Greenleaf Point at the mouth of the Washington Ship Channel). These sewers drain an area of approximately 14 km^2, or 22% of the drainage area of the Anacostia River within the District of Columbia. Approximately 54% of the total drainage area of the Anacostia Basin is urban (ICPRB, 1988).

Field Sampling

Field sampling was designed to meet the objectives of determining the concentration/ distribution and volume of contaminants in sediment cores taken from the tidal Anacostia River and was accomplished in two phases during the summer 1995.

Phase 1: Sediment Profiling. A high resolution seismic-reflection profiling was employed to determine the depth distribution of the various sediment layers. Seismic-reflection or sub-bottom profiling at high frequencies (1 to 5 kHz) is a tool frequently used for determining sediment inventories and sedimentary structure. What is sought are any significant boundaries in the sediment column (i.e., sand layers, Holocene gravel layer). The survey also employed a high resolution fathometer operated at 205 kHz to delineate the water-sediment interface. The survey provided information about the sedimentary depositional structure in the Anacostia River and helped to focus site selection for sediment coring.

Profiling was accomplished with a Ferranti Ocean Research Equipment (ORE) Geo-Pulse

Profiler using a frequency range of 1 to 5 kHz on May 24, 1995. The District of Columbia's Water Resources Boat, an 18 ft. Boston Whaler, was outfitted with the profiler and recorder as well as a Magellan Global Positioning System (GPS) receiver interfaced with a PC based navigation program. The profiling area started at Hains Point and profiling transects were obtained up the Anacostia River up to the Conrail Railroad Bridge (Figure 1). Included were profiles in the Washington Ship Channel from the area above the Interstate 395 Bridge to the confluence with the Anacostia River, near Hains Point. Appendix I provides the description of the profile lines. Actual positions while profiling were determined using satellite navigation (GPS) as well line of site recordings and USGS topographic sheets (Appendix I). Data was recorded on tape and chart recorder that combined both time, depth of water, possible sediment layers, and position.

Phase II: Sediment Coring. Once the profiling was complete and the information evaluated, sites were selected for coring and sediment retrieval. These sites were located in the most representative areas based on the sub-bottom profiling, location of potential contaminant sources, and the overall geography of the river. The coring platform for this study was the RV Orion from the University of Maryland Center for Estuarine and Environmental Sciences (CEES). To obtain the sediment cores, a three meter aluminum Kasten corer was used (Kuehl et al., 1985). This is a gravity corer that can take sediment samples up to three meters long in semi-consolidated organic rich sediments.

Eight cores were taken for this study (Table 1) on July 18, 1995. Two cores were taken within the Washington Ship Channel and six cores were obtained within the Anacostia River from near Hains Point to upstream of the Anacostia Bridges at 11 Street, S.W.. A core was to be obtained between the Conrail Railroad bridge and the Pennsylvania Ave. bridge but construction and barge traffic prohibited the vessel from going above the Pennsylvania Ave. bridge. Prior to sampling the core barrel was disassembled and scrubbed with dilute Alconox and rinsed with ambient water. The corer was then inspected for any oil and grease, which was not found. At each station, the corer was carefully lowered and allowed to penetrate the sediment. Only cores that appeared intact and achieved full penetration were kept. The total length of sediment retrieved (to estimate

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compaction which was generally less than 10%) and other visual characteristics were obtained

Table 1. Station locations for sites in Washington Ship Channel and Anacostia River.

Station Location/ID.	Latitude (N)	Longitude (W)	Comments
Washington Ship Channel			
WC-1	38 52' 55"	77 01' 41"	Washington Ship Channel: Hot Spot; By Bridges
WC-2	38 52' 13"	77 01' 15"	Washington Ship Channel: Dating/Old Core 17
Anacostia River			
ANC-1	38 51' 20"	77 01' 12"	Hains Point
ANC-2	38 51' 32"	77 01' 00"	Greenleaf Point
ANC-3	38 51' 45"	77 00' 35"	Buzzards Point
ANC-4	38 52' 14"	77 00' 14"	Near Ost area: Hot Spot: Outfalls
ANC-5	38 52' 21"	76 59' 20"	Near COE docks: Hot Spot
ANC-6	38 52' 32"	76 58' 55"	Downstream of PA Ave. Bridge
ANC-7	38 52' 45"	76 58' 19"	Above PA Bridge (Not taken due to barges)

prior to sectioning and sampling. Between stations the corer was scrubbed with a brush and ambient water and inspected for oil and grease and sediment from the prior station.

Each core was sectioned on the deck of the ship using pre-cleaned plastic or stainless steel spatulas or scoops. Cores were sectioned at different depth intervals dependent on the location (Table 2) and varied from 2.5 cm to up to 50 cm. Each section was placed into a precleaned glass bowl and homogenized, visually, to similar color and texture. The homogenized-sediment was then placed into various containers for trace metal, organic contaminants, ^{210}Pb dating, grain size, and other parameters. Containers were cleaned differently dependent on the parameter to be measured: plastic bags for metals, and grain size; pre-combusted jars for organic contaminants; glass vials for ^{210}Pb and sediment density.

Table 2a. Core depth intervals and sediment descriptions*.

Core/Depth	Description
WC-1	
0 - 20	Slurry, brown/olive color throughout core.
20 - 70	Gas bubbles near bottom of section, shell material around 50 cm; mild petro. smell.
70 - 120	Gas pockets, midway down section were horizontal black streaks; petro. smell.
120-170	Gas pockets, not as much black color; petro. smell, some leaf material.
170 - 216	Gas pockets, black color, some leaf material.
Bottom 20	Woody material near bottom; saved a piece of it; some sandier material, no petrol smell
Additional:	234 cm recovery; 267 cm penetration
WC -2	
0 - 5	Mixed brown/black color, no shell material.
5 - 10	" "
10 - 15	" "
15 - 20	" "
20 - 25	" "
25 - 30	" "
30 - 40	Black color sediments
40 - 50	" "
50 - 60	" "
60 - 70	" "
70 - 80	" "
80 - 90	" "
90 - 100	" "
100 - 125	Black color; more gas pockets
125 - 150	Black color sediments
150 - 175	" "
175 - 200	" "
200 - 250	" "
250 - 275	" "
Bottom 10	Clay layer
Additional:	280 cm recovery; 300 cm penetration: 23 cm of compaction

*All depths in cm.

Table 2b. Core depth intervals and sediment descriptions.*

Core/Depth	Description
ANC -1	
0 - 5	Shell fragments, olive color
5 - 10	" "
10 - 15	" "
15 - 20	" "
20 - 25	" "
25 - 30	" "
30 - 40	Gas bubbles throughout
40 - 50	" "
50 - 60	" "
60 - 70	0.5 Corbicula (sp.) shell (3 cm); Gas bubbles
70 - 80	Gas bubbles throughout
80 - 90	" "
90 - 100	Shell fragments and gas bubbles
100 - 125	" "
125 - 150	" "
150 - 175	Thin brown layer (red brown)
175 - 200	Petrol smell
200 - 250	Gas bubbles/blacker color
250 - 275	More compacted
Bottom 20	Shell fragments; clay like material
ANC - 2	
0 - 20	Less consolidated; large bivalve found and saved; brown/black color
20 - 70	Many gas pockets throughout section, brown/black color
70 - 120	Brown/black color
120-170	" "
170 - 220	" "
220 - 257	" "
Bottom 20	Very compacted

*All depths in cm.

Table 2c. Core depth intervals and sediment descriptions*.

Core/Depth	Description
ANC - 3	
0 - 20	Many gas pockets; brown and black streaks throughout
20 - 70	" " "
70 - 120	" " "
120-170	Many gas pockets; brown and black streaks throughout; woody debris
170 - 220	Many gas pockets; brown and black streaks throughout; distinct petrol smell; some shell fragments
Bottom 20	Many gas pockets; brown and black streaks throughout; very consolidated
Additional:	220 cm penetration; top 10 cm unconsolidated
ANC - 4	
0 - 20	Black flecks (1-2 mm); brown color, gas pockets
20 - 70	" "
70 - 120	" "
120 - 170	Light petrol smell; brown and black streaks
170 - 220	Heavier petrol smell; brown and black streaks; plant material
220 - 270	Light petrol smell; some plant material
270 - 295	Light petrol smell
Bottom 5	Small piece of Al foil
Additional:	300 cm penetration; top 30 cm more unconsolidated, appears sandier with black specs.
ANC - 5	
0 - 20	Hydrocarbon sheen to mud; more fluid mud; some leaf material
20 - 70	Leaf fragments
70 - 120	" "
120 - 170	Black/brown layers with petrol smell
170 - 220	" "
220 - 252	" "
Bottom 10	" "
Additional:	Approx. 30 cm compaction

*All depths in cm.

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Table 2d. Core depth intervals and sediment descriptions.*

Core/Depth	Description
ANC - 6	
0 - 5	Olive/brown color; leaves, plant material; some more sand and shell fragments.
5 - 10	" " "
10 - 15	" " "
15 - 20	" " "
20 - 25	" " "
25 - 30	" " "
30 - 40	Mostly brown color.
40 - 50	" "
50 - 60	" "
60 - 70	" "
70 - 80	" "
80 - 90	" "
90 - 100	" "
100 - 125	" "
125 - 150	Black layering-streaks, petrol smell; gas pockets
150 - 175	" "
175 - 200	Petrol smell, shell fragments, brown olive sediment color
200 - 250	" "
250 - 300	" "
Bottom 20	Clay like, petrol smell
Additional:	Full 300 cm penetration and recovery

*All depths in cm.

CHEMICAL AND PHYSICAL ANALYSES.

Inorganics (Total Metals). The analytical techniques used were atomic absorption spectrophotometry (AAS), in flame mode for those elements in high enough concentration and graphite furnace (GF-AAS) or cold vapor when necessary. Samples were digested in 50 ml closed all-teflon "bombs" (Savillex Co.) (Brooks et al., 1988). Accurately weighed sediment aliquots (ca. 200 mg) were digested at 130°C in a mixture of nitric, perchloric and hydrofluoric acids. A saturated boric acid solution was then added to complete dissolution of the sediment and the digest was brought to a known volume. Various dilutions were made on the clear digest solutions to bring them into the

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working range for AAS. Standard reference materials and blanks were digested and analyzed with every batch of samples (See Appendix II).

Because of its relative freedom from matrix interference and its high sample throughput, flame AAS was used whenever possible. However, only Fe, Mn and Zn were consistently in high enough concentrations to be determined by flame AAS. Most of the other elements (Cd, Cu, Ni, Pb) were determined by GF-AAS. The flame AAS work was conducted on a Perkin-Elmer Model 306 instrument, following the manufacturer's recommendations. Working curves were constructed from commercial standards and resulting concentrations were verified by analyzing NSIT and other standard materials (NRCC) with every sample set.

A Perkin-Elmer Zeeman 3030, equipped with an HGA-600 furnace and AS-60 autosampler, was used for GF-AAS. Matrix modifiers and analytical conditions for the furnace and spectrophotometer were based on the manufacturer's recommendations, with modifications as appropriate to maximize sensitivity and minimize interferences. Standard reference materials and spiked samples were used to evaluate instrument performance and furnace conditions were changed when necessary. Based on 10 separate analyses of the reference materials over the course of the project, the accuracy and precision of the analyses are approximately ± 10 % for all metals.

Mercury was determined by cold vapor AAS on an aliquot of the same digest used to determine other trace elements. A "head space" sampling procedure was used in contrast to the more common "stripping" procedure. One ml of sample or standard (more if needed) was put into a 25 ml Erlenmeyer flask and the flask was closed with a rubber serum stopper. The flask was injected with 0.5 ml of a 10 percent $SnCl_2$ solution and shaken for 30 seconds to reduce Hg to the metal and allow it to transfer into the head space. A syringe needle connected to the mercury analyzer by a short piece of tygon tubing was next pushed through the serum stopper. Finally, a syringe needle connected to a water reservoir by tygon tubing was pushed through the serum stopper. Water was allowed to flow into the flask at a rate that filled it in about 10 seconds. The water forced air from the flask, with its Hg, into the Hg analyzer where it was measured. A Laboratory Data Control Co. UV monitor with a 30 cm path length cell was used for Hg determinations.

Glassware, plasticware, and reaction vessels were cleaned first by soaking in Micro cleaning

solution for 24 hrs and then rinsed with distilled water. Glassware and the reaction vessels were then soaked in an acid bath (50%v/v HNO_3) for 24 hrs, rinsed with distilled deionized water (DDW), and air dried in a laminar flow hood in a dust free environment. Other plasticware used in these procedures were either used only a single time or reused after washing with Micro solution, appropriate acids (i.e., either HCl or HNO_3, depending upon resistance to attack) and DDW. As stated previously, method blanks, matrix spikes, and standard reference materials were run with each set of samples. In Appendix II the raw data along with the QA/QC information is presented; for the QA/QC plan see Velinsky et al. (1995).

Organics (PAH, PCB, and Pesticides). Approximately 10 grams of freeze-dried sediment was used for the analysis. The freeze-dried sample was soxhlet-extracted with methylene chloride. The solvent was concentrated to approximately 20 ml in a flat-bottomed flask equipped with a three-ball Snyder column condenser. The extract was then transferred to Kuderna-Danish tubes, which were heated in a water bath (60°C) to concentrate the extract to a final volume of 2 ml. During concentration of the solvent, hexane was exchanged for dichloromethane.

The extracts were fractionated by alumina:silica gel open column chromatography. Silica gel was activated at 170°C for 12 hours and partially deactivated with 3% (v/w) distilled water. Twenty grams of silica gel were slurry packed in dichloromethane over ten grams of alumina. Alumina was activated at 400°C for four hours and partially deactivated with 1% distilled water (v/w). The dichloromethane was replaced with pentane by elution, and the extract was applied to the top of the column. The extract was sequentially eluted from the column with 50 ml of pentane (aliphatic fraction) and 200 ml of 1:1 pentane-dichloromethane (aromatic- pesticide fraction). The fractions were then concentrated to 1 ml using Kuderna-Danish tubes heated in a water bath at 60°C.

Quality assurance for each set of twenty samples included a procedural blank and a sampled spiked with all calibrated analytes (matrix spike) and a referance sediment which were carried through the entire analytical scheme. In addition a laboratory reference material consisting of solvent spiked with an oil was used to check the quality control of each sample set analysed for PAHs. All internal standards (surrogates) were added to the samples prior to extraction and were used for quantification.

Aromatic hydrocarbons were separated and quantified by gas chromatography-mass spectrometry (GC-MS) (HP5890-GC and HP5970-MSD). The samples were injected in the splitless mode onto a 0.25 mm x 30 m (0.32 μm film thickness) DB-5 fused silica capillary column (J&W Scientific Inc.) at an initial temperature of 60°C and temperature programmed at 12°C/min to 300°C and held at the final temperature for 6 minutes. The mass spectral data were acquired using selected ions for each of the PAH analytes. The GC-MS was calibrated and linearity determined by injection of a standard component mixture at five concentrations ranging from 0.01 ng/μl to 1 ng/μl. Sample component concentrations were calculated from the average response factor for each analyte. Analyte identifications were based on correct retention time of the quantitation ion (molecular ion) for the specific analyte and confirmed by the ratio of quantitation to confirmation ion.

A calibration check standard was run three times during the sample runs (beginning, middle and end), with no more than 6 hours between calibration checks. The calibration check was confirmed to maintain an average response factor within 10% for all analytes, with no one analyte greater than 25% of the known concentration. With each set of samples a laboratory reference sample (oil spiked solution) was analyzed to confirm GC-MS system performance. The internal standards (surrogates) for the PAH analysis were d8-naphthalene, d10-acenaphthene, d10-phenanthrene, d12-chrysene, and d12-perylene, and were added at a concentration similar to that expected for the analytes of interest. To monitor the recovery of the PAH surrogates, chromatography internal standards d10-fluorene and d12-benzo(a)pyrene were added just prior to GC-MS analysis.

The pesticides and PCBs were separated by gas chromatography in the splitless mode using an electron capture detector (ECD). The output from the detector was collected by an automated HP-LAS 3357 data acquisition software package. A 30 m x 0.32 mm I.D. fused silica column with DB-5 bonded phase (J&W or equivalent) was used. Four calibration solutions containing the pesticides and the PCBs used for quantitation were used to generate a line fit (non-linear), and the calibration standards were in the range of 5 to 200 ng/ml. A sample containing only PCBs was used to confirm the identification of each PCB congener. The internal standards (surrogates) for pesticide and PCB analysis were added prior to extraction and were DBOFB (dibromooctafluorobiphenyl), PCB-103 and PCB-198. To monitor the recovery of the pesticides and PCB surrogates, the

chromatography internal standard TCMX (Tetrachloro-m-xylene) was added prior to GC-ECD analysis. The chromatographic conditions for the pesticide-PCB analysis were 100°C for 1 min, then 5°C/min till 140°C, hold for 1 min, then 1.5°C/min to 250°C, hold for 1 min, and then 10°C/min to 300°C and finally held for 5 min.

Glassware was cleaned by detergent washing (Micro cleaning solution) and rinsing with tap and distilled water. The glassware was then pre-heated in a muffle furnace at 400°C for at least 4 hrs. Solvent rinses of acetone followed by methylene chloride was substituted for the heating by the muffle furnace heating when determined to be appropriate. After drying and cooling, the glassware was wrapped in pre-heated Al foil and stored in a dust free environment. Blanks, matrix spikes and standard reference materials were extracted and run with each set of samples as appropriate. In Appendix II the raw data along with the QA/QC information is presented; for the QA/QC plan see Velinsky et al. (1995).

^{210}Pb Analyses. Wet sediments were weighed and dried at 65 °C to determine percent water. Percent waters were not corrected for dissolved solids due to the coring environment being in the tidal freshwater section of the estuary. After reweighing, sediments were ground to a powder with a ceramic mortar and pestle.

The measurement of ^{210}Pb ($T_{1/2}$ = 22.3 years) was carried out by the analysis of its daughter radionuclide, ^{210}Po ($T_{1/2}$ = 138 days). The extraction procedure for ^{210}Po generally followed that of Sugai (1990). A mixture of concentrated HNO_3 and HCl was added to 1-2 grams of dried sediment and the slurry heated for 1.5 hours at sub-boiling temperatures. Prior to extraction, NIST-calibrated ^{209}Po was added as a yield tracer. After centrifugation, the acid was decanted into a glass beaker and evaporated by heating at low heat. The Po was plated on a silver disk in a weak HCl solution. Plates were then counted in a Nucleus/Tennelec/Ortec counting system for 1-2 days. Activities were calculated based on the known ^{209}Po addition and counting statistics were determined.

The unsupported ^{210}Pb activity was determined from the asymptote of the ^{210}Pb profiles. Cumulative mass (g cm^{-2}) was determined from percent water data and a sediment density of 2.5 g cm^{-3} that is typical of Chesapeake Bay sediments. The constant initial concentration sedimentation model was used (Robbins 1978) and a simple linear regression of unsupported ^{210}Pb versus

cumulative mass was used to provide a sedimentation rate in g cm^{-2} yr^{-1}. The age at a given sediment depth was estimated by dividing the cumulative mass by the sedimentation rate.

Organic Carbon and Grain Size. Organic carbon (OC) was determined by infra-red absorption after combustion in an O_2 stream, using a LECO WR-12 Total Carbon System. Samples, 100 to 500mg, were acidified using dilute HCl in methanol and then dried overnight at 50°C. Method blanks and duplicate samples were analyzed every 20 samples. Data are reported as μg of C per gram dry weight. All glassware and utensils were pre-heated prior to use.

Sediment grain size was determined by the procedure of Folk (1974), utilizing sieving to separate gravel and sand fractions from the clay and silt fractions. The latter fractions were subsequently separated by the pipette (settling rate) method. Detailed descriptions of the methods utilized in measuring OC, CaCO3 and grain size are reported in Brooks et al. (1988).

RESULTS

Sub-Bottom Profiling of the Anacostia River and Washington Ship Channel

The sub-bottom survey produced limited results in the Anacostia River and Washington Ship Channel (see Appendix I). The ability of the acoustic signal to penetrate the Holocene sediment was limited by gaseous layers throughout the column and the relatively shallow water in this reach of the river. The gases, a mixture of methane, nitrogen, and carbon dioxide, are produced by the microbial decomposition of sediment organic matter by various processes. The Tertiary horizon was not detectable in the seismic records which prevented delineation of the thickness of the Holocene layer.

The sub-bottom records did not indicate any structure below the sediment-water interface except at transect 8 near the Washington Navy Yard (Figures 1 and 3). This transect reveals a layer of sediment approximately four to five meters below the sediment-water interface that becomes shallower towards either bank. Other transects did not show this structure due to either its absence or the presence of a strong gas bubble layer within the upper sections of the sediments. The bubble layers acts to baffle the sound waves and causes multiple reflectance of the sound pulse (see transect 11 for example; Figure 4). Coring data support the seismic records in that no distinct sub-bottom layer or sediment discontinuity in either structure or thickness was observed in the upper 3 m of sediment. Samples of the sub-bottom layer from the area of transect 8 were not obtained due to the depth of the layer (4 to 5 m below the sediment-water interface) and length of the Casten core (3 m).

Bathymetric records collected during the survey were utilized in conjunction with the coring data to calculate minimum volumes for contaminated sediments along the cored profiles. The bathymetric survey revealed several deeper channels in the lower Anacostia River between the Conrail Railroad Bridge and Hains Point (Appendix I).

Chemical Contaminants in the Tidal Anacostia River and Washington Ship Channel

The cores were sectioned at various intervals for both sediment volume and historical dating calculations, however not all sections were analyzed (Table 2). Below is a description of the forty

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ANACOSTIA RIVER
SUB-BOTTOM CROSS-SECTION
LINE 11

---SURFACE

---BOTTOM
~ 7 METERS

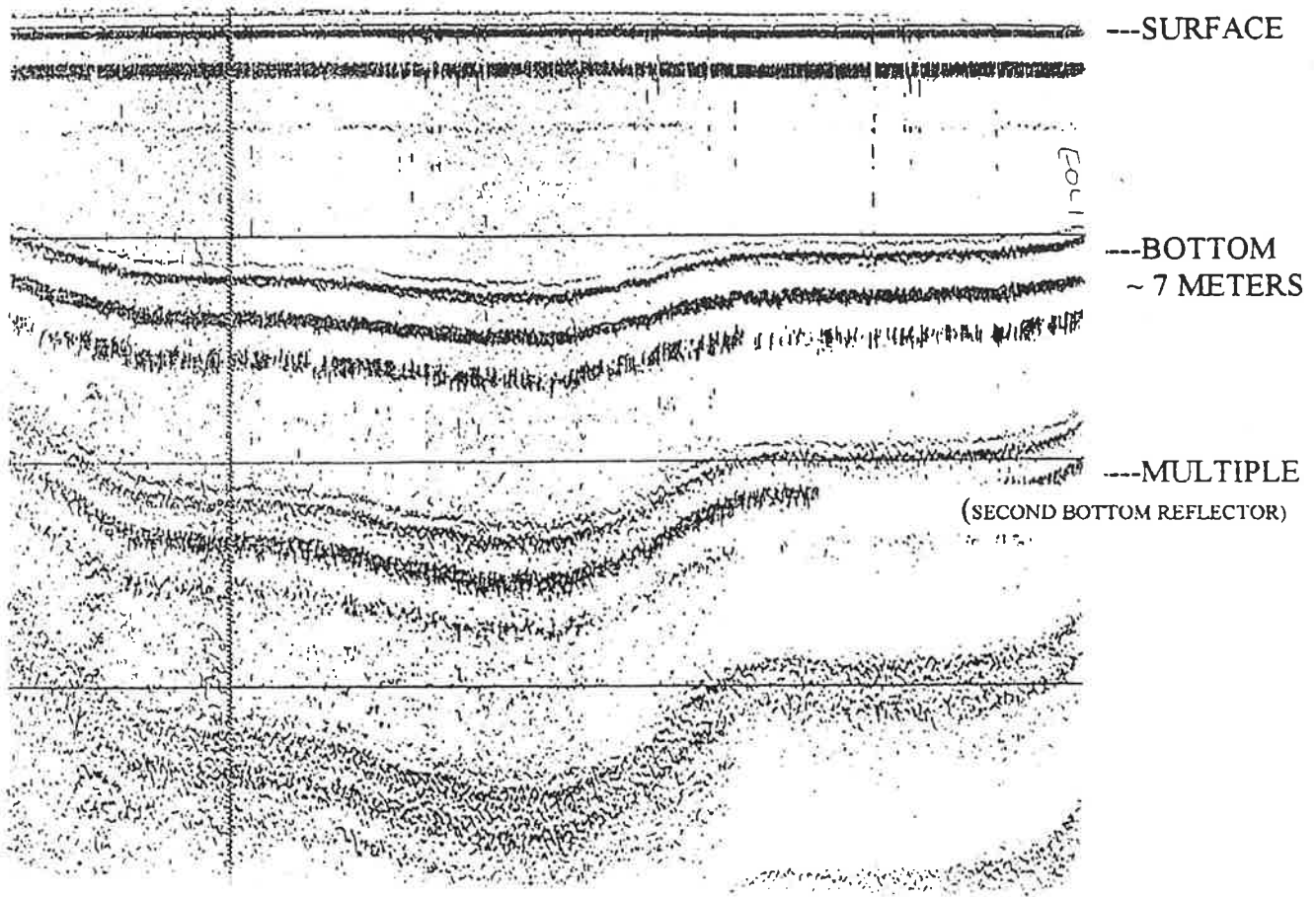----MULTIPLE
(SECOND BOTTOM REFLECTOR)

Figure 3. Sub-bottom profile of transect line 8 located above the Washington Navy Yard (see Appendix I).

