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Potential Long-Term Ecological Impacts Caused by Disturbance of Contaminated Sediments: A Case Study

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ABSTRACT / Several submerged barges were recently removed from the Passaic River, New Jersey, USA, in two areas (areas 1 and 2) where contaminated sediments are known to exist. During removal of the single barge in area 1, elevated turbidity levels and chemical parameters were measured. Greater increases were measured in area 2, where several barges were removed. In both areas, water column concentrations of polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs) and several metals exceeded one or more water quality criteria; turbidity levels in area 2 also exceeded regulatory criteria. Potential chemical bioaccumulation from the water column into residential aquatic receptors was estimated using standard models and assumptions. The modeled results predicted that steady-state tissue concentrations of bioaccumulative chemicals would not occur as a result of the brief increase in water column concentrations that occurred during barge removal but that metals and PCDD/Fs could bioaccumulate to levels that exceed regulatory ecological criteria during long-term sediment disturbance activities. In addition, based on some simplistic assumptions regarding settling of suspended sediments, we estimate that chemical bioaccumulation from surface sediments into the food web could result in substantial increases in PCDD/F body burdens in the benthic forage fish, mummichog. Our findings are consistent with the limited number of field studies that have measured increased body burdens of bioaccumulative chemicals following dredging. We suggest that, prior to consideration of extensive dredging as a remedial alternative for any river system, the potential significant and long-term impacts on the food web must be evaluated.

Sediment volumes in excess of 250 million m^3 are dredged annually in the United States using a wide variety of mechanical and hydraulic methods (US EPA 1998). The nature and extent of the environmental impacts typically associated with physical disturbance of sediments have been reviewed extensively and can include direct destruction of benthic habitat; the associated adverse impacts to aquatic, terrestrial, and avian food webs; and the decreased water quality as a result of sediment suspension (Kennish 1992, Lee and Jones 1992, Johnston 1981, Allen and Hardy 1980). While the mechanical and physical disturbances alone are often sufficient cause for concern, the impacts to ecological health might be even more extensive if the disturbed sediment contains elevated chemical concentrations. Specifically, suspension and dispersal of contaminated sediments could have ecological impacts that extend well beyond the time the physical effects caused by mechanical disturbances (e.g., water column turbidity) have returned to baseline conditions, particularly if persistent and bioaccumulative chemicals are involved. Sustained suspension of contaminated particulates in the water column could result in increased chemical body burdens in aquatic biota via partitioning from

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sediment to water to tissue. Dispersal and settling of contaminated sediment particles may create a new, and more contaminated, chemical horizon in the surface sediment in and near the dredged area (US EPA 1997).

Very little is known regarding the long-term chemical impacts of sediment disturbance. Some investigations have assessed the uptake of chemicals into caged fish and shellfish during and shortly after dredging, and in general these studies have reported no significant bioaccumulation in tissues. However, most of these studies did not attempt to measure the food web impacts that might occur long after dredging was completed. Presumably, significant bioaccumulation of persistent compounds into the upper trophic species might not be apparent until well after the sediment disturbance has subsided. Indeed, in one of the few published long-term studies (Rice and White 1987), increased PCB uptake into caged clams and fish was still occurring six months after dredging of the Shiawassee River, Michigan, USA. The paucity of information regarding long-term impacts would appear to be a potentially significant data gap, particularly since some regulatory authorities believe that remediation of contaminated sediments should be given high priority in the coming years. Specifically, the US Environmental Protection Agency (US EPA) recently identified 96 watersheds in the United States which it believes contain contaminated sediments that pose significant health and ecological risks; these waterways have been targeted as areas where additional research and sediment treatment and/or removal are considered to be a high priority (US EPA 1997).

One of the waterways noted by US EPA is the Hackensack–Passaic watershed, which feeds into Newark Bay, part of the Port of New York and New Jersey. According to the Interim Dredged Material Management Plan for the Port of New York and New Jersey, approximately 4 million yd^3 of sediment must be dredged annually for maintenance of New York/New Jersey (NY/NJ) channels and Newark Bay, in return for an economic benefit of $463 million from shipping traffic (US ACE 1996). We recently reviewed the decisions of 15 dredge permit applications initiated in the NY/NJ harbor in the last decade and found that, in a majority of the cases, the US Army Corps of Engineers (US ACE) and US EPA concluded that the concentrations of bioaccumulative chemicals in sediments exceeded the applicable criteria for permitting unrestricted open ocean disposal of the dredged sediments. As a result, the sediments were dredged and disposed of using other means. These dredging operations were not accompanied by any evaluation of the actual chemical impacts in the dredged areas themselves. Indeed, to our knowledge, no such data have been collected in the watershed despite the removal of very large volumes of sediment during this same period of time. It would seem reasonable to conclude that, if the sediments are sufficiently contaminated to warrant concern regarding their method of disposal, then there should be cause for concern regarding chemical impacts in and near the areas of dredging activity.

Recently, water quality data were collected near an area where the US ACE removed several abandoned barges that were partially entrenched in the bottom of the Passaic River, New Jersey. The Passaic River is within the Newark Bay region, an area that has been a center of industrialization for over two centuries. Extensive sediment investigation efforts have characterized the elevated levels of metals (Bonnevie and others 1992, 1993, 1994, Gillis and others 1993, Wenning and others 1994), polycyclic aromatic and petroleum hydrocarbons (Huntley and others 1993, 1995, Wenning and others 1994); DDT, DDD, and DDE (Gillis and others 1995), PCBs (Kennish and Ruppel 1996, Wenning and others 1994, Kennish and others 1992); and polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans (PCDD/Fs) (Belton and others 1985, Bopp 1988, Tong and others 1990, Bopp and others 1991, Finley and others 1990, Rappe and others 1991, Ehrlich and others 1994) that exist in Passaic River sediments. Chemical concentration profiles from sediment coring studies by Huntley and others (1993, 1997), Wenning and others (1994), and Gillis and others (1995) have shown that contaminant levels generally increase with depth. The surface and subsurface sediments near the proposed barge removal areas are known to contain many of the aforementioned chemicals. Prior to the removal of the barges, the US ACE conducted studies at this location and issued an environmental impact study (EIS) that concluded that any removal-related impacts to the mudflats, vegetation, wildlife, and aquatic resources would be temporary and insignificant (US ACE 1995). The scope of the EIS, however, focused on physical disturbance of the local habitat and transient water column impacts and did not consider increased bioaccumulation that might occur as a result of sediment disturbance.

During the barge removal activities, which occurred in April 1999, unfiltered water and filtered solids samples were collected and analyzed for numerous chemicals, in addition to several other water quality parameters, near the barge removal areas. In this paper, these measurements are used to develop a preliminary evaluation of the potential short- and long-term ecological impacts associated with sediment disturbance in the Passaic River. The aforementioned water column data consisted of grab samples and therefore represent brief snapshots of the conditions that occurred during barge

removal. In this analysis, these data are used to develop a timeline of elevated water column concentrations that could reasonably be expected to exist for a range of different time periods. We also estimate the surface sediment concentrations of bioaccumulative chemicals that might exist after settling of the disturbed sediments. By employing standard food web models and assumptions regarding chemical partitioning between different media, the water column and surface sediment concentrations are used to estimate increased chemical body burdens in representative benthic macroinvertebrates and finfish. For the purposes of this analysis, we estimate increased body burdens of bioaccumulative chemicals in blue crab and mummichog; these are two dominant resident species in the Passaic River that would be most likely to experience body burden increases due to disturbance of contaminated sediments. The estimated increase in body burdens provides a basis for assessing the potential adverse impacts to ecological receptors that might result for a variety of different sediment disturbance events.

