PHASE 2 REPORT- REVIEW COPY
FURTHER SITE CHARACTERIZATION AND ANALYSIS

VOLUME 2E - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBs REASSESSMENT RI/FS

AUGUST 1999

For

U.S. Environmental Protection Agency
Region II
and
U.S. Army Corps of Engineers
Kansas City District

Book 3 of 3
Appendices

TAMS Consultants, Inc.

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VOLUME 2E- BASELINE ECOLOGICAL RISK ASSESSMENT
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SITE DESCRIPTION AND CHARACTERIZATION

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APPENDIX A
SITE DESCRIPTION AND CHARACTERIZATION

The Hudson River PCBs Reassessment Site extends for 40 river miles (RM) in the Upper Hudson River and an additional 160 RM in the Lower Hudson River. The size of this Superfund site and the varied habitats found along the river required that representative sampling locations be selected for the baseline ecological risk assessment (ERA). These locations were selected based on:

Reconnaissance surveys performed in October 1992 and May 1993;
PCB field-screening in the Thompson Island (TI) Pool (see Appendix B);
Historical data;
Phase 1 Report (USEPA, 1991);
High-resolution coring program (USEPA, 1997); and
Hudson River Significant Tidal Habitats Report (New York State Department of State, 1990).

Nineteen stations were selected for sampling for the August 1993 ecological field sampling program. Seven stations were sampled for sediment, benthic invertebrates and fish; three sites were sampled for benthic invertebrates and sediment; and the remaining nine stations were sampled for sediment and fish. The locations of the Upper and Lower Hudson sampling stations are shown in Figures A-1 and A-2, respectively. The sampling station descriptions provided below are based on field observations and published information.

Upper Hudson Stations

A.1 RM 203.3/Station 1: Background Station above Sherman Island Dam

The background station (Station 1) is located approximately 15 feet (ft) off the western shore of an unnamed, undeveloped, island between the Sherman Island Dam and the Feeder Dam. Previous sampling and analyses done for the high-resolution coring program had shown the sediments to be relatively clean, with only low concentrations of PCBs detected (USEPA, 1997). Sediment and fish were sampled at this station.

The island at the background station is undeveloped and forested with temperate deciduous trees. Residential houses are found on the western shore of the Hudson River. Aquatic grasses and water lilies are present at the station. Aside from the Sherman Island power plant, no major industries are present along the shore. The sediments appeared clean, with normal odors and no deposits other than organic matter and leaf detritus.

A.2 RM 196.9/Station 20: "Canoe Carry" Below Remnant Deposits

The area around the “canoe carry” (Station 20) is heavily wooded. The river bottom is dominated by a mixture of boulders and rocks with occasional deposits of fine sediments. Rocks are generally covered with algae. Sediment samples were collected from three locations.

A.3 RM 194.1/Station 2: Rogers Island

The Rogers Island Station (Station 2) is located off the eastern shore of the Hudson River across from Rogers Island. Sampling took place near the mud flats off the Champlain Canal spillway, approximately 0.25 mile upriver from the Ft. Edward sewage treatment plant outfall. A strong organic odor was present, originating from the direction of the waste treatment plant. However, the sediments did not have a strong odor associated with them. A moderate sheen of oil was present at the surface of the sediment. Aquatic vegetation and killifish were present and the water was slightly turbid.

A.4 RM 191.5/Station 3: South of Snook Kill/ TI Pool

The Snook Kill Station (Station 3) is located on the eastern shore of the river south of Snook Kill. No oils or strong odors were associated with the sediment collected at Station 3. Large wood chips and water celery (*Vallisneria americana*) were observed at the sampling area. Land use around the station is a mixture of residential houses and agricultural fields.

A.5 RM 189.6/Station 4: Griffin Island/TI Pool

The Griffin Island Station (Station 4) is located on the western shore of the Hudson River, south of Griffin Island. Although the western side of Griffin Island was relatively navigable in May, by August water chestnuts (*Trapa natans*) had completely overgrown the area, preventing boat access to the western side of the island. Rip-rap has been placed along the western bank below Griffin Island. No unusual odors or deposits were observed in the sediment, but a slight oil sheen was seen at the surface of the water. Samples were taken from the southern border of the water chestnut mat, which is considered to be good habitat for fish and benthic invertebrate species. The predominant land use in the area is residential.

A.6 RM 189.0/Station 5: Canal Divider/TI Pool

The Canal Divider Station (Station 5) is located on the western side of the concrete canal divider that separates the mainstem of the Hudson River and the Champlain Canal above the TI Dam. Aquatic vegetation present included water chestnuts and water lilies (Nymphaeaceae). No unusual sediment odors or deposits were observed, but an oil sheen was seen at the surface of the sediment.

A.7 RM 188.7/Station 6: Western TI Dam/TI Pool

The Western TI Dam Station (Station 6) is located off the west bank of the Hudson River, slightly above the dam. The bank is approximately 50 ft from River Run Road, which runs parallel to the western shore of the river. Some erosion was noted near the shoreline, possibly resulting from the spring runoff. The vegetation at this station was dominated by pickerelweed (*Pontederia cordata*), arrow-arum (*Peltandra virginica*), and other aquatic grasses. Wood chips were present in the sediment and there was a slight sheen of oil on the surface.

A.8 RM 188.5/Station 7: Eastern TI Dam/TI Pool

The Eastern TI Dam Station (Station 7) is located on the eastern shore of the Hudson River, approximately 1,000 ft upriver from the TI Dam. The bank of the river is forested, and abundant emergent and submergent vegetation are found along the river's edge. An anaerobic odor was present in the sediment and there was a slight sheen of oil on the sediment surface.

A.9 RM 169.5/Station 8: Stillwater Pool

The Stillwater Pool Station (Station 8) is located approximately 15 miles below the TI Pool, about 500 yards (yds) upriver of a small scale junkyard. The surrounding area has both residential and commercial land-use. There were no noticeable odors or oils present at the site.

A.10 RM 159.0/Station 9: Undeveloped Island Above Lock 1 Dam

Station 9 is located off the southern shore of an unnamed, undeveloped, island near Campbell Island. The eastern shore is primarily used for residential housing (low-density). There were some fallen trees near the shoreline where sampling took place. The sediment had a slight sheen of oil and the odor from the sediments indicated anaerobic conditions.

Lower Hudson Stations

A.11 RM 143.5/ Station 10: South Albany Turning Basin

The South Albany Turning Basin (Station 10) was formerly used for large boats to turn around to return downriver. It is no longer used for boat traffic; however, the area is heavily industrialized with the remains of a pier at the southern end. A strong petroleum odor was present at this station and oil slicks were seen on both the water and sediment. Vegetation, primarily water celery, was observed at the edges of turning basin. The dark brown water at this station was slightly turbid.

A.12 RM 137.2/ Station 11: Binnen Kill Below Shad Island

Sediment samples were taken from the mouth of the Binnen Kill (Station 11), located at the southwestern end of Shad Island. Shad Island is located on the western shore of the Hudson River, across from Castleton-on-Hudson. Shad Island and nearby Schermerhorn Island cover approximately 1,000 acres of riverine littoral zones, freshwater wetlands, floodplain forest, cliffs, tributary streams and active agricultural lands (NYSDOS, 1990). Portions of the habitat have been modified by dredged spoil disposal. No unusual sediment odors or deposits were noted. A slight sheen of oil was seen on the sediment, and the water was slightly turbid, probably owing to recent rainfall.

The Binnen Kill, which enters the Hudson River at the base of Shad Island, provides spawning and feeding habitat for American shad (*Alosa sapidissma*), blueback herring (*A. aestivalis*), alewife (*A. pseudoharengus*), and white perch (*Morone americana*), as well as resident freshwater species (NYSDOS, 1990). Terrestrial portions of the area provide quality habitat for a variety of upland wildlife species, including whitetail deer (*Odocoileus virginianus*), eastern cottontail (*Sylvilagus floridanus*), ruffled grouse (*Bonasa umbellus*), and many passerine bird species. The small wetland areas in and around Shad and Schermerhorn Islands support limited numbers of waterfowl and furbearing mammals.

Significant freshwater intertidal mudflats and freshwater tidal marshes have been documented in the area (NYSDEC Natural Heritage Program, 1999).

A.13 RM 122.4/Station 12: Stockport Creek and Flats

Stockport Creek and Flats (Station 12) is the northernmost site of the four sites comprising the Hudson River National Estuarine Research Reserve (NERR). This 1,543 acre reserve is located on the eastern shore of the Hudson River in Columbia County, covering about five miles from north to south. Samples were taken approximately 0.5 miles upriver of Stockport Creek, off the eastern shore of Stockport Middle Ground. Stockport Creek is a tributary of major significance in the Hudson River Estuary. Large expanses of nearly all coastal related habitats occur in the Stockport Creek and Flats area. The Hudson River is entirely freshwater at Stockport Flats with an average tidal range of 4.0 feet.

Stockport Flats is primarily freshwater tidal wetlands. Subtidal shallows support communities of submerged plants, with water celery most abundant. The tidal marshes are dominated by narrow-leaf cattail (*Typha angustifolia*), wild rice (*Zizania aquatica*), spatterdock (*Nuphar advena*), and pickerelweed. Tidal and floodplain swamps are dominated by mature mixed deciduous forest characteristic of river bottoms. During the summer months water chestnuts dominate the shallow regions of Stockport Flats. The abundance of water chestnuts precluded entry into some areas of Stockport Flats.

Stockport Creek is formed by the confluence of Kinderhook Creek and Claverack Creek, and provides approximately three miles of accessible waters for fish spawning. Stockport Flats is a spawning and/or nursery ground for anadromous and freshwater fish species including alewife, blueback herring, American shad, rainbow smelt (*Osemerus mordax*), striped bass (*Morone saxatilis*) and white perch. Most anadromous species enter the area to spawn between April and June. The adults leave the area soon after spawning, and within several weeks the eggs have hatched and larval fish begin moving downstream to nursery areas. Smallmouth bass (*Micropterus dolomieui*) occur in Stockport Creek throughout the year. Adult bass move into the upper section of the creek in May or early June to spawn and return to the main river as water temperatures rise.

Stockport Creek and Flats provide valuable feeding and resting habitat for large concentrations of waterfowl during the fall and spring migrations. Approximately 10,000 canvasback ducks, along with various other waterfowl species, have been reported in the area during seasonal migrations. The area is also an important wintering ground, especially for redhead (*Aythya americana*) and canvasback (*A. valisineria*) ducks. Wetland areas here provide habitat for various marsh nesting birds, including the green heron (*Butorides striatus*), American bittern (*Botaurus lentiginosus*), black duck (*Anas rubripes*), wood duck (*Aix sponsa*), Virginia rail (*Rallus limicola*), sora (*Porzanz carolina*), fish crow (*Corvus ossifragus*), and marsh wren (*Cistothorus palustris*). Wading, shore and song birds use Stockport Creek and Flats for feeding.

The NYSDEC Natural Heritage Program has classified Stockport Flats as having important freshwater intertidal shore and freshwater tidal marsh communities and providing important habitat for anadromous fish and waterfowl (NYSDEC Natural Heritage Program, 1999).

Rare bird species found at Stockport Flats include bald eagles (*Haliaetus leucocephalus*) and osprey (*Pandion haliaetus*). Rare plants found in this area include the heart leaf plantain (*Plantago cordata*), estuary beggar ticks (*Bidens bidentoides*), kidneyleaf mud-plantain (*Heteranthera reniformis*) and spongy arrowhead (*Sagittaria calycina var. spongiosa*).

A.14 RM 113.8/Station 13: Roger's Island

Roger's Island (Station 13), located approximately two miles southwest of the city of Hudson, encompasses approximately a 650-acre area, including tidal forested wetlands on Roger's Island. The island proper is a wildlife management area administered by the NYSDEC, and includes both tidal forested area and extensive mudflats and littoral zones (NYSDOS, 1990). Samples were taken from the western shore of the island, in a cove surrounded by forest. No usual odors or oils were present in the sediment.

A heavy concentration of American shad use the littoral zone around Roger's Island for spawning. This area is also used as for spawning and as a nursery and feeding area by striped bass, alewife, blueback herring, white perch, and a variety of resident freshwater species.

The interspersion of forest and wetland cover provides favorable nesting areas for green heron, black duck, wood duck and many passerine bird species. The mudflats and littoral zones in this area provide valuable feeding and resting habitat for large concentrations of waterfowl during spring and fall migrations. Approximately 10,000 canvasbacks, along with other waterfowl, shorebirds, and passerine species have been reported in the area during seasonal migrations. When open water is available, this area also provides an important waterfowl wintering area in the upper Hudson Valley region, especially significant for redhead and canvasback duck.

Several important waterfowl food plants, including the golden club (*Orontium aquaticum*), are found at Roger's Island. The golden club is a regionally rare plant species that has declined from 25 acres to less than 10 acres here.

A.15 RM 100.0/Station 14: Tivoli Bays

Tivoli Bays, a 1,722-acre site in Dutchess county, is comprised of two large coves, North and South Tivoli Bays. Tivoli Bays are one of the four sites comprising the Hudson River NERR. North Tivoli Bay is predominantly intertidal marsh, with a well-developed network of tidal creeks and pools. South Tivoli Bay is a large, shallow cove with mudflats exposed at low tide.

Sediment, benthic invertebrate and fish samples were taken from North Tivoli Bay, off the northeastern tip of Cruger Island. The surrounding land is a wildlife preserve with railroad lines running through it.

Freshwater tidal marshes at North Tivoli Bay are dominated by narrowleaf cattail, spatterdock, and invading purple loosestrife (*Lythrum salicaria*), and common reed (*Phragmites communis*). Subtidal shallows support communities of submerged plants similar to those described for Stockport Flats, and fresh intertidal mudflats and shore communities are also present. South Tivoli Bay is dominated by water chestnut. Rare species found in the Tivoli Bays complex include the heartleaf plantain (*Plantago cordata*), golden club, ovate spikerush (*Eleocharis ovata*), Parker's pipewort (*Eriocaulon*), Eaton's bur-marigold (*Bidens* sp.), estuary beggar-ticks, swamp lousewort (*Pedicularis lanceolata*), and a rare species of panic grass (*Panicum* sp.).

The Tivoli Bays site is a spawning and/or nursery ground for a variety of anadromous and freshwater fish species. Anadromous fish include striped bass, alewife and blueback herring, while resident freshwater species include largemouth bass (*Micropterus salmoides*), smallmouth bass, white perch and various minnows (Cyprinidae). Regionally rare species found in Tivoli Bays include the American brook lamprey (*Lampetra appendix*), central mudminnow (*Umbra limi*), northern hogsucker (*Hypentelium nigricans*), and bridle shiner (*Notopis bifrenatus*). A large snapping turtle (*Chelydra serpentina*) population exists in and around North Tivoli Bay.

Populations of least bittern (*Ixobrychus exilis*), Virginia rail, and marsh wren regularly use Tivoli Bays for breeding, while the sora rail, common moorhen, and king rail use the area less

frequently. Waterfowl use this area extensively during migration for resting and feeding, including dabbling ducks in the marshes and diving ducks in the river shallows.

Rare animals that have been sighted at Tivoli Bays include osprey, bald eagle, king rail (*Rallus elegans*), least bittern, golden eagle (*Aquila chrysaetos*), map turtle (*Graptemys geographica*), and American brook lamprey.

A.16 RM 88.9/Station 15: Esopus Meadows

Esopus Meadows (Station 15) is a relatively large, undisturbed area of shallow, freshwater tidal flats located adjacent to a natural deepwater habitat. Because of the adjacent deepwater habitat, the area is not subject to disturbance from periodic maintenance dredging.

Sampling was conducted off Esopus Meadows Point on the western shore of the Hudson River. The surrounding land use is predominantly residential. In August 1993, water chestnut choked much of the water bordering the western shore. Samples were taken at the southern edge of the water chestnut mat.

The shallow subtidal aquatic beds in Esopus Meadows provide spawning, nursery, and feeding habitats for anadromous species such as striped bass, American shad and white perch. Resident fish species include largemouth bass, carp, brown bullhead, yellow perch and shiners. Concentrations of spawning anadromous fish generally occur in the area between mid-March and July, with substantial numbers of young of the year fish remaining well into the fall (October-November). Esopus Meadows is also an important feeding area for populations of the endangered shortnose sturgeon (*Acipenser brevirostrum*) wintering in the adjacent deepwater channel, especially in spring.

Significant concentrations of waterfowl occur in the Esopus Meadows area. Dense growths of submergent vegetation provide valuable feeding areas for many duck species of ducks, and are especially important during spring (March-April) and fall (mid-September-early December) migrations. Concentrations of diving ducks, such as scaups (*Aythya* sp.), redhead, canvasback, common goldeye (*Bucephala islandica*), and mergansers (Merginae), are regularly found in this area. This open water area is also used by dabbling ducks (Anatinae), including mallard (*Anas platyrhychos*), black duck (*A. rubripes*), and blue-winged teal (*A. discors*), especially during calm weather. Much of the area provides refuge from hunting pressures in shoreline areas.

A.17 RM 58.7/Station 16: Plum Point, North of Moodna Creek

Sediment and fish samples were taken off Plum Point (Station 16), on the western shore of the Hudson River. The sediment had no unusual odors or attributes. Nearby Moodna Creek is one of five major tributaries that empties into the lower portion of the Hudson estuary. The fish and wildlife habitat is an approximate three and one-half mile segment of Moodna Creek. The marsh at the mouth

of Moodna Creek is also significant for rare plants and natural communities including brackish intertidal mudflats and brackish tidal marsh (NYSDOS, 1990).

Moodna Creek is an important spawning area for anadromous fish including alewife, blueback herring, smelt, white perch, tomcod (*Microgadus tomcod*), and striped bass. The extensive flats at the creek mouth provide spawning and nursery habitat for these species. Anadromous species generally enter the stream to spawn between April and June and the adults leave the area shortly after spawning. The eggs hatch within several weeks and the larval fish begin moving downstream to nursery areas within the Hudson River. Moodna Creek also supports a substantial warmwater fish community year-round. Resident species include largemouth bass, bluegill (*Pomatomus saltatrix*), pumpkinseed (*Lepomis gibbosus*), and brown bullhead (*Ictalurus nebulosus*).

The bay and flats area at the mouth of Moodna Creek comprise a diverse and productive tidal freshwater wetland. Probable or confirmed breeding bird species in the area include green heron, least bittern, Canada goose (*Branta canadensis*), mallard, black duck, wood duck (*Anas sponsa*), Virginia rail, spotted sandpiper (*Actitis macularia*), belted kingfisher (*Megaceryle alcyon*), fish crow, marsh wren (*Cistothorus palustris*), common yellowthroat (*Geothlypis trichas*), hooded warbler (*Wilsonia citrina*), red-winged blackbird (*Agelaius phoeniceus*), downy woodpecker (*Picoides pubescens*), yellow-shafted flicker (*Colaptes auratus*), eastern kingbird (*Tyrannus tyrannus*), and swamp sparrow (*Melospiza georgiana*). Locally significant concentrations of herons, waterfowl and shorebirds are found in the area, particularly during spring and fall migrations. Moodna Creek is also reported to be a major crossing point for raptors migrating through the Hudson Valley, along the northern slope of the Hudson Highlands.

A.18 RM 47.3/Station 17: Iona Island

Iona Island (Station 17) is a 556-acre bedrock island in the middle of the Hudson Highlands, bordered to the west and southwest by Salisbury and Ring Meadows, two large tidal marshes, and a series of shallows and mudflats. In addition to being designated as one of the four sites comprising the Hudson River NERR, it is also registered as a National Natural Landmark with the US Department of the Interior. Iona Island is part of Bear Mountain State Park in Rockland County.

The salinity of the Hudson River at Iona Island ranges from slightly brackish (6 parts per thousand [ppt]) to freshwater. Most of the area is very shallow, ranging from 1 to 3 ft deep. Sediments in the tidal marshes consist of peat and silt.

Sampling was conducted off the western shore of Round Island, southeast of Iona Island. No sediment oils or odors were noted. The surrounding land use is recreational consisting of a section of Bear Mountain State Park, with the exception of the railroad along the western shore of the Hudson River.

Vegetation is dominated by narrowleaf cattail, with moderate amounts of common reed and swamp rose mallow (*Hibiscus palustris*). The island and mainland slopes are covered with deciduous forest, with abundant red oak (*Quercus rubra*), chestnut oak (*Q. prinus*) and pignut hickory (*Carya glabra*). Substantial areas of Iona Island's tidal shallows are bare mud, although water celery and other submerged plants also occur there. Brackish intertidal mudflats, brackish water tidal marsh, and freshwater tidal marsh are all present.

Iona Island is recognized as a waterfowl concentration area. Many heron and shorebirds feed in and around the marshes, and many bird species breed within the site. Probable or confirmed breeding species include green-backed heron, least bittern, Canada goose, mallard, wood duck, Virginia rail, sora rail, common moorhen (*Gallinula chlorops*), spotted sandpiper, belted kingfisher, marsh wren, red-winged blackbird and swamp sparrow. Muskrat (*Ondatra zibethica*), mink (*Mustela vison*), amphibians (in non-tidal areas), snapping turtles, and blue claw crab (*Callinectes sapidus*) are also found here. Offshore shallows are used for spawning and/or nursery for anadromous and resident freshwater resident fishes including alewife, blueback herring, white perch, striped bass, banded killifish (*Fundulus diaphanus*) and mummichog (*F. heteroclitus*).

Rare animals found at or offshore Iona Island include least bittern, bald eagle, golden eagle, osprey, peregrine falcon (*Falco peregrinus*), shortnose sturgeon (*Acipenser brevirostrum*), and five-lined skink (*Eumeces fasciatus*).

A.19 RM 25.8/Station 18: Piermont Pier

Piermont Pier (Station 18) is part of Piermont Marsh, a 1,017-acre site located slightly south of the Tappan Zee Bridge on the western shore of the Hudson River. Most of the marsh is part of the Tallman State Park and is one of the four sites comprising the Hudson River NERR. The salinity of the Hudson River at Piermont Pier is generally considered brackish, although it ranges from freshwater to 12 ppt. The average tidal range is 3.2 ft. The sediments present at the station are peat and organic silt. These deposits are at least 40 ft deep in the western part of the marsh, which has been developing for nearly 5,000 years.

Sampling was conducted off the northern shore of Piermont Pier. The shoreline is a mixture of common reed and smooth cordgrass (*Spartina alterniflora*), with a man-made breaker providing refuge against high waves. Shell fragments appeared to be more abundant here than other areas. No unusual odors or sediment textures were noted.

Piermont Marsh habitats include brackish tidal marsh, shallows, and intertidal flats. Substantial parts of offshore shallows are bare mud, although submerged plants are also present. Water celery, curlyleaf pondweed (*Potamogeton crispus*), sago pondweed (*P. pectinatus*), and horned pondweed (*Zannichellia palustris*) are found in the shallows.

The mudflats are used extensively by herons and egrets. Waterfowl, wading birds and shorebirds feed in the area during migration. Large numbers of resident and breeding bird species, Fiddler crabs (*Uca minax*), blue crabs (*Callinectes sapidus*), resident fishes, and lesser numbers of furbearers (muskrat, mink and raccoon [*Procyon lotor*]), snapping turtles, and northern water snakes (*Nerodia sipendon*) are also present. The area is recognized as an anadromous fish concentration area. Probable or confirmed breeding bird species in the area include pied-billed grebe (*Podilymbus podiceps*), green heron, mallard, black duck, gadwall (*Anas strepera*), wood duck, American woodcock (*Philohela minor*), marsh wren, red-winged blackbird, and swamp sparrow. Concentrations of herons, waterfowl, and shorebirds occur in the tidal flats and shallows during spring (March-April) and fall (September-November) migrations, but the extent of use by these birds has not been documented.

Rare species found at Piermont Marsh include least bittern, osprey, golden eagle, peregrine falcon, and diamondback terrapin (*Malaclemys terrapin*).

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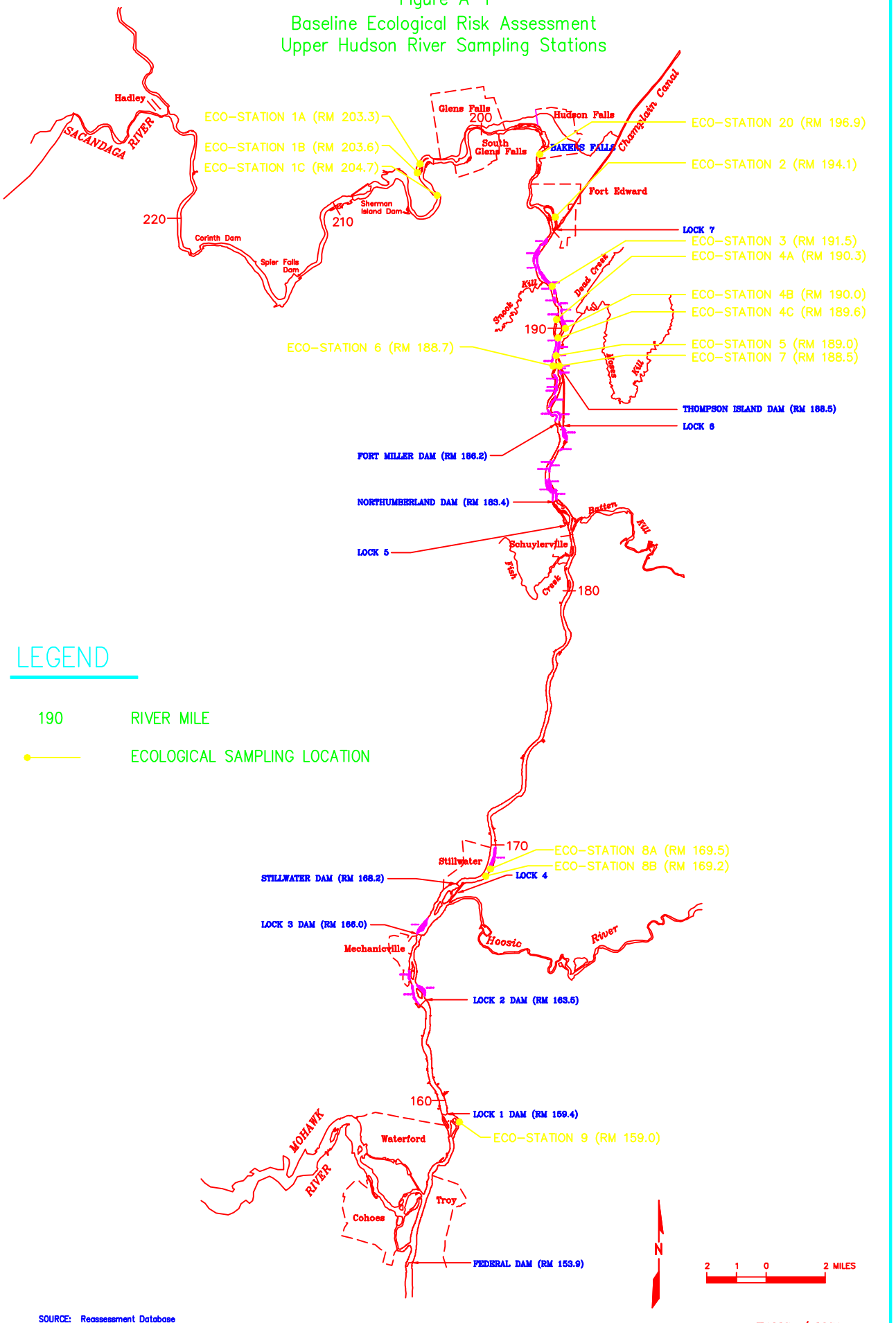
Figure A-1
Baseline Ecological Risk Assessment
Upper Hudson River Sampling Stations



SOURCE: Reassessment Database

TAMS / MCA

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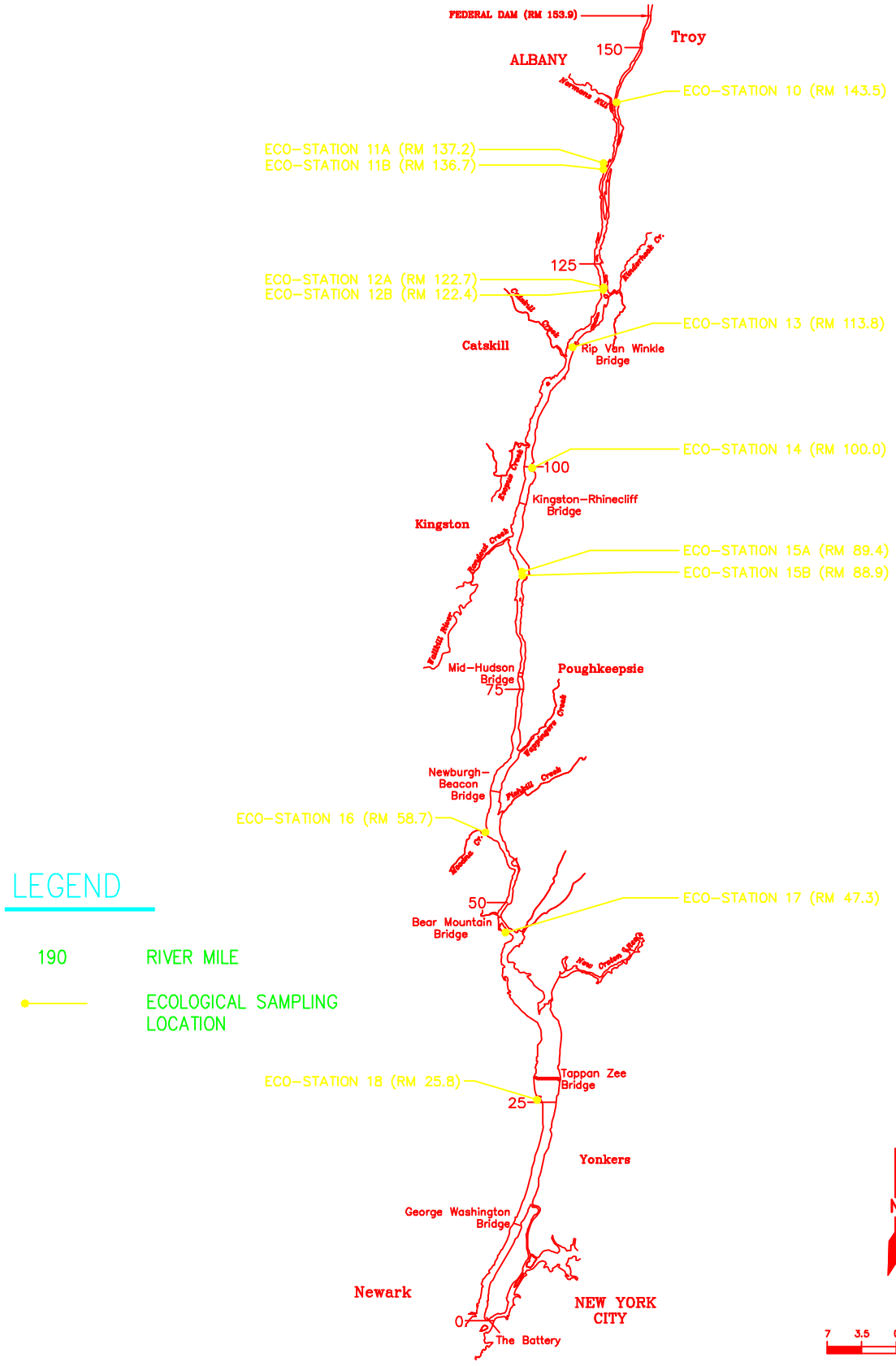
Figure A-2
Baseline Ecological Risk Assessment
Lower Hudson River Sampling Stations



SOURCE: Reassessment Database

TAMS / *MCA*

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APPENDIX B

ECOLOGICAL FIELD SAMPLING PROGRAM

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ECOLOGICAL FIELD SAMPLING PROGRAM

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ECOLOGICAL FIELD SAMPLING PROGRAM

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APPENDIX B
ECOLOGICAL FIELD SAMPLING PROGRAM

The Hudson River PCB Reassessment ecological sampling program was conducted to assess the effects of PCBs on aquatic communities in the Hudson River. Sediment, benthic invertebrates, and fish were collected and analyzed to correlate PCB concentrations in the sediment to concentrations in the biota and provide data for food chain modeling. Water column sampling was performed in a separate simultaneous field program (USEPA, 1997) and data pertaining to the ecological sampling period and locations were used. A matrix of samples taken at each station is provided in Table B-1.

The United States Environmental Protection Agency (USEPA) sampling effort was conducted from August 2 to 26, 1993. New York State Department of Environmental Conservation (NYSDEC) in conjunction with National Oceanic and Atmospheric Administration (NOAA) collected fish samples from each of the sampling stations from August 2 to September 1, 1993. Sampling started upriver at the background station (RM 203.3) and progressed downriver.

Each sampling station was triangulated using shoreline markers and a compass and recorded in the field log book. River mile designations for sampling locations were determined as follows:

> Upstream of RM 197.7 using USGS topographic quadrangles, using the county boundary lines to approximate the center of the Hudson River channel;
> Between RM 182 and RM 197.7 from the TAMS Hudson River centerline derived from TAMS bathymetric data;
> Between RM 153.9 and RM 182 from Hudson River Survey Maps (Normandeau Associates, Inc. 1976 to 1977); and
> Downstream of RM 151.7 were estimated from approximate 25-mile increment river mile tick marks on a digitized map of the Lower Hudson River.

The sediment, benthic invertebrate, fish, and water column sampling efforts are described below. A summary of the qualitative vegetation survey performed at each sampling station is also provided (subchapter B.5).

B.1 Sediment Sampling

The Phase 2B ecological sediment sampling was designed to provide sediment concentration data for the assessment of ecological risk to the biotic community. Surface sediment samples (top 5 cm) were collected from the same locations as the fish and invertebrates to provide an estimate of biological uptake of PCBs from the sediment. Pilot samples taken during the field reconnaissances in October 1992 and May 1993 indicated that most biological activity was found in the upper 5-cm (2-in) of the sediment. After discussion with other agencies, a surficial sampling depth of 5-cm was selected. The 5-cm sampling depth is less than the depth at which bioturbation can occur, but as the

ERA focuses on the exposure of the ecological community to PCBs the 5-cm sampling depth was used. Sampling locations are shown in Figures A-1 and A-2.

The sediment sampling was not intended to provide a definitive picture of PCB contamination along the Hudson River for feasibility study purposes. It does, however, examine a number of ecologically significant areas as determined by USEPA, NYSDEC, and NOAA. The high resolution coring technique (USEPA, 1997) was adapted to collect 2.5" diameter sediment samples. The coring apparatus was gently lowered into the sediment and pushed down. It was then slowly lifted up towards the water surface, and a cap was placed over the open core bottom prior to breaking the surface of the water. The core was removed from the water and the bottom of the sediment core was taped and placed upright in a holding rack. A total of ten (10) cores were collected at each station, excluding QA/QC samples. Two sequential 5-cm core sections were homogenized in a decontaminated stainless bowl and aliquots were taken for PCB congener, TOC, and metals analyses.

Grain size samples were obtained by using an Ekman Grab (6"x 6" x 6") with a clear acrylic liner. A stainless steel slicing plate was used to cut off the top 5 cm of sediment, which was then transferred into a 500-ml glass jar.

Sediments were analyzed for congener-specific PCBs, grain size, and total organic carbon. Temperature, pH, conductivity, and dissolved oxygen were measured as standard indicators of water quality conditions (Table B-2). Inorganic analytes on the USEPA Target Analyte List (TAL; Table B-3) were also analyzed in background and TI Pool samples.

B.1.1 PCB Screening in the Thompson Island (TI) Pool

Sediments and benthic invertebrate communities were collected at five locations in the TI Pool to determine if PCB concentrations affected community structure. PCB field screening was performed in the TI Pool to select locations with varying PCB concentrations using PCB RISc test field screening kits (Ensys, Inc., 1992; EPA draft Method 4020).

PCB field screening was performed for samples from nine locations within the TI Pool and a background area (RM 202.8). Samples were collected from the top 5 cm (2 in) of sediment on 5 August 1993 using an Ekman grab with a clear acrylic liner. Samples were placed into jars and stored in a cooler with ice.

The following morning (6 August 1993), samples were analyzed for PCBs using the PCB RISc Soil test (Ensys, 1992). Samples were tested for the presence of PCBs at three levels: 2 parts per million (ppm); 10 ppm; and either 40, 80, or 100 ppm.

The results of the TI Pool PCB field screening (Table B-4) showed that PCBs were present in all of the TI Pool test sediments. The majority of screening stations had PCB concentrations greater than 10 ppm, but less than 80 or 100 ppm. Screening station 2s was selected to provide a TI Pool site with relatively low PCB concentrations. Screening stations 1s, 3s ,4s and 8s were selected to assess

higher levels of contamination in the TI Pool. The habitat characteristics at these stations were also more similar than between the remaining samples. Screening stations 4s and 8s were considered to be the most appropriate stations for fish sampling, and were used as such. Stations were renumbered during the ecological field sampling program.

B.2 Water Column Sampling

Two distinct sampling studies were conducted as part of the Phase 2 sampling program. The Water-Column Transect Study examined instantaneous (i.e., one point in time) water-column concentrations. Each transect was conducted so as to follow, in a general fashion, the same parcel of water as it traveled down the Upper Hudson River. Examining each water parcel (i.e., transect) provided data on how the water-column inventory of PCBs varied as the river traveled down the Upper Hudson River basin. The water-column sampling schedule for a given transect event was estimated from a time-of-travel model, calibrated to USGS dye-study data, and the knowledge of instantaneous flow at the USGS Fort Edward telemeter gauge. Seven sampling events were conducted from January to September 1993, encompassing both low-flow and high-flow conditions.

Thirteen stations were sampled in the Upper Hudson River basin, including four tributaries, and four stations were sampled in the Lower Hudson River. Sampling began at the background station above the General Electric facilities near Glens Falls (RM 199.5) and continued down to the Lower Hudson River Estuary near Kingston (RM 77), covering a distance of over 120 river miles.

The Flow-Averaged Water-Column Sampling Study provided a measure of mean total PCB transport in the Upper Hudson River. Six 15-day sample collection events were conducted over a period of six months from April through September 1993. Daily samples were composited over a 15-day period to account for variations in river flow, suspended-matter load, sediment scour, and contaminant concentrations to determine longer-term averages of water-column conditions.

Flow-averaged samples were collected at the following four water-column transect locations:

RM 197.6, above Bakers Falls, upstream of the GE Hudson Falls source areas;
RM 194.6, the northern tip of Rogers Island, downstream of both GE facilities and the remnant deposits but above the TI Pool sediment source (NYSDEC Hot Spots 1 through 20 are found in this pool);
RM 188.5, the TI Dam, considered the downstream end of the TI Pool; and
RM 156.5, Waterford, considered the downstream end of the Upper Hudson basin (not including the Mohawk River).

In both Phase 2 water-column studies, the water samples were filtered in the field to determine PCB concentrations in both the "dissolved" and particulate phases. Additional detail on the sampling design, methodology, and analytical techniques can be found in the Phase 2A Sampling and Analysis/Quality Assurance Project Plan (SAP/QAP$_j$P) (USEPA, 1992a) for the water-column

transect study, and in the Phase 2B, Volume 1 SAP/QAP$_j$P (USEPA, 1992b) for the flow-averaged water-column sampling program.

B.3 Benthic Invertebrate Sampling

The two primary objectives of the Phase 2B Benthic Invertebrate Sampling were to examine benthic invertebrate community structure in the TI Pool and analyze benthic invertebrate PCB-congener body burdens (USEPA, 1993).

Community structure (i.e., species abundance and diversity) can serve as an indicator of the general health of a biological community. For example, the Phase 1 Report (USEPA, 1991) noted that a significant improvement in water quality in the Upper Hudson River from 1972 to 1977 and through 1988 led to greater species diversity and an increase of pollution intolerant species in the river. The effect of PCBs on the community structure of macroinvertebrates (i.e., organisms that are retained on a 0.5 mm mesh), has not been well-documented and therefore the macroinvertebrate community structure in the TI Pool was analyzed to determine if any effects could be measured.

An Ekman grab with an acrylic liner was used to collect benthic invertebrate samples under NYSDEC license number LCP92-499. The Ekman grab was slowly lowered to the river bottom. After tripping the closure mechanism the grab was gently pulled back up to the boat and water was allowed to drain out. The grab was placed into a plastic holding pan and the sample was visually examined to determine if a minimum uniform penetration depth of 5 cm was obtained. If the sample was determined to be inadequate, the apparatus was rinsed with river water and the procedure was repeated. Otherwise, the top 5 cm of sediment was placed into a wash bucket and samples were washed with river water until the sample was considered clean. The organisms and associated debris were then placed into precleaned 500-ml jars for analysis.

Four grabs were made for each benthic invertebrate replicate (i.e., sample). The first three grabs were used for the PCB congener analysis and the remaining grab was used to for the community analysis. Biomass in the Lower Hudson was generally low. Therefore, only three of the five replicates were analyzed for community structure at each station and the remaining two grabs were used to increase the macroinvertebrate mass for PCB body burden analyses.

Benthic macroinvertebrate communities were quantified by sorting and identifying to the lowest possible taxonomic level organisms from the top 5 cm of sediment. Five replicates of a 14 cm wide by 14 cm long by 5 cm deep (980 cm^3) washed sample of sediment were analyzed at each location.

Prior to sorting, the total benthic biomass (wet weight) of each sample was determined. After sorting, each major taxonomic group was weighed in order to establish relative contributions to the total biomass. A subset of discrete taxon samples with sufficient mass (i.e., yzed for PCB congeners.

Sediment sampling was conducted at the same time and locations as the benthic invertebrate sampling to examine the relationship between sediment PCB loads and macroinvertebrate body burdens. Both benthic invertebrates and sediment were taken from the top 5 cm of material, which was generally a single stratigraphic layer (i.e., one depositional zone). Samples were packed on ice and shipped to the appropriate laboratory for analysis within one day of collection with a completed, signed chain of custody form in each sample cooler.

B.4 Fish Sampling

NYSDEC and NOAA collected fish during the same time period as the sediment collection. Fish were collected from a total of 17 of the 20 sediment stations, eight in the upper river and nine in the lower river. Two basic categories of fish species were targeted (Table B-1):

> Resident (Type A) species, or specific life-history stages of species, that could be expected to be resident in the sample collection area for at least two months prior to collection.
>
> Mobile (Type B) species that were not assumed to remain in the sampling area for an extended period of time. Mobile fish were often larger, older specimens.

Fish were collected using beach seines and electroshocking. A 14 ft (4 m) aluminum skiff with a 25 horsepower motor was used to haul gear and personnel to sampling locations. NYSDEC personnel used a 19 ft (6 m) Boston Whaler to conduct some of the beach seine collections in the lower river. An 18 ft (5.5 m) aluminum flat-bottom electroshocking boat was used in several locations and was effective in sampling the outer edges of water chestnut beds and other areas that could not be effectively fished with a beach seine. A backpack electroshocker was used at the station just below Bakers Falls (RM 196.8) because of the relative inaccessibility of the area.

The primary gear used for collecting the Type A fish species was a fine-mesh beach seine 10 m (33 ft) long and 1.5 m (5 ft) deep, used by two people with chest-waders. Because of the small size of the seine and the very soft bottom at some stations, only selected areas could be fished. The seine was effective for sampling the fringes of dense water chestnut beds and in areas with shoreline debris. Table B-5 lists fish collection methods used for each sampling location.

At the time of collection, samples were assigned identification numbers, tagged, bagged, and labeled for later sorting. Species, date collected, collector names, location of collection, total lengths and weights were recorded on field collection record forms. Chain of custody sheets were maintained separately for each collection location and date. Samples from each collection station and date were grouped for PCB analysis by species and size. Samples of smaller fish (e.g., spottail shiner) generally represent composites of 10 individual fish. Larger fish (e.g., yellow perch) were either composited in small groups of three to five individuals, or analyzed individually, depending on the size and the number of fish of each species collected at each location.

Upon collection and as soon as processing was completed, all specimens were placed in coolers on ice or transported to freezer facilities. Fish were in frozen storage within 12 hours of collection. The main storage facility was a 17 cubic ft (0.5 cu m) chest freezer located in Salem, New York. Temperatures ranged from 0 °C to -27 °C), according to the freezer temperature log maintained during the course of the study. Freezer readings were begun on August 13, 1993 and continued until December 14, 1993. Temperatures were recorded whenever the freezer was opened to receive more fish samples or to prepare samples for shipment to analytical laboratories.

B.5 Vegetation Survey

During the field sampling effort, a baseline vegetative survey was performed at eighteen stations along the Hudson River. A plant ecologist conducted the survey by identifying dominant submergent and emergent vegetation observed in intertidal, bank, and upland areas, when possible. A list of all species identified throughout the field investigation is provided in Table B-6.

Vegetation was observed and documented to determine habitat similaritty between stations and obtain a general idea of the ecological "health" of the river. A variety of algae, crustaceans, snails, and insect larvae live on the surfaces of submerged plants, using them for food, habitat, and reproduction. Aquatic vegetation also provides feeding and nursery grounds for some fish species.

All the Upper Hudson River Stations (1 to 9) were classified as freshwater, nontidal wetlands. Similar plants were present at these stations, including nearly all the same dominant submergent plants. Emergent species were located at Stations 2, 6, 7 and 9. Vegetation observed on the river bank varied, but a majority of locations included silver maple *(Acer saccharinum)* and white ash *(Fraxinus americana).*

The most diverse sampling location (qualitative assessment) within the upper reaches of the Hudson River sampling was at Station 7 (RM 189) above the eastern TI Pool. The water slows within this bend of the river, just before the dam. This reduction in flow velocity allows fine-grained sediments to settle out, providing favorable conditions for plant growth.

In the Lower Hudson River the freshwater tidal reaches included Stations 10 to 15. Below the Troy Dam, the river is generally wider, less channelized, and enjoys a greater diversity of plant species. Stations 16 to 18 are located within brackish waters of the river, although salinity may vary seasonally. Saline species such as cord grass (*Spartina alterniflora*) were seen at Station 18 (RM 25.8), which has a salinity of approximately 9 parts per thousand (ppt).

The most prevalent aquatic plant noted was water chestnut (*Trapa natans*), which was abundant along nearly the entire river. Many places were inaccessible due to the thick mats of water chestnut encountered.

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Table B-1
Hudson River PCB Reassessment Phase 2 Ecological Sampling

Station	River Mile	Samples Collected			
		Sediment	Benthic Invertebrates.	Resident Fish	Mobile Fish
1a	203.3	x		x	x
1b	203.6			x	x
1c	204.7			x	x
2	194.1	x		x	x
3	191.5	x	x	x	
4a	190.3			x	x
4b	190.0			x	x
4c	189.6	x	x	x	x
5	189.0	x	x		
6	188.7	x	x		
7	188.5	x	x		
8a	169.5	x			
8b	169.2			x	x
9	159.0	x		x	
10	143.5	x		x	
11a	137.2	x			
11b	136.7			x	
12a	122.7			x	
12b	122.4	x	x		
13	113.8	x		x	x
14	100.0	x	x	x	
15a	89.4			x	x
15b	88.9	x	x		
16	58.7	x	x	x	x
17	47.3	x	x	x	
18	25.8	x	x	x	
20	196.9	x		x	x

Note: Only sampling performed or funded by the USEPA is listed in this table. Other agencies, such as NOAA, NYSDEC, and USFWS, have independently performed ecological sampling in the Hudson.

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Table B-2

Field Sampling
Water Quality Measurements

Station	Temperature	pH	Conductivity	Dissolved	Salinity
	oC		uMHOS		ppt
1.0	25.0	7.3	52.0	R	0.0
2.0	25.0	7.4	110.0	R	0.0
3.0	24.1	7.0	92.0	8.8	0.0
4.0	25.0		90.0	8.3	0.0
5.0	24.0	7.2	98.0	7.2	0.0
6.0	24.0	6.8	95.0	8.4	0.0
7.0	25.0	7.4	96.0	8.1	0.0
8.0	26.0	8.2	125.0	8.0	0.0
9.0	25.0	6.9	122.0	8.1	0.0
10.0	25.1	7.1	225.0	7.8	0.0
11.0	25.0	6.8	220.0	8.2	0.0
12.0	26.4	6.3	249.0	8.0	0.0
13.0	26.0	7.1	160.0	7.9	0.0
14.0	25.5	7.6	240.0	8.1	0.0
15.0	25.4	7.9	240.0	8.0	0.0
16.0	29.1	8.3	2600.0	7.6	1.1
17.0	26.0	7.7	7000.0	8.0	4.0
18.0	27.5	7.4	16000.0	7.9	9.0
Average	25.5	7.3	1545.2	8.0	0.8
Minimum	24.0	6.3	52.0	7.2	0.0
Maximum	29.1	8.3	16000.0	8.8	9.0
R- Rejected value					

Table B-3

Target Analyte List Metals

Analyte	Soil/Sediment USEPA Contract Required Quantitation Limit (mg/kg)
Aluminum	40
Antimony	12
Arsenic	2
Barium	40
Beryllium	1
Cadmium	1
Calcium	1,000
Chromium	2
Cobalt	10
Copper	5
Cyanide	2
Iron	20
Lead	1
Magnesium	1,000
Manganese	3
Mercury	0.04
Nickel	8
Potassium	1,000
Selenium	1
Silver	2
Sodium	1,000
Thallium	2
Vanadium	10
Zinc	4

Table B-4

PCB Field Screening Results

Sampling Station	Reading at 2 ppm	Reading at 10 ppm	Reading at next level[1]	Interpretation
Background	0.26	0.55	NA	< 2 ppm
1s	-1.24	-0.29	0.60 (100 ppm)	10-100 ppm
2s	-1.04	0.36	NA	2-10 ppm
3s	-1.24	-0.24	0.76 (80 ppm)	10-80 ppm
4s	-1.24	-0.13*	0.57 (80 ppm)	10-80 ppm
5s	-1.24	-0.18*	1.09 (80 ppm)	10-80 ppm
6s	-0.80	0.08*	0.73 (100 ppm)	2-10 ppm
7s	-0.95	0.26	0.74 (100 ppm)	2-10 ppm
8s	-1.24	-0.57	0.28 (40 ppm)	10-40 ppm
9s	-1.24	-0.62	0.66 (100 ppm)	10-100 ppm

Notes:
[1] Sediment samples were tested at 2, 10 and either 40, 80 or 100 ppm.
* indicates that value was within errors limits set for standards (<0.20)
NA- Not Applicable

Table B-5
Fish Collection Methods and Locations

Station	River Mile	Description	Date Sampled (1993)	Small Beach Seine	Large Beach Seine	Electro-shock Boat	Backpack Electro-shocker
1	203.3-204.7	Above Feeder Dam	8/7	A			
			9/1			A/B	
20	196.10	Below Bakers Falls	8/11				A
2	194.1	Rogers Island	8/8	A			
			8/27			A/B	
3	191.5	Thompson Island Pool - Opposite Snook Kill	8/10	A			
			8/31			A/B	
4	190.3-189.6	Thompson Island Pool - Griffin Island	8/10	A/B			
			8/31			A/B	
8	169.2	Stillwater	8/9	A/B			
			8/31			A/B	
9	159	Below Lock 1	8/12	A/B			
10	143.5	Albany Turning Basin (South)	8/16			A/B	
			8/26			A/B	
11	136.7	Shad Island	8/19		A/B		
12	122.7	Stockport Flats - Little Nutten Hook	8/25			A/B	
13	113.8	Catskill/Rogers Island	8/25			A/B	
14	100	Tivoli Bay Area - Cruger Island	8/17			A/B	
15	89.4	Esopus Meadows	8/17			A/B	
16	58.7	Plum Pt./Moodna Creek	8/16		B		
			8/30		A		
17	47.3	Iona Island	8/19	A/B	A/B		
18	25.8	Piermont Marsh	8/18	A			
			8/24		B		

Notes: Fish were collected by NYSDEC and NOAA.

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Table B-6
Hudson River Ecological Assessment Vegetation List

SCIENTIFIC NAME	COMMON NAMES	STATION NUMBER																	
		Freshwater									Freshwater Tidal						Brackish Tidal		
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
SUBMERGENTS																			
Lemna minor	duckweed																		
Myriophyllum brasiliense	milfoil																		
Nuphar advena	spatter dock																		
Sagittaria sp.	broad leaf arrowhead																		
Trapa natans	water chestnut																		
Vallisneria americana	water celery																		
EMERGENTS																			
Cyperus strigosus	straw colored nut sedge																		
Iris versicolor	blue flag																		
Juncus effusus	soft rush																		
Lindera benzoin	spice bush																		
Lythrum salicaria	purple loosestrife																		
Peltandra virginica	arrow arum																		
Pontederia cordata	pickerelweed																		
Spartina alterniflora	cord grass																		
Typha sp.	cattail																		
Zizania aquatica	wild rice																		
BANK																			
Acer rubrum	red maple																		
Acer saccharinum	silver maple																		
Ailanthus altissima	tree-of-heaven																		
Alnus rugosa	speckeled alder																		
Cornus ammomum	silky dogwood																		

Table B-6 (continued)
Hudson River Ecological Assessment Vegetation List (Continued)

SCIENTIFIC NAME	COMMON NAMES	STATION NUMBER																	
		Freshwater									Freshwater Tidal						Brackish Tidal		
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
BANK																			
Eupatorium maculatum	boneset																		
Eupatorium purpurea	spotted joe-pye weed																		
Fraxinus americana	white ash																		
Lonicera sp.	honeysuckle																		
Mimulus ringens	square-stemmed monkey-flower																		
Onoclea sensibilis	sensitive fern																		
Osmund cinnamomea	cinnamon fern																		
Phragmites australis	common reed																		
Plantanus occidentalis	sycamore																		
Populus deltoides	cotton wood																		
Prunus serotina	black cherry																		
Quercus velutina	black oak																		
Rhus typhina	staghorn sumac																		
Robina pseudo-acacia	black locust																		
Rosa sp.	rose																		
Sambucus canadensis	elderberry																		
Solidago alnifolia	elm-leaved goldenrod																		
Thuja occidentalis	northern white cedar																		
Tilia americana	basswood																		
Ulmus	elm																		
Verbena stricta	hoary vervain																		
Viburnum dentatum	arrow wood																		
Vitis sp.	grape vine																		
	bridal wreath																		

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PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

APPENDIX C

LIFE HISTORY AND ECOLOGY OF DOMINANT MACROINVERTEBRATE RECEPTORS

TAMS/MCA

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PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

APPENDIX C

LIFE HISTORY AND ECOLOGY OF DOMINANT MACROINVERTEBRATE RECEPTORS

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APPENDIX C

LIFE HISTORY AND ECOLOGY OF DOMINANT MACROINVERTEBRATE RECEPTORS

Invertebrates in the Upper and Lower Hudson River provide linkages between PCBs in the sediment and fish and wildlife. Eight invertebrate taxonomic groups were found to be the dominant macroinvertebrates during the 1993 ecological sampling program. Taxa were considered dominant if they comprised at least 5% of all the organisms collected at each station. Profiles of the following taxa are provided in this appendix.

- Isopods;
- Chironomids;
- Oligochaetes;
- Amphipods;
- Pelecypods;
- Cladocerans;
- Hydrobids; and
- Polychaetes.

C.1 Isopods (Sowbugs)

Caecidotea racovitzai and *Cyathura polita* were the two dominant isopod species found during the 1993 sampling program. It is unusual to find more than one species of isopod inhabiting a restricted area (Pennak, 1989). Most isopods have limited ability to maintain position in lotic environments (i.e., moving water). If the current is too swift, they are often washed downstream until they can establish a footing in more protected areas (Pennak, 1989). The life histories of *C. racovitzai* and *C. polita* are provided below.

C.1.1 *Caecidotea racovitzai*

The most abundant isopod collected in the TI Pool was *C. racovitzai*. This species is epifaunal and feeds on detritus, dead and injured invertebrates, and all types of decomposing aquatic vegetation in the surficial sediments (Pennak, 1989). Gut content analyses typically reveal organic debris, microcrustaceans, and algae (Kerr, 1978). *C. racovitzai* is generally absent in more erosional areas where the turnover of the surficial sediments make potential food sources less available (Kerr, 1978). Representatives of the genus *Caecidotea* have been extensively documented as characteristic organisms in areas of high sewage loads and organic pollution (Moon, 1957; Klein, 1957; Ellis, 1961; Kerr, 1978).

Although isopods can generally breed throughout the year, there is typically one major breeding season in late spring and early summer (Kerr, 1978). The number of eggs per brood may range from 20 to 250. Newly hatched young are retained in the marsupium of the female for approximately 20 to 30 days (Pennak, 1989). Little is known about the total number of molts during the life cycle but the life span is thought to be approximately one year. *C. racovitzai* ranges in size from one to two mm for the juveniles to approximately 12 mm for the adults, with most individuals ranging from six to eight mm (Kerr, 1978).

C.1.2 *Cyathura polita*

Cyathura polita is a common bottom dwelling estuarine isopod. It requires a fairly stable substrate and is generally not found in erosional areas or in extremely silt-ladened environments (Burbanck, 1967). *C. polita* was found at every station sampled during the NYSDEC 1983 macrobenthic study of the main channel of the Hudson River from RM 141.1 to RM 67.4 (Simpson et al., 1984). In the current study, *Cyathura polita* was considered to be a dominant organism at only Piermont Pier (RM 25.8).

Williams et al. (1975; 1973) found Hudson River (RM 37) *C. polita* in association with a number of benthic organisms including oligochaetes, chironomids, nematodes, the polychaete *Hobsonia florida*, and the amphipod *Gammarus fasciatus*. In addition to their normal benthic mode of existence, cyathurans may also be capable of limited swimming as evidenced by *C. polita* found in Hudson River (RM 38) plankton samples (Ecological Analysts, 1979).

C. polita can tolerate a wide range of salinity, temperature, and quality and quantity of food (Burbanck, 1967). Although *C. polita* is capable of living and developing normally over a wide range of salinities, the highest densities have been reported in salinities that range from 2 to 7 ppt (Kelley and Burbanck, 1976; Dean and Haskin, 1964). *C. polita* is intolerant of high organic content and low dissolved oxygen levels (Burbanck, 1967; Dean and Haskin, 1964).

As other isopods, cyathurans are considered to be omnivores and feed on diatoms and detritus. In addition, cyathurans have been documented to prey on gammarid amphipods, dead fish and other cyathurans (Burbanck, 1962).

The reproductive season generally extends from May through August with the number of eggs in the marsupia of females varying from 1 to 36. Juvenile cyathurans have relatively poor mobility (Kelley and Burbanck, 1976) and may be interstitial in areas where there is an insufficient algae or detritus mat (Burbanck, 1967; 1962). Adults are generally associated with the surficial sediment layers (Watling et al., 1974) and commonly live in tubes. They construct tubes or they inhabit tubes of other organisms, such as tube building polychaetes.

Population studies of *C. polita* in the Hudson River area of Indian Point (RM 45) indicate that the highest benthic population levels occur from late summer to early fall and decrease through the fall

and early winter (Texas Instruments, 1976). Data from Texas Instruments (1976) support the three year life span originally suggested by Burbanck (1962).

C.2 Chironomids (Non-biting Midges)

The family Chironomidae represents a diverse group of aquatic dipterans (two-winged flies) consisting of a total of five subfamilies: Tanypodinae, Podonominae, Diamesinae, Chironominae and Orthocladiinae. Chironomid eggs are laid in masses or in strings which are frequently attached to various substrates including the top of sediments and aquatic vegetation. There are about 2,000 to 2,500 eggs per mass, which hatch in 3 to 17 days, depending on temperature (Fuller, 1974). The life cycle is variable and can range from several generations in one year to some overwintering larvae with only one generation per year (Pennak, 1989).

Chironomid larvae occur in all types of aquatic habitats from marine to freshwater systems and are found in association with sediments, aquatic vegetation and detritus. Merritt and Cummins (1978) summarized the ecological and distributional data for various chironomids in terms of their primary mode of existence or habit into *clingers, sprawlers* and *climbers* that inhabit the surfaces of aquatic plants, detrital debris and fine sediments and *burrowers*, that tunnel into the sediment or construct tubes composed of algae, fine silt or sand grains. Feeding strategies were categorized as follows:

- *Collectors* are detritivores which gather, filter or collect Fine Particulate Organic Matter (FPOM) less than 2.0 mm in size from the sediments.
- *Shredders* are herbivores or detritivores that ingest either living or decomposing plant material and Coarse Particulate Organic Matter (CPOM) greater than 2.0 mm in size.
- *Scrapers* graze on attached algae and organic material with some *scrapers* able to pierce the plant cells and suck out the contents.
- *Piercers* puncture prey and suck the tissue.
- *Engulfers* consume whole animal tissue.