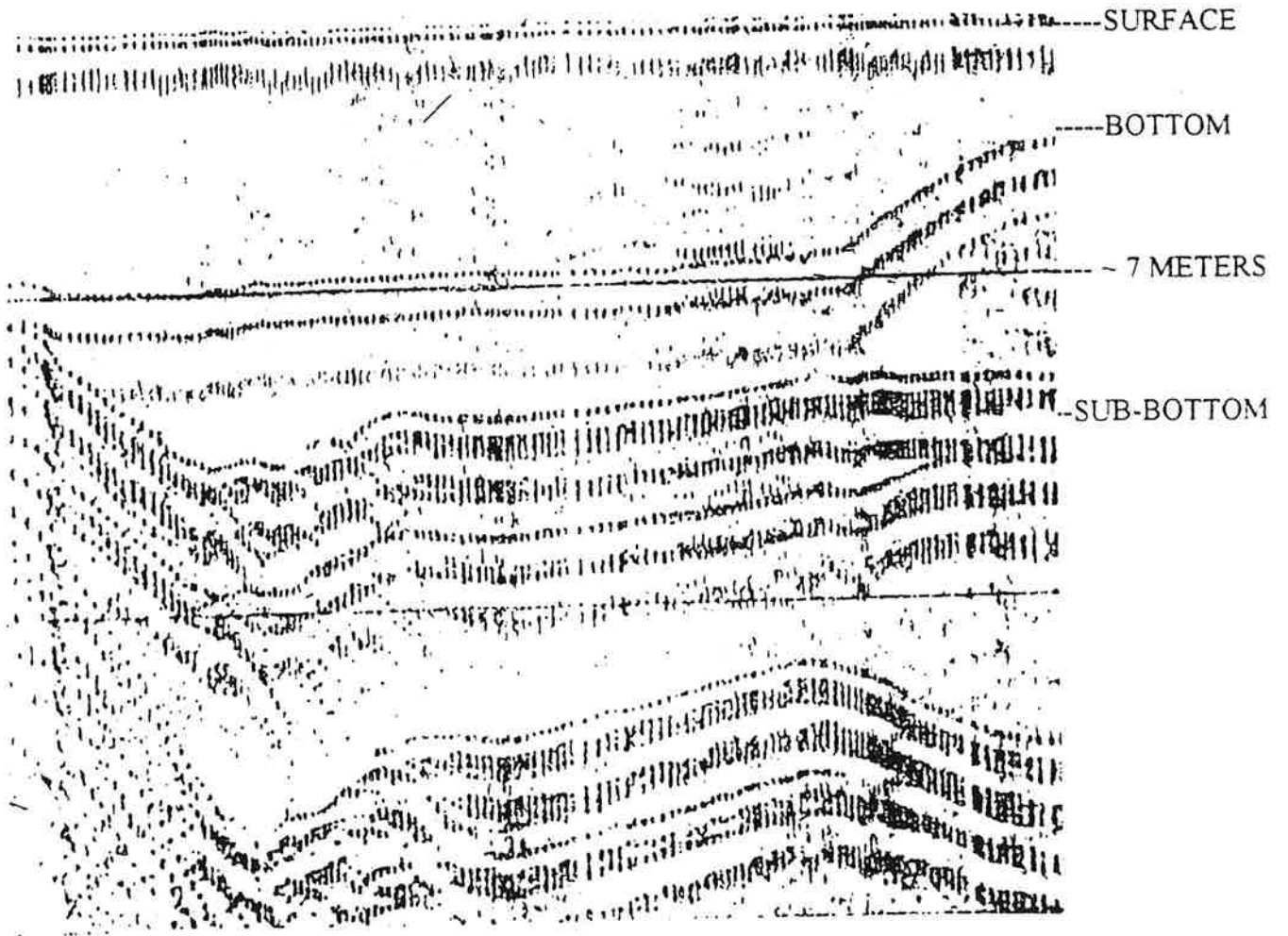# ANACOSTIA RIVER
SUB-BOTTOM CROSS-SECTION
LINE 8

Figure 4. Sub-bottom profile of transect line 11 located above the Anacostia Bridge (see Appendix I).

sediment sections that were analyzed for the eight cores.

<u>^{210}Pb and ^{137}Cs Analyses.</u> The activities of ^{210}Pb (reported as disintegrations per minute per gram of sediment; dpm/g) are reported in Table 3 and shown in Figure 5. In general, the counting statistics show a standard error of < 5% for these analyses. Three of the eight cores were sectioned at smaller intervals (variable starting at 0-2.5 or 5 cm; ANC-1, ANC-6, and WC-2) for ^{210}Pb analysis. However, due to the complexity of the Anacostia system, other cores were analyzed to provide a more accurate assessment of the sedimentation rates within the area. These cores were sectioned in larger intervals (0 to 20 or 50 cm; ANC-3, ANC-4, ANC-5, WC-1), limiting their use in historical evaluations of contaminant inputs, but providing some information regarding sedimentation rates in the river (see below).

Activities of ^{210}Pb for all cores ranged from approximately 4.5 to 7.4 dpm/g at the surface to between 1.5 and 3.2 dpm/g at the bottom (variable depths; Table 3). These activities are similar to those determined by Brush et al. (1982) for cores taken throughout the Potomac estuary, including the Washington Ship Channel, and from the mid-Chesapeake Bay (Cornwell et al., 1996). There are, however, substantial variations in ^{210}Pb activities profiles between cores of this study area, reflecting the urban landscape and sediment dynamics (i.e., transport, dredging, runoff) within the tidal river.

There are three different distributions observed from the cores taken within the tidal river. For cores ANC-1, ANC-6, and to a lesser extent WC-2, activities of ^{210}Pb are fairly uniform (4 to 6 dpm/g) in the surface 50 cm, decreasing to approximately 1.6 to 2.5 dpm/g in the bottom sections (Figure 5). The second distribution was observed from the core taken at ANC-4. The activities of ^{210}Pb start at 5.5 dpm/g at the surface then decrease slightly and appear to be fairly constant at 3.2 dpm/g from 70-120 cm to the bottom. Lastly, cores ANC-3, ANC-5 and WC-1 exhibit decreasing activities of ^{210}Pb from the surface to bottom layers (Figure 5). It is these latter three profiles that exhibit the exponential character expected from "normal" depositional patterns. The profiles of ^{210}Pb activity are shaped by 1) the rate of ^{210}Pb input, 2) the rate and time course of sediment inputs, 3) the activity of bioturbating organisms that mix the upper sediments and 4) other disturbances such as dredging. In a physically dynamic environment such as the Anacostia,

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Table 3a. Lead-210 activities for sediment cores sections from the Anacostia River.

Core ID	Depth Range (cm)	Mid-Point (cm)	Water (%)	Cumulative Mass (g/cm²)	Pb-210 Activity (dpm/g)	Counting Error (dpm/g)
WC-1	0-20	10	72.7	8.69	7.24	0.19
	20-70	45	62.0	37.46	3.92	0.13
	70-120	95	58.2	76.96	2.88	0.12
	120-170	145		96.39		
	170-220	193	46.9	113.60	1.99	0.09
	220-240	248	46.5	130.82		
WC-2	0-5	3	71.7	2.16	7.40	0.58
	5-10	8	78.9	6.58	6.87	0.18
	10-15	13	69.6	10.96	7.80	0.20
	15-20	18	73.7	15.28	7.31	0.40
	20-25	23	78.6	19.72		
	25-30	28	55.3	23.87	4.72	0.13
	30-40	35	69.1	30.00	4.49	0.16
	40-50	45	64.8	38.35		
	50-60	55	63.1	46.51	4.49	0.17
	60-70	65	60.2	54.52		
	70-80	75	57.1	62.32	5.24	0.16
	80-90	85	58.0	70.04		
	90-100	95	59.9	77.86		
	100-125	113	59.7	91.65	3.49	0.15
	125-150	138	57.9	111.18	2.76	0.11
	150-175	165	50.3	129.82	2.46	0.10
	175-200	188	47.2	147.42		
	200-250	225	46.5	173.17	2.38	0.05
	250-270	260				
	280-300	290	44.3			

Table 3b. Lead-210 activities for sediment cores sections from the Anacostia River.

Core ID	Depth Range (cm)	Mid-Point (cm)	Water (%)	Cumulative Mass (g/cm²)	Pb-210 Activity (dpm/g)	Counting Error (dpm/g)
ANC-1	0-5	3	68.2	2.11	4.46	0.29
	5-10	8	62.8	6.24	4.86	0.15
	10-15	13	61.1	10.25	5.04	0.20
	15-20	18	59.2	14.20	4.55	0.08
	20-25	23	56.0	18.06	4.00	0.17
	25-30	28	58.7	21.91		
	30-40	35	58.3	27.75	5.24	0.16
	40-50	45	55.2	35.42	4.89	0.20
	50-60	55	53.9	42.92	4.99	0.21
	60-70	65	51.6	50.28		
	70-80	75	49.6	57.47	2.56	0.06
	80-90	85	49.3	64.56		
	90-100	95	51.4	71.73		
	100-125	113	50.1	78.94		
	125-150	138	45.5	90.96	2.47	0.06
	150-175	165	49.5	108.29		
	175-200	188	46.6	125.74		
	200-250	225	42.4	142.41	1.85	0.05
	250-270	263	38.8	158.18		
	280-300	290	36.0	171.68		
ANC-2	0-20	10	65.8	8.28		
	20-70	45	56.2	35.61		
	70-120	95	51.7	72.87		
	120-170	145	47.5	108.41		
	170-220	195	47.0	142.97		
	220-260	239	48.0	173.10		
	280-300	290	46.2	192.82		

Table 3c. Lead-210 activities for sediment cores sections from the Anacostia River.

Core ID	Depth Range (cm)	Mid-Point (cm)	Water (%)	Cumulative Mass (g/cm²)	Pb-210 Activity (dpm/g)	Counting Error (dpm/g)
ANC-3	0-20	10	65.7	8.28	6.44	
	20-70	45	56.3	35.64	5.38	
	70-120	95	51.4	72.87	3.86	
	120-170	145	49.8	108.84	2.77	
	170-220	195	37.7	141.72	1.63	
	280-300	290	36.5	162.67		
ANC-4	0-20	10	56.2	7.63	5.49	0.19
	20-70	45	52.0	33.50		
	70-120	95	48.3	69.25	3.51	0.13
	120-170	145	48.5	104.28		
	170-220	195	49.9	139.66	3.44	0.18
	220-270	245	48.3	174.99		
	270-280	275	45.8	204.36	3.23	0.17
	280-300	290	46.1	223.05		
ANC-5	0-20	10	60.9	7.96	7.21	0.12
	20-70	45	54.7	34.70	8.05	0.14
	70-120	95	47.3	70.79	4.80	0.14
	120-170	145	40.9	103.94		
	170-220	195	42.1	135.91	3.22	0.08
	220-250	236	45.4	162.84	3.08	0.11
	280-300	290	46.6	180.50		

Table 3d. Lead-210 activities for sediment cores sections from the Anacostia River.

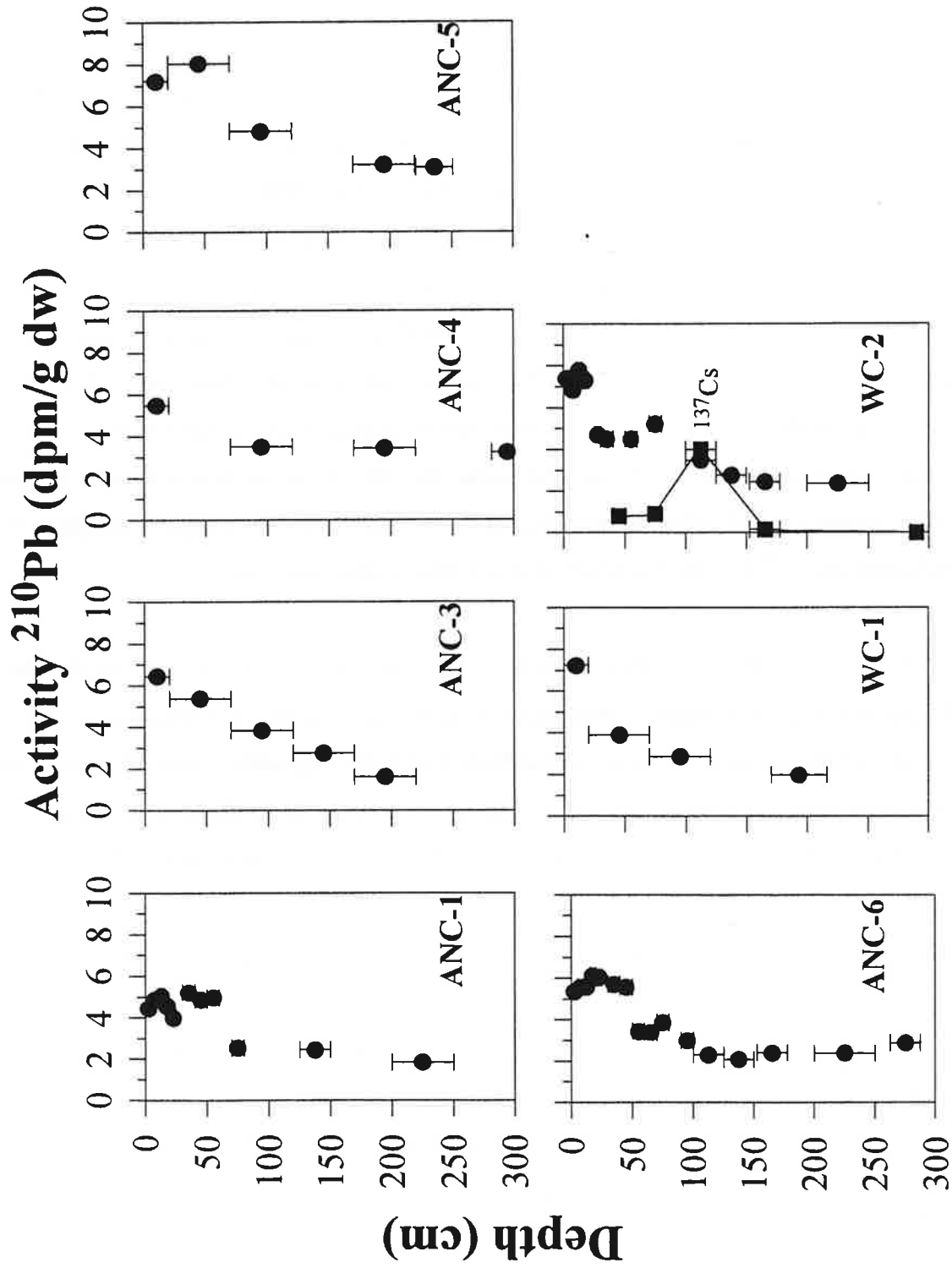| Core ID	Depth Range (cm)	Mid-Point (cm)	Water (%)	Cumulative Mass (g/cm²)	Pb-210 Activity (dpm/g)	Counting Error (dpm/g)
ANC6	0-5	3	51.9	1.82	5.36	0.57
	5-10	8	50.0	5.43	5.58	0.04
	10-15	13	49.3	8.99	5.59	0.10
	15-20	18	51.3	12.57	6.13	0.10
	20-25	23	50.5	16.18	6.03	0.44
	25-30	28	47.3	19.71		
	30-40	35	48.4	24.94	5.70	0.10
	40-50	45	46.7	31.88	5.56	0.10
	50-60	55	39.1	38.39	3.46	0.07
	60-70	65	39.8	44.59	3.41	0.08
	70-80	75	37.6	50.71	3.89	0.10
	80-90	85	34.8	56.57		
	90-100	95	37.6	62.43	3.01	0.06
	100-125	113	37.0	72.87	2.32	0.07
	125-150	138	41.7	88.33	2.10	0.05
	150-175	165	41.8	104.38	2.43	0.05
	175-200	188	45.3	120.83		
	200-250	225	45.3	146.11	2.41	0.06
	250-280	265	44.7	179.69	2.91	0.12
	280-300	290	46.0	203.21		

Figure 5. Sediment Depth Profiles of ^{210}Pb activity Within the Anacostia River and Washington Ship Channel.

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variable sediment inputs and dredging disturbances may have a large impact on ^{210}Pb profiles and on the calculation of apparent rates of sedimentation. In general though, higher activities of ^{210}Pb are indicative of newer sediments.

Selected intervals were analyzed for the activity of ^{137}Cs in the sediments. These samples by Dr. Thomas Church at the University of Delaware using a Ge-Si gamma counting system. Cesium-137 is a human-made radionuclide produced from the testing and use of atomic bombs and energy. Extensive testing of atomic bombs commenced in the early 1950's with a peak in the early 1960's (i.e., 1963) enables the presence and distribution of ^{137}Cs to be useful in determining specific sediment-time horizons in many environments (see Ritchie and McHenry, 1990).

Sediments from the bottom sections from all cores as well as specific intervals from core WC-2 were counted. For all bottom sections ^{137}Cs activities were below the detection limit (< 0.05 dpm/g) indicating that the bottom of the cores were approximately at a time horizon of 1950 or older. In core WC-2, activities of ^{137}Cs peaked within the 100-125 cm section at 4.0 ± 0.1 dpm/g (activity ± 1σ counting error), with low activities at 150-175 cm (0.16 ± 0.03 dpm/g) (Figure 5). As with the other cores, ^{137}Cs was not detected at the base of this core ages.

Grain Size and Organic Carbon. Grain size distributions in the various cores throughout the study were fairly uniform with the sediments predominately in the clay and silt size fraction (<63 μm; Table 4). The silt+clay fraction (<63 μm) varied from 84.1 to 99.7%, with an overall average and standard deviation of 96.1 ± 3.6% (n=40). Other than two core top sections for ANC-4 and ANC-6, which had silt+clay of 85% and 84% respectively, all sections were greater than 90% silt+clay.

There was a slight increase with depth in the silt+clay fraction for most cores (Table 4). The largest increase was for ANC-6 in which the percent silt+clay increased from 84% in the top 5 cm to 99% at approximately 290 cm. Other cores exhibited increases of only a few percent in the amount of silt+clay.

Sediment organic carbon (OC) concentrations ranged between 2.0 and 7.6 % on a dry weight basis (dw) for all sediment samples with an average of 3.5 ± 1.2 %OC (± standard deviation; Table 4; Figure 6). Generally, there was a slight decrease in organic carbon content with depth

Table 4a. Trace metals, grain size, and organic carbon for core intervals from the Anacostia River and Washington Ship Channel*.

Sta ID/ (cm)	Mid-Depth (cm)	Al (%)	Cd (µg/g)	Cr (µg/g)	Cu (µg/g)	Fe (%)	Hg (µg/g)	Ni (µg/g)	Pb (µg/g)	Zn (µg/g)	Sand (%)	Silt (%)	Clay (%)	Silt+Clay (%)	TOC (%)
ANC 1: 0-5	2.5	7.59	0.72	83.0	46.0	4.43	0.19	51.6	41.1	196.0	5.6	49.4	45.0	94.4	4.30
ANC 1: 20-25	22.5	7.86	0.74	77.0	55.0	4.59	0.19	54.2	46.7	230.0	6.5	58.1	35.4	93.5	3.93
ANC 1: 50-60	55.0	8.46	0.79	98.0	52.0	5.00	0.18	58.4	54.9	257.0	3.1	47.3	49.6	96.9	3.32
ANC 1: 90-100	95.0	8.53	0.88	101.0	50.0	4.70	0.19	58.4	69.7	273.0	1.6	46.7	51.7	98.4	3.27
ANC 1: 280-300	290.0	7.66	0.65	88.5	56.5	4.74	1.03	57.4	167.8	249.2	3.1	61.2	35.6	96.8	2.09
ANC 2: 0-20	10.0	6.76	0.78	88.4	50.6	4.87	0.23	62.5	50.3	240.7	3.5	64.8	31.7	96.5	3.70
ANC 2: 20-70	45.0	8.65	0.81	96.0	49.0	4.93	0.21	57.8	54.5	251.0	1.6	44.7	53.8	98.4	3.55
ANC 2: 120-170	145.0	7.43	1.33	110.1	68.3	5.10	0.38	69.5	131.1	374.8	1.5	57.1	41.4	98.5	2.77
ANC 2: 170-220	195.0	9.29	1.21	152.0	66.0	5.12	0.40	63.9	94.4	449.0	1.2	39.2	59.6	98.8	2.92
ANC 2: 280-300	290.0	7.94	1.75	120.1	77.4	4.70	0.34	65.5	100.9	356.0	1.0	58.7	40.3	99.0	2.82
ANC 3: 0-20	10.0	7.56	1.24	97.6	69.0	4.84	0.26	67.7	91.3	330.4	1.2	57.0	41.8	98.8	3.45
ANC 3: 20-70	45.0	9.27	1.20	109.0	74.0	5.29	0.25	64.2	96.8	295.0	1.0	44.7	54.3	99.0	3.11
ANC 3: 120-170	145.0	8.21	2.03	142.0	87.2	4.84	0.83	64.7	147.4	416.5	2.2	49.2	48.5	97.8	2.93
ANC 3: 170-220	195.0	9.00	1.20	95.0	64.0	4.86	0.78	47.4	109.0	244.0	1.8	43.2	55.0	98.2	2.12
ANC 3: 280-300	290.0	7.44	0.15	70.8	27.6	4.27	0.07	51.9	23.9	106.9	1.2	56.0	42.8	98.8	1.51
ANC 4: 0-20	10.0	6.85	2.95	154.2	151.3	4.10	0.73	82.3	525.7	648.8	14.8	48.0	37.1	85.2	5.14
ANC 4: 20-70	45.0	9.51	3.27	210.0	125.0	4.73	0.60	80.3	392.0	574.0	3.5	38.0	58.5	96.5	3.94
ANC 4: 120-170	145.0	8.67	2.67	226.3	137.0	4.98	0.99	65.0	441.7	587.9	6.9	40.6	52.5	93.1	3.92
ANC 4: 220-270	195.0	6.69	3.21	150.0	122.0	3.85	1.23	41.8	311.0	461.0	9.8	34.7	55.5	90.2	7.19
ANC 4: 295-300	297.0	7.66	5.19	168.9	153.3	4.38	2.24	51.7	519.7	744.2	6.2	35.3	58.4	93.8	7.69
ANC 5: 0-20	10.0	7.01	1.95	113.5	81.8	4.48	0.26	72.3	154.5	404.0	3.1	59.7	37.2	96.9	3.60
ANC 5: 20-70	45.0	8.44	1.89	123.0	81.0	4.81	0.29	64.7	147.0	388.0	4.3	56.4	39.3	95.7	4.07
ANC 5: 120-170	145.0	9.03	2.03	141.7	66.2	4.84	0.27	72.2	240.8	389.6	2.7	52.0	45.3	97.3	2.06
ANC 5: 170-220	190.0	10.22	1.69	214.0	72.0	5.12	0.34	57.0	237.0	401.0	2.1	38.4	59.5	97.9	2.25
ANC 5: 280-300	290.0	9.36	1.73	157.2	81.1	5.30	0.52	53.0	227.2	299.2	1.9	33.7	64.4	98.1	2.61
ANC 6: 0-5	2.5	7.13	1.68	102.0	66.0	4.09	0.25	59.8	132.0	328.0	15.9	56.1	28.0	84.1	3.95
ANC 6: 40-50	45.0	7.81	2.08	129.0	73.0	4.28	0.30	63.8	187.0	389.0	8.2	57.0	34.8	91.8	4.05
ANC 6: 90-100	95.0	8.45	3.06	150.0	66.0	4.20	0.27	67.6	289.0	439.0	5.3	53.2	41.5	94.7	2.55
ANC 6: 280-300	290.0	9.88	6.33	131.0	117.0	5.09	0.68	41.1	162.0	424.0	1.3	39.1	59.6	98.7	3.43

* All concentrations are on a dry weight basis.

Table 4b. Trace metals, grain size, and organic carbon for core intervals from the Anacostia River and Washington Ship Channel*.