Methods

Barge Removal Operations

The Passaic River Drift Removal Project of the US ACE focused on five derelict barges, including their associated parts and contents, located in three areas within the Passaic River (US ACE 1998). In the segment containing these three areas, the river has a bank-to-bank width of approximately 45–90 m and the depth, as a federal navigational channel, was maintained at 9 m until navigation dredging of this area was suspended in the 1980s. Barge 1, located within area 1, was situated against the western shore retaining wall. The remains consisted of piled wood that was partially embedded within the sediment and appeared to extend several feet above the water surface (Figure 1). Barges 2, 3, and 4, located within area 2, roughly 1.5 km upstream from area 1, were situated perpendicular to the eastern shore and about 30–45 m apart. Barge 5, located within area 3 roughly 7 km upstream from area 2, was not monitored in this study.

The removal of barges 1–4 took place on 9–12 April 1999. The operations were intended to be conducted during low-tide conditions to minimize water quality impacts. The removal equipment included a barge-mounted clamshell dredge, a debris barge, and a tugboat. The dredging barge was equipped with four spuds for anchoring; the spuds were deployed into the river bottom once the removal barge was positioned roughly 7.5 m from the abandoned barges. To prevent floating debris from escaping from the removal area, a tapered boom was prepared, intended to surround the abandoned barges and removal equipment. The removal procedure consisted of the clamshell excavating materials from both above and below the water surface and placing those materials on the debris barge (BB&L 2000). The removal of each barge was conducted generally over two days and the removal chronology is illustrated in Figure 2.

Monitoring and Sample Collection

Physicochemical parameters of the water column were monitored and composite samples of unfiltered water and filterable solids were collected for chemical analyses (BB&L 2000). Sampling and measurements were made to characterize changes in water column chemistry within a feasible vicinity and tidal depth near the barges. Physicochemical parameters measured included temperature, alkalinity (pH); dissolved oxygen (DO), and turbidity. Unfiltered water samples were analyzed for inorganics, total cyanide, extractable petroleum hydrocarbons, ammonia-nitrogen, Kjeldahl nitrogen, total organic carbon, dissolved organic carbon, total suspended solids, PCB congeners, 2,3,7,8-substituted PCDD/Fs, herbicides, pesticides and Aroclors, polycyclic aromatic hydrocarbons (PAHs), and semivolatile organics. In addition, water samples were collected separately and filtered to obtain filterable solid samples for PCB and PCDD/F analyses. All chemical analyses were performed by Lancaster Laboratories (Lancaster, Pennsylvania, USA), with the exception of PCB and PCDD/F congener analyses, which were conducted by Alta Analytical Laboratories (El Dorado, California, USA).

In area 1, physicochemical parameters were monitored within each of three subareas (A, B, and C) surrounding barge 1, at every 0.6 m (2 ft) at depths up to roughly 4.5 m. Physicochemical parameters were monitored in each subarea one day prior to and one day after the 9 April and 12 April barge 1 removal activities; monitoring was also conducted in the subareas at two different times during the 9 April removal. Two unfiltered water samples were collected for chemistry parameters during the 9 April barge 1 removal at 80% of the water depth in subareas B (PR99A102SW1) and C (PR99A103SW1). Each sample consisted of a composite of water collected in two locations roughly 6 m (20 ft) apart within each subarea, at roughly 9 m (30 ft) outside the tapered boom. A filterable-solids sample was also collected at a similar depth from subarea C (PR99A1FS1) during the 9 April removal activities. No samples were collected in area 1 during the 12 April removal activities. Figure 2 summarizes the chronology of the turbidity measurements in areas 1 and 2. Tables 1 and 2 summarize selected chem-

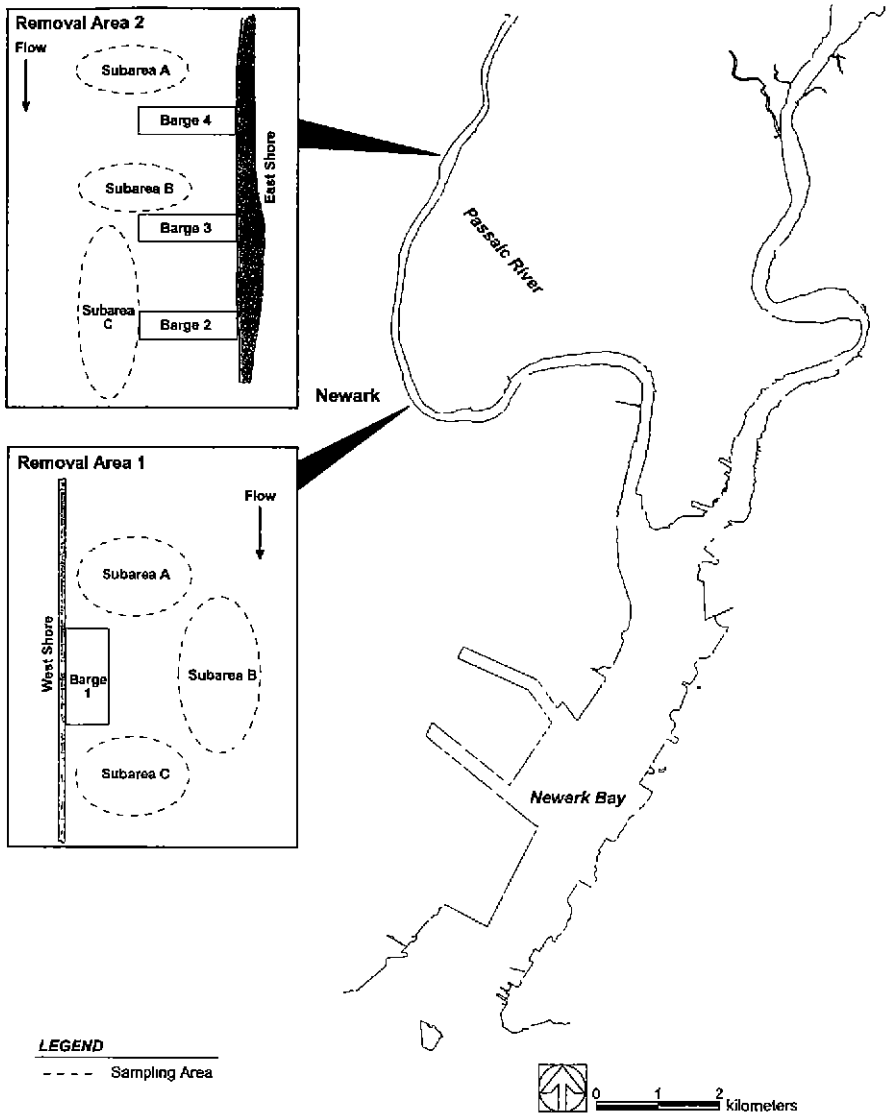


Figure 1. Barge removal areas 1 and 2

istry data for the water and filterable solids samples, respectively, collected in area 1.