The life histories of the three dominant chironomid subfamilies (Tanypodinae, Orthocladinae, and Chironominae) found during the 1993 sampling are described below.

C.2.1 Tanypodinae

Members of the Tanypodinae include mostly sprawlers with some burrowers. Most Tanypodinae are predaceous and feed on other Tanypodinae, other midge larvae, crustacea and small worms.

Species within the genus *Procladius* include many sprawlers, which are usually found in depositional zones on the surface of fine sediments (Roback, 1980). Modes of feeding include engulfing and collecting. The engulfers prey on a variety of protozoa, microcrustacea, Ephemeroptera (mayflies), oligochaetes, and other dipterans, while the collectors feed on the FPOM. *Procladius*

inhabited all silty sand or silty substrate stations sampled during the 1983 macrobenthic study of the freshwater main channel of the Hudson River (Simpson et al., 1984). The two other predominant genera found, *Clinotanypus* and *Coelotanypus*, are considered burrowers and also inhabit depositional zones. *Clinotanypus* are engulfers and prey on oligochaetes, ostracods, cladocerans and other chironomid larvae (Merritt and Cummins, 1978; Roback, 1969).

C.2.2 Orthocladiinae

Members of the Orthocladiinae include mostly tube building burrowers with some sprawlers. Orthocladiinae larvae are generally collectors and scrapers with some herbivorous shredders. *Cricotopus trifascia* was the dominant species of Orthocladiinae found in the TI Pool. *Cricotopus* are usually associated with algal mats and detritus. Collectors within this genus derive energy from the FPOM of the sediments, while shredders chew living or decomposing plant material and CPOM. Some members of this genus burrow into plant roots.

C.2.3 Chironominae

The Chironominae include mostly burrowers and clingers. Chironominae larvae are generally collectors that gather or filter deposits/suspensions of decomposing FPOM. Many of the larvae build fragile tubes composed of organic debris, silt, and small sand grains. Other mud inhabiting larvae burrow into soft sediments. *Tanytarsus* sp., *Dicrotendipes* sp., *Polypedilum* sp., *Tribelos jucundus*, *Tribelos* sp. and *Chironomus* were the six Chironominae most frequently found in the TI Pool. *Dicrotendipes* sp. and *Polypedilum* sp. were also found in the Lower Hudson.

Tanytarsus sp. is classified as a net spinning clinger and climber, inhabiting the surfaces of aquatic plants, detrital debris and fine sediments. Species of this genus feed primarily by filtering and gathering, with a few species feeding by scraping. Simpson and Bode (1980) found *Tanytarsus* over a wide range of habitats in the Hudson River, ranging from fast flowing stream headwaters to more slow flowing and turbid canals.

Dicrotendipes sp. is a burrower in soft sediments and gathers or collects FPOM from the surficial sediments. As is the case with most of the Chironominae, this genus can be found over a wide range of conditions.

Midges of the genus *Polypedilum* are climbers and net spinning clingers living on floating plant material and detrital debris. They have a range of feeding modes and have been classified as herbivorous shredders, collectors, and carnivorous engulfers (Merritt and Cummins, 1978). Some larval members of this genus construct conical nets and feed on suspended particles that are trapped in the net (Walshe, 1951). *Polypedilum* have been found in organically enriched waters (Simpson and Bode, 1980). Some species in the genus *Polypedilum* are able to colonize a wide variety of habitats. Simpson et al. (1984) found *Polypedilum* to be most abundant in areas of silty sand in the Hudson River.

Tribelos jucundus and *Tribelos* sp. are clingers that build tubes. These species scrape and collect algae and FPOM from a variety of surfaces.

Chironomus sp. are tube building burrowers found in soft sediments. They are generally classified as collectors and herbivorous shredders. Members of this genus gather FPOM deposits from the surface of sediments or shred decomposing plant material and CPOM. Although members are generally considered detritus grazers, some reports indicate that they may consume tubificid oligochaetes (Loden, 1974). *Chironomus* is a versatile genus capable of colonizing a wide variety of habitats Simpson and Bode (1980).

C.3 Aquatic Oligochaetes (Aquatic Worms)

Oligochaetes collected in the Upper and Lower Hudson were not identified to lower taxonomic groups. Hence, this section discusses the characteristics and ecology of the class Oligochaeta with an emphasis on the family Tubificidae, the major taxon of oligochaetes found in the freshwater portion of the Hudson River (Simpson et al., 1984).

Oligochaetes are segmented worms that are extremely common in mud and detritus substrates. They move by a series of circular and longitudinal muscle contractions of the body wall, which enables them to crawl and burrow. Most oligochaetes feed by ingesting sediments containing microorganisms and various plant and animal CPOM. They play an important role in mixing the sediments of the river bottom and in the exchange of nutrients and toxic pollutants between water and sediment (Stanne et al., 1996). Oligochaetes have a thin body wall that allows gas exchange to take place across the surface of the body. Reproduction includes both asexual budding and sexual copulation and reciprocal sperm transfer. The entire life cycle generally takes one to two years (Brinkhurst and Jamieson, 1972; Barnes, 1987; Pennak, 1989).

Most Tubificidae live in tubes in the sediment and generally prefer silt laden substrates. The family is usually associated with organically rich areas and can tolerate relatively low dissolved oxygen levels (Brinkhurst and Jamieson 1972; Pennak 1989). The most concentrated populations of tubificids are often found in areas polluted with sewage (Pennak, 1989). Typically they form dense aggregations on the bottom in response to local food resources (Chekanovskaya, 1962). Tube dwelling tubificids usually burrow with the anterior end in the sediment and the posterior end in the water column. If dissolved oxygen levels become low, the posterior end moves more vigorously in the water column in order to increase gas exchange across the integument (Chekanovskaya, 1962).

C.4 Amphipods (Scuds or Sideswimmers)

Gammarus fasciatus was the dominant amphipod found during the field sampling program. *G. fasciatus* is an epibenthic amphipod that is widespread and abundant in a variety of shallow waterbodies. It is found among the leaves of submergent plants, under dying vegetation at the base of plants (Lippson and Lippson, 1984), or amongst bottom debris (Pennak, 1989).

G. fasciatus is omnivorous and derives much of its energy by feeding on detritus, algae, fungi and dead animal matter (Clemens, 1950; DeLong et al., 1993). Bek (1972) estimates that the daily consumption of detritus for representatives of the family Gammaridae may approach 60% of the body weight for adults and 100% of the body weight for juveniles. Thus, they are considered voracious eaters of an extremely wide range of food resources. *G. fasciatus* is primarily a freshwater species and is seldom found in areas of salt water intrusion in the Hudson River (Consolidated Edison, 1978). However, it was a dominant organism both in the TI Pool in the Upper Hudson and at Iona Island and Piermont Pier in the Lower Hudson.

DeLong et al. (1993) studied the influence of diet on the growth of *G. fasciatus*, feeding them one of the following four diets: (1) filamentous algae and diatoms, (2) dead chironomids, (3) CPOM and (4) FPOM. They found no significant differences in growth during the first three weeks of the newly emerged first generation, but from week 4 to 6, amphipods fed either algae or dead chironomids were significantly larger than those fed on either CPOM or FPOM. In a parallel study of natural populations (DeLong et al., 1993), gut contents indicated that *G. fasciatus* consumed all the food diets used in the laboratory study with the greatest proportion of algae and chironomids found in the largest size-classes of amphipods. Therefore, *G. fasciatus* may feed primarily on detritus (CPOM and FPOM) until it reaches a size in which it can efficiently graze filamentous algae or scavenge larger dead animal material such as chironomids or oligochaetes.

While *G. fasciatus* is considered a bottom dweller, it seems to have more rapid and greater mobility than other benthic organisms. In abundance studies conducted in the vicinity of Indian Point (RM 43), it was shown to be primarily epibenthic with some individuals able to move into the water column at night (Consolidated Edison, 1978).

The eggs of *G. fasciatus* are fertilized in the brood chamber of the female. There is generally one annual brood of 15 to 50 eggs, depending on the size and age of the female, with an incubation period lasting from one to three weeks. Development is direct with fully formed juveniles retained by the female an additional one to eight days. The juveniles usually emerge during late spring, summer, or early fall. This time period coincides with the first molt of the female following copulation. The number of molts and the interval between molts varies depending on a variety of conditions including temperature and food. Pennak (1989) indicates that some amphipods may have as many as 15 to 20 molts with an intermolt period ranging from 3 to 40 days. The entire life span is generally about one year. Juvenile *G. fasciatus* range in size from 2 to 5 mm, while adults are approximately 6 to 10 mm, with some individuals as long as 14 mm.

C.5 Pelecypods (Clams)

With the exception of *Elliptio*, which accounted for less than one percent of the number of individual organisms collected, *Pisidium* sp. was the only pelecypcod collected in the Upper Hudson. *Pisidium* are small burrowing infaunal bivalves that range in size from approximately 2.0 mm to 8.0

mm in the Hudson River basin (Strayer, 1987). *Pisidium* can be very abundant in the Hudson River basin, where 12 species belonging to this genus have been documented (Strayer, 1987).

Simpson et al. (1984) note that different species of *Pisidium* have different habitat preferences, but most burrow in relatively soft substrates. According to Hynes (1966), *Pisidium* can burrow relatively deep in soft sediments and extend their long siphons into the sediment water interface.

The diet of most suspension feeding pelecypods consists of a variety of microscopic particles including organic detritus and phytoplankton filtered from the sediment water interface. Some pelecypods are extremely efficient suspension feeders and are capable of removing particles as small as one micron from the water column (Pennak, 1989). In addition to filtering particulates from the sediment water interface, organic detritus temporarily placed into suspension during burrowing may also be filtered and taken into the digestive system. This process, known as interstitial suspension-feeding, has been observed in *Pisidium* burrowing below the mud surface (Lopez and Holopainen, 1987).

Pisidium are hermaphrodites with self-fertilization occurring in the reproductive ducts (Pennak, 1989). The fertilized eggs are brooded between the gills of the parent and undergo direct development into young or juvenile clams. There may be from one to many young in various stages of development at any particular time within a single parent (Pennak, 1989). Although reproduction occurs throughout the year, few young are released during the winter (Pennak, 1989). Juveniles are not released into the environment until they are fully formed with all the morphological features of the parent and are often 1/4 to 1/3 the size of the parent when they are discharged from the gills (Heard, 1965). Their life span generally lasts from one to two years (Pennak, 1989).

C.6 Cladocerans (Water Fleas)

The dominant cladocerans collected during the field sampling program belong to the family Chydoridae. The family Chydoridae contains numerous genera that range in size from 0.8 mm to 5.0 mm. Although many cladocerans are capable of vertical migrations in the water column and are often classified as zooplankton, members of the family Chydoridae are considered to be benthic species with their first two pair of legs adapted for seizing, clinging or grasping the substrate. They are commonly found in the interstices of sand and gravel deposits of many streams and ingest a wide variety of organic material including algae and bacteria (Pennak, 1989). Most are suspension feeders with the setae on the thoracic appendages to collect food particles and transfer the filtered particles to the mouth (Barnes, 1987).

Brood size is variable and depends on a variety of factors including temperature, food, and the maternal body size (Hann, 1985). Generally, females can produce multiple clutches consisting of 2 to 20 eggs. The mean number of eggs per brood in some Chydoridae increases for the first four reproductive instars and then declines. In others, the mean brood size remains fairly constant throughout the reproductive period (Hann, 1985). Development is usually direct and the fully

developed juveniles are released from the brood chamber of the female in approximately two days (Pennak, 1989).

Many Chydoridae reproduce once during the warmer summer months (Barnes, 1987; Hann, 1985; and Pennak, 1989). As in most cladocerans, reproduction is generally parthenogenetic with only females being produced. Males may be produced at times of population stress including changes of food conditions or water temperatures (Pennak, 1989). The fertilized eggs of Chydoridae remain with the exoskeleton after the molt. The encapsulated fertilized eggs may overwinter and are able to withstand a variety of adverse environmental conditions. Life cycles of the Chydoridae are extremely variable and may last from days to months (Hann, 1984).

The cladoceran *Bosmina longirostris* is abundant in the freshwater portion of the Hudson River, but also occurs in brackish water up to salinities of 8 ppt (Stanne et al., 1996). It typically reaches peak densities in June, when there are large larval fish populations, making it an important food source for these fish.

C.7 Hydrobids (Mud Snails)

Hydrobia minuta was found only in the brackish waters of Piermont Marsh. It is a member of the deposit feeding Hydrobiidae family of mud snails and is generally less than 6 mm in length (Lippson and Lippson, 1984). Gosner (1971) describes this species as occurring in brackish waters or salt marshes and is often associated with the common marine green algae *Ulva* sp., as well as other plants and detritus. *H. minuta* has been renamed either *H. totteni* or *H. truncata*, which are currently considered the same species (Herschler, 1995). The following discussion focuses on the general biology of estuarine hydrobiid snails with particular reference to the genus *Hydrobia*.

In a review of particle feeding by deposit-feeders, Levinton (1980) identified a number of parameters that influence resource availability in the genus *Hydrobia*. He concluded that the feeding behavior of *Hydrobia* is complex and depends on a number of factors, including particle size, quality of food, space limitations, and renewal rates of diatoms and bacteria. Since the mouth is generally in direct contact with the substrate, detrital particles may be ingested as the snail moves along over the surficial sediments. The radula, a toothed chitinoid structure inside the mouth, is capable of both grinding food that is ingested and scraping algae off the particulates. However, many *Hydrobia* seem to derive their main nutritional requirements from algae and bacteria associated or attached to ingested detrital particles taken directly into the mouth (Fenchel, 1977; Levinton, 1980).

Hydrobia have separate sexes. The females deposit multiple egg capsules, which generally contain one to two fertilized eggs per capsule (Fenchel, 1977). Depending on the species, the eggs can either develop directly into juvenile snails or into short-lived pelagic larvae. The eggs that develop directly into juvenile snails that become reproductive adults the following spring. The life span is typically one year, but may extend up to two years (Fish and Fish, 1974).

C.8 Polychaetes (Segmented Worms)

Hobsonia florida, previously known as *Hypaniola grayi*, was the dominant polychaete found at Iona Island and Piermont Marsh in the Lower Hudson. It is a relatively small worm with a total length of approximately 15 mm belonging to the Ampharetidae family.

The Ampharetidae are tube dwelling polychaetes that can be categorized as selective deposit feeders (Barnes, 1987). Specialized feeding tentacles with ciliated grooves gather food particles from the surface of the sediments or between sand particles and convey the deposited material to the mouth. Unlike other tentacular feeders, the tentacles of ampharetids can be completely retracted into the mouth.

Ampharetidae reproduce sexually, with mature worms carrying either eggs or sperm. The gametes are shed into the surrounding water column and the fertilized eggs develop into trochophore larvae. Species that spawn only once generally produce a large number of small eggs that have feeding planktonic trochophore larvae. Other species that reproduce more than once, or have relatively short life spans, generally have a smaller number of large eggs and a non-feeding benthic trochophore larvae (Fauchald, 1983). After a period that varies from days to weeks, the trochophore larva metamorphoses into the adult form.

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LIFE HISTORY AND ECOLOGY OF FISH RECEPTORS

Species of interest include:

- Pumpkinseed *(Lepomis gibbosus)*
- Spottail Shiner *(Notropis hudsonius)*
- Brown Bullhead *(Ictalurus nebulosus)*
- White Perch *(Perca flavescens*)
- Yellow Perch *(Morone americana)*
- Largemouth Bass *(Micropterus salmoides)*
- Striped Bass *(Morone saxatilis)*
- Shortnose Sturgeon *(Acipenser brevirostrum).*

These species represent fish that experience a wide variety of exposures, including pelagic and demersal feeders, stationary and migratory species, and various trophic levels.

Information on the feeding ecology of Hudson River fish species was taken from the literature and from several studies on the river. Important sources of information include:

- Hudson River aquatic ecology studies performed by LMS Engineers in Haverstraw Bay (LMS, 1975a), above Newburgh (LMS, 1975c), and in the vicinity of Kingston (LMS, 1975b);

- Observations on white perch feeding made as part of the TAMS/Gradient Phase II sampling effort;

- Analyses of gut contents along with invertebrate investigations by Exponent (1998a, 1998b); and

- Analysis of several fish species collected by New York State Department of Environmental Conservation in (NYSDEC) 1997 and 1998 and analyzed for the Hudson River PCBs Reassessment. Additional insight into feeding ecology for fish collected from the river was obtained from Gladden et al. (1988) and Feldman (1992).

The prey base ecology evaluation relied on information obtained from the literature, observations in the Hudson River reported by Exponent (1998a, 1998b), observations made by Charles Menzie on the ecology of zooplankton, epibenthos, and infauna in the Lower Hudson

River invertebrates during 1971 to 1975 while employed by LMS, and observations reported in Gladden et al. (1988), Simpson and Bode (1980), and Feldman (1992).

D.1 Pumpkinseed *(Lepomis gibbosus)*

The pumpkinseed, *Lepomis gibbosus*, is the most abundant and widespread fish in New York State (Smith, 1985). In the Hudson River, they feed exclusively upon epiphytic water column organisms. Pumpkinseed are important forage for predatory fishes.

D.1.1 Foraging

Pumpkinseed are diurnal feeders in areas with low light intensity and migrating to cooler, deeper water at night. They do not feed in winter and only begin to feed when the water temperature rises above 8.5°C. Pumpkinseed forage on hard shelled gastropods and are able to exploit food sources not available to other fish, particularly mollusks (Sadzikowski and Wallace, 1976). Food is mainly a variety of insects and, secondarily, other invertebrates. Small fish or other vertebrates, e.g., larval salamanders, can also contribute significantly to the pumpkinseed diet (Scott and Crossman, 1973).

Early juvenile pumpkinseed prefer chironomid larvae, amphipods, cladocerans, and, to a lesser extent, copepods as food items (Sadzikowski and Wallace, 1976). Juvenile pumpkinseed in the Connecticut River feed primarily upon benthic organisms (Domermuth and Reed, 1980). A study conducted in the St. Lawrence River near Massena found that juvenile pumpkinseed between 77 and 113 mm in length consumed 94% chironomids (Johnson, 1983). Feldman (1992) found that juvenile pumpkinseed taken from Thompson Island Pool in the Hudson River consumed zooplankton such as cladocerans, copepods, ostracods, chironomids and talitrids. Adults consumed mostly gastropods on plants. No sediment source of food was noted.

Adult pumpkinseed primarily prefer insects and secondarily prefer other invertebrates. As the fish age and increase in size, other fish and invertebrates other than insects constitute a larger portion of the diet, up to 50% of the diet.

A small subset of the pumpkinseed samples taken as part of the USEPA Phase 2 activities were analyzed for gut contents. A large number of chironomid were found and identified to evaluate the relative contribution of sediment and water sources to the diet of pumpkinseed resident in the Hudson River. These gut content analyses demonstrate that pumpkinseed in the Hudson River appear to feed largely upon species associated with plants or other surface substrates.

Additional data on the diet of pumpkinseed sunfish is available from the collections of yearling fish made by Exponent (1998a, 1998b). These data indicated that the diet of the fish was comprised invertebrate commonly associated with benthic environments. Predominant prey items included small clams, snails, amphipods, isopods, and insect larvae. However, most of the invertebrate prey items live at or on the surface of substrates rather than deep within the sediments. Gastropod snails were a predominant item in the diet similar to the observations of

Feldman (1992) who observed that these were an important part of the diet of adult fish; he presumed they were eating gastropods living on plants. The composition of the chironomid insect larvae in the gut contents of yearling sunfish is also suggestive that yearling fish feed on surface substrates rather than on burrowing animals; *Dicrotendipes* spp. were commonly observed while *Procladius* spp. were rarely seen in the gut contents. The amphipod *Gammarus* spp. is also an important item in the diet and is considered epibenthic and meroplanktonic.

The diet of pumpkinseeds changes with size and age as noted above. Young-of-the-year fish may consume a proportionally greater amount of smaller invertebrates associated with the water column while larger juvenile and adult sunfish may consume a proportionally greater amount of benthic invertebrates. These benthic invertebrates largely include species that live on or at the surface of substrates. Gastropods, for example, feed on surface substrates and are likely exposed to water conditions directly above sediments or around stands of plants. The diet of pumpkinseed sunfish consist of invertebrates that may be more influenced by conditions at and above the water/sediment interface than by conditions deeper in the sediments.

D.1.2 Range, Movement and Habitat within the Hudson River

Pumpkinseed are restricted to freshwater and are found in shallow quiet areas with slow moving water. Pumpkinseed are usually found in clear water with submerged vegetation, brush or debris as cover. They rely on the littoral zone as a refuge from predators and for foraging material (Feldman, 1992).

Several investigators have noted the ability of pumpkinseed to return to a home range, even after significant displacement (Hasler and Wisby, 1958; Fish and Savitz, 1983; Shoemaker, 1952; Gerking, 1958).

Pumpkinseed are found throughout the Upper Hudson River above the Federal Dam, primarily in wetland, stream mouth, and embayment habitats (MPI, 1984).

D.1.3 Reproduction

Spawning occurs during early spring and summer although it can extend into late summer (Scott and Crossman, 1973). Nests are built in water that is 6 to 12 inches deep, forming colonies close to aquatic vegetation and other pumpkinseed nesting areas. Nesting occurs when the water temperature reaches 60°F and lasts approximately 11 days. Nesting substrates include sand, sandy clay, mud, limestone, shells and gravel. Females lay from 600 to 5,000 eggs (Smith, 1985). Males guard the nest for one week after hatching.

D.2 Spottail Shiner *(Notropis hudsonius)*

The spottail shiner, *Notropis hudsonius*, consumes plankton, aquatic insects, and some bottom-dwelling organisms, and is therefore exposed to sediment and water column. The spottail shiner is consumed by virtually all other fish, including larger spottail shiners.

D.2.1 Foraging

Spottail shiners are morphologically suited for bottom foraging in that they have rounded snouts that hang slightly over their mouths. They do not however feed exclusively upon benthic organisms. Spottail shiners are considered omnivorous and opportunistic feeders, feeding upon cladocerans, ostracods, aquatic and terrestrial insects, spiders, mites, fish eggs and larvae, plant fibers, seeds, and algae (Texas Instruments, 1980; Scott and Crossman, 1973; Smith, 1987). Based on work in the Lower Hudson River, Gladden et al. (1988) consider zooplankton to be a major part of the spottail shiners diet.

In Lake Nipigon, Ontario (Scott and Crossman, 1973), 40% of the diet was made up of *Daphnia* spp. Other cladocerans were also present, and aquatic insect larvae, including chironomids and ephemeropterids, comprised another 40% of the spottail shiner diet.

In Lake Michigan, Anderson and Brazo (1978) found that terrestrial dipterans and fish eggs represented the major components of the spottail shiner's diet in the spring and summer. In the fall, chironomid larvae and terrestrial insects represent the major diet components.

Information on the diet of spottail shiners in the Hudson River was obtained by Exponent (1998a, 1998b). We evaluated these data qualitatively and found that the major food items appeared to be organisms with a high water column association (algae, cladocera, and copepods) and species that live in close association with surface substrates (ostracods, amphipods, chironomid larvae and caddisfly larvae). The composition of the predominant chironomid larvae in spottail shiner gut contents are considered surface sprawlers or epiphytic rather than sediment burrowers. As such, these prey items may derive the bulk of their exposure via water column sources although strictly speaking are considered to be benthic organisms.

Observations on feeding behavior of spottail shiner suggest they can range from benthic feeders to water column feeders. Many of the benthic invertebrates include surface dwellers that are influenced by surface water conditions. We estimate spottail shiners primarily eat invertebrates that are more directly influenced by surface water conditions than by conditions below the surface of sediments. However, benthic invertebrates could be an important part of the diet based on the literature.

D.2.2 Range, Movement and Habitat within the Hudson River

Spottail shiners prefer clear water and can be found at depths up to 60 feet (Smith, 1987), but tend to congregate in larger numbers in shallow areas (Anderson and Brazo, 1978). Spottail shiners in the Upper Hudson River were primarily taken in wet dumpsite habitat areas (MPI, 1984).

D.2.3 Reproduction

Spottail shiners spawn in the spring and early summer in habitats with sandy bottoms and algae (Scott and Crossman, 1973). In New York waters, spawning usually occurs at the mouths of streams in June or July. Ovarian egg counts range from 100 to 2,600 eggs per female, depending upon total size (Smith, 1985).

D.3 Brown Bullhead *(Ictalurus nebulosus)*

The brown bullhead, *Ictalurus nebulosus*, is a demersal omnivorous species occurring near or on the bottom in shallow, warmwater situations with abundant aquatic vegetation and sand to mud bottoms. Brown bullhead are sometimes found as deep as 40 feet, and are very tolerant of conditions of temperature, oxygen, and pollution (Scott and Crossman, 1973).

D.3.1 Foraging

The brown bullhead feeds on or near the bottom, mainly at night. Adult brown bullhead are truly omnivorous, consuming offal, waste, molluscs, immature insects, terrestrial insects, leeches, crustaceans including crayfish and plankton, worms, algae, plant material, fishes, and fish eggs. Raney and Webster (1940) found that young bullheads in Cayuga Lake near Ithaca, New York fed upon crustaceans, primarily ostracods and cladocerans, and dipterans, mostly chironomids. For brown bullhead in the Ottawa River, algae have also been noted as a significant food source (Gunn et al., 1977).

Information on the diet of brown bullhead in the Hudson River is available for the river north of Newburgh (LMS, 1975). This work indicated that brown bullhead displayed a varied and seemingly opportunistic feeding behavior. Smaller bullheads consumed primarily chironomid insect larvae, amphipods, odonata, and oligochaete worms. Larger bullheads displayed a similar feeding behavior but also ate young-of-the-year fish. Observations made on gut contents of brown bullheads collected in the Kingston area indicated that oligochaete worms were a major part of the diet.

Additional information on feeding habits of Hudson River fish is available from Exponent (1998a, 1998b) and for fish collected in Spring 1997 and analyzed in the Baseline Modeling Report (USEPA, 1999). The available data from these studies indicates that the diet reflects a large benthic invertebrate component. Only one fish was observed in a gut of one bullhead. Our analysis of the Exponent data indicate that predominant prey items for bullheads included small clams, amphipods (*Gammarus*), isopods (*Caecidotea*), a few of the cladoceran species, and chironomid insect larvae that are typically considered to burrow into sediments (e.g., *Procladius*). The Baseline Modeling Report also reported that the diet of brown bullhead frequently contain oligochaete setae (worms are usually quickly digested or unidentifiable).

A qualitative assessment of the Exponent data suggests that 71% to 83% of the invertebrates are associated with sediments and 17% to 29% are associated with water. Because oligochaete

worms may be a major food item, the benthic percentage is probably even higher and we estimate that it may be as high as 95%. Data for the Lower Hudson River reported by LMS (1975) also support a high component of the diet as benthic in nature in that are large component was comprised of oligochaete worms. These organisms are digested more quickly that insects and crustaceans and are probably underrepresented in the Exponent and Baseline Modeling Report analyses. Fish are considered to be a minor component of the diet (less than 5%).

D.3.2 Range, Movement and Habitat within the Hudson River

Brown bullhead, a freshwater demersal fish, resides in water conditions that are shallow, calm and warm. In the summer, bullheads can be found in coves with ooze bottoms and lush vegetation, especially water clover, spatterdock and several species of pond weed (Raney, 1967). Carlson (1986) found that the vegetated backwaters and offshore areas are the most common habitats for brown bullheads. McBride (1985) found bullhead abundant in river canal pools.

Brown bullhead were most frequently taken in wetland and embayment habitats (MPI, 1984). Brown bullhead prefer wetlands, embayments, and shallow habitats. Carlson (1986) found bullheads most frequently in backwaters, but also in other, deeper areas such as the channel border. This species prefers silty bottoms, slow currents, and deeper waters.

D.3.3 Reproduction

Brown bullhead reach maturity at two years and spawn for two weeks in the late spring and early summer. Smith (1985) noted that in New York, brown bullhead spawn when water temperatures reach 27°C in May and June.

They prefer to spawn among roots of aquatic vegetation, usually near the protection of a stump, rock or tree, near shores or creek mouths. Males, sometimes aided by females, build nests under overhangs or obstructions (Smith, 1985). Eggs are guarded.

D.4 White Perch *(Perca flavescens)*

White perch, *Morone americana*, are resident throughout the Hudson River Estuary below the Federal Dam. They are semi-anadromous and migrate to the lower lock pools of the Upper Hudson River to spawn. They are one of the most abundant species in the region and are the dominant predatory fish in the Lower Hudson River (Bath and O'Connor, 1981; Wells et al., 1992).

D.4.1 Foraging

Adult white perch are benthic predators, with older white perch becoming increasingly piscivorous (Setzler-Hamilton, 1991). Insect larvae and fishes comprise the principal food of white perch, and dipteran larvae, especially chironomids, represent the most important insect

prey. White perch have two peak feeding periods: midnight and noon. Midnight is the most important foraging time.

In a study of Hudson River larvae, Hjorth (1988) found that white perch larvae fed almost exclusively upon microzooplankton. Adults and copepods of *Eurytemora affinis* were the preferred food, but when they were not present, white perch larvae consumed rotifers, cladocerans, and other seasonal zooplankton.

From August through October, young-of-the-year white perch in the Hudson River feed predominantly on amphipods supplemented by copepods and mysids (Klauda et al., 1988). In a study of white perch taken from the Hudson River between Haverstraw and Bear Mountain (Bath and O'Connor, 1985), gammarid amphipods occurred most frequently in the stomachs of immature and mature white perch. Mature fish ate a higher proportion of isopods and annelid worms than did immature fish during the spring and summer. During May and June, mature fish contained between 2 and 8.6% by occurrence, while gammarid amphipods were the predominant food item in July, 64%, and November, 75%. Insect larvae occurred in fewer than 2% of mature fish during May and June, and were not found again during the remainder of the sampling year. White perch in this oligohaline sector of the river fed primarily at or near the sediment-water interface. Their preferred prey items consisted of epibenthic crustaceans and insects.

In 1973 and 1974, LMS conducted an extensive biomass and stomach content analysis in the lower Hudson River on behalf of Central Hudson Gas & Electric Corporation (LMS, 1974). Their study found that the dominant food item consumed by the 49 white perch obtained from Roseton and Danskammer Point during the spring were amphipods, representing 93% of the total identified food volume. During fall sampling, amphipods (*Gammarus* spp. and *Leptochierus plumulosus*) were the dominant food item consumed by the 36 white perch captured. Copepods were found to be a dominant prey item for smaller white perch, but were infrequently found in larger white perch. During the 1974 sampling season, the largest size range of white perch (>17 cm) consumed amphipods and isopods, supplemented by chironomid larvae during the spring and summer, and the decapods *Rithropanopeus harrissi* and *Crangon septemspinosa* during the fall and winter. The data on gut contents indicate that white perch feed primarily on benthic invertebrates and select arthropods such as amphipods and chironomid insect larvae. This fish species probably makes use of all depths in the river for foraging based on collections made using bottom trawls and bottom gill nets in the Lower Hudson River (C. Menzie, 1980).

A small subset of the white perch samples taken as part of the USEPA Phase 2 activities were analyzed for gut contents. A large number of chironomid were found and identified to evaluate the relative contribution of sediment and water sources to the diet of white perch resident in the Hudson River. White perch in the Hudson River generally consume chironomid equally associated with both the water column and sediment (USEPA, 1999).

Another group of 40 white perch from the NYSDEC 1996 sampling effort were also evaluated for gut contents in the Baseline Modeling Report (USEPA, 1999). These fish were collected in the river at Troy and at Catskill Creek in the spring of 1997. Chironomid insect larvae were the most common food item in the diet (75% of fish) and amphipods were the next

most common dietary item (35% of fish). These observations are similar to those made on the fish collected during the USEPA Phase 2 sampling.

The data on feeding behavior for white perch indicate that this species eats invertebrates. The species can make use of near-shore areas as well as the main river bottom for foraging. Feeding is selective for arthropods such as chironomid insect larvae and amphipods. In nearshore areas where rooted aquatic plants are present, the species probably feeds on arthropods associated with both sediments and plants. In areas along the main river bottom, the species probably feeds primarily on benthic invertebrates. Benthic invertebrates include species that vary in the degree of surface water, pore water, and sediment exposure. Oligochaete worms form a small part of the white perch diet which suggests that this species does consume organisms that are closely associated with sediment. This is also suggested by the presence of chironomid insect larvae such as *Tanytarsus, Procadius, Chironomus* and *Cryptochironomus* in their digestive system that are also reported to burrow into sediments rather live on surfaces of plants and substrates (Simpson and Bode, 1980). However, white perch also eat benthic organisms that may be more strongly influenced by surface water exposure. These include chironomid insect larvae such as *Polypedilum illinoense grp.* and *Dicrotendipes neomodestus* that tend to live on the surface of substrates. The amphipod *Gammarus* is also likely to be influenced strongly by water exposures because it lives on or near surface sediments and also swims into the water column.

Based on available information we estimate that the diet of white perch contains 75% invertebrates that are influenced primarily by sediments and 25% of invertebrates that are influenced by water. This estimate is uncertain. If we assume that benthic species are more likely to be exposed to sediment than to water, we estimate that 50% to 100% of the white perch diet consists of invertebrates that are primarily influenced by sediment exposure.

D.4.2 Range, Movement and Habitat within the Hudson River

White perch prefer shallow areas and tributaries, generally staying close to rooted vegetation. The position of this fish relative to the water surface varies somewhat based on size (Seltzer-Hamilton, 1991). White perch are bottom oriented fish that accumulate in areas with dissolved oxygen of at least 6 mg/L (Seltzer-Hamilton, 1991). Gladden et al. (1988) compared the spatial segregation of a number of fish species in the Hudson River estuary and found the majority of white perch over the course of three years to prefer the main channel bottom.

Because white perch make spawning migrations, they are considered semi-anadromous. Spawning occurs in the upper reaches of the Lower Hudson River. Eggs, larvae, and juveniles gradually disperse downstream throughout the summer. Young-of-the-year white perch often congregate in the Tappan Zee and Croton-Haverstraw regions, with a smaller peak from Saugerties to Catskill (LMS, 1992).

During the summer, white perch move randomly within the local area. Adult white perch tend to accumulate at 4.6-6 meters depth during the day and move back to the surface during the night (Seltzer-Hamilton, 1991). White perch spend the winter in depths of 12-18 meters, but

occasionally can be found at depths as low as 42 meters. Hudson River white perch are acclimated at 27.8°C and avoid temperatures that are below 9.5°C or above 34.5°C.

White perch prefer shallow and wetland areas to other habitats, but undertake extensive migrations within the estuary (Carlson, 1986). White perch were most often found in tributaries, vegetated backwaters, and shore areas in the Lower Hudson River. Carlson observed the greatest increase in summertime abundance between RM 102 and 131. By winter, the majority of white perch move downriver, although some overwinter in the upper estuary in areas over 32 feet deep (Texas Instruments, 1980).

In the Upper Hudson River, white perch were taken in the lower two lock pools (MPI, 1984). They were collected primarily in shallow and wetland habitats.

All ages of white perch are adversely affected by high levels of suspended solids. Adult white perch can be found in water with pH ranges between 6.0 and 9.0 and avoid areas with moderate turbidity at 45 NTU, although they can be found in either clear or highly turbid areas (Seltzer-Hamilton, 1991).

D.4.3 Reproduction

Spawning is episodic, usually occurring in a two week period from mid-May to early June when the water temperatures are between 16° and 20°C. Hudson River white perch tend to spawn beginning in April when the water temperature reaches 10° to 12°C, and continue spawning through June. In years when the water temperature increases gradually, the peak spawning period lasts from four to six weeks (Klauda et al., 1988).

White perch prefer to spawn in shallow water, such as flats or embankments, and tidal creeks. They generally spawn over any bottom type (Scott and Crossman, 1973). Spawning is greatest in the fresh water regions around Albany, and between RM 86 and 124 (McFadden et al., 1978; Texas Instruments, 1980).

Fecundity of Hudson River white perch age 2 to 7, the maximum age of white perch in the river, ranges from less than 15,000 to more than 160,000 eggs per female (Bath and O'Connor, 1981). Mean fecundity in that study was 50,678 eggs per female and was dependent upon size.

D.5 Yellow Perch *(Morone americana)*

Yellow perch, *Perca flavescens*, are gregarious fish that travel in schools of 50-200. They feed omnivorously on organisms from the sediment and in the water column. Yellow perch are an important freshwater sport fish.

D.5.1 Foraging

Yellow perch feed actively early in the morning or late in the evening, with less feeding taking place later in the day. At night the fish are inactive and rest on the bottom (Scott and Crossman, 1973).

Young fish feed primarily upon cladocerans, ostracods, and chironomid larvae (Smith, 1985). As they grow, they shift to insects. Chabot and Maly (1986) found that fish that were one to one and a half years old preferred large zooplankton species. Larger fish eat crayfish, small fish, and odonate nymphs (Smith, 1985). Piavis (1991) found that approximately 25% of the diet of yearling yellow perch was made up of other perch. From May through August, chironomids generally comprise between 30% and 60% of the diet. Piavis noted that adult yellow perch forage on midge larvae, anchovies, killifish, silversides, scuds, and caddsisfly larvae. Adults also forage on pumpkinseed.

Information on feeding behavior of yellow perch in the Hudson River is available from the work conducted by Exponent (1998a, 1998b) and fish collected by NYSDEC in Spring 1997 and analyzed in the Baseline Modeling Report (USEPA, 1999). The Exponent data set consists of fish that are in the range of 6.1 to 14.6 cm. The fish analyzed were larger (median = 21.5 cm, maximum = 31.8 cm). Both data sets indicate that yellow perch feed primarily on invertebrates. Based on the literature fish may be eaten by larger yellow perch. The diet of yellow perch indicates they eat a wide variety of invertebrates from the water column, from plants, and from sediments. Amphipods (especially *Gammarus*), isopods (*Caecidotea*), cyclopoid copepods, and most of the cladoceran species were predominant in yellow perch stomachs. Analyses performed in the Baseline Modeling Report (USEPA, 1999) indicated that larger yellow perch also eat small clams and snails as well as oligochaete worms; all of these are common benthic species. Predominant insect larvae in the guts of yellow perch (6 – 14 cm length) included species that are readily available on the surfaces of plants and on sediments as well as diptera pupa which tend to be planktonic.

Our qualitative assessment of the Exponent (1998a, 1998b) data for yellow perch in the 6-14 cm size range suggests that benthic invertebrates could comprise as much as 70% of the diet. However, for this ERA it has been estimated that up to 56% of the diet could consist of invertebrates that live primarily in the water (e.g., zooplankton and on plants). Some of the benthic invertebrates associated with the sediments could also be strongly influenced by surface water (e.g., *Gammarus* spp.). Therefore, the component of the invertebrate diet that is exposed to surface water could be even greater than that indicated from a simple division of benthic and non-benthic. It was estimated that this component could be as much as 65% (and might be even higher).

Oligochaete worms were observed in the gut contents of a number of larger yellow perch (11 to 32 cm) indicating that these fish forage directly in the sediments. Larger yellow perch also probably eat fish although none were observed in the gut contents examined by MCA (USEPA, 1999). Fish are probably a small part of the diet of large yellow perch (i.e., less than 10%).

D.5.2 Range, Movement and Habitat within the Hudson River

Yellow perch are most abundant in waters that are clear and have moderate vegetation and sand, gravel or mucky bottoms. Abundance decreases with increases in turbidity or with decreases in abundance of vegetation. Adult perch prefer slow moving waters near the shore areas where there is moderate cover.

Yellow perch studied in the freshwater Cedar Lake in Illinois stayed within a 5 to 20 kilometer home range (Fish and Savitz, 1983). The fish preferred heavy and light weeded as well as sandy areas, and were virtually never seen in open water.

Yellow perch are found throughout the Upper Hudson River (MPI, 1984), particularly near RM 153 (Federal Dam) and again up near the Thompson Island Pool area.

Yellow perch prefer wetlands, embayments and shallow areas to other habitats, but can be found in all types of habitats to some degree. They primarily inhabit the freshwater portion of the estuary with an apparently even distribution of early life stage abundance from RM 77 through 153 (Texas Instruments, 1976; Carlson, 1986).

Yellow perch require a minimum dissolved oxygen concentration for all life stages of 5 mg/L-1. Seasonal lethal dissolved oxygen is 0.2 mg/L in winter and 1.5 mg/L in summer. Yellow perch are poikilothermic, requiring less oxygen in winter. Suboptimal dissolved oxygen may have acute implications, in that if a preferred habitat contains less dissolved oxygen than necessary, then fish may leave the area, subjecting them to predation, or they may experience retarded growth, impacting survivability (Piavis, 1991).

D.5.3 Reproduction

Yellow perch are among the earliest spring spawners, with spawning occurring near vegetated areas and in upstream, tidal tributaries (Carlson, 1986). In the Chesapeake River, adult yellow perch migrate from downstream stretches of tidal waters to spawning areas in less saline upper reaches in mid February through March (Piavis, 1991). Spawning occurs when water temperatures reach 45-52_F in April and May in New York waters (Smith, 1985). Males arrive at the spawning ground first. Spawning occurs in 5 to 10 feet of water over sand, rubble, or vegetation. Eggs are often draped over logs or vegetation.

D.6 Largemouth Bass *(Micropterus salmoides)*

The largemouth bass, *Micropterus salmoides*, is a relatively large, robust fish that has a tolerance for high temperatures and slight turbidity (Scott and Crossman, 1973). It occupies waters with abundant aquatic vegetation. Largemouth bass show a low tolerance for low oxygen conditions. The largemouth bass represents a top predator in the aquatic food web, consuming primarily fish. Benthic invertebrates comprise a small but significant component of the largemouth bass diet.

D.6.1 Foraging

Young largemouth bass feed on algae, zooplankton, insect larvae, and microcrustaceans (Boreman, 1981). Largemouth bass can grow to 136 grams on a diet consisting of insects and plankton. Larger prey are required to continue growth after reaching a total length of 20 mm. Young largemouth bass compete for food with a variety of other warmwater and bottom-feeding fishes.

Johnson (1983) found that the diets of juvenile fish foraging in the St. Lawrence River varied somewhat by location and length of the fish. Fish, insects including corixids, and other invertebrates made up the diets in varying proportions.

Largemouth bass longer that 50 mm total length usually forage exclusively on fish. Prey species include gizzard shad, carp, bluntnose minnow, silvery minnow, golden shiner, yellow perch, pumpkinseed, bluegill, largemouth bass, and silversides (Scott and Crossman, 1973). Cannibalism is more prevalent among largemouth bass than among many species. Cannibalism is present among largemouth bass with 10% of the food of largemouth bass 20 mm and longer consisting of their own fry (Scott and Crossman, 1973).

Largemouth bass take their food at the surface during morning and evening, in the water column during the day, and from the bottom at night. They feed by sight, often in schools, near shore, and almost always close to vegetation. Feeding is restricted at water temperatures below 10°C and decreases in winter and during spawning. Largemouth bass do not feed during spawning.

Information on feeding habits of largemouth bass in the Upper Hudson River was obtained for 73 juvenile and adult fish collected in Spring 1997 by NYSDEC and analyzed in the Baseline Modeling Report (USEPA, 1999). Sample locations included Griffin Island, Stillwater, Troy, and Catskill Creek. Thirty-one of the bass (42%) had fish remains in their digestive system and represented the most common food item for adult bass. Crayfish were occasionally consumed at most river locations, but as many as six of 20 bass collected at Catskill Creek showed evidence of crayfish consumption. Benthic invertebrates were observed in the diet of juvenile bass. It is difficult to reconstruct the amount of food eaten on a percentage basis because of many factors, including inter- and intra-species variability in biomass and differential digestion rates for different species eaten by fish. On the basis of the available data it is estimated that fish comprise between 75% and 90% of the diet. The Spring 1997 data indicate that the balance of the diet is made up of benthic invertebrates.

Exponent (1998a, 1998b) conducted gut analyses of 32 adult largemouth bass from Griffin Island, Thompson Island Pool, and Stillwater in Fall 1997 and 21 bass collected from Griffin Island and at Coveville in Spring 1998. Results were similar to those observed in the Baseline Modeling Report (USEPA, 1999). Thirty-one of the bass (58%) had fish in their digestive systems and crayfish were occasionally eaten. Smaller invertebrates (insects and crustaceans) were commonly present. Frogs were also occasionally eaten.

The Baseline Modeling Report (USEPA, 1999) analyzed the Exponent (1998a, 1998b) data to evaluate the composition of invertebrates eaten by bass. These analyses were qualitative and focused on the composition of predominant species in the gut contents of the fish. The analyses looked for associations between invertebrates in the gut contents and those that Exponent (1998a, 1998b) collected in sediments and on plants. Based on knowledge of the river, the possibility that some invertebrates are zooplankton members was also considered (not explicitly evaluated by Exponent). The analyses revealed that largemouth bass feed on a variety of invertebrates that inhabit sediments, live on plants, or are part of the zooplankton. Predominant invertebrate species observed in the gut contents of bass include amphipods (both *Hyallella* and *Gammarus)*, isopods (*Caecidotea*), cladocerans (*Bosmina, Chydorus, Eurycercus, and Simocephalus)*, cyclopoid copepods, ostracods (e.g., *Podocopa*), and some chironomid larvae (Table D-1). The crustacea observed include a number of species that inhabit the water column (e.g., *Bosmina*), occupy the littoral area and also open water (e.g., *Chydorus sphaericus*) , and live in close association with surface sediments (e.g., *Gammarus* and *Caecidotea*). The amphipod *Gammarus* spp. also occur in the plankton of the river and are likely influenced by both water and surficial sediment exposures. The isopod is probably a surface deposit feeder and is also probably influenced by surface water as well as surficial sediment exposure.

It is difficult to reconstruct the amount of food eaten on a percentage basis because of many factors, including inter- and intra-species variability in biomass and differential digestion rates for different species eaten by fish. Further, food consumption varies seasonally due to changes in the availability of different prey items. Therefore, any estimate based on a few sampling dates and locations must be viewed as a rough indication of feeding preference. On the basis of the available Hudson River data we estimate that fish comprise between 75% and 90% of the average adult largemouth bass diet. The balance of the diet is made up primarily of invertebrates including crayfish. Our estimates consider the relative size of the prey organisms as well as the frequency of prey animals in the diet. Terrestrial animals are also occasionally eaten. A qualitative assessment of the Exponent (1998a, 1998b) data suggests that 54% and 68% of the invertebrates are associated with sediments and 34% to 46% are associated with water column exposures. Invertebrates associated with sediments such as amphipods and isopods are also likely influenced by water exposures. The extent to which water or sediment affect the body burdens of surface deposit feeders and meroplanktonic animals such as *Gammarus* is not known.

D.6.2 Range, Movement and Habitat within the Hudson River

Largemouth bass have distinct home ranges and are generally found between 8 and 9 kilometers of their preferred range (Kramer and Smith, 1960). Kramer and Smith found that 96% of the fish remained within 91 meters of their nesting range. Fish and Savitz (1983) found that bass in Cedar Lake, Illinois, have home ranges from 1,800 to 20,700 square meters. The average home range was 9,245 square meters and the average primary occupation area, defined as that area within the home range in which the fish spends the majority of its time, including foraging, was 6,800 square meters.

Largemouth bass are almost universally associated with soft bottoms, stumps, and extensive growths of a variety of emergent and submerged vegetation, particularly water lilies, cattails, and various species of pond weed. It is unusual to find largemouth bass in rocky areas. Largemouth

bass are rarely caught at depths over 20 feet, although they often move closer to the bottom of the river during the winter.

Mobility of largemouth bass also varies seasonally. Daily movements increase with temperature from March through June, but decrease sharply during the hottest months (Mesing and Wicker, 1986). Activity during warmer seasons occurs primarily near dawn and dusk, while cool-water activity is most extensive in the afternoon.

A 1984 Malcolm-Pirnie report prepared for New York State describes the results of a fish survey taken that same year. The results are reported as number of fish by habitat type as well as number of fish by lock pool for the Upper Hudson River and associated canals. The numbers shown are not significant in terms of absolute numbers, but rather provide a qualitative indication as to the relative distribution of fish within each habitat area and within each lock pool. Largemouth bass were found in each of the lock pools.

Largemouth bass were found throughout the Upper Hudson River in significant numbers. Major concentrations of fish were within areas where submerged and emergent vegetation, overhang, and bottom debris provided adequate cover (MPI, 1984). Largemouth bass were not found in the main, natural channel of the river nor in the rapids.

In the Lower Hudson River Estuary, Carlson (1986) found that largemouth bass preferentially winter in five major areas:

- Coxsackie Bay (roughly RM 130)
- The mouth of the Catskill Creek (RM 115)
- The mouth of the Esopus Creek (RM 103)
- The mouth of the Rondout Creek (RM 92)
- The mouth of the Wappinger Creek (RM 67)

Largemouth bass prefer to establish habitats near dense vegetation not just during winter, primarily near milfoil (*Myriophyllum verticillatum*) (Carlson, 1992). A study of largemouth bass in two freshwater lakes in central Florida found a positive correlation between the use of specific habitats in proportion to the availability of those habitats to the fish (Mesing and Wicker, 1986). Vegetative habitat covers included *Panicum* spp., cattails (*Typha* spp.), and water lilies (*Nuphar* spp.).

In a 1982 survey of the Lower Hudson River Estuary (Carlson, 1986), largemouth bass were found to prefer vegetated backwater and tributary locations, with a few fish caught in rock piles and tailwater. This suggests a preference for nearshore areas rather than the main channel.

D.6.3 Reproduction

Largemouth bass mature at age five and spawn from late spring to mid-summer, in some cases as late as August. Male largemouth bass construct nests in sand and/or gravel substrates in areas of nonflowing clear water containing aquatic vegetation (Nack and Cook, 1986). This

aquatic vegetation generally consists of water chestnut (*Trapa natans*), milfoil (*Myriophyllum verticillatum*), and water celery (*Valisneria americana*).

Females produce 2,000 to 7,000 eggs per pound of body weight (Smith, 1985) and leave the nest after spawning.

D.7 Striped Bass *(Morone saxatilis)*

The striped bass, *Morone saxatilis*, is an anadromous species that enters the Hudson River to spawn throughout the estuarine portion of the river, but particularly upstream from the saltfront. NOAA (1985) reported that striped bass spawn primarily 80 to 113 kilometers north of the Battery around Poughkeepsie in May and June. While most adults return to the sea after spawning, some remain within the estuary for a period of time. Juveniles remain in the Hudson River estuary until their second year when they begin their annual offshore migration (NOAA, 1985). Young of the year gradually move downstream during the summer months and move out of the river, primarily remaining in the lower estuary, during the winter. Historically, striped bass were an important Hudson River fisheries species, but high polychlorinated biphenyl levels closed the fishery in 1976.

D.7.1 Foraging

Striped bass are voracious, carnivorous fish that feed in groups or schools and alternate periods of intense feeding activity with periods of digestion (Raney, 1952). Peak foraging time for juveniles is at twilight. Adults feed throughout the day, but forage most vigorously just after dark and just before dawn. Adults typically gorge themselves in surface waters, then drop down into deeper waters to digest their food. Seasonally, adult feeding intensity lessens in the late spring and summer. Feeding ceases during spawning.

Striped bass feed primarily upon invertebrates when they are young, consuming larger invertebrates and fish as they grow larger. Post yolk-sac larvae feed upon zooplankton. Hjorth (1988), in a study of Hudson River striped bass larvae, found that copepods and adults of the calanoid copepod *Eurytemora affinis* were the most frequently selected prey item. Hudson River striped bass larvae also fed upon cladocerans, especially *Bosmina* spp. Copepods and cladocerans are the most common zooplankton in the Hudson River during times that striped bass larvae are present (Texas Instruments, 1980).

A study by the Hudson River power authorities (Texas Instruments, 1980) found that striped bass up to 75 mm preferred amphipods *Gammarus* spp., calanoid copepods, and chironomid larvae. Fish from 76-125 mm preferred *Gammarus* and calanoid copepods. Those from 126-200 mm preferred a fish prey, *Microgadus* tomcod.

Fish are generally considered to make up the bulk of the diet of adult striped bass. Researchers commonly find engraulids and clupeids the most the most common prey

(summarized in Setzler et al., 1980). Because striped bass feed in schools, schooling species of fish generally comprise a large portion of the diet. Striped bass are known to gorge themselves upon schooling clupeids and engraulids, concentrating their feeding activity upon whatever species is most abundant. Many other species have also been noted in striped bass diets, for example, mummichogs, mullet, white perch and tomcod. Invertebrates also may persist in the diet of adult striped bass. Schaefer (1970) found that in Long Island Sound, fish from 275-399 mm fork length fed primarily (85% by volume) upon invertebrates, primarily the amphipods *Gammarus* spp. and *Haustorius canadensis* and the mysid shrimp *Neomysis americana*. Fish from 400-599 mm divided their diet between fish (46%) (bay anchovy, Atlantic silverside, and scup) and amphipods. Sixty percent of the diet of fish from 600-940 mm in length was made up of fish, but even these larger animals consumed amphipods, mysids, and lady crabs. Schaefer hypothesized that the continued importance of invertebrates in larger fishes diets may have resulted from turbidity in the surf zone making it difficult to pursue fast-swimming fish.

D.7.2 Range, Movement and Habitat within the Hudson River

Striped bass are anadromous, spawning in tidal rivers, then migrating to coastal waters to mature. Abundant data on distribution and abundance of early life history stages of striped bass are available, because the Hudson River utilities have conducted annual surveys of the distribution of striped bass in the Hudson River since 1973. Field sampling has been conducted from New York City, the George Washington Bridge at RM 12, to the Federal Dam. Since 1981 the sampling programs have been adjusted to emphasize collection of striped bass. Additionally, the utilities have sponsored mark-recapture studies of striped bass (e.g., McLaren et al., 1981). These studies documented movement of the species within and outside the river.

The upstream spring migration of adult striped bass begins in March and April and ranges up to the Federal Dam. As young striped bass grow during the summer, they move downstream. Even at the egg stage, striped bass can be found throughout the Hudson River Estuary, although peak abundances of eggs and larvae are usually found from the Indian Point to Kingston reaches of the river, approximately RMs 100-150 (LMS, 1992). Downstream movement is partially determined by flow rate.

At approximately 13 mm total length, striped bass form schools and move into shallow waters (Raney, 1952). In the Hudson River, young-of-the-year striped bass begin to appear in catches during early July. They move shoreward as well as downstream throughout the summer and are usually found over sandy or gravel bottoms (Setzler et al., 1980). The utilities' studies typically find peak catches of young-of-the-year fish at RM 35, at the southern end of Croton-Haverstraw Bay (LMS, 1992).

Dovel (1992) summarized movements of young striped bass within the river based upon studies conducted by the utilities and others. He found that young striped bass congregate in the vicinity of the salt front during the winter, although movements in the Lower Hudson River continue throughout the winter. During the spring, some yearling striped bass continue to emigrate from the river, while other move upstream. Some young-of-the year fish leave the estuary during the summer and fall (Dovel, 1992). By their second year, most striped bass have left the river, except for their returns during spawning migrations.

D.7.3 Reproduction

In the Hudson River, striped bass spawn above the salt front and potentially as far upstream as the Federal Dam At RM 153. On average, however, they do not spawn as far upstream as white perch and will generally spawn 80 to 113 kilometers north of the Battery around Poughkeepsie in May and June. During periods of low freshwater flow, striped bass spawn further upstream than in years of high flow. Age at sexual maturity of striped bass depends upon water temperature (Setzler et al., 1980). Males mature at approximately two years, and females mature later. Spawning is triggered by sudden rises in temperature and occurs at or near the surface. Spawning occurs in brief, explosive episodes. Eggs are broadcast into the water, where a single female may be surrounded by as many as 50 males.

D.8 Shortnose Sturgeon *(Acipenser brevirostrum)*

The shortnose sturgeon, *Acipenser brevirostrum*, is the smaller of two sturgeons that occur in the Hudson River. Both the shortnose and Atlantic sturgeons have been prized for their flesh and their eggs for caviar, but sturgeons were also purposely destroyed when they became entangled in the shad nets that were once common on the Hudson River. The shortnose sturgeon has been listed on the federal endangered species list since 1967.

The life cycle of the shortnose sturgeon is typically divided into four intervals (Bain, 1997):

- Non-spawning adults;
- Spawning adults;
- Eggs and larvae; and,
- Juveniles.

D.8.1 Foraging

No field studies have documented the diets of larval shortnose sturgeon. Buckley and Kynard (1981) observed post yolk-sac larvae that they had hatched in the laboratory to feed upon zooplankton.

Juvenile shortnose sturgeon feed mostly upon benthic crustaceans and insect larvae (summarized in Gilbert, 1989). Juveniles of 20-30 cm fork length have been recorded as feeding extensively upon cladocerans. Adult fish feed indiscriminately upon bottom organisms and off emergent vegetation. Food items of juvenile and adult fish include polychaete worms, molluscs, crustaceans, aquatic insects, and small bottom-dwelling fishes (Gilbert, 1989).

Juveniles and adults generally feed by rooting along the bottom, consuming considerable mud and debris with food items. As much as 85-95% of their stomachs may contain mud and other non-food material. Conversely, shortnose sturgeon may also feed upon gastropods that live

upon vegetation. Shortnose sturgeon from New Brunswick and South Carolina have been reported as including almost exclusively gastropods with no non-food matter.

During periods of intense growth and feeding (late spring through early fall), shortnose sturgeon consume primarily insects and crustaceans, with molluscs comprising somewhere between 25 and 50% of the diet (Bain, 1997).

Shortnose sturgeon mostly feed at night or when turbidity is high, when they move into shallow water to feed. Adults move into areas as shallow as 1-5 m and forage among the weeds and river banks. Feeding occurs in deeper water during the summer, possibly in response to water temperature. The relatively little feeding occurs during the winter also occurs in deeper waters.

Shortnose sturgeon are not thought to feed in groups or schools. Mark-recapture data (Dovel et al., 1992) suggest, however, that fish tend to move as groups. Fish of the same group would therefore tend to eat in the same general areas.

D.8.2 Range, Movement and Habitat within the Hudson River

Shortnose sturgeon are found throughout the portion of the Hudson River below the Federal Dam. They are considered anadromous because they are sometimes taken by commercial fishermen at sea. However, their movements are more restricted than Atlantic sturgeon, and most of the Hudson River population probably does not leave the river. The fish does not require a marine component to its life cycle: a landlocked population in the Holyoke Pool, part of the Connecticut River system, persisted from 1848 until a fish ladder was constructed in 1955.

Adult shortnose sturgeon can be found in the center channel of the river from late spring through early fall (Bain, 1997). Eggs typically adhere to solid objects along the bottom of the river, and newly hatched embryos also tend to congregate on the bottom (Bain, 1997).

Adult shortnose sturgeon have been shown to overwinter in Esopus Meadows, approximately at RM 90 (Dovel et al., 1992), in the Croton-Haverstraw region, approximately RM 35 (Geoghegan et al., 1992). It has been shown that nonspawning adults behave differently from sapwning adults. Nonspawning adults tend to concentrate in brackish waters. Thus, it is typically nonspawning adults overwintering in the Haverstraw region.

Adult fish migrate upstream to spawn in the upper reaches of the portion of the Hudson River south of the Federal Dam in spring and then disperse downstream to feed during the summer. They can be taken throughout the fresh waters of the tidal portion of the river during the summer months.

The size of the nursery area for shortnose sturgeon larvae and young is difficult to determine, because few specimens are collected. Based upon the utilities' collections of young of the year in Haverstraw Bay, Dovel et al. (1992) presume that the young fish occupy the same freshwater portion of the estuary as do the adults of the species.

D.8.3 Reproduction

Shortnose sturgeon do not spawn every year, and non-spawning adults have shown different migratory behavior from spawning adults (Bain, 1997). When spawning, shortnose sturgeon spawn in the upper reaches of the estuarine portion of the Hudson River, approximately RMs 130-150. Spawning is limited to the last two weeks in April and the first two weeks in May. Throughout its range, the shortnose sturgeon spawns at water temperatures of 9-14°C (summarized in Crance, 1986). Dovel and his co-workers (1992) found that in 1979 and 1980, spawning in the Hudson River occurred at water temperatures of 10-18°C.