Sta ID/ (cm)	Mid-Depth (cm)	Al (%)	Cd (µg/g)	Cr (µg/g)	Cu (µg/g)	Fe (%)	Hg (µg/g)	Ni (µg/g)	Pb (µg/g)	Zn (µg/g)	Sand (%)	Silt (%)	Clay (%)	Silt+Clay (%)	TOC (%)
WC 1: 0-20	10.0	6.92	1.31	100.1	125.6	5.56	0.51	74.9	295.8	413.3	6.1	57.7	36.2	93.9	3.99
WC 1: 20-70	45.0	8.63	1.47	131.0	121.0	5.03	0.97	76.4	264.0	384.0	9.7	44.1	46.2	90.3	4.82
WC 1: 120-170	145.0	8.03	1.12	104.6	98.7	5.66	6.54	54.5	435.9	406.5	4.7	33.1	62.2	95.3	3.94
WC 1: 170-220	190.0	9.62	0.67	146.0	65.0	5.50	7.15	57.0	1026.0	308.0	2.4	25.5	72.0	97.6	3.48
WC 1: 220-240	230.0	6.86	0.36	101.9	41.4	4.61	10.84	52.8	604.5	200.2	2.1	35.7	62.3	97.9	2.97
WC 2: 0-5	2.5	8.29	0.94	103.7	83.7	5.71	0.41	71.1	115.9	366.8	1.6	52.0	46.4	98.4	2.95
WC 2: 5-10	7.5	8.99	0.96	103.0	83.0	5.39	0.35	68.4	107.0	305.0	2.8	44.0	53.2	97.2	3.50
WC 2: 20-25	22.5	9.04	0.98	102.0	78.0	5.47	0.33	69.1	98.0	316.0	1.3	39.5	59.2	98.7	3.26
WC 2: 50-60	55.0	8.37	1.42	141.9	83.2	5.60	0.56	77.5	210.0	421.4	0.4	34.0	65.7	99.7	2.67
WC 2: 90-100	95.0	9.21	1.51	199.8	94.3	6.06	0.84	78.0	198.9	456.2	0.5	27.8	71.8	99.5	2.59
WC 2: 290-300	295.0	8.08	1.48	114.5	75.4	5.45	1.36	62.7	185.5	386.4	1.1	35.6	63.3	98.9	2.00

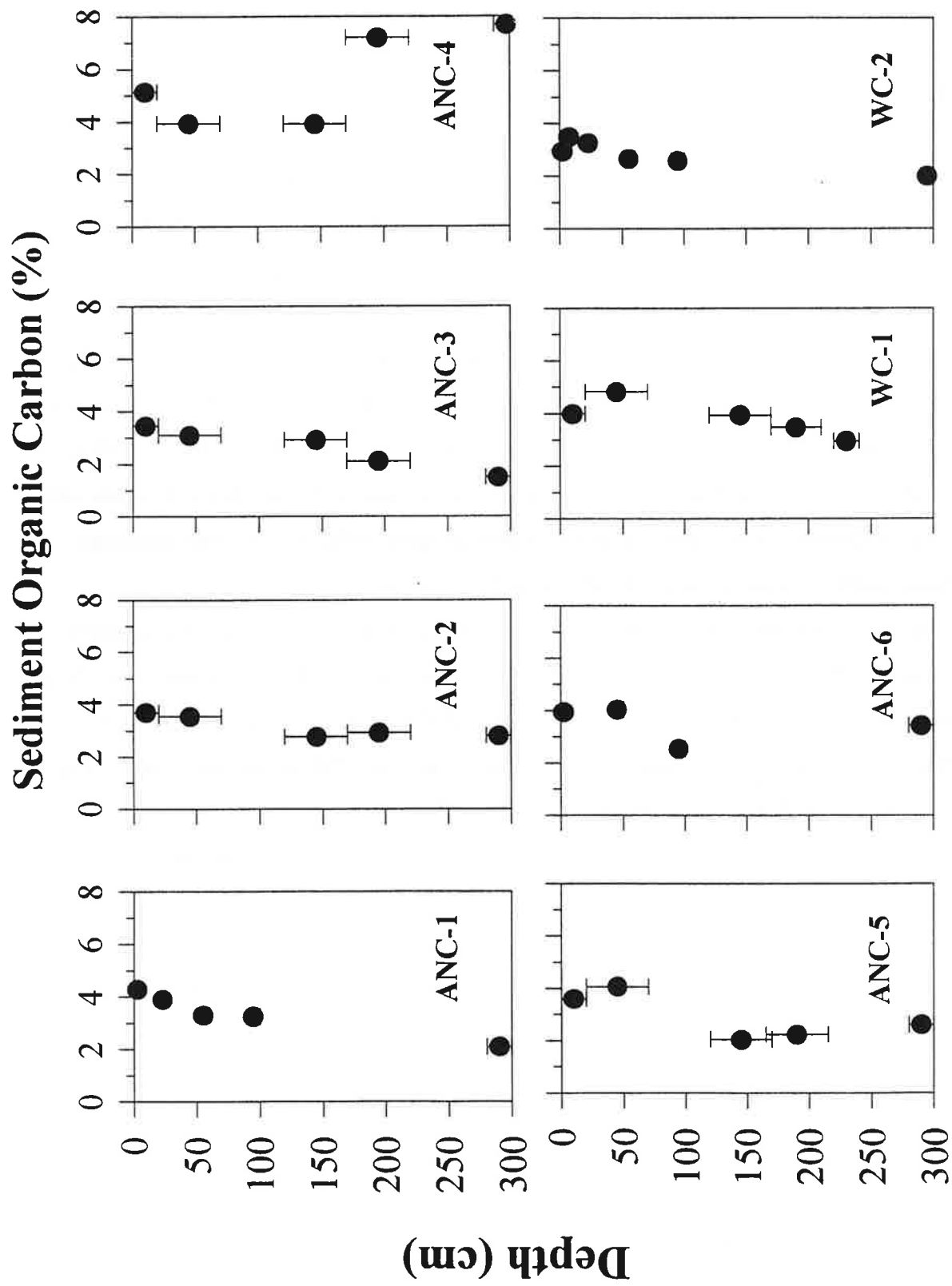* All concentrations are on a dry weight basis.

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Figure 6. Sediment Depth Profiles of Organic Carbon Within the Anacostia River and Washington Ship Channel.

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for all cores except ANC-4. In this core organic carbon increased in the bottom sections to greater than 7% OC. This was the highest for any core section analyzed. Sediment organic carbon profiles with depth in the cores are presented in Figure 6. Concentrations of organic carbon generally decrease from 4 to 5 %OC to 2 to 3 %OC near the bottom.

Sedimentary Trace Metals. Table 4 provides the trace metal data for all cores that were taken as part of this study. It should be pointed out that the points shown on the depth distribution graphs (Figure 7) are the mid-point of specific depth ranges that were homogenized to provide samples for analysis.

While the sediment concentration of many trace metals were both horizontally and vertically variable, the sedimentary concentrations and distributions of both iron (Fe) and aluminum (Al) were fairly constant with averages (± standard deviation) of 4.91 ± 0.50% and 8.26 ± 0.92% respectively (Figures 7a,b). Sedimentary Fe and to a lesser degree Al were fairly invariant with depth. These elements, which are less prone to anthropogenic influence, can help determine background levels of other trace metals in this area (see below).

Cadmium (Cd) concentrations ranged from 0.16 to 6.3 µg Cd/g dw. Lower concentrations were measured throughout core ANC-1 and highest overall concentrations, between 5 and 6 µg Cd/g dw in the bottom sections of cores ANC-4 and ANC-6 (Figure 7c). In core ANC-3, concentrations ranged between 1 and 2 µg Cd/g dw in the upper 150 cm and decreased to the lowest levels measured in the all cores of 0.16 µg Cd/g dw at 280 cm.

Sediment concentrations of nickel (Ni) ranged from 41.1 to 82.3 µg Ni/g dw for all cores. Highest concentrations were found in the upper sections (0 to 70 cm) of ANC-4 which is located between the Navy Yard and the South Capital Street Bridge (Figure 7d). Concentrations of chromium (Cr) and copper (Cu) ranged between 70.8 and 226.3 µg Cr/g dw and 27.6 and 153.3 µg Cu/g dw for all cores. For both trace metals, concentrations were highest in core ANC-4 and lowest in ANC-1 or the bottom section of ANC-3 (Figures 7e,f). Concentrations of Ni, Cr, and Cu exhibited little variation with depth in most cores (Figure 7d-f).

Concentrations of both mercury (Hg) and lead (Pb) exhibited substantial differences between core areas and with depth in some cores (Figure 7g,h). Overall concentrations ranged from 0.07
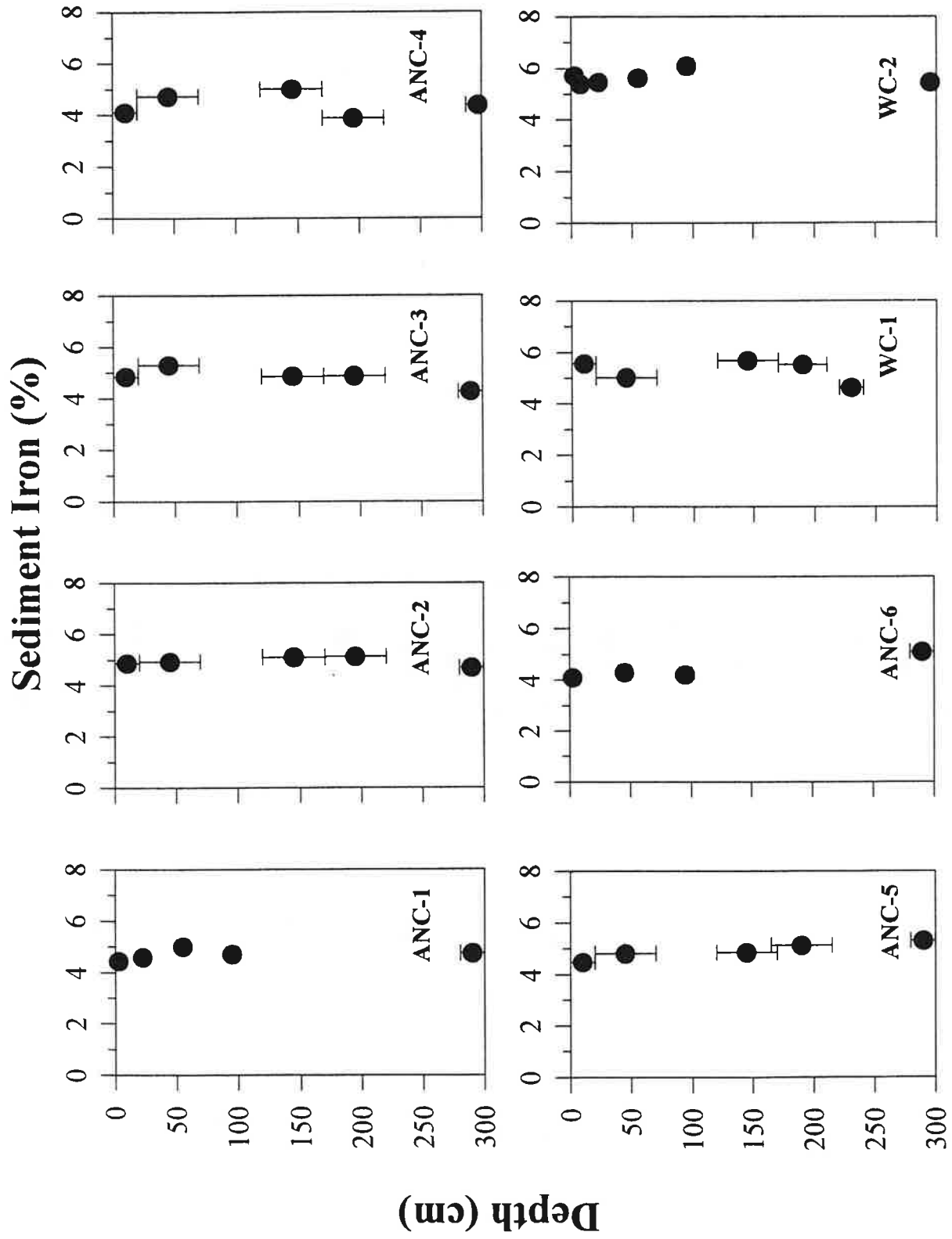
34

Figure 7a. Sediment Depth Profiles of Iron Within the Anacostia River and Washington Ship Channel.
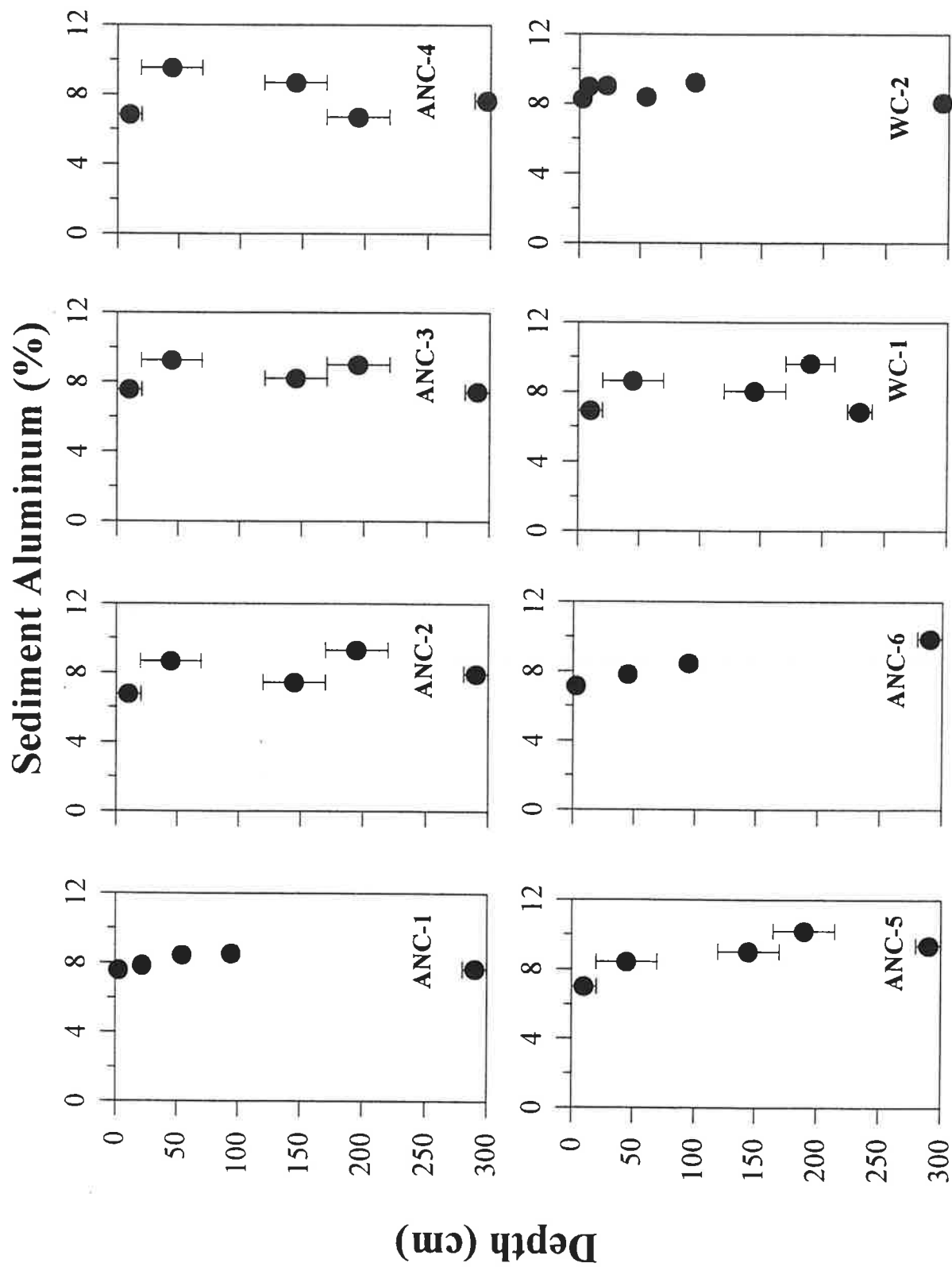
35

Figure 7b. Sediment Depth Profiles of Aluminum Within the Anacostia River and Washington Ship Channel.
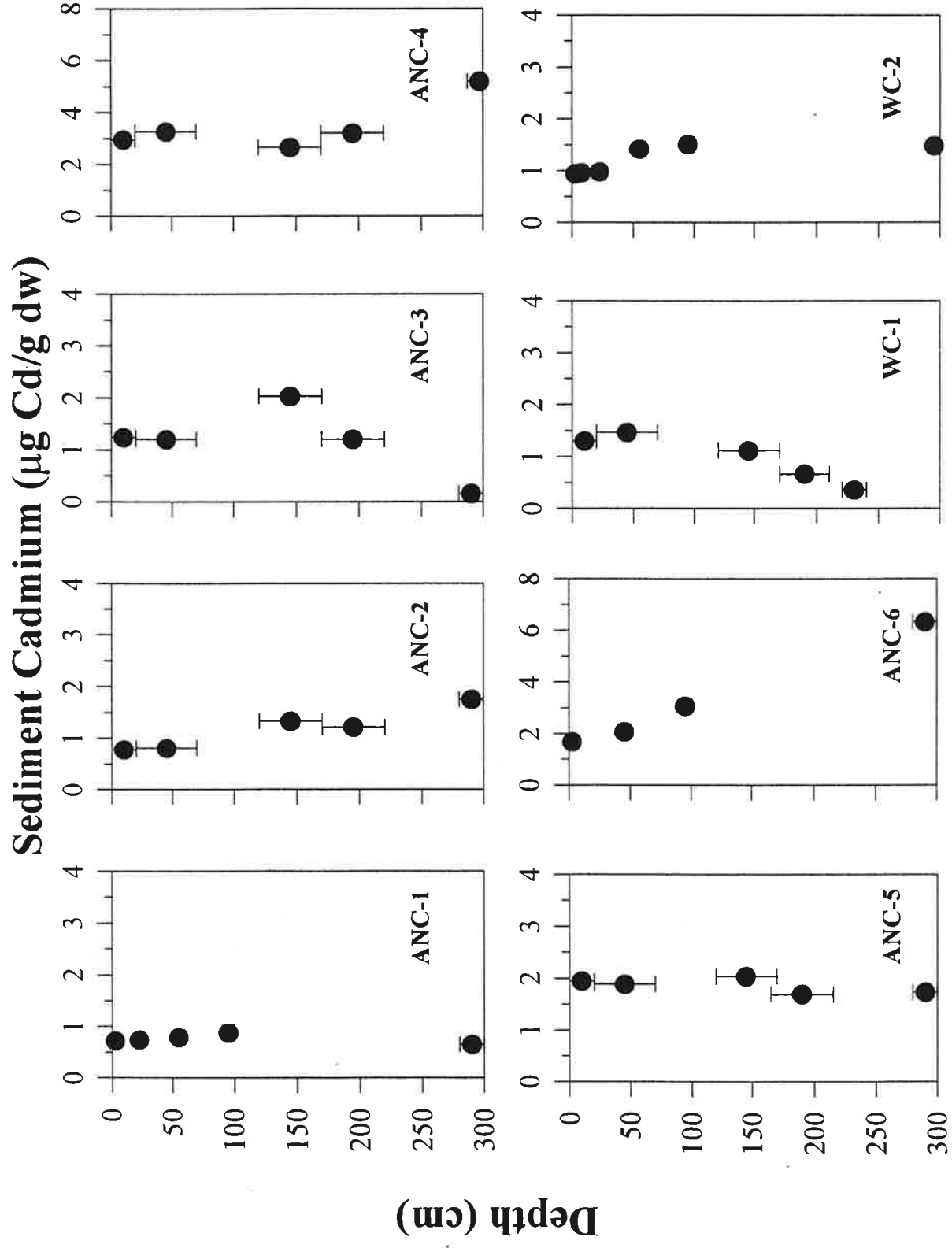
36

Figure 7c. Sediment Depth Profiles of Cadmium Within the Anacostia River and Washington Ship Channel.

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Figure 7d. Sediment Depth Profiles of Nickel Within the Anacostia River and Washington Ship Channel.

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Figure 7e. Sediment Depth Profiles of Chromium Within the Anacostia River and Washington Ship Channel.

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Figure 7f. Sediment Depth Profiles of Copper Within the Anacostia River and Washington Ship Channel.

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Figure 7g. Sediment Depth Profiles of Mercury Within the Anacostia River and Washington Ship Channel.

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Figure 7h. Sediment Depth Profiles of Lead Within the Anacostia River and Washington Ship Channel.
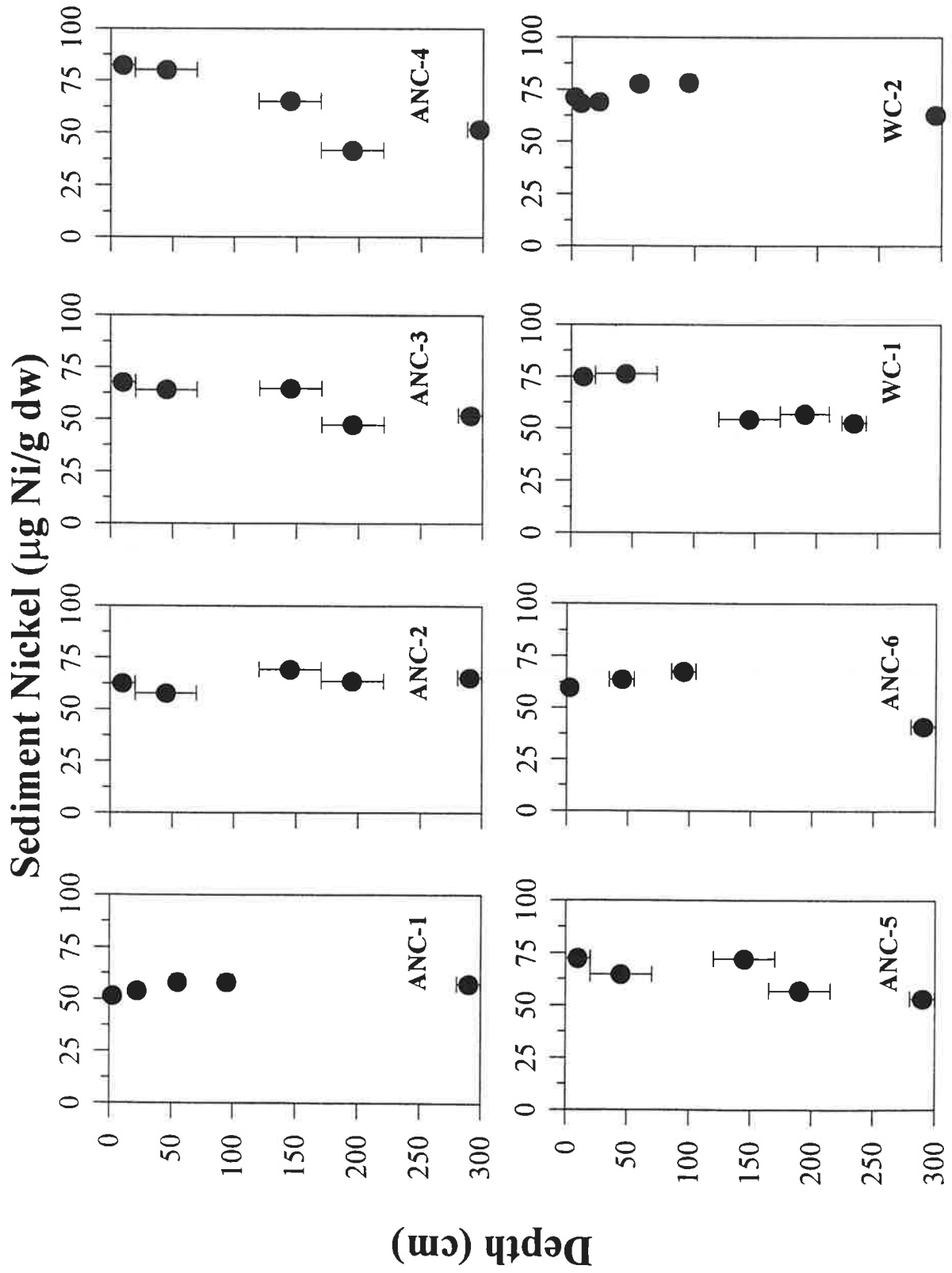
42