In area 2, physicochemical parameters were monitored in subareas A and C during the 9 April removal of barge 4. Additional monitoring of physicochemical parameters was conducted in subareas A and B during the 10 April barge 4 removal and repositioning. One filterable-solids sample was collected at subarea A (PR99A2FS1) during the 9 April barge 4 removal at 80% of the water depth, and two unfiltered water samples for chemistry were collected in subarea B (PR99A201SW1 and duplicate, PR99A202SW1) at similar depth during the 10 April barge 4 removal. Prior to barge 3 removal, a water sample was collected for chemistry parameters in subarea A (PR99A203SW1) at 80% of the water depth. During the 10 April barge 3 removal, physicochemical parameters were monitored at every 0.6 m (2 ft), and an unfiltered water sample for chemistry parameters (PR99A204SW1) was collected at 80% of water depth in subarea C. During the 11 April barge 3 removal physicochemical parameters were monitored and a filterable-solids (PR99A2FS2) and an unfiltered water (PR99A205SW1) sample were collected within subarea C. Barge 2 was removed during low tide and, as a result, samples were not collected during the removal activity due to insufficient water depth. Physicochemical parameters were monitored 1 h before barge 2 removal in subarea C on 12 April and one day after barge 2 removal in subareas A, B, and C for every 2 ft in water depth. Tables 2 and 3 summa-

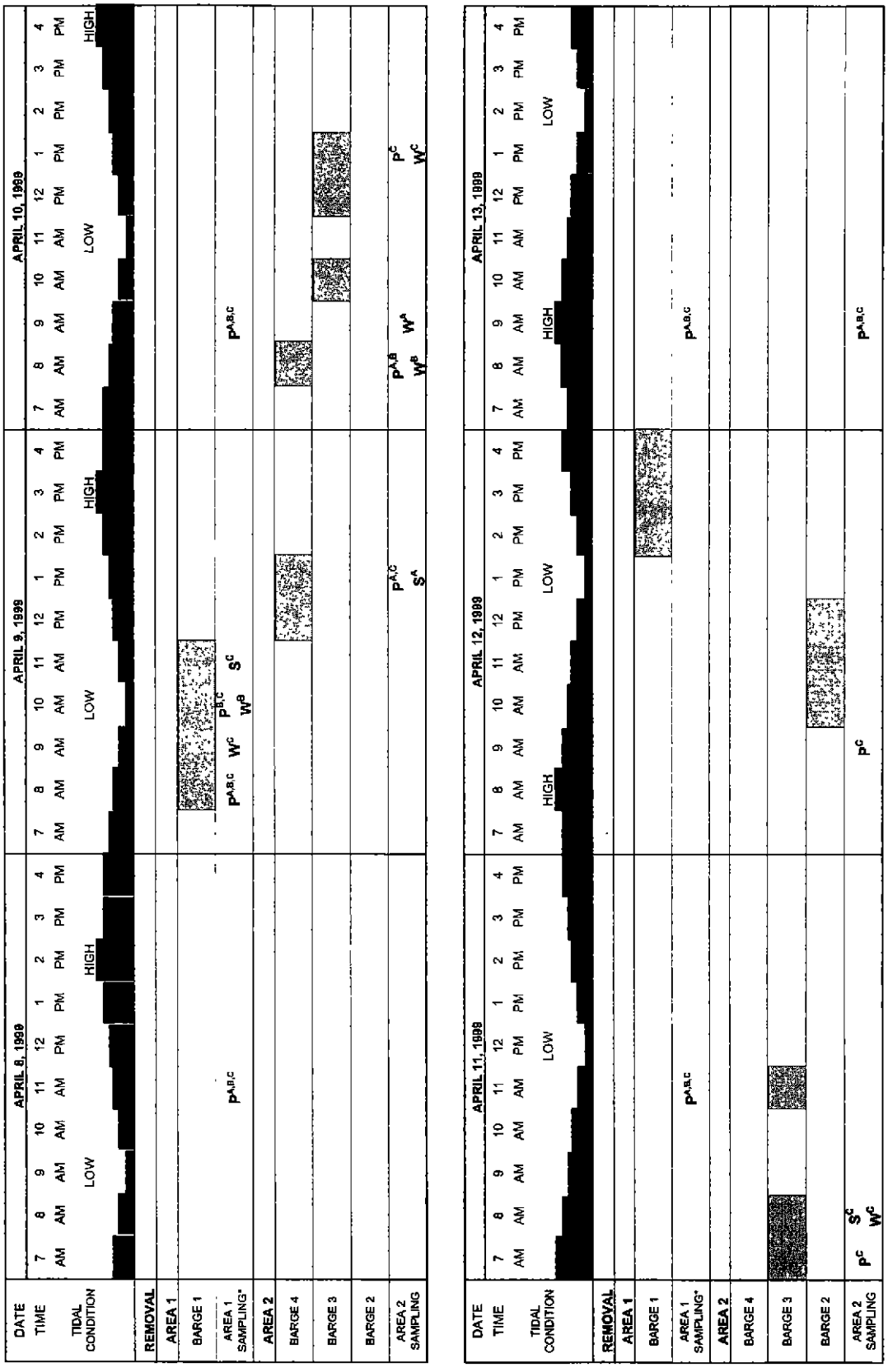


Figure 2. Chronology of barge removal and sampling activities. Sampling type: P, physicochemical of water column; S, filterable solids; W, water column; superscripts A, B, and C denote sampling subareas. ▨ Barge removal in progress.

Table 1. Selected chemistry data for unfiltered water samples collected 9 April 1999 in area 1, barge 1

			Arithmetic mean
Sample ID:	PR99A102SW1	PR99A103SW1	
Subarea:	B	C	
2,3,7,8-PCDDs/PCDFs[a] (pg/liter)			
2,3,7,8-TCDD	7.4[b,c]	23[b,c]	15.20
1,2,3,4,6,7,8-HpCDD	12	43	27.50
OCDD	180	580	380.00
2,3,7,8-TCDF	ND (1)[d]	2.9	1.70
1,2,3,4,7,8-HxCDF	ND (1.9)	10	5.48
1,2,3,4,6,7,8-HpCDF	10	42	26.00
OCDF	18	76	47.00
TEQ[c]	7.65	21.67	16.16
PCB congeners (ng/liter)			
2,3,3′,4,4′-PeCB (PCB 105)	ND (0.5)	0.69	0.47
2,3′,4,4′,5-PeCB (PCB 118)	0.77	1.5	1.14
TEQ[c]	5.10E-06	1.10E-05	8.03E-06
Metals (mg/liter)[e]			
Aluminum	0.23	0.93[e,f]	0.58
Lead	0.0044	0.0143[b,c]	0.0094
Manganese	0.088	0.111[b,c]	0.100

[a]Only those congeners that have WHO fish TEFs and were detected in at least 1 sample are presented. TEQs are calculated from WHO fish TEFs (Van den Berg and others 1998).

[b]Concentration exceeded New Jersey Department of Environmental Protection Surface Water Quality Criteria for protection of human health (NJDEP 1998)

[c]Concentration exceeded USEPA Recommended Water Quality Criteria for protection of human health (US EPA 1999b)

[d]ND: Not detected, number in parentheses is the limit of detection

[e]Only metals whose concentration exceeded one or more regulatory water quality standard in one or more sample are shown.