Age and size of the fish at maturity varies by latitude (Gilbert, 1989). In the Hudson River, females first spawn at approximately 9-10 years and males at 11-20 years. Spawning does not occur each year and is most likely controlled by environmental factors rather than by endocrinology.

Shortnose sturgeons produce approximately 40,000-200,000 eggs per spawning in New York waters.

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TABLE D-1
PREDOMINANT FOOD ITEMS IN HUDSON RIVER FISH
(NOTE: LESS COMMON ITEMS ARE NOT LISTED)

	Largemouth bass	Pumpkinseed sunfish	Brown bullhead	Yellow perch	Spottail shiner	White perch
PLANT MATTER						
Algae	**				***	
Vegetation						
BRYOZOA						
Bryozoa statoblasts					***	
BIVALVE MOLLUSCS (CLAMS)						
Pisidium		***		X		
Sphaerium		***	***			
GASTROPOD MOLLUSCS (SNAILS)						
Gastropods		***		X		
Planorbidae		***				
Valvata bicarinata		***				
OLIGOCHAETE WORMS						
Oligochaete worms			XX	X		
AMPHIPOD CRUSTACEANS						
Amphipod	**	***	***	***,XXX	****	XX
Gammarus spp.	**	****	****	****		
Hyalella azteca	**			**		
ISOPOD CRUSTACEANS						
Caecidotea	**	**	***	***,XXX		
CLADOCERAN CRUSTACEANS						
Bosmina longirostris	**					
Camptoceerus				***	**	
Chydorus				***	**	
Chydorus sphaericus	***		***	***	****	
Cladocera			**	***	****	
Eurycercus	***		**	****	****	
Pleuoxus denticulatus					***	
Sida				***		
Simocephalus serrulatus	**		**	***		
COPEPOD CRUSTACEANS						
Cyclopoid copepods	**			****	**	

TABLE D-1
PREDOMINANT FOOD ITEMS IN HUDSON RIVER FISH
(NOTE: LESS COMMON ITEMS ARE NOT LISTED)

	Largemouth bass	Pumpkinseed sunfish	Brown bullhead	Yellow perch	Spottail shiner	White perch
OSTRACOD CRUSTACEANS						
Ostracod					****	
Podocopa	**			**	**	
AQUATIC INSECTS						
Chaoborida						
Chaoborus			**			
Chironomidae						
Ablabesmyia annulata		**				
Ablabesmyia amallochi				**		
Chironomus spp.	**	**	**	**	****	XX
pupa		***		***	***	
Cryptochironomus						XX
Cricotopus/OrthocaldiusOrtho		**				
Dicrotendipes modestus		***		**	***	XX
Dicrotendipes neomodestus		***			***	XX
Polypedilum		**				XXX
Procaldius bellus			**			
Procaldius			**			XX
Tanytarsus spp.		***				XX
Ephemeroptera						
Caenis				**		
Odonata						
Coenargi				**, X		
Enallagma				**		
Tabanidae						
Tabanidae					**	
Trichoptera						
Oecetis					***	
Orthotrichia					**	
Trichoptera larvae unid.					***	
ARACHNIDA						
Fish (unidentified species)	****		observed		***	

Notes: *s are based on Exponent (May, 1998); Xs are based on MCA analysis (USEPA, May 1999)
number of */x indicate quantitative estimate of prevalence

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LIFE HISTORY AND ECOLOGY OF AVIAN RECEPTORS

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APPENDIX E
LIFE HISTORY AND ECOLOGY OF AVIAN RECEPTORS

Hundreds of birds species representing various trophic levels are found along the Hudson River. Avifauna are primarily exposed to Hudson River PCBs via the aquatic food chain. The tree swallow, mallard, belted kingfisher, great blue heron, and bald eagle were selected as receptors to represent insectivorous, piscivorous, and omnivorous feeding strategies.

E.1 Tree Swallow (*Tachycineta bicolor)*

Tree swallow sexes are dimorphic in plumage. The adult male is dark, greenish-blue on the dorsum and pure white on the ventrum, while the adult female is dullish brown on the dorsum fading to a lighter brownish hue before changing to a dull white on the breast and ventrum (Farrand, 1983). The tree swallow can be differentiated from other swallow species by its forked tail.

North American range adults and Upper Hudson River meristic measurements (body mass, wing chord lengths) are provided in Table E-1. Tree swallows are seasonal residents in the Hudson River Valley and migrate to warmer climates by early fall. They are gregarious and flock in large numbers for feeding and migration.

Table E-1
Tree Swallow - Meristic Measurements

Measurement	N. American Range [1]	Hudson River Female[2] (n=18)	Hudson River Male[2] (n=22)
Total length (cm)	13 - 18	NA	NA
Wing cord length (mm)	110.0 - 125	116.2 +/- 2.3	118.7 +/- 3.0
Adult body mass (gm)	17.0 - 25.5	20.6 +/- 1.5	21.0 +/- 1.6
Source**:** 1: Robertson et al. 1992; Dunning 1993; Peterson 1980**;** 2: Secord et al. (1997)**.** Notes: NA = Not Available			

E.1.1 Habitat, Home Range, and Migration

Tree swallows are found in open areas, in the vicinity of water, where flying insect prey are abundant. Preferred habitats include river valleys, lakes, marshes, and flooded swamps and beaver ponds with many dead and hollow trees standing in the water or along the shore. However, they are common around fields and meadows if nesting sites are available and there is some open water nearby (Andrle and Carroll, 1988). Tree swallows are indigenous to the Hudson River shoreline where they actively feed over the river channel on flying insects. No data exist regarding average distance traveled between nest site and feeding location, but in the absence of suitable open space individuals may fly several kilometers to suitable feeding areas (Robertson et al., 1992).

Annual migration occurs between wintering range and breeding ranges. Winter range extends from the Carolinas southward although in mild winters a holdover population may be present in the southern extremes of the breeding range including areas of eastern Massachusetts and Long Island, New York (Robertson et al., 1992). Most populations from the East Coast and Great Lake areas migrate along the Atlantic Coast as far South as Florida and the Caribbean. Tree swallows arrive in the northern breeding areas from February through April, with the typical peak in March. Fall migration from the breeding areas to the winter range from July through September with late August being the peak migrational period (Tyler, 1942 cited in Robertson et al., 1992). Tree swallows are assumed to spend April to July at a minimum and February to September at a maximum period in the northeast, equaling a residency period of 122 to 242 days per year.

E.1.2 Feeding Habits and Diet

Tree swallows are insectivores that feed upon flying insects. Feeding flights occur within the open areas at heights up to 50 m or more. Dense feeding groups of swallows can gather over surface water and fields when large swarms of insects form during periods of little or no wind (Tyler, 1942). Tree swallows are highly gregarious feeders by nature. They often feed in the open habitat in the vicinity of their nest cavities. Hudson River tree swallows appeared to prefer feeding areas near their nest sites (Secord and McCarty, 1997).

A variety of insect orders make up the diet of tree swallows and include terrestrial forms of semi-aquatic insects belonging to the Diptera, Hemiptera, Ephemeroptera, Zygoptera, Anisoptera, Plecoptera and Trichoptera. Because they feed over open water or running water, their diet consists almost exclusively of emergent forms of aquatic insects. Secord and McCarty (1997) collected insect boluses (N=27) from nesting tree swallows along the Hudson River in Saratoga County, New York. Analyses of insect taxa in the boluses revealed that flying insects with partial aquatic life histories represented 50% to 98 % of the insects captured. Ephemeroptera was the dominant order (60%) using total number of prey items in all

boluses (Secord and McCarty, unpublished data). Percent composition represented by other insect orders included Diptera 25%, Plecoptera 3.6%, Zygoptera/Anisoptera 3.3%, Trichoptera 1.5%, and Hemiptera 0.8%.

The size of the prey captured ranged from 1.0 to 42.0 mm total length, although there is an apparent selectiveness for prey items less than 10 mm (Secord and McCarty, unpublished data). Insects greater than 10 mm in length accounted for less than 10% of the insects captured by tree swallows (Quinney and Ankney, 1985). The predominance of flying insects was also observed in Ithaca, NY populations. McCarty and Winkler (in press) found the diet on average to consist of 58% Diptera, 17.6 % Hemiptera, 13.7% Odonata and 10.9% other insects.

In the absence of insect prey, early spring migrants and overwintering populations in the North have been documented as feeding upon wax myrtle berries (*Myrcia* sp.). Swallows will on occasion capture emerging insects directly from the water surface; however, most are captured in flight.

Feeding typically occurs throughout the day with the most intensive feeding period being between late morning through late afternoon (Cohen, 1984). McCarty and Winkler (in press) found that feeding tree swallows remain close to their nest boxes (100 to 200 m) and feed within 12 m of the ground.

E.1.3 Reproduction

Tree swallows nest in abandoned or excavated woodpecker holes, natural cavities in standing trees, and nest boxes. They compete directly with other secondary cavity nesting species for suitable nesting sites. Edge areas for nesting include those with standing snags with cavities to provide suitable nesting sites and snags or free standing poles with wires to provide resting perches. The presence of suitable nesting cavities is often an important limiting factor in the distribution and density of breeding pairs.

Although gregarious during feeding, males and breeding pairs are highly territorial with regard to nest site selection and defense of nest cavities. Preferred nest spacing appears to be 10 to 15 m, although under high density cavity availability, pairs may nest as closely as 1 to 3 m (Robertson and Rendell 1990; Harris 1979).

Males arrive first at the breeding area and select and defend a suitable nest cavity. Upon arrival of the females, breeding pairs are formed and the females begin to build nests. In southern Ontario, Stutchbury and Robertson (1987) found nest building to begin during the period of late April through early May. Principal building materials of the nest cup are vegetation (mostly grasses) and feathers, with nesting material being dependent upon availability of materials in vicinity of the nest. Nest cup quality and construction has been a casual indicator of nestling growth and nest success (Secord and McCarty, 1997). Eggs are laid 10 to 14 days following completion of the nest. In New York State (NYS) populations, egg

laying has been documented to occur from early May through mid-July (May 5 - July 18) (Andrle and Carroll, 1988).

A single brood typically consists of three to five eggs. If the first brood fails, a second brood may be laid (Robertson et al., 1992). Across North America, the incubation period is 11 to 19 days and in NYS populations ranges 13 to 16 days (Robertson et al. 1992; Andrle and Carroll 1988). At hatching, nestlings typically weigh 1.5 to 1.7 grams (Robertson et al., 1992). Secord and McCarty (1997) reported mean nestling weights of 1.56 to 1.80 grams from Hudson River populations. Nestling growth and development is rapid with both adults feeding the nestlings. Adults make about 10 to 20 trips per hour and average 28 mgs of insects per trip (Leffelaar and Robertson, 1986; Quinney, 1986; and Williams, 1988 cited in Robertson et al., 1992). In NYS populations, fledgling occurs 16 to 24 days following hatching. This period has been documented to occur from early June to early August (June 6 to August 2) with peak occurring in late June (Andrle and Carroll, 1988).

E.2 Mallard (*Anas platyrhychos*)

The mallard (*Anas platyrhychos*) is one of the most common species of waterfowl which is associated with the Atlantic flyway and NYS waterfowl populations (Bull, 1998). The species is dimorphic in plumage coloration with males being more brilliant in coloration than females. Drakes (i.e., males) are characterized by a yellow bill, dark green plumage about the head and crown, a narrow white ring above the base of the neck which demarks dark brown plumage on the breast extending to the ventrum of the body. The remainder of the body is slate gray with a black rump and the legs and feet are bright orange. Hens (i.e., females) are light brown to tan with dark brown mottling on the plumage extending from the base of the neck to the rump. The speculum of secondary feathers is dark purple with a trailing edge of white along the tips (Farrand, 1983). Body mass between the sexes is comparable with males typically weighing a little more than the females. Drakes average 1,246 grams (+/- 108) while hens average 1,095 (+/- 106) in body mass. Drake wing length range 290 to 294 mm and the range for hens is 270 to 275 mm (Delnicki and Reinecke, 1986). Body weights fluctuate between the spring/summer and winter/fall seasons due to build up and use of fat stores (Heitmeyer and Fredrickson, 1990).

E.2.1 Habitat, Home Range, and Migration

The mallard is among the most tolerant of waterfowl to human disturbance and development and even abides human incursion at feeding and resting areas. It is a common inhabitant of urban ponds and lakes and developed riverine corridor (Peterson, 1980), although the mallard prefers tidal and non-tidal freshwater wetlands, river and estuaries (USEPA, 1993a). Mallards in riverine wetland habitats in Minnesota occupied an average home range of 540 to 620 hectares with a range in size of 40 to 1,440 hectares (Kirby et al., 1985). Gilmer et al. (1975 cited in USEPA, 1993a) reported that home range

during laying periods were smaller than during nesting periods and that females defend a smaller range than male.

Mallards live in the Hudson River Valley both year-round and seasonally. They are gregarious in nature and flock in small to large numbers for feeding, resting, and migration. Migration is dependent upon the severity of the winter and the availability of open water. The Hudson River Valley has been identified as supporting one of the largest resident and migrant populations of mallards in NYS (Bull, 1998). Mallards in the Hudson River Valley are expected to be year-round residents as long as there is open water. A noticeable increase in the mallard population during spring suggests that many migrants supplement the resident population (Bull, 1998).

Migratory and resident mallards have been increasing throughout NYS (Andrle and Carroll, 1988). This increase is attributed to the establishment of resident, non-migrant populations in many urban and suburban parks and historical introductions by NYSDEC. Migration in mallards can take the form of both local and long distance movements, with distance being dependant on proximity of open water with suitable foraging areas. In addition to the local patterns in movement of mallards during the winter to ice free areas, long distance migrants from Canada also join more locally resident birds at overwintering locations. Palmer (1976 cited in USEPA 1993a) reported that northern populations of the mallard migrate to overwintering areas from late September through November. Spring migration, where individuals return to breeding areas, extends from late March through April in northern areas, inclusive of NYS.

E.2.2 Feeding Habits and Diet

Mallards are omnivorous in diet with the majority of the diet being composed of various parts of plants and aquatic invertebrates. The percent composition of these items remains highly seasonal with plant material dominating in the fall and winter and aquatic invertebrates dominating during the spring and fall. Feeding upon submerged vegetation and aquatic invertebrates in aquatic habitats is known as “dabbling”. This feeding behavior consists of submersion of the head, neck and breast beneath the water surface, allowing the mallard to filter through bottom material in search of aquatic plants and invertebrates in the detritus and surficial sediments. This method of feeding restricts the mallard from feeding in deep water areas, although it can feed on submerged matter near the water surface.

A variety of feeding studies involving crop and stomach analyses have documented that the majority of the diet in wild mallards in riverine and wetland habitats is derived from aquatic sources (USEPA, 1993a). Agricultural crops including grains and corn, when available, can also contribute to the diet. Review of the studies summarized in USEPA (1993b) reveal that the fall and winter diet consists of largely of plant material (mean percent composition of 98.7%; range 97 -100%) with a minor component of animal material (mean percent composition of 1.3%; range 0-3.2%). During the spring and summer, the diet appears to be near equal in composition of plant and animal matter. Composition of the diet averages 57%

animal matter (range 8.4 - 99%) and 43% plant matter (range 1.0-99.6%). It is assumed that the diet of Hudson River populations would be similar. Dominant animal matter consumed by mallards are benthic invertebrates including both insect and non-insect taxa. Dominant non-insect taxa includes gastropods (snails), annelids (worms), amphipods (scuds), chironomids (midge larvae), and isopods (aquatic sow bugs) (Fredrickson and Reid, 1988; Delnicki and Reinecke, 1986).

E.2.3 Reproduction

The mallard has been documented to breed from almost every county in NYS and is a confirmed breeder in both the Upper and Lower Hudson River (Andrle and Carroll, 1988). It frequently breeds and nests in urban parks associated with a water body (Bull, 1998; Peterson, 1980). Although the mallard tolerates human disturbance during breeding, it prefers to nest in habitats such as wetlands with open water or marginal corridor habitats associated with developed river banks. Mallards prefer to nest in dense groundcover. Nests are located on the ground and can be some distance from a water body (Andrle and Carroll, 1988). The mallard hen selects the nest site. The nest is constructed from vegetation in the vicinity and down feathers plucked from the hens breast (Andrle and Carroll, 1988).

Clutch size varies but typically ranges from 9 to 10 eggs per nest with yearling females producing fewer eggs than mature females (Orthmeyer and Ball, 1990). One clutch per year is normal, but two clutches have been documented from NYS when the first is lost or if conditions are favorable (Andrle and Carroll, 1988). The first clutch of eggs is laid in late March to early April. The incubation period lasts approximately 23 to 29 days. A second clutch may be laid in late June or early July (Andrle and Carroll, 1988).

At hatching, ducklings are fully covered in down feathers and weigh an average of 32.4 grams (Lokemoen et al., 1990). The ducklings accompany the hen to the water and remain with the hen until they are ready to fly. Mallard broods suffer significant losses due to predation and more than half of all ducklings do not survive the first thirty days (Orthmeyer and Ball, 1990). Predators include raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), Norway rats (*Rattus rattus*), ring-billed gulls (*Larus delawarensis*), and crows (*Corvus brachyrhynchos*) (Schemnitz, 1980; Doutt et al., 1977). In addition to these predators, snapping turtles (*Chelydra serpentina*), foxes (*Vulpes vulpes*), and large predatory gamefish such as largemouth bass (*Micropterus salimoides*) and muskellunge (*Esox masquinongy*) also eat ducklings (Orthmeyer and Ball, 1990; Scott and Crossman, 1976). Ducklings grow rapidly and are ready to fly 49 to 56 days after hatching (Andrle and Carroll, 1988).

E.3 Belted Kingfisher (*Ceryle alcyon*)

The belted kingfisher is distinguished by a blue-gray dorsal plumage and mostly white underparts, a large, heavy bill, and a double peaked crest of feathers on the crown. It has a white throat and a broad white and blue-gray collar around the neck, a small white spot near the eye, and is spotted on the ventral portion of the wings and tail. The ventral side of the tail feathers remains distinctly barred with gray and white banding. The sole distinctive plumage characteristic between the sexes is the presence of a distinct rufous band crossing the chest in the female. Kingfishers have broad wing areas relative to their body size and fly with a wing beat characteristic of a deep and rapid irregular pace (Farrand, 1983).

Across their North American range adults are 31.0 to 36.0 cm total length (Farrand, 1983), and weigh 136.0 to 155.0 gms (Brooke and Davis, 1987; Dunning, 1993; Poole, 1938). Recorded meristic measures for a single Hudson River male was 31.0 cm total body length and 53.0 cm in total wing expanse (Bopp, 1999). Belted kingfishers have been reported from along the entire length of the Hudson River and breeding pairs have been documented in both the non-tidal and tidal areas of the river (Andrle and Carroll, 1988).

E.3.1 Habitat, Home Range, and Migration

Belted kingfishers are found along the shoreline of rivers, streams, ponds, and lakes, including both freshwater and brackish areas. The kingfisher diet is almost exclusively aquatic prey items and nesting usually occurs in close proximity to feeding areas. Preferred riparian areas include areas with mature woody vegetation with numerous overhangs above the water surface. The overhangs are critical for use as perching posts from which aquatic prey may be observed. Clear water conditions assist in prey capture (Bent, 1940). Artificial perches for feeding include overhead wires above the water surface and bridges.

Typically the streams and rivers selected for feeding areas are larger (4 to >16 m) permanent lotic environments with a diverse assemblage of microhabitats (i.e., riffles, pools, runs etc.) of varying depths (0.17-0.50 m) (Brooks and Davis, 1987). Banks can be steep or gradual in inclination and remain well vegetated. Feeding can occur in aquatic microhabitats with higher water velocities (i.e., riffles and runs) or more quiescent conditions (i.e., pools and runs). Generally feeding occurs in both lentic and lotic habitats, although lotic environments appear to be favored (Brooks and Davis, 1987).

Nesting always occurs in a cavity in close proximity to the feeding area. Nesting occurs in cavities that have been excavated in the steep, exposed banks of the shoreline or in riparian areas associated with the feeding habitat. Use of abandoned woodpecker holes and wood duck nests has been documented but are uncommon relative to earthen cavity sites (Andrle and Carroll 1988). The vertical inclination and height of the embankment slope appears to be a critical factor and may act as a deterrent to predators, allow for easy excavation, and prevent the nest from flooding during high flows. Brooks and Davis (1987) observed

an average inclination of 55 to 89% and a height of one to two meters above the ground in nest embankments in Ohio and Pennsylvania populations. Eroded tracks at the base of the hole from the adults dragging their feet in flight when entering the nest cavity are characteristic of kingfisher nests. Embankments subject to severe erosion and rock outcrops are characteristics that may limit nest site selection. Suitable nest sites appear to be a limiting factor in the distribution of mating pairs (Brooks and Davis, 1987).

Home range is typically defined by length of shoreline defended by mated pairs (breeding territory) and feeding areas defended by solitary adults (non-breeding). Generally, breeding pairs defend a larger habitat than solitary individuals, although considerable overlap in size occurs. Davis (1982) reported that non-breeding individuals occupied an average home range of 0.39 km of shoreline and that breeding pairs defend an average home range of 1.03 km of shoreline in Pennsylvania and Ohio populations. NYS populations are expected to occupy similar home ranges.

The kingfisher is native throughout North America. In NYS, the kingfisher can be both a seasonal migrant or a resident species throughout the year. Migrations in the northeast are dependent upon the severity of the winter season, in particular the degree of ice cover on feeding waters. During severe conditions (i.e., persistent cold and continuous ice cover) northeast populations will migrate as far south as portions of the Carolinas and Virginia. Fall migration in NYS occurs from September through October and spring migration occurs from April through June (Bent, 1940). During milder winters, the kingfisher can remain in NYS as long as a steady food supply is available and aquatic habitats remain free of ice (USEPA, 1993b). Annual residence time of this species in NYS, inclusive of the Hudson River Valley, ranges from 245 days/year (migrants) to 365 days/year (full time resident).

E.3.2 Feeding Habits and Diet

Throughout their North American range belted kingfishers are opportunistic piscivores with smaller fish species dominating the diet and larger aquatic invertebrates like crayfish supplementing the diet. While amphibians, reptiles, and small mammals have been documented as occurring in the diet, wholly aquatic prey (fish and crayfish) are the principal diet components in northeast populations (USEPA, 1993b).

Kingfishers locate aquatic prey by perching above the water surface and visual detecting the prey. All feeding occurs by sight with detection of prey being based upon movement. Capture of aquatic prey consists of the kingfisher diving from its perch into the water and physically seizing the prey with its bill. Prey detection and capture occurs within a few inches of the water surface (Davis, 1982). Water turbidity is thought to contribute to feeding success. A reduction in feeding duration during peak or storm flow periods has been observed (Brooks and Davis, 1987).

Diet studies of northeast and central North American populations (Michigan, New York, Pennsylvania, and Ohio) indicate that the typical diet of belted kingfishers ranges from 46-100% fish, 5-

41% crayfish and other aquatic invertebrates, and 0-6% amphibians, reptiles or small mammals (USEPA, 1993b). Stomach content analyses from 25 individuals from south-central NYS revealed an average diet of 72% fish, 22% crayfish/invertebrates, and 6% amphibian/reptiles (Gould, unpublished data cited in USEPA 1993b). Comparison of these data to the observed North American range shows the diets to be comparable. Fish consumed from NYS waters include salmonids, cyprinids, percids, ichtrarcids and centrarchids (USEPA, 1993b). Prey species selectivity appears to be based upon local abundance within in the aquatic community rather than species specificity. Davis (1982) observed that all fish captured by belted kingfishers in Ohio and Pennsylvania populations ranged from 4.0 to 14.0 cm in length. It is anticipated that NYS kingfisher populations would have similar size selectivity.

E.3.3 Reproduction

Males typically arrive prior to females and select and defend a breeding territory. Kingfishers are highly territorial and do not congregate in large numbers (Davis, 1982). Because of limitations of suitable excavation/nest sites breeding pairs may nest some distance away from the foraging area (Andrle and Carroll, 1988). The male and female excavate a cavity in an earthen tunnel for nesting. Tunnels are circular 8.9 to 10.0 cm wide and 7.6 to 8.9 cm high and can be excavated into the embankment up to 4.6 meters. Established breeding pairs often return to the same excavated nest cavities year after year. Excavations are often associated with other species that use earthen cavities to nest, including bank swallows (*Riparia riparia*) and rough winged swallows (*Steligidopteryx serripenniss*). Nests are devoid of nest lining material and eggs are laid on the earthen floor (Andrle and Carroll, 1988). Although belted kingfishers prefer areas with as little disturbance as possible for nest site locations, they will tolerate human incursion and have been found nesting in roadway cuts and gravel and sand quarries (Hamas, 1974).

Eggs in NYS populations are laid from April to June and a single brood is common (Andrle and Carroll, 1988). Five to eight eggs are generally laid in North American kingfishers (Peterson, 1980). Incubation lasts approximately 17 to 24 days in NYS. Both male and female feed the nestlings. At hatching, nestlings typically weigh 10.0 to 12.0 gms and grow at a rate of five to six grams per day. At fledgling, generally occurring from July through August, individuals weigh 149 to 169 gms (Brooks and Davis, 1987). The diet of nestlings and fledglings is comparable to the adult diet.

E.4 Great Blue Heron (*Ardea herodias*)

The great blue heron is the largest heron species (order Ciconiiformes) indigenous to NYS. It is a common wading bird that inhabits both freshwater and estuarine portions of the Hudson River. The USFWS considers it a migratory, non-game avian species. NYS populations are monitored by the NYSDEC Non-game Species Program (Hicks, 1999).

The sexes are similar in body size, wing span and coloration, although males are slightly larger in body mass and wing span than females (Peterson, 1980). Body size ranges 104.0 to 132.0 cm with a wing span of 1.8 to 2.2 m and a height of 1.2 to 1.5 m (Farrand, 1983). A single, male specimen collected from the Hudson River Valley and curated in the NYSM was measured to be 104 cm in body size (neck and body) (Bopp, 1999). Dunning (1993) lists average body masses as 2,576 gms for males and 2,204 gms for females. Plumage in both sexes is identical. Adults have a white head with the sides of the crown and nape being black with short plumes projected to the rear; the neck is light gray, with a whitish ventral stripe; the bill is large and yellowish; the body is blue gray; and the legs are dark brown to black in coloration (Farrand, 1983).

E.4.1 Habitat, Home Range, and Migration

Preferred habitats for feeding and breeding are riparian habitats along the shoreline of rivers, streams, lakes, and wetlands. These include both non-tidal and tidal portions of rivers and estuaries. When feeding along the shoreline of aquatic habitats, the great blue heron diet is composed almost exclusively of aquatic prey. It is semi-tolerant of human disturbance and is common along drainage ditches and river banks associated with human development, but will readily flush when approached on foot (Eckert and Karalus, 1983). Heronries are typically located in standing trees and dead snags in secluded areas with minimal human disturbance (Andrle and Carroll, 1988).

Home range can be considered in terms of both distance traveled to feeding grounds from heronries and defended foraging areas used for feeding. USEPA (1993a) gives mean ranges of 3.1 to 8.0 km linear distance (max. 24.4 km). Unit areas for foraging varied by habitats with an average area of 0.6 ha in a Oregon freshwater marsh to 8.4 ha in an Oregon estuary (USEPA, 1993a). No NYS home range data were available, but values are expected to be similar to those observed in other areas of the continental US.

In NYS, the great blue heron can be both a seasonal migrant or a resident species throughout the year as long as open water persists (Bull, 1998). Results of the Audubon Christmas Bird Count show that the great blue heron is an uncommon winter resident (CBC, 1999). Migrations in the northeast are highly dependent upon the severity of the winter season, primarily the degree of ice cover on feeding waters. During severe conditions (i.e., persistent cold and continuous ice cover) northeast populations will migrant south to portions of the Carolinas and Virginia. Fall migration in NYS populations remains unclear given the tendency of this species to linger or reside in summer grounds during the winter period. Fall migration may begin as early as mid-July. Spring migrants typically return to NYS habitats from late-March through early April (Bull, 1998). Annual residence of this species in NYS (inclusive of the Hudson River Valley) can range up to 365 days/year for year-round residents.

E.4.2 Feeding Habits and Diet

The feeding behavior in great blue herons can be characterized as a stalking and ambush approach to prey capture (Eckert and Karalus, 1983). Great blue herons are typically solitary hunters along shorelines of aquatic habitats. However, when prey is abundant (e.g., baitfish stranded in tidal mudflat shallows) great blue herons will congregate in large numbers to feed (Krebs, 1974). Feeding typically occurs throughout the day with greatest activity occurring during dawn and dusk.

Solitary feeding behaviors consists of a slow and deliberate pace in shallow water with prey being detected based upon visible movement. Maximum depth in which feeding occurs is approximately 1.5 to 1.6 m with firm bottom substrates (USEPA, 1993a). No quantitative dietary studies for the Hudson River Valley were available. Stomach contents of adults and nestlings from a southwestern Lake Erie population were found to consist of 100% fish with most fish eaten being less than 20 cm total length (Hoffman, 1978). Fish species indigenous to the Hudson River which were found in the Lake Erie study include: carp and minnows (Cyprinidae) 50% to 53%, perch (Percidae) 10% to 28%, sunfish and bass (Centrarchidae) 7% to 10%, drum (Sciaenidae) 4% to 10%, catfish (Ictaluridae) 0% to 5%, herrings and shad (Clupeidae) 0% to 5%, and aquatic invertebrates (crayfish, aquatic insects) 5% to 31% (USEPA, 1993a). While herons prefer to feed on fish, amphibians/reptiles, small mammals and insects are taken on occasion (USEPA, 1993a; Eckert and Karalus, 1983).

Herons capture fish by impaling them with their bill. They realign fish in the beak and then swallow them whole. Fish up to 0.6 m long and up to one kilogram can be captured and swallowed (Eckert and Karalus, 1983). Krebs (1974) found that smaller prey were selected more frequently because of greater abundance and less handling time. Through field observations, Krebs categorized fish size based upon comparative size of the fish captured to the length of the herons bill (assuming a 12.7 cm bill length) using the categories of small fish (< ½ bill length), medium fish (>½ to l bill length), and large fish (> l bill length). Results of the field investigation revealed a distribution in prey size of 73.4% small fish (<6.0 cm total length [TL]); 19.4 % medium fish (approximately 6.0-13 cm TL) and 7.4 % large fish (> 13.0 cm TL).

E.4.3 Reproduction

Great blue herons are colonial nesters and form heronries that in NYS range from less than 50 nests to up to 1,000 nests, given optimal nesting habitats (Bull, 1998; Andrle and Carroll, 1988). Confirmed heronries have been found in the Hudson River Valley, although the number nests appears to be smaller than those observed from central NYS (Andrle and Carroll, 1988). Selection of nesting sites remains highly selective with the availability of densely distributed large trees or standing snags or dense scrub, a local foraging habitat and minimal human disturbance being three of the most critical characteristics for location of heronries (Eckert and Karalus, 1983). Nests vary greatly in their dimensions from flimsy new

platforms of sticks 0.5 m across to bulky older structures 0.9-1.2 m across. Nests are usually 7.6 to 30.5 m above the ground (Andrle and Carroll, 1988). Mating occurs from late March through early April and egg are laid between April 15 to June 9. The nestling stage extends for approximately 60 days after hatching and fledglings leave the nest by July in NYS (Andrle and Carroll, 1988).

E.5 Bald Eagle *(Haliaeetus leucocephalus)*

The bald eagle is an opportunistic piscivore, indigenous to the US and NYS. The bald eagle, the national bird, was originally listed as an endangered species by the US Fish and Wildlife Service (USFWS) but was relisted in 1997 as threatened in the continental US. It remains listed as endangered in NYS due to low reproductive success and small population size in the state. The eagle is a large, highly distinctive species with the female being larger in body mass and wing span than the male. The adults of both sexes are similar in plumage coloration with chocolate-brown body plumage, white head and tail plumage, and eyes with pronounced yellow irises. Juvenile and sub-adult plumage is variable ranging from brownish-black to light-tan, with white spotting and marbling on the ventral side of the wings. Adult plumage is fully developed by the third or fourth year (Farrand, 1983).

Across their North American range adult, female body mass averages 5.3 kg and adult males average 4.1 kg (Dunning, 1993). Body length and wing span for North American populations (both sexes combined) range 76 to 109 cm body length and 1.8 to 2.5 m in expanse. Three specimens (2 female/1 male) collected from the Hudson River Valley (Saratoga, Albany, Rensselaer, Greene, Columbia, Ulster and Dutchess Counties) are curated in the New York State Museum (NYSM) in Albany. Body lengths recorded at the time of collection range 84 to 91 cm and body mass is 3.86 and 4.19 kg for the female specimens and 3.36 kg for the single male specimen (Bopp, 1999). The Hudson River Valley supports a limited breeding population of eagles (1 to 3 nests) and a consistent overwintering population in the Upper Hudson River area. The over-wintering population consists of both resident and migrants from more northern habitats (Nye, 1999).

Currently, the NYSDEC Endangered Species Program is conducting an intensive evaluation of bald eagle nesting success, feeding characteristics, and exposure to organochlorine contamination on nesting pairs in the Hudson River Valley (Nye, 1999).

E.5.1 Habitat, Home Range, and Migration

Bald eagles are always associated with large aquatic habitats including rivers, lakes, reservoirs, coastal marshes and bays. The close association with aquatic habitats is reflected in a largely fish derived diet for both sexes. Nesting habitat is typically secluded, and requires a tall tree (living or dead), cliff shelf, or in some cases a power tower for nest construction. Selection of a nesting site often is selected based upon height of the identified structure. Bald eagles have been documented using abandoned osprey

(*Pandion haliaetus*) nests for nesting along coastal habitats. Although the bald eagle may tolerate limited human disturbance during non-nesting periods, it is intolerant to disturbance during nesting (USEPA, 1993a). Once a nest site has been established, and suitable nesting conditions remain intact, a breeding pair will return to the same location annually to breed and nest (Andrle and Carroll, 1988).

Bald eagle home range varies with breeding state. Solitary adults occupy a larger home range than juveniles or breeding pairs. No home range data were available for resident bald eagles in the Hudson River Valley; however, reported territory sizes from other populations include studies summarized in USEPA (1993a). When expressed on a kilometer of river mile, territory sizes range 0.56 to 0.72 km during incubating and brooding periods and three to seven kilometers in the over-wintering area. Once fledged, juveniles often accompany adults who greatly expand their foraging range after abandoning the nest site (USEPA 1993a).

Bald eagles frequently migrate between wintering and breeding areas. Movement to wintering areas is largely controlled by persistent cold and freezing of water at aquatic foraging sites. One common characteristic of wintering areas is the presence of open water and a steady food supply. In winter, bald eagles often gather below hydroelectric and once through cooling power plants that provide areas of open water and a steady supply of dead and stunned fish during their operation. This occurrence has been documented on the Upper Hudson River and the Upper Delaware River. Currently a resident population of eagles has established itself near Albany on the Upper Hudson River. It is considered to be a year-round population, augmented in winter by eagles migrating south from Ontario and Quebec, Canada (Nye, 1999). The influx of winter migrants into the Hudson River Valley may double the number of eagles present in the overwintering areas. Winter migrants from Hudson Bay, Canada arrive in the Upper Hudson River area in December and overwinter in the area before returning north in late March or early April (Journey North, 1999).

E.5.2 Feeding Habits and Diet

The bald eagle is an opportunistic carnivore that feeds on variety of both living and dead animals. Fish are the principal prey and are captured live from the water surface or scavenged dead from the water and shoreline. Aquatic invertebrates appear to be negligible in the diet. Other potential prey include waterfowl, small mammals, and carrion. Eagles are highly opportunistic in diet preference and will take advantage of fortuitous circumstances in the abundance of any one prey item, though fish remain a staple in the diet (USEPA, 1993a).

In general, studies of diet characteristics summarized in USEPA (1993b) show that summer diets from inland waters are dominated by fish (>70%) with birds and mammals representing a smaller component (<30%). Winter diets appear to be more variable with fish and birds/small mammals consumed in almost equal proportion. Fish species from freshwater habitats observed in eagle diets from Maine

populations during the summer include bullheads (*Ictalurus* sp.), suckers (*Catastomus* sp.), pike and pickerel (Esocidae), smallmouth bass (*Micropterus dolomeui*), perch (Percidae), eel (*Anguilla rostrata*), carp (*Cyprinus carpio*) and herring and shad (*Alosa* sp.) and combined account for approximately 77% of the diet (Todd et al., 1982). Eagles have been observed to congregate in large numbers along stretches of rivers below hydroelectric power dams to feed on dead or stunned fish passing through the facility turbines. This behavior has been observed in NYS wintering populations in both the Delaware and Hudson River drainage basins (Nye, 1999; Nye and Suring, 1978).

Based upon limited sampling of prey remains from nests by NYSDEC staff, fish appears to dominate the diet in resident Hudson River populations. In addition, the fish component appears to remain consistent on a seasonal basis (Nye, 1999). While no quantitative dietary studies from the Hudson River species are available, observations by NYSDEC staff indicate that fish are the dominant dietary component based upon nest remains. American eel (*Anguilla rostrata*) remains were found frequently in Hudson River bald eagle nests.

E.5.3 Reproduction

The bald eagle has been the focus of a reintroduction program across NYS that began in the 1970's. Prior to 1997, the Upper Hudson River supported a small population of non-reproducing eagles (Nye, 1999). In 1997, a pair of bald eagles successfully nested and fledged a single eaglet from a nest near Albany, and in 1998 two nesting pairs in the Upper Hudson produced two fledglings, and a third nesting pair produced a single egg that failed to hatch (Nye, 1999).

In NYS, bald eagles lay their eggs in early March and a single brood is produced (Andrle and Carroll, 1988). Incubation ranges 28 to 46 days and the nestling stages lasts approximately 10 to 11 weeks. Fledglings are reported from late May and immature eagles may be found with adults through the rest of the summer. Eagles hatched from Hudson Valley nests appear to remain in the area year-round with the existing population (Nye, 1999). No breeding or resident populations have been reported in the Lower Hudson River.

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PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

APPENDIX F

LIFE HISTORY AND ECOLOGY OF MAMMALIAN RECEPTORS

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APPENDIX F
LIFE HISTORY AND ECOLOGY OF MAMMALIAN RECEPTORS

The diverse habitats found along the Hudson River Valley support a wide variety of indigenous mammals. Many mammals use the Hudson River for feeding, shelter, or reproduction. Four representative mammals were selected to serve as receptor species to assess risks in this assessment. The life history of these four species, the little brown bat, raccoon, mink and river otter are discussed below.

F.1 Little Brown Bat (*Myotis lucifugus*)

The little brown bat is a small insectivorous bat species found throughout the US and Canada. This species is indigenous to New York State (NYS) where it is considered a non-game species and is regulated by NYSDEC (Hicks, 1999). Little brown bats are nocturnal and feed in open forest canopies, open shorelines, and basins of rivers, lakes, streams, and wetlands.

Little brown bats are characterized by dark walnut-brown fur on their dorsal side, lighter fur on their belly, and only limited hair on their wings. They have small ears and a simple snout (Doutt et al., 1977; Burt and Grossenheider, 1976). Males and females are similar in size, although pregnant females may weigh twice as much as males. Meristic measurements are provided in Table F-1. Measurements of little brown bats in the New York State Museum (NYSM) from Saratoga, Albany, Rensselaer, Greene, Columbia, Ulster, and Dutchess Counties, New York (NY) were similar to other North American populations (Bopp, 1999).

Table F-1
Little Brown Bat - Meristic Measurements

Measurement	N. American Range	NYSM Female (n=6)	NYSM Male
Body length (cm)	5.0 - 5.72	8.5 - 9.0 ([1]	7.6 - 10.9 ([1]
Tail length (cm)	2.30 - 2.54	NA	NA
Wing span (cm)	25.4	NA	NA
Forearm length (cm)	5.59	NA	NA
Adult body mass (gm)	5.0 - 7.0	4.2 - 9.4 (	5.5 - 7.7 (

Source: Bopp 1999; Doutt et al. 1977; Burt and Grossenheider 1976.
[1] Body length recorded is sum of body and tail lengths. NA= Not Available.

F.1.1 Habitat and Home Range

Little brown bats are highly social, congregating in large numbers to form both winter and summer colonies. They require sheltered locations for summer roosts and winter hibernation. Because they are nocturnal and relatively inactive during the day, they are tolerant of human disturbance and may use man-made structures for roosting (Doutt et al., 1977). Their tolerance to human disturbance makes them one of the more commonly encountered bat species in developed areas.

During the summer little brown bats can be found in a wide variety of areas with suitable summer roosts. Summer roosts are typically any protected refuge that allows for shelter from normal weather conditions and direct sunlight. In developed areas preferred summer roosts are in abandoned barns/structures and attics of older buildings, while in undeveloped areas they include caves, rock crevices, trees, and abandoned mines (Rabe et al., 1998; McManus and Esher, 1971; Davis and Hitchcock, 1965). In general, the larger and more secluded the roost, the larger the colony will be in size. Well established summer roosts can host colonies as large as several hundred individuals. Summer colonies in NYS range from 12 to 1,200 individuals and average 280 to 400 individuals (Davis and Hitchcock, 1965). Large summer roosts are dominated by females. For example, summer colonies near Cincinnatus and Malloryville, NY were 97% and 100% female, respectively.

Little brown bats are true hibernators and congregate in large numbers in winter hibernacula (i.e., hibernating shelters). Winter hibernacula are typically deep mines or caves that are often shared by several species. Deeply recessed tunnels, such as mine shafts, provide stable environmental conditions for bat hibernacula (McManus and Esher, 1971). Key environmental characteristics in winter hibernacula include seclusion (i.e., lack of light penetration), temperature between 2.0- 8.9 , 0-100% humidity, and minimal draft (Doutt et al., 1977; McManus and Esher, 1971; Orr, 1966). As the outside temperature fluctuates, atmospheric conditions in shallower depths near the mine or cave entrance also change. In response to changes hibernating colonies may move within the hibernaculum to follow preferred environmental conditions, but colonies rarely move between hibernacula (Orr, 1966). Up to 25,000 individual bats were observed in several large winter hibernacula in Essex County, NY (Davis and Hitchcock, 1965), while other NYS hibernating colonies had 200 to 800 individuals (Griffin, 1945). Winter hibernating colonies have a more equal distribution of males and females than summer colonies (Wimbatt, 1945; Griffin, 1940).

In response to colder temperatures and diminishing prey, little brown bats move from summer roosting and maternity colonies to winter hibernacula that can be up to several hundred miles from summer refuge (Cockrum, 1956; Griffin, 1940). Seasonal movements occur before and after hibernation. While not truly migration in definition, this movement results in temporary displacement of little brown bat populations from summer refuges/feeding areas and dispersal to other summer and winter refuges. Bats from colonies in Albany and Schoharie Counties, NY traveled 14.4 to 33.8 km between summer and winter locations (Davis and Hitchcock, 1965), while New England populations were found to travel 8.7 to 105 km between summer colonies and winter hibernacula (Griffin, 1945).

In Essex County, NY little brown bats were found to return to winter hibernacula from September to October (Davis and Hitchcock, 1965). Hibernating colonies can be found from

September through early June. Females leave the hibernaculum earlier (April to mid-May) than males (mid-May to early June) to disperse or move to summer colonies.

F.1.2 Feeding Habits and Diet

Little brown bats are nocturnal. Their major activity at night is feeding, which occurs on the wing. Flying insects are located and captured using echolocation (Burt and Grossenheider, 1976). Bats capture smaller insect prey directly with the mouth in flight and use their body, tail, and wings to "cup" and direct larger prey into the mouth. All insect prey is masticated and devoured in flight.

The main feeding periods in NY and other northeast populations are during the hours of 2100 to 0100 and 0330 to 0500 (Anthony and Kunz, 1977; Buchler, 1976). The amount of prey ingested during feeding varies by sex, age, and reproductive state. On average, pregnant bats ingested 2.5 gms of prey, lactating females ingested 3.7 gms, and juveniles ingested 1.8 gms per feeding flight. Digestion of ingested prey begins after the stomach is full and the bat has returned to its colony. Transit time in the gut is rapid, complete digestion and excretion of one stomach volume ranged from 0.5 to 1.5 hours depending upon degree of activity for the Millbrook population (Buchler, 1975).

The diversity of insect species eaten by little brown bats varies with the emergence patterns of aquatic insects during the spring and summer. Prey selection in little brown bats depends on insect size, prey abundance, and breeding state of the individual, although in general insects 3 to 10 mm long are preferred (Anthony and Kunz, 1977; Belwood and Fenton, 1976; Buchler, 1976). As in other northeastern populations, NYS colonies of little brown bats have a diet dominated by the terrestrial stages of aquatic insects, although other insects are also taken (Belwood and Fenton, 1976; Buchler, 1976). The predominant insect orders in their diet are Diptera (midges), Trichoptera (caddisflies), and Ephemeroptera (mayflies). They also feed on Lepidoptera (butterflies and moths) and Hymenoptera (bees, ants, and wasps), and other insects associated with the tree canopy layer. Adult bats from Mid-Atlantic and New England populations (inclusive of NYS populations) had the following dietary composition for the period of June to August based upon scat and bolus analysis:

Chironomidae- 2.8-76.4%;
Other Diptera 0-25.5%;
Trichoptera 2.8-36.6%;
Ephemeroptera 0-81% ;
Lepidoptera 0-15.2%; and
Other non-aquatic insect orders 0.56-10.5%.

Pregnant and lactating females favor larger prey such as Ephemeroptera, Trichoptera, and Lepidoptera (Belwood and Fenton, 1976; Buchler, 1976).

The time and foraging radius of feeding flights depend upon prey selection and abundance within the foraging area (Buchler, 1976; Davis and Hitchcock, 1965). Reproductive adult females and males remained close to the summer colony while foraging and typically had shorter foraging periods

than non-breeding adults (Buchler, 1976). A NY summer colony (n= 400) feeding near a pond about 100 meters from their roost spent approximately 20 minutes per feeding flight (Buchler, 1976).

F.1.3 Reproduction

In NY and Pennsylvania (PA), mating occurs in the winter hibernaculum during September/October (Wimbatt, 1945). After mating the female stores the sperm in her uterus during the fall and winter. Ovulation occurs in late April or early May, at which time the sperm is released and the eggs are fertilized. A single fertilized egg implants itself in the uterus wall and the remaining eggs are reabsorbed. Bats in the colonies near Ithaca, NY gave birth from the end of June through the first half of July.

Female bats typically give birth to a single individual, although twins are occasionally born. Newborn weights ranged between 1.5 and 1.9 gms at birth in NY colonies (Wimbatt, 1945). Young little brown bats are nursed during the summer and mature (i.e., birth to flying adult) in 21 to 28 days (Doutt et al., 1977; O'Farrell and Studier, 1973).

F.2 Raccoon (*Procyon lotor*)

The raccoon is a medium-sized opportunistic omnivore found throughout North America. It is indigenous to NYS where it is considered a furbearer species and its take is regulated by NYSDEC (Batcheller, January 1999). Raccoons are commonly associated with aquatic habitats and frequent the shoreline and shallows of rivers, lakes, streams and wetlands, although they are also found in upland habitats (Doutt et al., 1977).

Raccoons are characterized by dark grayish guard hairs over a dark brown to black under coat. A mask of black fur across their eyes and black and gray banded tail make them an easily recognizable species (Burt and Grossenheider, 1976). Meristic data for raccoons is provided in Table F-3. The NYSM has 23 raccoon specimens from the Hudson River Valley (Saratoga, Albany, Rensselaer, Greene, Columbia, Ulster and Dutchess Counties); however, body length and weight recorded at the time of collection are only available for five of the specimens.

Table F-2
Raccoon - Meristic Measurements

Measurement	N. American Range	NYSM Female (n=2)	NYSM Male (n=3)
Body length (cm)	46 - 71	69.8 - 71.0	70.8 - 83.2 (
Tail length (cm)	20 - 30	ND	ND
Adult body mass (kg)	7.0 - 8.3 males 5.6 - 7.1 females	2.77 - 5.7	2.44 - 4.75 (
Source**:** Sanderson, 1984; Burt and Grossenheider, 1976; Bopp 1999. ND = No Data.			

F.2.1 Habitat and Home Range

Raccoons are found in a diverse range of habitats, but prefer areas with permanent water sources, such as stream banks, lake shores, and marshes. They are highly adaptable and can inhabit a wide range of habitats including riparian areas, forested uplands, freshwater and saltwater marshes, mangroves, and hardwood swamps (Kaufmann, 1982). Where natural habitat has been replaced by human encroachment, the raccoon will readily forage on garbage (Kaufmann, 1982; Doutt et al., 1977; Urban, 1970). The raccoon's tolerance of disturbance/development makes it one of the most successful mammals in the contiguous US (Doutt et al., 1977). Raccoons are common in the Hudson River Valley.

Regardless of habitat, raccoons typically den in a protected cavity. In upland and riverine areas this often takes the form of a hollow tree, an abandoned den of another animal, or rock-rubble. In more developed areas, potential denning sites include storm drains, chimneys, abandoned buildings, and beneath patios.

Depending upon the nature of habitat, home range varies in size and is typically expressed as a unit area of habitat. Home range varies by gender, as male raccoons have larger territories than females. Urban (1970) documented an average home range (for both sexes) of 48 hectares for a Lake Erie marsh population, with most activity (73%) occurring in shallow water areas. In riparian habitat in Michigan home ranges varied from 18.2 to 814 hectares for adult males and 5.3 to 376 hectares for adult females (Stuewer, 1943). Ranges decrease during the winter ranges and with young (Stuewer, 1943) and vary by age class and breeding condition (Urban, 1970).

Raccoons do not migrate on a seasonal basis, but occupy and defend a resident territory throughout the year. This excludes local movements related to territory or dispersal of sub-adults from defended territories (Smith, 1980; Doutt, et al., 1977; Urban, 1970). Populations within the study area of the Hudson River Valley are considered year-round, non-migratory residents.

F.2.2 Feeding Habits and Diet

Raccoons are nocturnal in habit and highly opportunistic in diet. Although they are nocturnal, raccoons will modify their foraging behavior to coincide with tidal fluctuations to take advantage of prey items on tidal mudflats during the day (Ivey, 1948). During non-breeding periods, raccoons are solitary and actively feed on both plants and animals. They are active year round, although during periods of cold weather they may hibernate. In northern raccoon populations hibernation may last from late November to April (Whitney and Underwood, 1952). While hibernating, individuals can lose up to half their body weight (Mech et al., 1968).

Raccoons exploit seasonally abundant food items including aquatic invertebrates, fish, berries, fruit, or refuse. Although smaller prey items are preferred, raccoons can catch and feed upon larger prey, such as waterfowl and small mammals, and are significant waterfowl egg predators (Doutt et al., 1977).

Prey items identified in feeding studies demonstrate the diverse nature of the raccoon's diet. Fruits, berries, and vegetation make up the greatest proportion of the diet with fish, aquatic invertebrates (e.g., crayfish), amphibians, reptiles, small birds, and mammals comprising most of the rest of the diet. Riverine populations consume more aquatic prey than wetland, upland, or agricultural populations, which tend to eat more plant matter (Hamilton, 1940). Analysis of raccoon scat from Montezuma Marsh, NY showed fruits (e.g., wild cherries, apples, dogwood berries) to dominant the diet during the summer months. In contrast, summer diets of raccoons from the Schoharie River near Middleburg, NY were dominated by crayfish with only a minor component of fruit and corn (Hamilton, 1951). Crayfish accounted for 59% of the diet of riverine raccoons in Michigan (Dearborn, 1932) and 67% of the diet of riverine raccoons in Minnesota (Schoonover and Marshall, 1951). Review of several diet studies on a seasonal basis indicate that animal prey typically comprises a higher proportion of spring and winter diets (Tabatabai and Kennedy, 1988; Llewellyn and Uhler, 1952).

On an annual basis, the dietary composition of several North American populations consists of : 0-3% fish, 1.4-37.0% aquatic invertebrates, 0-40.0% reptiles/amphibians, 0-15.8% terrestrial invertebrates, 0-8.0% small mammals/birds, 0-93.0% fruits, berries, vegetation, and 0-1.5% carrion (Tabatabai and Kennedy, 1988; Llewellyn and Uhler, 1952; and Hamilton, 1951). Carrion appears to be a minor dietary component and its consumption depends upon finding it by chance. Raccoons were observed scavaging dead fish (Clupeidae) along the shoreline of an upper NYS reservoir following a sudden change in water temperature that resulted in thermal shock to the fish (Nye and Suring, 1978) and scavenging dead and stranded fish following periods of high water in rivers (Doutt et al., 1977). Carrion has been estimated to comprise 1.5 % of the raccoon diet in NY (Hamilton, 1951).

Beyer et al. (1994) included raccoons in an investigation of incidental ingestion of abiotic material (as insoluble material) by wildlife. Beyer et al. (1994) estimated a percent soil in diet of 9.4% for raccoons. This high fraction of abiotic material is attributable to the diverse diet and feeding behaviors of the raccoon.

F.2.3 Reproduction

Raccoons use a variety of refugia for dens, based upon availability in the surrounding habitat. Natural dens include hollow trees or stumps, abandoned animal dens, muskrat and beaver lodges, and caves/crevices (Doutt et al., 1977; Urban, 1970). In developed areas, dens may include chimneys, crawl spaces beneath homes, storm sewers, and debris in abandoned lots (Doutt et al., 1977).

In northern populations, mating occurs from January to March (Doutt et al., 1977; Johnson, 1970). In NYS, mating usually occurs during the first half of February (Hamilton, 1943). Raccoons have a gestation period of about 63 days and give birth from April to May in the northeast US (Doutt et al., 1977). There are 3 to 6 kits in an average litter (Doutt et al., 1977; Burt and Grossenhieder, 1976). Sexual maturity is typically reached by 10 months to a year (Doutt et al. 1977; Burt and Grossenhieder 1976, Stuewer 1943).

Raccoon populations suffer significant mortality from disease (incl. parasites), starvation, and predation. Parasitism and starvation were the two greatest causes of juvenile mortality in a Minnesota population (Mech et al., 1968). Monthly mortality for this population was 0-16% for adults and 0-39% for juveniles, with the highest mortality in both age groups occurring in winter (January through March).

F.3 Mink (*Mustela vison)*

The mink is a small opportunistic carnivore found throughout North America. It is indigenous to NYS, where as a furbearer species its take is regulated by NYSDEC (Batcheller, 1999). Mink have dark brown fur rich in long coarse hairs, known as guard hairs, that cover and protect their soft underfur. Male minks are larger than females (Table F-2). Meristic measurements for mink are provided in Table F-2.

Table F-3

Mink - Meristic Measurements

Measurement	N. American Range wild populations	N. American captive populations	NYSM (n=1)
Body length (cm)	33 - 43 males 30 -36 females	ND	59.0 male
Tail length (cm)	18 - 23 males 13 - 20 females	ND	ND
Body mass (gm)	681 - 1,233 males 567 - 586 females	1,734 male 974 female	1,100 male

Source: Bopp, 1999; Hornshaw et al., 1983; Burt and Grossenheider, 1976; Mitchell, 1961.
ND = No Data.

F.3.1 Habitat and Home Range

Mink are semi-aquatic in habit and are found around the shoreline and shallows of rivers, lakes, streams, and wetlands. They prefer wetlands and riparian habitat with irregular shorelines, good cover (i.e., woods and shrub), and suitable den sites. Bulkheaded and channelized shorelines with sparse vegetation are poor habitat for minks (Allen, 1986). Mink are reasonably tolerant of human disturbance/development as long as prey abundance is not affected (Allen, 1986). Regardless of the type of habitat used, mink dens are always associated with water and typically are 5 to 100 meters from a water body. Mink are expected to potentially inhabit all counties of NYS and can be found in the Hudson River Valley (Batcheller, 1999). The decline of mink populations along portions of the Hudson River (Foley et al., 1988) and the Niagara River (Newell et al., 1987) may be related to development and/or organochlorine contamination.

Depending upon the nature of habitat, (i.e., wetland vs. riverine), mink home range has been expressed either as per unit area of wetland or per length of river shoreline. Riverine home ranges in Sweden varied from 1.8 to 5.0 river km for adult males and 1.0 to 2.8 river km for adult females (Gerell, 1970), while adult female mink from a Montana riverine population had home ranges of 7.8 hectares in heavy vegetation and 20.4 hectares in sparse vegetation (Mitchell, 1961). Male mink generally defend larger territories than female minks (Eagle and Whitman, 1987).

Mink are active year round and do not hibernate (Doutt, et al. 1977; Alexander, 1977). They occupy and defend a resident territory throughout the year, but do not migrate with the exception of local territorial movements by adults and dispersal of sub-adults from resident populations (Allen, 1986). Populations within the study area of the Hudson River Valley are year-round residents.

F.3.2 Feeding Habits and Diet

Mink are nocturnal in habit and entirely carnivorous in diet. Like other members of the weasel family, they are solitary (with exception of mating and courtship) aggressive predators and actively seek prey within their home range.

Generally, mink are opportunistic in their feeding habits and prey varies according to seasonal abundance of prey and habitat. Mink feed primarily on small aquatic and terrestrial animals, although they can feed upon prey items larger than themselves, such as waterfowl and muskrats (Sealander, 1943). Common prey items include fish, frogs, crayfish, salamanders, clams, insects, muskrats, voles, and rabbits (DeGraaf and Rudis, 1987; Sealander, 1943). Hunting in aquatic habitats occurs in shallow, nearshore areas where aquatic prey is captured and then moved to the shore prior to consumption (Allen, 1986; Doutt et al., 1977).

Riverine populations of mink have a greater proportion of aquatic prey in their diet than wetland populations. Quantitative dietary composition of prey items from Hudson River Valley populations was not available; however, dietary characteristics observed from other northern riverine populations would be expected to be similar (Batcheller, January 1999). In Michigan riverine populations mink diets consisted of 85% fish, 4% crayfish, 3% amphibians, 6 % birds/mammals, and 2% other matter/vegetation (Alexander, 1977).

Hamilton (1936) conducted a stomach content analysis study of 70 mink trapped throughout NYS. He found their winter diet to consist of: 54.1% mammals, 18.8 % fish, 16.5 % crayfish, 2.4% amphibians, and 7.0 % insects. A subsequent analysis of the summer mink in Montezuma Marsh, a wetland in the Finger Lakes region of NY, found the diet to consist of: 42.7% mammals, 27.3% fish, 13.9% aquatic invertebrates, 9.1% birds, and 4.5% reptiles/amphibians (Hamilton, 1940). The most abundant forage fish in Montezuma Marsh, the golden shiner (*Notemigonus crysoleucas*), comprised the greatest proportion of fish in the mink diet. Fish consumed were generally 7.6-10.5 cm, which is within the average adult size range of the golden shiner. The aquatic invertebrate fraction of the diet consisted almost entirely of adult predacious diving beetles (Family Dytiscidae).

The dominant fish prey of a Montana riverine population was the brook stickleback *(Culaea incostans)* (Gilbert and Nancekivell, 1982). The brook stickleback was abundant in the drainages studied and ranged from 3.8 to 6.4 cm in length. As the mink is an opportunistic feeder, prey selection probably depends on primarily on the abundance of fish or other prey species and secondarily on the size. For example, in wetlands managed for waterfowl populations waterfowl and muskrats appear to be the most important prey items for mink (Arnold and Fritzell, 1987).

Mink incidentally ingest a small quantity of vegetation/soil appears while feeding (Waller, 1962; Sealander, 1943). Hamilton (1940) recorded trace quantities of sand in a limited number of mink scat samples. Grasses were found at a relative frequency of 1.18% in mink stomachs (Hamilton, 1936). Soil and vegetation are probably incidentally ingested during feeding (Alexander, 1977). Based upon the observations of soil and vegetation in mink stomachs, a percent ingestion of sediments during feeding was assumed to be approximately one percent of the diet.

F.3.3 Reproduction

Mink build their dens below the ground in hollow logs, muskrat lodges or other abandoned animal dens, or under fallen trees or stumps (Allen, 1986; Doutt et al., 1977). They mate in early spring and have a gestation period of about 50 days. Implantation of the embryos can be delayed to allow birth to occur when the weather is warmer, generally from April to June, as the newborn kits are naked and blind (Eagle and Whittman, 1987). An average litter contains of 3 or 4 kits (Burt and Grossenhieder, 1976). Sexual maturity is typically reached by one year, although mating may occur as early as 10 months in captive populations (Burt and Grossenhieder, 1976; Enders, 1952).