Figure 7i. Sediment Depth Profiles of Zinc Within the Anacostia River and Washington Ship Channel.
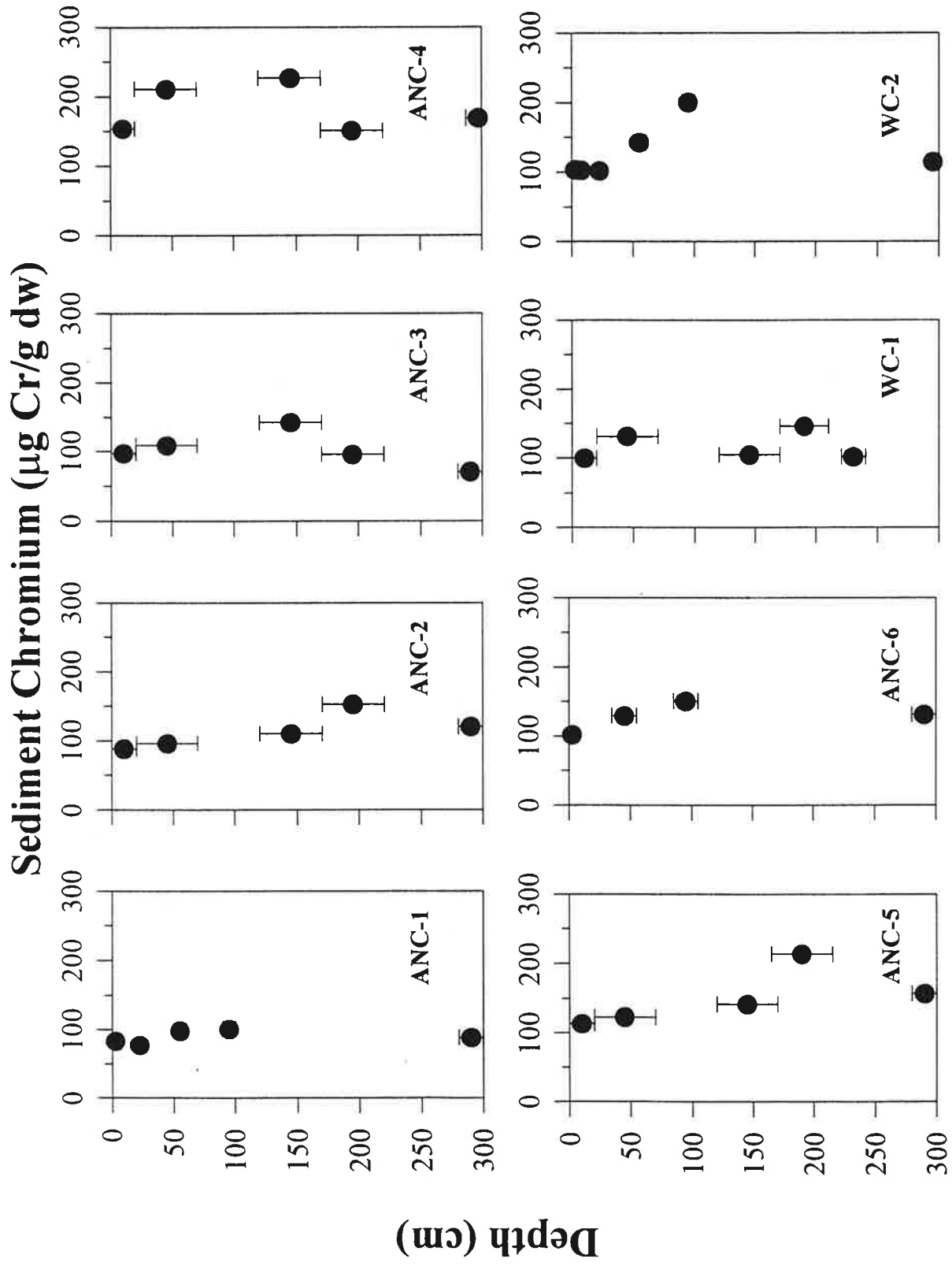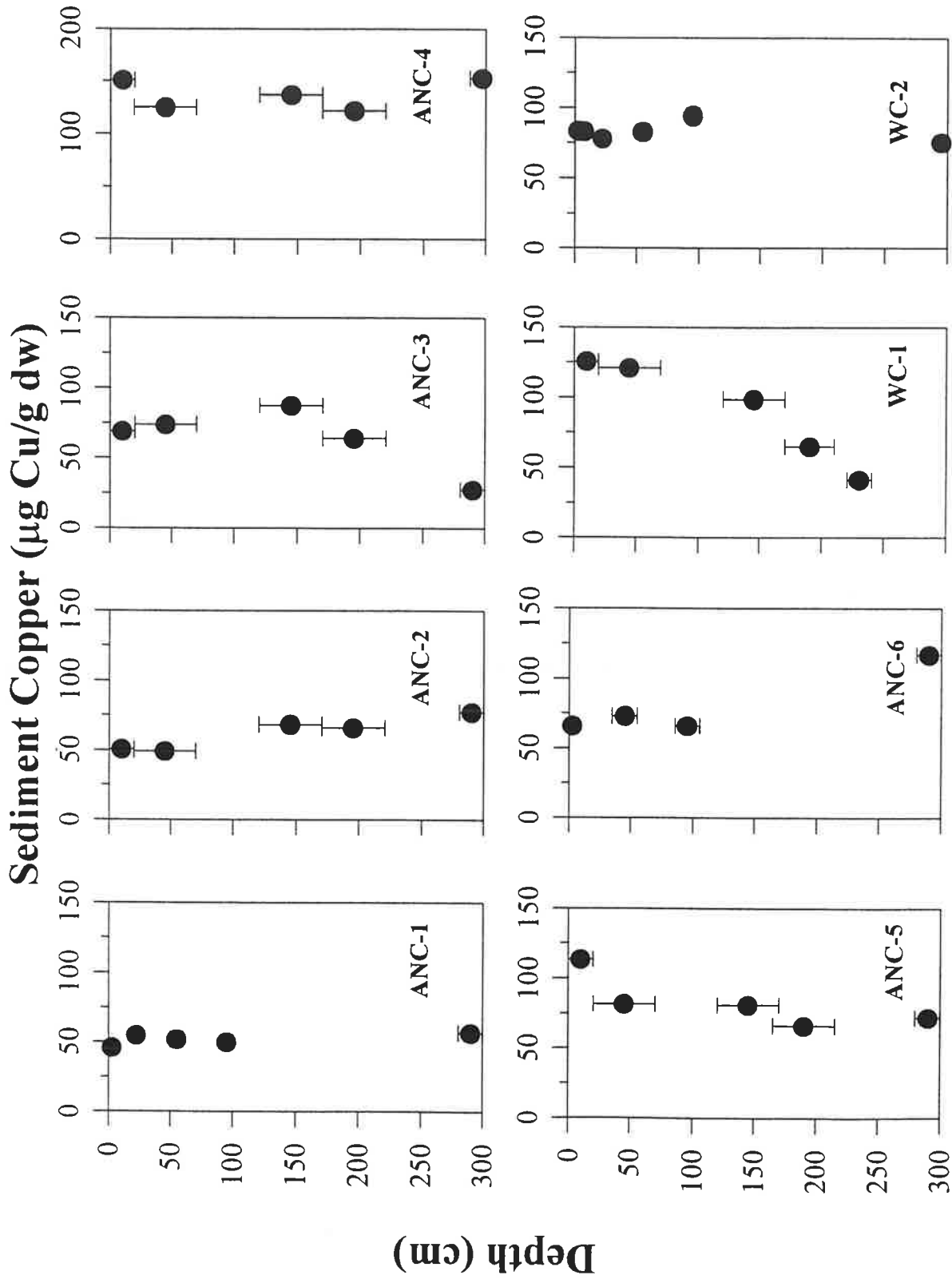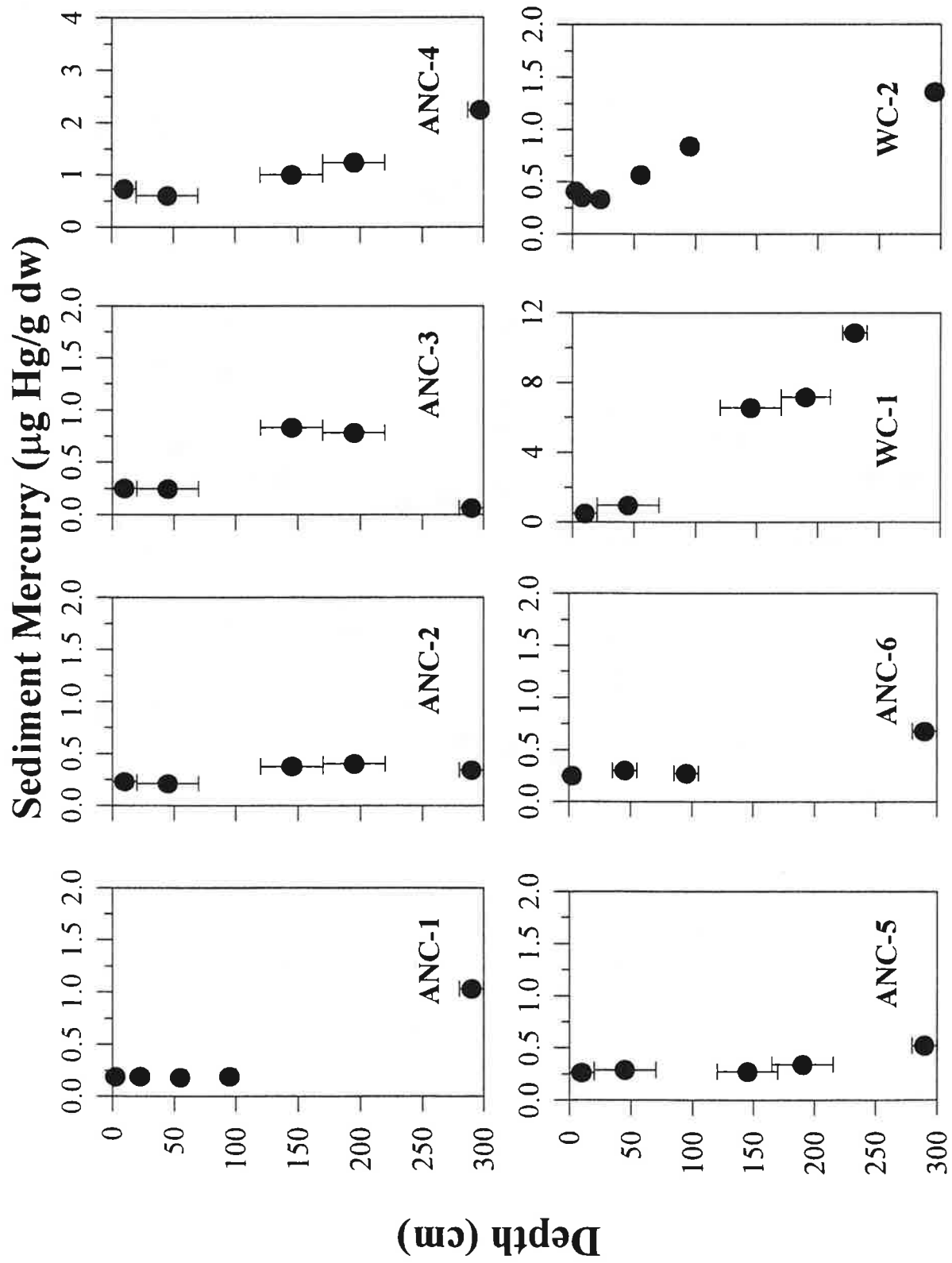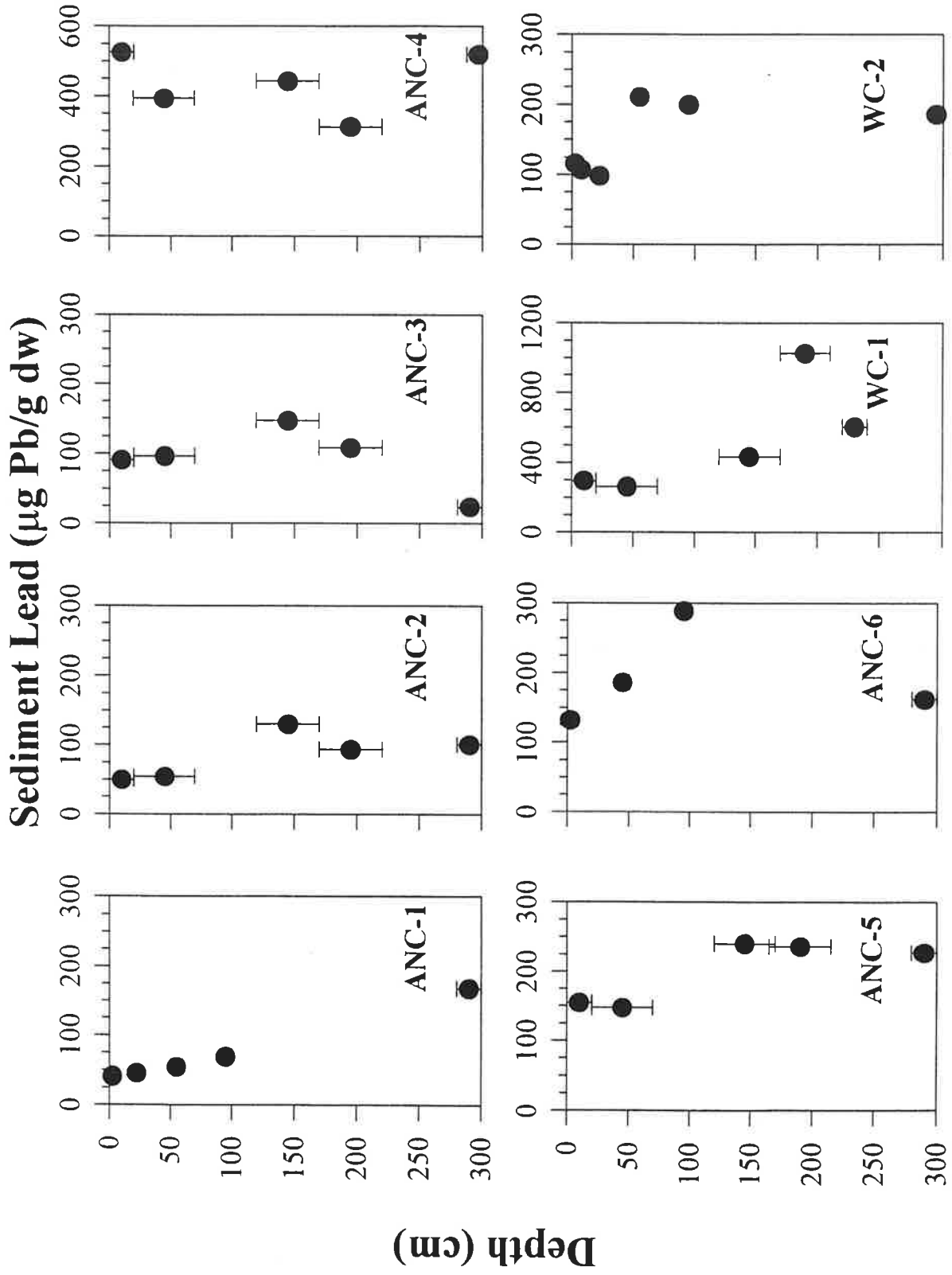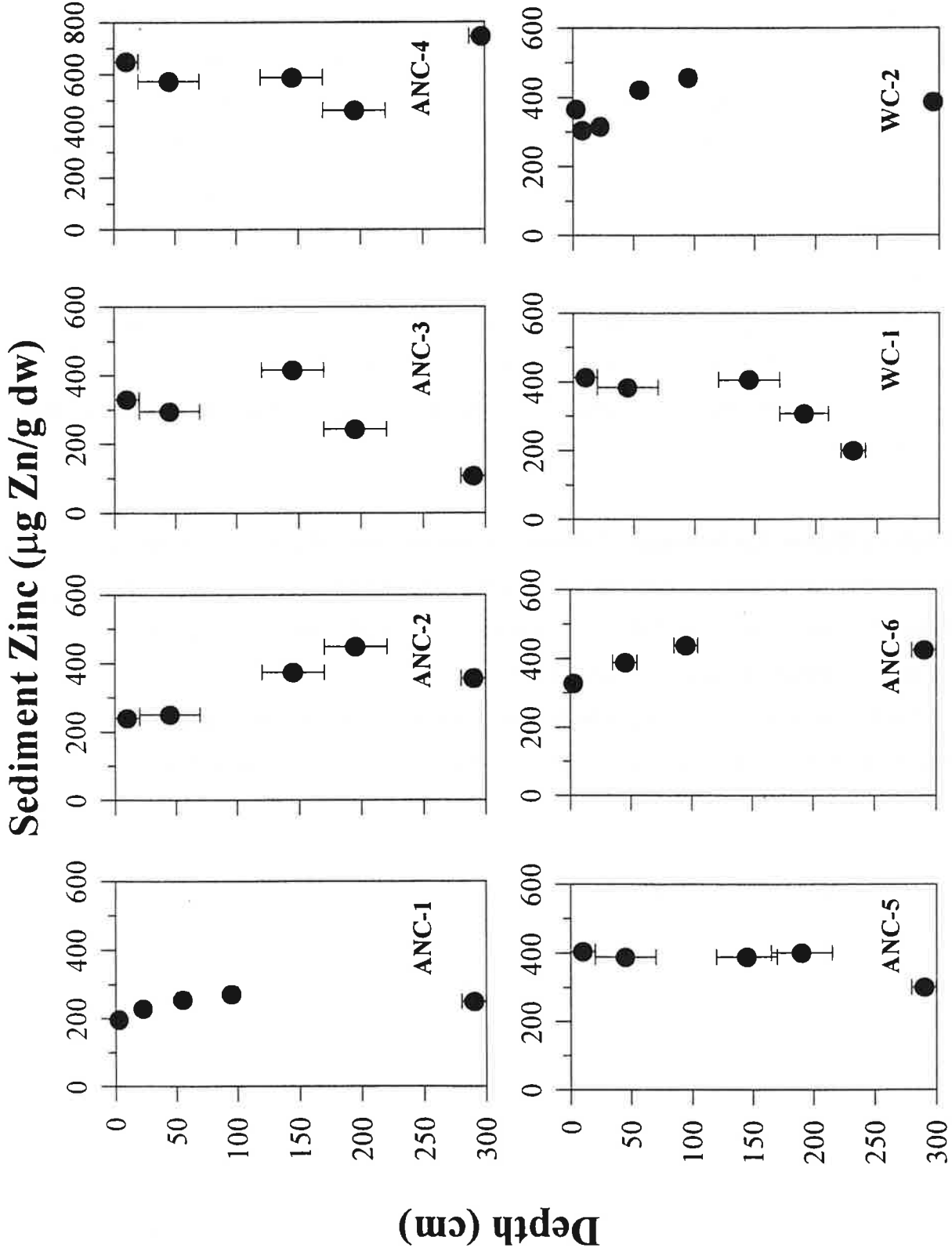
43

to 10.8 μg Hg/g dw for Hg and from 23 to 1,026.0 μg Pb/g dw for Pb. Lowest concentrations were observed for both trace metals in the bottom section (280-300 cm) of ANC-3. Sedimentary Hg was elevated at station ANC-4 compared to other Anacostia cores, and increased to 2.2 μg Hg/g dw in the 295-300 cm section. Highest concentrations were found in the deeper sections of WC-1 in the upper Washington Ship Channel. Concentrations increased from 0.51 μg Hg/g dw in the surface section (0-20 cm) to 10.8 μg Hg/g dw in the bottom 280-300 cm section. Similar surface concentrations were observed in WC-2, which is located down channel (Figure 1), with a steady increase to over 1 μg Hg/g dw in the bottom section (Figure 7g). Concentrations of Pb exhibited a similar distribution with elevated concentrations throughout the core ANC-4 within the Anacostia River (Figure 7h). Sedimentary concentrations of Pb in core ANC-4 ranged from 311 to 525 μg Pb/g dw with no substantial variation with depth. Again, in core WC-1, concentrations of Pb increased substantially from the surface (295 μg Pb/g dw) to 1,026 μg Pb/g dw at 170-216 cm.

<u>Sedimentary Organic Contaminants.</u> Sedimentary concentrations of polycyclic aromatic hydrocarbons (PAHs), total congener specific polychlorinated biphenyls (PCBs), total DDTs, and total chlordane are presented in Table 5. The concentrations of individual compounds that compose these groups are given in Appendix II.

Total sedimentary hydrocarbons can be broken down into three fractions: saturated hydrocarbons (SHC), polycyclic aromatic hydrocarbons (PAH), and the unresolved complex mixture (UCM). The UCM contains co-eluting compounds that are not resolved by current capillary gas chromatographic techniques, and is thought to be mainly alicyclic hydrocarbons. Saturated hydrocarbons are the sum of normal alkanes from n-C_{13} to n-C_{34} including the isoprenoids pristane and phytane, while PAHs are the sum of 44 individual aromatic hydrocarbons (Appendix II). For this study, only PAHs were determined because they are potentially more harmful to aquatic organisms then the other fractions.

DDT is the sum of DDT (1,1-(2,2,2-trichloroethylidene)bis[4-chlorobenzene]) and its breakdown products, DDE (1,1-(2,2,2-trichloroethenylidene)bis[4-chlorobenzene]) and DDD (1,1-(2,2-dichloroethylidene)bis[4-chlorobenzene]), including the o+p and p+p isomers.

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Table 5a. Selected organic contaminants, grain size and organic carbon for core sections from the Anacostia River and Washington Ship Channel*.

Sta ID/cm	Mid-Depth (cm)	∑Chlordane's (ng/g)	∑DDT's (ng/g)	∑PCB's (ng/g)	∑PAHs (ng/g)	∑LMW PAHs (ng/g)	∑HMW PAHs (ng/g)	Sand (%)	Silt (%)	Clay (%)	Silt+Clay (%)	TOC (%)
ANC 1: 0-5	2.5	14.1	15.9	117.6	5407.8	1703.1	3704.7	5.6	49.4	45.0	94.4	4.30
ANC 1: 20-25	22.5	12.3	33.3	120.3	4247.1	1501.6	2745.6	6.5	58.1	35.4	93.5	3.93
ANC 1: 50-60	55.0	10.7	26.2	163.8	3548.3	1117.8	2430.5	3.1	47.3	49.6	96.9	3.32
ANC 1: 90-100	95.0	10.7	24.3	135.8	3694.5	1472.6	2221.9	1.6	46.7	51.7	98.4	3.27
ANC 1: 280-300	290.0	1.8	1.1	19.2	13465.1	7216.6	6248.5	3.1	61.2	35.6	96.8	2.09
ANC 2: 0-20	10.0	16.7	21.5	128.3	3801.9	1292.5	2509.4	3.5	64.8	31.7	96.5	3.70
ANC 2: 20-70	45.0	13.3	27.3	164.7	3408.7	1459.5	1949.2	1.6	44.7	53.8	98.4	3.55
ANC 2: 120-170	145.0	14.1	37.1	358.2	5771.3	3232.5	2538.8	1.5	57.1	41.4	98.5	2.77
ANC 2: 170-220	195.0	17.4	48.1	576.6	4629.6	2724.9	1904.8	1.2	39.2	59.6	98.8	2.92
ANC 2: 280-300	290.0	17.1	101.0	601.3	6265.0	3620.6	2644.4	1.0	58.7	40.3	99.0	2.82
ANC 3: 0-20	10.0	35.9	27.8	216.8	7778.0	1871.5	5906.5	1.2	57.0	41.8	98.8	3.45
ANC 3: 20-70	45.0	23.4	37.6	271.7	5899.6	2546.8	3352.8	1.0	44.7	54.3	99.0	3.11
ANC 3: 120-170	145.0	20.7	166.2	1053.6	8338.7	5103.3	3235.4	2.2	49.2	48.5	97.8	2.93
ANC 3: 170-220	195.0	0.4	12.9	81.9	8838.4	4850.2	3988.2	1.8	43.2	55.0	98.2	2.12
ANC 3: 280-300	290.0	0.1	1.0	3.8	106.5	63.0	43.5	1.2	56.0	42.8	98.8	1.51
ANC 4: 0-20	10.0	180.7	209.8	2514.9	48063.9	23980.4	24083.5	14.8	48.0	37.1	85.2	5.14
ANC 4: 20-70	45.0	86.8	106.5	1803.1	20203.9	13435.2	6768.8	3.5	38.0	58.5	96.5	3.94
ANC 4: 120-170	145.0	167.6	686.9	3811.7	27049.5	15816.1	11233.4	6.9	40.6	52.5	93.1	3.92
ANC 4: 220-270	195.0	120.6	467.8	1891.9	34883.6	21483.4	13400.1	9.8	34.7	55.5	90.2	7.19
ANC 4: 295-300	297.0	54.4	440.1	1475.8	49328.8	25737.2	23591.7	6.2	35.3	58.4	93.8	7.69
ANC 5: 0-20	10.0	122.6	64.7	584.8	23992.2	7432.2	16560.0	3.1	59.7	37.2	96.9	3.60
ANC 5: 20-70	45.0	63.3	56.8	496.5	13809.5	4576.0	9233.5	4.3	56.4	39.3	95.7	4.07
ANC 5: 120-170	145.0	120.5	99.8	1200.1	8461.7	4817.3	3644.3	2.7	52.0	45.3	97.3	2.06
ANC 5: 170-220	190.0	44.4	150.2	928.5	8369.5	4762.4	3607.1	2.1	38.4	59.5	97.9	2.25
ANC 5: 280-300	290.0	39.1	199.1	1064.5	28224.7	14757.1	13467.6	1.9	33.7	64.4	98.1	2.61
ANC 6: 0-5	2.5	128.3	81.9	481.5	16426.0	6401.5	10024.4	15.9	56.1	28.0	84.1	3.95
ANC 6: 40-50	45.0	90.6	50.1	470.7	19434.0	9564.8	9869.2	8.2	57.0	34.8	91.8	4.05
ANC 6: 90-100	95.0	78.0	63.1	690.0	10507.0	5589.3	4917.7	5.3	53.2	41.5	94.7	2.55
ANC 6: 280-300	290.0	12.7	62.6	208.6	31064.4	12236.3	18828.1	1.3	39.1	59.6	98.7	3.43

*All concentrations are on a dry weight basis. LMW and HMW PAHs are the sum of 2-3 and 4-5 ring compounds respectively. (see text)

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Table 5b. Selected organic contaminants, grain size and organic carbon for core sections from the Anacostia River and Washington Ship Channel*.

Sta ID/cm	Mid-Depth (cm)	ΣChlordane's (ng/g)	ΣDDT's (ng/g)	ΣPCB's (ng/g)	ΣPAHs (ng/g)	ΣLMW PAHs (ng/g)	ΣHMW PAHs (ng/g)	Sand (%)	Silt (%)	Clay (%)	Silt+Clay (%)	TOC (%)
WC 1: 0-20	10.0	27.9	92.3	990.1	17771.7	6099.1	11672.6	6.1	57.7	36.2	93.9	3.99
WC 1: 20-70	45.0	19.6	233.2	1419.3	17907.0	12016.7	5890.3	9.7	44.1	46.2	90.3	4.82
WC 1: 120-170	145.0	9.0	3.3	166.1	17004.0	7170.8	9833.1	4.7	33.1	62.2	95.3	3.94
WC 1: 170-220	190.0	2.0	0.6	63.1	19525.8	9254.7	10271.2	2.4	25.5	72.0	97.6	3.48
WC 1: 220-240	230.0	1.6	1.5	14.3	5593.6	2342.5	3251.1	2.1	35.7	62.3	97.9	2.97
WC 2: 0-5	2.5	19.6	42.3	291.5	6768.2	2556.0	4212.1	1.6	52.0	46.4	98.4	2.95
WC 2: 5-10	7.5	14.9	52.6	244.5	5075.4	1509.5	3565.9	2.8	44.0	53.2	97.2	3.50
WC 2: 20-25	22.5	12.1	46.5	264.0	5137.6	1604.0	3533.6	1.3	39.5	59.2	98.7	3.26
WC 2: 50-60	55.0	11.4	65.6	521.4	5063.3	2116.3	2947.0	0.4	34.0	65.7	99.7	2.67
WC 2: 90-100	95.0	14.8	197.7	1326.7	4424.5	2259.4	2165.1	0.5	27.8	71.8	99.5	2.59
WC 2: 290-300	295.0	3.4	36.5	278.1	4614.2	1902.9	2711.3	1.1	35.6	63.3	98.9	2.00

*All concentrations are on a dry weight basis. LMW and HMW PAHs are the sum of 2-3 and 4-5 ring compounds respectively. (see text)

Polychlorinated biphenyls (PCB) are the sum of the 209 possible individual PCB congeners, while total chlordane is the sum of $\gamma+\alpha$-chlordane, cis+trans-nonachlor, and heptachlor and its breakdown product heptachlor epoxide.