[f]Concentration exceeded USEPA Recommended Water Quality Criteria for protection of aquatic life (US EPA 1999b)

rize selected chemistry data for the water samples from area 2 and filterable solids samples from both areas, respectively.

Exposure Scenarios

The benthic forage fish, mummichog (*Fundulus heteroclitus*) and blue crab (*Callinectes sapidus*) are the subject of the exposure analysis since they are key members of the local aquatic foodweb that is susceptible to changes in water column and sediment chemistry. Their role in the local food web has been described by Iannuzzi and others (1996).

For the purposes of this analysis, bioaccumulation of chemicals from the water column into mummichog and blue crab during sediment disturbance activities is estimated for several periods: two days, three months, six months, nine months, and one year. These periods are intended to represent the duration of several different sediment disturbance activities ranging from short-term (e.g., barge removal) to long-term (extensive wide-area dredging). The water column concentrations are assumed to remain constant (and equivalent to the values measured in this analysis) throughout these time periods. A food web model is also used to estimate bioaccumulation from surface sediment following the sediment disturbance activities.

Estimating Bioaccumulation from Water Column

Bioaccumulation of chemicals from the water column into mummichog and blue crab tissue was estimated using the measured chemical concentrations in the unfiltered water column samples and chemical-specific bioconcentration factors (BCFs). For the purposes of this analysis, bioaccumulation from water was calculated for metals detected at concentrations that exceeded one or more water quality criteria in one or more samples and PCB and PCDD/F congeners that were detected in at least one sample and for which the World Health Organization has established fish toxic equivalency factors (TEFs) as described by Van den Berg and others (1998).

The mean of the water column concentrations measured during the 9 April removal of barge 1 and the mean of the water column concentrations measured during the 10 and 11 April removal of barges 3 and 4 were used to represent conditions in areas 1 and 2, respectively. The mean values are shown in Tables 1 and 3, respectively.

Table 2. Selected chemistry data for unfiltered water samples collected in area 2

Sample ID:	PR99A201SW1	PR99A202SW1	PR99A203SW1	PR99A204SW1	PR99A205SW1	
Barge:	4	4	4	3	3	
Subarea:	B	B	A	C	C	Arithmetic
Sample Date:	10 April 1999	10 April 1999	10 April 1999	10 April 1999	11 April 1999	mean
2,3,7,8-PCDDs/PCDFs (pg/liter)[a]						
2,3,7,8-TCDD	1000[b,c]	1300[b,c]	29[b,c]	510[b,c]	1300[b,c]	827.80
1,2,3,7,8-PeCDD	4 80	6.1	ND (1.4)[d]	1.5	ND (4.7)	132.62
1,2,3,4,7,8-HxCDD	3 9	4.6	1.2	1.8	ND (11)	2.77
1,2,3,6,7,8-HxCDD	17	26	3.7	5.9	ND (18)	11.62
1,2,3,7,8,9-HxCDD	8 5	11	2.2	3	ND (13)	6.74
1,2,3,4,6,7,8-HpCDD	230	320	50	67	290	134.70
OCDD	2800	3700	610	900	3100	1660.00
2,3,7,8-TCDF	15	18	2.2	7.4	20	628.52
1,2,3,7,8-PeCDF	7.2	8.5	ND (1.8)	2.5	ND (13)	7.82
2,3,4,7,8-PeCDF	24	30	3.5	8	27	14.40
1,2,3,4,7,8-HxCDF	120	100	16	20	100	56.60
1,2,3,6,7,8-HxCDF	24	23	5.5	6	28	31.70
1,2,3,7,8,9-HxCDF	6.8	5.8	1.7	2	ND (16)	8.86
2,3,4,6,7,8-HxCDF	15	17	3.6	4.9	ND (12)	9.70
1,2,3,4,6,7,8-HpCDF	570	380	140	84	460	236.00
1,2,3,4,7,8,9-HpCDF	17	14	4.5	3.8	ND (12)	99.86
OCDF	1200	660	240	140	890	449.20
TEQ	543.20	694.37	22.02	266.32	689.63	443.11
PCB congeners (ng/liter)[a]						
3,3',4,4'-TeCB (PCB 77)	23	16	1.6	8.8	5.9	9.88
3,4,4',5-TeCB(PCB 81)	2.1	1.6	ND (0.51)	0.78	ND (1 0)	1.05
2,3,3',4,4'-PeCB (PCB 105)	55	37	3.8	17	19	22.56
2,3,4,4',5-PeCB (PCB 114)	3	2.5	ND (0.51)	0.84	ND (1.0)	1.42
2,3',4,4',5-PeCB (PCB 118)	130	89	8.8	39	46	53.36
2',3,4,4',5-PeCB (PCB 123)	3.2	2.1	ND (0.51)	0 97	ND (1.0)	1.41
3,3',4,4',5-PeCB (PCB 126)	0.7	ND (0.5)	ND (0.51)	ND (0.52)	ND (1.0)	0.39
2,3,3',4.4',5-HxCB (PCB 156)	13	9.2	1.2	4.6	4.1	5.60
2,3,3',4.4',5'-HxCB (PCB 157)	3	2.2	ND (0.51)	1 1	ND (1.0)	1.41
2,3',4,4',5,5'-HxCB (PCB 167)	5 10	3.30	ND (0.51)	1.90	ND (1.0)	2 21
2,2',3,3',4,4',5-HpCB (PCB 170)	33	23	4 1	15	10	15 02
2,3,3',4,4',5,5'-HpCB (PCB 189)	1 2	0.92	ND (0.51)	0.58	ND (1.0)	0 69
TEQ[a]	7.92E-03	4.38E-03	1 64E-03	2.90E-03	3.70E-03	4.11E-03
Metals (mg/liter)						
Aluminum	2.12[e]	4.43[e]	0.68	0.98[e]	0.79[e]	1.8
Arsenic	ND (0.01)[f]	0.012[b,c]	ND (0.01)[f]	ND (0.01)[f]	ND (0.01)[f]	0.0052
Copper	0.069[e,g]	0.123[e,g]	ND (0.025)[h]	0.042[e,g]	0.026[e,g]	0.055
Lead	0.103[b]	0.175[b]	0.0197[b]	0.0628[b]	0.0257[b]	0.077
Zinc	ND (0.197)[h]	0 365[e]	0.046	0.143[e]	0.074	0.083

[a]Only those congeners that have WHO fish TEFs and were detected in at least 1 sample are presented. TEQs are calculated from WHO fish TEFs (Van den Berg and others 1998)

[b]Concentration exceeded New Jersey Department of Environmental Protection Surface Water Quality Criteria for protection of human health (NJDEP 1998).

[c]Concentration exceeded US EPA Recommended Water Quality Criteria for protection of human health (US EPA 1999b).

[d]Not detected number in parentheses is the limit of detection.

[e]Concentration exceeded US EPA Recommended Water Quality Criteria for protection of aquatic life (US EPA 1999b).

[f]Detection limit reported exceeds New Jersey criteria.

[g]Concentration exceeded New Jersey Department of Environmental Protection Surface Water Quality Criteria for protection of aquatic life (NJDEP 1998)

[h]Detection limit reported exceeds Federal criteria.