F.4 River Otter (*Lutra canadensis)*

The river otter is a medium sized piscivorous carnivore found throughout most of North America. It is indigenous to NYS, and occurs within the Hudson River Valley. As a furbearer species, its take is regulated by NYSDEC. Currently the river otter is the focus of a reintroduction program in western NYS where trapping of the species has been suspended pending results of the reintroduction program (Penrod, 1999). Otter have rich, dark brown fur dense with many guard hairs on their dorsal side and dark brownish sheen underfur on their ventrum (Burt and Grossenheider, 1976). Adults are sexually dimorphic in body size and mass with males being slightly larger than females. General meristic measurements of river otters are provided in Table F-4. Observed measurements for river otters from Hudson River counties are consistent with those from other North American populations.

Table F-4
River Otter - Meristic Measurements

Measurement	N. American Range wild populations	NYSDEC River Otter Project	NYSM Female (n=1)	NYSM Male (n=2)
Body length (cm)	66.0 - 76.0 males and females	ND	64.0	61.3-82.4
Tail length (cm)	30.0 - 43.0 males and females	ND	36.4	39.1-64.9
Body mass (Kg)	5.0 - 10.0 males 4.0 - 7.0 females	7.3-8.3 males 5.2-6.3 females[1]	ND	8.23-12.7

Source: Bopp, 1999; Spinola et al., 1999; Toweill and Tabor, 1982; Burt and Grossenheider, 1976.
ND =No data.
[1] Females were all classified as juveniles.

F.4.1 Habitat and Home Range

River otter are always closely associated with aquatic habitats. They prefer open water areas of lakes, wetlands, rivers and streams with irregular shorelines with a permanent food supply, good cover (e.g., woods and shrub), and suitable den sites. Otters will tolerate minor disturbance by human encroachment, though less so than mink and raccoon, but prefer habitats that are minimally impacted by human disturbance. River otter have a general preference for rivers and streams although they are also found in lakes and reservoirs (Melquist and Horcocker, 1983). For riverine and lacustrine populations, habitats are defined by riparian margins which parallel the shoreline of the water body. In wetland areas which consist of multiple, smaller stream drainages, home range appears polygonal in shape and is expressed as a unit area (Melquist and Horcocker, 1983).

Regardless of the type of habitat used, river otters use various denning and latrine sites that are always associated with water. Use of these areas is dependant upon behavioral activity and reproductive state. During non-reproductive periods, resting dens are temporally utilized between feeding periods and as temporary cover for sleeping. These dens are usually distributed throughout the home range of the family unit. Multiple, common latrine sites are utilized at discrete locations within the familiar home range. Birthing dens include excavated dens abandoned by other animals, log piles/jams, and abandoned beaver lodges (Toweill and Tabor, 1982; Melquist and Dronkert, 1987).

Currently as part of the NYSDEC river otter reintroduction program, a radio tracking study of relocated river otters revealed ranges of 1.5 km to 22.4 km with an average of 10 km for fourteen individuals in western New York State individuals (Spinola et al., 1999). Riverine home ranges in Idaho populations varied from 23 to 50 km for adult females (mean = 31 km); and 10 to 78 km for yearling males and females (mean = 38 km) (Melquist and Horncocker, 1983). On a unit area basis, home range estimates range from 295 to 5,700 hectares (USEPA, 1993). Home range size fluctuates between sexes and reproductive state with females occupying smaller home ranges than males and smaller still with young.

River otter are active year round and do not hibernate (Doutt et al., 1977). They occupy and defend a resident territory throughout the year, and do not migrate with the exception of local territorial movements by adults and dispersal of sub-adults from resident populations. Populations within the study area of the Hudson River Valley are year-round residents.

F.4.2 Feeding Habits and Diet

River otter are entirely piscivorous/carnivorous in diet. They are most active at dawn and dusk, though they remain active throughout the day. Males are solitary, with the exception of mating and courtship. Breeding females and juveniles form small family aggregations following weaning. They are good swimmers and divers and actively pursue fish underwater. Throughout their range, they are largely piscivorous in diet throughout the year with amphibians, birds and small mammals

representing minor components of the diet (<5%) (Toweill and Tabor, 1982). No dietary data on a seasonal basis for river otter in the Hudson River Valley were available, but dietary characteristics for other regionally proximal populations are expected to be representative of Hudson River populations.

Hamilton (1961) examined stomach/intestine contents from 141 trapped individuals from the Adirondacks in NYS and found the diet to consist of: 70% fish, 35% crayfish, 25% amphibians, 13.5% aquatic insects, and 4.3% mammals. This evidence for an aquatic prey diet, primarily fish prey, of NYS populations of river otter is consistent with other populations from surrounding regions. In evaluation of risks to river otter in the Niagara River watershed, Newell, et al. (1987) considered the river otter diet in NYS to consist of mostly aquatic prey and used a 90% fish diet in the Niagara River assessment. Preliminary observations from scat analysis involving radio telemetry of river otters in western NYS indicate that diets are entirely piscivorous in nature during the winter (Spinola, 1999). Diets may vary slightly during other times of the year when other potential prey items, such as amphibians and reptiles, become available.

Otters select fish in direct proportion to abundance and inverse proportion to the fish species swimming ability (Ryder, 1955; Sheldon and Toll, 1964). Otters actively feed in coves and shallow water areas and avoid deeper areas. Analysis of their scat showed that fish species are eaten in proportion to their abundance in shallow water habitats. Principal fish species consumed included percids (perches), centrarchids (sunfish and freshwater bass), catastomids (suckers), cyprinids (minnows), and esocids (pickerel). Based upon predominance of recovered dorsal and pelvic spines in middens, Sheldon and Toll, (1964) postulated that ictalurids (bullheads) are likely a significant component as well, although they may be underestimated in the scat because of their small scales.

Size of fish consumed by river otter appears to be secondary to general abundance by habitat in prey selectivity. Size of fish consumed by river otter in North American populations range 7.6 to 41.0 cm depending upon species captured (Gilbert and Nancekivell, 1982; Greer, 1955). The fraction of aquatic invertebrates in the diet appears secondary to the fish component. Detailed descriptions of representative invertebrate prey include crayfish, mussels and clams, aquatic beetles, dragonfly nymphs, stonefly nymphs and scuds (Amphipoda).

River otter appear to incidentally ingest a small quantity of vegetation/sediments while feeding (Toweill, 1974; Liers, 1951). This ingestion occurs largely through the searching out and recovery of invertebrate prey (e.g., crayfish, mussels, etc.) from benthic substrates. Additionally, the river otter may be incidentally exposed during specific non-feeding behaviors. The river otter is an active groomer (e.g., combing of the fur by the front paws) and cleaner (e.g., licking of the fur) and spends a significant amount of time on these activities (Liers, 1951). During such behavior, the otter may incidentally ingest sediment on its fur. No estimate of incidental sediment/soil ingestion for the river otter was available from the literature, but the percent incidental ingestion of sediments during feeding is considered to be less than observed in the raccoon. Therefore, based upon a largely piscivorous diet and best professional judgement, it was assumed that incidental ingestion of sediments was one percent of the diet.

F.4.3 Reproduction

River otter build birthing dens in excavated animal burrows, in hollow logs, abandoned beaver lodges, other abandoned animal dens, or under fallen trees or stumps (Doutt et al., 1977). A detailed description of reproduction in NYS river otter populations is provided in Hamilton and Eadie (1961). In New York populations, mating occurs from March to April. Implantation of the developing blastocysts can be delayed up to eight months and implantation into the uterine wall usually occurs during January and February. Following implantation, fetal development is rapid with NYS river otters giving birth during the period of March to April. The newborn kits are fully furred and their eyes are closed. An average litter contains two to three kits, though litters of up to four are possible. Sexual maturity is typically reached in two years for both sexes. Reproduction in females does not occur before two years of age. Females nurse the young and males leave the den soon after the birth. Juveniles remain with the female and the formation of family groups is common (Greer, 1955).

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PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

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THREATENED, ENDANGERED, AND SPECIAL CONCERN SPECIES

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APPENDIX G
THREATENED, ENDANGERED, AND SPECIAL CONCERN SPECIES

The federal Endangered Species Act (16 USC Subchapter 1531, et seq.) divides rare animals and plants into two categories, "endangered" and "threatened". New York State (NYS) maintains a list of animals and plants that are considered endangered or threatened at the state level, plus a third category for animal species of "special concern" (6 NYCRR Part 182). These federal and state categories reflect the level of concern regarding extinction of the species. Endangered species are faced with imminent extinction. Threatened species are in less danger, but require special protection to maintain their populations. Special concern species have no legal protection but are listed because a welfare concern or risk of endangerment has been documented in New York State.

The NYSDEC Natural Heritage Program has inventoried single or multiple occurrences of 35 rare and endangered plants, 19 rare and endangered animals, and multiple occurrences of 9 rare community types in the Hudson River Estuary (NYSDEC, 1998). The information in this section is based on the searches performed by the US Fish and Wildlife Service (USFWS, 1999) and NYS Natural Heritage Program (NYSDEC Natural Heritage Program, 1999), and NYSDOS report on the Hudson River Tidal Habitats (NYSDOS, 1990). Profiles of species presented in this appendix are taken from NYSDEC (1999) and NYSDOS (1990).

G.1 Plants

The Hudson's tidal habitats support a number of rare plant species. Many of these species are recognized by the state as "protected native plants" that are listed as either endangered, threatened, exploitatively vulnerable or rare (Article 9, Section 15 of the Environmental Conservation Law). A list of rare plant species found along the Hudson River is provided in Table G-1.

G.2 Invertebrates

No legally-recognized endangered invertebrate species are currently found in the Hudson River. The Karner blue butterfly (*Lycaeides melissa samuelis*), a federal and NYS-listed endangered species, is found in areas adjacent to the Hudson River.

G.2.1 Karner Blue Butterfly *(Lycaeides melissa samuelis)*

The Karner blue is a small butterfly with a wing span of approximately one inch. In the male, the upper surface of all four wings is a deep violet-blue fringed with white. In the female, the upper surface is a dusky brownish blue with orange spots on the edge of the hindwing. The lower surface is a pale silver with white- ringed black spots and rows of bright orange and blue markings near the edge of the hindwings. The protective coloration of the larva, which reaches half an inch in length before changing into a pupa, perfectly matches the green leaves of the vegetation. The larva is covered with very fine hairs.

Like all butterflies, the Karner blue has four stages in its life cycle - the egg, the larva (caterpillar), the pupa (chrysalis), and the adult (butterfly). There are two generations per year. The first generation adults appear in late May to mid-June. Females lay eggs on the underside of a leaf or stem of the food plant, blue lupine (*Lupinus perrennis*). These eggs hatch in seven to eight days. Forty to fifty percent of the eggs survive to the adult stage. The resulting second brood adults, emerging in mid-July to early August, lay their eggs singly in dried lupine seed pods or near the ground on the stems. Eggs of the second brood overwinter, to hatch the next May. Karner blue adults are nectar-feeders, aiding in the pollination of a variety of wildflowers. The larvae, however, are highly specialized, feeding exclusively on the wild blue lupine leaves. Without blue lupine, the Karner blue would not survive.

The Karner blue is found in scattered localities from Minnesota to New Hampshire. In New York, the butterfly is found in certain parts of the Hudson Valley sand belt which extends from the Albany Pine Bush north to the Glens Falls area. Within its range, this species is restricted to dry sandy areas with open woods and clearings supporting wild blue lupine. This type of habitat is usually associated with pitch pine/scrub oak or oak savannah communities that are maintained by fire at an early stage of plant succession.

The sandy habitat essential to the blue lupine, and therefore the Karner blue, occurs mostly along river valleys and outwash plains. Because of the location and topography of such areas, they have been heavily favored as settlement sites. Extinctions of entire populations of the Karner blue have occurred around large urban centers such as Chicago and New York City. Other populations, such as those in the Albany Pine Bush, have been reduced both by habitat destruction from urbanization and by loss of lupine through natural succession resulting from fire suppression. The most intact populations remain in Saratoga County.

G.3 Fish

The federal and State-listed endangered shortnose sturgeon *(Acipenser brevirostrum)* (profiled in Appendix D) is the only listed fish species found in the Hudson River.

G.4 Reptiles and Amphibians

NYS-listed endangered herpetofauna potentially found along the Hudson River include the northern cricket frog (*Acris crepitans*) and bog turtle (*Clemmys muhlenbergii*). The northern population of the bog turtle (*Clemmys muhlenbergii*) is also a federally-listed threatened species (US Federal Register: November 4, 1997). NYS-listed threatened herpetofauna include Blanding's turtle (*Emydoidea blandingii*) and the timber rattlesnake (*Crotalus horridus*).

Amphibians of special concern listed by NYS potentially found near the Hudson River include the Jefferson salamander (*Ambystoma jeffersonianum*), blue-spotted salamander (*Ambystoma laterale*, and spotted salamander (*Ambystoma maculatum).* Reptiles of special concern include spotted turtle (*Clemmys guttata*), wood turtle (*Clemmys insculpta*), diamondback terrapin *(Malaclemys terrapin*), and fence lizard *(Sceloporous undulatus).*

G.4.1 Northern Cricket Frog *(Acris c. crepitans)*

The northern cricket frog is one of New York State's smallest vertebrates. This frog is an aquatic species, and although it belongs to the tree-frog family, Hylidae, which includes such well-known climbers as the spring peeper (*Pseudacris crucifer*) and gray treefrog (*Hyla versicolor*), it does not climb very much. It is, however, among the most agile of leapers and can jump surprisingly long distances (5-6 feet) for its small size.

Adults average only 1 inch (2.5 cm) in length; the male is usually smaller than the female. Cricket frogs exhibit a myriad of patterns and combinations of black, yellow, orange or red on a base of brown or green. Distinguishing characteristics are small size, dorsal warts, a blunt snout, a dark triangular-shaped spot between the eyes, and a ragged, longitudinal stripe on the thigh. The webbing on the hind foot is extensive, reaching the tip of the first toe and the next to last joint of the longest toe. This frog was named for its breeding call, which sounds very much like the chirp or trill of a cricket, "gick, gick, gick...," repeated for 20 or more beats. The sound has been likened to two pebbles being clicked together, slowly at first, then picking up in speed.

This frog, which may be reproductively active for 3-10 years, is one of the last frogs in the northern part of its range to come into full chorus in New York. Breeding occurs from June to July. A single female may lay several dozen filmy egg masses on aquatic vegetation, each containing 5-10 eggs. In about 4 days, tadpoles with black-tipped tails emerge. They develop relatively slowly, feeding mostly on algae and zooplankton, until transforming into subadults by mid-September, often at a length as small as frog spends the coldest winter months burrowed in muck or peat below the frost line, although there is evidence in New York that some individuals may overwinter in upland sites.

In the eastern US, cricket frog populations reach their northern limit in the Hudson Highlands - Shawangunk region of New York. As late as the 1920's, it also occurred commonly on Long Island and Staten Island. Recently, a population of these frogs was discovered on the east side of the Hudson River in Dutchess County. Within its range, the cricket frog inhabits sunny, shallow ponds with abundant vegetation in the water or on the shores. Slow moving, algae-filled water courses with sunny banks are the preferred habitat. Deep water is generally avoided. Males are typically found calling from floating mats of vegetation and organic debris.

By the 1940's, most historically known populations in New York State had been extirpated. This diminutive frog is now only locally present in a few scattered populations which still occur in the Hudson Highlands and Shawangunk area. The decline of the cricket frog apparently began in the 1800's with the clearing, drainage and alteration of thousands of acres of wetland habitat. Aerial spraying of DDT and other chlorinated hydrocarbon pesticides in the 1950's and 1960's is thought to have contributed to the decline of most remaining populations. Other factors that may have contributed to the cricket frog's decline are contamination of ponds by road salt and the introduction of predatory fish, which feed on their eggs.

G.4.2 Bog Turtle *(Clemmys muhlenbergii)*

The bog turtle is New York's smallest turtle, reaching a maximum length of 4.5 inches. It is one of seventeen species of turtles found in New York State, including marine turtles. A bright yellow or orange blotch on each side of its head and neck are a distinctive feature of this species. The body color is dark with an orange-red wash on the inside of the legs of some individuals. The carapace (upper shell) is domed and somewhat rectangular, often with prominent rings on the shell plates (scutes). In some older individuals, or those that burrow frequently in coarse substrates, the shell may become quite smooth and polished. Although generally black, the carapace is sometimes highlighted by a chestnut sunburst pattern in each scute. The plastron (lower shell) is hingeless, with a pattern of cream and black blotches. As with most turtles, the plastron of the male is slightly concave while the female's is flat.

In New York, the bog turtle emerges from hibernation, often spent in an abandoned muskrat lodge or other burrow, by mid-April. In New York bog turtles often hibernate communally with other bog turtles and with spotted turtles (*Clemmys guttata*). Generally both the air and water temperature must exceed 50 degrees F for the turtle to become active.

Mating occurs primarily in the spring but may also occur in the fall and may be focused in or near the hibernaculum (winter shelter). In early to mid-June, a clutch of two to four eggs is laid in a nest which is generally located inside the upper part of an unshaded tussock. The eggs hatch around mid-September. Some young turtles spend the winter in the nest, emerging the following spring. The adults enter hibernation in late October. Sexual maturity may be reached at eight years or as late as eleven. A bog turtle may live for more than 30 years.

Although generally very secretive, the bog turtle can be seen basking in the open, especially in the early spring just after emerging from hibernation. It is an opportunistic feeder, eating what it can get, although it prefers invertebrates such as slugs, worms, and insects. Seeds, plant leaves, and carrion are also included in its diet.

The bog turtle is found in the eastern United States scattered in disjunct colonies from New York and Massachusetts south to southern Tennessee and Georgia. This is a semi-aquatic species, preferring habitat with cool, shallow, slow-moving water, deep soft muck soils, and tussock-forming herbaceous vegetation. In New York, the bog turtle is generally found in open, early successional types of habitats such as wet meadows or open calcareous boggy areas generally dominated by sedges (*Carex* spp.) or sphagnum moss. Like other cold-blooded or ectothermic species, it requires habitats with a good deal of solar penetration for basking and nesting. Plants such as purple loosestrife (*Lythrum salicaria*) and reed (*Phragmites australis*) can quickly invade such areas resulting in the loss of basking and nesting habitat.

More than half of the 74 historic bog turtle locations in New York still contain apparently suitable habitat. Only one quarter of these sites, however, are known to support extant populations, primarily in southeastern New York. The bog turtle is potentially found in wetland habitats along the Hudson River.

G.4.3 Blanding's Turtle *(Emydoidea landingii)*

The Blanding's turtle is a medium sized turtle with an average shell length of approximately seven to nine inches and a maximum length of 10 inches. A distinguishing feature of this turtle is the bright yellow chin and throat. The carapace, or upper shell, is domed, but slightly flattened along the midline, and is oblong when viewed from above. The carapace is speckled with numerous yellow or light-colored flecks or streaks on a dark background. The plastron is yellow with dark blotches symmetrically arranged. The head and legs are dark, and usually speckled or mottled with yellow. The Blanding's turtle is also called the "semi-box" turtle, for although the plastron is hinged, the plastral lobes do not shut as tight as the box turtle's.

Mating probably occurs in April and early May with nesting beginning in early June and lasting throughout the month. The clutch size varies from region to region. In New York, the clutch size ranges from 5-12 eggs with an average of eight. The Blanding's is a timid turtle and may plunge into water and remain on the bottom for hours when alarmed. If away from water, the turtle will close itself up within its shell. It is very agile and a good swimmer. The Blanding's turtle overwinters under or near water, in mud or under vegetation or debris. During the nesting season, a female Blanding's turtle may be found more than a kilometer from where it hibernated. It is omnivorous, eating crustaceans and other invertebrates, fish, plants, carrion and vegetable debris. It is capable of catching live fish. Blanding's turtles take 18-22 years to reach sexual maturity and may live to be 70 years old.

G.4.4 Timber Rattlesnake *(Crotalus horridus)*

Measuring from 3-4.5 feet (91-137 cm) or more in length, the timber rattlesnake is the largest venomous snake in New York. Despite their size, cryptic coloration allows them to be easily concealed. Two color patterns are commonly found: a yellow phase, which has black or dark brown crossbands on a lighter background color of yellow, brown or gray, and a black phase, which has dark crossbands on a dark background. Black or dark brown stippling also occurs to varying degrees, to the extent that some individuals appear all black. Scales are ridged, giving this rattlesnake a rough-skinned appearance.

Like other members of the pit-viper family, the timber rattlesnake has a temperature- sensitive opening, or pit, on either side of the face between and a little below the eye and nostril. This sensory organ is used to detect prey and potential predators. This rattler feeds primarily on small mammals, but occasionally takes small birds, amphibians and other snakes. Another feature distinctive of rattlesnakes is the rattle itself, which is made of loosely attached horny segments. A new segment is added each time the snake sheds. When vibrated, the rattle makes a buzzing sound characteristic of a disturbed rattlesnake.

Timber rattlesnakes are active from late April until mid-October, although in northern New York they may not emerge until mid-May. Upon emerging from the den, they are very lethargic. Little feeding occurs early in the spring. Mating occurs in the spring and fall. Males are especially active at this time, seeking out females by following the pheromone (sex attractant odor) they emit. The

gestation period is 4-5 months. Females give birth to 4-14 (average 9) young every three to five years during late August to mid-September. The young are approximately 1 foot (30 cm) in length at birth and emerge individually from the female, encased in a transparent membrane which is shed in a few minutes. Each is equipped with venom, hollow fangs and a tiny rattle segment called a "button." Their skin has a velvety texture and the coloring is essentially the same as the adult's. They remain in the area for 1-2 weeks before shedding their skin and dispersing. The young follow the adult's scent trail back to the den. Males are sexually mature in 5 years, females in 7-11 years. Their average life span is 16-22 years, with a maximum age of about 30 years.

During winter, dozens of timber rattlers may congregate together in a den to hibernate below the frost line in association with copperheads (*Akgistrodon contortrix*), other snakes, and skinks (*Eumeces* spp.). Dens are generally on open, steep, south facing slopes with rock fissures or talus surrounded by hardwood forests.

The range of the timber rattler extends from southern New Hampshire south through the Appalachian Mountains to northern Georgia and west to southwestern Wisconsin and northeastern Texas. Populations were once found on Long Island and in most mountainous and hilly areas of New York State, except in the higher elevations of the Adirondacks, Catskills and Tug Hill region. They are now found in isolated populations in southeastern New York, the Southern Tier and in the peripheral eastern Adirondacks.

Timber rattlesnakes are generally found in deciduous forests in rugged terrain. In the summer, gravid (pregnant) females seem to prefer open, rocky ledges where temperatures are higher, while the males and non-gravid females seem to prefer cooler, thicker woods where the forest canopy is more closed. Rattlers generally migrate from 1.3 to 2.5 miles (2 to 4 km) from their den each summer, with a maximum movement of 4.5 miles (7.2 km) observed.

G.5 Birds

The Hudson River Valley is home to many bird species, including a number of endangered and threatened species and species of special concern. The peregrine falcon (*Falco peregrinus*) is listed as endangered by both federal and state governments. The bald eagle (*Haliaeetus leucocephalus*) is a NYS-listed endangered species and a federal-listed threatened species.

The osprey (*Pandion haliaetus*), northern harrier (*Circus cyaneus*), and red-shouldered hawk (*Buteo lineatus)* are NYS-listed threatened species found in the Hudson River Valley.

Avian NYS species of special concern found in the vicinity of the Hudson River are the least bittern (*Ixobrychus exilis*), Cooper's hawk (*Accipiter cooperii*), upland sandpiper (*Bartramia longicauda*), short-eared owl (*Asio flammeus*) barn owl *(Tyto alba),* king rail *(Rallus elegans),* common nighthawk (*Chordeiles minor*), eastern bluebird, (*Sialia sialis*), grasshopper sparrow (*Ammodramus savannarum*), and vesper sparrow (*Pooecetes gramineus).*

Brief profiles of the species listed as endangered or threatened are provided here, with the exception of the bald eagle. The bald eagle was selected as a receptor in the Hudson River ERA and a detailed profile of it can be found in Appendix E.

G.5.1 Peregrine Falcon *(Falco peregrinus)*

The peregrine falcon (*Falco peregrinus*) is a federal and NYS-listed endangered species. This crow-sized falcon is admired for its incredible speeds which are seldom exceeded by any other bird. Plunging from tremendous heights, the peregrine falcon can reach up to 180 mph in pursuit of prey. It feeds primarily on birds, which it takes on the wing. Adult peregrines are slate-grey above and pale below, with fine dark bars and spots on their underparts. Both adults and immature birds have a wide, dark "moustache" mark below the eye. The tail is narrow and the wings long and pointed. Juveniles are brown overall, with dark streaking below. Airborne, this falcon can be recognized by characteristic rapid wingbeats interspersed with long glides.

The peregrine falcon feeds mainly on birds. It hunts close to waterways, spotting its prey from the tops of cliffs. Prey species include waterfowls, song birds, sea birds and shore birds.

Peregrine falcons generally return to the same nesting territory annually and mate for life. The courtship flight is a spectacular sight. The pair climbs high in the air and performs a precise acrobatic act of whirling spirals and steep rapid dives, often touching in midair. The average clutch consists of three to four eggs which hatch after an incubation period of 29-32 days. The single brood fledges after 35-42 days. Both parents participate in incubation and brooding activities, but the female remains at the nest for the majority of the time while the male hunts and brings food to her and the young. Young falcons may stay in the area for about six weeks after the fledge, developing their flying and hunting skills. Sexual maturity is generally reached at two years of age, but one-year-olds have been known to produce young. Individuals may live as long as 20 years.

The peregrine falcon prefers open country from tundra, savannah and sea coasts, to high mountains, as well as open forests and tall buildings. Nests are built on high ledges, 50 to 200 feet off the ground. The nest itself is a well rounded scrape and is occasionally lined with grass.

Like many other birds of prey, peregrine falcons have suffered from the use of pesticides. Exposure to DDT and other chemical contaminants has caused population declines since the 1940's. These pesticides cause eggshell thinning which drastically lowers breeding success. At one time, there were approximately 350 breeding pairs in the eastern US, including 40-50 historic eyries (nest sites) in New York. By 1965, all were gone and populations in other parts of the country showed similar declines. Release programs initiated by the Peregrine Fund in the mid 1970's have resulted in peregrine falcons breeding in New York once again. In 1998, 38 pairs were present in New York, 36 bred, 31 were successful and 69 young fledged. In the first half of 1999, 11 pairs of peregrine falcons were sighted in the Hudson River Valley (Loucks, 1999). These birds were spotted nesting on bridges from Albany south to the Verrazano Narrows. Ten of the eleven pairs bred and eight of the pairs have produced a total of 18 fledglings to date.

The USFWS published a proposal to delist the peregrine falcon (Federal Register, August 26, 1998); however no action has been taken to date.

G.5.2 Osprey (*Pandion haliaetus*)

The osprey is a NY state-recognized threatened species found along large bodies of open water. Osprey breed during the summer in the northern US and Canada, and overwinter in the southern US, the Caribbean, and Latin America. In New York, osprey migrate seasonally to most parts of the State and nest in the northern Adirondacks and on Long Island.

Mature osprey feed exclusively on live fish. Osprey hunt by hovering above the water and then plunging, talons first, into the water to catch its prey. Osprey will take most fish species, but tend to concentrate on those that form large schools. Osprey breeding may be timed to take advantage of concentrations of anadromous fish during spawning runs (Greene et al., 1983).

The female lays one to four, but usually three, eggs in the spring in a large nest of sticks constructed at the top of a dead tree. Breeding osprey pairs return year after year to the same nest, which consists of a bulky stick structure situate high up in a tree or on poles or other artificial platforms (DeGraaf and Rudis, 1986). They also occasionally nest on the ground. The nest is often used year after year and can become quite large (up to 10 feet high) as more material is added prior to each nesting season. The young fledge at about eight weeks of age, then remain in the area of the nest for about two months.

The primary habitat requirement for osprey is a plentiful and constant supply of fish. Consequently, osprey are found only near large lakes, rivers, and estuaries. Within these locations, areas of shallow water are preferred where fish swim close to the surface (DeGraaf and Rudis, 1986). Despite lengthy annual migrations, osprey do not disperse readily from their natal breeding sites and are slow to colonize new breeding areas. Breeding ospreys are extremely sensitive to organochlorine pesticide residues that interfere with eggshell formation (e.g., DDT), resulting in shells that are too thin to survive incubation. The presence of successfully breeding osprey indicates a pesticide-free local environment (Henry, 1983).

The osprey population along the Hudson declined over most of the twentieth century, but has been increasing over the past decade. Although pesticides have no doubt had a significant impact, habitat destruction seems to have also played an important role. Most breeding sites along the Hudson including those at Hyde Park, West Point, Croton Point, and Yonkers have all been inactive since the late 1800's, well before the development of synthetic pesticides (Bull, 1985). Breeding osprey persisted, on the other hand, at less disturbed sites such as at Tivoli Bay until well into the 1950's when pesticides presumably became a factor (Andrle and Carroll, 1988). Currently, there are no known osprey breeding sites along the River but numerous sites including Schodack Island, North Tivoli Bay, Esopus Estuary, Moodna Creek, Wappinger Creek, and Fishkill Creek provide important osprey feeding grounds during the spring and fall migration periods.

G.5.3 Northern Harrier (*Circus cyaneus*)

The northern harrier or marsh hawk is a state-recognized threatened species found in freshwater wetlands throughout northern North America in summer and in the southern US and Latin America during winter. It breeds throughout New York but has been undergoing decline in recent years (Andrle and Carroll, 1988).

This 16-24 inch (41-61 cm), slender-bodied hawk has a long tail and wings, long yellow legs, distinct facial disks and a conspicuous white rump patch. In flight, the wings are held in a shallow "V." The adult male is pale gray on the head, back and wings. The gray tail is banded with 6-8 gray-brown bars. There is cinnamon-brown spotting on the legs and flanks, and the wing linings and undertail are white. The eyes of an adult male are yellow. Female plumage is browner overall with dark streaks on the breast. The female is born with brown eyes which turn yellow at about three years of age. Juveniles resemble adult females, but have gray eyes. When startled, this species makes a rapid, nasal chattering "ke-ke-ke-ke-ke."

This raptor is considered one of the most agile and acrobatic in North America. During the breeding season, the male performs an elaborate courtship flight consisting of a series of U-shaped maneuvers. Unlike most other hawks, harriers build their nests on the ground where they are prone to high predation rates Bull (1998). The nest is a flimsy structure built of sticks and grass and can be found in dense vegetation or situated in a slightly elevated position. The clutch averages 5 eggs. Incubation lasts 30-32 days and begins before the last egg is laid, so the young vary in size. The young fledge in 30-41 days, then remain near the nest, dependent on their parents for 3-4 weeks. Clutches are larger and reproductive success is higher during years when vole populations are high.

The northern harrier hunts primarily on the wing and may cover up to 100 miles per day. Its prey is detected using extremely keen hearing. Mature harriers feed primarily on small mammals and birds, reptiles, insects, and carrion (DeGraaf and Rudis, 1986). The harrier hunts almost exclusively over marsh areas and meadows, flying at low altitudes and diving on its prey. Harriers are thought to mate for life; occasionally a male may be paired with two females.

The primary habitat requirement for the harrier is large expanses of open marsh and meadow for both feeding and nesting. Although the harrier will hunt over pastures and agricultural lands, it is more prevalent in natural open areas Bull (1998). Nestlings are best able to hide from potential predators when they are well concealed among herbaceous or low woody vegetation (DeGraaf and Rudis, 1986) which is most commonly found in cattail marshes and other wetland areas Bull (1998). The effects of human disturbance on harrier populations is not discussed in the literature, but it seems likely that the decline of the species in New York is related to an overall loss of marshes.

Although no specific census of the Hudson harrier populations has been conducted, it is likely that the species occurs in most suitable upper marsh areas along the Hudson River. Populations would likely benefit if these key nesting and feeding wetlands were protected from human disturbance.

G.5.4 Red Shouldered Hawk *(Buteo lineatus)*

The red-shouldered hawk is a slim, narrow-winged, long-tailed buteo. It obtains prey by still-hunting from perches and scanning the ground below. The 17-24 inch (43-61 cm) adult is blackish-brown above with extensive black and white checkering, especially apparent on the wings. Rufous streaking and edging is apparent all over the body, but is most evident on the shoulders. The tail is blackish with three or four narrow white bands. The breast, belly and wing linings are rufous with black streaks. Immatures are brownish above with little or no rufous coloring. Their undersides are cream-colored, heavily streaked, and blotched with dark brown. The tail is brownish-gray with narrow, light bands.

From courtship to the start of incubation red-shouldered hawks scream a loud "kee-yar;" during the remainder of the year they are predominantly silent. During the courtship display, one to four birds may soar together. They flap, swoop and descend while calling before diving to the original perch. They may rise in wide spirals 1,500 to 2,000 feet above the nest. The male and female build a nest together. Nesting almost always occurs near water, such as a swamp, river or pond. It is usually placed in the crotch of the main trunk of a tree, 20-60 feet high. The nest is made of sticks and twigs, lined with strips of inner bark, fine twigs, dry leaves, evergreen sprigs, feathers and down. The clutch averages three eggs. Incubation lasts for 33 days and the young fledge in 39-45 days. First breeding usually occurs at two years of age.

In New York, nesting populations have been found in the Appalachian Plateau, Catskill Peaks, the Delaware, Mongaup and Rensselaer hills, the Tug Hill Plateau, and Lake Champlain Valley.

G.6 Mammals

The federal and State-listed endangered Indiana bat *(Myotis sodalis)* is known to occur along the Hudson River or within one mile of it (USFWS, 1999). Adjacent upland forest may provide habitat for the NYS-listed endangered eastern woodrat *(Neotoma magister)*. Although there are no endangered marine whales and dolphins in the Hudson River, the Hudson River Estuary contributes to the marine food web. Federal and NYS endangered species such as the finback whale (*Balaenoptera physalus*), blue whale (*Balaenoptera musculus*), sei whale (*Balaenoptera borealis*), and the humpback whale (*Megaptera novaenagliae)* pass the mouth of the Hudson River during their annual migrations and may feed on organisms originating in the Hudson River Estuary.

G.6.1 Indiana Bat *(Myotis sodalis)*

The Indiana bat is roughly 2 inches (5 cm) in length and weighs approximately 0.2-0.3 ounces (6-9 gm). It is distinguished from its closest look alike, the little brown bat *(Myotis lucifugus)*, by several rather obscure features. For example, the Indiana bat is uniformly dark grey to grayish-brown in color and often has a pinkish colored nose while the little brown bat has brown fur with slightly darker ears and nose, giving the appearance of a faintly contrasting dark mask. Indiana bat's feet are smaller than little browns, with few if any hairs. Indiana bats are generally found in tightly packed clusters, while little browns generally occur in loose clusters.

Towards spring, Indiana bats disperse from their winter homes (i.e., hibernacula), some going hundreds of miles. They feed solely on flying insects and presumably males spend the summer preparing for the breeding season and winter that follows. Females congregate in nursery colonies, only a handful of which have ever been discovered. These were located along the banks of streams or lakes in forested habitat, under the loose bark of dead trees, and contained from 50-100 females. A single young is born to each female, probably late in June, and is capable of flight within a month. In August or early September, Indiana bats swarm at the entrance of selected caves or mines. This is when mating takes place. Sperm is stored in the female's body; eggs are fertilized in the spring.

Like other hibernating species, the Indiana bat accumulates layers of fat which sustain it over the winter period of dormancy. Indiana bats spend the winter months in secluded caves or mines which average 37 to 43 degrees F. Criteria for selecting hibernacula are not clearly understood; many apparently suitable sites are not occupied. Where this species is found, however, it can be extremely abundant, congregating in densities of more than 300/sq. ft. Year after year, bats often return to exactly the same spots within individual caves or mines. Hibernation can begin as early as September and extend nearly to June.

The Indiana bat is found within the central portion of the eastern United States, from Vermont to Wisconsin, Missouri and Arkansas and south and east to northwestern Florida. In New York, knowledge of its distribution is limited to known wintering locations-caves and mines in which they hibernate. There are eight hibernacula currently known in Albany, Essex, Warren, Jefferson, Onondaga and Ulster Counties. It is certain that the summer range of this species extends well beyond these counties since the animals disperse to breeding areas and other habitats to feed and raise their young. In New York, approximately 13,000 Indiana bats are known to exist in 8 of the 120 sites searched to date.

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TABLE G-1

NYS RARE AND LISTED PLANT SPECIES FOUND ALONG THE HUDSON RIVER

Common Name	Scientific Name	NYS Status	State Rank	Precision Value
Plants - known occurrences (i.e., precision value S)				
American waterwort	*Elantine americana*	Endangered	S1	S
Bicknell's sedge	*Carex bicknelli*	Rare	S2/S3	S
Carey's smartweed	*Polygonum careyi*	Unprotected	S2	S
Clustered sedge	*Carex cumulata*	Rare	S2S3	S
Corn-salad	*Valerianella umbilicata*	Unprotected	SH	S
Davis' sedge	*Carex davisii*	Rare	S1	S
Estuary beggar-ticks	*Bidens bidentoides*	Threatened	S3	S
False hop sedge	*Carex lupiformes*	Rare	S3	S
Fissidens (non-vascular)	*Fissidens Fontanus*	Unprotected	S3?	S
Frank sedge	*Carex frankii*	Unprotected	S1	S
Glaucous sedge	*Carex Flaccosperma var. glaucodea*	Rare	S1	S
Golden club	*Orontium aquaticum*	Unprotected	S2	S
Golden seal	*Hydrastis canadensis*	Threatened	S2	S
Gypsy-wort	*Lycopus rubellus*	Unprotected	S1	S
Heartleaf plantain	*Plantago cordata*	Threatened	S3	S
Illinois pinweed	*Lechea racemulosa*	Rare	S3	S
Liliaeopsis	*Lilaeopsis chinensis*	Unprotected	S2	S
Lined sedge	*Carex striatula*	Unprotected	S1	S
Long's bittercress	*Cardamine longii*	Unprotected	S2	S
Marsh straw sedge	*Carex hormathodes*	Rare	S2/S3	S
Midland sedge	*Carex mesocorea*	Unprotected	S1	S
Mock-pennyroyal	*Hedeoma hispidum*	Rare	S2/S3	S
Narrow-leaved sedge	*Carex amphibola var. amphibola*	Unprotected	S1	S

TABLE G-1 (CONTINUED)

NYS RARE AND LISTED PLANT SPECIES FOUND ALONG THE HUDSON RIVER

Common Name	Scientific Name	NYS Status	State Rank	Precision Value
Saltmarsh bulrush	*Scirpus novae-angliae*	Endangered	S1	S
Schweinitz's flatsedge	*Cyperus schweinitizii*	Rare	S3	S
Slender crabgrass	*Digitaria filiformis*	Rare	S2	S
Small-flowered crowfoot	*Ranunculus micranthus*	Unprotected	S2	S
Smooth bur-marigold	*Bidens laevis*	Rare	S2	S
Southern yellow flax	*Linum medium var. texanum*	Threatened	S2	S
Southern dodder	*Cuscata obtusiflora car. glandulosa*	Unprotected	S1	S
Spongy arrowhead	*Sagittaria calycina var. spongiosa*	Rare	S2	S
Starwort	*Callitriche terrestris*	Unprotected	S2S3	S
Swamp lousewort	*Pedicularis lanceolata*	Rare	S2	S
Swamp cottonwood	*Populus heterophylla*	Threatened	S2	S
Taxiphyllum (non-vascular)	*Taxiphyllum taxirameum*	Unprotected	S1	S
Violet wood-sorrel	*Oxalis violacea*	Unprotected	S1S2	S
Violet lespedeza	*Lespedeza violacea*	Rare	S3	S
Water pigmyweed	*Crassula aquatica*	Endangered	S1	S
Weak stellate sedge	*Carex seorsa*	Rare	S2	S

Notes:
State Rank:
S1 = Typically 5 or fewer occurrences, very few remaining individuals, acres or miles of stream in NYS
S2 = Typically 6 to 20 occurrences, very few remaining individuals, acres or miles of stream in NYS
S3 = Typically 21 to 100 occurrences, limited acreage or miles of stream in NYS
S4 = Apparently secure in NYS
S5 = Demostrably secure in NYS
Precision Rank:
A precision value of "S" indicates that a species is known to be found along the Hudson River.
A precision value of "M" indicates that a species may occur along the Hudson River in an appropriate habitat.
Source: NYSDEC, May 1999.

PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

APPENDIX H

BENTHIC MACROINVERTEBRATE COMMUNITY ANALYSIS

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PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

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BENTHIC MACROINVERTEBRATE COMMUNITY ANALYSIS

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BENTHIC MACROINVERTEBRATE COMMUNITY ANALYSIS

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APPENDIX H
BENTHIC INVERTEBRATE COMMUNITY STUDIES

Benthic community responses to environmental perturbations have been used to assess the impact of municipal, industrial, oil, and agricultural wastes, as well as impacts from other land uses on natural water bodies. Because the benthic community is closely associated with sediment and pore water, relying on the association for habitat, food, and exchange of gases, the characteristics of the community are strongly affected by, and in turn reflect, the quality of the sediment that the organisms inhabit. Therefore, the examination and evaluation of benthic community structure may provide useful information when evaluating sediment quality at hazardous waste sites (Maughan, 1993).

The Thompson Island (TI) Pool Benthic Invertebrate and Sediment Study quantitatively investigates macroinvertebrate populations in areas of varying PCB concentrations within the lower reach of the TI Pool (RM 188.5 to 191.5). Benthic community structure, measured by diversity (D_s), evenness (E_s), and dominance (l), is evaluated to determine whether there are differences that may affect the benthic invertebrate community as a food source for local fish and wildlife.

Benthic invertebrate communities were also sampled at five Lower Hudson River stations in order to characterize the dominant benthic invertebrates at a subset of the significant habitats found in the Lower Hudson.

As part of a weight of evidence approach to determine potential PCB effects on macroinvertebrates in the Hudson River, sediment and water column PCB concentrations are compared to guidelines and criteria (i.e., NOAA, 1999; NYSDEC, March 1998; Washington Department of Ecology, 1997; Ingersoll et al., 1996; Long et al., 1995; Persaud et al., 1993; and USEPA, 1980).

H.1 TI Pool Study Design

A stratified random sampling design was employed within the TI Pool for improved sampling efficiency and accuracy of population estimates. Unlike simple random sampling of an entire area, stratified sampling is a directed effort and focuses on specific areas, or strata, of concern. The strata are generally designated as sampling stations and an equal field effort is made at each station to secure random samples. Stratified random sampling is extensively used in benthic surveys to ascertain whether there are any observed differences in community structure among the various strata or areas (Elliot, 1979). The stratified random design employed within the TI Pool examined the benthic community of five benthic stations or strata, each with a different range of PCB concentrations, but with apparently similar physical characteristics.

A field reconnaissance and associated pilot sampling were performed from May 13-16, 1993 to select benthic sampling stations in the TI Pool and a comparable background station. During the reconnaissance a physically comparable background station was not found upstream of Ft. Edward. Since the relevance of benthic communities from a dissimilar habitat could not be ascertained, it was decided to locate all stations within the lower reach of the TI Pool to provide similar benthic habitats and compare various PCB concentrations in the sediment. The reduced spatial scale increased the likelihood of associating differences in benthic populations to differences in PCBs in contrast to populations responding to other abiotic or biotic factors occurring over larger spatial scales.

Benthic communities were examined in the TI Pool because of the high PCB concentrations found there. Although there is small-scale variability in the distribution of PCBs (USEPA, 1997), an attempt was made to select areas with a range of PCB concentrations by PCB-screening techniques (see Appendix B).

The final selection of the five ecological stations was, by necessity, a compromise between habitats and PCB concentrations. The five ecological sampling stations (3, 4, 5, 6, and 7) were selected based on the following five criteria:

Location in shallow areas less than 1.5 meters total depth;

Similarity of physical characteristics (e.g., temperature, turbidity, and conductivity) as determined by TAMS biologists during field reconnaissances conducted May 13-16, 1993 and August 5, 1993;

Depositional areas dominated by fine-grained sediments;

Major taxonomic groups of benthic organisms were well represented in the 0-5 cm layer (results obtained from the ecological field reconnaissance conducted from May 13-16, 1994); and

PCB field screening results were used to obtain a mix of lower and elevated PCB concentrations.

Five replicates were taken at each station. The benthic invertebrate field sampling effort is described in Appendix B.

The overall aim of the TI Pool study was to create a general profile of community characteristics and to determine if ecologically based linkages and relationships to PCBs could be inferred. Macroinvertebrate communities were characterized through analysis of species richness (number of taxa), abundance (number of individuals), species diversity (a combination of richness and equitability), and biomass. These parameters were then analyzed and statistically significant differences in community characteristics between stations were viewed in relation to physical

properties that could account for the observed variations (i.e., statistically significant differences in grain size, total organic carbon, PCBs, and metals).

To determine statistically significant differences between stations parametric (e.g., analysis of variance [ANOVA]) and non-parametric (e.g., Kruskal-Wallis Test and Mann-Whitney U-test) tests were used (see Ludwig and Reynolds, 1988 and Sokal and Rohlf, 1981 for detailed explanations of tests). ANOVAs were used to test whether samples from each station could be considered to belong to the same population and if differences between sample means are within the accepted error of the population mean (Elliot, 1979). The probability (p) for all statistical tests conducted during this study was set at P=0.05, which corresponds to less than a five percent chance of an event occurring randomly. ANOVAs were used to determine statistical differences in total PCB concentrations, grain size, total organic carbon (TOC), and metal concentrations in sediments between the five stations in the TI Pool.

H.2 Community Characteristics

A total of 86 taxa were collected from the five TI Pool stations from August 9-12, 1993. Table H-1 lists the taxa in rank order, excluding the colonial bryozoans. Colonial bryozoans were excluded from the numerical ranks as it was not possible to assign a discrete abundance value to the various modular colonies.

Approximately 90% of the total taxa collected were members of the following five major taxonomic groups (see Appendix C for profiles):

Isopods (sow bugs);
Chironomids (midge larvae);
Oligochaetes (aquatic worms);
Amphipods (scuds/sideswimmers); and
Pelecypods (mussels and clams).

The mean relative abundance of each of these five groups at the TI Pool stations is provided in Table H-2.

Of the dominant groups observed, chironomids, oligochaetes, and pelecypods are considered primarily infaunal (i.e., existing within the sediment), while the isopods and amphipods are considered to be primarily epifaunal (i.e., living at the surface of the sediment). These taxa are typically found in slow-flowing areas of the river in fine-grained sediments high in organic content. They represent a variety of trophic groups including suspension-feeders, deposit feeders, and predators.

H.2.1 Species/Taxa Richness

There were significant differences between species/taxa richness (s) at the five TI Pool stations (Figure H-1; Table H-3). Station 3 had the highest species richness with 56 taxa present (s=56), representing 65% of the 86 taxa collected in TI Pool during the study. Species richness at Stations 6 (s=48) and 4 (s=46) represented approximately 55% of the total taxa collected. Stations 5 (s=36) and 7 (s=28) had substantially lower species richness, representing approximately 42% of all taxa sampled in the TI Pool.

H.2.2 Species/Taxa Diversity

Diversity indices can be used to measuring the quality of the environment and the effect of induced stress on the structure of a benthic community. The advantage of species/taxa diversity is that it considers the evenness of occurrence of individuals of various species, while species richness does not. For example if there are two communities, one with a significant proportion of individuals belonging to just one species and the other with a more ecologically diverse community but the same number of individuals, species richness could not differentiate between the two components. To incorporate both the evenness and richness components, species diversity indices are often utilized (Ludwig and Reynolds, 1988).

The Simpson index (D_s) was employed in the TI Pool study to calculate species diversity. The Simpson index is more sensitive to the relative abundances of species and to dominance as opposed to evenness of species abundances (Magurran, 1988). Given the shifts in relative abundance in the TI Pool (Table H-3), the Simpson index was considered the most appropriate choice. Simpson (1949) showed that if two individuals are taken at random from a community, the probability that two will belong to the same species is given by:

$$l \quad \frac{n_i (n_i \quad 1)}{N(N \ 1)} \qquad \text{Equation H-1}$$

where:

l = Simpson dominance
n_i = abundance of species I
N = total abundance of all species

The Simpson dominance is used to calculate the Simpson diversity which is simply defined as:

$$D_s \quad l \quad 1 \qquad \text{Equation H-2}$$

The maximum possible diversity for N individuals among s species occurs when the abundance of each species (n_i) = N/s. Thus, the maximum possible value for D_s is given by:

$$D_{max} \quad \left(s \quad \frac{1}{s} \right) \left(\frac{N}{N \quad 1} \right) \qquad \text{Equation H-3}$$

Evenness (E_s) is a measure of the distribution of individuals among the component taxa; the higher the E_s, the more even the distribution. E_s is expressed as the nearness of the diversity index for the observed data D_s to the maximum theoretical diversity with perfect evenness equal to one (1.0):

$$E_s \quad \frac{D_s}{D_{max}} \qquad \text{Equation H-4}$$

The results of the diversity (D_s), evenness (E_s), and dominance (*l*) indices are provided in Table H-3. Diversity, evenness, and dominance were not significantly different between Stations 3, 4, and 6 (i.e., $p>0.05$). These three stations approached the maximum theoretical diversity of one (i.e., $E_s = 1$) and their low dominance indices, ranging from 0.13 at Station 3 to 0.17 at Station 6, indicate that these stations had taxonomic diversity. At Stations 5 and 7 all indices (D_s, E_s and *l)* were not significantly different from each other, but both stations were significantly different from the other three stations (Table H-4). The mean D_s and E_s values at Stations 5 and 7 were significantly lower than the other stations and the relatively high dominance indices (*l*) of 0.31 at Station 5 and 0.43 at Station 7 indicate that both stations were dominated by a single taxon. The dominant taxon at both Station 5 and 7 was the isopod *Caecidotea racovitzai* (Table H-2). The extremely large numbers collected of this species in comparison to other species accounts for the low diversity and evenness values at these two stations.

H.2.3 Abundance

The numerical abundance of benthic invertebrates at each station is provided in Table H-2 and shown in Figure H-2. Station 5 had a significantly greater number of benthic invertebrates than all the other stations due to the large numbers of *C. racovitzai* and Chironomidae present (Table H-2). There were no significant differences in the mean (total) number of benthic invertebrates between Stations 3, 4, 6, and 7 (Table H-4).

Analysis of the relative abundance of dominant groups is another medium by which to examine community-level characteristics. The numerical and relative percent abundance of the isopods, chironomids, oligochaetes, amphipods, and pelecypods collected at the five stations in the TI Pool are shown in Figures H-2 and H-3, respectively. Since representatives within these five major taxonomic groups collectively account for approximately 90% of all organisms collected during the study, these groups were used to examine community-level properties. Station 5 had approximately 2 to 22 times more individual isopods (14,256/m^2) and 2 to 5 times more chironomids (7,619/m^2) than all other

stations (Figure H-2; Table H-2). Overall, there is a distinct shift in the biotic profile from an isopod-dominated community at Stations 5 and 7 (51% and 61%, respectively) to a more equitable distribution of the major taxa at Stations 3, 4, and 6. For example, the relative percent abundances of oligochaetes, chironomids, amphipods (*Gammarus fasciatus*) and isopods (*C. racovitzai*) at Station 6 were fairly evenly distributed at 23%, 21%, 20%, and 15%, respectively (Figure H-3).

H.2.4 Biomass

Biomass at the TI Pool stations ranged from 38 to 233 gm/m^2 dry weight (Figure H-4). Several large pelecypods (*Elliptio* sp.) were sampled at Station 7 and a number of gastropods (*Valvata* sp.) were found at Station 3. Excluding these two species decreased the range of biomass sampled at the TI Pool stations (Figure H-4). Despite having the highest absolute number of individuals per sq m, Station 5 had the lowest unadjusted biomass of any of the stations.

The isopod *C. racovitzai* was the only species that had sufficient biomass replicates at each of the five stations to determine if there were significant species-specific biomass differences among the TI Pool stations. An analysis of the dry weight of *C. racovitzai* collected at all five stations within the TI Pool indicated that the average dry weight per organism at Station 5 (0.06 mg) was significantly lower than all the other TI Pool ecological stations. The average dry weight per isopod at Stations 3 (0.11 mg), 4 (0.10 mg), 6 (0.09 mg), and 7 (0.09 mg) did not differ significantly from one another. Although no data are available on the specific lengths of isopods from the community study, observations noted during the live sorting of invertebrates for PCBs analyses indicated that the isopod population at Station 5 contained a large number of juveniles.

H.2.5 Overall Community Similarity and Faunal Affinity

To quantitatively assess overall community similarity in a more robust fashion, an index was used to compare all species rather than just the dominant taxa. The Morisita index (I_m) of community similarity was selected for use since it is based on the Simpson's index of dominance (l). It ranges from zero (no similarity) to approximately 1.0 (identical) and is calculated as follows:

$$I_m \quad \frac{2s \quad x_i y_i}{(l_1 \quad l_2)(N_1 \quad N_2)} \qquad \text{Equation H-5}$$

where:

I_m	=	Morisita index
s	=	total number of species
l_1	=	Simpson dominance index for community 1
x_i	=	number of species I in community 1
N_1	=	total number of individuals in community 1
l_2	=	Simpson dominance index for community 2

y_i = number of species I in community 2
N_2 = total number of individuals in community 2

The Morisita indices calculated for the five stations are provided in Table H-4 and Figure H-5 presents a dendrogram formed by complete linkage clustering of the Morisita Indices. A hierarchical clustering technique was used to organize the similarity data into a series of partitions that range from a single similarity cluster, containing all the ecological stations, to individual station-to-station similarity clusters.

Two distinct clusters emerged from the dendrogram. One cluster, composed of Stations 5 and 7, has extremely high similarity (0.87) and is linked to the other major cluster, comprised of Stations 4, 3 and 6, of more moderate similarity (0.57) at a 0.21 fusion value. All the biotic data analyzed including the D_s, *l*, and E_s indices indicate that there are two major clusters of stations within the TI Pool. Stations 5 and 7 form a similar, lower richness, lower diversity, lower evenness, and higher dominance cluster and Stations 3, 4, and 6 form a higher richness, higher diversity, higher evenness, and lower dominance cluster.

H.2.6 Infauna Analysis

Analysis of the of the biotic profiles revealed that the dominance of one taxa, *C. racovitzai*, at Stations 5 and 7, accounts for the significant differences between the two clusters (one cluster being Stations 5 and 7, while the other cluster being Stations 3, 4, and 6). Because *C. racovitzai* is an epibenthic organism (see Appendix C), the data were reanalyzed excluding epifauna.

Two epifaunal taxa, the isopod *C. racovitzai* and the amphipod *G. fasciatus* were included in the fifteen taxa comprising more than 1% of the total number of individuals collected at the TI Pool. *C. racovitzai* accounted for about 35% of the total organisms collected, while *G. fasciatus* accounted for about 10%. In addition, at least one of the midge larvae *(Polypedilum* sp.) could be considered epifaunal since it is generally found in floating plant material and in detritus. Because *Polypedilum* only constituted slightly over 1% of the total number of organisms, only *C. racovitzai* and *G. faciatus* were removed from the species list for the infaunal analysis. Species diversity, including dominance and evenness, species richness, and abundance, were then calculated for infauna for each of the five stations.

The results of the infauna analysis for each station are presented in Table H-5. The results indicate that when epifaunal organisms are removed from the analysis, the species diversity, dominance, and evenness are similar between the five stations. With respect to abundance, Station 7 with 2,387 individuals per square meter, exhibited much lower mean values than the other four stations, which ranged from 8,825 at Station 4 to 13,044 at Station 5. Total abundance at each station declined sharply, with the exception of Station 3, when epifauna were removed from the analysis. Because the two epifaunal taxa removed for the infauna analysis were found at all stations, there were no meaningful changes in species richness.

The overall result of the infauna analysis indicate that when epifauna are excluded, the macroinvertebrate community characteristics, typically used to assess the quality of the environment (i.e., species diversity, dominance, and evenness), are comparable among the five TI Pool stations sampled. However, the abundance of individuals at Station 7 was much lower than at the other four stations.

H.3 Sediment Characterization

Benthic community structure is dependent on both biotic and abiotic parameters. Grain size, TOC, PCBs, and metals were measured at each station to evaluate the effects of abiotic parameters on community structure.

H.3.1 Grain Size

The relative percent distribution of medium sand, fine sand, silt and clay for the five stations sampled in the TI Pool is shown in Figure H-6. Overall, there is a shift from a more even distribution of sediment types at Station 3 to a greater percentage of fine sands and some silt at Stations 4 and 6, to more silt with some fine sands at Stations 5 and 7. There were significant differences in each grain size class among the TI Pool stations with the exception of the clays which ranged from about 1% to 2%.

Station 3 had the most even distribution of medium sand (31%), fine sand (33%), and silt (35%). In contrast, Stations 5 and 7 were dominated by silt with 58% and 56%, respectively. Stations 4 and 6 were predominately fine sand habitats. There were no significant differences in the percentage of fine sands at Stations 4 (60%) and 6 (61%).

H.3.2 Total Organic Carbon (TOC)

Percent TOC in sediment samples was not significantly different between Stations 3, 4, and 6 or between Stations 5 and 7 (Figure H-7; Table H-6). Stations 5 and 7 had a significantly higher percentage of TOC than Stations 3, 4, and 6. The TOC results correspond to what would be predicted based on the grain size data. Generally, depositional silt-laden environments have higher percentages of TOC than sand environments.

H.3.3 PCB Concentrations and Guideline Comparison

Total PCB concentrations at the five stations fell into the same two general groups as the Morisita index, TOC, and grain size. The total PCB concentration (29.32 mg/kg) at Station 5 was significantly greater than at Stations 3 (9.29 mg/kg), 4 (10.49 mg/kg), and 6 (14.33 mg/kg), and the total PCB concentration at Station 7 (18.51 mg/kg) was significantly greater than at Stations 3 and 4 (Table H-6; Figure H-8) . There were no significant differences in total PCBs between Stations 5 and

7 and among Stations 3, 4, and 6. Stations 3, 4, and 6 constitute a lower total PCB concentration cluster and Stations 5 and 7 comprise a higher total PCB concentration cluster.

Comparisons to various PCB sediment criteria and guidelines are shown in Table H-7. Consensus-based sediment effect concentrations (SECs) for PCBs in the Hudson River Basin were developed to support an assessment of the potential for injury to sediment-dwelling organisms (NOAA, 1999). The consensus-based SECs reflect the agreement that exists among various types of guidelines and:

- Provide an unifying synthesis of existing sediment quality guidance (SQG);
- Reflect causal rather than correlative effects; and
- Account for the effects of contaminant mixtures.

The SEC for PCBs refers to all of the polychlorinated biphenyls found in the Hudson River, plus the degradation products and metabolites of these chemicals. The SECs do not consider the potential for: 1) bioaccumulation in fish or other species that live in the water column; 2) bioaccumulation in aquatic organisms; or 3) potential effects that could occur throughout the food web as a result of the bioaccumulation of PCBs. The Hudson River SECs and the NYSDEC Technical Guidance for Screening Contaminated Sediments (NYSDEC, March 1998) were used as the primary sediment guidelines in this assessment.

The threshold effect concentration (TEC) is intended to identify the concentration of total PCBs below which adverse effects on sediment-dwelling organisms are unlikely to be observed (NOAA, 1999). The mid-range effect concentration (MEC) represents the concentration of total PCBs above which adverse effects on sediment-dwelling organisms are expected to be frequently observed and adverse effects are expected to be usually or always observed at PCB concentrations above the extreme effect concentration (EEC). Mortality was used as the measurement endpoint to determine adverse effects. Sediment guidance values for total PCBs were exceeded at all TI Pool stations (i.e., Stations 2 to 7; Table H-7), indicating the potential for adverse effects on local biota for chronic and acute exposures.

Water column samples taken from January through September 1993 from RM 194.6 to 156.5 were compared to NYSDEC Water Quality Criteria (WQC) (1998) in Table H-8. The chronic freshwater WQC was exceeded in some water column samples taken during May, June, and July (mean total PCB conc. 0.071 ug/L; max. total PCB conc. 0.226 ug/L). Wildlife criteria were exceeded by all water column samples taken.