Total polycyclic aromatic hydrocarbons (PAHs) ranged from 0.11 to 49.3 µg/g dw with an overall average of 13.3 µg/g dw (Table 5; Figure 8a). The lowest concentration (0.11 µg/g dw) was observed at the bottom of ANC-3 located downstream of the South Capital Street Bridge. This core/section also had some of the lowest trace metal concentrations of all cores. The highest concentrations were found throughout core ANC-4 located downstream of the Washington Navy Yard near the O Street stormwater-combined sewer outfall. Concentrations of total PAHs in this core ranged from 20.2 to 49.3 µg/g dw with similar and highest concentrations near the surface and bottom of the core (Figure 8a). Cores, ANC-1 and WC-2, located at or near the mouth of the Anacostia River, Washington Ship Channel, and Potomac River has some of the lowest concentrations of total PAHs (Table 5; Figures 1 and 8a). Total PAH concentrations generally ranged from 3.5 to 6.7 µg/g dw in these cores except for the 280-300 cm section of core ANC-1 which was 13.4 µg/g dw. There was no consistent trend with depth for total PAHs between cores (Figure 8a).

Concentrations of total chlordane (see above) exhibited substantial variations between location and with depth in many cores. Concentrations ranged from 0.12 to 181 ng/g dw with lowest concentrations in the bottom two sections of ANC-3 and highest concentrations in ANC-4 (Table 5; Figure 8b). The surface sections of most cores exhibited higher concentrations and generally decreased with depth (Figure 8b). The exception is in core ANC-2 in which concentrations were similar from top to bottom with a range of 13 to 17 ng/g dw. Surface sediment concentrations of total chlordane were highest at station ANC-4 and stayed above 100 ng/g dw farther upstream.

The concentrations of total polychlorinated biphenyls (PCBs) exhibited substantial differences between cores and in some cases with depth in each core (Table 5; Figure 8c). Overall, concentrations ranged from 3.8 to 3,810 ng/g dw with the lowest concentration found in the bottom section (280-300 cm) of ANC-3 and highest concentrations throughout core ANC-4. Surface sediment concentrations for total PCBs generally increased from the mouth of the Anacostia and Potomac River (ANC-1) to just below the Washington Navy Yard (ANC-4), with
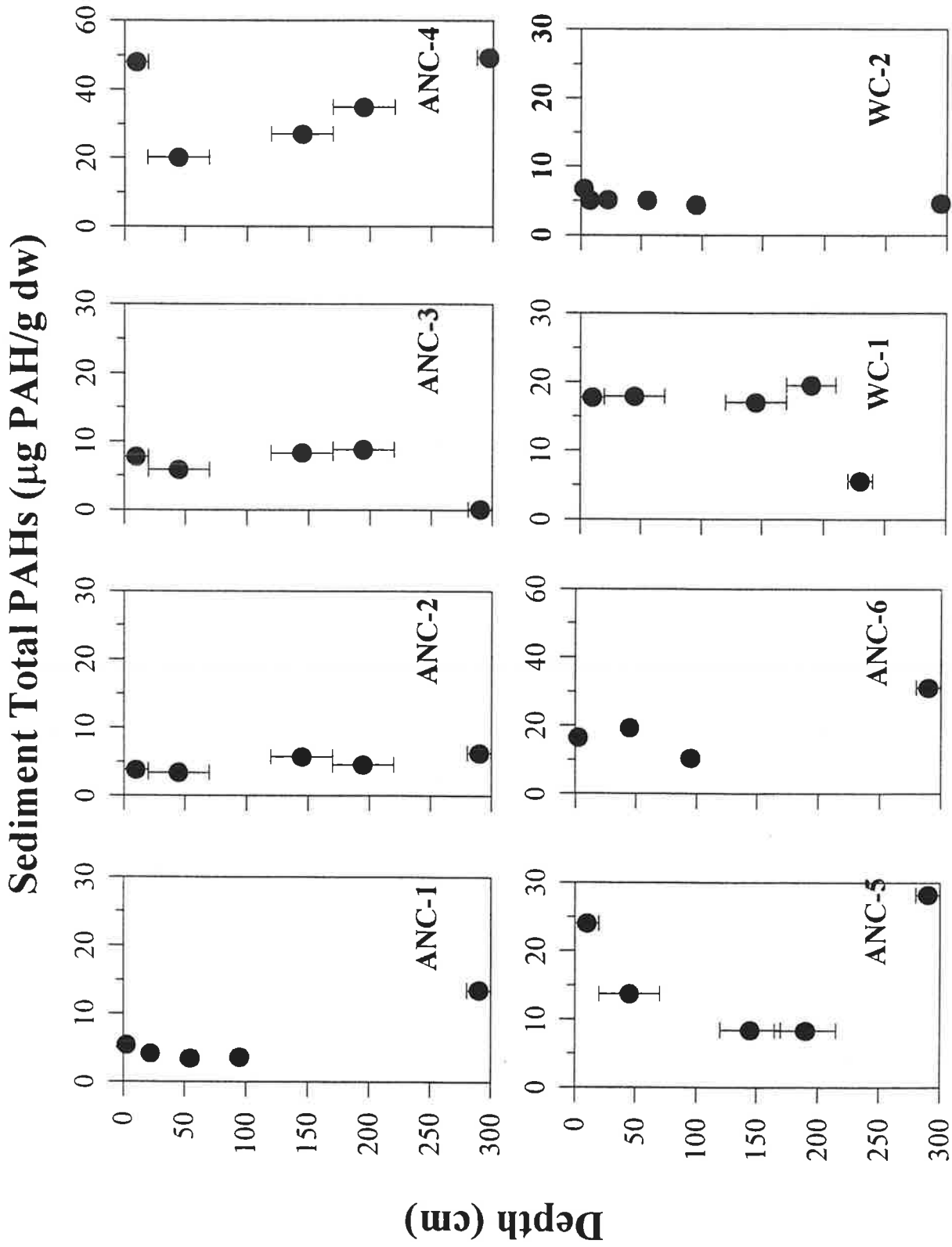
Figure 8a. Sediment Depth Profiles of Total Polycyclic Aromatic Hydrocarbons (PAHs) Within the Anacostia River and Washington Ship Channel.

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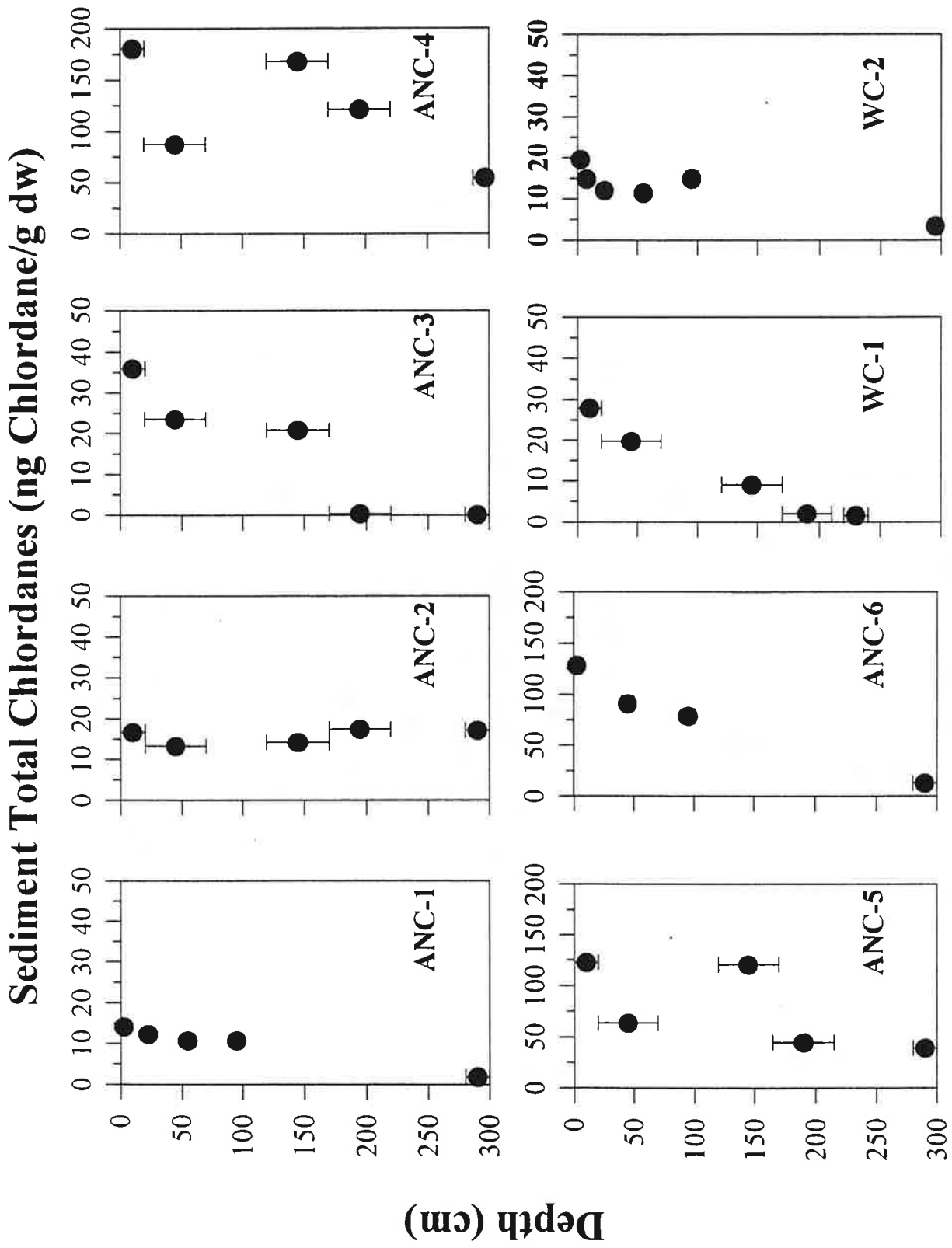# Sediment Total Chlordanes (ng Chlordane/g dw)

Figure 8b. Sediment Depth Profiles of Total Chlordanes Within the Anacostia River and Washington Ship Channel.
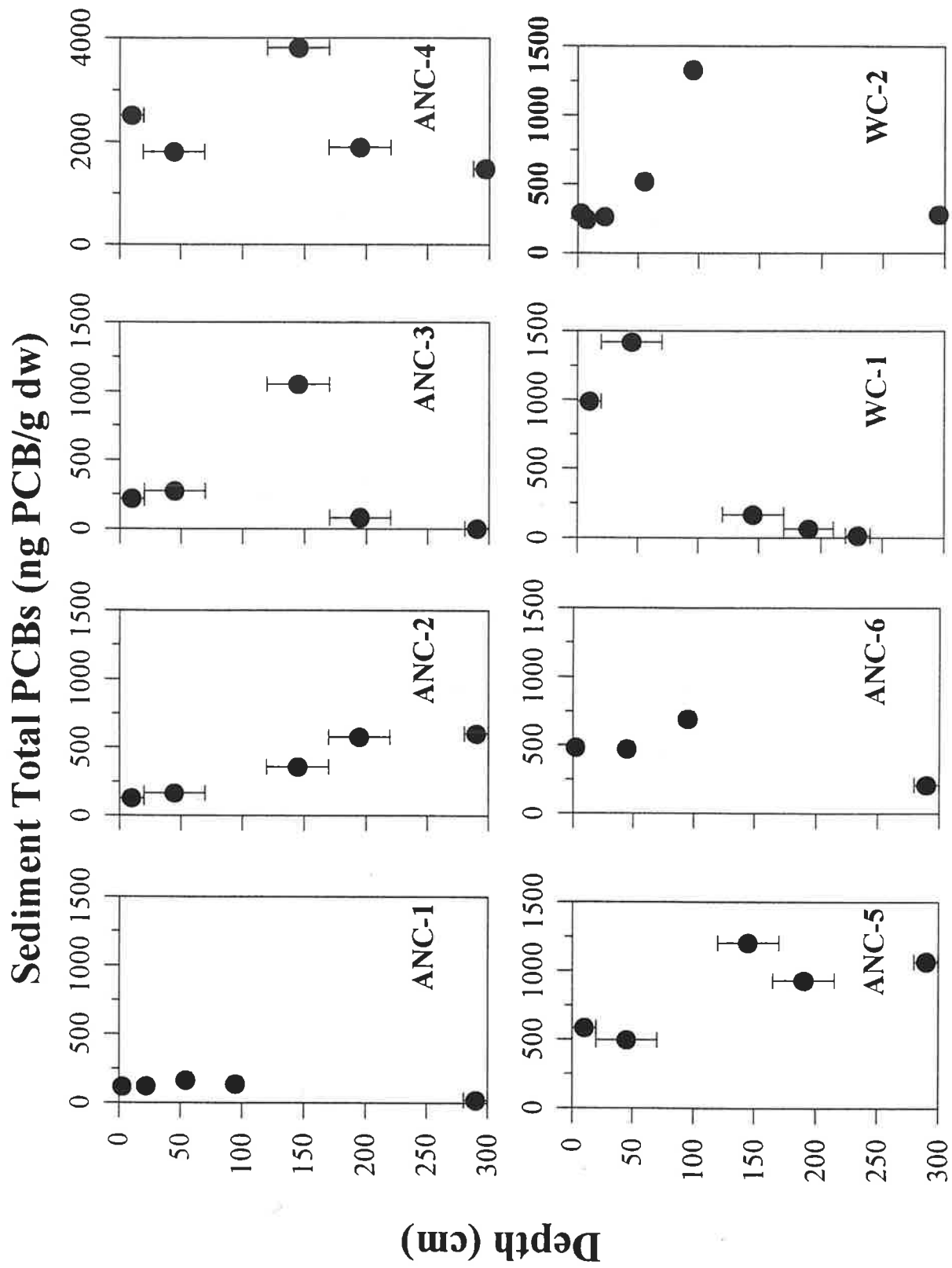
49

Figure 8c. Sediment Depth Profiles of Total Polychlorinated Biphyenls (PCBs) Within the Anacostia River and Washington Ship Channel.
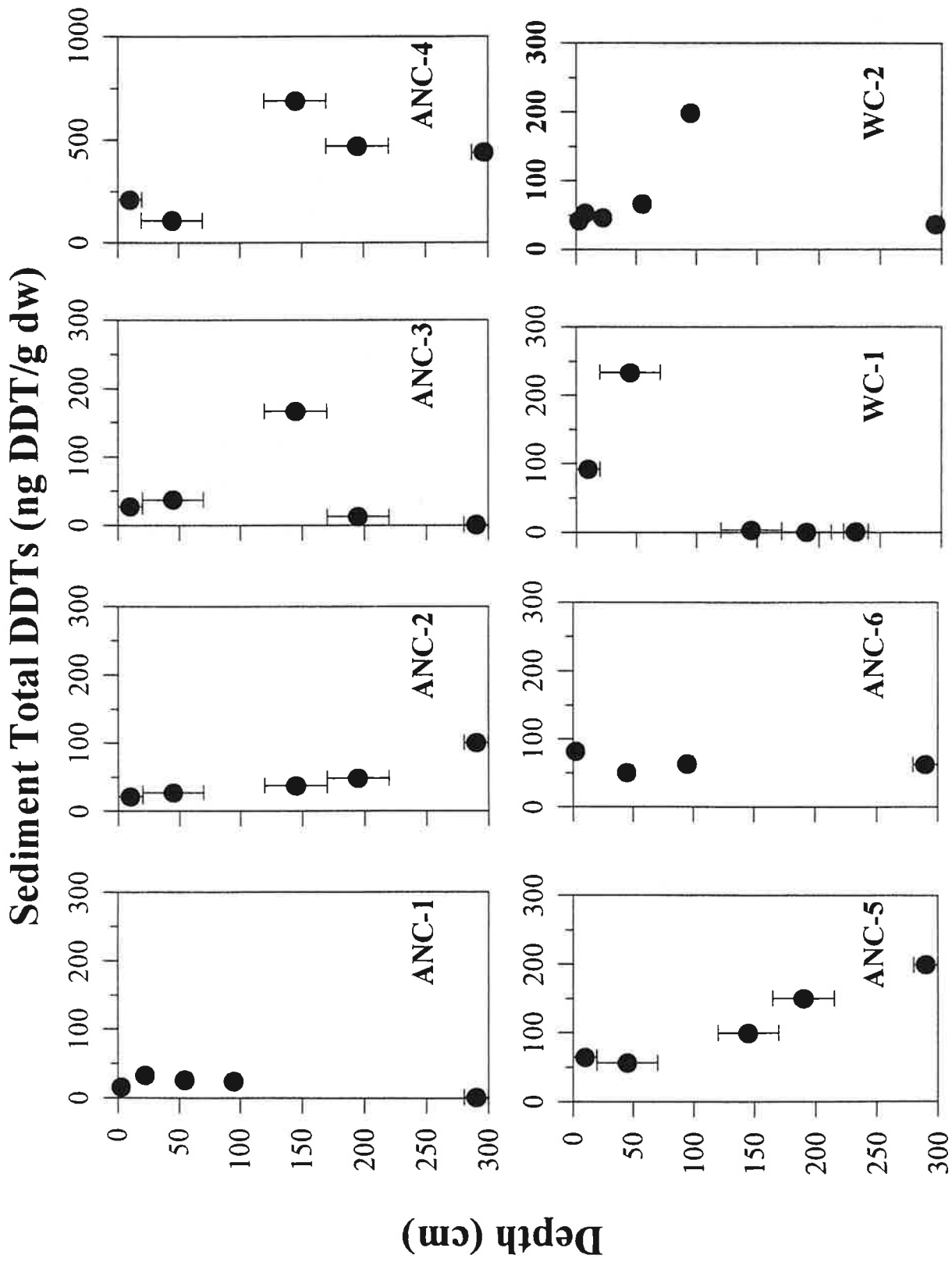
50

Figure 8d. Sediment Depth Profiles of Total DDTs Within the Anacostia River and Washington Ship Channel.

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a slight decrease farther upstream to ANC-6 (Figure 1; Table 5). In the Washington Ship Channel, concentrations exhibited substantial downcore variations. In WC-1, located near the head of the channel (Figure 1), concentrations of total PCBs ranged from 990 to 1,440 ng/g dw in the upper 70 cm and decreased to 14 ng/g dw in the 280-300 cm section. Farther down the ship channel at WC-2, concentrations ranged between 240 and 560 ng/g dw in the upper 60 cm increasing to 1,326 ng/g dw between 90 to 100 cm. Concentrations decreased to approximately 280 ng/g dw in the bottom 290-300 cm section. Unfortunately, similar sections were not analyzed between cores (Table 2).

As stated previously, total DDT is the sum of DDE, DDD, and parent DDT (both o+p forms) (Appendix II, Table 5). Concentrations of total DDT ranged from 0.65 to 687 ng/g dw for all core sections. As with many other parameters, the lowest concentrations were found in core ANC-3 at 280-300 cm and highest concentrations were observed in ANC-4 at 120-170 cm (Figure 8d). The depth distribution of total DDTs is very similar to that of total PCBs. There was a significant linear correlation between the concentrations of total DDT and PCBs for all samples ($r^2 = 0.78$, n = 38, p < 0.001) indicating similar sources and cycling.

DISCUSSION

The data obtained from this study will be used to assess the historical and recent levels of chemical contamination and determine the amount and volume of contaminated sediments in the lower tidal river. This information will be used to determine the potential cost for sediment removal/remediation as part of an preliminary sediment management strategy.

Historical Analysis of Contaminant Concentrations

The activity of ^{210}Pb (reported as disintegrations per minute per gram of sediment; dpm/g) with depth can be used to determine the sedimentation rate and historical record of contaminants in sediments (Valette-Silver, 1993). This information is necessary for the construction of sediment budgets (Schubel and Hirschberg, 1977; Brush et al., 1982; Officer et al., 1984; Huntley et al., 1995), locate areas of contaminant accumulation (Owens and Cornwell, 1995; Cornwell et al., 1996; Bopp et al., 1991; Sanders et al., 1993). Also, historical records from dated sediment cores can provide information as to the changes in loadings of contaminants over time. Such information is especially useful in tracking the effectiveness of various management actions designed to reduce inputs to specific areas (Owens and Cornwell, 1995; Christensen and Zhang, 1993; Beurskens et al., 1993). However, there are many variables that can affect the usefulness of ^{210}Pb and ^{137}Cs dating in a given area (Appleby and Oldfield, 1983; Anderson et al., 1987; Crusius and Anderson, 1991; Valette-Silver, 1993; Stephenson et al., 1996). These include mixing of the sediment by benthic organisms (i.e., burrowing organisms), physical mixing from dredging and storm events, post-depositional movement of contaminants, and additional inputs of ^{210}Pb from sources in the urban landscape (i.e., urban sediment focusing). Urban sediment focusing may be considered the transport of atmospherically-derived ^{210}Pb from urban landscape (i.e., impervious surfaces) to the river system. This ^{210}Pb could reach the water system without additional sediment thereby yielding high inventories of unsupported ^{210}Pb activity.

The ^{210}Pb profiles in sediment cores ANC-3, ANC-6, WC-1, and WC-2 exhibited an exponential decrease in ^{210}Pb activity with depth. In cores ANC-6 and WC-2 there was a surface mixed layer in the upper 50 cm below which there was an exponential decline (Figure 5). The other cores exhibited either no change with depth or were more variable. The constant initial concentration

model (CIC) was used to determine sedimentation rates and potentially assign dates to specific sediment layers. This model assumes that there is a constant input flux of both ^{210}Pb and sediment particles (i.e., an increased flux of sedimentary particles from the water column will remove proportionally more amounts of ^{210}Pb from the water to the sediments) (Robbins, 1978; Appleby and Oldfield, 1983). For this model, the supported ^{210}Pb activity was determined from the asymptote of the ^{210}Pb profile at depth. Supported ^{210}Pb activity is due to the decay of ^{226}Ra naturally present within the sediments, while unsupported ^{210}Pb is derived from atmospheric deposition (i.e., decay of gaseous ^{222}Rn). While supported or background activities in most cores (1.7 to 2.5 dpm/g; Table 3) were similar to other areas of Chesapeake Bay and Potomac estuary (Schubel and Hirschberg, 1977; Brush et al., 1982), it is not clear if the actual background ^{210}Pb levels were reached in these cores. Brush et al. (1982) used a background activity of 2.2 dpm/g, as determined from ^{226}Ra (parent of supported ^{210}Pb), for a core in the Washington Ship Channel. This is similar to the activities found in core WC-2 obtained from the same location. Cumulative mass (g/cm^2) was calculated using the water content within each interval and a sediment density of 2.4 g/cm^3 and then plotted against the unsupported activity of ^{210}Pb (unsupported activity = total activity - supported activity) to determine a sedimentation rate in g/cm^2-yr. The age at a given depth was estimated by dividing the cumulative mass by the sedimentation rate.

The mass sedimentation rates for the four cores ranged from 0.4-0.5 g/cm^2-yr at WC-2 to 1.3-1.5 g/cm^2-yr at ANC-3 ($\approx$2 to 4 cm/yr). Within the Washington Ship Channel, core (WC-2) was taken at the same location as Brush et al. (1982; Core 17 obtained in 1978) and could be a useful comparison. Unfortunately, information obtained from the Baltimore District of the Army Corps of Engineers suggests that this area was dredged as recently as 1984 and therefore could limit an accurate comparison between cores. From Brushes et al.'s (1982) analysis, the mass sedimentation rate was determined to be 0.93 g/cm^2-yr via the ^{210}Pb method and 0.52 g/cm^2-yr using pollen analysis ($\approx$0.8 to 1.6 cm/yr). Our rates for the same location are similar, but slightly lower (0.4-0.5 g/cm^2-yr). Therefore, it appears that the sedimentation rates determined for the Washington Ship Channel core obtained in this study are fairly consistent with previous evaluations. For all cores these rates were similar to those determined by Scatena (1978) for a series of cores in the lower tidal river using pollen analysis and mass balance sediment transport

calculations

The CIC model provides dates, relative to 1995, for each sediment layer (Table 6). These temporal horizons can be compared to specific chemical markers (i.e., ^{137}Cs activity and chlorinated hydrocarbons) to provide an internal check as to the accuracy of the time horizons. As stated previously, ^{137}Cs is an atomic fission product first introduced via bomb testing in the early 1950's and with peak deposition at approximately 1963. The chlorinated hydrocarbon analyzed for in this study are all industrially produced (i.e., no natural source). For example, DDT was first used starting in the late 1940's with large applications between 1950's and 1960's (Brooks, 1977; Kennish, 1992). By 1970 approximately 2×10^9 kg of the pesticide had been produced, mainly in the United States, with a total ban in the United States starting in 1972. Similarly, technical chlordane, a multi-component mixture of polychloro-methanoindenes, was developed in the 1940's and approximately 7×10^7 kg have been produced since 1946 (Brooks, 1977; Dearth and Hites, 1991). The use of chlordane was halted in 1988 after a phased reduction in use since approximately 1975. Lastly, PCBs were mainly produced by the Monsanto Corporation from 1930 to 1977 and it is estimated that 5.4×10^8 kg has been produced within the United States (Kennish, 1992). While PCB production in the United States was halted in 1977, older transformers and capacitors are still in use. Therefore, the presence of detectable amounts of ^{137}Cs and these chlorinated hydrocarbons are an indicator of a sediment horizon from the early 1950s, late 1940's and early 1930's, respectively and can be compared to those from the ^{210}Pb dating method (Table 6).