Table 3. Selected chemistry data for filterable solids samples collected in areas 1 and 2 during barge removal

Removal area	1	2	2	
Sample ID:	PR99A1FS1	PR99A2FS1	PR99A2FS2	
Barge:	1	4	3	
Subarea:	G	A	C	Area 2
Sample date:	9 April 1999	9 April 1999	11 April 1999	arithmetic mean
2,3,7,8-PCDDs/PCDFs (ng/kg)[a]				
2,3,7,8-TCDD	340	5,600	8,100	6,850.00
1,2,3,7,8-PeCDD	10	16	43	29.50
1,2,3,4,7,8-HxCDD	9.4	12	27	19.50
1,2,3,6,7,8-HxCDD	33	48	130	89.00
1,2,3,7,8,9-HxCDD	24	23	55	39.00
1,2,3,4,6,7,8-HpCDD	580	690	1,600	1,145.00
OCDD	7,300	8,900	19,000	13,950.00
2,3,7,8-TCDF	23	37	100	68.50
1,2,3,7,8-PeCDF	14	17	40	28.50
2,3,4,7,8-PeCDF	33	56	160	108.00
1,2,3,4,7,8-HxCDF	110	180	530	355 00
1,2,3,6,7,8-HxCDF	40	56	160	108 00
1,2,3,7,8,9-HxCDF	7 7	8.6	27	17 80
2,3,4,6,7,8-HxCDF	19	33	78	55.50
1,2,3,4,6,7,8-HpCDF	520	860	2,200	1,530.00
1,2,3,4,7,8,9-HpCDF	24	29	76	52.50
OCDF	900	1,500	5,500	3,500.00
TEQ	398.13	5,691.79	8,351.66	7,021.73
PCB congeners (ug/kg)[a]				
3,3',4,4'-TeCB (PCB 77)	7	12	40	26.00
3,4,4',5-TeCB(PCB 81)	ND (1.4)[b]	1.6	3.3	2.45
2,3,3',4,4'-PeCB (PCB 105)	22	34	100	67.00
2,3,4,4',5-PeCB (PCB 114)	ND (1.4)	1.2	8.8	5.00
2,3',4,4',5-PeCB (PCB 118)	52	81	240	160 50
2',3,4,4',5-PeCB (PCB 123)	ND (1.4)	1.5	ND (0.57)	0 89
3,3',4,4',5-PeCB (PCB 126)	2	ND (0.92)	1.6	1.03
2,3,3',4,4',5-HxCB (PCB 156)	6 5	7.9	23	15.45
2,3,3',4,4',5'-HxCB (PCB 157)	ND (1.4)	1 7	4.7	3.20
2,3',4,4',5,5'-HxCB (PCB 167)	3.4	ND (0.92)	9.7	5.08
2,3,3',4,4',5,5'-HpCB (PCB 189)	ND (1.4)	ND (0.92)	2.4	1.43
TEQ	0.011	0.0049	0.015	0.010

[a]Only those congeners that have WHO fish TEFs and were detected in at least 1 sample are presented. TEQs are calculated from WHO fish TEFs (Van den Berg and others 1998)

[b]Not detected, number in parentheses is the limit of detection

Table 4. Estimated metal body burden in aquatic biota attributable to water column uptake and levels in previous mummichog samples

			Removal area		
Chemical	Kd_{sw} (liters/kg)[a]	BCF (liters/kg)[b]	1 (mg/kg)	2 (mg/kg)	1995 samples (mg/kg)[c]
Copper	28,500	710	ND	7 2	3.5
Lead	900	0.090	0.00081	0 0060	1.8
Zinc	62	2059	69	296	45

[a]Sediment-to-surface water partition coefficient from US EPA (1999a)

[b]BCF values from US EPA (1998).

[c]Mean of five Passaic River mummichog composite samples collected in 1995.

Metals, PCDDs/Fs, and PCBs have a high affinity for organic particulates, only that fraction that is freely dissolved is available for bioaccumulation into tissue via the water column. Accordingly, an estimate of the freely dissolved fraction must first be derived from the unfiltered water column samples. The freely dissolved concentration of the metals was estimated by:

$$C_{fd} = \frac{1}{1 + Kd_{su} \times TSS \times 10^{-6}} \times C_w$$

based on US EPA (1999a) where Kd_{su} is the suspended sediment-to-surface water partition coefficient (liters per kilogram) and TSS is total suspended solid concentration (milligrams per liter). The partition coefficients used in this analysis are those recommended by US EPA (1999a). Whole-body concentrations of metals were then calculated as the product of BCF and freely dissolved chemical concentration.

The BCF values for the metals are based on studies representing various exposure durations (US EPA 1999a). The values for the suspended sediment-to-surface water partition coefficient (Kd_{su}) and BCFs for the metals are shown in Table 4.

Levels of freely dissolved PCDD/F and PCB congeners were estimated by the following equation:

$$C_{fd} = \frac{1}{1 + \frac{(DOC)(K_{ow})}{10} + (POC)(K_{ow})} \times C_w$$

where C_{fd} is the concentration of freely dissolved chemical, DOC is the concentration of dissolved organic carbon (kilograms per liter), K_{ow} is the octanol–water partition coefficient of the chemical, POC is the concentration of particulate organic carbon (kilograms per liter), and C_w is the total concentration of the organic chemical in water (US EPA 1995). K_{ow} values for these chemicals were obtained from US EPA (1995) for PCDD/F congeners and Hawker and Connell (1988) for PCB congeners.

BCF values for PCDD/F and PCB congeners reported in the literature are not applicable for this analysis because they were derived from estimates or measurements under steady-state conditions (between water and fish tissue). Steady-state conditions for PCBs and PCDD/Fs will not be reached in many of the time periods considered in this analysis because a significant amount of time is required for PCBs and PCDD/Fs to accumulate in tissues to the point where a steady-state (no further increase in tissue concentration) is reached. For this analysis, the polynomial equations presented by Connell and Hawker (1988) that account

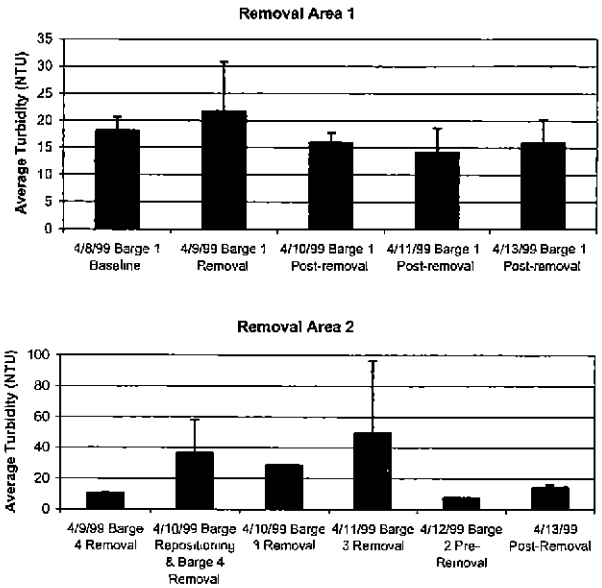


Figure 3. Average turbidity levels measured during various stages of barge removal in areas 1 and 2. Error bars show 1 SD for three or more measurements.

for time-dependent uptake and depuration kinetics were applied to estimate lipid-based PCDD/F and PCB congener BCF values. The expressions for estimating BCF from log K_{ow} values at time t (in days) are the following:

$$BCF_t\,(lipid) = BCF_{eq}\,(1 - e^{-k_2 t})$$

$$BCF_{eq} = 6.9 \times 10^{-3} (\log K_{ow})^4 - 1.85 \times 10^{-1} (\log K_{ow})^3 + 1.55 (\log K_{ow})^2 - 4.18 (\log K_{ow}) + 4.79$$

$$\log\left(\frac{1}{k_2}\right) = 6.9 \times 10^{-3} (\log K_{ow})^4 - 1.85 \times 10^{-1} (\log K_{ow})^3 + 1.65 (\log K_{ow})^2 - 5.34 (\log K_{ow}) + 5.27$$

where BCF_{eq} is the steady-state bioconcentration factor, k_2 the elimination rate, and t the exposure duration in days. The estimated BCFs using Connell and Hawker (1988) methodology were reported to be quite consistent with experimentally measured values for PCDD/F congeners under near steady-state conditions (Loonen and others 1994).