H.3.4 Metals

Metals were analyzed in the samples collected at the five TI Pool stations, Rogers Island station (RM 194.1), and the background station (RM 203.3) to provide a broader focus of factors that may affect benthic community structure. The standard EPA Target Analyte List (TAL) of metals was analyzed consisting of aluminum, antimony, arsenic, barium, beryllium, cadmium, calcium, chromium,

cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, potassium, selenium, silver, sodium, thallium, vanadium, and zinc.

Many of the metals analyzed occur naturally at low levels in soils and sediment. Therefore, levels of analytes detected at the background station (Station 1) were used to screen out metals found at or below background concentrations. Metals were retained for further discussion if the mean concentration was greater than at the background station and the maximum concentration detected was greater than the severe effects level (SEL) of the NYSDEC sediment screening guidance (1998). Table H-9 contains a summary of the mean concentrations of metals detected at the background station, TI Pool benthic invertebrate stations, and NYSDEC sediment screening values.

Based on the screening, cadmium, chromium, lead, and mercury were retained for further examination. The concentrations of each of these metals are provided in Table H-10 and histograms for each station are shown in Figures H-9 to H-12.

A one-way analysis of variance (ANOVA) was performed on each analyte to determine if there were significant differences between any of the groups, and if so, which groups differed significantly. To test the differences among the means, the least significant difference (LSD), used for planned comparisons, was calculated (Table H-6).

Station 5, with a mean cadmium concentration of 8.8 ppm had a significantly higher concentration of cadmium than all other stations ($p< 0.05$) and neared the NYSDEC cadmium SEL of 9 ppm. All stations exceeded the NYSDEC LEL of 0.6 ppm.

Stations 5 and 7 with mean chromium values of 192 and 121 ppm, respectively had concentrations above the chromium SEL of 110 ppm. The concentrations at these two stations were significantly greater than concentrations detected at all other stations ($p< 0.005$). Mean chromium concentrations at all stations exceeded the LEL of 26 ppm.

Lead was detected at an average of 187 ppm at Station 5, which was above the SEL and significantly higher than all other stations ($p< 0.005$). Station 7 with a mean concentration of 81 ppm, had a significantly higher concentration of lead than Stations 4 and 6 ($p < 0.05$). All stations had average lead concentrations above the LEL of 31 ppm.

Station 5 had a significantly higher concentration of mercury than all other stations ($p<0.005$). The mean concentration of 1.9 ppm at Station 5 was above the SEL of 1.3 ppm, and the remainder of the stations, with the exception of Station 7, had average concentrations above the LEL of 0.15 ppm.

H.3.4.1 Metals Toxicity

Cadmium is a known teratogen and carcinogen, a probable mutagen, and has been implicated as the cause of severe deleterious effects on fish and wildlife (Eisler, 1985). Concentrations of cadmium in freshwater above 10 ppb are associated with high mortality, reduced growth, inhibited

reproduction and other effects. Effects are most pronounced in waters of comparatively low alkalinity. Adsorption and desorption rates of cadmium are rapid on mud solids and particles of clay, silica, humic material, and other naturally occurring solids. The cadmium Threshold Effect Level (TEL) for the amphipod *Hyalella azteca* in 28 day toxicity tests was 0.58 ppm, and the Probable Effects Level (PEL) was 3.2 ppm (Ingersoll et al., 1996).

Chromium toxicity is dependent on speciation, with the hexavalent form considered the most toxic. Although chromium is an essential trace element in many species, at high environmental concentrations chromium is a mutagen, teratogen, and carcinogen (Eisler, 1986). The cadmium TEL for *H. azteca* in 28 day toxicity tests was 0.58 ppm, and the PEL was 3.2 ppm (Ingersoll et al., 1996).

Lead is a cumulative metabolic poison and can cause mutagenic, teratogenic and carcinogenic effects (Eisler, 1988). It can also impair reproduction and thyroid functions, and interferes with resistance to infectious diseases (USEPA, 1979). Lead poisoning in fish results in neurological and muscular disorders as well as changes in blood chemistry. The TEL for *H. azteca* in 28 day toxicity tests was 37 ppm and the PEL was 82 ppm (Ingersoll et al., 1996). Avian receptors appear to be more sensitive to lead exposure than mammalian receptors. Acute lead poisoning in waterfowl has been identified as a major problem associated with the use of lead shot by waterfowl hunters.

Mercury is a mutagen, teratogen, and carcinogen that can cause cytochemical and histopathological effects. Bioavailability of mercury is generally determined by biologically mediated reactions, mainly bacterial transformation of inorganic mercury to methylmercury. Methylmercury is the most readily bioaccumulated form of mercury in aquatic systems and is both highly persistent and toxic.

The concentrations of metals in sediments is influenced by several factors including grain size. Fine-grained sediments usually have higher concentrations of contaminants than coarser sediments and benthic communities show the most diversity in heterogeneous sediment types. Cluster 1 (Stations 5 and 7) had the highest concentrations of PCBs, metals, and a higher percentage of fine-grained sediment and lower species richness and eveness than Cluster 2 (Stations 3, 4, and 6).

Generally, concentrations of metals, PCBs, and TOC are positively correlated. All four of the metals examined here were significantly correlated with total PCBs ($p<0.05$). The R^2 ranged from 0.22 (mercury) to 0.49 (lead) for the log-normalized regressions.

H.4 Summary of the TI Pool Study

The benthic invertebrate communities present in the TI Pool can be divided into two distinct clusters. The first cluster (Cluster 1), comprised of Stations 5 and 7, exhibited lower species richness, species diversity, dominance, and diversity than Cluster 2, which was made up of Stations 3, 4 and 6. Differences in community characteristics may be a function of sediment characteristics. The stations in Cluster 1 have a higher proportion of fine-grained silty material than stations in Cluster 2. With the exception of the more even grain size distribution at Station 3, Cluster 2 can be

characterized as a predominately fine sand habitat with a lower percentage of silt. Metals, PCBs, and other contaminants tend to accumulate in depositional areas with fine-grained sediments with relatively high TOC concentrations.

With respect to grain size, sediments with more heterogeneous distribution of grain size classes can often support a greater variety of species than more homogeneous environments. These heterogeneous environments generally present a greater range of food and afford more areas of protection from predation pressures. The greater heterogeneity of grain sizes at the Cluster 2 stations is likely associated with greater species richness and species diversity found at those stations.

It is generally recognized that the genus *Caecidotea* is characteristically found in organically rich areas with relatively high content of particulate matter (Kerr 1978; Smith, pers. comm.). Hence, the more silt-laden, high TOC environment of the Cluster 1 stations provides more suitable habitat for this isopod than do the stations comprising Cluster 2. The overwhelming dominance of this one species of isopod at the Cluster 1 stations accounts for the low diversity and evenness values compared to the Cluster 2 stations.

The infauna analysis provides information on ecologically based linkages and relationships to PCBs. The abundance of the isopods at the Cluster 1 stations, which is what drives the significant differences in community characteristics, can be explained by the silt-laden, high TOC environments of the Cluster 1 stations. In addition, pore water is considered to be the toxic component within the sediment that potentially impacts benthic organisms. Infaunal organisms have significantly more contact with pore water than the epifauna that are more closely associated with the sediment/water interface and the surfaces of submergent vegetation.

Species diversity, dominance, and evenness are similar between all five stations when only infauna are considered. However, the abundance of infauna at Station 7 (one of the two Cluster 1 stations) is strikingly low compared to the other four stations. In contrast, Station 5 with the highest abundance had the lowest total biomass of any of the TI Pool stations. The abundance or growth and reproduction of benthic macroinvertebrates may be restricted at these two stations, possibly by the higher PCBs.

Comparison of total PCB concentrations in the sediment to federal and NY State guidance indicates that the levels of PCBs found in the TI Pool may cause adverse effects to aquatic life. Sediment and water quality guidance also indicate that wildlife may be more severely affected (via bioaccumulation) by the concentrations of PCBs present in the Hudson River than the aquatic organisms living in the river.

H.5 Lower Hudson Benthic Invertebrate Community Analysis

Macroinvertebrates in the Lower Hudson River represent a heterogeneous group of organisms with a wide range of life history strategies. Numerous studies and reviews of invertebrates throughout

the Lower Hudson indicate that they are distributed in distinct spatial patterns (e.g., Ristich et al., 1977; Weinstein, 1977; Gladden et al., 1988; and Moran and Limburg, 1986). The salt water reaches below RM 25 support a typical marine assemblage of benthic invertebrates including marine oligochaetes, polychaetes and crustaceans. The brackish reaches from RM 25 to 60 have a mixture of freshwater and marine forms, and the upper reaches above RM 60 are dominated by freshwater arthropods and oligochaetes.

Simpson et al. (1985) found that the freshwater macrobenthic fauna of the main channel of the Hudson River between Glenmont (RM 141.1) and New Hamburg (RM 67.4) consisted primarily of oligochaete worms, midge larvae, crustaceans and bivalves. The most abundant taxa were the oligochaetes, which represented approximately 54% to 79% of the total macrofauna. Benthic invertebrate populations in the middle reaches of the Lower Hudson River estuary are numerically dominated by oligochaetes, polychaetes, amphipods, and isopods. These taxa may account for more than 70% of the benthos in many regions (Texas Instruments, 1976).

The following discussion focuses on the dominant benthic invertebrates at each of five stations in the Lower Hudson River. The five stations are designated as significant coastal fish and wildlife habitats (NYS Dept. of State, 1990), and four of the five stations are part of the Hudson River Natural Estuarine Research Reserve (NERR) administered by NYS in partnership with NOAA. Benthic invertebrates sampled at each station and their relative percent abundance are summarized in Table H-11. Figures H-13, H-14A to H-14E illustrate the total species richness and relative percent abundance of macroinvertebrates for the Lower Hudson stations.

H.5.1 Stockport Creek and Flats (Station 12)

Stockport Creek and Flats had an average abundance of 5,289 individuals/m^2 and a mean biomass of 63 mg/m^2 (dry weight) (Table H-12). The community was dominated by oligochaetes and chironomids (Table H-11). Average species richness and diversity (D_s) were 14 and 0.70, respectively (Table H-12). The bottom sediments are primarily composed of silt (52%) and fine sand (40%) with relatively small percentages of medium sand and clay (Figure H-15).

The dominant chironomids sampled here included *Procladius* sp., *Polypedilum* sp., and indeterminate members of the family Chironomidae and subfamily Chironominae (Figure H-14A; Table H-11). These species are generally associated with depositional areas and organically enriched waters. *Polypedilum* sp. was found to be one of the most abundant chironomids in areas of silty sand throughout other freshwater sections of the Hudson River (Simpson et al., 1984). Oligochaetes, the other dominant taxa, are well documented members of organically enriched zones (Pennak, 1989).

H.5.2 Tivoli Bays (Station 14)

Tivoli Bays had an average abundance of 4,524 individuals/m^2 and a biomass of 126 mg/m^2 (Table H-12). Average species richness and diversity were 16 and 0.82, respectively (Table H-11). The bottom sediments are predominantly silt (77%) with some fine sand (16%) and relatively small percentages of medium sand and clay (Figure H-15).

Chironomids were the dominant macroinvertebrates sampled (Figure H-14B; Table H-11). Species found included *Dicrotendipes* sp., *Procladius* sp., *Polypedilum* sp., *Clinotanypus* sp., and indeterminate members of the subfamily Chironominae (Table H-11). All four genera of chironomids identified prefer silty sediments. For example, *Dicrotendipes* sp. is a burrower in soft sediments and gathers fine particulate organic matter from the surficial sediments. *Clinotanypus* sp. is also a burrower in depositional zones and preys on oligochaetes, ostracods, cladocerans and other chironomid larvae.

H.5.3 Esopus Meadows (Station 15)

At the Esopus Meadows station the macrofauna was dominated by oligochaetes, cladocerans (Chydoridae family), and the chironomids *Coelotanypus* sp., *Clinotanypus* sp., and *Polypedilum* sp. (Figure H-14C; Table H-11). The average abundance of 2,551 individuals/m^2 (Table H-12) was the lowest of all the stations sampled in the Upper and Lower Hudson, although the species present were distributed relatively evenly. Biomass was also low at 65 mg/m^2 (Table H-12). Average species richness and diversity were 11 and 0.86, respectively (Table H-12). The bottom sediments were characterized as a silt (59%) and fine sand (27%) environment, with some medium sand (11%), and a relatively small percentage of clay (Figure H-15).

The dominant genera of chironomids found at this station were those generally associated with more depositional zones and organically enriched waters. *Clinotanypus* sp. and *Coelotanypus* sp. are both considered burrowers and prefer more silt laden sediments. As previously indicated, *Polypedilum* sp. also prefers more silt laden environments. In addition, oligochaetes are also well represented in many depositional areas.

H.5.4 Iona Island (Station 17)

Iona Island had an average abundance of 5,136 individuals/m^2 and a biomass of 365 mg/m^2 (Table H-12). *Hobsonia florida* (polychaete), oligochaetes, *Gammarus fasciatus* (amphipod), and *Clinotanypus* sp. were the most frequent species found here (Table H-11; Figure H-14D). Average species richness and diversity were 9 and 0.71, respectively (Table H-12). The bottom sediments were predominantly silt (81%) with some fine sand (13%) and relatively small percentages of medium sand and clay (Figure H-15).

The silty environment at this station favors the establishment of deposit feeding polychaetes, such as *H. florida*, and euryhaline oligochaetes. As mentioned previously, the chironomid *Clinotanypus* sp. is a burrower in depositional zones and is known to prey on oligochaetes as well as other crustaceans. The amphipod *G. fasciatus* is an epibenthic omnivore that feeds on a variety of detritus and dead animal matter characteristic of depositional areas.

H.5.5 Piermont Marsh (Station 18)

Piermont Marsh had an average abundance of 6,480 individuals/m^2 and a mean biomass of 291 mg/m^2 (Table H-12). Oligochaetes, *Cyathura polita* (isopod), *H. florida*, *Hydrobia minuta* (gastropod), unidentified isopods, *Clinotanypus* sp., and *G. fasciatus* were fairly evenly distributed here (Figure H-14E; Table H-11). Average species richness and diversity were 9 and 0.84, respectively (Table H-12). At the time of sampling, the salinity was 9.0 ppt. The bottom sediments are predominately silt (78%) with some fine sand (12%) and relatively small percentages of medium sand and clay (Figure 6.1-15).

This silt laden station is dominated by deposit feeders, such as polychaetes and oligochaetes. The presence of the mud snail *H. minuta* also indicates that this environment favors many deposit feeding benthic invertebrates. The isopod *C. polita* was found in depositional areas throughout the brackish Lower Hudson and was the second most abundant organism collected at Piermont Marsh.

H.5.6 Sediment Characterization

H.5.6.1 Grain Size

Silt dominated the Lower Hudson River stations (Figure H-15). There was an increase in fine grained sediments (silt and clay) at the two stations closest to the salt front, Stations 17 and 18 (Iona Island and Piermont Marsh, respectively).

H.5.6.2 Total Organic Carbon

TOC levels were relatively consistent along the Lower Hudson River, ranging from a mean of 2.0% at Station 18 to 3.6% at Station 15 (Figure H-16). Stations 15 and 17 had significantly higher percentages of TOCs than the other Lower Hudson stations ($p<0.05$; one-way ANOVA).

H.5.6.3 Total PCBs

The mean total PCB concentration varied from 367 µg/kg at Tivoli Bays (Station 14) to 1,313 µg/kg at Iona Island (Station 17) (Figure H-17). Iona Island and Stockport Flats (Stations 17 and 12, respectively) had significantly higher concentrations of total PCBs than Stations 14, 15, and 18 ($p<0.05$; one-way ANOVA). Esopus Meadows (Station 15) had a higher PCB concentration than Tivoli Bays or Piermont Marsh (Stations 13 and 18).

All stations had mean total PCB concentrations above all the TEC (0.04 ug/kg) and MEC (0.4 ug/kg), with the exception of Stations 14 (Tivoli Bays) and Station 18 (Piermont Marsh), which were above the TEC but slightly below the MEC.

H.5.7 Lower Hudson River Summary

The Lower Hudson River benthic macroinvertebrate communities reflect the varied habitats and conditions found along the river. Because of the habitat diversity and salinity gradient found in the Lower Hudson, it is difficult to make direct comparisons between any of the stations. Station 14 had the highest species richness of the Lower Hudson stations (Figure H-12) and the lowest mean total PCB concentration.

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Table H-1

Benthic Invertebrates Collected at TI Pool Stations

Taxa in Rank Order	Common Name	Mean % of Total Ind. Collected
Caecidotea racovitzai	Isopod (sowbug)	34.6
Chironomidae[1]	Midges	-30.2
Oligochaeta	Aquatic worms	14.3
Gammarus fasciatus	Amphipod	10.3
Pisidium sp.	Pill Clam	5.0
Canthocamptes sp.	Harpacticoid copepod	1.5
Nematoda	Nematods (worms)	1.1
Phylocentropus sp.	Caddis fly larvae	<1.0
Dubiraphia sp.	Beetle larvae	<1.0
Menetus sp.	Caddis fly larvae	<1.0
Valvata sp.	Snail	<1.0
Sialis sp.	Alderfly larvae	<1.0
Oecetis sp.	Caddisfly larvae	<1.0
Probezzia sp.	Biting midges	<1.0
Enallagma sp.	Damselfly nymph	<1.0
Chydoridae	Water fleas (Cladoceran)	<1.0
Acariformes	Mites	<1.0
Amnicola sp.	Snail	<1.0
Mystacides sp.	Caddisfly larvae	<1.0
Diaphanosoma sp.	Water fleas (Cladoceran)	<1.0
Ceratopogonidae	Biting midges	<1.0
Helobdella fusca	Leech	<1.0
Arthropoda	Arthropods	<1.0
Eukiefferiella sp.	Biting Midges	<1.0

Table H-1 (Continued)

Benthic Invertebrates Collected at TI Pool Stations

Taxa in Rank Order	Common Name	Mean % of Total Ind. Collected
Turbellaria	Flatworms	<1.0
Dugesia tigrina	Flatworm	<1.0
Bithynia tentaculata	Snail	<1.0
Trichoptera	Caddisfly larvae	<1.0
Chydorus sp.	Water fleas (Cladoceran)	<1.0
Caenis sp.	Mayfly nymph	<1.0
Physa sp.	Snail	<1.0
Helobdella sp.	Leech	<1.0
Mesocyclops sp.	Cyclopoid copepods	<1.0
Orthotrichia sp.	Caddis fly larvae	<1.0
Aeschnidae	Dragonfly nymph	<1.0
Hexagenia sp.	Mayfly nymph	<1.0
Hirudinea	Leeches	<1.0
Neureclipsis sp.	Caddisfly larvae	<1.0
Culicoides sp.	Mosquito larvae	<1.0
Corixidae	Water boatman	<1.0
Neoperla sp.	Stonefly nymph	<1.0
Caenidae	Mayfly nymph	<1.0
Donacia sp.	Beetle	<1.0
Hemiptera	True bugs	<1.0
Molanna sp.	Caddisfly larvae	<1.0
Copepoda	Copepods	<1.0
Insecta	Insects	<1.0
Baetidae	Mayfly nymph	<1.0
Macronychus sp.	Riffle beetle	<1.0

Table H-1 (Continued)

Benthic Invertebrates Collected at TI Pool Stations

Taxa in Rank Order	Common Name	Mean % of Total Ind. Collected
Tipulidae	Cranefly larvae	<1.0
Cymatia sp.	Water boatman	<1.0
Notonecta sp.	Water boatman	<1.0
Talitridae	Amphipod	<1.0
Baetis sp.	Mayfly nymph	<1.0
Dromogomphus sp.	Dragonfly nymph	<1.0
Oxyethira sp.	Caddis fly larvae	<1.0
Diptera	Flies and midges	<1.0
Atherix sp.	Snipe fly	<1.0
Tabanidae	Horse fly larvae	<1.0
Elliptio sp.	Eastern elliptio mussel	<1.0

Notes: Taxa are listed in order of absolute abundance.
Mean Percent of individuals is based on the mean of Stations 3 to 7.
[1] Chironomidae were primarily composed of Chironominae, *Proclaudius* sp., *Tanytarsus* sp., *Dicrotendipes* sp., *Polypedilum* sp., *Clinotanypus* sp., *Tribelos jucundus*, and Tanypodinae.

Table H -2

Relative Abundance of Five Dominant Taxonomic Groups at TI Pool Stations

Group/Taxa	Station 3		Station 4		Station 5		Station 6		Station 7	
	Abund.	Percent	Abund.	Percent	Abund.	Percent	Abund.	Percent	Abund.	Percent
	ind/m^2		ind/m^2		ind/m^2		ind/m^2		ind/m^2	
Total Dominant Isopoda	**653**	**5.6%**	**3245**	**24.6%**	**14256**	**50.9%**	**2347**	**15.2%**	**7286**	**60.9%**
Caecidotea racovitzai										
Total Dominant Chironomids	**3775**	**32.3%**	**3959**	**30.1%**	**7619**	**27.2%**	**3277**	**21.3%**	**1561**	**13.0%**
Unidentified Chironomidae	1398	12.0%	122	0.9%	2232	8.0%	293	1.9%	398	3.3%
Unidentified Chironominae	510	4.4%	1490	11.3%	374	1.3%	1378	8.9%	41	0.3%
Procladius sp.	479	4.1%	204	1.5%	1474	5.3%	128	0.8%	296	2.5%
Tanytarsus sp.	255	2.2%	0	0.0%	1409	5.0%	26	0.2%	0	0.0%
Dicrotendipes sp.	479	4.1%	337	2.6%	560	2.0%	38	0.2%	204	1.7%
Polypedilum sp.	82	0.7%	102	0.8%	396	1.4%	281	1.8%	224	1.9%
Clinotanypus sp.	51	0.4%	133	1.0%	200	0.7%	332	2.2%	194	1.6%
Tribelos jucundus	0	0.0%	867	6.6%	0	0.0%	0	0.0%	0	0.0%
Unidentified Tanypodinae	112	1.0%	571	4.3%	131	0.5%	38	0.2%	0	0.0%
Tribelos sp.	214	1.8%	51	0.4%	194	0.7%	128	0.8%	204	1.7%
Chironomus sp.	41	0.3%	41	0.3%	650	2.3%	0	0.0%	0	0.0%
Cricotopus trifascia	102	0.9%	41	0.3%	0	0.0%	306	2.0%	0	0.0%
Unidentified Orthocladiinae	51	0.4%	0	0.0%	0	0.0%	332	2.2%	0	0.0%
Total Dominant Oligochaeta	**2918**	**25.0%**	**2245**	**17.0%**	**2681**	**9.6%**	**3584**	**23.3%**	**71**	**0.6%**
Unidentified Oligochaeta										
Total Dominant Amphipoda	**1030**	**8.8%**	**1102**	**8.4%**	**682**	**2.4%**	**3176**	**20.6%**	**2296**	**19.2%**
Gammarus fasciatus										
Total Dominant Pelecypoda	**1245**	**10.6%**	**1581**	**12.0%**	**49**	**0.2%**	**1097**	**7.1%**	**0**	**0.0%**
Pisidium sp.										
Subtotals	**9621**	**82.3%**	**12132**	**92.1%**	**25287**	**90.4%**	**13482**	**87.5%**	**11214**	**93.7%**
Total Abundance (all taxa)	**11691**		**13172**		**27983**		**15407**		**11968**	

Table H-3

Summary of Diversity Indices and Abundance Data - TI Pool

Station Mean	D_s	I	D_{max}	E_s	Species Richness	Abundance Ind./Sq M
Station 3	0.87	0.13	0.96	0.90	27	11,691
Station 4	0.83	0.17	0.95	0.87	21	13,172
Station 5	0.69	0.31	0.95	0.73	19	27,983
Station 6	0.84	0.16	0.96	0.88	24	15,407
Station 7	0.57	0.43	0.91	0.61	14	11,968
TI Pool Mean	0.76	0.24	0.95	0.80	21	16,044

Table H-4

Statistical Summary of TI Pool Data

	D_s	*I*	D_{max}	E_s	Abundance	Biomass	Morisita
					No. Ind./m^2	gms/m^2	Index
Comparison:							
Station 3 vs. Station 4	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	0.57
Station 3 vs. Station 5	**3>5***	**5>3***	n.s.	**3>5***	**5>3***	n.s.	0.32
Station 3 vs. Station 6	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	0.69
Station 3 vs. Station 7	**3>7***	**7>3***	n.s.	**3>7***	**n.s.**	n.s.	0.21
Station 4 vs. Station 5	**4>5***	**5>4***	n.s.	**4>5***	**5>4***	n.s	0.62
Station 4 vs. Station 6	n.s.	n.s.	n.s.	n.s.	n.s.	n.s	0.69
Station 4 vs. Station 7	**4>7***	**7>4***	n.s.	**4>7***	n.s.	n.s	0.56
Station 5 vs. Station 6	**6>5***	**5>6***	n.s.	**6>5***	**5>6***	**6>5***	0.46
Station 5 vs. Station 7	n.s.	n.s.	n.s.	n.s.	**5>7***	n.s	0.87
Station 6 vs. Station 7	**6>7***	**7>6***	n.s.	**6>7***	n.s.	n.s.	0.46

Notes: *Significant at p<0.05

n.s. = not significant

Biomass excludes mollusks found at Stations 3 and 7

Table H-5

Summary of Infauna and Total Benthos Indices - TI Pool

Station	Simpson Diversity D_s		Simpson Dominance I		Eveness Distribution		Species Richness		Abundance No. Ind./Sq M	
	Infauna	Total Benthos	Infauna	Total Benthos	Infauna	Total Benthos	Infauna	Total Benthos	Infauna	Total Benthos
3	0.84	0.87	0.16	0.13	0.88	0.90	25	27	10,008	11,691
4	0.79	0.83	0.21	0.17	0.84	0.87	19	21	8,825	13,172
5	0.81	0.69	0.19	0.31	0.87	0.73	17	19	13,044	27,983
6	0.78	0.84	0.22	0.16	0.82	0.88	22	24	9,884	15,407
7	0.84	0.57	0.16	0.43	0.95	0.61	12	14	2,387	11,968
TI Pool Grand Mean	0.81	0.76	0.19	0.24	0.87	0.80	19	21	8,830	16,044

Notes: Total benthos equals the sum of infaunal and epibenthic macroinvertebrates

Table H-6

Summary of TI Pool ANOVAs

	Cadmium	Chromium	Lead	Mercury	TOC	Total PCBs
Comparison:						
Station 3 vs. Station 4	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Station 3 vs. Station 5	**5>3***	**5>3****	**5>3****	**5>3****	**5>3****	**5>3****
Station 3 vs. Station 6	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Station 3 vs. Station 7	n.s.	**7>3****	n.s.	n.s.	**7>3****	**7>3***
Station 4 vs. Station 5	**5>4****	**5>4****	**5>4****	**5>4****	**5>4****	**5>4****
Station 4 vs. Station 6	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Station 4 vs. Station 7	n.s.	**7>4****	**7>4***	n.s.	**7>4***	**7>4***
Station 5 vs. Station 6	**5>6****	**5>6****	**5>6****	**5>6****	**5>6****	**5>6****
Station 5 vs. Station 7	**5>7***	**5>7****	**5>7****	**5>7****	**5>7***	**5>7***
Station 6 vs. Station 7	n.s.	**7>6****	**7>6***	n.s.	**7>6****	n.s
Statistics: One-way ANOVA followed by LSD						
*Significant at p<0.05						
**Significant at p<0.005						
n.s. = not significant						

Table H-7

Selected Sediment Screening Guidelines: PCBs

		Total PCBs	Aroclor 1254	Aroclor 1248	Aroclor 1016	Aroclor 1260	Aroclor 1242
Sediment Guidelines/Effect Levels							
Hudson River Sediment Effect Concentrations (NOAA, 1999) - mg/kg (ppm)							
	Threshold Effect Concentration	0.04					
	Mid-range Effect Concentration	0.4					
	Extreme Effect Concentration	1.7					
NYSDEC (1998) Freshwater (ug/gOC)							
	Benthic Aquatic Life Acute Toxicity	2760.8					
	Benthic Aquatic Life Chronic Toxicity	19.3					
	Wildlife Bioaccumulation	1.4					
Ontario Ministry of the Environment							
Freshwater Guidelines (Persaud et al., 1993)							
	No Effect Level (ug/g)	0.01					
	Lowest Effect Level (ug/g)	0.07	0.06	0.03	0.007	0.005	
	Severe Effect Level (ug/g OC)	530	34	150	53	24	
Long et al. (1995) Marine & Estuaries- ppb							
	Effects-Range-Low	22.7					
	Effects-Range-Median	180					
Ingersoll et al. (1996) Freshwater Guidelines							
based on *Hyallela azteca* - ppb							
	Effects-Range-Low	50					
	Effects-Range-Median	730					
	Threshold Effect Level	32					
	Probable Effect Level	240					
	No Effect Concentration	190					
Washington State (1997) Freshwater - ppb							
	Probable Apparent Effects Threshold - Microtox	21	7.3	21			
	PAET - *Hyalella azteca*	450	240				100
	Apparent Effects Threshold - Microtox	21	7.3				
	AET - *Hyalella azteca*	820	350				100
	Apparent Effects Threshold - Microtox mg/kg OC	2.6	0.73				
	AET - *Hyalella azteca* mg/kg OC		18				
Jones et al. (1997) ppb; Eq-P-derived assuming 1% OC							
Recommended TOC adjustment							
	Secondary Chronic Values		810	1000		4500000	
Notes: All values are provided in dry weight unless noted							
	Mean PCB conc.Upper Hudson benthic stations: 9.292 - 29.320 ppm						
	Mean PCB conc.Lower Hudson benthic stations: 0.367 - 1.313 ppm						

Table H-8

Federal and New York State PCB Water Quality Criteria

			Total PCB Water Quality Criteria			Upper Hudson	
					(ug/L)	1993 (ug/L)	
USEPA/NYSDEC - Benthic Aquatic Life						Average	Maximum
	Acute Toxicity - Freshwater				2		
	Acute Toxicity - Saltwater				10		
	Chronic Toxicity - Freshwater				0.014	0.071	0.226
	Chronic Toxicity - Saltwater				0.03		
NYSDEC - Wildlife Bioaccumulation							
	Freshwater				0.001	0.071	0.226
	Saltwater				0.001		
NYSDEC Surface Water Standards							
	Wildlife Criterion				0.00012	0.071	0.226
Sources: NYSDEC June, 1998 and March 1998; USEPA, 1991							

Table H-9

Comparison of Metals Concentrations to NYSDEC Guidance

	Background	Average	Maximim	NYSDEC	NYSDEC	Retained ?
	Station 1-ave.	Stations 3 -7	Stations 3 -7	LEL	SEL	
Metals	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	
Aluminum	13167	11907	23100			NO
Antimony	15	8	15	2	25	NO
Arsenic	2	1	4	6	33	NO
Barium	67	100	180			NO
Beryllium	1	1	2			NO
Cadmium	1	4	12	0.6	9	YES
Calcium	5137	4174	6430			NO
Chromium	13	100	234	26	110	YES
Cobalt	10	8	16			NO
Copper	26	29	53	16	110	NO
Iron	18767	17136	27900	2%	4%	NO
Lead	55	79	264	31	110	YES
Magnesium	2537	3264	5570			NO
Manganese	292	270	525	460	1100	NO
Mercury	0	1	3	0.15	1.3	YES
Nickel	18	17	36	16	50	NO
Potassium	1194	1557	2910			NO
Selenium	1	1	1			NO
Silver	2	1	3	1	2.2	NO
Sodium	633	393	629			NO
Thallium	5	1	2			NO
Vanadium	32	26	48			NO
Zinc	322	152	292	120	270	NO
Notes:						
1) NYSDEC Lowest Effect Level (LEL) and Severe Effect Level (SEL)						
are based on Persaud et al. (1993) and Long et al. (1995)						
2) Metals were retained for further discussion if the average was above background levels						
and the maximum concentration was greater or equal to the SEL						

Table H -10

Selected Metals, PCB, and TOC Concentrations

Contaminant		Station 1	Station 2	Station 3	Station 4	Station 5	Station 6	Station 7
Cadmium	mg/kg	1.2	0.8	4.0	0.8	8.8	2.1	3.7
Chromium	mg/kg	12.8	60.4	62.7	51.5	191.6	74.7	121.4
Lead	mg/kg	55.4	87.1	49.7	43.8	186.6	32.8	80.9
Mercury	mg/kg	0.21	0.14	0.47	0.16	1.92	0.08	0.13
PCBs	ug/kg	72	20626	9292	10491	29320	14325	18515
TOC	%	8.3	4.9	2.4	3.5	7.4	2.6	5.8

Table H-11

Relative Percent Abundance of Macroinvertebrates - Lower Hudson River

Station 12		Station 14		Station 15		Station 17		Station 18	
Species/Group	%	Species/Group	%	Species/Group	%	Species/Group	%	Species/Group	%
Oligochaeta	42.4%	Chironominae Indet.	36.1%	Oligochaeta	22.0%	*Hobsonia florida*	36.1%	Oligochaeta	18.4%
Chironominae Indet.	12.9%	*Dicrotendipes sp.*	10.5%	Chydoridae	17.3%	Oligochaeta	32.8%	*Cyathura polita*	16.5%
Chironomidae Indet.	10.3%	*Procladius sp.*	10.2%	*Coelotanypus sp.*	14.0%	*Gammarus fasciatus*	11.3%	*Hobsonia florida*	14.2%
Procladius sp.	8.0%	*Polypedilum sp.*	9.0%	Nematoda	7.3%	*Clinotanypus sp.*	6.3%	*Hydrobia minuta*	11.5%
Polypedilum sp.	7.1%	*Clinotanypus sp.*	6.4%	*Clinotanypus sp.*	6.0%	Nemotoda	3.3%	Isopoda	10.8%
Pisidium sp.	2.9%	Oligochaeta	4.1%	*Polypedilum sp.*	5.3%	*Cyathura polita*	2.0%	*Clinotanypus sp.*	10.0%
Tribelos sp.	2.9%	*Gammarus fasciatus*	2.6%	Acariformes	4.0%	*Coelotanypus sp.*	2.0%	*Gammarus fasciatus*	9.7%
Cryptotendipes sp.	2.9%	*Pisidium sp.*	2.3%	*Dicrotendipes sp.*	4.0%	*Procladius sp.*	1.7%	Ostracoda	4.5%
Tanytarsus sp.	2.3%	Chironomidae Indet.	2.3%	*Cladotanytarsus sp.*	3.3%	Pelecypoda	1.3%	*Neanthes succinea*	1.3%
Chironomus sp.	1.9%	*Amnicola limosa*	1.9%	*Amnicola sp.*	3.3%	*Neanthes succinea*	1.0%	Pelecypoda	1.3%
Gammarus fasciatus	1.0%	*Cladotanytarsus sp.*	1.5%	*Synorthocladius sp.*	2.7%	Bryozoa	0.7%	*Procladius sp.*	1.0%
Acariformes	0.6%	*Orthotrichia sp.*	1.1%	*Pisidium sp.*	2.7%	*Balanus improvisus*	0.7%	*Rhithropanopeus harrisii*	0.5%
Tanypodinae Indet.	0.6%	Nematoda	1.1%	*Tribelos sp.*	2.0%	Isopoda	0.3%	*Coelotanypus sp.*	0.3%
Clinotanypus sp.	0.6%	Gastropoda	1.1%	Cyclopoida	1.3%	Orthocladiinae	0.3%		
Coleoptera	0.3%	*Cricotopus bicinctus*	1.1%	*Gammarus fasciatus*	1.3%	*Dicrotendipes sp.*	0.3%		
Bithynia tentaculata	0.3%	*Tanytarsus sp.*	1.1%	Hydroptilidae	1.3%				
Valvata sp.	0.3%	*Triaenodes sp.*	0.8%	*Cyathura polita*	0.7%				
Nematoda	0.3%	Orthocladiinae Indet.	0.8%	*Hydroptila sp.*	0.7%				
Cyathura polita	0.3%	*Chironomus sp.*	0.8%	*Chironomus sp.*	0.7%				
Ostracoda	0.3%	Acariformes	0.4%						
Leptoceridae	0.3%	*Dugesia tigrina*	0.4%						
Ceratopognidae	0.3%	*Diaphanosoma sp.*	0.4%						
Hemiptera	0.3%	*Hydroptila sp.*	0.4%						
Nilothauma sp.	0.3%	*Probezzia sp.*	0.4%						
Cryptochironomus sp.	0.3%	*Bithynia tentaculata*	0.4%						
		Tanypodinae Indet.	0.4%						
		Synorthocladius sp.	0.4%						
		Tribelos sp.	0.4%						
		Djalmabatista sp.	0.4%						
		Labrundinia sp.	0.4%						
		Coelotanypus sp.	0.4%						
		Synorthocladius sp.	0.4%						
		Cryptotendipes sp.	0.4%						

Table H -12

Summary of Diversity Indices and Abundance Data for Lower Hudson Stations

	D_s	*I*	D_{max}	E_s	Species Richness	Abundance per m^2	Biomass mg/m^2
Station 12 - Stockport Flats							
Mean	0.70	0.30	0.92	0.76	14	5289	63
Station 14 - Tivoli Bay							
Mean	0.82	0.18	0.95	0.86	16	4524	126
Station 15 - Esopus Meadows							
Mean	0.86	0.14	0.93	0.93	11	2551	65
Station 17 - Iona Island							
Mean	0.71	0.29	0.90	0.79	9	5136	365
Station 18 - Piermont Marsh							
Mean	0.84	0.16	0.90	0.93	9	6480	291
Grand Mean	0.79	0.21	0.92	0.85	12	4796	182

Notes:

D_s: Simpson Diversity

I: Simpson Dominance

D_{max}: Maximum possible diversity of D_s

E_s: Evenness of distribution of N individuals among s species

Table H -13

Lower Hudson River - Abiotic Parameters

			Station 12	Station 14	Station 15	Station 17	Station 18
Grain Size (average per fraction)							
	Medium sand		2.5%	1.2%	4.3%	10.0%	2.5%
	Fine sand		38.1%	13.3%	23.1%	10.4%	10.2%
	Silt		53.3%	73.1%	55.3%	79.5%	73.0%
	Clay		5.0%	8.7%	6.4%	9.3%	11.2%
TOC (average percent)			2.1%	2.5%	3.6%	3.5%	2.0%
Total PCBs ug/kg DW			2245	755	2077	3675	971

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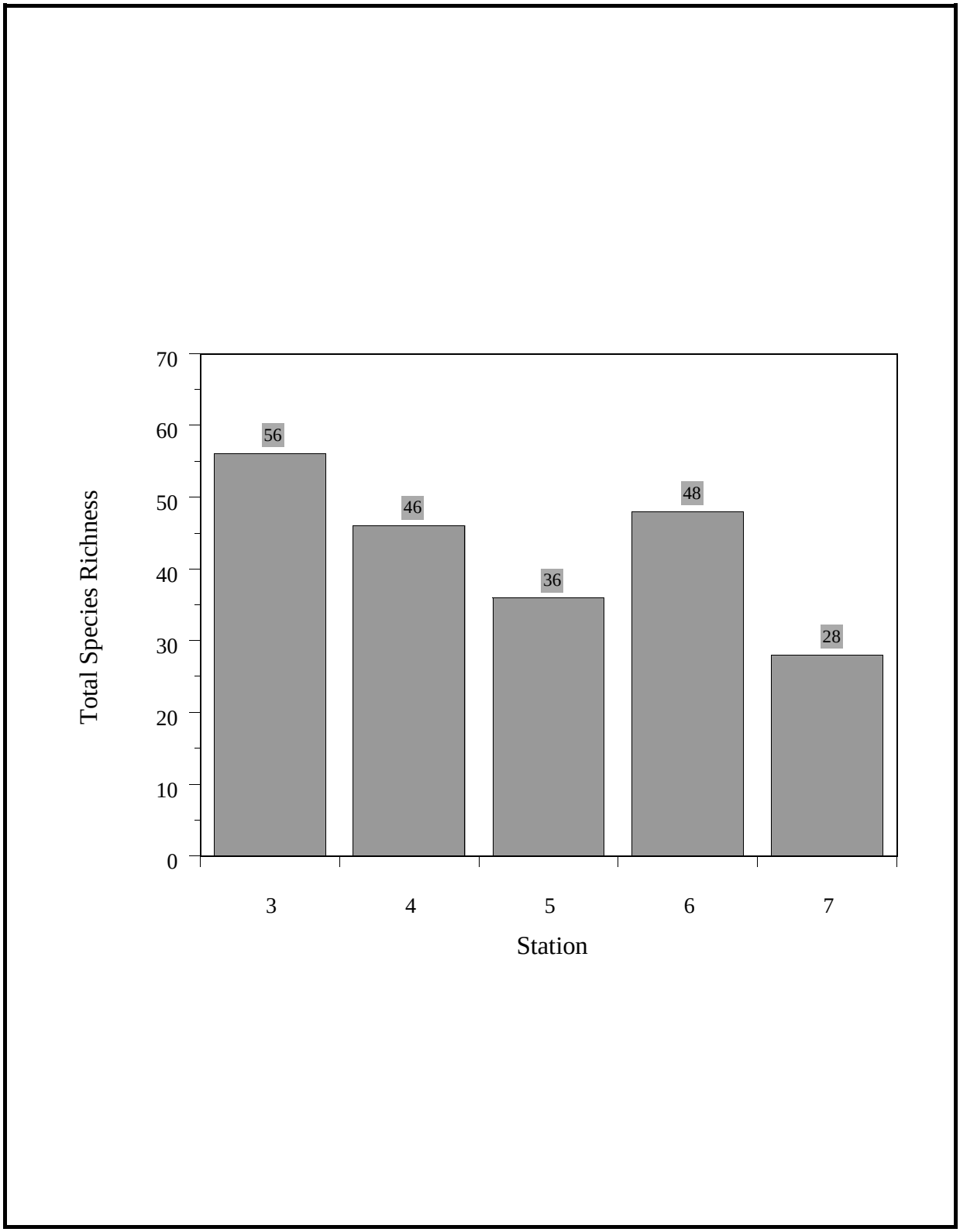
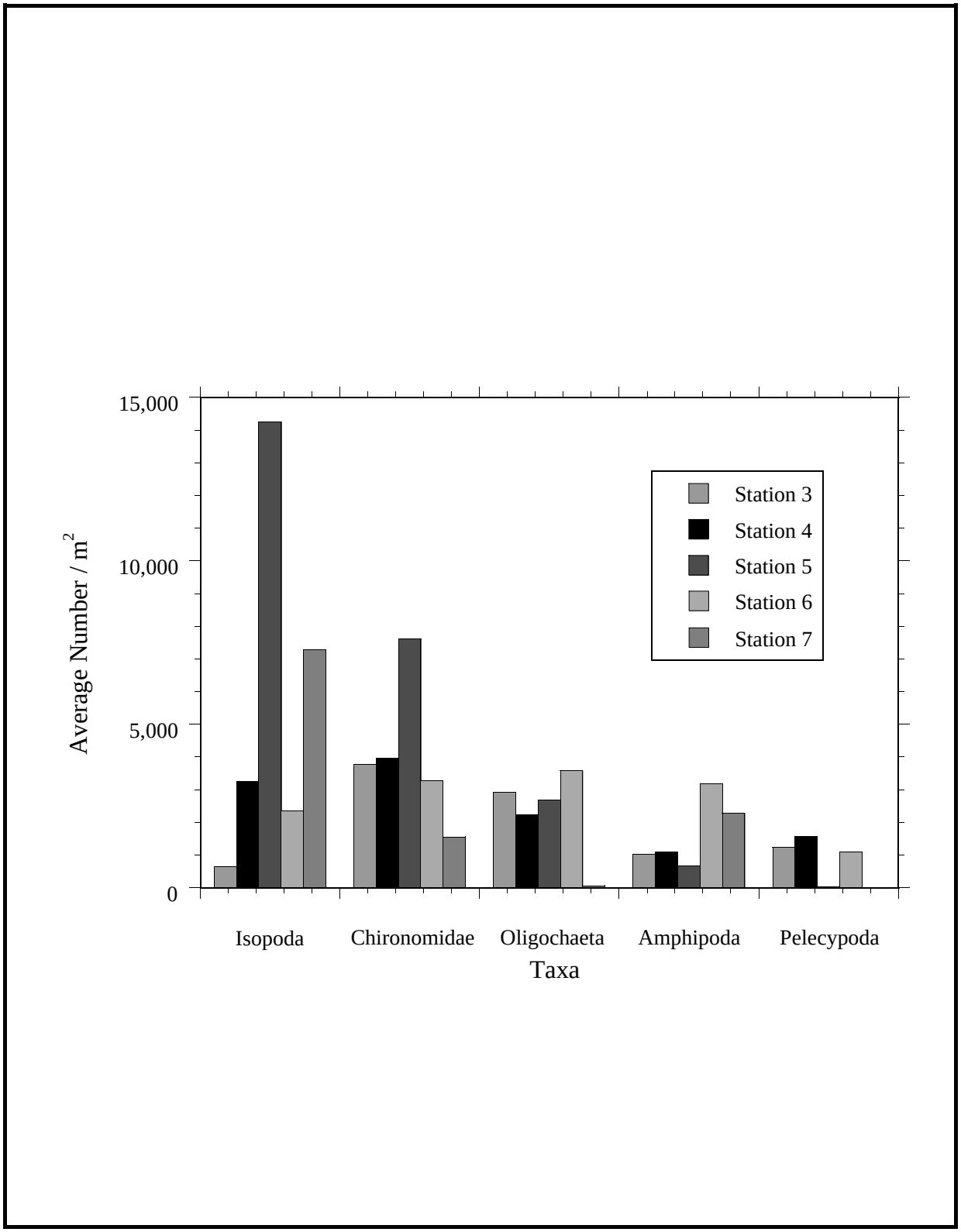


TAMS/MCA

Figure H-1
Total Species Richness at TI Pool Stations



TAMS/MCA

Figure H-2
Numerical Abundance of Dominant Taxa

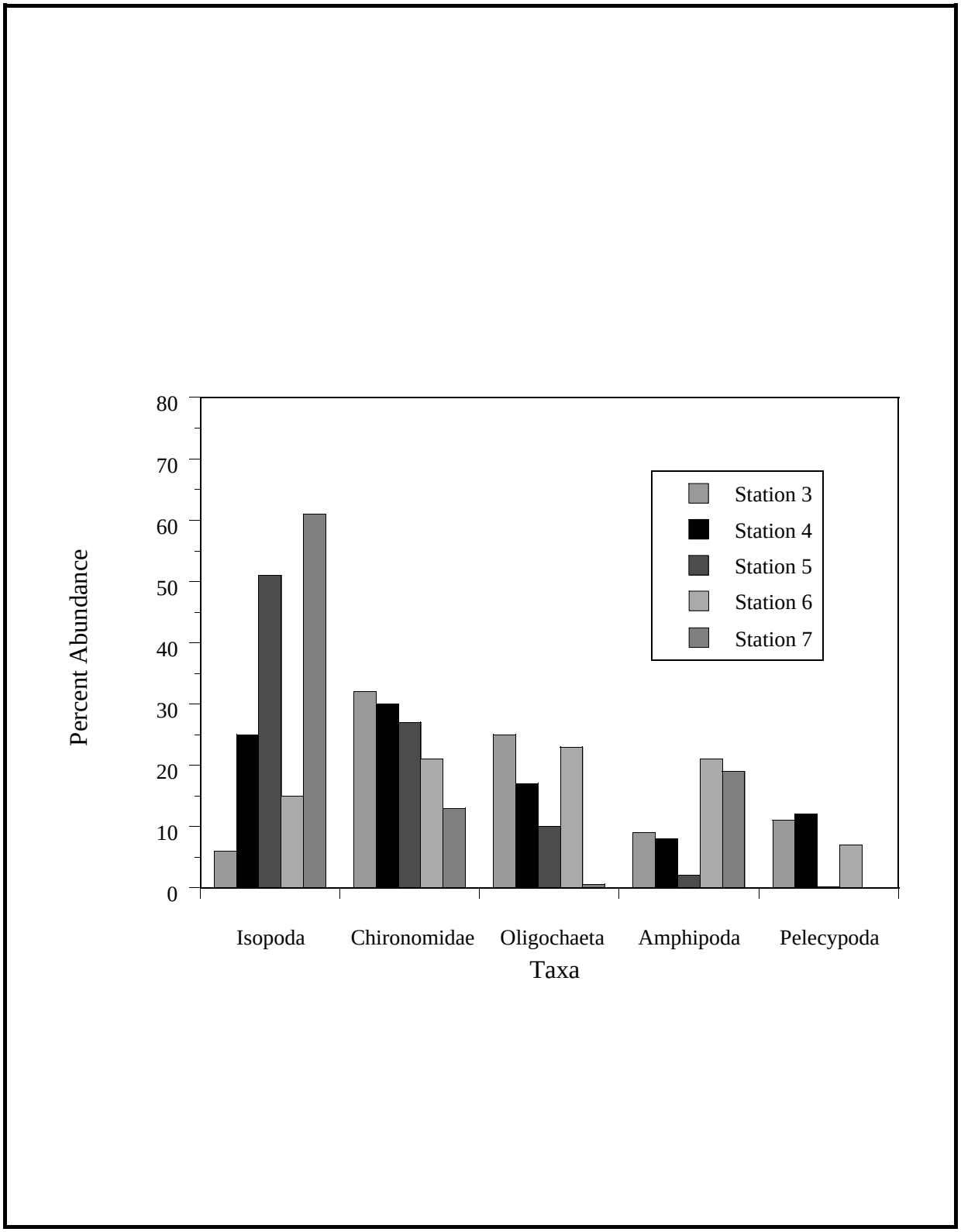
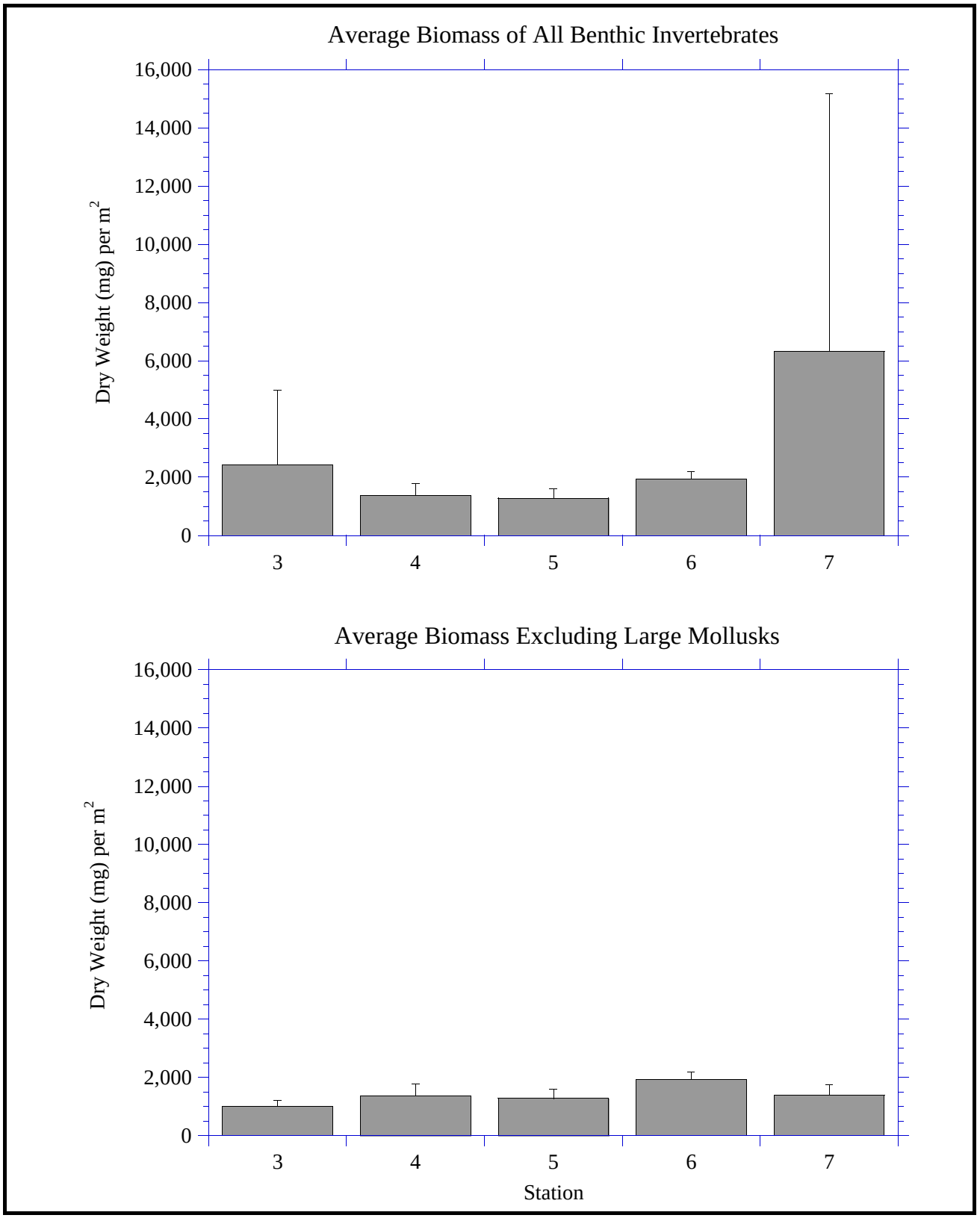
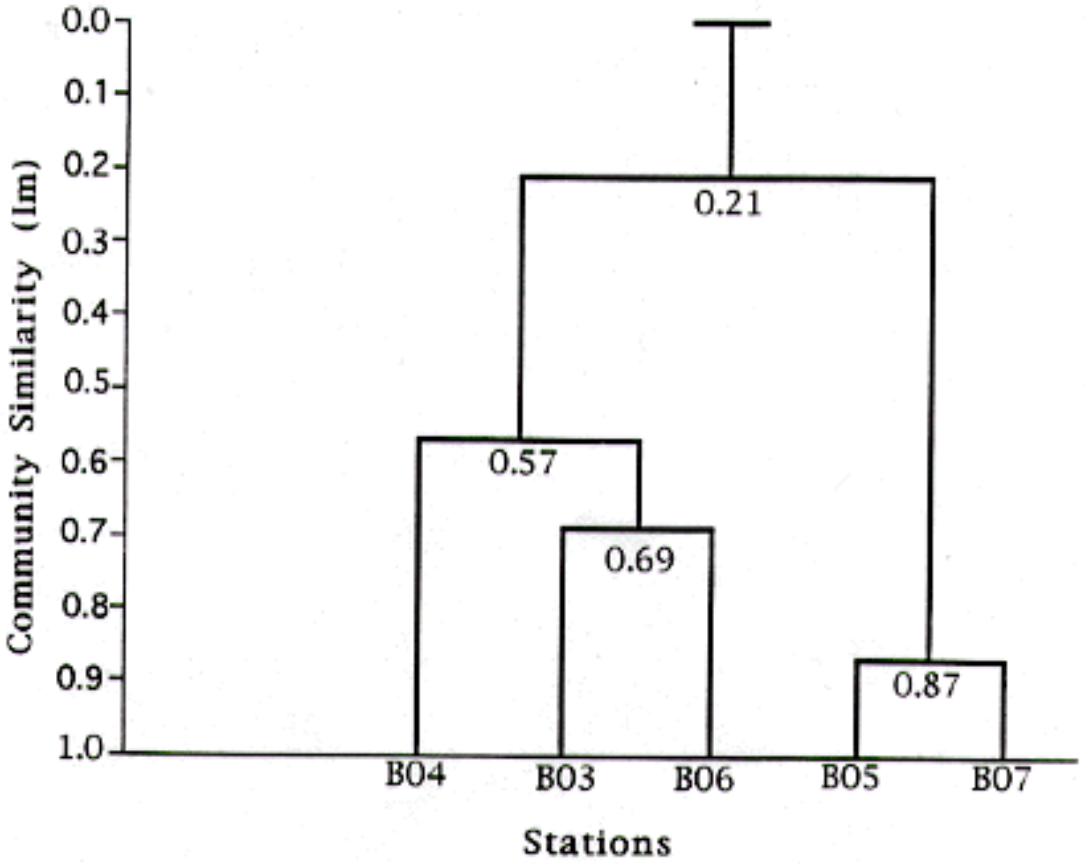


Figure H-3
Relative Percent Abundance of Numerically Dominant Taxa



TAMS/MCA

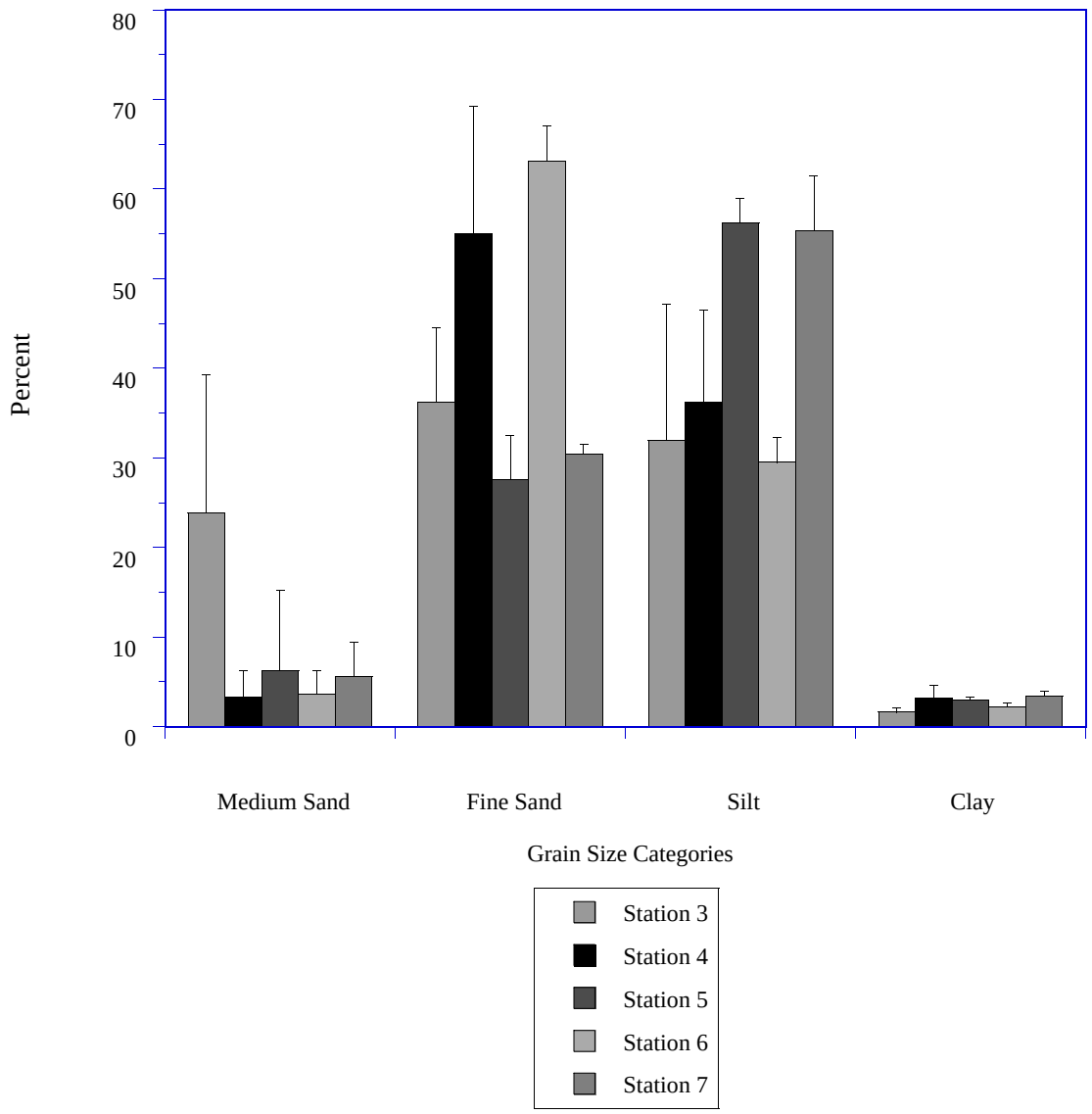
Figure H-4
Biomass of Benthic Invertebrates



Note: The dendrogram is based on Morista's Index (I_m) of community similarity and the computed fusion value of each junction is given.