For cores WC-1 and ANC-3 this analysis is severely hindered by the sediment intervals (i.e., up to 50 cm per interval) that were obtained and analyzed from the cores. In other words, a smaller sediment interval would provide better concentration resolution, especially with the estimated sedimentation rates for the river. The average time interval associated with the 50 cm section is too large for accurate analysis. Also, not every interval was analyzed for contaminants as part of the current project (i.e., missing concentrations were interpolated using values from sections above and below). With these limitations in mind, the information in Table 6 suggest that the ^{137}Cs and contaminants are present below the depth in which the ^{210}Pb dating would predict. For example, for core ANC-3 all three contaminants (PCBs, chlordane, and DDT) are present at a

predicted time horizon of 1920. Unless there is substantial post-depositional chemical migration of all three contaminants, this is not possible given that the chemicals did not exist in the 1920s. Also, in core WC-2 detectable activities of ^{137}Cs were found at a depth corresponding to the 1930s which is not possible given that 137Cs did not exist until the mid to late 1940s.

Table 6. Dating of specific horizon from Anacostia River cores*.

Core	Depth Interval (cm)	Predicted Date	Presence of: ^{137}Cs Activity	Chlordanes	DDTs	PCBs
WC-1	70-120	1941	ND	Yes[1]	Yes[1]	Yes[1]
	120-170	1900	ND	Yes	DL[2]	Yes
	170-220	ND[3]	DL	DL	DL	Yes
WC-2	80-90	1945	Yes	Yes[1]	Yes[1]	Yes[1]
	90-100	1930	Yes	Yes	Yes	Yes
	100-125	ND	Yes	Yes	Yes	Yes
	170-220	ND	Yes	Yes	Yes	Yes
	290-300	ND	DL	DL	Yes	Yes
ANC-3	70-120	1945	ND	Yes[1]	Yes[1]	Yes[1]
	120-170	1920	ND	Yes	Yes	Yes
	280-300	ND	DL	DL	DL	DL
ANC-6	60-70	1945	ND	Yes[1]	Yes[1]	Yes[1]
	70-80	1930	ND	Yes[1]	Yes[1]	Yes[1]
	280-300	ND	DL	Yes	Yes	Yes

*These cores exhibited the best fit for ^{210}Pb data. [1]These are based on interpolated data, sections above and below contained these contaminants. [2] Concentration at the detection limit (~1-3 ng/g or < 0.03 dpm/g for ^{137}Cs). [3]ND - Not Determined

It is unclear why the ^{210}Pb model did not accurately predict the sediment deposition rate for this area. Core sections may have been contaminated during sampling, but it is difficult to conceive of how significant contamination at the high levels measured could occur for ^{210}Pb and the contaminants, given the precautions that were taken (i.e., clean utensils and containers, sampling from the center of the core). Mixing of the sediments by benthic organisms, methane emanations, or dredging would have mixed both contaminants and ^{210}Pb and produced profiles similar to those

observed for core ANC-4. It is possible that high loadings of ^{210}Pb relative to the sediment inputs (i.e., urban focusing) could have resulted in erroneous dating calculations. Also, not all sections were analyzed, and importantly the interval width of many sections were large (i.e., 10 to 50 cm) thereby limiting dating and contaminant resolution. It may be possible to use the contaminant and ^{137}Cs data, given their source function history, to help define sedimentation rates for the river. However, previous analysis in the area by Brush et al. (1981) and Scatena (1987) have yield accurate dating methods (e.g., ^{210}Pb and pollen analysis). Further sediment analysis, possibly with deeper cores, may allow the complexity of the system to be resolved.

Enrichment of Trace Metals in the Sediments of the Anacostia River

Baseline or natural concentrations of sedimentary trace metals are determined largely from inorganic material resulting from the weathering of geological deposits. This sedimentary material consists of quartz, feldspars, clay minerals and iron and manganese oxides and have trace metal concentrations dependent on the source metals present at the time of formation. When anthropogenic trace metals are introduced to a system they can be adsorbed onto fine-grain material in the water column. Inorganic and organic coatings on the fine-grain suspended material are the main phase for adsorption (e.g., Fe/Mn oxides) and, once trace metals are bound to particulate matter, the sediment-associated metals can settle and be incorporated into sediment. The metals would therefore enrich the "background" metal concentrations within the sediments.

Normalization of the sediment to a reference element not associated with anthropogenic influences is a convenient approach to determining the degree of sediment contamination (Daskalakis and O'Connor, 1995). Elements such as aluminum (Al) (Windom et al., 1989; Schropp et al., 1990), lithium (Li) (Loring et al., 1990) and iron (Fe) (Trefrey and Presely, 1976; Sinex and Helz, 1981; Helz et al., 1985; Velinsky et al., 1994a) have been used in the past. For this study Fe was chosen as a normalizing element because 1) it is the fourth most abundant metal in the earth with a crustal average of 3.5% (Wedepohl, 1971); 2) in most cases anthropogenic sources are small compared to the amount of Fe naturally present; 3) the ratios of most metals to Fe are fairly constant in the Earth's crust. The major caveat in using Fe as a normalizing element, instead of Al or Li, is that Fe undergoes important diagenetic reactions in many sediments. These

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reactions include dissolution and precipitation of Fe oxides and Fe-sulfur minerals (i.e., pyrite). However, in the previous study by Velinsky et al. (1994) and in the present study, the potential changes in speciation and mobility did not significantly affect the total concentration of sedimentary iron with depth or location (Table 4, Figure 7a). Therefore, Fe was used as normalizing agent for this study.

A useful tool in expressing the degree to which a sediment is impacted from anthropogenic sources of trace metals is the enrichment factor (EF) (Trefrey and Presely, 1976; Sinex and Helz, 1981; Helz et al., 1985; Windom et al., 1989; Velinsky et al., 1994). When normalizing to Fe, the enrichment factor is defined as: $EF = (X/Fe)_{sediment} /(X/Fe)_{unimpacted\ sediment}$, where X/Fe is the ratio of the trace metal (X) to the amount of Fe in the sample. In using the EF, a comparison to a sediment that is unimpacted by anthropogenic sources is necessary $[(X/Fe)_{unimpacted}]$ (i.e., critical in this analysis is the choice of metal to Fe ratio for "unimpacted" sediments). Enrichment factors of 1 indicate no enrichment, while EFs greater than 1 (dependent on the variability of the reference sediments) indicate anthropogenic sources of metals to the sediments. Past studies have compared sediments to the distribution of trace metals in the earth's crust (Sinex and Helz, 1981; Helz et al., 1985). While this approach is useful, it may not account for natural variations in sediment types of different geological regions. One way to account for this variability is to derive a ratio from "unimpacted" sediments in the general area of interest (Windom et al., 1989; Schropp et al., 1990).

In the present study, most samples have the potential to be impacted above natural levels. Therefore, data from samples taken in the Chesapeake Bay drainage area (including the Potomac River) were used to derived metal abundances in the general area (NOAA, 1991). Sixteen stations in Chesapeake Bay were used, some of which were sampled over multiple years. The major criterion for the selection of these stations is that they are relatively remote from areas with known anthropogenic sources of metals (i.e., Baltimore Harbor and Elizabeth River).

The ratios obtained from the regression of the NOAA (1991) data are presented in Table 7 along with data from other areas. The ratios derived from Helz et al. (1985) are from the average composition of coastal plain deposits from northern Chesapeake Bay, while the data from core 1314 (Goldberg et al., 1978) are from a location just south of the mouth of the Potomac River.

These data, along with values from average continental crust and soils, are similar in magnitude. The metal to Fe ratios for one sample from core ANC-3 (280-300 cm) are similar to those presented in Table 7. It appears that the metal concentrations in the bottom interval of ANC-3 are at background levels; the only section for the entire study. For the calculations presented below, the average metal to Fe ratio were used to calculate the EF for each station (Table 7).

The degree to which sediments in the study area are enriched in trace metals from anthropogenic sources varies from metal to metal. These variations can be due to a number of factors including 1) choice of $(X/Fe)_{unimpacted}$; 2) biogeochemistry and redox behavior of the metal; and 3) sources of metals to the study area. While these calculations use the average $(X/Fe)_{unimpacted}$, these values can vary. For example, the Cd/Fe value ranges from 0.01 to 0.09,

Table 7. Metal to Iron Ratios used for the Calculation of Enrichment Factors (EF).

Cd	Cr	Cu	Hg	Ni	Pb	Zn	Location
0.03	20.0	8.5	0.01	9.8	4.2	17.0	Continental Crust[a]
ND	18.8	9.2	ND	10.6	9.4	25.0	Soils[b]
0.01	11.8	3.6	ND	5.5	4.4	14.1	St. Mary's County Coastal Deposit[c]
0.01	24.0	2.1	ND	4.9	3.9	14.5	Anne Arundel County Coastal Deposits[c]
0.43	14.3	3.4	ND	5.3	3.3	22.1	Calvert County Coastal Deposits[c]
0.09	9.4	8.1	0.06	ND	NS	NS	Chesapeake Bay Sediments[d]
0.05	23.0	8.1	ND	12.0	NS	NS	Core 1314[e], Mouth of Potomac River
0.04	16.6	6.5	0.02	12.2	5.6	25.1	Core ANC-3; 280-300 cm
0.10	17.3	6.2	0.03	8.6	5.2	19.6	Average
0.15	5.2	2.8	0.03	3.3	2.2	5.1	± Standard Deviation (1 σ)

Values are the ratio of total metal (µg/g) to total Fe (%). [a]Wedepohl, 1971; [b]Martin and Whitfield, 1983; [c]Helz et al., 1985; [d]NOAA, 1991; [e]Goldberg et al., 1978. [f]Present Study. ND - No Data; NS - regression between metal and iron was not significant at the p<0.05 level, whereas other metals were significant at the p <0.01 level (n > 50); NC - Not Calculated.

while the Pb/Fe value ranges from 4.0 to 9.4 (Table 8). While the choice of these values affect the magnitude of the EF, the relative geographic trends should not change. In light of these factors some general trends and features are obtained from the EF data.

The EF calculation indicates that there are substantial anthropogenic inputs of many trace

metals to the sediments of the Anacostia River and Washington Ship Channel (Figure 9). The most enriched metals appear to be Hg, Pb, and Cd (Table 8, Figure 9), similar to those determined in Velinsky et al. (1994). Highest EFs for Hg and Pb were determined at either station ANC-4 or WC-1 and for Cd at station ANC-4. Levels of Zn and Cu and especially Cr and Ni do not appear to be substantially enriched in the overall study area except for sediments at station ANC-4 (Figure 9).

In many cores there was a change in the magnitude EF for some metals with depth (Figure 9; Table 8). The EF for Hg shows only slightly higher values throughout the core at stations ANC-1 to ANC-3 and ANC-5 to ANC-6. At ANC-4, WC-1 and to a lesser extent at WC-2, the EF increases with depth. At station WC-1 the Hg EF increases from 6 at the surface to over 150 at

Table 8. Summary of trace metal enrichments for Anacostia River sediment cores.

Core/Metal	Pb	Hg	Cd	Cu	Zn	Cr	Ni
ANC-1	2 - 6	2 - 14	4 - 5	2	2	1	1
ANC-2	2 - 5	3 - 5	4 - 10	2 - 3	2 - 3	1 - 2	1
ANC-3	1 - 5	1 - 11	1 - 12	1 - 3	1 - 3	1 - 3	1
ANC-4	14 - 23	8 - 33	15 - 33	4 - 6	5 - 7	5 - 7	1 - 2
ANC-5	5 - 9	4 - 6	9 - 12	2 - 3	2 - 4	2 - 4	1
ANC-6	6 - 12	4 - 9	11 - 34	2 - 4	3 - 4	3 - 4	1
WC-1	9 - 33	6 - 152	2 - 8	1 - 4	2 - 3	2 - 3	1
WC-2	3 - 7	4 - 16	5 - 8	2	2 - 3	2 - 3	1

ANC - Anacostia River
WC - Washington Ship Channel

280-300 cm, while at ANC-4 the EF increases from 12 at the surface to 33 at 280-300 cm. The downcore distribution of the Pb EF was similar for most cores taken in the Anacostia River (Figure 9). Higher values were observed at station ANC-4 (14 to 23) with no trend with depth. At station WC-1, within the upper Washington Ship Channel, the Pb EF increases with depth from 9.5 at the surface to over 30 at 170-216 cm. The Pb EF was low and did not substantially increase with depth at station WC-2 in the lower Washington Ship Channel.

Figure 9a.   Sediment Depth Profiles of Cadmium Enrichment Factors within the Anacostia River and Washington Ship Channel.

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Figure 9b. Sediment Depth Profiles of Chromium Enrichment Factors within the Anacostia River and Washington Ship Channel.

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Figure 9c. Sediment Depth Profiles of Copper Enrichment Factors within the Anacostia River and Washington Ship Channel.

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Figure 9d. Sediment Depth Profiles of Mercury Enrichment Factors within the Anacostia River and Washington Ship Channel.

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Figure 9e. Sediment Depth Profiles of Nickel Enrichment Factors within the Anacostia River and Washington Ship Channel.
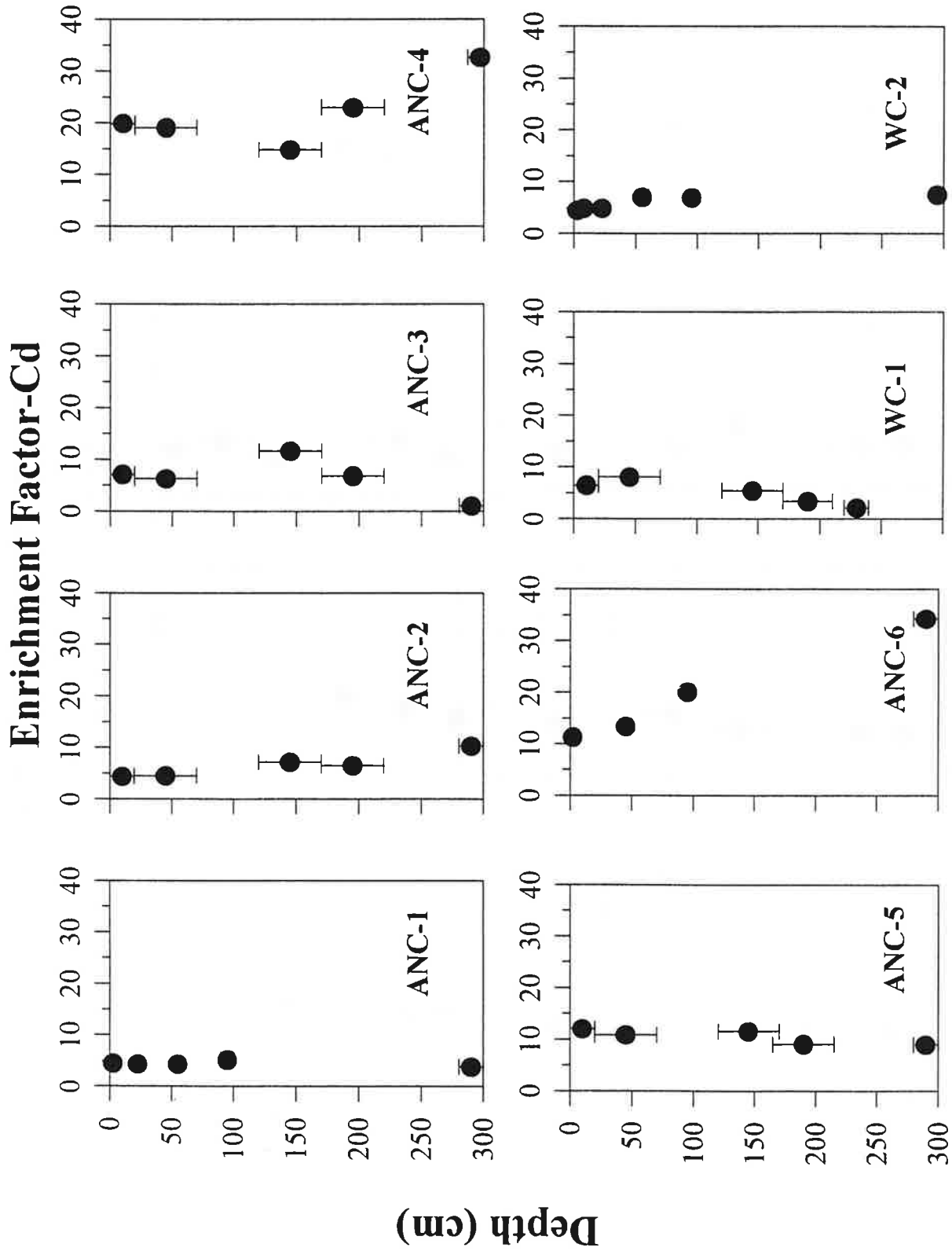
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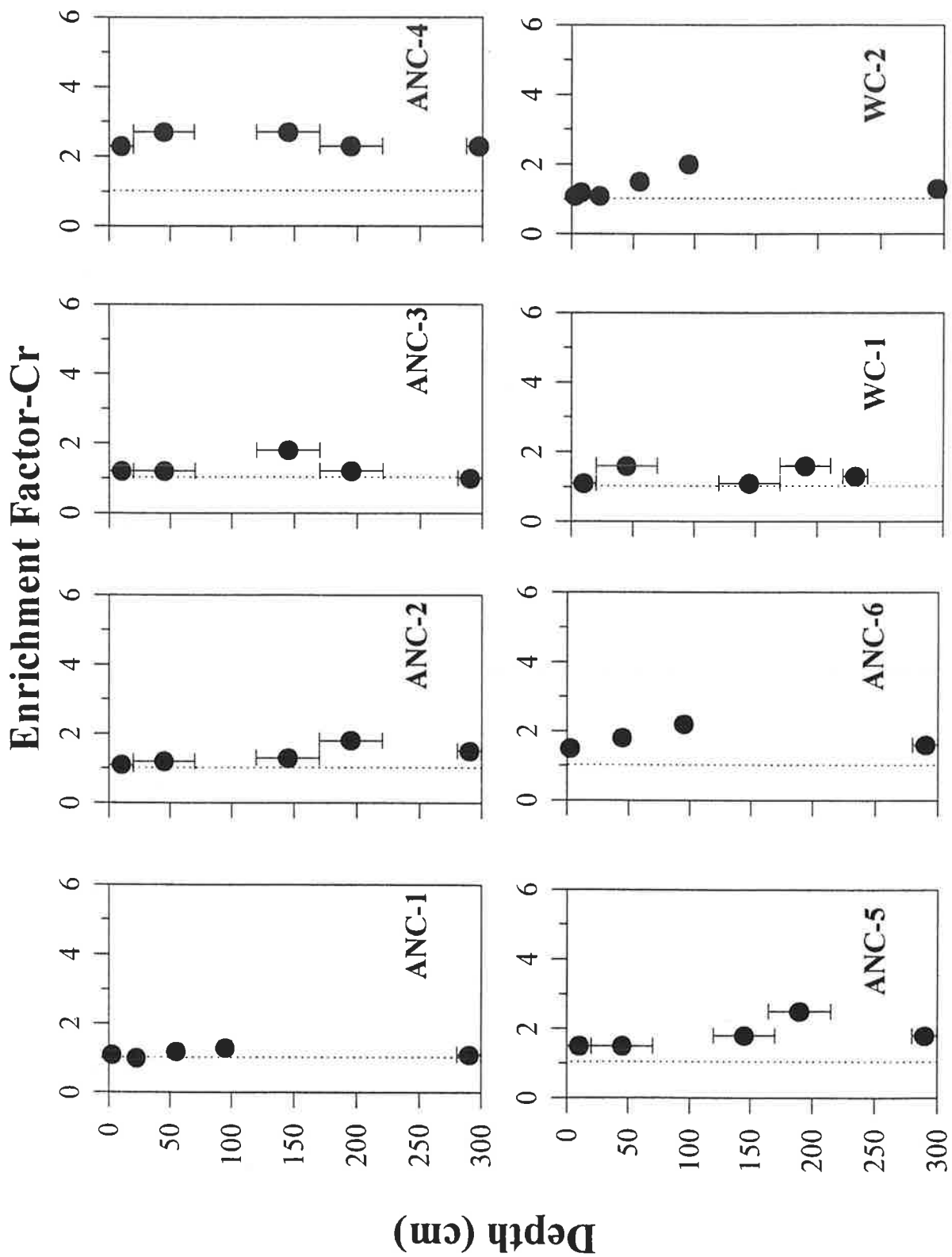Figure 9f. Sediment Depth Profiles of Lead Enrichment Factors within the Anacostia River and Washington Ship Channel.