The incremental increase in PCDD/F and PCB body burden, in terms of lipid concentration, may be estimated by:

$$C_{lipid} = BCF_{lipid} \times C_{fd}$$

where C_{lipid} is the estimated concentration of PCDDs/PCDFs and PCBs in mummichog and blue crab lipid,

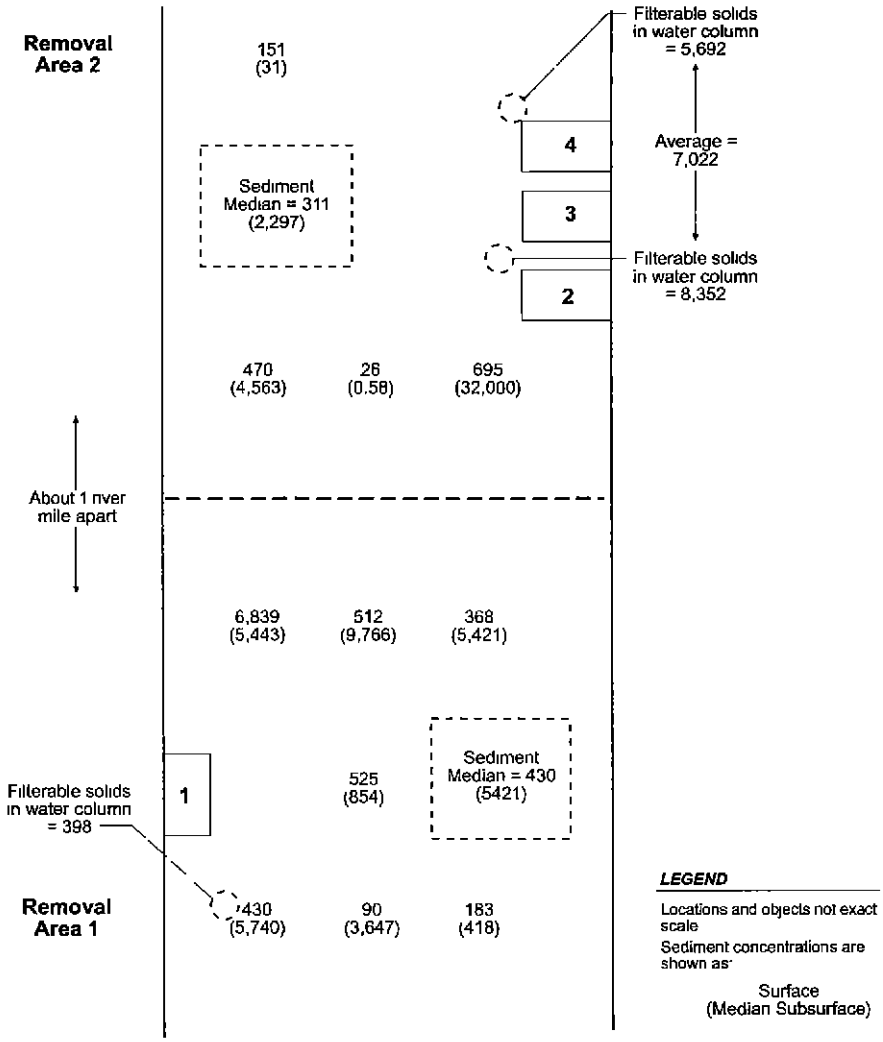


Figure 4. PCDD/F WHO Fish TEQ Concentration (ng/kg) in suspended particulates and historical sediment samples.

BCF_{lipid} is the lipid-based bioconcentration factor, and C_{fd} is the freely dissolved concentration. Whole body wet weight concentrations are then estimated as follows:

$$C_{WB} = C_{lipid} \times f_{lipid}$$

where C_{WB} is the whole body wet weight concentration of the organism and f_{lipid} is the lipid fraction of the organism. For the purpose of this analysis, f_{lipid} values for mummichog and blue crab were assumed to be 2.2 and 2.8%, respectively, based on Iannuzzi and others (1996).

Estimating Bioaccumulation from Surface Sediment

In general, metals do not demonstrate significant food-chain bioaccumulation, and therefore they are not considered in the food web analysis. The increase in PCB and PCDD/F body burdens associated with settling of contaminated sediments was quantified using the food web model of Iannuzzi and others (1996). In this model, the uptake from sediment, water, and prey are represented by a simple food web consisting of dominant species from various trophic levels including the polychaete (*Nereis virens*), mummichog, blue crab, and striped bass (*Morone saxatilis*). This model has previously been used to accurately estimate PCB and PCDD/F body burdens in the Passaic food web (Finley and others 1997, Iannuzzi and others 1996).

Postdisturbance surface sediment concentrations of PCDD/Fs and PCBs were assumed to be equivalent to the concentrations measured in the filterable solids. The filterable solids sample collected during the 9 April removal of barge 1 is considered representative of sediment suspension in area 1; the mean of the filterable solids samples collected during the 9 and 11 April

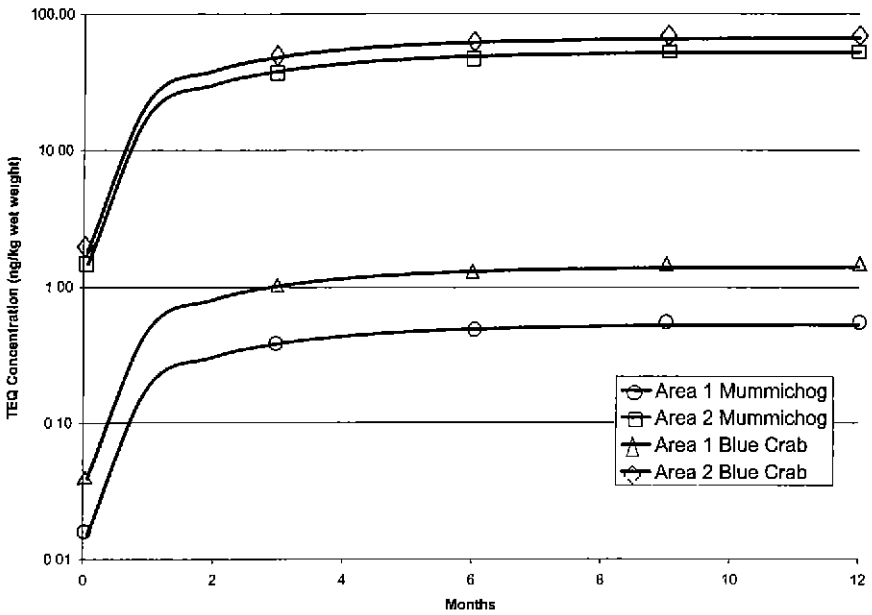


Figure 5. Time dependency of estimated mummichog and blue crab 2,3,7,8-TCDD toxic equivalent (TEQ) concentration attributable to increased water column concentration. TEQ based on WHO toxic equivalent factors for fish and shellfish from Van den Berg et al. (1998).

removal of barges 4 and 3, respectively, was considered representative of sediment suspension in area 2. These values are summarized in Table 2.