TAMS/MCA

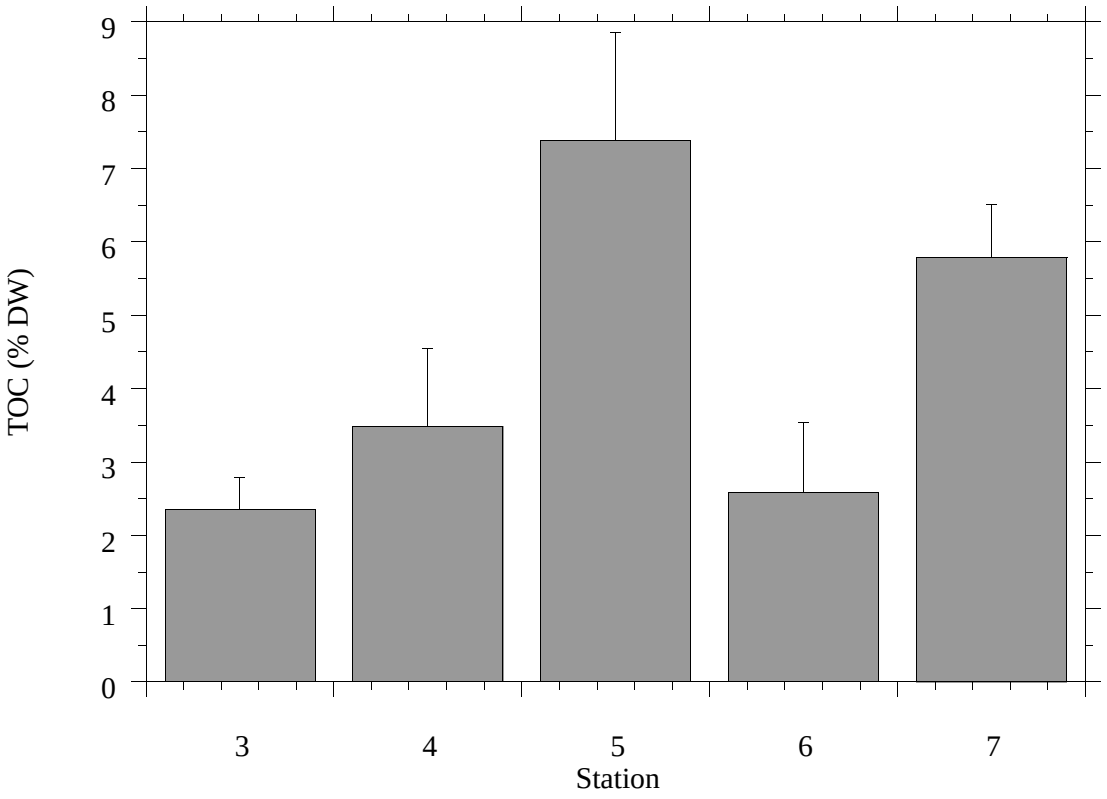
Figure H-5
Complete Linkage Clustering TI Pool Stations



Note: Error bars represent one standard deviation.

TAMS/MCA

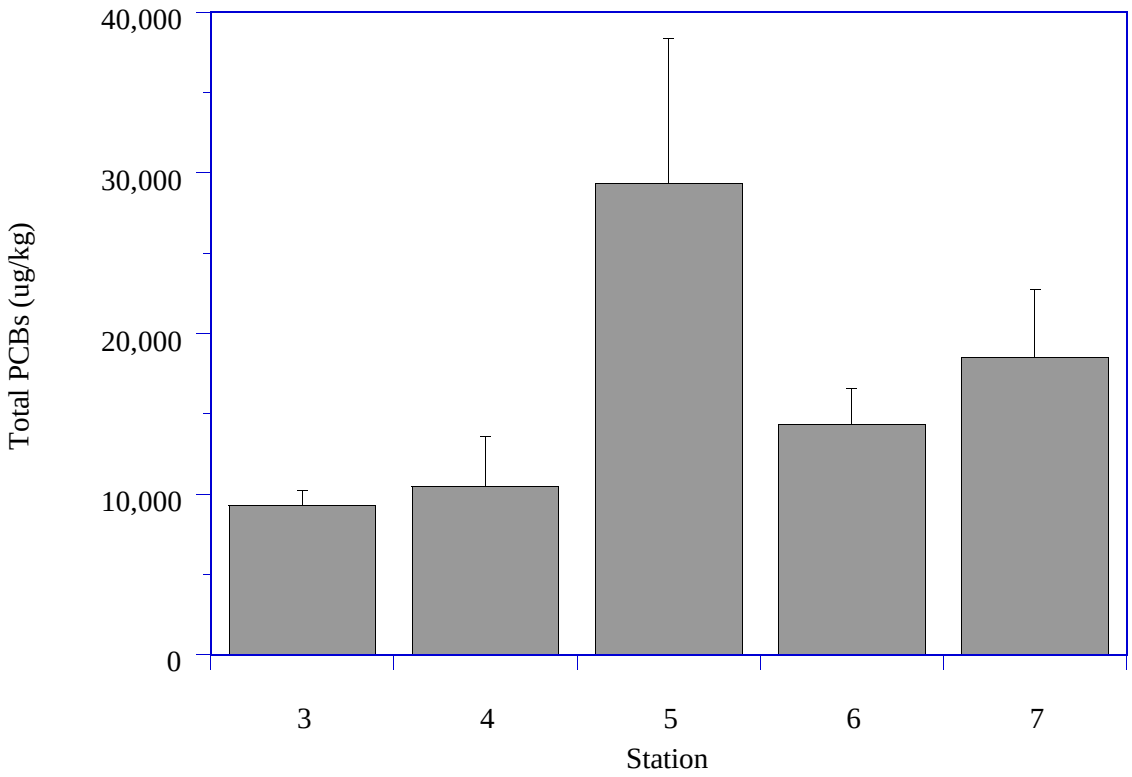
Figure H-6
Relative Percent Grain Size Classes in the TI Pool



Note: Error bars represent one standard deviation.

TAMS/MCA

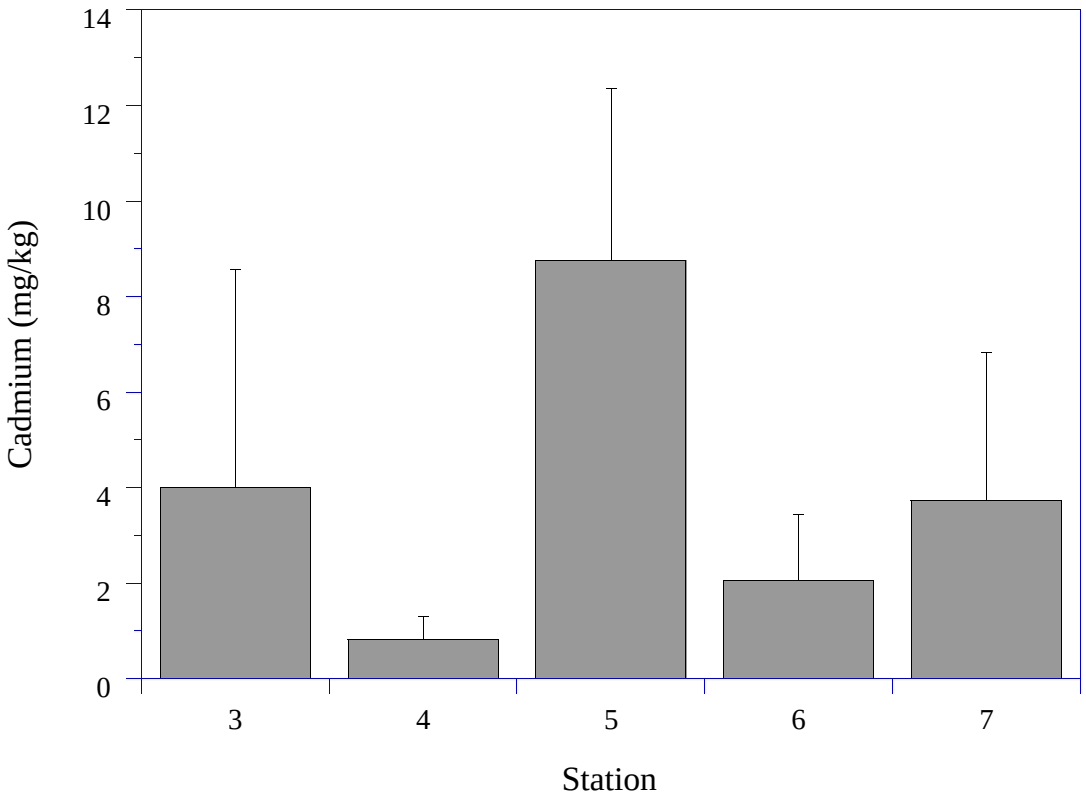
Figure H-7
Mean Sediment TOC at TI Pool Stations



Note: Error bars represent one standard deviation.

TAMS/MCA

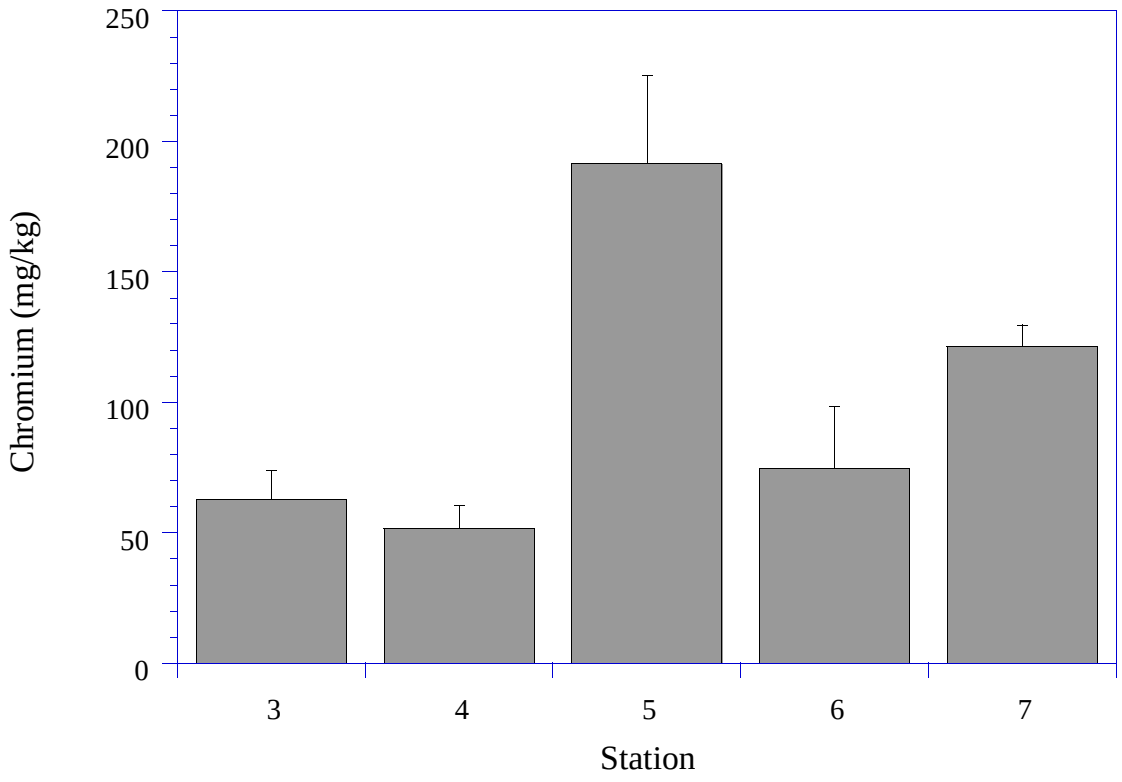
Figure H-8
Mean Total PCB Concentration in Sediments - TI Pool



Note: Error bars represent one standard deviation.

TAMS/MCA

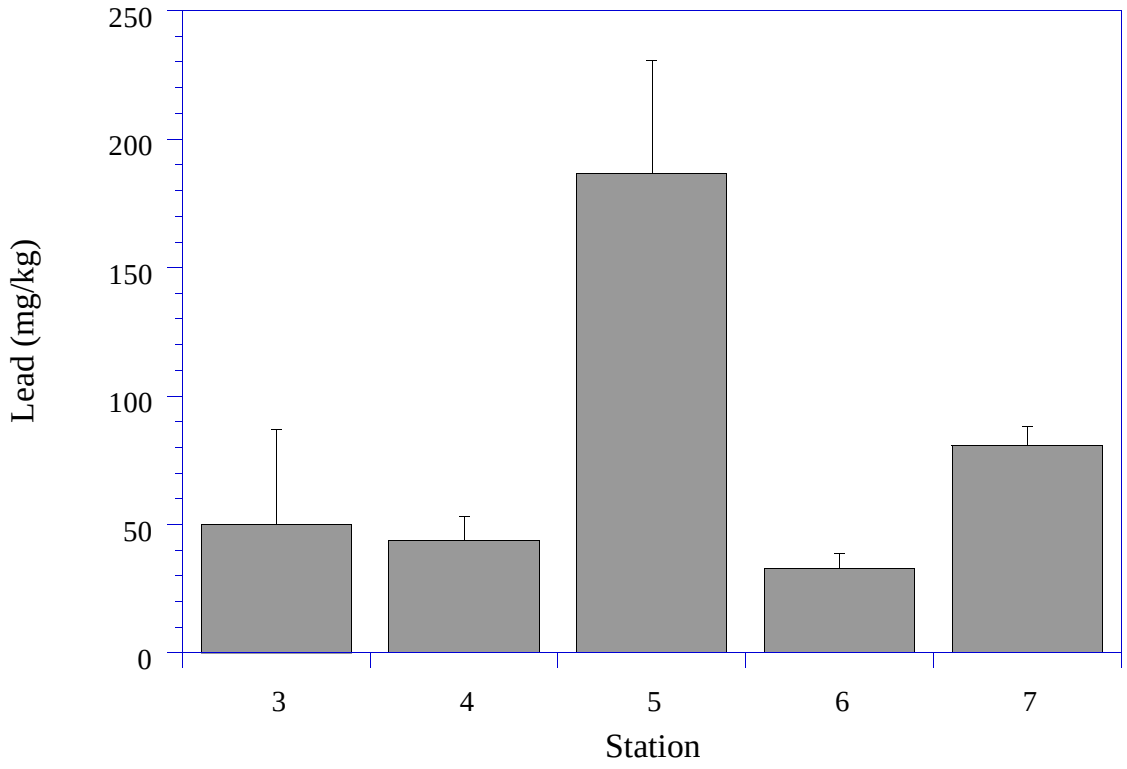
Figure H-9
Mean Cadmium Concentration at TI Pool Stations



Note: Error bars represent one standard deviation.

TAMS/MCA

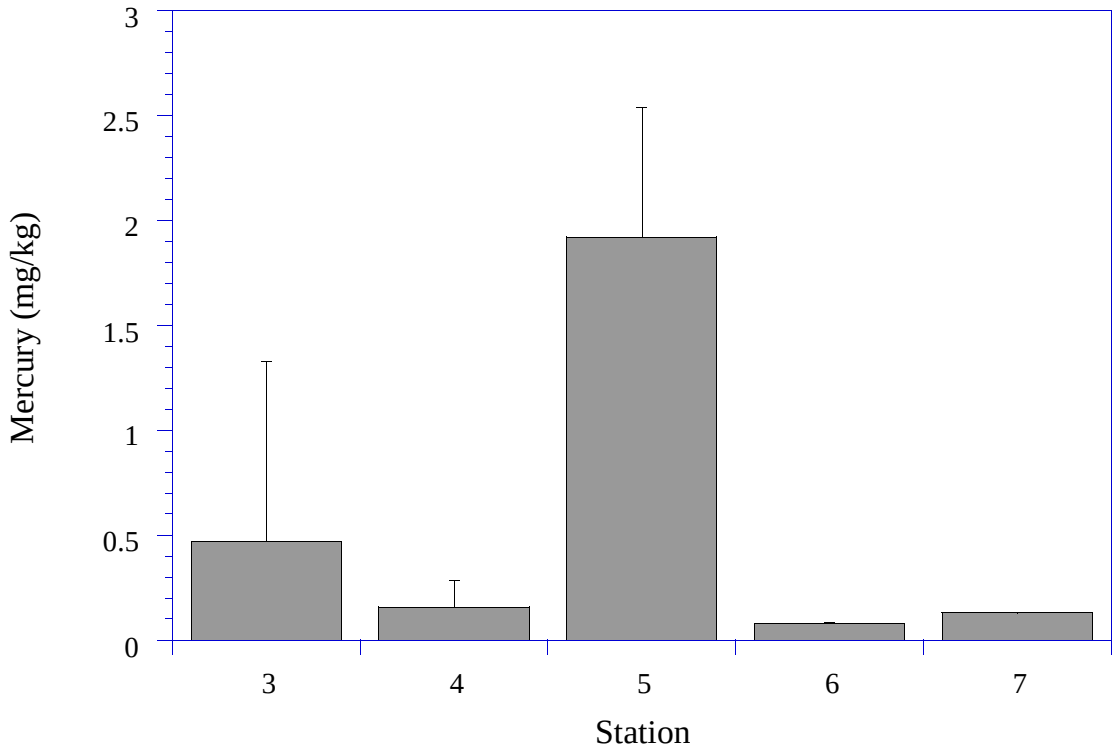
Figure H-10
Mean Chromium Concentration at TI Pool Stations



Note: Error bars represent one standard deviation.

TAMS/MCA

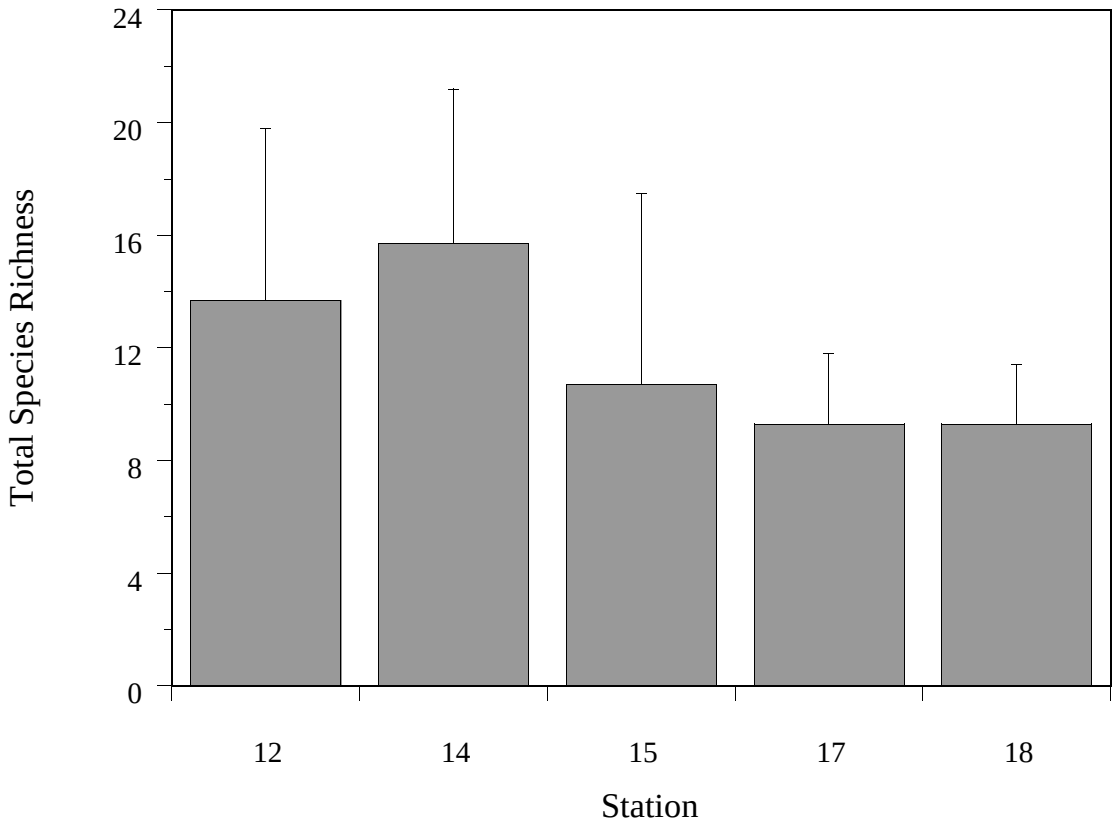
Figure H-11
Mean Lead Concentration at TI Pool Stations



Note: Error bars represent one standard deviation.

TAMS/MCA

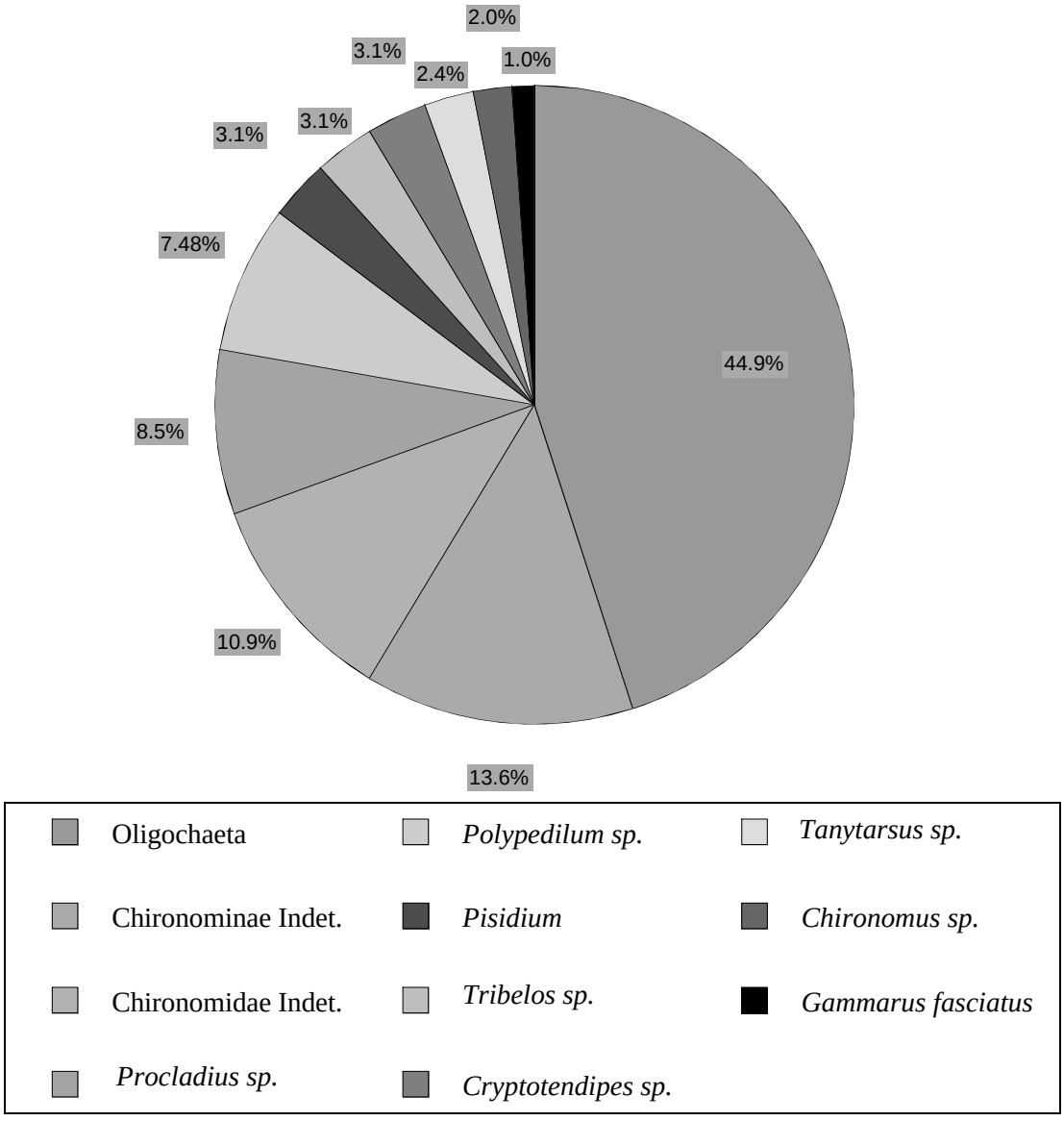
Figure H-12
Mean Mercury Concentration at TI Pool Stations



Note: Error bars represent one standard deviation.

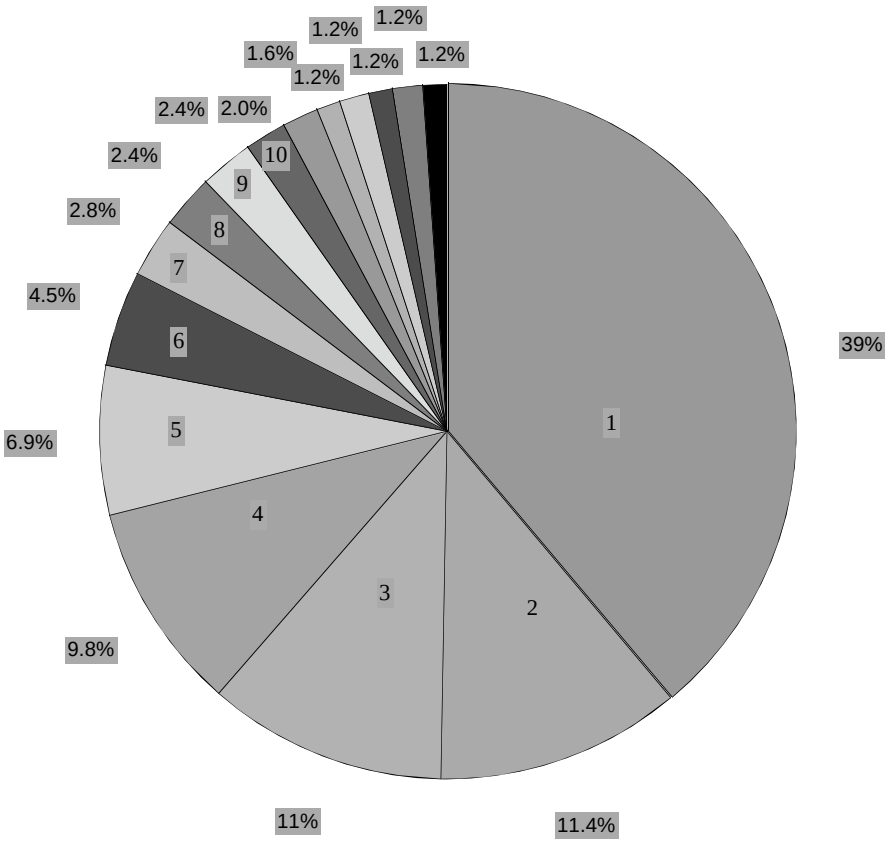
TAMS/MCA

Figure H-13
Mean Species Richness at Lower Hudson Stations



TAMS/MCA

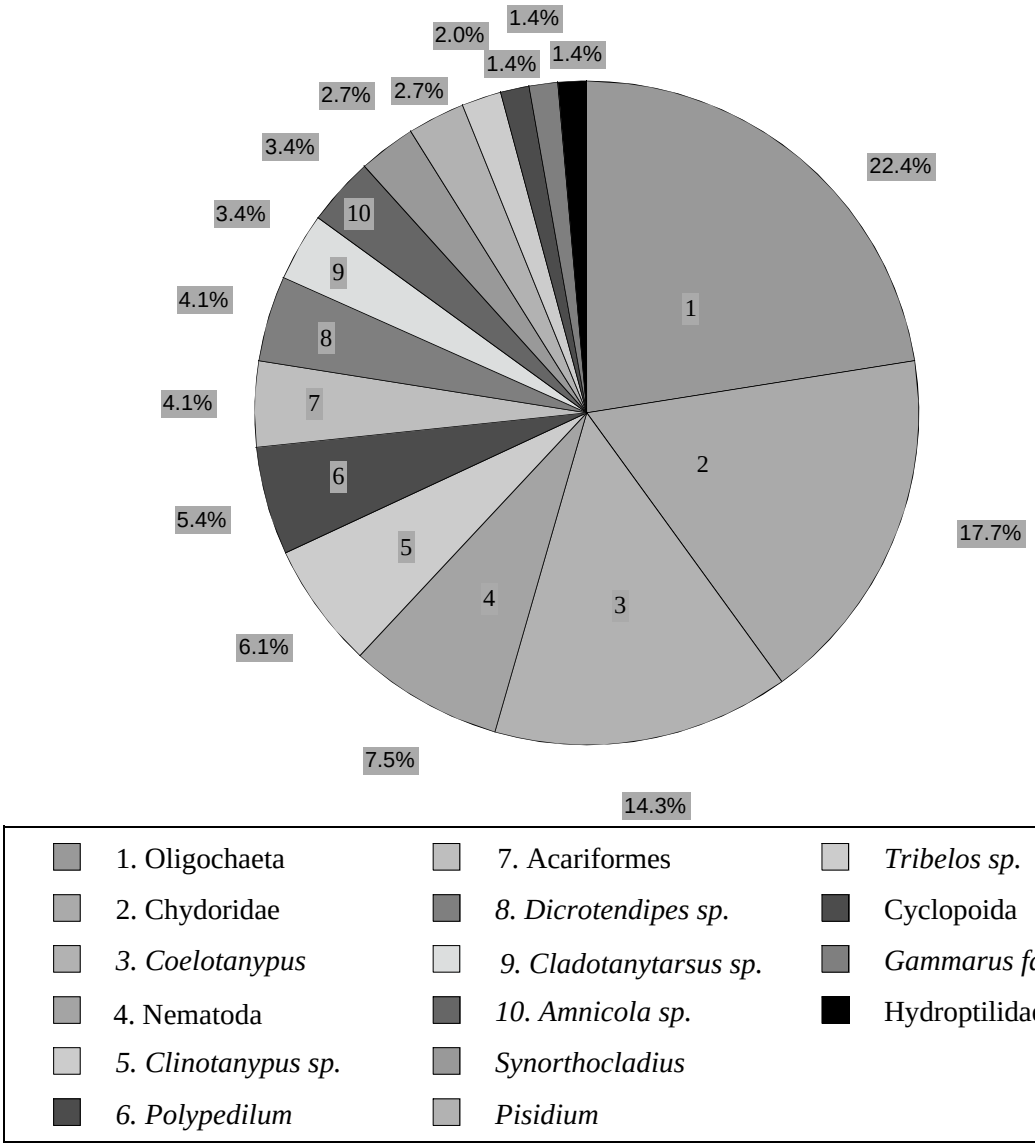
Figure H-14A
Relative Percent Abundance of Macroinvertebrates at Station 12



1. Chironominae Indet.	*7. Gammarus fasciatus*	Nematoda
2. Dicrotendipes sp.	*8. Pisidium*	Gastropoda
3. Procladius sp.	9. Chironomidae Indet.	*Cricotopus bicinctus*
4. Polypedilum sp.	*10. Amnicola limosa*	*Tanytarsus sp.*
5. Clinotanypus sp.	*Cladotanytarsus sp.*	
6. Oligochaeta	*Orthotrichia sp.*	

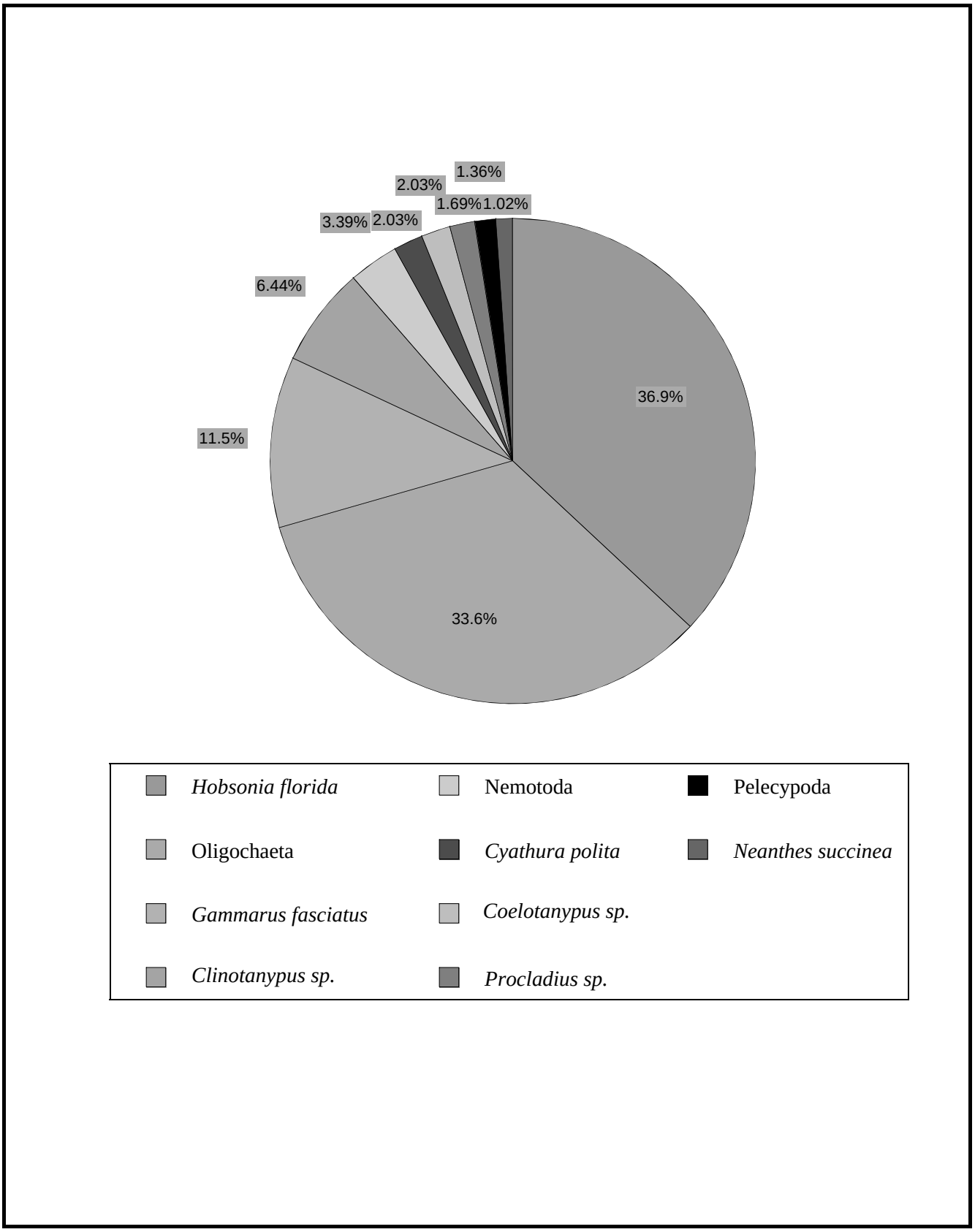
TAMS/MCA

Figure H-14B
Relative Percent Abundance of Macroinvertebrates at Station 14



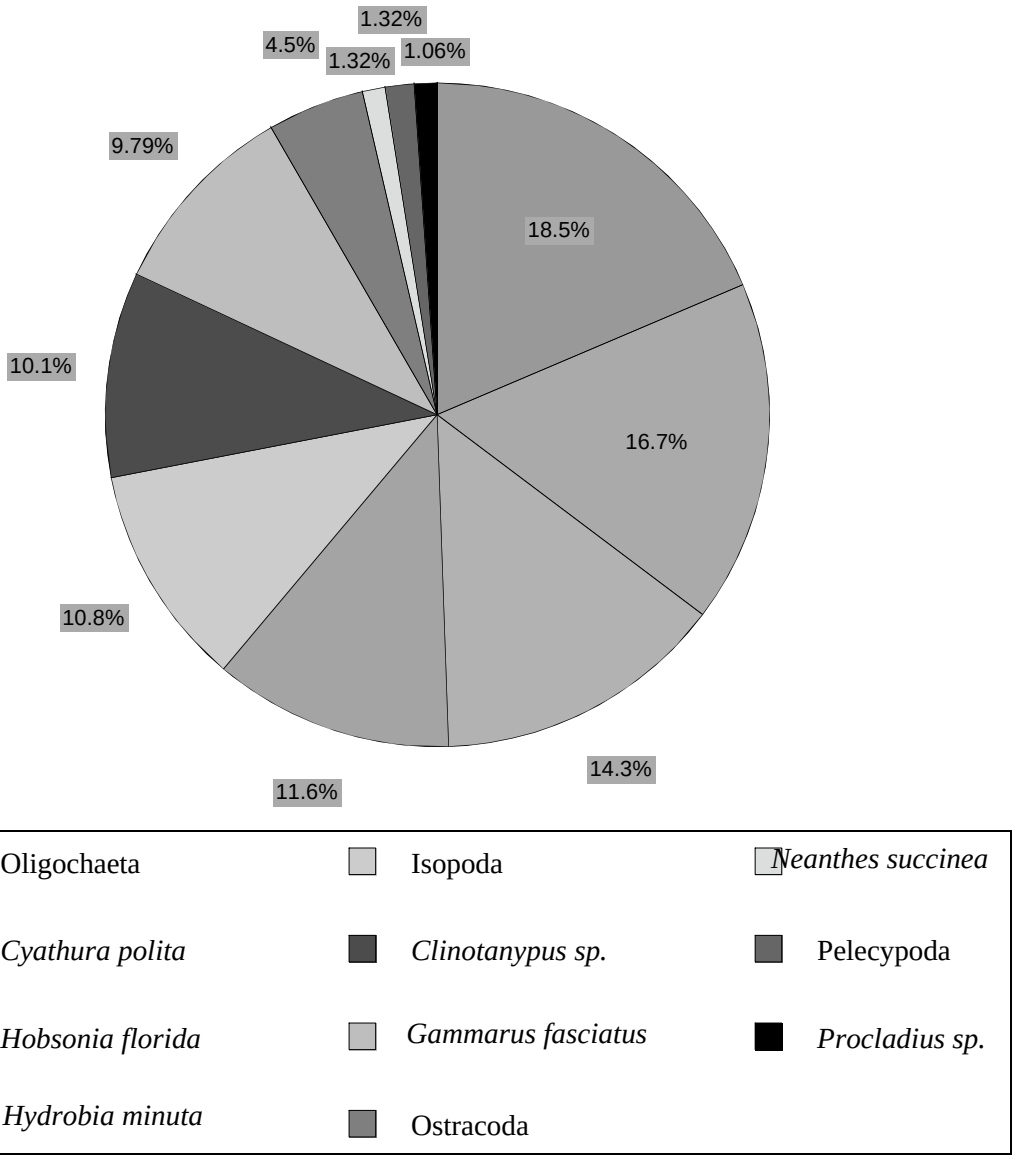
TAMS/MCA

Figure H-14C
Relative Percent Abundance of Macroinvertebrates at Station 15



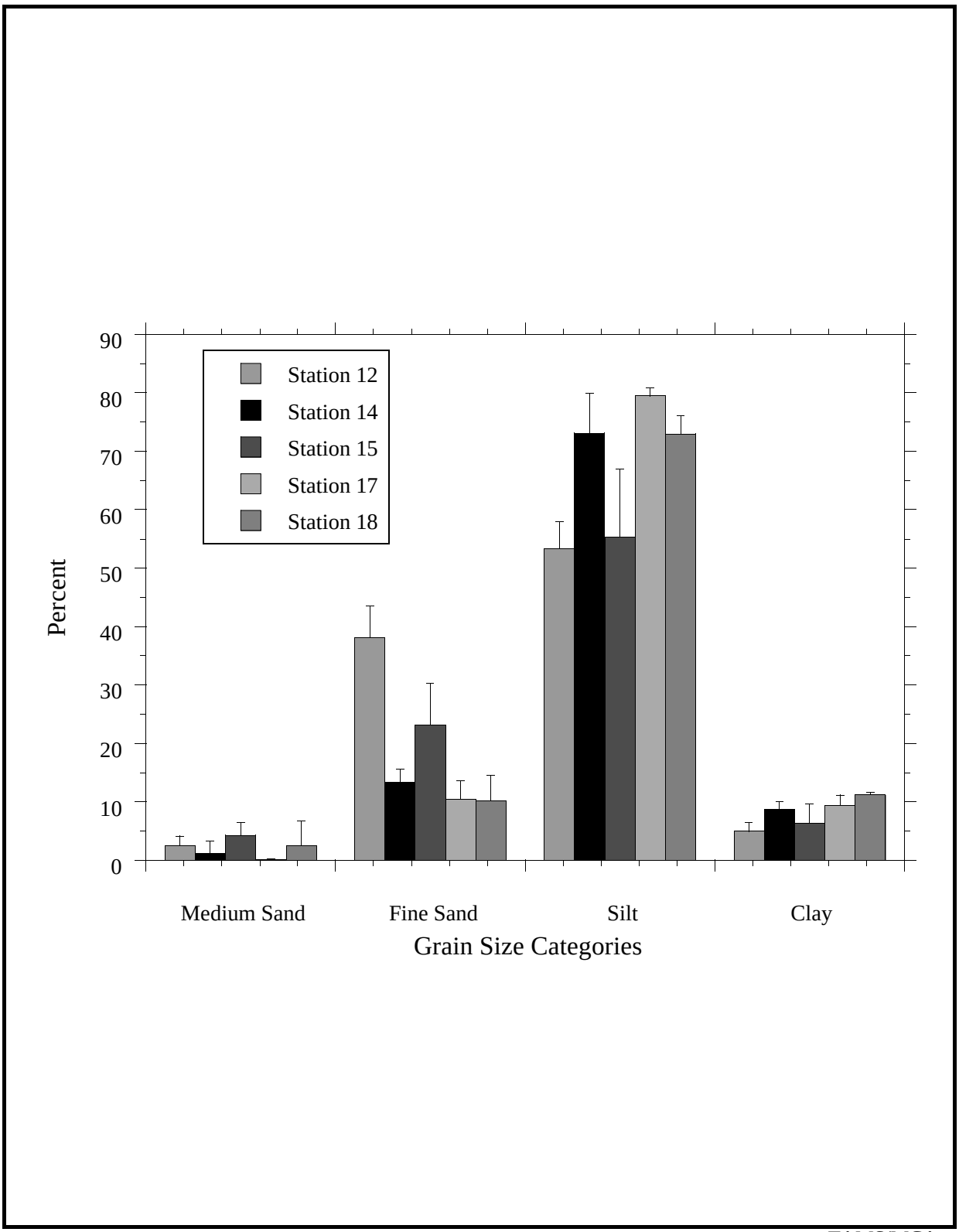
TAMS/MCA

Figure H-14D
Relative Percent Abundance of Macroinvertebrates at Station 17



TAMS/MCA

Figure H-14E
Relative Percent Abundance of Macroinvertebrates at Station 18



TAMS/MCA

Figure H-15
Relative Percent Grain Size Classes in Selected Lower Hudson Stations

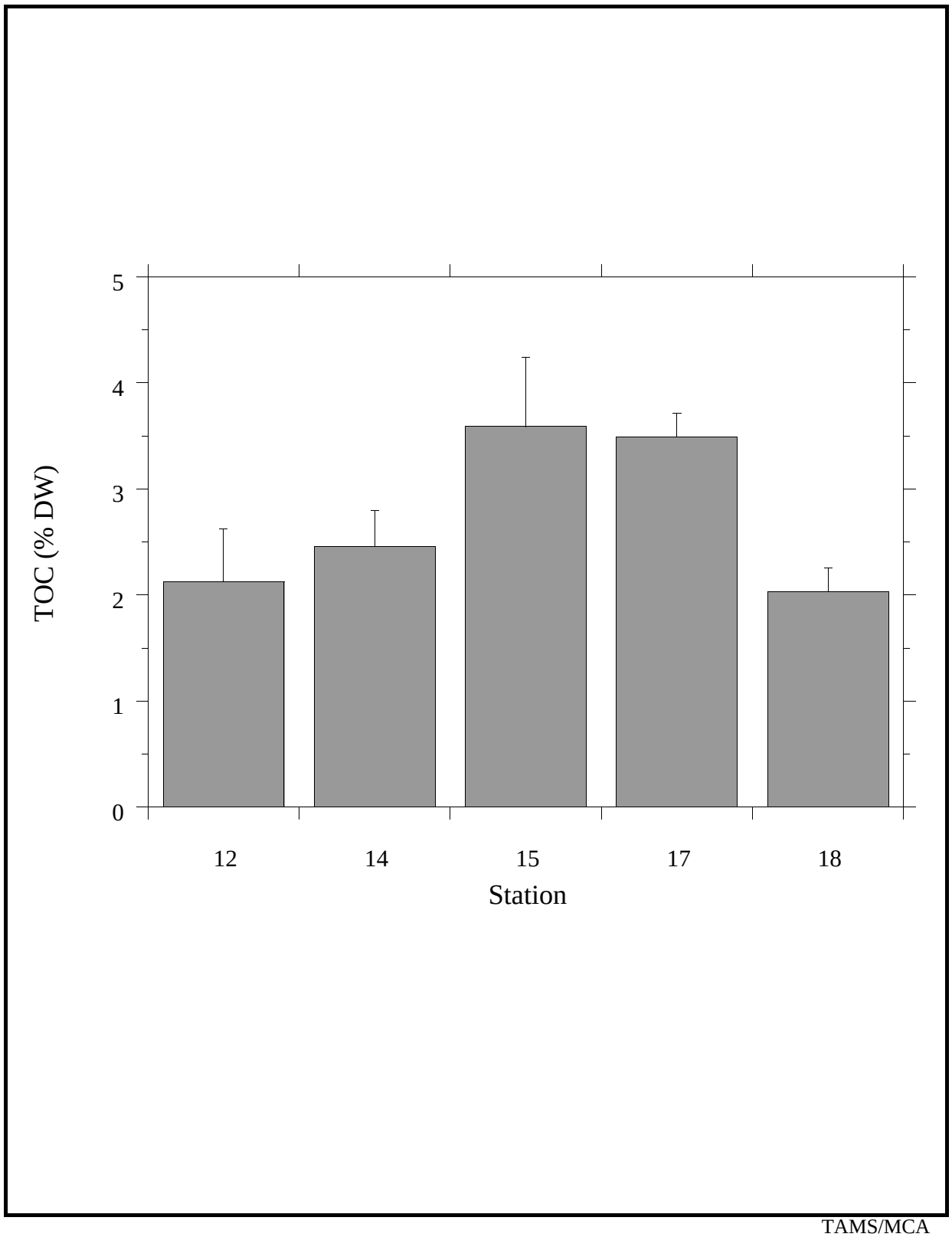
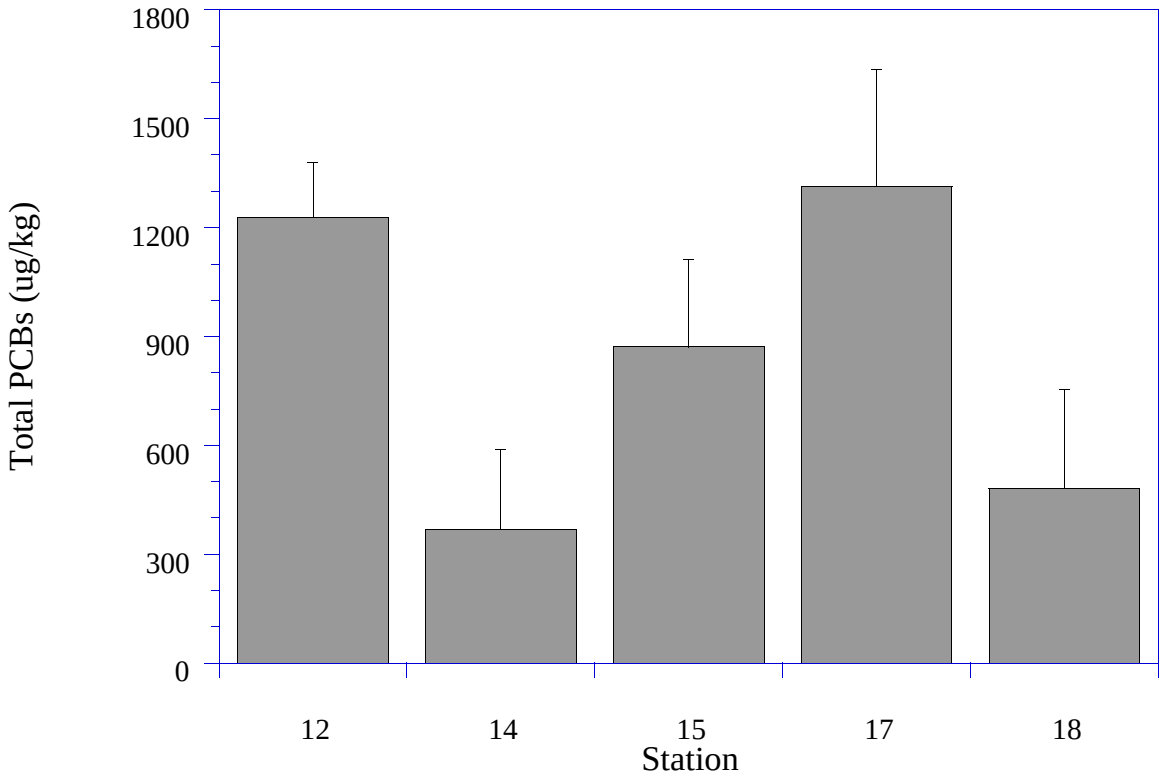


Figure H-16
Mean Sediment TOC at Selected Lower Hudson Stations



Note: Error bars represent one standard deviation.

TAMS/MCA

Figure H-17
Mean Total PCB Concentration in Sediments - Lower Hudson Stations

PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

Appendix I

DATA USABILITY REPORT FOR PCB CONGENERS ECOLOGICAL STUDY

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APPENDIX I
DATA USABILITY REPORT FOR PCB CONGENERS ECOLOGICAL STUDY

I.1 Introduction

The usability of data relates directly to the data quality objectives of the environmental investigation (Maney and Wait, 1991; USEPA, 1993a, 1994). The Hudson River PCB congener chemistry program required sophisticated, high resolution gas chromatography analyses with stringent quality control criteria. In addition, various inorganic and physical parameters were analyzed to define the chemical context within which the PCB congeners exist. This approach was necessary to delineate the concentration of PCB congeners within the context of geochemical and biological processes occurring in the river.

This report focuses on the usability of the PCB data generated by the Ecological Study, one of several studies including the High Resolution Sediment Coring Study and the Low Resolution Sediment Coring Study, that taken together constitute the overall Phase 2 program. The data usability assessment was done in a manner consistent with that used during the assessment of the PCB data generated during the High Resolution Sediment Coring Study.

A total of 90 PCB congeners were selected as target congeners based on their significance in environmental samples and the availability of calibration standards at the start of the overall program. For the ecological program one of these target congeners, BZ #192, was used as a surrogate compound rather than a target congener. In addition, Aquatec obtained qualitative and quantitative information for an additional 53 to 58 PCB congeners (non-target congeners) from each sample analysis using relative retention time information detailed in the literature, and more recently verified with actual standards. During the ecological study, Aquatec calibrated for eighteen of these non-target congeners on a daily basis. Certain target congeners are of particular importance in evaluating geochemical and biological processes within the Hudson River sediments. These are the 12 "principal" target congeners, which consist of BZ #1, 4, 8, 10, 18, 19, 28, 52, 101, 118, 138, and 180. The focus of this report will be on the usability of the analytical data for these 12 principal congeners.

This report serves as an overall evaluation of the PCB congener analyses performed for the Hudson River ecological study. The evaluation is based on the assessment of data quality relative to the objectives of the study. The report first provides a synopsis and assessment of the field sampling, analytical chemistry and data validation programs, and then evaluates data usability for all congeners analyzed, with particular emphasis on the 12 principal target congeners and an evaluation of the toxic equivalency factor congeners.

I.2 Field Sampling Program

The ecological study was designed to provide site specific data for the ecological assessment, to examine the PCB concentrations in sediments along the Hudson River in ecologically significant areas, and to estimate the potential adverse ecological effects from exposure to present levels of PCBs. The ecological collection program, sampling procedures, analytical protocols, and quality control/quality assurance requirements are described in Volume 2 of the "Phase 2B Sampling and Analysis Plan/Quality Assurance Project Plan - Hudson River PCB Reassessment RI/FS" (USEPA, 1993b, referred to in this report as the Phase 2B SAP/QAPP). Samples were collected from the Thompson Island (TI) Pool and at Upper and Lower Hudson River locations using an Ekman Grab Sampler, with two grab samples collected at each location and one additional grab sample collected at the TI Pool locations. The sample design consisted of collecting replicate samples within a 10 meter by 10 meter grid at each of the sampling locations. Once collected, the field sampling team sieved the grab sample designated for PCB congener analysis to separate out the benthic invertebrates which underwent PCB congener, biomass, and percent lipid determinations. The field sampling team sub-sampled one of the grab samples for total organic carbon (TOC), target analyte list (TAL) metals, grain size, total carbon/total nitrogen (TC/TN), and total inorganic carbon (TIC) analyses. The field sampling team collected an additional grab sample at the TI Pool locations to provide benthic macroinvertebrate for abundance and diversity determination. One sample of epibenthic organisms were also collected at each station location by dragging the river bottom with a net; these samples were analyzed for PCB congeners, biomass, and percent lipids.

Scientists from TAMS (contracted by the US Environmental Protection Agency [EPA]), the New York State Department of Environmental Conservation (NYSDEC), and the National Oceanic and Atmospheric Administration (NOAA) performed sampling for the ecological program from August 3 to September 1, 1993. TAMS scientists collected a total of 93 sediment and 83 invertebrate samples from 19 station locations. In addition, NYSDEC and NOAA scientists collected at total of 120 fish samples funded by EPA and 115 fish sample funded by NOAA at TI Pool and at Upper and Lower Hudson River locations. The laboratory allocated these sediment, invertebrate, and fish samples into a total of 27 sample delivery groups (SDGs). The Program Quality Assurance Officer (QAO) conducted a field sampling audit on August 11, 1993 to assess compliance of the sampling procedures with the Phase 2B SAP/QAPP. The audit findings indicate that the sampling program was being conducted in a technically acceptable manner consistent with the Phase 2B SAP/QAPP (Wait, 1993). St. John's University in Jamaica, New York received the invertebrates for abundance and diversity analyses.

I.3 Analytical Chemistry Program

I.3.1 Laboratory Selection and Oversight

EPA retained a number of analytical laboratories to perform the analyses required for this program. To verify that the selected laboratories had the capacity, capabilities, and expertise to

perform sample analyses in strict accordance with the specified methodologies, each qualifying laboratory underwent an extensive audit by TAMS/Gradient's senior chemists. EPA retained the following laboratories to perform ecological sample analyses for the Hudson River RI/FS program: Aquatec Laboratories, a division of Inchcape Testing Service located in Colchester, Vermont; St. John's University in Jamaica, New York; and Ohio State University. Aquatec was the sole analytical laboratory which conducted the PCB congener analyses for the entire program.

Routine laboratory audits were conducted during the ecological study to verify compliance of the laboratories contracted by EPA (Aquatec and St. Johns) with the Phase 2B SAP/QAPP requirements.

Unique requirements of the PCB congener method necessitated refinements of previously published methods. In conjunction with these changes, Aquatec conducted Method Detection Limit (MDL) studies for the sediments, invertebrate, and fish and Extraction Efficiency (EE) studies for the sediments and fish to evaluate the adequacy of the methods. To conduct these studies, seven replicate Hudson River sediment samples were collected . For the sediment MDL study, samples were collected upstream from the zone of major PCB contamination, while for the sediment EE study samples were collected from within the zone of major PCB contamination. For the fish MDL study, Aquatec obtained seven replicate fish tissue samples from Fernwood Trout Hatchery. Ron Sloan of the NYSDEC provided Aquatec with four replicate fish tissue samples that contained significant level of PCBs for the EE study. Aquatec performed an MDL study using seven replicate mussel tissue samples to represent invertebrates. Aquatec could not produce usable EE study results on the mussel tissue samples because the initial PCB concentrations in the samples were too low. Synopses of these MDL/EE studies are provided in TAMS/Gradient memoranda dated July 12, 1993 and December 29, 1993 (Cook, 1993a and 1993b). The TAMS/Gradient Program QAO oversaw and approved the method refinements through out the process.

I.3.2 Analytical Protocols for PCB Congeners

The method used for the determination of PCB congeners in Phase 2B is a program-specific method based on NYSDEC's Analytical Services Protocol Method 91-11 (NYSDEC, 1989) for PCB congeners. Appendix A4 of the Phase 2B SAP/QAPP describes procedures for the calibration, analysis, and quantitation of PCB congeners by fused silica capillary column gas chromatography with electron capture detection (GC/ECD). The method is applicable to samples containing PCBs as single congeners or as complex mixtures, such as commercial Aroclors. Aquatec extracted sediment and tissue samples with hexane, and performed applicable cleanup procedures prior to analysis by GC/ECD, as detailed in Appendix A3 and A9 of the Phase 2B SAP/QAPP. Aquatec analyzed hexane extracts for PCB congeners on a dual capillary-column GC/ECD, as detailed in Appendix A4 of the Phase 2B SAP/QAPP and identified PCB congeners using comparative retention times on two independent capillary columns of different polarity. Aquatec used calibration standards for each target congener to define retention times. In addition, Aquatec routinely analyzed Aroclor standards and mixtures of Aroclor standards to verify identification and quantification of the primary calibration

standards. Due to the non-linear nature of the ECD over any significant calibration range (for this project 1 to 100 ppb in extract), Aquatec generated the calibration curves used for quantitation from a quadratic-weighted least-squares regression model where the correlation coefficient is greater than 0.99 (McCarty and Lesnik, 1995; USEPA, 1986 - Method 8000B, proposed 1995 update). For each PCB congener which elutes as a single congener on each GC column, Aquatec reported the result as the lower of the two values. Although this quantification scheme is compliant with EPA CLP guidelines for dual-column analyses (USEPA, 1991), it may introduce a slight low bias when calculating homologue and total PCB sums. TAMS/Gradient compared data in the database relative to absolute results on both columns and found the bias was usually negligible, and on a worst-case basis, may be 2% to 10% low. For situations where coelution occurred on one column, Aquatec quantitated the result from the column not displaying coelution. If only coelution results were available, Aquatec performed a calculation to decipher concentrations using response factors derived by Mullen (1984). The 12 principal congeners, BZ #1, 4, 8, 10, 18, 19, 28, 52, 118, 138, and 180 eluted as either a single congener peak on both GC columns or a single congener peak on one column and coeluted on the other column. BZ #101 coeluted on both columns, and therefore, was always reported with BZ #90.

Approximately 10% of all samples analyzed by GC/ECD also underwent additional analysis using a GC-ion trap detector (ITD) as an additional means of confirming PCB congener identifications, as detailed in Appendix A5 of the Phase 2B SAP/QAPP. When possible, Aquatec selected samples with the highest concentrations of PCB congeners for confirmation analysis by GC/ITD. Usually, Aquatec performed two GC/ITD analyses per SDG, even if congener concentrations were minimal throughout the SDG.

At the start of the Phase 2B sampling and analysis program, TAMS/Gradient and Aquatec selected 90 target PCB congeners. These target congeners are listed in Table I-1 and identified by BZ number (Ballschmiter and Zell, 1980). The selection of these 90 PCB congeners was based on their significance in environmental samples and the commercial availability of calibration standards. PCB congeners for which calibration standards were available are referred to as "target congeners". To verify that congener response for these calibration standards were reproducible over time, TAMS/Gradient examined calibration data from November 1992 and October 1993 and found temporal consistency to be acceptable on both GC columns (Bonvell, 1994a).