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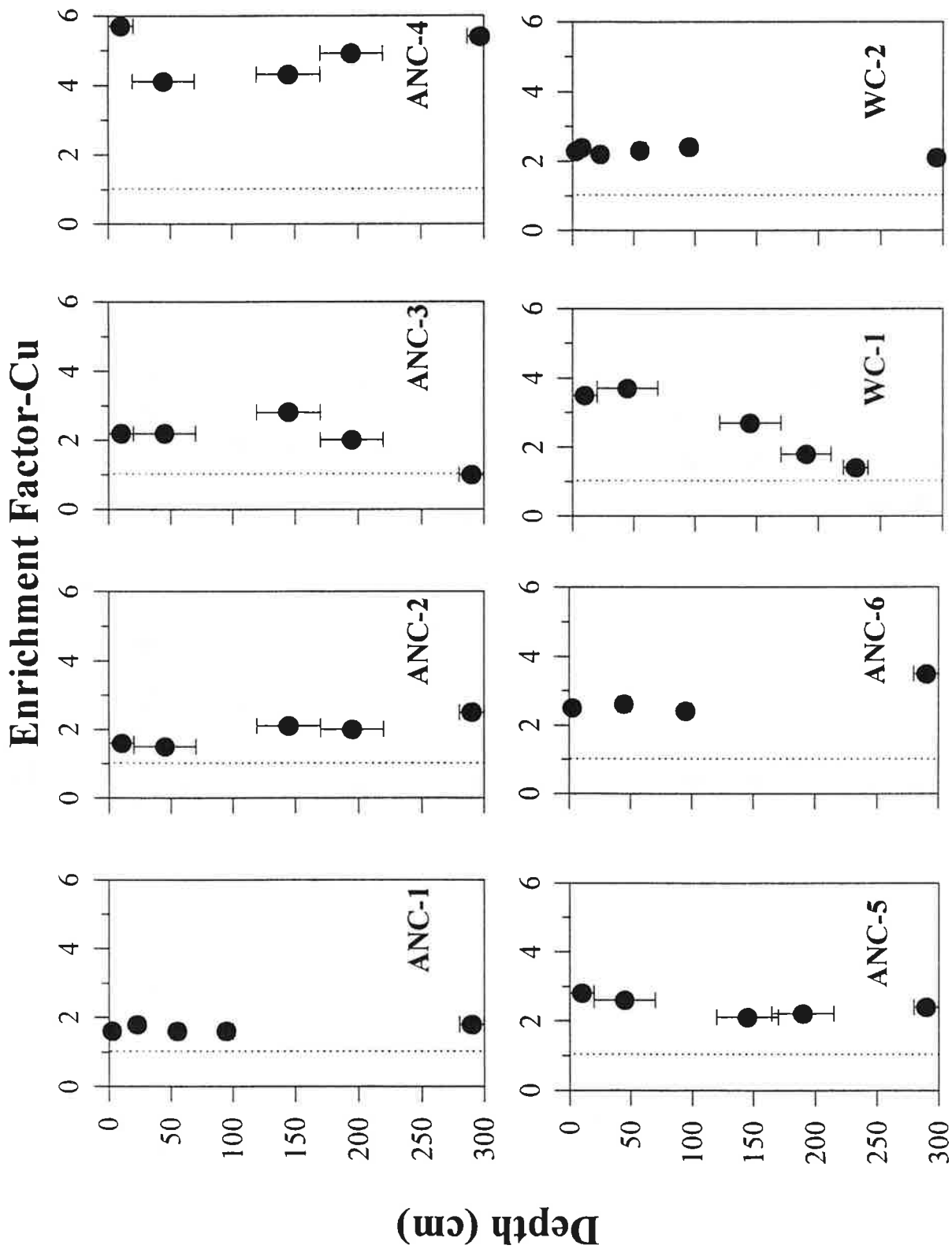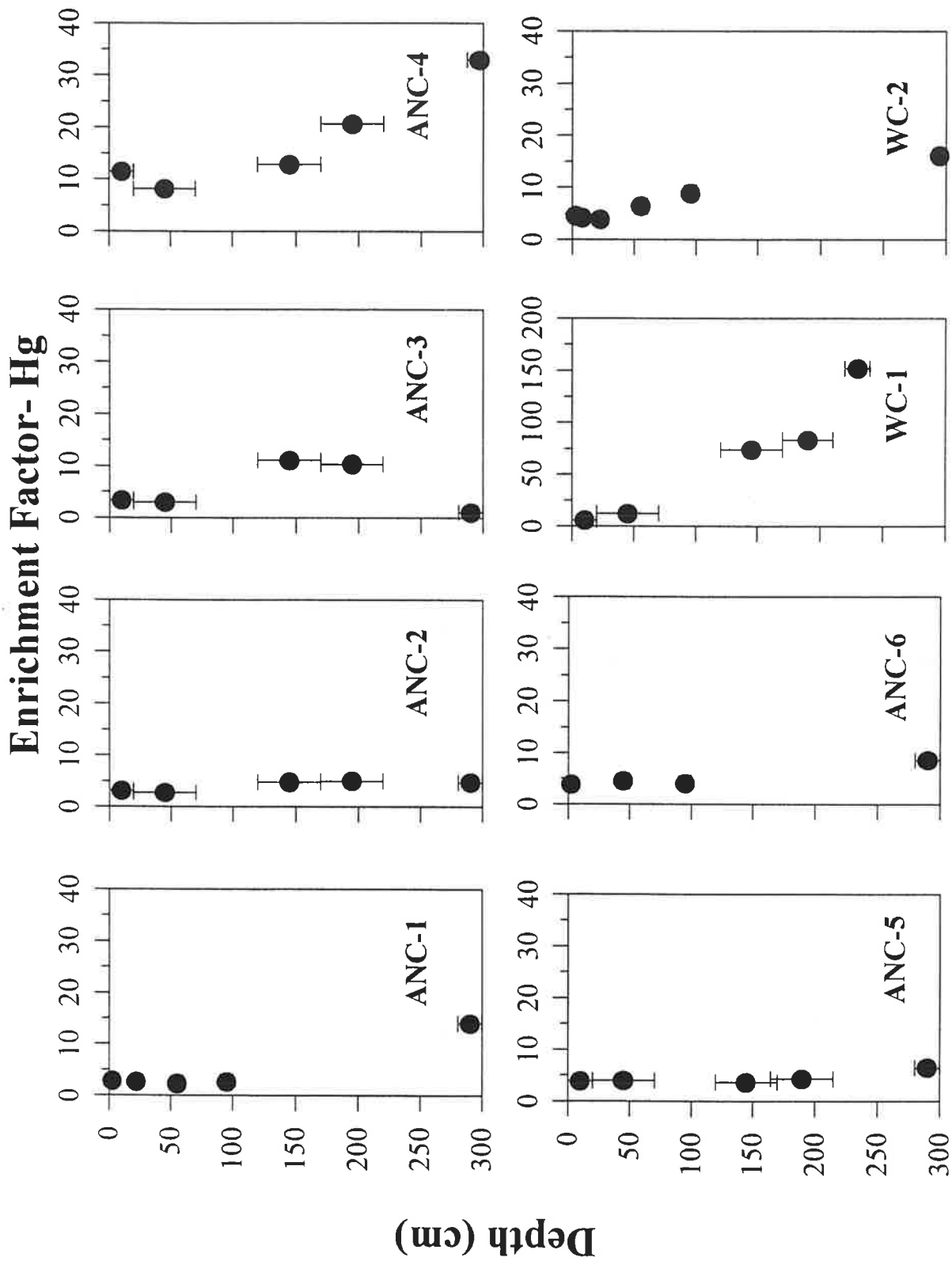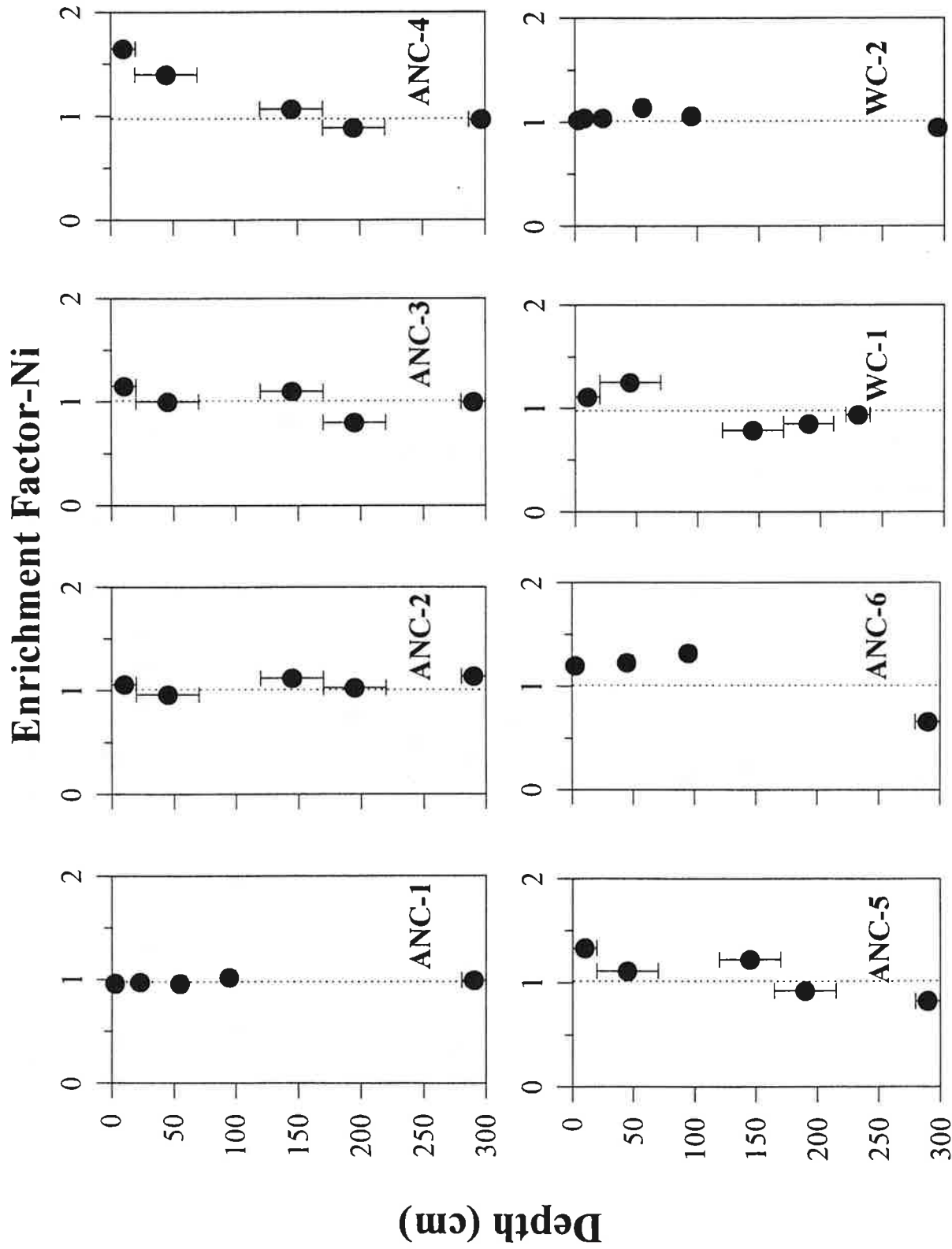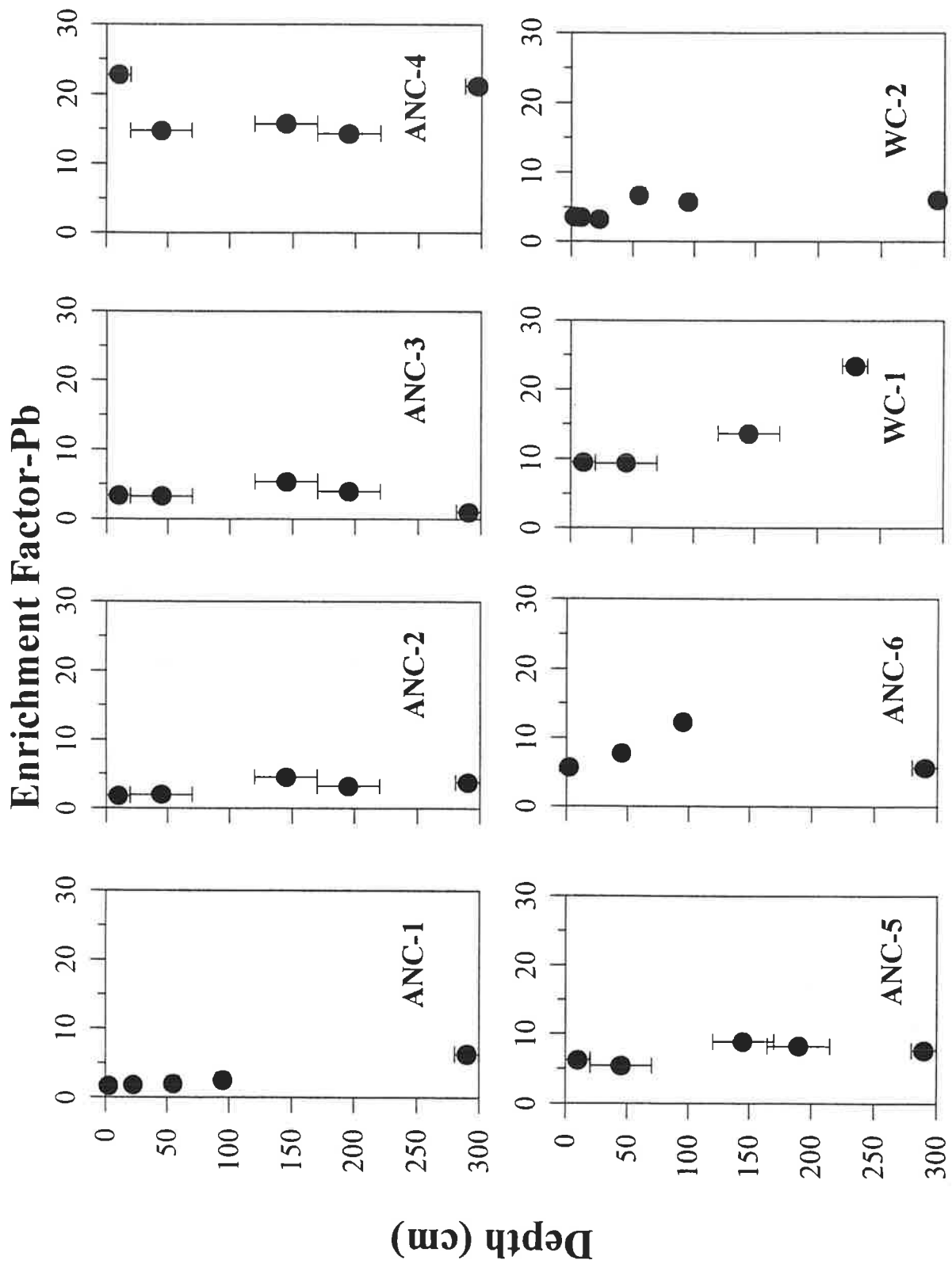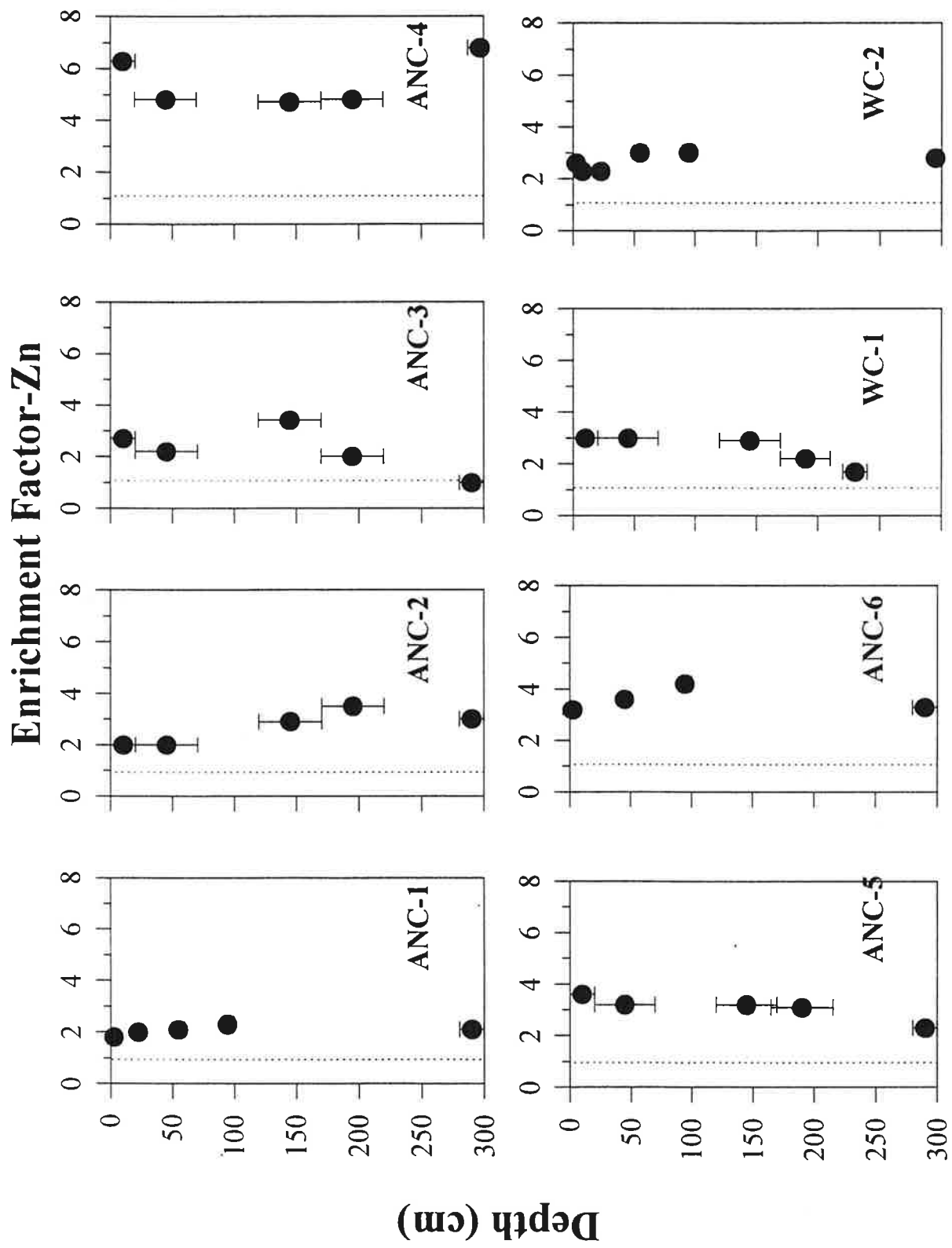Figure 9g. Sediment Depth Profiles of Zinc Enrichment Factors within the Anacostia River and Washington Ship Channel.

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The distribution of most metal EFs indicate that sediments throughout the core at ANC-4 were highly enriched (Table 8). Mercury and Pb were even further enriched in the deeper sections of the core at WC-1. These data suggest higher inputs or deposition of metals near station ANC-4 in the Anacostia River and for Hg and Pb at station WC-1. Unfortunately, sediment cores were not deep enough to allow assessment of the entire sediment layer with enriched Hg and Pb. Analysis of deeper cores would help in the determination of the entire volume and mass of contaminated or enriched sediments in the river.

Comparison of Chemical Contaminants to Indicators of Biological Response

Chemical analyses revealed relatively high levels of sediment contamination at all sites throughout most cores. A broad variety of contaminant types were identified, including trace elements (i.e., heavy metals), polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and chlorinated pesticides (i.e., chlordane and DDT). Previous studies showed effects on the sediment benthic community and laboratory toxicity (Velinsky et al., 1992; Schlekat et al., 1994). While the levels of contaminants in the sediments can not be directly related to these biological impacts, these chemicals have been shown to bioaccumulate in clams (Velinsky et al., 1992) and fish tissue collected in the Anacostia and Potomac rivers (Velinsky and Cummins, 1994; 1996). High levels of Hg, PCBs, and chlorinated pesticides are found in various fish species, with higher concentrations in bottom feeding benthic species and those with higher lipid content (e.g., American eel and channel catfish). Concentrations of specific aromatic hydrocarbons (i.e., PAHs) were also detected in many fish species; and the distribution pattern of PAHs (i.e., alkylated and substituted aromatic compounds) reflected a sediment source for bottom feeding fish (Velinsky and Cummins, 1996). Therefore, a preponderance of evidence suggests that the levels found in the river sediments can have an ecosystem and possible human health effect.

For specific sediment management actions, however, it is important to determine at what concentration(s) are these biological effects observed. The identification of sediment contaminant levels associated with adverse biological effects provides a means of grossly identifying potential causative factors in sediment-effects assessment and can assist in sediment management decisions.

There are numerous approaches to developing sediment quality guidelines for assessment of sediment contamination (Marcus, 1991; Eskin et al., 1996 for a review). These include spiked-sediment bioassays, equilibrium partitioning models, sediment chemical-biological triad, apparent effect threshold (AET), screening level concentrations (SLC) and others. The guidelines or criteria (i.e., concentrations) produced from these methods can be used as consistent benchmarks for evaluating sediment quality for the protection of aquatic life and, importantly, represent one management tool for protecting marine and freshwater ecosystems.

The guidelines used for this study are those derived using the National Status and Trends Program (NSTP) approach for freshwater sediments by Environment Canada (CCME, 1995; Smith et al., 1996). This approach uses a weight of evidence of the available toxicological information generated from various field studies that associate sediment concentrations with specific biological responses (Long and Morgan, 1990; Long et al., 1995). Guidelines are statistically developed from a large database of sediments in which chemical mixtures occur, and establishes associations, *not cause and effect relationships*, between chemical concentrations and adverse biological responses, including acute and chronic toxicity or changes in benthic community structure.

For the Canadian guidelines (CCME, 1995) the original procedure used by Long and Morgan (1990) was modified to calculate two assessment values using both an effects and no effects database. The lower value, the threshold effect level (TEL), represents the concentration below which adverse effects are expected to occur rarely. For each chemical, the TEL was calculated as the geometric mean of the lower 15th percentile concentration of the effects data set and the 50th percentile concentration of the no-effect data set. The upper value, the probable effect level (PEL), defined the level above which adverse effects are expected to occur frequently. It was calculated as the geometric mean of the 50th percentile of the effect data set and the 85th percentile concentration of the no effect data set. Concentrations between the TEL and PEL are expected to occasionally be associated with adverse biological effects. These guideline are considered tentative until further studies are conducted (i.e., spiked sediment toxicity tests) to complement the current analysis. Also, the freshwater guidelines are not as statistically strong as those developed using a larger data set for marine sediments. In this regard, it must be pointed

out that these type of guidelines (i.e., concentrations derived using the NSTP approach) remain controversial in their use due to many factors including ecosystem relevance, lack of specific cause and effect relationships regarding biological interactions (Lee and Jones-Lee, 1993; Sampson et al., 1996; MacDonald et al., 1996) and site-specific characteristics that modify sediment contaminant-aquatic organism interactions. However, these guidelines can be used as interpretive tools, along with other tests (toxicity tests, benthic community assessments, TIE methods, and others) as guides to assessing sediment quality (MacDonald et al., 1996).

Comparing the published interim TEL and PEL concentrations against the concentrations determined in this study reveals multiple exceedences within each core (Table 9, Table 10a,b). In most core sections (except for the very bottom sections of a few cores), most chemicals were between the TEL and PEL, indicating concentrations that are expected to occasionally be associated with adverse biological effects. Chemicals that were mostly above the PEL, where frequent effects are expected, included Cr, Ni, Hg, Pb, chlordane, and total PCBs. Total PAHs were above the PEL in only two instances in core ANC-4, while total DDTs were never above the PEL concentration. It is interesting to note that concentrations of Cr and Ni (and sometimes Zn) were above the TEL and PEL in many cases but were not substantially enriched overall (see previous section). Also, in many instances concentrations of Cd were not above the PEL yet it was one of the trace metals most substantially enriched (Table 10). These observations indicate the importance of obtaining site specific background conditions in performing any assessment of sediment contamination and possible biological impact.

For some chemicals there are proposed guidelines that take into account specific controlling factors in the bioavailability of sediment contaminants. The amount of sulfide (both dissolved and sediment bound) can reduce the bioavailability and uptake of many divalent metals (see review by Ankley et al., 1997). For many nonpolar organic contaminants the amount (and type) of organic carbon present in the sediments has been shown to have a controlling influence on porewater concentrations and biological availability (Di Toro et al., 1991). The establishment of sediment quality criteria for non-ionic organic compounds is currently under consideration by the U.S.

Table 9. Effects related concentrations for sediment contaminants*.

Chemical Analyte	TEL	PEL
Trace Metals (μg/g dw)		
Cadmium (Cd)	0.59	3.53
Chromium (Cr)	37.3	90.0
Copper (Cu)	35.7	197.0
Lead (Pb)	35.0	91.3
Mercury (Hg)	0.17	0.48
Nickel (Ni)	18.0	35.9
Zinc (Zn)	123.	315.
Chlorinated Hydrocarbons (ng/g dw)		
p,p'-DDD	3.54	8.51
p,p'-DDE	1.42	6.75
Total DDTs	6.98	4450.
Chlordane	4.50	8.9
Total Polychlorinated Biphenyls (PCBs)	34.1	277.
Aromatic Hydrocarbons (ng/g dw)		
Benzo[a]anthracence (1)	31.7	385.
Benzo[a]pyrene (2)	31.9	782.
Chrysene (3)	57.1	862.
Flouranthene (4)	111.	2,355.
Phenanthrene (5)	41.9	515.
Pyrene (6)	53.0	875.
Total Polycyclic Aromatic Hydrocarbons (PAHs)	4,020.	44,790.

[1]Freshwater interim concentrations from Smith et al. (1996) except total PAHs which
are ER-L and ER-M taken from Long et al. (1995). See text for details.

Table 10a. Contaminants that exceeded either the TEL or PEL for entire data set.

Sta ID/Depth (cm)	TEL Exceedences	PEL Exceedences
ANC 1: 0-5	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Ni, Chlordane
ANC 1: 20-25	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Ni, Chlordane
ANC 1: 50-60	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PCBs	Cr, Ni, Chlordane
ANC 1: 90-100	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PCBs	Cr, Ni, Chlordane
ANC 1: 280-300	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total PAHs	Cr, Ni, Hg
ANC 2: 0-20	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PCBs	Ni, Chlordane
ANC 2: 20-70	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PCBs	Cr, Ni, Chlordane
ANC 2: 120-170	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 2: 170-220	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 2: 280-300	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 3: 0-20	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDTs, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane
ANC 3: 20-70	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Chlordane
ANC 3: 120-170	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Hg, Zn, Chlordane, Total PCBs
ANC 3: 170-220	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Total PAHs, Total PCBs	Cr, Ni, Hg, Pb
ANC 3: 280-300	Cr, Ni	Ni
ANC 4: 0-20	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PAHs, Total PCBs
ANC 4: 20-70	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 4: 120-170	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 4: 220-270	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 4: 295-300	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cd, Cr, Hg, Ni, Pb, Zn, Chlordane, Total PAHs, Total PCBs

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Table 10b. Contaminants that exceeded either the TEL or PEL for entire data set.

Sta ID/Depth (cm)	TEL Exceedences	PEL Exceedences
ANC 5: 0-20	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 5: 20-70	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 5: 120-170	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 5: 170-220	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 5: 280-300	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Chlordane, Total PCBs
ANC 6: 0-5	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 6: 40-50	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 6: 90-100	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
ANC 6: 280-300	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cd, Cr, Hg, Ni, Pb, Zn, Chlordane
WC 1: 0-20	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Zn, Chlordane, Total PCBs
WC 1: 20-70	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Zn, Chlordane, Total PCBs
WC 1: 120-170	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Chlordane, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Zn, Chlordane
WC 1: 170-216	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb
WC 1: 280-300	Cr, Cu, Hg, Ni, Pb, Zn	Cr, Hg, Ni, Pb
WC 2: 0-5	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane, Total PCBs
WC 2: 5-10	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Chlordane
WC 2: 20-25	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Ni, Pb, Zn, Chlordane
WC 2: 50-60	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Zn, Chlordane, Total PCBs
WC 2: 90-100	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Chlordane, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Zn, Chlordane, Total PCBs
WC 2: 290-300	Cd, Cr, Cu, Hg, Ni, Pb, Zn, Total DDT, Total PAHs, Total PCBs	Cr, Hg, Ni, Pb, Zn, Total PCBs

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Environmental Protection Agency. The basis for these criteria lies in the observation that biological effects of non-ionic organic compounds in sediment can be correlated to the pore water concentration of the compound, and, further, that the pore water concentration of these compounds is governed by the partitioning between sediment organic carbon and the pore water phase of a sediment (Di Toro et al., 1991; U.S. EPA, 1994). Di Toro et al. (1991) suggest that sediment quality criteria (SQC) be established using the final chronic value (FCV) from water quality criteria (WQC) normalized by a coefficient that approximates the partitioning between organic carbon and pore water, which yields the organic carbon-normalized SQC concentration, or SQC_{OC}. The partitioning of a chemical between sediment and pore water is calculated using two factors, the hydrophobicity of a chemical and the sorption capacity of a sediment as indicated by sediment organic carbon.

For the present study, the hydrophobicity of five organic compounds, DDT, chlordane, endrin, dieldrin, flouranthene, acenaphthene, phenanthrene, and PCBs, were estimated using the most recent octanol/water partitioning coefficients available for each compound (Table 11). SQC_{OC} were then calculated using the FCV available for each chemical (Table 11). The SQC_{OC}s that were calculated for endrin, dieldrin, flouranthene, acenaphthene and phenanthrene are those estimated by the U.S. EPA, while those for DDT, PCBs, and chlordane were calculated from K_{oc}s and FCV obtained from various studies. The criteria were also calculated with uncertainty limits assuming log-normal distribution at the 95% level (Table 11).