Results

Physicochemical Parameters

There was very little variation in temperature, dissolved oxygen, and pH throughout the barge removal activities in areas 1 and 2 (data not shown). The measured levels of dissolved oxygen and pH were all within acceptable US EPA and NJDEP water quality ranges (NJDEP 1998, US EPA 1999b). As shown in Figure 3, there was a slight increase in turbidity in area 1 during the removal of barge 1 on 9 April (data points shown are averages of all depths and subareas combined). A much greater increase in turbidity was measured in area 2. Specifically, mean turbidity levels during the 10 April barge 4 removal/repositioning and 11 April barge 3 removal were 37 and 50 Nephelometric Turbidity Units (NTU), respectively, while the turbidity levels measured at all other times were less than 20 NTU. Two measurements taken from the water surface during 10 April barge 4 removal and 11 April barge 3 removal exceeded the NJDEP's maximum allowable turbidity criteria of 50 NTU.

Water Column Chemistry

Tables 1 and 3 summarize the results for areas 1 and 2, respectively, for selected chemicals. All PCDD/Fs and PCB congeners detected in one or more samples and all metals that exceeded one or more criteria in one or more samples are summarized. SVOCs and PAHs were not detected in the water samples.

In general, higher concentrations were detected in samples collected in locations downstream from the barges. For example, in area 1 (Table 1) the chemical concentrations in the downstream sample (subarea C) were generally at least threefold higher than in the sample collected upstream in subarea B. A similar gradient was seen in area 2 (Table 3), where downstream samples in subareas B and C contained chemical concentrations that were generally higher (up to 10-fold) than in the upstream sample in subarea A. These patterns are consistent with the turbidity measurements.

Filterable Solids Chemistry

As shown in Table 2, and consistent with the turbidity and water chemistry results, more chemicals were detected, and at higher concentrations, in area 2 than area 1 and chemical concentrations in downstream samples (subarea C in area 2) were higher than in upstream samples (subarea A in Area 2). Chemical concentrations

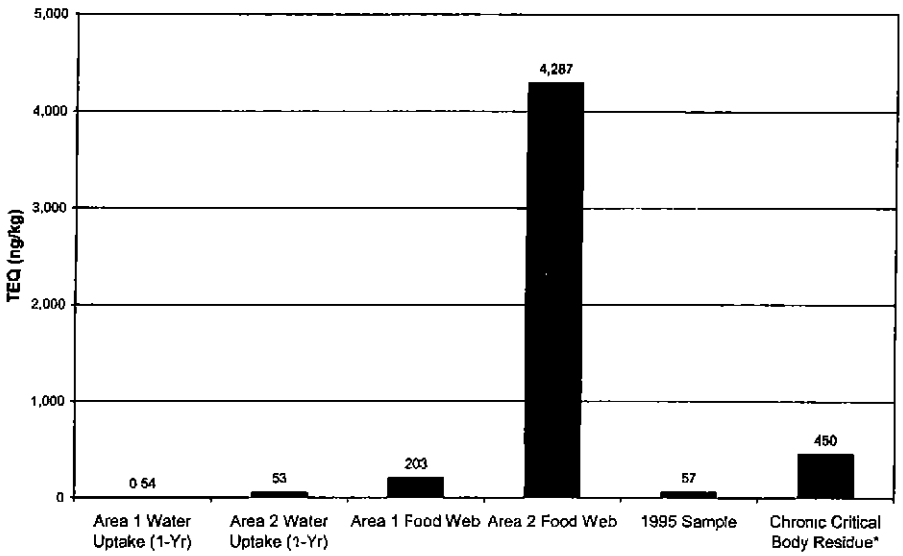


Figure 6. Estimated and measured PCB and PCDD/F TEQ concentrations (ng/kg) in mummichog from the Passaic River. *Maximum chronic TEQ critical body residue cited in screening level human and ecological risk assessment (ChemRisk 1995).

measured in the filterable solids can be compared to values previously reported for sediments in nearby locations (BB&L 1999, Ehrlich and others 1994, Wenning and others 1994, Finley and others 1990). As shown in Figure 4, the total PCDD/F TEQ in the area 1 filterable solids sample (398 ng/kg), based on WHO TEFs for fish provided by Van den Berg and others (1998), was comparable to the median TEQ concentration measured in seven nearby surface sediment samples (430 ng/kg TEQ), but far less than the median TEQ of the subsurface sediments from these seven borings (5421 ng/kg TEQ). Conversely, in area 2, the mean PCDD/F TEQ in the filterable solids samples (7022 ng/kg TEQ) was much higher than the median TEQ in four nearby surface sediment concentrations (311 ng/kg TEQ) and more comparable to the median of the subsurface sediment concentrations (2297 ng/kg TEQ).

Estimated Body Burden Increases

As shown in Table 4, estimated metal body burdens from water column uptake in some cases exceed background metal concentrations. Specifically, estimated body burdens of copper in area 2 and zinc in areas 1 and 2 exceed the mean concentrations of these metals measured in five composited mummichog samples collected from the Passaic River in 1995 (BB&L 1999). Estimated body burdens of lead are far lower than recently measured concentrations.

As shown in Figure 5, the total body burden TEQs (PCBs and PCDD/Fs) from water column uptake are expected to reach steady-state concentrations after approximately six months. Estimated blue crab body burdens are slightly higher than mummichog due to the higher whole-body lipid content.

As shown in Figure 6, the mummichog TEQ body burden associated with food web uptake is estimated to be much greater (80 to 380-fold) than the contribution from water uptake, even after steady-state water uptake conditions have been reached (one year of water column exposure). The measured TEQ in the 1995 composited mummichog samples (57 ng/kg) is greater than the estimated steady-state body burdens associated with water column uptake in area 1 (0.54 ng/kg) and area 2 (53 ng/kg) but is less than the estimated body burdens associated with the total food web uptake (200 ng/kg in area 1; 4290 ng/kg in area 2). The estimated body burdens associated with food web uptake in area 2 are much higher than the maximum chronic critical body residue value of 450 ng/kg TEQ used in the Screening Level Human Health and Ecological Risk Assessment for the Passaic River Study Area (ChemRisk 1995). All other estimated concentrations are below this value. As expected, estimated TEQ body burdens (for both water column uptake and sediment uptake) are higher in area 2 than area 1. The PCDD/F congeners comprised the majority of the total TEQ in both areas 1 and 2 (which is also true for the 1995 mummichog data).