The high resolution column chromatography techniques employed by Aquatec produced acceptable PCB resolution for numerous congeners not contained in the target congener calibration standards. Thus, TAMS/Gradient decided during method refinement to report approximately 50 additional PCB congeners. The laboratory identified these additional PCB congeners based upon the relative retention times reported in the published literature (Mullen, 1984; Schulz, 1989; Fischer and Ballschmiter, 1988, 1989). Aquatec calibrated these additional "non-target" congeners using the calibration curve for target congener BZ #52. Aquatec chose BZ #52 because it elutes as a single congener peak in the middle region of the chromatogram for both GC columns and is a major component of Aroclor 1242, the Aroclor anticipated in Hudson River samples. Using additional

congener calibration standards which became commercially available by August 1993, Aquatec performed analyses to verify and refine the historical relative retention times, and to determine individual congener calibration parameters. These analyses confirmed a majority (36) of the historical non-target congener relative retention times. For all analyses performed prior to August 1993, the results for 14 non-target compounds were not confirmed by this analysis, TAMS/Gradient considered these results unusable and deleted them from the database. A review of the high resolution sediment coring data indicated that the 36 confirmed non-target congeners represent a significant percentage, up to 25 percent, of the total PCB mass. Therefore, it was decided to include the non-target congener results to calculate homologue and total PCB masses in the Hudson River. If these non-target congener results were deleted, the resulting calculations for homologue and total PCBs would have been significantly biased low. Thus, 36 non-target congeners are included in this report, as shown in Table I-1. Since the non-target congener results were to be included in the calculations of homologue and total PCB mass, TAMS/Gradient applied an individual correction factor to each congener's results based on the analysis of the additional congener standards. The application of these correction factors served to minimize the uncertainty associated with quantitation of non-target congeners. A series of memoranda describe the method for deriving these calibration correction factors (Bonvell, 1993a,b,c) and a listing of the derived calibration correction factors is provided in Bonvell (1994b).

To establish a method of quantitating total Aroclor concentrations from PCB congener data, Aquatec performed duplicate analyses of seven Aroclor standards (1016, 1221, 1232, 1242, 1248, 1254, 1260). The quantitation of an Aroclor for this program was defined as the sum of all congeners present in the standard Aroclor mixture at a concentration greater that 0.1% of the total Aroclor mass. In this manner, the percentage of the total mass represented by the detected target and non-target congeners greater than 0.1% of the Aroclor mass was then compared to the actual concentrations of each Aroclor standard. The results produced the following mass yields for the seven Aroclor standards: Aroclor 1016=93.3%, Aroclor 1221=86.8%, Aroclor 1232=91.0%, Aroclor 1242=90.6%, Aroclor 1248=89.2%, Aroclor 1254=95.8%, and Aroclor 1260=87.0%. Thus, in each case, the 90 target and 36 non-target congeners represented more than 87% of the original Aroclor mass. For those Aroclors most important to the Hudson River based on General Electric's reported usage (Brown et al., 1984) these congeners represented better than 90% of the Aroclor mass (i.e., Aroclors 1242, 1254, and 1016).

I.4 Data Validation

An essential aspect of understanding the uncertainties of the Phase 2 ecological data is understanding the significance of the qualifiers associated with the results. Each result has an associated qualifier. Qualifiers denote certain limitations or conditions that apply to the associated result. Initially, the analytical laboratories applied qualifiers to the results, and then the data validators modified the qualifiers, as necessary, based on the established validation protocols. Data reporting and validation qualifiers direct the data users concerning the use of each analytical result. Two sets of qualifiers were used in the database, one set for PCB congener data, and a second set for non-PCB

chemical and physical data. Aquatec developed an extensive list of data reporting qualifiers to be applied to the PCB congener data. The list is based on standard EPA qualifiers used for organic analyses, with additional qualifiers provided to note unique issues concerning PCB congener analysis (e.g., the quantitation scheme). The data reporting qualifiers for PCB congener data, as applied by Aquatec, are defined in detail in Table I-2.

During validation, the validators made modifications to the data qualifiers which are reflected in the database. CDM Federal Programs Corporation and their subcontractors, under a separate EPA contract, performed data validation for the ecological study. Validation procedures employed by CDM for GC/ECD analyses are detailed in Appendix A6 of the Phase 2B SAP/QAPP, and validation guidelines for GC/ITD analyses are provided in Appendix A7 of the Phase 2B SAP/QAPP. TAMS/Gradient devised the validation procedures to reflect the data quality objectives of the program, as well as to conform with EPA (1988, 1992a) standards as appropriate. USEPA Region II concurred with these method-specific validation protocols. In addition, TAMS/Gradient designed comprehensive data validation templates to facilitate consistency of approach and actions during validation. Prior to validation of the PCB data, Gradient conducted a training workshop to aid CDM in properly performing the validation. Gradient reviewed and commented on the initial CDM validation reports and provided real-time QA oversight. USEPA Region II (Lockheed ESAT) revalidated data for an earlier phase of the program to ensure that CDM had performed the validations properly. Lockheed ESAT noted no significant problems.

The initial data validation efforts for the ecological samples were completed between January 1994 and April 1995. The results were subsequently incorporated into the EPA Phase 2 database and available for review in March 1996. In April 1995, it became clear that the validation results differed markedly but randomly for the nonvalidated data for the high resolution core samples. Upon further investigation, the source of some of these differences was the result of incorrect data validation procedures largely pertaining to blank corrections. Specifically, it was found that blank samples were sometimes incorrectly associated with environmental samples and blank values were transcribed incorrectly among validation records, among other concerns. The same incorrect data validation procedures that were applied to the high resolution core samples also were applied to the ecological study and the low resolution core study. These problems were found to be extensive enough that EPA, in agreement with TAMS/Gradient, decided to have the entire PCB analysis data validation program redone to minimize manual data manipulation and transcription (e.g., Garvey, 1995). TAMS developed a computer spreadsheet macro for data validation in July 1995. This macro electronically applied blank qualification criteria (i.e., the "B" qualifier) to the electronic data files using an algorithm developed from the data validation procedures. These files were then used to generate the standard data validation forms incorporated in the validation packages. Subsequent to the electronic validation, CDM reviewed all data for blank qualifier assignment before approving the data validation packages. As a result of this review, minor changes in the macro had to be made to handle unusual data packages (e.g., extra congeners reported). Using the data validation macro, CDM completed the revalidation of the PCB samples in September 1995.

As an overall assessment of data quality, the TAMS/Gradient Program QAO reviewed pertinent aspects of the sampling and analysis program (*e.g.*, historical data, implementation of sampling protocols, laboratory performance) relative to the data quality objectives. Decisions on data usability sometimes overrode data qualification codes, as justified in this report. All qualifier changes made by the TAMS/Gradient Program QAO, as reflected in this data usability report, are noted in the final database (code Y in QA Comment field of database). For the ecological study, the QAO modified 16 qualifiers out of 59,063 PCB congener data records as a result of data usability issues, representing 0.03% of the data. Specifically, the QAO un-rejected data for 16 BZ #18 results. CDM rejected certain positive BZ #18 detects due to poor dual column precision. The QAO changed the rejection qualifier "R" to the presumptively present qualifier "N". The QAO based this decision on the routine presence of BZ #18 in historical sediment samples containing PCBs, and the consistent PCB congener pattern distribution present throughout the Hudson River sediments. Both the preponderance of BZ #18 retention time data and BZ #18 identification verification by GC/ITD for the associated ITD-confirmed samples warrants inclusion of this principal congener in the database. The QAO also corrected five result values due to transcription errors and corrected many result qualifiers due to data validation procedure errors (Wait and Cook, 1996 and Hunt, 1996).

I.5 Data Usability

I.5.1 Approach

TAMS/Gradient established a quality assurance system for this program to monitor and evaluate the accuracy, precision, representativeness, and sensitivity of the results relative to the data quality objectives. These are all important elements in evaluating data usability (e.g., USEPA, 1992b, 1993a). Accuracy is a measure of how a result compares to a true value. Precision indicates the reproducibility of generating a value. Representativeness is the degree to which a measurement(s) is indicative of the characteristics of a larger population. Sensitivity is the limit of detection of the analytical method.

In the following sections each of these parameters is evaluated for the each ecological study medium (i.e., sediment, fish, and invertebrates). Accuracy was assessed using holding times, instrument performance and calibrations for both the GC/ECD and GC/ITD, internal standard performance for the GC/ITD, surrogate criteria for the GC/ECD, spike recoveries, matrix spike/matrix spike duplicate (MS/MSD) recovery results, compared identification results, and GC/ITD confirmation results. Precision was assessed by comparing matrix spike (MS) and matrix spike duplicate (MSD) results, representativeness was evaluated by comparing field duplicate results (Tables I-3 to I-4), and sensitivity was assessed using blank results and the sample-specific quantitation limits achieved.

Comparability and completeness are two other important data quality attributes. Comparability expresses the confidence with which data are considered to be equivalent (USEPA,

1992b). Comparable data allowed for the ability to combine the analytical results obtained from this study with previous Hudson River studies. In addition, Gauthier (1994) has provided Aroclor translation procedures for Hudson River capillary column GC data relative to previous packed column GC studies. Completeness is a measure of the amount of usable data resulting from a data collection activity (USEPA, 1992b). For this program, a 95% completeness goal was established. A discussion of completeness for the ecological study is provided in the conclusions section of this report.

Most previous studies of PCB chemistry in Hudson River sediments have focused on the concentration of specific Aroclors, total PCBs and/or the distribution of PCB homologues. The current assessment of PCB fate and distribution in the Hudson River required TAMS/Gradient scientists to implement sophisticated equilibrium chemistry and transport modeling studies requiring concentration ratios of certain PCB congeners. Of the 90 target and 36 non-target congeners, 12 target congeners are of particular importance. The usability of these "principal" congeners is key to the ecological study.

Principal congeners will be employed in the following studies by the data users:

- Dechlorination product ratio - The molar sum of BZ #1, 4, 8, 10, and 19 are compared to the molar sum of all congeners analyzed. This ratio is then compared to a similar index for Aroclor 1242 to assess, calculate, and evaluate the extent of dechlorination.

- Transport modeling - BZ #4, 28, 52, 101, and 138 are considered independently as compounds modeling PCB transport.

- Aroclor 1016 and 1242 - BZ #18 is used to estimate the potential contribution of Aroclor 1016 and 1242 to Hudson River sediments.

- Aroclor 1254 - BZ #118 is used to estimate the potential contribution of Aroclor 1254 to Hudson River sediments.

- Aroclor 1260 - BZ #180 is used to estimate the potential contribution of Aroclor 1260 to Hudson River sediments.

Thus, 12 principal congeners (BZ #1, 4, 8, 10, 18, 19, 28, 52, 101, 118, 138, and 180) are the focus of this usability report. However, the remaining target and non-target congeners have important implications to the ecological study. These congeners were used to calculate the concentrations of total PCBs, PCB homologues, and Aroclor mixtures, toxic equivalency as well as for congener pattern analysis.

I.5.2 Usability - General Issues

The data quality objectives for the Hudson River Reassessment required the development of a sensitive program-specific gas chromatography method. Available standard agency methods were not adequate to achieve the congener-specific identifications and detection limits needed for the project. TAMS/Gradient based the method utilized on a modified NYSDEC ASP Method 91-11 (1989) protocol encompassing information published in the literature, as well as in-house research conducted by Aquatec. This research included Method Detection Limit (MDL) studies and Extraction Efficiency (EE) studies conducted in accordance with USEPA (1984, 1986) guidance. During the course of these studies, various nuances in the methods were noted that required refinement. As such, TAMS/Gradient and Aquatec made modifications to some of the original protocols. This section will discuss some of the more significant changes, and ramifications of those changes.

Additional Calibrated Congeners

Aquatec increased the number of PCB congeners contained in the calibration standards from the original 90 target congeners selected to include an additional eighteen congeners. These additional congeners are as follows: BZ #17, 20, 33, 42, 45, 59, 72, 74, 110, 135, 143, 156, 165, 168, 174, 176, 178, and 179. Aquatec selected these additional congeners for daily calibration due to their presence in Aroclor mixtures. This change occurred before the analysis of the ecological study, but after analysis of the high resolution core, and water column and transect studies. Use of these additional target congener data should be limited since they are not consistently quantitated for all data sets. Comparison of the concentrations of these congeners between the ecological study and the previous studies is not appropriate as the two methods of quantitation are not comparable. None of the additional congeners were selected as principal congeners, and therefore, the data analyses efforts should not be affected.

Identification of Non-Target Congeners

At the beginning of the overall program, Aquatec identified non-target congeners based on historical relative retention times reported in the literature. In August 1993, Aquatec analyzed calibration standards for each of the non-target congeners. Using these additional calibration standards, Aquatec performed analyses to confirm historical relative retention times. Though these analyses verified a majority of the historical non-target congener relative retention times, some of the historical relative retention times used to identify non-target congeners did not match the relative retention times determined by the analyses of the non-target congener standards. TAMS/Gradient deleted fourteen non-target congeners from the database for all analyses performed prior to August 1993 due to these unconfirmed identifications. The 14 non-target congeners deleted were: BZ #35, 39, 46, 100, 104, 130, 131, 132, 134, 162, 165, 173, 176, and 179. Aquatec identified and confirmed these 14 congeners based on the current laboratory-derived relative retention times for

samples analyzed during and after August 1993. Therefore, the results for these 14 non-target congeners remain in the database for all samples analyzed during and after August 1993, which includes all of the ecological analyses. Use of these non-target congener data should be limited since they are not consistently available for all data sets. If a situation arises where information for the deleted non-target congeners is critical to a data user, an in-depth review of the chromatograms and re-calculation of the concentrations could potentially produce usable results for some of these congeners.

Quantitation of Non-Target Congeners

The laboratory originally quantitated non-target congeners using the calibration curve determined for BZ #52. Since the non-target congener results were to be included in the calculations of homologue and total PCB mass, TAMS/Gradient desired a more accurate method of quantifying the non-target congeners. Aquatec analyzed calibration standards for the non-target congeners in September 1993, and again in April 1994, for the determination of congener-specific response factors. Based on this information, TAMS/Gradient calculated correction factors for each non-target congener and applied these to the laboratory data within the database (Bonvell, 1994b).

GC Column Change

Initially, Aquatec used a HP-5 (or RTx-5) column and a SB-octyl-50 GC column for PCB congener analyses. In November 1993, Aquatec obtained new SB-octyl-50 columns for pending analyses of Phase 2B ecological samples. Each of the new SB-octyl-50 columns showed signs of column degradation resulting in severe peak retention time shifts. Due to the concern that an acceptable SB-octyl-50 column would not be obtainable, TAMS/Gradient solicited approval from USEPA Region II for a replacement column, Apiezon_L. TAMS/Gradient was concerned about data comparability for the overall program, but had no alternative. USEPA Region II concurred with the replacement of the SB-octyl-50 column with the Apiezon_L column in December 1993. The Apiezon_L column was selected for the following reasons:

- The Apiezon_L column phase is similar to the SB-octyl-50 column phase.
- The Apiezon_L column provides PCB congener separations similar to the SB-octyl-50 column.
- The PCB congener retention times on the Apiezon_L column are more stable than on the SB-octyl-50 column.
- The NYSDEC analytical laboratory performing Hudson River PCB congener analyses was using the Apiezon_L column successfully for fish samples.

In February 1994, Aquatec performed a comparison study for the two column sets, HP-5/SB-octyl-50 and HP-5/Apiezon_L (Cook, 1994). Aquatec analyzed four Phase 2B pilot fish samples on

both the HP-5/SB-octyl-50 column combination and also the RTx-5/Apiezon_L column combination. The PCB congener results compared well qualitatively and quantitatively with few exceptions. The results for BZ #15 and 37 were consistently 2 to 10 times higher on the SB-octyl-50 column pair. Data users are cautioned that the results for BZ #15 and 37 reported through March 1994 and the same congeners reported after March 1994 are not comparable due to differences in the method of quantitation. For example, comparisons between high resolution sediment data and the ecological data are not appropriate for BZ #15 and 37.

Lower Column Concentration Bias

The USEPA CLP protocol requires that for dual column GC analyses, the lower of the two values from each column will be reported (USEPA, 1991). TAMS/Gradient incorporated this same quantitation scheme into this program. This quantitative method may introduce a slight low bias when calculating homologue and total PCB sums. TAMS/Gradient determined that this bias was usually negligible, and on a worst-case basis, may be as much as 2 to 10% low. Therefore, the data user should consider these totals as usable, but estimated values, due to the uncertainties of the individual results which are summed to form these values.

Confirmation by GC/ITD

Aquatec analyzed approximately 10% of all samples analyzed by GC/ECD by GC/ITD to provide an additional mechanism to verify congener identification and, as a secondary objective, quantification of congeners. The ITD is not as sensitive as the ECD (approximately an order of magnitude less sensitive); therefore, when possible, samples with the highest concentration of PCBs were selected for GC/ITD confirmation. Although this may result in a program bias for only confirming high concentration samples, the overall effect does not impair data usability.

In addition, there is the potential for some quantitative bias associated with the GC/ITD results relative to the GC/ECD results. Aquatec quantified each congener detected in the GC/ITD analysis using an average response factor per level of chlorination rather than using response factors determined specifically for each individual congener. As such, potential bias, which will vary for each congener within a chlorination homologue group, is present with the GC/ITD results.

Inconsistencies Between CDM's and EcoChem's Validation Procedures of Fish Data

USEPA funded the analysis of 120 fish samples as part of the Phase 2B program and NOAA funded the analysis of 115 fish samples. The usability of NOAA's fish analyses is addressed in this report as these results were included in the project database. Aquatec performed the PCB analyses on both the USEPA and NOAA fish samples using the same tissue extraction and analysis methods; therefore, the analytical results are comparable. CDM performed the data validation of the EPA fish samples according to the Appendices A-6 and A-7 of the Phase 2B SAP/QAPP. CDM performed the validation of the high resolution coring, water column and transect, and low resolution coring data

using the same validation procedures. EcoChem, Inc. of Seattle, Washington performed the data validation of the NOAA fish samples. EcoChem's validation approach differed from CDMs on two significant issues. EcoChem only calculated blank action levels for PCB congeners that were confirmed on both analytical columns, whereas CDM calculated blank action level for all PCB congeners detected on either column as specified in Appendix A-6 of the Phase 2B SAP/QAPP. EcoChem did not qualify results based on dual column imprecision, whereas CDM qualified results as estimated, presumptively present, or rejected on the degree of dual column imprecision as specified in Appendix A-6. These differences resulted in many fewer NOAA results qualified for blank actions and dual column imprecision. EcoChem performed full validation on two SDGs and cursory validation on seven SDGs. The cursory validation consisted of reviewing only the data summary forms and not the associated raw data. CDM performed full validation of all fish SDGs. EcoChem's less conservative approach to data validation may have resulted in the reporting of false positive congener results, especially at low concentrations.

I.5.3 Usability of Sediment Data - Accuracy, Precision, Representativeness, and Sensitivity

I.5.3.1 Accuracy

Holding Times

Exceedance of holding times may indicate a possible loss of PCB congeners due to volatilization, chemical reactions, and/or biological alterations. Due to the persistent nature of PCBs, only severe exceedance should be considered deleterious to quantitative accuracy. For the sediment samples, USEPA established an extraction holding time of seven days from sampling, followed by an analysis holding time of 40 days from extraction.

Aquatec missed the analytical holding times for three sediment samples by 11 days. CDM appropriately qualified all data affected by missed holding times as estimated. Aquatec has routinely demonstrated that the stability of PCB congener standards in solvent is at least six months. Therefore, TAMS/Gradient considered the data for these samples to be usable as estimated values.

GC/ECD Instrument Performance

Adequate chromatographic resolution and retention time stability throughout an analytical sequence are essential attributes for qualitative identification of congeners on a GC. The criteria for congener resolution and retention time windows are defined in the Phase 2B SAP/QAPP. For the SB-octyl-50 column, resolution must be greater than 50% between BZ #5 and 8, 40 and 41, 183 and 185, and BZ #209 and OCN. On the HP-5 column, resolution must be greater than 25% between BZ #4, 10 and tetrachloro-m-xylene (TCMX), and between BZ #31 and 28. Resolution must be greater than 50% between BZ #84 and 101/90, and between BZ #206 and OCN. On the Apiezon_L column, resolution must be greater than 25% between BZ #9 and TCMX, between TCMX and BZ

#7, and between BZ #187 and BZ #128. Aquatec initially established retention time windows for both columns to be ±0.3% relative to the average initial calibration retention times for all target congeners and surrogates.

CDM noted the only congener calibration standard coelution problems for BZ #5 with BZ #8 were on the SB-octyl-50 column in SDG 38514 with resolution <10%. CDM appropriately qualified all data affected by this issue as estimated. TAMS/Gradient considered these data to be usable as estimated values. Only one SDG (38866) had any exceedances for retention time criteria requiring qualification. CDM qualified several BZ #31 results in this SDG as presumptively present due to retention times shifts in the calibration standards. Other compounds within this SDG had retention times outside of the established retention time windows. However, these retention times were within an expanded retention time window of ±0.5 (as agreed to by EPA Region II), and therefore did not affect identification.

GC/ECD Calibration

Instrument calibration (IC) requirements were established to verify the production of acceptable quantitative data. Initial calibrations using 5-level standard concentration curves demonstrate an instrument is capable of acceptable performance prior to sample analysis. The IC criteria are 20% relative standard concentration error (% RSCE) for monochlorobiphenyls and 15% RSCE for all remaining PCB congeners, and a correlation coefficient $ 0.995. Continuing calibration standards document maintenance of satisfactory performance over time. TAMS/Gradient noted no significant continuing calibration problems.

Surrogate Spike Recoveries

Aquatec spiked surrogate compounds into all sediment samples prior to extraction to monitor recoveries. Recoveries may be indicative of either laboratory performance or sample matrix effects. For the ecological study, Aquatec used TCMX and BZ #192 as surrogates. CDM appropriately qualified as estimated any data associated with samples that had TCMX or BZ #192 recoveries outside of a range of 60-150% including one sample in SDG 38514 and two samples in SDG 38701. TAMS/Gradient considered these results to be usable as estimated values.

Matrix Spike/Matrix Spike Duplicate Recoveries

Within each SDG, two aliquots of a representative sediment sample were spiked with a suite of 20 congeners (BZ #8, 18, 28, 44, 52, 66, 77, 101, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, and 209). The purpose of the spikes was, in part, to evaluate the accuracy of the analytical method relative to laboratory performance and specific sample matrix. The advisory limits for spiked congener recoveries are 60-150%. TAMS/Gradient noted MS/MSD recovery exceedances in two SDGs. SDG 38514 had low recoveries for BZ #8, BZ #28, and BZ #52, and SDG 38866 had low recoveries for all spiked congeners in the MS. CDM appropriately qualified the associated results

in the nonspiked samples as estimated. TAMS/Gradient considered these results to be usable as estimated values. MS/MSD analyses were conducted for five ecological sediment samples. This represents a frequency of 5.4%, which exceeds the 5% requirement stipulated in Phase 2B SAP/QAPP.

Compound Identification

TAMS/Gradient established qualitative criteria to minimize erroneous identification of congeners. An erroneous identification can be either a false positive (reporting a compound present when it is not) or a false negative (not reporting a compound that is present). The calculated concentrations for congeners detected in both columns should not differ by more than 25% between columns (%D # 25%). This criterion applies to only those congeners which can be resolved as individual congeners on both columns. If the %D for the results between the two columns is > 25% but # 50% the results were qualified as estimated. If the %D was > 50% but # 90%, the results were qualified as estimated and presumptively present. If the %D between columns was > 90%, the results were considered unusable.

TAMS/Gradient noted extensive problems with congener identifications as a result of dual column imprecision for all SDGs. In fact, a majority of the estimated and rejected data for the ecological study were a result of dual GC column imprecision. CDM qualified the following congeners as rejected at frequencies greater than 10% as a result of dual GC column imprecision: target congener BZ #1 (20%), BZ #2 (11%), BZ #3 (27%), BZ #5 (12%), target congener BZ #8 (11%), BZ #12 (37%), BZ #16 (14%), BZ #77 (13%), BZ #119 (16%), and BZ #165 (12%). With the level of background organic material present in Hudson sediments, resultant interferences, particularly for congeners with low concentrations, likely caused these differences between the dual GC column results.

GC/ITD Instrument Performance

GC/ITD performance required evaluating GC column resolution, ion trap detector sensitivity, and ion trap calibration. The GC resolution criteria required baseline separation of BZ #87 from BZ #154 and BZ #77. The ion trap sensitivity requires the signal/noise ratio for m/z 499 for BZ #209 and m/z 241 for chrysene-d_{12} to be greater than 5. For ion trap calibration, the abundance of m/z 500 relative to m/z 498 for BZ #209 must be $ 70% but #95%. TAMS/Gradient noted no significant ITD performance problems for samples analyzed during the ecological study.

GC/ITD Calibration

The initial calibration criterion for acceptable quantitative data for GC/ITD analyses required percent relative standard deviations (RSD) of the congener relative response factor (RRF) to be less than 20%. For continuing calibration, the RRF for each congener must be within 20% of the mean

calibration factor from the 5-level calibration at the beginning and end of each calibration sequence. For the ecological study, TAMS/Gradient noted no significant GC/ITD calibration problems.

GC/ITD Internal Standard Performance

To demonstrate the stability of the ITD, internal standard performance criteria were monitored. Internal standard area counts must not vary by more than 30% from the most recent calibration or by more than 50% from the initial calibration. In addition, the absolute retention time of the internal standard must be within 10 seconds of the retention time in the most recent calibration, and ion abundance criteria must be met for chrysene-d_{12} and phenanthrene-d_{10}. The response for chrysene-d_{12} in several samples exceeded criteria, for which CDM appropriately qualified the associated data as estimated. TAMS/Gradient considered these results to be usable as estimated values.

Confirmation by GC/ITD

CDM qualified all positive GC/ITD results that had signal/noise ratios of less than 3 as nondetect due to uncertainty in the identification. TAMS/Gradient considered these results to be usable as nondetects at the reported quantitation limits.

Aquatec analyzed approximately 10% of all samples analyzed by GC/ECD by GC/ITD to provide an additional mechanism to verify congener identification and, as a secondary objective, quantification of congeners. Since the ITD method was not designed to be a primary quantitative tool, some variations in quantitative results were expected. TAMS/Gradient considered quantitative differences between the GC/ITD and GC/ECD results less than a factor of five acceptable, while differences greater than five times were considered unacceptable. CDM qualified GC/ECD results that were detected at concentrations above the GC/ITD quantitation limit but that were not confirmed by GC/ITD with an "Q". TAMS/Gradient converted all "Q" qualifiers to "JN" due to the potential reporting of false positive results. CDM qualified eight sediment results with "Q" qualifiers; TAMS/Gradient considered these results to be presumptively present. CDM qualified GC/ECD results that were not detected or were less than one-fifth the GC/ITD results with an "M". TAMS/Gradient converted all "M" qualifiers to "R" as the nondetect GC/ECD may be a false negative or the GC/ECD result may be significantly biased low. CDM qualified 93 sediment results (of which 10 were principal congeners) with "M" qualifiers; TAMS/Gradient considered these results to be unusable.

I.5.3.2 Precision

Matrix Spike/Matrix Spike Duplicate Comparison

The analysis of MS and MSD samples can also provide valuable information regarding method precision relative to laboratory performance and specific sample matrix. The advisory limit for

relative percent difference (RPD) of spiked congeners in a MS/MSD pair is 40%, and for nonspiked congeners, the precision criterion is 40% RSD.

TAMS/Gradient noted MS/MSD precision exceedances in three SDGs - 38514, 38701, and 38860. The RPDs for all spiked and nonspiked congeners in SDG 38860 exceeded the criterion of 40% due to low recoveries in the MS analyses. Spiked congeners BZ #8, BZ #28, and BZ #52 in SDG 38514, and spiked congener BZ #8 and nonspiked congeners BZ #1, BZ #3, BZ # 83, and BZ #95 in SDG 38701 exceeded the RPD criterion of 40%. CDM appropriately qualified the results for these congeners in the associated nonspiked samples. TAMS/Gradient considered these data to be usable as estimated values.

I.5.3.3 Representativeness

Field Duplicate Results

Analysis of field co-located samples provides an indication of the overall precision of the sampling and analysis program. These analyses measure both field and laboratory precision; therefore, the results will likely have more variability than laboratory duplicates and MS/MSD samples, which only measure laboratory precision. Data validators used a 50% RPD criterion for evaluating field duplicate precision. Any congener precision greater than 50% RPD was qualified as estimated.

A total of five field co-located samples were analyzed for the ecological sediment samples. This represents a frequency of 5.4%, which exceeds the 5% requirement stipulated in the Phase 2B SAP/QAPP. For the field co-located samples reported in SDGs 38866 and 38940, CDM qualified as estimated only four and five compounds, respectively, due to poor field duplicate precision. The three remaining co-located samples had significant numbers of congeners (target and non-target) with RPDs greater than the criterion of 50%: SDGs 38514 (63), SDG 38655 (16), and SDG 39045 (66). Table I-3 summarizes the duplicate precision results for the congeners for each field co-located sediment sample. CDM appropriately qualified these results as estimated. TAMS/Gradient scrutinized the data validation reports for these three SDGs for errors, but found none. The differences may be a result of sample heterogeneity. TAMS/Gradient considered these data to be usable as estimated values.

I.5.3.4 Sensitivity

Blanks

An important data quality objective associated with the ecological study was to obtain detection limits as low as the analytical method could produce. One effect of this approach is to register low level blank contamination during the preparation and analysis of the sediments. As such, numerous congeners in all samples in all SDGs required blank contamination qualifications.

TAMS/Gradient reviewed the distribution of blank contaminants and found most contamination associated with the monochlorobiphenyls, particularly with BZ #2. Blank levels for BZ #2 usually ranged up to 45 ppb in extract. Since BZ #2 is neither a dechlorination product, a major Aroclor component, nor a principal congener, TAMS/Gradient did not consider this to be a serious data quality problem. CDM qualified six principal congeners in several samples due to blank contamination including: BZ #1 (6 results), BZ #4 (17 results), BZ #8 (18 results), BZ #19 (34 results), BZ #28 (15 results), and BZ #180 (9 results). TAMS/Gradient considered these results to be usable as s.

CDM qualified results during data validation with a "B", which indicated that the result was within five times of the blank action level. TAMS/Gradient converted all "B" qualified results in the database to nondetect results due to uncertainty in this detection. Table I-5 summarizes the congener detects changed to nondetects for the sediment samples. TAMS/Gradient considered these results to be usable as nondetects at the reported quantitation limit.

Quantitation Limits

Evaluating bioaccumulation in benthic and epibenthic invertebrates necessitated obtaining low detection limits for the associated sediment samples. TAMS/Gradient and Aquatec devised analytical methods to achieve lower detection limits. This, in part, required employing sample/extract cleanup methods to remove matrix interferences, and maximizing sample size when possible. For the ecological sediment study, TAMS/Gradient defined optimum detection limits as 1 µg/Kg for monochlorobiphenyls, 0.5 µg/Kg for dichlorobiphenyls through hexachlorobiphenyls, and 0.5-1 µg/Kg for heptachloro-biphenyls through decachlorobiphenyl. Results of the MDL study necessitated raising the detection limit for BZ #2 (a monochlorobiphenyl) significantly above these requirements (approximately a factor of 3).

In general, achieving appropriate detection limits for the sediment samples was not a problem. Whenever TAMS/Gradient noted raised detection limits, the affected samples contained high organic content; specifically the presence of PCBs. The relative ratio of congeners detected within each high-concentration sample remained reasonably consistent, therefore the raised detection limit for nondetect congeners did not affect data usability. Aquatec achieved adequate detection limits for critical low level samples used for describing biological processes in the Hudson River sediments.

I.5.4 Usability of Fish - Accuracy, Precision, Representativeness, and Sensitivity

EPA funded the analysis of 120 fish samples as part of the Phase 2B program and NOAA funded the analysis of 115 fish samples. TAMS/Gradient addresses the usability of NOAA’s fish analyses in this report as these results were included in the project database.

I.5.4.1 Accuracy

Holding Times

Exceedance of holding times may indicate a possible loss of PCB congeners due to volatilization, chemical reactions, and/or biological alterations. Due to the persistent nature of PCBs, only severe exceedance should be considered deleterious to quantitative accuracy. TAMS and NYS Fish and Wildlife personnel froze the fish samples immediately after collection. TAMS/Gradient established an extraction holding time of seven days from sample thawing, followed by an analysis holding time of 40 days from extraction.

Aquatec missed the analytical holding times for two fish samples by 38 days. CDM appropriately qualified all data affected by missed holding times as estimated. Aquatec has routinely demonstrated that the stability of PCB congener standards in solvent is at least six months. Therefore, TAMS/Gradient considered the data for these samples to be usable as estimated values.

GC/ECD Instrument Performance

Adequate chromatographic resolution and retention time stability throughout an analytical sequence are essential attributes for qualitative identification of congeners on a GC. TAMS/Gradient defined criteria for congener resolution and retention time windows in the Phase 2B SAP/QAPP. For the SB-octyl-50 column, resolution must be greater than 50% between BZ #5 and 8, 40 and 41, 183 and 185, and BZ #209 and OCN. On the HP-5 column, resolution must be greater than 25% between BZ #4, 10 and TCMX, and between BZ #31 and 28. Resolution must be greater than 50% between BZ #84 and 101/90, and between BZ #206 and OCN. On the Apiezon_L column, resolution must be greater than 25% between BZ #9 and TCMX, between TCMX and BZ #7, and between BZ #187 and BZ #128. Aquatec initially established retention time windows for both columns to be ±0.3% relative to the average initial calibration retention times for all target congeners and surrogates.

CDM noted the only congener calibration standard coelution problems for BZ #5 with BZ #8 and BZ #209 and OCN were on the SB-octyl-50 column in SDG 203001 with resolution less than criteria; CDM determined that no action was necessary. TAMS/Gradient noted no significant exceedances for retention time criteria requiring qualification. Several SDGs had retention times outside of the established retention time windows. However, these retention times were within an expanded retention time window of ±0.5 (as agreed to by EPA Region II), and therefore did not affect identification.

GC/ECD Calibration

Instrument calibration (IC) requirements were established to verify the production of acceptable quantitative data. Initial calibrations using 5-level standard concentration curves demonstrate an instrument is capable of acceptable performance prior to sample analysis. The IC

criteria are 20% RSCE for monochlorobiphenyls and 15% RSCE for all remaining PCB congeners, and a correlation coefficient $ 0.995. Continuing calibration standards document maintenance of satisfactory performance over time. TAMS/Gradient noted no significant continuing calibration problems.

Surrogate Spike Recoveries

Aquatec spiked surrogate compounds into all fish samples prior to extraction to monitor recoveries. Recoveries may be indicative of either laboratory performance or sample matrix effects. For the ecological study, Aquatec used TCMX and BZ #192 as surrogates. CDM appropriately qualified as estimated one sample in SDG 5F that had TCMX and BZ #192 recoveries outside of a range of 60-150%. TAMS/Gradient considered these results to be usable as estimated values.

Matrix Spike/Matrix Spike Duplicate Recoveries

Within each SDG, two aliquots of a representative fish sample were spiked with a suite of 20 congeners (BZ #8, 18, 28, 44, 52, 66, 77, 101, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, and 209). The purpose of the spikes was, in part, to evaluate the accuracy of the analytical method relative to laboratory performance and specific sample matrix. The advisory limits for spiked congener recoveries are 60-150%. TAMS/Gradient noted MS/MSD recovery exceedances in SDG 1F, SDG 2F, and SDG 3F. TAMS/Gradient did not considered these exceedance to be serious, and CDM appropriately qualified the associated results in the nonspiked sample as estimated. TAMS/Gradient considered these results to be usable as estimated values. MS/MSD analyses were analyzed for seven EPA fish samples. This represents a frequency of 5.8%, which exceeds the 5% requirement stipulated in Phase 2B SAP/QAPP.

Compound Identification

TAMS/Gradient established qualitative criteria to minimize erroneous identification of congeners. An erroneous identification can be either a false positive (reporting a compound present when it is not) or a false negative (not reporting a compound that is present). The calculated concentrations for congeners detected in both columns should not differ by more than 25% between columns (%D # 25%). This criterion applies to only those congeners which can be resolved as individual congeners on both columns. If the %D for the results between the two columns is > 25% but # 50% the results qualified as were estimated. If the %D was > 50% but # 90%, the results qualified as were estimated and presumptively present. If the %D between columns was > 90%, the results were considered unusable.

TAMS/Gradient noted extensive problems with congener identifications as a result of dual column imprecision for all SDGs for the EPA funded fish analyses. In fact, a majority of the estimated and rejected data for the fish study were a result of dual GC column imprecision. CDM qualified the following congeners as rejected at frequencies greater than 10% as a result of dual GC

column imprecision: BZ #2 (18%) and BZ #6 (44%). With the complex composition of tissue samples, resultant interferences, particularly for congeners with low concentrations, likely caused these differences between the dual GC column results.

Positive Control Sample Recoveries

NOAA submitted ten positive control samples (*i.e.*, Saginaw Bay Carp) to the analytical laboratory, one with each sample batch. EcoChem selected fourteen congeners for evaluation because these congeners were free of coelutions and had congener concentrations above the calibrated quantitation limit. Four of those selected were principal congeners: BZ #18, BZ #28, BZ #52, and BZ #180. EcoChem compared these results to the average results provided by the U.S. Fish and Wildlife (Positive Control Carp Tracking, 10/9/93) and considered results in a window of plus or minus two standard deviation from the mean to be outliers. Of the 140 results evaluated, 71 results (~50%) were outside of the evaluation limits. EcoChem did not qualify data due to positive control outliers as no certified values are available for the positive control samples.

GC/ITD Instrument Performance

GC/ITD performance required evaluating GC column resolution, ion trap detector sensitivity, and ion trap calibration. The GC resolution criterion required baseline separation of BZ #87 from BZ #154 and BZ #77. The ion trap sensitivity requires the signal/noise ratio for m/z 499 for BZ #209 and m/z 241 for chrysene-d_{12} to be greater than 5. For ion trap calibration, the abundance of m/z 500 relative to m/z 498 for BZ #209 must be $ 70% but #95%. CDM qualified the GC/ITD results in SDG 3F as estimated due to failure to meet the 70-95% criterion for m/z 500 relative to m/z 498 for BZ #209. TAMS/Gradient considered these results to be usable as estimated values. TAMS/Gradient noted no other ITD performance problems for fish samples analyzed during the ecological study.

GC/ITD Calibration

The initial calibration criteria for acceptable quantitative data for GC/ITD analyses required RSD of the congener RRF to be less than 20%. For continuing calibration, the RRF for each congener must be within 20% of the mean calibration factor from the 5-level calibration at the beginning and end of each calibration sequence. For the fish samples analyzed during the ecological study, TAMS/Gradient noted no significant GC/ITD calibration problems.

GC/ITD Internal Standard Performance

To demonstrate the stability of the ITD, internal standard performance criteria were monitored. Internal standard area counts must not vary by more than 30% from the most recent calibration or by more than 50% from the initial calibration. In addition, the absolute retention time of the internal standard must be within 10 seconds of the retention time in the most recent calibration, and ion abundance criteria must be met for chrysene-d_{12} and phenanthrene-d_{10}. The response for the

internal standards in samples included in SDGs 1F, 2F, 4F, and 203001 exceeded response and/or ion ratio criteria. CDM appropriately qualified the associated data as estimated and TAMS/Gradient considered these results to be usable as estimated values.

Confirmation by GC/ITD

CDM qualified all positive GC/ITD results that had signal/noise ratios of less than 3 as nondetect due to uncertainty in the identification. TAMS/Gradient considered these results to be usable as nondetects at the reported quantitation limits.

Aquatec analyzed approximately 10% of all samples analyzed by GC/ECD by GC/ITD to provide an additional mechanism to verify congener identification and, as a secondary objective, quantification of congeners. Since the ITD method was not designed to be a primary quantitative tool, some variations in quantitative results were expected. TAMS/Gradient considered quantitative differences between the GC/ITD and GC/ECD results less than a factor of five acceptable, while differences greater than five times were considered unacceptable. CDM and EcoChem qualified GC/ECD results that were detected at concentrations above the GC/ITD quantitation limit but that were not confirmed by GC/ITD with an "Q". TAMS/Gradient converted all "Q" qualifiers to "JN" due to the potential reporting of false positive results. CDM and EcoChem qualified a total of eight fish results including one principal congener with "Q" qualifiers; TAMS/Gradient considered these results to be presumptively present. CDM and EcoChem qualified GC/ECD results that were not detected or were less than one-fifth the GC/ITD results with an "M". TAMS/Gradient converted all "M" qualifiers to "R" as the nondetect GC/ECD may be a false negative or the GC/ECD result may be significantly biased low. CDM and EcoChem qualified 45 fish results, of which one was a principal congener, with "M" qualifiers; TAMS/Gradient considered these results to be unusable.

I.5.4.2 Precision

Matrix Spike/Matrix Spike Duplicate Comparison

The analysis of MS and MSD samples can also provide valuable information regarding method precision relative to laboratory performance and specific sample matrix. The advisory limit for RPD of spiked congeners in a MS/MSD pair is 40%, and for nonspiked congeners, the precision criterion is 40% RSD.

TAMS/Gradient noted MS/MSD precision exceedances in three SDGs - 38514, 38701, and 38860. The RPDs for all spiked and nonspiked congeners in SDG 38860 exceeded the criterion of 40% due to low recoveries in the MS analyses. Spiked congeners BZ #8, BZ #28, and BZ #52 in SDG 38514, and spiked congener BZ #8 and nonspiked congeners BZ #1, BZ #3, BZ # 83, and BZ #95 in SDG 38701 exceeded the RPD criterion of 40%. CDM appropriately qualified the results for these congeners in the associated nonspiked samples. TAMS/Gradient considered these data to be usable as estimated values.

Positive Control Sample Comparison

As previously noted, NOAA submitted ten positive control samples (*i.e.*, Saginaw Bay Carp) to the analytical laboratory. The ten positive control sample analyses represent replicate analyses. TAMS/Gradient evaluated the precision, measured as RSD, between these ten analyses and noted excellent precision (*i.e.*, %RSDs less than 15%) for the fourteen evaluated congeners including principal congeners BZ #18, BZ #28, BZ #52, and BZ #180.

I.5.4.3 Representativeness

Field Duplicate Results

Analysis of field duplicate or co-located samples provides an indication of the overall precision of the sampling and analysis program. These analyses measure both field and laboratory precision; therefore, the results will likely have more variability than laboratory duplicates and MS/MSD samples, which only measure laboratory precision. Duplicate samples were not prepared for the fish analyses, therefore, TAMS/Gradient could not assess representativeness for the fish analyses.

I.5.4.4 Sensitivity

Blanks

An important data quality objective associated with the ecological study was to obtain detection limits as low as the analytical method could produce. One effect of this approach is to register low level blank contamination during the preparation and analysis of the fish. As such, numerous congeners in all samples in all SDGs required blank contamination qualifications. TAMS/Gradient reviewed the distribution of blank contaminants and found most contamination associated with the monochlorobiphenyls, particularly with BZ #2 and BZ #3. Blank levels for BZ #2 usually ranged up to 98 ppb in extract and up to 22 ppb for BZ #3. Since BZ #2 and BZ #3 are neither dechlorination products, major Aroclor components, nor principal congeners, TAMS/Gradient did not consider this to be a serious data quality problem. CDM qualified nine principal congeners in the EPA funded fish samples due to blank contamination including: BZ #8 (59 results), BZ #10 (10 results), BZ #18 (7 results), BZ #28 (4 results), BZ #52 (4 results), BZ #118 (2 results) and BZ #180 (19 results). EcoChem qualified six principal congeners in the NOAA funded fish samples due to blank contamination including: BZ #8 (1 result), BZ #52 (3 results), BZ #101 (2 results), BZ #118 (2 results), BZ #138 (2 results), and BZ #180 (1 result). TAMS/Gradient considered the results qualified by CDM and EcoChem to be usable as nondetects.

CDM and EcoChem qualified results during data validations with a "B", which indicated that the result was within five times of the blank action level. TAMS/Gradient converted all "B" qualified results in the database to nondetect results due to uncertainty in this detection. Tables I-6 and I-7

summarize the congener detects changed to nondetects for the EPA and NOAA fish analyses. TAMS/Gradient considered these results to be usable as nondetects at the reported quantitation limit.

Quantitation Limits

Evaluating bioaccumulation of PCB congeners in fish necessitated obtaining low detection limits. TAMS/Gradient and Aquatec devised analytical methods to achieve lower detection limits. This, in part, required employing sample/extract cleanup methods to remove matrix interferences, and maximizing sample size when possible. For the fish analyses, TAMS/Gradient defined optimum detection limits as 4-8 µg/Kg for monochlorobiphenyls, 2-4 µg/Kg for dichlorobiphenyls through hexachlorobiphenyls, and 2-4 µg/Kg for heptachlorobiphenyls through decachlorobiphenyl. Results of the MDL study supported these detection limits.

In general, achieving appropriate detection limits for the fish analyses was not a problem. Whenever TAMS/Gradient noted raised detection limits, the affected samples contained high organic content; specifically the presence of PCBs. Aquatec reported multiple analyses for fish sample requiring dilutions in order to report the low detection limits and low concentration congener results, while also reporting high concentration congeners within the instrument calibration range. TAMS/Gradient detailed the approach to dilution analyses in a letter to Aquatec (DiBernardo, 1994).

I.5.5 Usability of Invertebrate Data - Accuracy, Precision, Representativeness, and Sensitivity

I.5.5.1 Accuracy

Holding Times

Exceedance of holding times may indicate a possible loss of PCB congeners due to volatilization, chemical reactions, and/or biological alterations. Due to the persistent nature of PCBs, only severe exceedance should be considered deleterious to quantitative accuracy. St. John's personnel froze the invertebrate samples within four days of sample collection. St. John's transferred the invertebrate samples to Aquatec, where the samples remained frozen until sample extraction. TAMS/Gradient established an extraction holding time of seven days from sample thawing, followed by an analysis holding time of 40 days from extraction. CDM misinterpreted associated sample custody logs and inferred that the invertebrate samples remained unthawed from mid-June to the end of August when the samples were extracted. Based on the misconception that the invertebrate samples remained unfrozen for a considerable length of time, CDM qualified all invertebrate results are estimated. Due to the difficulty in differentiating between the qualifiers applied solely due to this issue and qualifiers applied for other data quality issues, TAMS/Gradient decided to leave the qualifiers CDM applied due to the holding time issue in the database.

GC/ECD Instrument Performance

Adequate chromatographic resolution and retention time stability throughout an analytical sequence are essential attributes for qualitative identification of congeners on a GC. TAMS/Gradient defined criteria for congener resolution and retention time windows in the Phase 2B SAP/QAPP. For the SB-octyl-50 column, resolution must be greater than 50% between BZ #5 and 8, 40 and 41, 183 and 185, and BZ #209 and OCN. On the HP-5 column, resolution must be greater than 25% between BZ #4, 10 and TCMX, and between BZ #31 and 28. Resolution must be greater than 50% between BZ #84 and 101/90, and between BZ #206 and OCN. On the Apiezon_L column, resolution must be greater than 25% between BZ #9 and TCMX, between TCMX and BZ #7, and between BZ #187 and BZ #128. Aquatec initially established retention time windows for both columns to be ±0.3% relative to the average initial calibration retention times for all target congeners and surrogates.

CDM noted the one congener calibration standard coelution problem for TCMX and BZ #7 on the Apiezon_L column in SDG 4B with resolution less than criteria; CDM appropriately qualified BZ #7 in the associated samples as estimated. Gradient qualified the nondetect detect results for nine samples included in SDG 202834 as rejected due to severe retention time shifts; Gradient confirmed the positive results in these samples by comparison to revised RTWs established from the preceding and following continuing calibration standards. CDM considered the positive BZ #2, BZ #3, and BZ #136 results for several samples in SDG 2B to be not detected due to exceedance of the retention time windows. Several SDGs had retention time outside of the established retention time windows. However, these retention times were within an expanded retention time window of ±0.5% (as agreed to by EPA Region II), and therefore, did not affect identification.

GC/ECD Calibration

Instrument calibration requirements were established to verify the production of acceptable quantitative data. Initial calibrations using 5-level standard concentration curves demonstrate an instrument is capable of acceptable performance prior to sample analysis. The IC criteria is 20% RSCE for monochlorobiphenyl and 15% RSCE for all remaining PCB congeners, and a correlation coefficient $ 0.995. Continuing calibration standards document maintenance of satisfactory performance over time. TAMS/Gradient noted no significant continuing calibration problems.

Surrogate Spike Recoveries

Aquatec spiked surrogate compounds into all invertebrate samples prior to extraction to monitor recoveries. Recoveries may be indicative of either laboratory performance or sample matrix effects. For the ecological study, Aquatec used TCMX and BZ #192 as surrogates. TAMS/Gradient noted no significant surrogate spike recovery problems

Matrix Spike/Matrix Spike Duplicate Recoveries

Within each SDG, two aliquots of a representative invertebrate sample were spiked with a suite of 20 congeners (BZ #8, 18, 28, 44, 52, 66, 77, 101, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, and 209). The purpose of the spikes were, in part, to evaluate the accuracy of the analytical method relative to laboratory performance and specific sample matrix. The advisory limits for spiked congener recoveries are 60-150%. TAMS/Gradient noted MS/MSD recovery exceedances in SDG 2B. TAMS/Gradient did not considered these exceedance to be serious, and CDM appropriately qualified the associated results in the nonspiked sample as estimated. TAMS/Gradient considered these results to be usable as estimated values. MS/MSD analyses were conducted for five invertebrate samples. This represents a frequency of 6%, which exceeds the 5% requirement stipulated in Phase 2B SAP/QAPP.

Compound Identification

TAMS/Gradient established qualitative criteria to minimize erroneous identification of congeners. An erroneous identification can be either a false positive (reporting a compound present when it is not) or a false negative (not reporting a compound that is present). The calculated concentrations for congeners detected in both columns should not differ by more than 25% between columns (%D # 25%). This criterion applies to only those congeners which can be resolved as individual congeners on both columns. If the %D for the results between the two columns is > 25% but # 50%, the results were qualified as estimated. If the %D was > 50% but # 90%, the results qualified as were estimated and presumptively present. If the %D between columns was > 90%, the results were considered unusable.

TAMS/Gradient noted extensive problems with congener identifications as a result of dual column imprecision for all SDGs. In fact, a majority of the estimated and rejected data for the fish study were a result of dual GC column imprecision. CDM qualified the following congeners as rejected at frequencies greater than 10% as a result of dual GC column imprecision: BZ #2 (18%), BZ #29 (10%), BZ #41 (10%), BZ #168 (13%), BZ #180 (16%), BZ #189 (12%), BZ #194 (10%), BZ #195 (18%), and BZ #196 (10%). With the complex composition of tissue samples, resultant interferences, particularly for congeners with low concentrations, likely caused these differences between the dual GC column results.

GC/ITD Instrument Performance

GC/ITD performance required evaluating GC column resolution, ion trap detector sensitivity, and ion trap calibration. The GC resolution criterion required baseline separation of BZ #87 from BZ #154 and BZ #77. The ion trap sensitivity requires the signal/noise ratio for m/z 499 for BZ #209 and m/z 241 for chrysene-d_{12} to be greater than 5. For ion trap calibration, the abundance of m/z 500 relative to m/z 498 for BZ #209 must be $70% but #95%. CDM qualified the GC/ITD results in

SDG 1B as estimated due to failure to meet the 70-95% criteria for m/z 500 relative to m/z 498 for BZ #209. TAMS/Gradient considered these results to be usable as estimated values.
TAMS/Gradient noted no other ITD performance problems for samples analyzed during the ecological study.

GC/ITD Calibration

The initial calibration criterion for acceptable quantitative data for GC/ITD analyses required RSD of the congener RRF to be less than 20%. For continuing calibration, the RRF for each congener must be within 20% of the mean calibration factor from the 5-level calibration at the beginning and end of each calibration sequence. For the invertebrate samples analyzed during the ecological study, TAMS/Gradient noted no significant GC/ITD calibration problems.

GC/ITD Internal Standard Performance

To demonstrate the stability of the ITD, internal standard performance criteria were monitored. Internal standard area counts must not vary by more than 30% from the most recent calibration or by more than 50% from the initial calibration. In addition, the absolute retention time of the internal standard must be within 10 seconds of the retention time in the most recent calibration, and ion abundance criteria must be met for chrysene-d_{12} and phenanthrene-d_{10}. The response for the internal standards for samples included in SDG 202834 exceeded the response criterion. CDM appropriately qualified the associated data as estimated and TAMS/Gradient considered these results to be usable as estimated values.

Confirmation by GC/ITD

CDM qualified all positive GC/ITD results that had signal/noise ratios of less than 3 as nondetect due to uncertainty in the identification. TAMS/Gradient considered these results to be usable as nondetects at the reported quantitation limits.

Aquatec analyzed approximately 10% of all samples analyzed by GC/ECD by GC/ITD to provide an additional mechanism to verify congener identification and, as a secondary objective, quantification of congeners. Since the ITD method was not designed to be a primary quantitative tool, some variations in quantitative results were expected. TAMS/Gradient considered quantitative differences between the GC/ITD and GC/ECD results less than a factor of five acceptable, while differences greater than five times were considered unacceptable. CDM qualified GC/ECD results that were detected at concentrations above the GC/ITD quantitation limit but that were not confirmed by GC/ITD with an “Q”. TAMS/Gradient converted all “Q” qualifiers to “JN” due to the potential reporting of false positive results. CDM qualified a total of 26 invertebrate results including one principal congener with “Q” qualifiers; TAMS/Gradient considered these results to be presumptively present. CDM qualified GC/ECD results that were not detected or were less than one-fifth the GC/ITD results with an “M”. TAMS/Gradient converted all “M” qualifiers to “R” as the nondetect

GC/ECD may be a false negative or the GC/ECD result may be significantly biased low. CDM qualified 59 invertebrate results, of which two were principal congener results, with "M" qualifiers; TAMS/Gradient considered these results to be unusable.

I.5.5.2 Precision

Matrix Spike/Matrix Spike Duplicate Comparison

The analysis of MS and MSD samples can also provide valuable information regarding method precision relative to laboratory performance and specific sample matrix. The advisory limit for RPD of spiked congeners in a MS/MSD pair is 40%, and for nonspiked congeners, the precision criterion is 40% RSD.

TAMS/Gradient noted MS/MSD precision exceedances in all SDGs. The RPDs for four spiked congeners in SDG 2B exceeded the criterion of 40% due to low recoveries in the MS analyses. Nonspiked congeners in each SDG, ranging from four to 36 results, exceeded the RPD criterion of 40%. CDM appropriately qualified the results for these congeners in the associated nonspiked samples. TAMS/Gradient considered these data to be usable as estimated values.

I.5.5.3 Representativeness

Field Duplicate Results

Analysis of field duplicate samples provides an indication of the overall precision of the sampling and analysis program. These analyses measure both field and laboratory precision; therefore, the results will likely have more variability than laboratory duplicates and MS/MSD samples, which only measure laboratory precision. Data validators used a 50% RPD criterion for evaluating field duplicate precision. Any congener precision greater than 50% RPD was qualified as estimated.

A total of four field duplicate samples were analyzed for the invertebrate samples. This represents a frequency of 4.8%, which is slightly below the 5% requirement stipulated in the Phase 2B SAP/QAPP. For the field duplicate samples reported in SDGs 2B, CDM qualified as estimated eleven results due to poor field duplicate precision. CDM appropriately qualified these results as estimated. TAMS/Gradient noted no significant problems with the remaining three field duplicate samples. Table I-4 summarizes the duplicate precision results for the 12 principal congeners for each field co-located sample.

I.5.5.4 Sensitivity

Blanks

An important data quality objective associated with the ecological study was to obtain detection limits as low as the analytical method could produce. One effect of this approach is to register low level blank contamination during the preparation and analysis of the invertebrates. As such, numerous congeners in all samples in all SDGs required blank contamination qualifications. TAMS/Gradient reviewed the distribution of blank contaminants and found most contamination associated with the monochlorobiphenyls, particularly with BZ #2 and BZ #3. Blank levels for BZ #2 ranged up to 52 ppb in extract and up to 78 ppb in extract for BZ #3. Since BZ #2 and BZ #3 are neither dechlorination products, major Aroclor components, nor principal congeners, TAMS/ Gradient did not consider this to be a serious data quality problem. CDM qualified eight principal congeners in the invertebrate samples due to blank contamination including: BZ #1 (12 results), BZ #4 (3 results), BZ #8 (10 results), BZ #10 (3 results), BZ #19 (1 results), BZ #118 (2 results), BZ #138 (1 results), and BZ #180 (4 results). TAMS/Gradient considered these results to be usable as nondetects. CDM qualified results during data validation with a "B", which indicated that the result was within five times of the blank action level. TAMS/Gradient converted all "B" qualified results in the database to nondetect results due to uncertainty in this detection. Table I-8 summarizes the congener detects changed to nondetects. TAMS/Gradient considered these results to be usable as nondetects at the reported quantitation limit.

Quantitation Limits

Evaluating bioaccumulation of PCB congeners in invertebrates necessitated obtaining low detection limits. TAMS/Gradient and Aquatec devised analytical methods to achieve lower detection limits. This, in part, required employing sample/extract cleanup methods to remove matrix interferences, and maximizing sample size when possible. For the invertebrate analyses, TAMS/Gradient defined optimum detection limits as 4-8 µg/Kg for monochlorobiphenyls, 2-4 µg/Kg for dichlorobiphenyls through hexachlorobiphenyls, and 2-4 µg/Kg for heptachlorobiphenyls through decachlorobiphenyl. Results of the MDL study supported these detection limits.

In general, achieving appropriate detection limits for the invertebrate samples was not a problem. Whenever TAMS/Gradient noted raised detection limits, the affected samples contained high organic content; specifically the presence of PCBs. The relative ratio of congeners detected within each high-concentration sample remained reasonably consistent, therefore the raised detection limit for nondetect congeners did not affect data usability.

I.5.6 Usability - Principal Congeners

The 12 principal target congeners employed in the ecological study are key to describing biological processes in the Hudson River sediment, fish, and invertebrate. The following synopsis will

provide data users with the strengths and weaknesses of the principal target congener data within the context of this study. All percentages recorded below are based on the collection of 93 sediment, 120 EPA-funded fish, 115 NOAA-funded fish, and 83 invertebrate sample analyses.

BZ #1. Overall, the reported results for BZ #1 did not met the data quality objective of 95% completeness the for ecological study, as 6.5% of the results were not usable for project objectives. BZ #1 results in 19 sediment (20%) samples were rejected due to dual GC column imprecision, seven invertebrate (8%) samples were rejected due to dual GC column imprecision or severe retention time shift, and one EPA-funded fish (<1%) sample was rejected as a potential false negative. BZ #1 results in six sediment and 12 invertebrate samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #4. Overall, the reported results for BZ #4 met the data quality objectives of the program. BZ #4 results in four invertebrate (5%) samples were rejected due to dual GC column imprecision or severe retention time shift and one sediment (1%) sample was rejected as a potential false negative. BZ #4 results in 16 sediment, 38 EPA-funded fish, and three invertebrate samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #8. Overall, the reported results for BZ #8 met the data quality objectives of the program. BZ #8 results in 10 sediment (11%) samples were rejected due to dual GC column imprecision and four invertebrate (5%) samples were rejected due to dual GC column imprecision or severe retention time shift. BZ #8 results in 17 sediment, 59 EPA funded fish, one NOAA-funded fish, and 10 invertebrate samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #10. Overall, the reported results for BZ #10 met the data quality objectives of the program. BZ #10 results in one sediment (1%) sample was rejected due to dual GC column imprecision and four invertebrate (5%) samples were rejected due to dual GC column imprecision or severe retention time shift. BZ #10 results in 16 sediment, 10 EPA-funded fish, and 10 invertebrate samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #18. Overall, the reported results for BZ #18 met the data quality objectives of the program. Sixteen sediment results for BZ #18 were initially rejected by the data validator due to poor dual column precision. The TAMS/Gradient Program QAO changed the rejection qualifier to a presumptively present qualifier based on the presence of BZ #18 in historical sediment samples containing PCBs, the consistent PCB congener pattern distribution present throughout the Hudson River sediment, and GC/ITD confirmational analysis on about 10% of the data. Nonetheless, BZ #18 results in five sediment (6%) samples were rejected due to dual GC column imprecision and one sediment (1%) sample was rejected as a potential false negative. BZ #18 results in seven EPA-funded fish were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #19. Overall, the reported results for BZ #19 met the data quality objectives of the program. BZ #19 results in three sediment (3%) samples were rejected due to dual GC column imprecision and six invertebrate (7%) samples were rejected due to dual GC column imprecision or severe retention time shift. BZ #19 results in 32 sediment and one invertebrate sample were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #28. Overall, the reported results for BZ #28 met the data quality objectives of the program. No BZ #28 results were rejected. BZ #28 results in 14 sediment and four EPA-funded fish samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered all BZ # 28 results to be usable as reported in the project database.