Organic carbon-normalized concentrations of (as totals) chlordane, DDT, and PCBs were greater than SQC_{OC} for most depth intervals at all sites (Table 11). No exceedences were observed for dieldrin, endrin, acenaphthene, flouranthene, or phenanthrene within the 95% confidence interval. Interpretation of criteria exceedances by sediment chemical concentrations in the present study must be interpreted with caution because the U.S. EPA is currently reevaluating K_{OC}s and FCVs for PCB, DDT, and chlordane. It is likely that criteria issued by U.S. EPA for these compounds will differ from those estimated for the present study. Despite this, PCBs and chlordane (normalized to organic carbon) uniformly exceed their respective SQC_{OC}s indicating that these organic contaminants are potentially bioavailable to benthic organisms and up the food chain. In the previous study by Velinsky et al. (1992), these chemicals exhibited a significant

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Table 11. Sediment quality criteria derivation and number of exceedances for select non-ionic organic compounds.

Chemical	FCV (ug/L)	Log Koc	Koc	Lower 95% ug/g OC	Criteria ug/g OC	Upper 95% ug/g OC	Number of Exceedences
Acenaphthene[1]	23.000	3.77	5,823	62.4	133.9	287.7	0
Dieldrin[1]	0.063	5.25	177,623	5.2	11.1	23.8	0
Endrin[1]	0.061	4.84	68,650	1.9	4.2	9.0	0
Fluoranthene[1]	5.090	5.00	100,867	239.1	513.4	1102.7	0
Phenanthrene[1]	6.320	4.46	29,047	85.5	183.6	394.3	0
TChlordane[2]	0.004	3.57	3,715	0.01	0.02	0.03	39
TDDT[2]	0.001	5.15	141,254	0.07	0.14	0.30	35
TPCBs L Koc[2]	0.014	4.30	19,953	0.13	0.28	0.60	39
TPCBs H Koc[2]	0.014	5.08	120,226	0.78	1.68	3.61	39

[1] K_{oc}s and freshwater chronic values from US EPA (1993c-g). [2] K_{oc}s estimated from K_{ow}s of Kadez et al (1986) and freshwater chronic values as reported in the 1980 Federal Registrar.

relationship with sediment toxicity, strongly supporting the hypothesis that these compounds contribute to sediment toxicity.

Inventories of Chemical Contaminants in the Lower Anacostia River

To illustrate the capacity of the lower river to accumulate and retain contaminants, a sediment inventory of both trace metals and organic contaminants was determined. The inevitable assumptions involved in this type of calculation can produce results that either over or underestimate the actual inventory. These assumptions include the use of a limited number of cores over the area (8 for the entire area), limited core depth, and interpolation between sediment layers that were not analyzed. Also, in the past material from the river has been removed by dredging (1940's and 1984; U.S. ACE, 1980; RG&H, 1984; Garbarino, personal communication) and this material is not included in this inventory. However, this calculation can provide a first-order estimate of the mass of material that has accumulated in the lower river over the past century or so as indicated by the sediment accumulation rates (see previous section).

The lower river (surface area = 2.7×10^9 cm^2) was broken into eight cells relative to the location of each core and areas of known contamination (area data provided by MWCOG; Table 12). For each sediment interval sampled, the concentration of a contaminant was converted to an areal mass (g /cm^2) using the porosity (as derived from the percent water), a dry sediment density of 2.4 g/cm^3, and the thickness of the specific interval. Intervals that were not analyzed were linearly interpolated from sections that were analyzed above and below. Next, each areal mass was summed over the entire length of the core and multiplied by the surface area for that specific cell;

$$A\sum C_x \rho_s (1-\phi) I = \text{Total Inventory (kg)} \qquad (1),$$

where A is the surface area of each cell, C_x is the concentration of chemical x within the sediment interval, ρ_s is the dry sediment density, ϕ is the porosity within the sediment interval, and I is the thickness of the sediment interval. The chemical mass for each cell of the river was combined to determine the total inventory for the lower river from the Sousa Bridge to Hains Point including

the Washington Ship Channel.

As discussed above, a substantial portion of the trace metals that have accumulated in the Anacostia River sediments are derived from anthropogenic sources. To determine the mass of anthropogenic metals, the concentration of each metal in each interval was corrected for the amount of metals expected to be present from natural sources using the background metal to iron

Table 12. River sections based on core sample locations.

Station	Description	Areas (cm^2)
ANC 1	Hains Point to Eastern Shore of Washington Channel	4.39×10^9
ANC 2	Eastern Shore of Washington Channel to Buzzard Point	2.92×10^9
ANC 3	Buzzard Point to South Capitol Street Bridge	3.24×10^9
ANC 4	South Capitol Street Bridge to 11th Street Bridge	4.85×10^9
ANC 5	11th Street Bridge to Bottom of 15th Street, SE	1.64×10^9
ANC 6	Bottom of 15th Street, SE to Sousa Bridge	1.32×10^9
ANC 7	Sousa Bridge to Railroad Bridge	1.41×10^9
WC 2	Hains Point to Base of 7th Street, SW	6.50×10^9
WC 1	Base of 7th Street, SW to Tidal Basin	1.60×10^9

ANC = Anacostia Station, WC = Washington Channel Station. Area data provided by MWCOG.

ratios (see above). The mass of anthropogenic metals were then calculated for the entire lower Anacostia River using the equation described above.

The total inventory of the various contaminants and the amount of anthropogenic material is presented in Table 13. For organic contaminants such as total chlordane, DDTs, and PCBs, which are all related to human activities, the entire mass is anthropogenic and ranges from 0.23 metric tons (MT) for chlordane to 4.7 MT for total PCBs (1 MT = 1000 kg). For total PAHs, the amount of material is substantial (79 MT) and mostly anthropogenic (i.e., there are some naturally occurring aromatic hydrocarbons).

There also is a substantial amount of trace metals within the sediments, from 6 MT for Hg to 2,280 MT for Zn. The mass of Zn in the Anacostia River is approximately 10X lower than that derived by Sinex and Helz (1982) for the Patapsco River (i.e., Baltimore Harbor), but the

\Anacostia River is over 30X smaller than the Patapsco River (117 km^2 versus 0.27 km^2). It is unclear why the Anacostia River contains large amounts of Zn given that it is not an industrialized area compared to the Patapsco River. The amount of anthropogenic metal varies as the enrichment factors would suggest, with Cd, Hg, and Pb showing the greatest enrichments. Cadmium, Hg, Pb, and to a lesser extent Zn and Cu are derived from anthropogenic inputs. The anthropogenic influence is intermediate for Cr and slight for Ni.

Table 13. Sediment inventories for contaminants in the lower tidal Anacostia River*.

Contaminant	∑Chlor	∑DDTs	∑PCBs	∑PAHs	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Total Inventory	0.23	0.76	4.7	79	11.3	806	493	5.9	367	1,300	2,280
Anthropogenic	0.23	0.76	4.7	79	10.3	314	301	5.4	6.9	1,140	1,540
% Excess	100	100	100	~100	91	39	58	89	2	87	67

Units: metric tons (1000kg = 1 metric ton); Z = 300 cm, total surface area = 2.7 X 10^9 cm^2, total sediment volume = 8.1 X 10^{11}cm^3

Sinex and Helz (1982) also calculated an anthropogenic inventory for Zn in the Patapsco River and the amount is slightly greater than that in the Anacostia River (75% versus 67%). This, albeit slightly higher amount, reflects the greater industrialization and related sources within the Patapsco River watershed compared to the Anacostia. However, the amounts of trace metals and organic contaminants found in the sediments of the Anacostia River appear high in relation to the lack of present and past major industrial sources of these contaminants. A complete historical inventory and source identification with land use changes is needed to further evaluate the causes of the high levels of anthropogenic material.

Summary

This examination of the core data indicates substantial enrichment of many trace metals and organic contaminants over background concentrations. Also, the number of samples with contaminant concentrations exceeding guidelines associated with adverse biological effects indicate that the sediment bound contaminants within the Anacostia River and Washington Ship Channel pose a possible risk to aquatic organisms and human health via food chain

bioaccumulation. In fact sediment toxicity and substantial levels of contaminants in fish tissue has been shown in previous studies (Velinsky et al., 1992; Velinsky and Cummins, 1994; 1996; Schlekat et al., 1994). In only the bottom section (280-300 cm) of core ANC-3 did the concentrations of many metals and all organic contaminants fall below background concentrations. Therefore, any sediment management action must deal with the multiple potential contaminants throughout the river and at depths of up to and possibly exceeding 3 m.

The sub-bottom profiling from one transect indicates another possible sediment layer in the lower river at approximately 4 to 5 m below the sediment surface. Unfortunately this layer was not sampled during this study, therefore it is not possible to determine the full extent of contamination deeper in the sediments. The sub-bottom profiling at transect 8 (Figure 3) suggests that it is possible that core samples at 3 m were taken just above a cleaner, less contaminated sediment. This is also supported by the low concentration of trace metals and organic contaminants in the bottom section of core ANC-3. This core was taken off the channel closer to the northwest side of the Anacostia River where shoaling of the sediment layer may occur. This is also suggested in the sub-bottom results from transect 8 (Figure 3). Therefore, contaminated sediments may be only slightly deeper than 3 m in the river. A more concise determination of the depth of the contaminated sediment layer can be obtained only if deeper cores are taken within the system.

In the following section, the core information presented above, along with spatial variations in surface sediment contamination, were used to develop the preliminary scenarios for the remediation of the sediments in the tidal Anacostia River and Washington Ship Channel.

Contaminated Sediment Management Action

Remediation or clean-up of contaminated sediments in the river may be needed to decrease and eventually eliminate harmful direct and indirect exposure of chemical contaminants to aquatic organisms including fish. Clean up of contaminated sediments is a complex and costly undertaking and, therefore, careful consideration of a number of factors is required before deciding on a remediation plan. This section provides some general information about the remediation of contaminated sediments summarized from several sources (U.S. EPA, 1995; U.S. EPA, 1994a-b, d-e; U.S. EPA, 1993a,b), with additional comments on specifics in the Anacostia River (DCRA, 1996).

Considerations for Remediation of Contaminated Sediments

Source Identification and Control

Identification and control of the sources of the contaminants, either point or diffuse, from the watershed to the tidal river is a important part of any remediation plan. For example, Velinsky et al. (1993b) quantitatively determined that urban runoff is a major source of trace metals to river sediments. Studies that determine the flux of contaminants from upstream sources are currently nearing completion and should provide additional information concerning the magnitude of upper watershed sources. Once the sources are identified and quantified, steps must be taken to control them. Control methods include point source reduction (if appropriate), storm water runoff controls, combined sewer diversions, and stopping illegal dumping/discharges. Source controls will most likely be very costly but are necessary components of any sediment remediation action. If the sources of the contaminants are not controlled to the greatest extent possible, remediation efforts will provide only a temporary solution to the problem. Any plan to reduce the load of multiple contaminants from an area must be a balance of chemical specific versus source specific control strategies.

Sediment Characterization

To determine the best way to approach the remediation of contaminated sediments, the chemical and physical properties of the sediments in question must be characterized. In addition,

the environmental conditions that influence sediment behavior at the site (e.g., current, tides, boat traffic) and the potential for spreading contaminants to other areas during remediation must be assessed. The discussion within this report and those within the studies by Velinsky et al. (1992; 1993; 1994), Wade et al. (1994), Scheklat et al. (1994), and others help characterize the chemical composition of the sediments in the lower Anacostia River and Washington Ship Channel. Additional studies such as modeling of sediment and chemical transport will add to the understanding of sediment properties and should be a priority for future research.

Risk Characterization and Remediation Feasibility

Characterization of the risk from contaminated sediments is necessary to select what benefits will be sought as an end point to any remediation actions. An important tool in this evaluation is the use of a risk-based framework and model (U.S. EPA, 1992). These models help focus management, research, and financial information and needs into a sound framework for dealing with contaminated sediments.

Once additional information is gathered on chemical contaminant sources and effects, sediment transport dynamics, and the benefits of remediation, it will be necessary to conduct a detailed feasibility study on sediment remediation alternatives. The final selection of the remediation actions will be based on the characterization described above, the amount of risk to human and ecological health posed by the contamination, and the available funds for remediation activities. Funding for remediation will depends on the resources of local governments and on the assistance of the Federal Government.

Selecting Remediation Techniques

As discussed previously, there are a number of techniques that have been developed for the remediation of contaminated sediments. The sediments can be removed by traditional dredging techniques, although safe disposal of the dredged materials can be difficult and expensive. Containment, natural burial (no action), and active burial (capping) all seek to effectively eliminate contact between chemical contaminants and either the overlying water or living organisms. *In situ* treatment options have been used in some areas (e.g., New York/New Jersey Harbor area),

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depending on site and chemical specific variables. These methods include biological treatment, dechlorination, soil washing, solvent extraction, solidification/ stabilization, incineration, thermal desorption, and microbial degradation (U.S. EPA, 1993). These methods can be used on in-place sediments or after dredging with replacement back into the river system. Marcus (1991) noted that the level of uncertainty associated with each of treatment methods can easily slow down any management action plan.

Sediment remediation creates many formidable challenges including: environmental side effects upon removing, treating, and replacing large volumes of sediments; choosing specific clean up goals and actions in light of scientific and environmental uncertainty; and importantly, generally high costs for a limited amount of clear benefits over the short and long term. Many of these issues are beyond the scope of this study but, needless to say, these issues must be addressed within a risk-based framework in order to determine what action is needed to obtain specific clean-up goals.

Preliminary Sediment Remediation Scenarios: Removal

This section presents some preliminary scenarios for removal of sediments in the lower Anacostia River using traditional dredging techniques, followed in some cases by capping. It must be kept in mind that dredging is not necessarily the only or best method for dealing with contaminated sediments in the Anacostia River. The scenarios may not be technically or financially feasible and other more realistic remediation options may exist for the lower Anacostia River. Additionally, removal or other remediation actions may be necessary upstream of area where core samples were the core samples were collected in this study. The scenarios do, however, provide general estimates of the range of costs associated with sediment remediation.

Estimating Sediment Remediation

For the purposes of this analysis, whether sediments were classified as in need of remediation was determined by comparing the sediment core data for several chemical contaminants of particular concern in the Anacostia River (Pb, Ni, Cr, Cu, Hg, Cd, Zn, total PCBs, total DDTs, total chlordane, and total PAHs). These data were compared to the PELs (discussed above,

Table 10) as indicators for potential biological effects. The river and channel were divided into sections based on the core sampling locations (Table 12). It was assumed that the conditions at the sampling location were representative of those in the entire section, although significant variability in contaminant concentrations within each division may exist.

Table 14. Summary of contamination in sediment core samples.

Station[1]	Depth of "Contaminated" Sediments (cm)	Number of Parameters above PELs[2]	Relative Contamination Levels
WC-1	< 200	4 to 7	Middle-Higher
WC-2	300	4 to 6	Lower
ANC-1	< 300	3 to 4	Lower
ANC-2	300	2 to 5	Lower
ANC-3	< 200	1 to 8	Lower
ANC-4	300	6 to 9	Higher
ANC-5	300	6	Middle
ANC-6	300	6 to 7	Middle

[1]ANC = Anacostia Station, WC = Washington Channel Station;
[2]Range is for all sections of each core, total number of parameters evaluated is 11 (Table 9).

Sediment was considered in need of remediation if levels of any chemical contaminant exceeded its PEL. If any PEL was exceeded at the lowest depth for which data was available, that depth was considered to be the deepest extent of contamination needing remediation, even though contaminant levels may still have exceeded the PELs at greater depths. Tables 8 and 10 details the comparison of the available data to the guidelines and metal enrichments.

Prioritizing Hot Spots

To determine which areas should receive priority for remediation, the relative magnitude of the contamination in the sections of the river was assessed. Table 14 reports the ranges for the number of chemicals exceeding their PELs in the sections of a core. Based on these results, the sections of the river were assigned ratings from Higher to Lower. Although these ratings reflect the relative magnitude of the contamination, it is important to keep in mind that sediments in *all*

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of the cores exhibit concentrations of one or more contaminants that exceed the PELs and, therefore, may all be considered in need of remediation.

<u>Preliminary Removal/Capping Scenarios</u>

The scenarios presented below cover a range of possible dredging and capping options from comprehensive removal to a more limited capping project in the most highly contaminated areas. While oversimplified, these scenarios are examples of remediation actions that could be taken to address sediment problems in the lower Anacostia River. The comprehensive removal scenarios demonstrate the great magnitude of the problem, while the more limited scenarios provide an idea of how the larger problem may be approached in prioritized phases.

Scenario 1: Comprehensive Removal

Dredge the entire Washington Ship Channel and lower Anacostia River from Hains Point to the Sousa Bridge to a depth of 300 cm (ANC-1 through ANC-6; WC-1 and 2).

Scenario 2: Comprehensive Removal/Capping Combination

Dredge the entire Washington Ship Channel and lower Anacostia River from Hains Point to the Sousa Bridge to a depth of 100 cm (ANC-1 through ANC-6; WC-1 and 2), and place 60 cm of clean capping material in dredged area.

Scenario 3: Removal from Highly Contaminated Areas

Dredge to a depth of 300 cm from areas between the 11th Street Bridge and the South Capitol Street Bridge (ANC-4) and from the upper Washington Ship Channel (WC-1).

Scenario 4: Removal/Capping in Highly Contaminated Areas

Dredge to a depth of 100 cm from areas between the 11th Street Bridge and the South Capitol Street Bridge (ANC-4) and from the upper Washington Ship Channel (WC-1), and place 60 cm of clean capping material in dredged area.

Cost Estimate Calculations

Unit Costs

Rough unit cost estimates were calculated for each of the above scenarios based upon published costs for similar operations in the Anacostia River (U.S. ACE, 1993) and in the Great Lakes region (U.S. EPA, 1994d). A range of possible costs was calculated based on low and high cost estimates for various activities, as shown in Table 15.

The higher cost estimate for operation of the dredge allows for the added expense of using sediment barriers, resuspension controls, and of operating the dredge precisely, all of which will reduce unwanted transport of sediment contaminants away from the dredging area. High transport cost estimates are via truck or rail, while the corresponding low estimates are for barge transport. For disposal, the high cost estimates include locating and building a new confined disposal facility. Costs for fill materials and capping activities are usually fairly constant, except when volumes are large.

Table 15. Unit cost estimates for various remediation activities.

Activity	Low Cost Estimate ($ per cu yd)	High Cost Estimate ($ per cu yd)
Hydraulic Dredge Operation	5	15
Transport	2	13
Disposal	5	50
Capping/Fill	5	5

Calculations and Assumptions

A number of assumptions were made to simplify the cost estimate calculations. Although there are several different types of dredges available, use of a hydraulic dredge and pipeline transport system was assumed in all calculations. The U.S. Army Corps of Engineers (U.S. ACE) has determined that this method is the most cost-effective for navigational dredging projects in the

Anacostia River. The costs include pipeline transport to disposal facilities within about 3 km (2 mi) of the dredging site and not exceeding 6 m (~20 ft) of vertical rise (U.S. EPA, 1994a,d,e; U.S. ACE, 1993).

Disposal of dredged materials constitutes a considerable amount of the cost of sediment removal activities. These costs were figured into the estimates below assuming that space for approximately 250,000 cu. yd (1 cu. yd = 0.765 m³) of dredged material is available at or in the vicinity of current disposal sites near Bladensburg, MD (U.S. ACE, 1993). It was also assumed that up to 50,000 cu. yd of dredged materials would be used in a proposed U.S. ACE wetland restoration project in Kingman Lake (U.S. ACE, 1993). For volumes beyond this local disposal capacity, additional costs were added in to account for transport of the materials to an undetermined facility up to 50 miles away and subsequent offsite disposal. Lastly, the cost of obtaining and placing clean capping materials, both for the Kingman Lake project and the capping scenarios, was added when appropriate.

Cost Estimates

Table 16 summarizes the results of the cost estimate calculations. Because certain chemical

Table 16. Summary of Dredged Sediment Volumes and Cost Estimates.

Scenario	Volume of Dredged Sediment	Comparison to Disposal Capacity	Cost Estimates (Millions of Dollars)
1	10,324,258	34x	122 - 787
2	3,441,419	11x	50 - 260
3	4,431,267	15x	54 - 283
4	1,477,089	5x	20 - 101

Volumes in cubic yards. (1 cu. yd = 0.765 m³)

contaminants have been detected at high levels in sediments even at depths well below the sediment surface, the total volume of sediments estimated to be "contaminated" is many times the combined capacity of the readily available disposal options for dredged material in all of the scenarios investigated. Even if the dredging component of the comprehensive remediation options is deemed logistically and financially feasible, disposal of the dredged materials will be

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difficult if not impossible. For comparison, the volume of dredged materials removed from the Anacostia River during eight U.S. ACE projects since 1961 ranged from 30,000 to 145,000 cu. yds (U.S. ACE, 1993). Even the most limited dredging scenario requires the disposal of 5 times more dredged materials than the previous maximum amount. It is clear that high disposal costs will likely drive total costs toward the upper end of the estimate range for each of the scenarios.

Limiting remediation to a smaller area (Scenarios 3 and 4) reduces the volume of dredged materials and, therefore, the costs. In fact, additional chemical analyses and toxicity testing could lead to a refined delineation of the extent of the highly contaminated areas, thereby reducing the amount of dredged materials to within the local capacity. Costs for this level of remediation would likely be under 5 million dollars. Although noncomprehensive scenarios such as this are potentially more feasible, they leave in place large amounts of sediments that could be considered contaminated. There is no guarantee that only such limited remediation of the most highly contaminated sediments will result in the desired reduction of the impacts on human and ecological health. Further sampling and bioaccumulation modeling studies can help determine the benefits that might result from these and other scenarios.

Summary

As discussed previously, the benefits of any remediation activities will be only temporary unless current sources of chemical contaminants are first controlled. Controlling some nonpoint source contaminants, such as PAHs from automobiles, may require extensive, costly and time-consuming efforts throughout the watershed. However, it may be possible to control contaminants such as PCBs with less difficulty and in a shorter time-frame, since they appear to come from some relatively localized sources. As demonstrated by the cost estimates for simple dredging scenarios (Table 16), remediation of contaminated sediments will no doubt be an expensive undertaking. To justify the expense and provide the opportunity for remediation efforts to be successful in the long term, chemical contaminant sources must be identified and controlled at the onset.

It is difficult to predict to what extent any remediation action will contribute toward meeting the goal of eliminating chemical contaminant impacts on human and ecological health. As stated

above, mathematical models that estimate the ecological benefits of reducing the concentration (or availability) of sediment contaminants can be employed to help choose between various remediation alternatives. Such models could be developed and used as part of a more detailed study on the feasibility of the remediation of contaminated sediments.

This preliminary analysis of sediment remediation options suggests that comprehensive remediation of contaminated sediment in the tidal Anacostia River may not be feasible due to the magnitude and extent of the chemical contamination. However, once the sources of chemical contaminants are sufficiently controlled, a more detailed study of limited remediation alternatives (similar to Scenarios 3 and 4) could be conducted. Such a study would require some additional information on the extent of the contamination, the dynamics of sediment transport in certain areas of the river, and modeling estimates of the benefits from each of the alternatives.

Alternatively, a small-scale remediation project at the site with the highest contamination could be undertaken in the near term as a research project. The results of the project could be closely monitored and the lessons learned then applied to remediation in other areas in the river.

Although complete remediation of all contaminated sediments may not be possible in the short term, steps can be taken to reduce the sources of chemical contaminants and to plan feasible remediation activities that will ultimately lead to meeting the goal of eliminating the impacts of sediment contaminants on human and ecological health.

SUMMARY AND CONCLUSIONS

This study has provided a detailed first look at the vertical extent of sediment contamination in the lower Anacostia River. The results indicate that the problems in the lower river are even more extensive than previous studies of surface sediments had suggested. This increased understanding of conditions in Anacostia sediments will be a valuable guide for planning future assessments and other management actions.

The present study indicates that specific sedimentary areas are severely enriched and impacted by chemical contaminants within the District of Columbia. Areas with the highest sedimentary concentrations of many trace metals and organic contaminants, either at the surface or with depth (up to 300 cm), include the upper Washington Ship Channel and the Anacostia River near the Washington Navy Yard and the South Capitol Street Bridge. The examination of the core data indicates substantial enrichment of many trace metals and organic contaminants over background concentrations. Also, the number of multiple exceedences of guideline concentrations associated with adverse biological effects (TELs or PELs) indicate that the sediment bound contaminants within the Anacostia River and Washington Ship Channel pose a possible risk to aquatic organisms and human health via food chain bioaccumulation. In only the bottom section (280-300 cm) of core ANC-3, located below the South Capitol Street Bridge, did the concentrations of many metals and all organic contaminants fall below or at background concentrations. Therefore, any sediment management action must deal with multiple contaminants throughout the river and at depths of up to and possibly exceeding 3 m.

The benefits of any remediation activities will be only temporary unless current sources of chemical contaminants are first controlled. Controlling nonpoint source contaminants may require extensive efforts throughout the watershed. As demonstrated by the cost estimates for simple dredging scenarios, remediation of contaminated sediments will be an expensive undertaking. To justify the expense and provide the opportunity for remediation efforts to be successful in the long term, chemical contaminant sources must be identified and controlled at the onset.

It is difficult to predict to what extent any remediation action will contribute toward meeting the goal of eliminating chemical contaminant impacts on human and ecological health. The

preliminary analysis of sediment remediation options in this report suggests that comprehensive remediation of contaminated sediment in the tidal Anacostia River may not be feasible due to the magnitude and extent of the chemical contamination. However, once the sources of chemical contaminants are sufficiently controlled, a more detailed study of limited remediation alternatives could be conducted, or a small-scale remediation project at the site with the highest contamination could be undertaken in the near term. These projects will require additional information on the extent of the contamination, the dynamics of sediment transport in certain areas of the river, and modeling of contaminant fate and transport.

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