Discussion

The 1999 Passaic River barge removal activities resulted in an increase in turbidity and water column

concentrations of numerous chemicals. In area 1, where a single barge was removed, there was little change in physical parameters (turbidity) over time, and a comparison of upstream and downstream chemistry samples showed a moderate (approximately threefold) increase in chemical concentration. In area 2, where several barges were removed, the increases in turbidity and chemical concentrations were much higher. For example, turbidity levels increased to levels beyond the EPA maximum allowable criteria of 50 NTU in area 2 but not in area 1 (Figure 3). In general, turbidity levels returned to baseline conditions within 24–48 h of cessation of barge removal activities (Figure 3). Chemistry samples were not collected before or after barge removal in either area (Figure 2), but it is reasonable to conclude that the temporal nature of the increases roughly correlated with the changes in turbidity (given that metals, PCBs, and PCDD/Fs are primarily sediment-bound). In short, the barge removal activities resulted in a transient spike in turbidity and water column chemistry that was consistent with the degree of sediment disturbance (number of barges removed).

As shown in Tables 1 and 3, concentrations of PCDD/F TEQs and metals in several unfiltered water samples significantly exceeded water quality criteria established by NJDEP and US EPA (NJDEP 1998, US EPA 1999b). In particular, the mean PCDD/F TEQs exceeded the New Jersey and US EPA criteria of 0.014 pg/liter by more than 1000- and 10,000-fold in areas 1 and 2, respectively. While the increases in water column chemistry were transient, it is reasonable to expect that more extensive sediment disturbance activities in the Passaic River, such as dredging, could yield sustained increases in water column concentrations at levels similar to those measured here.

The PCDD/F TEQ concentrations in filterable solids also exceeded applicable criteria. As shown in Table 2, PCDD/F TEQ concentrations in filterable solids ranged from 398 to 8352 ng/kg, concentrations that exceed many of the sediment TEQ values that have typically been considered to be health-protective by different regulatory agencies (approximately 10–1000 ng/kg, depending on the agency and the nature of the receptor). PCDD/F concentrations in the barge removal areas tend to increase with depth (Figure 4). It is therefore interesting to note that in area 1, the PCDD/F TEQ concentration in the filterable solid sample (410 ng/kg) was similar to the concentrations measured in nearby surface sediments (mean = 430 ng/kg), while in area 2 the filterable solids concentrations (mean = 7050 ng/kg) was much higher than concentrations in surface sediments (mean = 310 ng/kg) and more similar to concentrations in subsurface sediments (mean = 5421 ng/kg). This is consistent with the fact that turbidity levels in area 2 were only about twice those in area 1, yet the water column concentrations of PCDD/Fs were much higher in area 2 than area 1. These observations suggest that in area 2 subsurface sediments comprised a majority of the suspended particulates in the water column. Subsurface sediments throughout the Passaic River are known to be highly contaminated with a wide range of chemicals, and sediment coring studies have shown that the chemical concentrations increase several-fold with depth (e.g., Finley and others 1990, Wenning and others 1994). The measurements obtained in this analysis suggest that even relatively minor, short-term activities are capable of suspending and dispersing the highly contaminated subsurface sediments.

As shown in Figure 5, total TEQ (PCBs and PCDD/Fs) biouptake from the water column is estimated to reach steady-state conditions after approximately six months. It is not inconceivable that dredging activities in the Passaic River could extend for this duration because of the large sediment volumes involved (US ACE 1996), and therefore the body burden estimates presented here are reasonable from an exposure-duration perspective. Clearly, appropriate controls such as booms and silt curtains will minimize the degree of sediment dispersal to points beyond the area of sediment disturbance; but even with total control of sediment dispersal, there may still be significant impacts to resident biota in the immediately impacted area. If the water column chemistry collected in area 2 is representative of the concentrations that typically exist in immediately impacted areas in the Passaic River, then it is almost certain that long-term sediment disturbance in the Passaic River will have a measurable impact on resident biota in the immediately impacted areas.

In this analysis it was assumed that the suspended sediments would settle and create a new surface sediment layer with chemical concentrations equivalent to those in the suspended sediments (in this case, the filterable solids). Clearly, there is a significant degree of uncertainty in this assumption. For example, the filterable solids samples were a snapshot and may not be entirely representative of the concentrations in the suspended particulates. Furthermore, it is admittedly simplistic to assume that suspended sediments will deposit homogenously in the manner assumed here. Still, although we are unaware of any studies that have explicitly examined this possibility, it is reasonable to suggest that, if highly contaminated subsurface sediments are suspended into the water column, the postdisturbance chemical concentrations in surface sediments could be

higher than before the activities were initiated (US EPA, 1997) and this could result in increased body burdens via food web bioaccumulation. The results in Figure 6 suggest that food web-related body burden increases in PCDD/F TEQ could be substantial and much greater than those associated with partitioning from the water column, under the assumptions used here. Figure 6 also indicates that in both areas 1 and 2, food web-related body burden increases would be much greater than the body burdens that already exist in the Passaic River, and this is simply due to the fact that the PCDD/F TEQ concentrations in the filterable solids were much higher than the PCDD/F TEQ concentrations currently present in surficial Passaic sediments. Finally, Figure 6 also indicates that the increased body burdens would exceed a chronic critical body residue of 450 ng/kg TEQ [see ChemRisk (1995) for derivation of this value]. In short, the most significant impact associated with disturbance of sediments in the Passaic River may be postdisturbance bioaccumulation from surface sediments.

Dredge-related body burden increases are site-specific and depend upon the degree of sediment contamination, the amount of sediment suspended, the duration of the disturbance, the species of interest, etc., and therefore it is difficult to directly compare these results to other published findings. Still, the estimated body burdens presented here are consistent with other studies that have measured food web impacts as a result of sediment disturbance. For example, Otto and other (1996) measured PCB uptake in caged rainbow trout during dredging of the St. Lawrence River at Massena, New York, USA. The fish were placed outside the silt curtain enclosing the dredged area. There was a reported increase in total PCB body burden of 117- and 34-fold after 21 and 42 days of dredging, respectively, which is consistent with the orders of magnitude increases in PCB and PCDD/F TEQ mummichog body burdens estimated in this analysis. Rice and White (1987) measured PCB uptake in caged fish before, during, and up to six months after dredging of the Shiawassee River, Michigan, USA. Moderate (about twofold) increases in PCB body burdens were measured 6.8 miles downstream six months after dredging had ceased. Interestingly, PCB concentrations in water were not elevated after-dredging, which indicates that the increased body burdens after dredging were from the food web (i.e., from sediment fallout) rather than from the water column.

It is important to note that sediments, particularly subsurface sediments, in other areas of the Passaic River contain chemical concentrations much higher than in the areas where the barge removals occurred. It has been suggested by some that sediment dredging is a cost-effective and technologically feasible remedy for the lower 6 miles (upstream of the confluence with Newark Bay) of the river. However, even the most refined and controlled dredging operation will likely suspend and disperse more sediment, and for longer periods of time, than the simple removal of partially buried barges. In summary, the increased turbidity and water column chemistry measured in this analysis, and the accompanying projections of increased body burdens in resident biota, are likely to far underestimate the magnitude of the impacts that would occur during a dredging operation in the Lower Passaic River. We suggest that further studies of actual food web impacts during Passaic sediment disturbance, particularly field studies using resident biota, should be considered before any remedies involving dredging can be seriously discussed. It may also be possible to construct reasonably accurate predictive models from the available field data, and these could be used to estimate food web impacts for defined sediment disturbance scenarios

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