BZ #52. Overall, the reported results for BZ #52 met the data quality objectives of the program. No BZ #52 results were rejected. BZ #52 results in four EPA-funded fish and three NOAA-funded fish samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered all BZ # 52 results to be usable as reported in the project database.

BZ #101. Data users should be aware that BZ #101 always coeluted with BZ #90 (on both GC columns), and therefore was always reported with BZ #90. BZ #101 results in one sediment (1%) sample was rejected due to dual GC column imprecision and one sediment (1%) sample was rejected as a potential false negative. BZ #101 results in two NOAA-funded fish samples were considered to be not detected as the potential

for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #118. Overall, the reported results for BZ #118 met the data quality objectives of the program. No BZ #118 results were rejected. BZ #118 results in two EPA-funded fish, two NOAA-funded fish, and two invertebrate samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered all BZ #118 results to be usable as reported in the project database.

BZ #138. Overall, the reported results for BZ #138 met the data quality objectives of the program. BZ #138 results in three sediment (3%) samples were rejected due to dual GC column imprecision and one sediment (1%) and two invertebrate (2%) samples were rejected due to potential false negatives. BZ #138 results in two NOAA-funded fish, and one invertebrate sample were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

BZ #180. Overall, the reported results for BZ #180 met the data quality objectives of the program. BZ #180 results in 13 invertebrate (16%) samples were rejected due to dual GC column imprecision or severe retention time shift. BZ #180 results in eight sediment, 19 EPA funded-fish, one NOAA-funded fish, and four invertebrate samples were considered to be not detected as the potential for false positives exists as indicated by blank contamination. TAMS/Gradient considered the rejected results to be unusable and considered all remaining results to be usable as reported in the project database.

I.5.7 TEQ Congeners Data Usability

One the 12 principal congeners (BZ #118) examined in the preceding sections is a Toxic Equivalents (TEQ) congeners (described in subchapter 4.1.3). Of the remaining 11 TEQ congeners, two (BZ #169 and 114) were "non-target" congeners, one (BZ #156) is an "additional calibrated congener," and seven (BZ #77, 126, 105, 123, 157, 167, and 189 as well as 118) are target congeners, and one (BZ #81) was not analyzed or reported. Quantitation of the two non-target congeners is estimated in all samples, since no calibration standards were analyzed for these two congeners.

While this appendix focuses on the 12 principal congeners, all reported congener data were reviewed, including the 11 TEQ congeners that were analyzed. Except as noted within this memo, there were no significant issues associated with TEQ congeners in ecological samples.

Four of the TEQ congeners (BZ #77, 105, 118, and 126) were part of the suite of matrix spike compounds. No issues specific to any of these congeners were noted; although it was noted that MS/MSD recoveries were high across the board in one of the invertebrate sample delivery groups.

Two of the TEQ congeners, BZ #77 and BZ #189 had more than 10% of the data in a single matrix rejected. Thirteen percent of the BZ #77 sediment data were rejected due to dual column imprecision (Table I-9); overall about 4.1% of the BZ #77 data were rejected from all matrices. Twelve percent of the BZ #189 data were rejected in the invertebrate samples (Table I-10) for the same reason; the overall rejection rate for BZ #189 was 2.4%. No other TEQ congeners were rejected in any of the three ecological media at frequencies of 10% or more, and the overall rejection rate for TEQ congeners in the four media analyzed was less than 5%, that is 95% or better completeness (Tables I-9 to I-12); ranging from 0.4% rejected TEQ congeners in the NOAA-funded fish data (Table I-12) to 3.1% rejected in the invertebrate data (Table I-10).

Results for BZ #118 were qualified in a small percentage (less than 2%) of the fish samples (both the EPA and NOAA fish) due to blank contamination, and also in two of the invertebrate samples. No other TEQ congeners were qualified in any of the other samples for blank contamination.

Other than noted above, there were no issues associated with TEQ congener data quality evident from the data usability report. It is noted that, overall, a high percentage of the ecological data (62%) were qualified as estimated; however, these data were considered usable for the purposes of the ecological risk assessment. Rejected data, which are not usable, amounted to about 1.6% of the total congener data generated for the ecological program. Review of the data for the TEQ congeners indicates a similar pattern, with about 64.6% of the data (for the 11 TEQ congeners analyzed) qualified as estimated, and about 1.7% rejected.

BZ #169 was detected in only one of the 411 analyses; and this one detection was the only BZ#169 data point which was qualified as estimated (0.2% of the total data points). However, for the other 10 TEQ congeners analyzed, the percentage of qualified data (the sum of estimated and rejected data) was much higher, ranging from about 49% (for BZ #123 and 126) to over 90% (93.4% of BZ #77 data, and 94.6% of BZ #105 data).

The ecological sediment data generally had the least qualifications, as slightly less than 44% of the sediment was qualified (41.1% estimated, and 2.8% rejected). Slightly over 70% of the data for each of the other three media were qualified: 72.7% of the EPA-funded fish data; 71.7% of the NOAA-funded fish data; and 74.7% of the invertebrate data.

I.6 Conclusions

The analytical chemistry program implemented by TAMS/Gradient for the Hudson River ecological study was extremely sophisticated, requiring the use of state-of-the-art GC methodology. A total of 93 sediment, 120 EPA funded fish, 115 NOAA funded fish, and 83 invertebrate samples were analyzed for 108 target and up to 38 non-target congeners. Considering the complexity of the program, TAMS/Gradient considers the outcome of the analytical chemistry program to have been successful.

Summaries of the number of qualifiers applied to each PCB congener presented by media are tabulated in Tables I-9 through I-12. For the ecological study, a total of 59,063 congener measurements were recorded, of which 925 values (1.6%) were rejected. A 98.4% completeness rate was achieved for the overall program, which successfully exceeded the 95% completeness requirement. The only principal congener which did not meet the completeness requirement was BZ #4 (93.5% completeness), however, this did not impair the overall integrity of the program.

A majority of all congener results (both detects and nondetects) were qualified as estimated or estimated and presumptively present (62%). Again, the main reason for most of the qualifications was detection at concentrations be
low the calibrated quantitation limit and/or exceedances in the dual GC column precision criteria. Numerous congeners for nearly all SDGs had calculated concentrations on each GC column which differed by more than 25%, but less than 50%, which warranted qualification as estimated values. With the level of background organic material present in Hudson sediments and in tissue samples, resultant interferences, particularly for congeners at low concentrations, likely caused these differences between the GC columns. Other problems contributing to data qualification included missed holding times, poor field duplicate precision, problems with MS/MSD recoveries and precision, and some GC/ECD calibration problems. Data users should consider all detect and nondetect results which were estimated to be usable relative to the data quality objectives of the program.

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TABLE I -1
PHASE 2 TARGET AND NON-TARGET PCB CONGENERS USED IN ANALYSES

Congener Number	Homologue Group	Congener Name	Target[a]
BZ #1	Mono	2-Chlorobiphenyl	Yes
BZ #2	Mono	3-Chlorobiphenyl	Yes
BZ #3	Mono	4-Chlorobiphenyl	Yes
BZ #4	Di	2,2N-Dichlorobiphenyl	Yes
BZ #5	Di	2,3-Dichlorobiphenyl	Yes
BZ #6	Di	2,3N-Dichlorobiphenyl	Yes
BZ #7	Di	2,4-Dichlorobiphenyl	Yes
BZ #8	Di	2,4N-Dichlorobiphenyl	Yes
BZ #9	Di	2,5-Dichlorobiphenyl	Yes
BZ #10	Di	2,6-Dichlorobiphenyl	Yes
BZ #12	Di	3,4-Dichlorobiphenyl	Yes
BZ #15	Di	4,4N-Dichlorobiphenyl	Yes
BZ #16	Tri	2,2N,3-Trichlorobiphenyl	Yes
BZ #17	Tri	2,2N,4-Trichlorobiphenyl	No - Cal
BZ #18	Tri	2,2N,5-Trichlorobiphenyl	Yes
BZ #19	Tri	2,2N,6-Trichlorobiphenyl	Yes
BZ #20	Tri	2,3,3N-Trichlorobiphenyl	No - Cal
BZ #21	Tri	2,3,4-Trichlorobiphenyl	No
BZ #22	Tri	2,3,4N-Trichlorobiphenyl	Yes
BZ #23	Tri	2,3,5-Trichlorobiphenyl	No
BZ #24	Tri	2,3,6-Trichlorobiphenyl	No
BZ #25	Tri	2,3N,4-Trichlorobiphenyl	Yes
BZ #26	Tri	2,3N,5-Trichlorobiphenyl	Yes
BZ #27	Tri	2,3N,6-Trichlorobiphenyl	Yes
BZ #28	Tri	2,4,4N-Trichlorobiphenyl	Yes
BZ #29	Tri	2,4,5-Trichlorobiphenyl	Yes
BZ #31	Tri	2,4N,5-Trichlorobiphenyl	Yes
BZ #32	Tri	2,4N,6-Trichlorobiphenyl	No
BZ #33	Tri	2N,3,4-Trichlorobiphenyl	No - Cal
BZ #34	Tri	2N,3,5-Trichlorobiphenyl	No
BZ #35	Tri	3,3',4-Trichlorobiphenyl	No
BZ #37	Tri	3,4,4N-Trichlorobiphenyl	Yes
BZ #39	Tri	3,4',5-Trichlorobiphenyl	No
BZ #40	Tetra	2,2N,3,3N-Tetrachlorobiphenyl	Yes
BZ #41	Tetra	2,2N,3,4-Tetrachlorobiphenyl	Yes
BZ #42	Tetra	2,2N,3,4N-Tetrachlorobiphenyl	No - Cal
BZ #44	Tetra	2,2N,3,5N-Tetrachlorobiphenyl	Yes
BZ #45	Tetra	2,2N,3,6-Tetrachlorobiphenyl	No - Cal
BZ #46	Tetra	2,2',3,6'-Tetrachlorobiphenyl	No
BZ #47	Tetra	2,2N,4,4N-Tetrachlorobiphenyl	Yes
BZ #48	Tetra	2,2N,4,5-Tetrachlorobiphenyl	No
BZ #49	Tetra	2,2N,4,5N-Tetrachlorobiphenyl	Yes
BZ #51	Tetra	2,2N,4,6N-Tetrachlorobiphenyl	No
BZ #52	Tetra	2,2N,5,5N-Tetrachlorobiphenyl	Yes

TABLE I-1
PHASE 2 TARGET AND NON-TARGET PCB CONGENERS USED IN ANALYSES
(CONTINUED)

Congener Number	Homologue Group	Congener Name	Target[a]
BZ #53	Tetra	2,2N,5,6N-Tetrachlorobiphenyl	Yes
BZ #56	Tetra	2,3,3N,4N-Tetrachlorobiphenyl	Yes
BZ #57	Tetra	2,3,3',5-Tetrachlorobiphenyl	No
BZ #58	Tetra	2,3,3N,5N-Tetrachlorobiphenyl	No
BZ #59	Tetra	2,3,3',6-Tetrachlorobiphenyl	No - Cal
BZ #60	Tetra	2,3,4,4N-Tetrachlorobiphenyl	No
BZ #63	Tetra	2,3,4N,5-Tetrachlorobiphenyl	No
BZ #64	Tetra	2,3,4N,6-Tetrachlorobiphenyl	No
BZ #66	Tetra	2,3N,4,4N-Tetrachlorobiphenyl	Yes
BZ #67	Tetra	2,3N,4,5-Tetrachlorobiphenyl	No
BZ #69	Tetra	2,3N,4,6-Tetrachlorobiphenyl	No
BZ #70	Tetra	2,3N,4N,5-Tetrachlorobiphenyl	Yes
BZ #72	Tetra	2,3',5,5'-Tetrachlorobiphenyl	No - Cal
BZ #74	Tetra	2,4,4N,5-Tetrachlorobiphenyl	No - Cal
BZ #75	Tetra	2,4,4N,6-Tetrachlorobiphenyl	Yes
BZ #77	Tetra	3,3N,4,4N-Tetrachlorobiphenyl	Yes
BZ #82	Penta	2,2N,3,3N,4-Pentachlorobiphenyl	Yes
BZ #83	Penta	2,2N,3,3N,5-Pentachlorobiphenyl	Yes
BZ #84	Penta	2,2N,3,3N,6-Pentachlorobiphenyl	Yes
BZ #85	Penta	2,2N,3,4,4N-Pentachlorobiphenyl	Yes
BZ #87	Penta	2,2N,3,4,5N-Pentachlorobiphenyl	Yes
BZ #88	Penta	2,2N,3,4,6-Pentachlorobiphenyl	No
BZ #90	Penta	2,2N,3,4N,5-Pentachlorobiphenyl	No
BZ #91	Penta	2,2N,3,4N,6-Pentachlorobiphenyl	Yes
BZ #92	Penta	2,2N,3,5,5N-Pentachlorobiphenyl	Yes
BZ #95	Penta	2,2N,3,5N,6-Pentachlorobiphenyl	Yes
BZ #96	Penta	2,2N,3,6,6N-Pentachlorobiphenyl	No
BZ #97	Penta	2,2N,3N,4,5-Pentachlorobiphenyl	Yes
BZ #99	Penta	2,2N,4,4N,5-Pentachlorobiphenyl	Yes
BZ #100	Penta	2,2',4,4',6-Pentachlorobiphenyl	No
BZ #101	Penta	2,2N,4,5,5N-Pentachlorobiphenyl	Yes
BZ #104	Penta	2,2',4,6,6'-Pentachlorobiphenyl	No
BZ #105	Penta	2,3,3N,4,4N-Pentachlorobiphenyl	Yes
BZ #107	Penta	2,3,3N,4,5N-Pentachlorobiphenyl	Yes
BZ #110	Penta	2,3,3N,4N,6-Pentachlorobiphenyl	No - Cal
BZ #114	Penta	2,3,4,4N,5-Pentachlorobiphenyl	No
BZ #115	Penta	2,3,4,4N,6-Pentachlorobiphenyl	Yes
BZ #118	Penta	2,3N,4,4N,5-Pentachlorobiphenyl	Yes
BZ #119	Penta	2,3N,4,4N,6-Pentachlorobiphenyl	Yes
BZ #122	Penta	2N,3,3N,4,5-Pentachlorobiphenyl	Yes

TABLE I-1
PHASE 2 TARGET AND NON-TARGET PCB CONGENERS USED IN ANALYSES
(CONTINUED)

Congener Number	Homologue Group	Congener Name	Target[a]
BZ #123	Penta	2N,3,4,4N,5-Pentachlorobiphenyl	Yes
BZ #126	Penta	3,3N,4,4N,5-Pentachlorobiphenyl	Yes
BZ #128	Hexa	2,2N,3,3N,4,4N-Hexachlorobiphenyl	Yes
BZ #129	Hexa	2,2N,3,3N,4,5-Hexachlorobiphenyl	Yes
BZ #130	Hexa	2,2',3,3',4,5'-Hexachlorobiphenyl	No
BZ #131	Hexa	2,2',3,3',4,6-Hexachlorobiphenyl	No
BZ #132	Hexa	2,2',3,3',4,6'-Hexachlorobiphenyl	No
BZ #134	Hexa	2,2',3,3',5,6-Hexachlorobiphenyl	No
BZ #135	Hexa	2,2N,3,3N,5,6N-Hexachlorobiphenyl	No - Cal
BZ #136	Hexa	2,2N,3,3N,6,6N-Hexachlorobiphenyl	Yes
BZ #137	Hexa	2,2N,3,4,4N,5-Hexachlorobiphenyl	Yes
BZ #138	Hexa	2,2N,3,4,4N,5N-Hexachlorobiphenyl	Yes
BZ #140	Hexa	2,2N,3,4,4N,6N-Hexachlorobiphenyl	No
BZ #141	Hexa	2,2N,3,4,5,5N-Hexachlorobiphenyl	Yes
BZ #143	Hexa	2,2N,3,4,5,6-Hexachlorobiphenyl	No - Cal
BZ #144	Hexa	2,2N,3,4,5N,6-Hexachlorobiphenyl	No
BZ #146	Hexa	2,2N,3,4N,5,5N-Hexachlorobiphenyl	No
BZ #149	Hexa	2,2N,3,4N,5N,6-Hexachlorobiphenyl	Yes
BZ #151	Hexa	2,2N,3,5,5N,6-Hexachlorobiphenyl	Yes
BZ #153	Hexa	2,2N,4,4N,5,5N-Hexachlorobiphenyl	Yes
BZ #156	Hexa	2,3,3N,4,4N,5-Hexachlorobiphenyl	No - Cal
BZ #157	Hexa	2,3,3N,4,4N,5N-Hexachlorobiphenyl	Yes
BZ #158	Hexa	2,3,3N,4,4N,6-Hexachlorobiphenyl	Yes
BZ #160	Hexa	2,3,3N,4,5,6-Hexachlorobiphenyl	No
BZ #162	Hexa	2,3,3',4',5,5'-Hexachlorobiphenyl	No
BZ #165	Hexa	2,3,3',5,5',6-Hexachlorobiphenyl	No - Cal
BZ #167	Hexa	2,3N,4,4N,5,5N-Hexachlorobiphenyl	Yes
BZ #168	Hexa	2,3N,4,4N,5N,6-Hexachlorobiphenyl	No - Cal
BZ #169	Hexa	3,3N,4,4N,5,5N-Hexachlorobiphenyl	No
BZ #170	Hepta	2,2N,3,3N,4,4N,5-Heptachlorobiphenyl	Yes
BZ #171	Hepta	2,2N,3,3N,4,4N,6-Heptachlorobiphenyl	Yes
BZ #172	Hepta	2,2N,3,3N,4,5,5N-Heptachlorobiphenyl	No
BZ #173	Hepta	2,2',3,3',4,5,6-Heptachlorobiphenyl	No
BZ #174	Hepta	2,2N,3,3N,4,5,6N-Heptachlorobiphenyl	No - Cal
BZ #175	Hepta	2,2N,3,3N,4,5N,6-Heptachlorobiphenyl	No
BZ #176	Hepta	2,2',3,3',4,6,6'-Heptachlorobiphenyl	No - Cal
BZ #177	Hepta	2,2N,3,3N,4N,5,6-Heptachlorobiphenyl	Yes
BZ #178	Hepta	2,2N,3,3N,5,5N,6-Heptachlorobiphenyl	No - Cal
BZ #179	Hepta	2,2',3,3',5,6,6'-Heptachlorobiphenyl	No - Cal
BZ #180	Hepta	2,2N,3,4,4N,5,5N-Heptachlorobiphenyl	Yes
BZ #183	Hepta	2,2N,3,4,4N,5N,6-Heptachlorobiphenyl	Yes
BZ #184	Hepta	2,2N,3,4,4N,6,6N-Heptachlorobiphenyl	No
BZ #185	Hepta	2,2N,3,4,5,5N,6-Heptachlorobiphenyl	Yes
BZ #187	Hepta	2,2N,3,4N,5,5N,6-Heptachlorobiphenyl	Yes

TABLE I-1
PHASE 2 TARGET AND NON-TARGET PCB CONGENERS USED IN ANALYSES
(CONTINUED)

Congener Number	Homologue Group	Congener Name	Target[a]
BZ #189	Hepta	2,3,3N,4,4N,5,5N-Heptachlorobiphenyl	Yes
BZ #190	Hepta	2,3,3N,4,4N,5,6-Heptachlorobiphenyl	Yes
BZ #191	Hepta	2,3,3N,4,4N,5N,6-Heptachlorobiphenyl	Yes
BZ #192	Hepta	2,3,3N,4,5,5N,6-Heptachlorobiphenyl	No
BZ #193	Hepta	2,3,3N,4N,5,5N,6-Heptachlorobiphenyl	Yes
BZ #194	Octa	2,2N,3,3N,4,4N,5,5N-Octachlorobiphenyl	Yes
BZ #195	Octa	2,2N,3,3N,4,4N,5,6-Octachlorobiphenyl	Yes
BZ #196	Octa	2,2N,3,3N,4,4N,5N,6-Octachlorobiphenyl	Yes
BZ #197	Octa	2,2N,3,3N,4,4N,6,6N-Octachlorobiphenyl	No
BZ #198	Octa	2,2N,3,3N,4,5,5N,6-Octachlorobiphenyl	Yes
BZ #199	Octa	2,2N,3,3N,4,5,6,6N-Octachlorobiphenyl	Yes
BZ #200	Octa	2,2N,3,3N,4,5N,6,6N-Octachlorobiphenyl	Yes
BZ #201	Octa	2,2N,3,3N,4N,5,5N,6-Octachlorobiphenyl	Yes
BZ #202	Octa	2,2N,3,3N,5,5N,6,6N-Octachlorobiphenyl	Yes
BZ #203	Octa	2,2N,3,4,4N,5,5N,6-Octachlorobiphenyl	No
BZ #205	Octa	2,3,3N,4,4N,5,5N,6-Octachlorobiphenyl	Yes
BZ #206	Nona	2,2N,3,3N,4,4N,5,5N,6-Nonachlorobiphenyl	Yes
BZ #207	Nona	2,2N,3,3N,4,4N,5,6,6N-Nonachlorobiphenyl	Yes
BZ #208	Nona	2,2N,3,3N,4,5,5N,6,6N-Nonachlorobiphenyl	Yes
BZ #209	Deca	2,2N,3,3N,4,4N,5,5N,6,6N-Decachlorobiphenyl	Yes

Homologue Group	Congener Ratio[b]
Mono	3:3
Di	9:12
Tri	18:24
Tetra	23:42
Penta	23:46
Hexa	19:42
Hepta	16:24
Octa	11:12
Nona	3:3
Deca	1:1
Sum	126:209

Notes:

[a]Yes: Target; No: Non-target; No - Cal: Calibrated non-target.
[b]Ratio of number of congeners used to total number of congeners in homologue group.

TABLE I-2
DATA QUALIFICATION CODES

Source of Qualifier	Definition of Qualifier Code	Data Validation/ Assessment Qualifier Code	Database Qualifier Code
Laboratory	Compound not detected above reporting limit of 0.1 ppb in extract for all PCB congeners (0.5 ppb in extract for the monochlorinated biphenyls). The reported value is the quantitation limit (QL).	U	U
Laboratory	Compound detected above reporting limit, but below calibration range. This qualifier is applied to any positive result that is less than the lowest calibration standard. The reported result is an estimated value, due to uncertainty in the reported value near the quantitation limit.	J	J
Laboratory	Compound concentration exceeds the calibration range. This qualifier is applied to any positive result that exceeds the calibration range. The laboratory may report some congeners with concentrations up to twice the concentration in the highest calibration standard, in order to report some very low concentrations and low quantitation limits. The reported result is an estimated value, due to uncertainty in the quantitation above the calibrated range of the instrument.	E	J
Laboratory	Specific column result used for quantitation due to confirmation column coelution. This qualifier designates congeners whose results are always quantitated from a specific column due to coelution with congeners or surrogates on the other column. The reported result should be considered an estimated value, due to inability to confirm the concentration of the result because of coelution on the other column. The S qualifier precludes the P qualifier since a %Difference (%D) between columns is excepted to be greater than 25% due to coelution on one column.	S	J
Laboratory	Tentative identification, specific column result used with no confirmation information. This qualifier designates congeners which could not be confirmed due to an interferant (or surrogate) peak, however, there is good reason to believe its presence. The reported value should be considered an estimated value, due to inability to confirm reported concentrations.	T	JN
Laboratory	Estimated concentration due to coelution on both columns. This qualifier designates congeners which coelute with congeners or surrogates on both analytical columns. In order to report a concentration for the congener of interest, the concentrations of the coeluting congeners are subtracted from it. Therefore, the reported result is an estimated value.	X	J

TABLE I-2
DATA QUALIFICATION CODES
(Continued)

Source of Qualifier	Definition of Qualifier Code	Data Validation/ Assessment Qualifier Code	Database Qualifier Code
Laboratory	Confirmation column result exceeds reported result by more than 25%. This qualifier is applied to a congener result if the concentration on the quantitation and confirmation columns exceed the percent difference (%D) criteria of 25. The reported result is an estimated value, due to poor precision of results between columns.	P	J
Laboratory	Specific column or estimated result exceeds confirmation result by more than 25% despite expected confirmation coelution. This qualifier is applied to a congener result if the result from the quantitation column exceeds the confirmation result by more than 25 %D, even though the confirmation column result was expected to be greater due to coelution on the confirmation column. Therefore, the reported result should be considered an estimated value, bias high.	H	J
Data Validation	Estimated data due to exceeded quality control criteria. This qualifier is applied to data if problems with data quality are noted and estimation of the data is deemed necessary. Justification for qualification are given in the data validation report.	G	J
Data Validation	Reject data due to exceeded quality control criteria. This qualifier is applied to data if serious problems with data quality are noted and rejection of the data is deemed necessary. Justification for rejection of data are given in the data validation report. Rejected data are not usable and do not meet the data quality objectives of the program. No numerical value is reported.	R	R
Data Validation	The compound was also detected in associated blank(s). This qualifier is applied to GC/ECD results that are within five times the concentration detected in the associated blanks. The reported result may be considered not detected; a false positive is suspected due to blank contamination.	B	U
Data Validation	GC/ECD result at concentration within GC/ITD calibration range, but not confirmed by GC/ITD analysis. This qualifier is applied to GC/ECD results that are not confirmed by GC/ITD analysis, even though the results are at sufficient concentration to be detected by GC/ITD. The reported result is suspect as it may be a false positive.	Q	JN

TABLE I-2
DATA QUALIFICATION CODES
(Continued)

Source of Qualifier	Definition of Qualifier Code	Data Validation/ Assessment Qualifier Code	Database Qualifier Code
Data Validation	Presumptive evidence for the presence of a material. This qualifier is applied to GC/ECD results that exceeded the compound identification criteria. The reported result is suspect as it may be a false positive.	N	N
Data Management	Results generated by decoupling BZ #4 and 10 using regression analysis.	L	J
Data Management	Results updated by Aquatec due to revisions in GC column performance.	K	--
Data Management	Results requalified by QAO due to decisions made during data usability assessment.	Y	J

TABLE I-3
ECOLOGICAL SEDIMENT PCB FIELD DUPLICATE SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

TAMS ID	Parameter	Sample Result and Qualifier		Duplicate Result and Qualifier		RPD (%)
EC-S02-0005	BZ#1	780	J	1800	J	-79
EC-S02-0005	BZ#4	840.336	JN	2228.94	JN	-90
EC-S02-0005	BZ#8	673	JN	1990	JN	-99
EC-S02-0005	BZ#10	134.48	JN	356.7	JN	-90
EC-S02-0005	BZ#18	608	J	1200	J	-65
EC-S02-0005	BZ#19	305	J	509	U	-50
EC-S02-0005	BZ#28	1030	JN	1350	JN	-27
EC-S02-0005	BZ#52	494	J	751	J	-41
EC-S02-0005	BZ#101 with BZ#[9(	165	J	208	J	-23
EC-S02-0005	BZ#118	146		162		-10
EC-S02-0005	BZ#138	78.5	J	72.4	J	8
EC-S02-0005	BZ#180	18.5	J	109	U	-142
EC-S03-0005	BZ#1	449		460		-2
EC-S03-0005	BZ#4	456.036	J	508.557	J	-11
EC-S03-0005	BZ#8	336	JN	342	JN	-2
EC-S03-0005	BZ#10	72.98	J	81.385	J	-11
EC-S03-0005	BZ#18	326		298		9
EC-S03-0005	BZ#19	138		141		-2
EC-S03-0005	BZ#28	639	JN	645	JN	-1
EC-S03-0005	BZ#52	259	J	260	J	0
EC-S03-0005	BZ#101 with BZ#[9(	98.8	J	95.4	J	4
EC-S03-0005	BZ#118	92.9		90.9		2
EC-S03-0005	BZ#138	45.1	J	42.7	J	5
EC-S03-0005	BZ#180	9.51	J	9.1	J	4
EC-S06-0005	BZ#1	933	J	2090	J	-77
EC-S06-0005	BZ#4	891.576	J	2139.27	J	-82
EC-S06-0005	BZ#8	581	JN	1430	JN	-84
EC-S06-0005	BZ#10	142.68	J	342.35	J	-82
EC-S06-0005	BZ#18	248	J	655	J	-90
EC-S06-0005	BZ#19	205	J	492	J	-82
EC-S06-0005	BZ#28	525	JN	1180	J	-77
EC-S06-0005	BZ#52	233	J	532	J	-78
EC-S06-0005	BZ#101 with BZ#[9(	70.9	J	154	J	-74
EC-S06-0005	BZ#118	61.2	J	139	J	-78
EC-S06-0005	BZ#138	35.2	J	69.2	J	-65
EC-S06-0005	BZ#180	8.09	J	18.1	J	-76
EC-S13-0005	BZ#1	5.88	U	3.39	R	NC
EC-S13-0005	BZ#4	17.8	U	20	U	NC
EC-S13-0005	BZ#8	28.1	U	33.6	U	NC
EC-S13-0005	BZ#10	17.8	U	20	U	NC
EC-S13-0005	BZ#18	22.5	JN	24.9	R	NC
EC-S13-0005	BZ#19	5.36	U	5.46	U	NC
EC-S13-0005	BZ#28	74.7		85.5	J	-13
EC-S13-0005	BZ#52	30.5	J	30.7	J	-1
EC-S13-0005	BZ#101 with BZ#[9(	13.4	J	13	J	3
EC-S13-0005	BZ#118	12.7		13.1	J	-3
EC-S13-0005	BZ#138	10.3	J	10.1	J	2
EC-S13-0005	BZ#180	3.87	U	4.48	U	NC
EC-S15-0005	BZ#1	4.7	R	2.98	R	NC

TABLE I-3
ECOLOGICAL SEDIMENT PCB FIELD DUPLICATE SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

TAMS ID	Parameter	Sample Result and Qualifier		Duplicate Result and Qualifier		RPD (%)
EC-S15-0005	BZ#4	9.36	U	5	U	NC
EC-S15-0005	BZ#8	26.8	J	19.1	J	34
EC-S15-0005	BZ#10	9.36	U	5	U	NC
EC-S15-0005	BZ#18	1.11	U	15.1	J	-173
EC-S15-0005	BZ#19	4.68	U	2.8	U	NC
EC-S15-0005	BZ#28	79.3		55		36
EC-S15-0005	BZ#52	28.8	J	19.3	J	40
EC-S15-0005	BZ#101 with BZ#[9(	12	J	8.82	J	31
EC-S15-0005	BZ#118	11.1		8.19		30
EC-S15-0005	BZ#138	8.12	J	6.03	J	30
EC-S15-0005	BZ#180	2.67	J	2.33	J	14
EC-S17-0005	BZ#1	7.46	R	1.72	R	NC
EC-S17-0005	BZ#4	13.4505	J	7.0967	J	62
EC-S17-0005	BZ#8	42.7	J	10.6	U	120
EC-S17-0005	BZ#10	2.1525	J	1.1357	J	62
EC-S17-0005	BZ#18	1.37	U	1.34	U	NC
EC-S17-0005	BZ#19	1.37	U	1.8	U	NC
EC-S17-0005	BZ#28	151	J	29.7	J	134
EC-S17-0005	BZ#52	48.4	J	13.3	J	114
EC-S17-0005	BZ#101 with BZ#[9(	27.2	J	9.1	J	100
EC-S17-0005	BZ#118	24.8	J	6.71	J	115
EC-S17-0005	BZ#138	18.9	J	6.22	J	101
EC-S17-0005	BZ#180	7.93	J	2.7	J	98

Note: all concentrations are given in ug/Kg DW

TABLE I-4
INVERTEBRATE PCB FIELD DUPLICATE SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

TAMS ID	Species	Parameter	Sample Result and Qualifier (ppb)	Duplicate Result and Qualifier (ppb)	RPD (%)
EC-B03-0004	ST	BZ#1	42.2	54.7 U	-26
EC-B03-0004	ST	BZ#4	301.035 J	359.961 J	-18
EC-B03-0004	ST	BZ#8	72.8 J	85.2 J	-16
EC-B03-0004	ST	BZ#10	48.175 J	57.605 J	-18
EC-B03-0004	ST	BZ#18	177	226	-24
EC-B03-0004	ST	BZ#19	103 J	138 J	-29
EC-B03-0004	ST	BZ#28	371	395	-6
EC-B03-0004	ST	BZ#52	240	292	-20
EC-B03-0004	ST	BZ#101 with BZ#[90]	111 J	120 J	-8
EC-B03-0004	ST	BZ#118	112 J	115 J	-3
EC-B03-0004	ST	BZ#138	41.7 J	44.5 J	-6
EC-B03-0004	ST	BZ#180	10	9.61	4
EC-B06-0003	ST	BZ#1	247 J	262 U	-6
EC-B06-0003	ST	BZ#4	1720 J	995 J	53
EC-B06-0003	ST	BZ#8	578 J	314 J	59
EC-B06-0003	ST	BZ#10	394 J	256 J	42
EC-B06-0003	ST	BZ#18	784 J	513 J	42
EC-B06-0003	ST	BZ#19	1030 J	525 J	65
EC-B06-0003	ST	BZ#28	984 J	719 J	31
EC-B06-0003	ST	BZ#52	1040 J	705 J	38
EC-B06-0003	ST	BZ#101 with BZ#[90]	300 J	255 J	16
EC-B06-0003	ST	BZ#118	238 J	186 J	25
EC-B06-0003	ST	BZ#138	21 J	90.8 J	-125
EC-B06-0003	ST	BZ#180	23.5 J	18.6 J	23
EC-B07-0005	ST	BZ#1	66.5 J	57.7 J	14
EC-B07-0005	ST	BZ#4	241 J	209 J	14
EC-B07-0005	ST	BZ#8	57.2 J	57.7 J	-1
EC-B07-0005	ST	BZ#10	71.9 J	59.2 J	19
EC-B07-0005	ST	BZ#18	117 J	110 J	6
EC-B07-0005	ST	BZ#19	164 J	134 J	20
EC-B07-0005	ST	BZ#28	204 J	199 J	2
EC-B07-0005	ST	BZ#52	269 J	238 J	12
EC-B07-0005	ST	BZ#101 with BZ#[90]	96.5 J	81 J	17
EC-B07-0005	ST	BZ#118	74.7 J	65.9 J	13
EC-B07-0005	ST	BZ#138	34.9 J	29.9 J	15
EC-B07-0005	ST	BZ#180	6.78 JN	6.75 J	0
EC-B18-0001	IS	BZ#1	17.6 U	17.5 U	NC
EC-B18-0001	IS	BZ#4	3.53 U	3.5 U	NC
EC-B18-0001	IS	BZ#8	3.53 U	2.46 U	NC
EC-B18-0001	IS	BZ#10	3.53 U	3.5 U	NC
EC-B18-0001	IS	BZ#18	3.53 U	3.5 U	NC
EC-B18-0001	IS	BZ#19	3.53 U	3.5 U	NC
EC-B18-0001	IS	BZ#28	2.18 J	2.01 J	8
EC-B18-0001	IS	BZ#52	3.41 J	3.47 J	-2
EC-B18-0001	IS	BZ#101 with BZ#[90]	5.97 J	5.08 J	16
EC-B18-0001	IS	BZ#118	5.02 J	4.23 J	17
EC-B18-0001	IS	BZ#138	13.9 J	10.1 J	32
EC-B18-0001	IS	BZ#180	15.4 J	15.2 J	1

Note: all concentrations are given in ug/Kg WW

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TABLE I-5
PCB DETECTS CHANGED TO NON-DETECTS
ECOLOGICAL SEDIMENT SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	Number of results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#1	6	93	6
BZ#2	22	93	24
BZ#3	14	93	15
BZ#4	16	93	17
BZ#7	16	93	17
BZ#8	17	93	18
BZ#9	28	93	30
BZ#10	16	93	17
BZ#16	23	93	25
BZ#19	32	93	34
BZ#20	16	93	17
BZ#22	14	93	15
BZ#27 & BZ#[24]	14	93	15
BZ#28	14	93	15
BZ#31	3	93	3
BZ#37	1	93	1
BZ#42	1	93	1
BZ#45	44	93	47
BZ#53	12	93	13
BZ#56	5	93	5
BZ#72	1	93	1
BZ#74	32	93	34
BZ#75	1	93	1
BZ#82	17	93	18
BZ#83	3	93	3
BZ#85	1	93	1
BZ#87	20	93	22
BZ#92	2	93	2
BZ#97	6	93	6
BZ#105 & BZ#[168	5	93	5
BZ#119	29	93	31
BZ#122	21	93	23
BZ#123	18	93	19
BZ#128	2	93	2
BZ#129	32	93	34
BZ#137	20	93	22
BZ#141	7	93	8
BZ#153	2	93	2
BZ#156	4	93	4
BZ#157	1	93	1
BZ#158	21	93	23
BZ#165	6	93	6
BZ#167	24	93	26
BZ#170	12	93	13
BZ#171	21	93	23
BZ#174	6	93	6
BZ#176	14	93	15

TABLE I-5
PCB DETECTS CHANGED TO NON-DETECTS
ECOLOGICAL SEDIMENT SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	Number of results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#177	11	93	12
BZ#178	15	93	16
BZ#179	20	93	22
BZ#180	8	93	9
BZ#183	8	93	9
BZ#185	8	93	9
BZ#187	1	93	1
BZ#189	4	93	4
BZ#190	29	93	31
BZ#193	1	93	1
BZ#194	29	93	31
BZ#195	16	93	17
BZ#196	13	93	14
BZ#200	5	93	5
BZ#201	27	93	29
BZ#202	11	93	12
BZ#205	15	93	16
BZ#206	13	93	14
BZ#209	15	93	16

Note * - Results were considered nondetect due to suspected false positive as indicated by blank contamination.

TABLE I-6
PCB DETECTS CHANGED TO NON-DETECTS
EPA-FUNDED FISH SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	Number of results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#2	7	120	6
BZ#3	10	120	8
BZ#4	38	120	32
BZ#5	1	120	1
BZ#6	36	120	30
BZ#7	2	120	2
BZ#8	59	120	49
BZ#9	27	120	23
BZ#10	10	120	8
BZ#12	1	120	1
BZ#15	49	120	41
BZ#16	10	120	8
BZ#17	26	120	22
BZ#18	7	120	6
BZ#20	14	120	12
BZ#26	7	120	6
BZ#27	3	120	3
BZ#28	4	120	3
BZ#31	4	120	3
BZ#33	1	120	1
BZ#37	19	120	16
BZ#40	6	120	5
BZ#41	9	120	8
BZ#42	8	120	7
BZ#44	4	120	3
BZ#47	5	120	4
BZ#49	4	120	3
BZ#52	4	120	3
BZ#56	2	120	2
BZ#59	9	120	8
BZ#66	3	120	3
BZ#70	2	120	2
BZ#72	11	120	9
BZ#74	2	120	2
BZ#75	22	120	18
BZ#77	19	120	16
BZ#85	4	120	3
BZ#95	5	120	4
BZ#99	2	120	2
BZ#105	5	120	4
BZ#110	2	120	2
BZ#115	50	120	42
BZ#118	2	120	2
BZ#119	34	120	28
BZ#122	37	120	31
BZ#123	3	120	3
BZ#126	29	120	24

TABLE I-6
PCB DETECTS CHANGED TO NON-DETECTS
EPA-FUNDED FISH SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	Number of results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#128	7	120	6
BZ#129	32	120	27
BZ#135	7	120	6
BZ#136	12	120	10
BZ#141	17	120	14
BZ#151	7	120	6
BZ#156	22	120	18
BZ#157	49	120	41
BZ#165	72	120	60
BZ#167	9	120	8
BZ#168	1	120	1
BZ#170	21	120	18
BZ#171	37	120	31
BZ#176	63	120	53
BZ#177	4	120	3
BZ#178	28	120	23
BZ#180	19	120	16
BZ#183	2	120	2
BZ#185	20	120	17
BZ#187	2	120	2
BZ#189	17	120	14
BZ#190	43	120	36
BZ#191	1	120	1
BZ#194	29	120	24
BZ#195	80	120	67
BZ#196	34	120	28
BZ#198	101	120	84
BZ#199	9	120	8
BZ#200	22	120	18
BZ#201	11	120	9
BZ#202	20	120	17
BZ#205	65	120	54
BZ#206	33	120	28
BZ#207	4	120	3
BZ#208	34	120	28
BZ#209	36	120	30
BZ#209	5	120	4

Note * - Results were considered nondetect due to suspected false positive as indicated by blank contamination.

TABLE I-7
PCB DETECTS CHANGED TO NON-DETECTS
NOAA-FUNDED FISH SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	Number of results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#3	4	115	3
BZ#8	1	115	1
BZ#26	2	115	2
BZ#31	2	115	2
BZ#44	2	115	2
BZ#47	1	115	1
BZ#49	3	115	3
BZ#52	3	115	3
BZ#56	1	115	1
BZ#66	2	115	2
BZ#70	2	115	2
BZ#74	2	115	2
BZ#83	1	115	1
BZ#84	2	115	2
BZ#85	2	115	2
BZ#87	2	115	2
BZ#91	1	115	1
BZ#92	2	115	2
BZ#95	2	115	2
BZ#97	3	115	3
BZ#99	2	115	2
BZ#101 with BZ#[9	2	115	2
BZ#105	2	115	2
BZ#107	1	115	1
BZ#110	2	115	2
BZ#118	2	115	2
BZ#119	2	115	2
BZ#128	2	115	2
BZ#135	2	115	2
BZ#136	1	115	1
BZ#137	1	115	1
BZ#138	2	115	2
BZ#141	2	115	2
BZ#143	1	115	1
BZ#149	1	115	1
BZ#153	2	115	2
BZ#156	2	115	2
BZ#158	2	115	2
BZ#165	6	115	5
BZ#167	1	115	1
BZ#170	3	115	3
BZ#180	1	115	1
BZ#198	5	115	4
BZ#201	2	115	2
BZ#205	2	115	2
BZ#209	5	115	4

TABLE I-7
PCB DETECTS CHANGED TO NON-DETECTS
NOAA-FUNDED FISH SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	Number of results considered nondetect*	Total number of results	Percentage of results considered nondetect

Note * - Results were considered nondetect due to suspected false positive as indicated by blank contamination.

TABLE I-8
PCB DETECTS CHANGED TO NON-DETECTS
INVERTEBRATE SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#1	12	83	14
BZ#2	12	83	14
BZ#3	12	83	14
BZ#4	3	83	4
BZ#6	53	83	64
BZ#7	2	83	2
BZ#8	10	83	12
BZ#10	3	83	4
BZ#12	3	83	4
BZ#15	25	83	30
BZ#16	3	83	4
BZ#17	9	83	11
BZ#19	1	83	1
BZ#20	4	83	5
BZ#22	4	83	5
BZ#25	5	83	6
BZ#26	9	83	11
BZ#41	2	83	2
BZ#44	14	83	17
BZ#49	6	83	7
BZ#53	4	83	5
BZ#56	4	83	5
BZ#66	3	83	4
BZ#70	2	83	2
BZ#74	1	83	1
BZ#75	5	83	6
BZ#77	5	83	6
BZ#83	10	83	12
BZ#84	2	83	2
BZ#87	1	83	1
BZ#95	1	83	1
BZ#99	6	83	7
BZ#105	12	83	14
BZ#105 & BZ#[168]	2	83	2
BZ#110	1	83	1
BZ#115	2	83	2
BZ#118	2	83	2
BZ#119	4	83	5
BZ#126	1	83	1
BZ#128	15	83	18
BZ#136	2	83	2
BZ#138	1	83	1
BZ#157	13	83	16
BZ#158	5	83	6
BZ#165	10	83	12
BZ#167	3	83	4
BZ#170	6	83	7
BZ#174	9	83	11

TABLE I-8
PCB DETECTS CHANGED TO NON-DETECTS
INVERTEBRATE SAMPLES
HUDSON RIVER RI/FS PCB REASSESSMENT

Congener Name	results considered nondetect*	Total number of results	Percentage of results considered nondetect
BZ#176	28	83	34
BZ#180	4	83	5
BZ#183	8	83	10
BZ#185	2	83	2
BZ#193	2	83	2
BZ#194	1	83	1
BZ#195	3	83	4
BZ#196	4	83	5
BZ#198	20	83	24
BZ#199	2	83	2
BZ#200	13	83	16
BZ#201	8	83	10
BZ#205	7	83	8
BZ#206	3	83	4

Note * - Results were considered nondetect due to suspected false positive as indicated by blank contamination.

Table I-9
Ecological Sediment Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#1	93	22	3	38	6	5	19	20%
BZ#2	93	66	17	0	0	0	10	11%
BZ#3	93	17	8	23	11	9	25	27%
BZ#4	93	2	33	4	44	9	1	1%
BZ#5	93	75	3	0	0	4	11	12%
BZ#6	93	6	0	63	11	10	3	3%
BZ#7	93	26	17	3	28	12	7	8%
BZ#8	93	6	16	6	7	48	10	11%
BZ#9	93	14	29	4	27	17	2	2%
BZ#10	93	2	33	4	44	9	1	1%
BZ#12	93	8	0	28	12	11	34	37%
BZ#15	93	13	0	5	52	21	2	2%
BZ#16	93	5	23	3	35	14	13	14%
BZ#17	93	13	0	5	52	21	2	2%
BZ#18	93	18	0	32	9	28	6	6%
BZ#19	93	26	14	43	1	6	3	3%
BZ#20	93	13	17	4	51	3	5	5%
BZ#22	93	5	9	61	12	6	0	0%
BZ#23NT	93	91	0	0	0	2	0	0%
BZ#25	93	9	0	71	7	4	2	2%
BZ#26	93	4	0	74	6	8	1	1%
BZ#27 & BZ#[24]	93	7	14	5	67	0	0	0%
BZ#28	93	4	10	34	10	35	0	0%
BZ#29	93	41	2	4	41	5	0	0%
BZ#31	93	1	2	66	20	4	0	0%
BZ#32NT	93	7	0	6	0	80	0	0%
BZ#33	93	12	1	6	72	2	0	0%
BZ#34NT	93	19	0	6	0	68	0	0%
BZ#35NT	93	36	0	5	52	0	0	0%
BZ#37	93	5	2	6	79	1	0	0%
BZ#39NT	93	92	0	0	0	1	0	0%
BZ#40	93	6	0	6	75	6	0	0%
BZ#41	93	2	0	6	80	5	0	0%
BZ#42	93	3	1	6	70	6	7	8%
BZ#44	93	0	0	78	14	1	0	0%
BZ#45	93	27	28	25	5	2	6	6%
BZ#46NT	93	13	0	6	0	74	0	0%
BZ#47	93	4	0	6	80	2	1	1%
BZ#48NT	93	50	0	3	16	24	0	0%
BZ#49	93	3	0	74	10	5	1	1%
BZ#51NT	93	8	0	6	0	79	0	0%
BZ#52	93	0	0	6	84	3	0	0%
BZ#53	93	32	13	4	16	21	7	8%
BZ#56	93	3	5	6	72	5	2	2%
BZ#58NT	93	83	0	1	0	9	0	0%
BZ#59	93	10	0	33	45	4	1	1%

Table I-9
Ecological Sediment Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#60NT	93	13	0	6	0	74	0	0%
BZ#63NT	93	29	0	5	0	59	0	0%
BZ#64NT	93	2	0	6	0	85	0	0%
BZ#66	93	0	0	6	79	5	3	3%
BZ#67NT	93	27	0	5	0	61	0	0%
BZ#69NT	93	82	0	1	0	10	0	0%
BZ#70	93	0	0	80	12	1	0	0%
BZ#72	93	14	2	21	31	20	5	5%
BZ#74	93	20	14	39	19	1	0	0%
BZ#75	93	89	4	0	0	0	0	0%
BZ#77	93	4	0	5	71	1	12	13%
BZ#82	93	28	14	19	25	3	4	4%
BZ#83	93	14	6	31	24	15	3	3%
BZ#84	93	3	1	76	11	1	1	1%
BZ#85	93	5	2	63	16	2	5	5%
BZ#87	93	5	20	5	55	2	6	6%
BZ#91	93	6	0	52	33	2	0	0%
BZ#92	93	7	2	5	66	6	7	8%
BZ#95	93	0	0	6	79	3	5	5%
BZ#96NT	93	35	0	5	0	53	0	0%
BZ#97	93	10	2	67	9	4	1	1%
BZ#99	93	1	0	78	11	2	1	1%
BZ#100NT	93	26	0	6	61	0	0	0%
BZ#101 with BZ#[9	93	0	0	6	84	1	2	2%
BZ#104NT	93	77	0	1	15	0	0	0%
BZ#105 & BZ#[168	93	8	7	6	68	0	4	4%
BZ#107	93	17	2	9	57	6	2	2%
BZ#110	93	1	0	6	84	2	0	0%
BZ#114NT	93	63	0	4	0	26	0	0%
BZ#115	93	87	4	0	0	0	2	2%
BZ#118	93	0	0	78	11	4	0	0%
BZ#119	93	24	29	3	15	7	15	16%
BZ#122	93	46	24	2	17	0	4	4%
BZ#123	93	27	21	4	35	0	6	6%
BZ#126	93	68	4	2	9	6	4	4%
BZ#128	93	8	4	32	45	3	1	1%
BZ#129	93	31	35	2	19	0	6	6%
BZ#130NT	93	58	0	3	32	0	0	0%
BZ#131NT	93	91	0	0	0	2	0	0%
BZ#132NT	93	10	0	6	77	0	0	0%
BZ#134NT	93	70	0	1	0	22	0	0%
BZ#135	93	12	1	5	66	2	7	8%
BZ#136	93	20	0	27	35	8	3	3%
BZ#137	93	32	23	3	33	0	2	2%
BZ#138	93	0	0	6	80	3	4	4%
BZ#140NT	93	93	0	0	0	0	0	0%

Table I-9
Ecological Sediment Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#141	93	23	9	23	32	6	0	0%
BZ#143	93	87	5	0	0	0	1	1%
BZ#144NT	93	40	0	6	0	47	0	0%
BZ#146NT	93	3	0	6	0	84	0	0%
BZ#149	93	1	0	6	80	3	3	3%
BZ#151	93	25	0	29	37	2	0	0%
BZ#153	93	1	2	6	78	5	1	1%
BZ#156	93	22	6	4	56	5	0	0%
BZ#157	93	66	3	1	23	0	0	0%
BZ#158	93	27	23	4	34	4	1	1%
BZ#162NT	93	84	0	2	7	0	0	0%
BZ#165	93	73	9	0	0	0	11	12%
BZ#167	93	33	26	4	23	4	3	3%
BZ#169NT	93	93	0	0	0	0	0	0%
BZ#170	93	18	8	30	34	1	2	2%
BZ#171	93	60	24	1	6	2	0	0%
BZ#172NT	93	46	0	4	0	43	0	0%
BZ#173NT	93	93	0	0	0	0	0	0%
BZ#174	93	13	8	6	56	7	3	3%
BZ#175NT	93	87	0	1	0	5	0	0%
BZ#176	93	74	17	0	0	0	2	2%
BZ#177	93	20	13	19	34	5	2	2%
BZ#178	93	67	17	1	8	0	0	0%
BZ#179	93	33	21	5	24	5	5	5%
BZ#180	93	4	8	5	74	2	0	0%
BZ#183	93	33	10	4	41	2	3	3%
BZ#184NT	93	93	0	0	0	0	0	0%
BZ#185	93	70	12	1	3	4	3	3%
BZ#187	93	7	1	52	31	2	0	0%
BZ#189	93	82	9	0	2	0	0	0%
BZ#190	93	61	31	0	1	0	0	0%
BZ#191	93	88	5	0	0	0	0	0%
BZ#193	93	85	7	0	0	0	1	1%
BZ#194	93	52	26	6	6	3	0	0%
BZ#195	93	63	19	0	2	2	7	8%
BZ#196	93	41	15	2	26	2	7	8%
BZ#197NT	93	93	0	0	0	0	0	0%
BZ#198	93	86	5	0	0	0	2	2%
BZ#199	93	83	5	0	5	0	0	0%
BZ#200	93	81	10	1	0	0	1	1%
BZ#201	93	19	29	2	37	5	1	1%
BZ#202	93	53	14	1	22	0	3	3%
BZ#203NT	93	24	0	6	0	63	0	0%
BZ#205	93	73	20	0	0	0	0	0%
BZ#206	93	62	16	0	4	9	2	2%
BZ#207	93	75	4	1	9	1	3	3%

Table I-9
Ecological Sediment Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#208	93	63	3	8	8	5	6	6%
BZ#209	93	52	18	12	4	3	4	4%
Total	13020	4704	979	1927	3514	1522	374	3%

Table I-10
Invertebrate Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#1	83	9	22	8	29	8	7	8%
BZ#2	83	22	46	0	0	0	15	18%
BZ#3	83	13	62	2	0	0	6	7%
BZ#4	83	3	17	3	54	2	4	5%
BZ#5	83	23	50	0	1	4	5	6%
BZ#6	83	18	50	4	4	0	7	8%
BZ#7	83	7	30	3	38	0	5	6%
BZ#8	83	1	16	3	52	7	4	5%
BZ#9	83	4	16	3	57	0	3	4%
BZ#10	83	3	18	3	55	0	4	5%
BZ#12	83	21	50	0	5	0	7	8%
BZ#15	83	3	26	2	50	0	2	2%
BZ#16	83	4	12	3	61	0	3	4%
BZ#17	83	1	9	3	69	0	1	1%
BZ#18	83	1	0	23	58	1	0	0%
BZ#19	83	1	4	5	62	5	6	7%
BZ#20	83	1	9	3	65	0	5	6%
BZ#21NT	11	1	0	0	10	0	0	0%
BZ#22	83	1	11	24	44	0	3	4%
BZ#23NT	83	21	0	3	59	0	0	0%
BZ#24NT	83	19	0	3	56	5	0	0%
BZ#25	83	1	11	25	43	2	1	1%
BZ#26	83	1	9	3	70	0	0	0%
BZ#27	65	1	1	2	60	0	1	2%
BZ#27 & BZ#[24]	18	0	0	1	17	0	0	0%
BZ#28	83	0	0	29	54	0	0	0%
BZ#29	83	25	50	0	0	0	8	10%
BZ#31	83	1	0	10	72	0	0	0%
BZ#32NT	83	1	0	5	1	76	0	0%
BZ#33	83	3	12	3	62	0	3	4%
BZ#34NT	83	28	0	3	0	52	0	0%
BZ#35NT	83	68	0	2	0	13	0	0%
BZ#37	83	1	8	3	68	0	3	4%
BZ#39NT	83	26	0	3	0	54	0	0%
BZ#40	83	0	10	4	66	0	3	4%
BZ#41	83	3	5	3	64	0	8	10%
BZ#42	83	0	3	4	75	0	1	1%
BZ#44	83	1	15	27	38	0	2	2%
BZ#45	83	1	2	19	57	3	1	1%
BZ#46NT	83	76	0	0	0	7	0	0%
BZ#47	83	1	0	3	79	0	0	0%
BZ#48NT	83	26	0	3	54	0	0	0%
BZ#49	83	2	5	9	65	2	0	0%
BZ#51NT	83	9	0	3	0	71	0	0%
BZ#52	83	0	0	28	55	0	0	0%
BZ#53	83	2	7	3	71	0	0	0%

Table I-10
Invertebrate Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#56	83	0	9	4	69	0	1	1%
BZ#57NT	72	30	0	3	0	39	0	0%
BZ#58NT	83	73	0	0	0	10	0	0%
BZ#59	83	2	10	8	60	1	2	2%
BZ#60NT	83	10	0	4	61	8	0	0%
BZ#63NT	83	10	0	4	0	69	0	0%
BZ#64NT	83	0	0	4	3	76	0	0%
BZ#66	83	0	3	4	76	0	0	0%
BZ#67NT	83	22	0	3	7	51	0	0%
BZ#69NT	83	79	0	0	0	4	0	0%
BZ#70	83	0	2	19	58	2	2	2%
BZ#72	83	3	16	8	52	0	4	5%
BZ#74	83	0	3	20	58	2	0	0%
BZ#75	83	1	14	3	62	0	3	4%
BZ#77	83	1	21	3	58	0	0	0%
BZ#82	83	0	12	25	42	1	3	4%
BZ#83	83	1	27	16	37	0	2	2%
BZ#84	83	0	5	4	73	0	1	1%
BZ#85	83	0	8	10	63	1	1	1%
BZ#87	83	0	7	6	67	2	1	1%
BZ#88NT	72	68	0	0	0	4	0	0%
BZ#91	83	1	2	3	76	0	1	1%
BZ#92	83	14	11	13	34	6	5	6%
BZ#95	83	1	2	4	76	0	0	0%
BZ#96NT	83	32	0	3	0	48	0	0%
BZ#97	83	0	7	27	48	1	0	0%
BZ#99	83	0	9	21	50	1	2	2%
BZ#100NT	83	28	0	3	0	52	0	0%
BZ#101 with BZ#[9(	83	0	0	4	78	1	0	0%
BZ#104NT	83	73	0	0	10	0	0	0%
BZ#105	65	1	21	2	40	0	1	2%
BZ#105 & BZ#[168]	18	0	2	1	15	0	0	0%
BZ#107	83	1	15	6	56	1	4	5%
BZ#110	83	0	1	4	78	0	0	0%
BZ#114NT	83	28	0	3	0	52	0	0%
BZ#115	83	6	21	3	46	0	7	8%
BZ#118	83	1	7	11	64	0	0	0%
BZ#119	83	3	24	6	46	0	4	5%
BZ#122	83	23	51	0	5	0	4	5%
BZ#123	83	17	37	1	22	0	6	7%
BZ#126	83	20	39	2	16	0	6	7%
BZ#128	83	2	27	8	45	0	1	1%
BZ#129	83	4	27	3	45	0	4	5%
BZ#130NT	83	17	0	3	0	63	0	0%
BZ#131NT	83	44	0	3	33	3	0	0%
BZ#132NT	83	14	0	4	0	65	0	0%

Table I-10
Invertebrate Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#134NT	83	33	0	3	0	47	0	0%
BZ#135	83	2	12	3	65	0	1	1%
BZ#136	83	4	28	3	42	3	3	4%
BZ#137	83	4	23	8	43	2	3	4%
BZ#138	83	0	5	16	54	6	2	2%
BZ#140NT	83	83	0	0	0	0	0	0%
BZ#141	83	3	16	8	55	0	1	1%
BZ#143	83	26	47	0	3	0	7	8%
BZ#144NT	83	30	0	3	0	50	0	0%
BZ#146NT	83	9	0	4	0	70	0	0%
BZ#149	83	1	7	3	71	0	1	1%
BZ#151	83	1	14	10	45	11	2	2%
BZ#153	83	0	4	4	73	0	2	2%
BZ#156	83	0	17	10	55	1	0	0%
BZ#157	83	18	55	0	6	0	4	5%
BZ#158	83	2	20	4	55	0	2	2%
BZ#160NT	72	72	0	0	0	0	0	0%
BZ#162NT	83	82	0	0	0	1	0	0%
BZ#165	83	22	54	0	2	0	5	6%
BZ#167	83	1	30	8	35	8	1	1%
BZ#168	83	23	49	0	0	0	11	13%
BZ#169NT	83	83	0	0	0	0	0	0%
BZ#170	83	1	19	9	54	0	0	0%
BZ#171	83	21	47	0	12	0	3	4%
BZ#172NT	83	45	0	3	0	35	0	0%
BZ#173NT	83	83	0	0	0	0	0	0%
BZ#174	83	3	29	8	41	1	1	1%
BZ#175NT	83	82	0	0	0	1	0	0%
BZ#176	83	16	60	0	1	0	6	7%
BZ#177	83	2	17	4	53	4	3	4%
BZ#178	83	3	23	4	51	0	2	2%
BZ#179	83	5	29	4	38	4	3	4%
BZ#180	83	0	15	12	34	9	13	16%
BZ#183	83	3	28	6	45	0	1	1%
BZ#184NT	83	40	0	4	0	39	0	0%
BZ#185	83	18	44	1	14	0	6	7%
BZ#187	83	1	10	27	43	1	1	1%
BZ#189	83	22	50	0	0	1	10	12%
BZ#190	83	1	19	4	57	0	2	2%
BZ#191	83	26	51	0	0	0	6	7%
BZ#193	83	23	52	0	2	1	5	6%
BZ#194	83	9	28	5	23	10	8	10%
BZ#195	83	18	39	2	8	1	15	18%
BZ#196	83	16	32	3	24	0	8	10%
BZ#197NT	83	83	0	0	0	0	0	0%
BZ#198	83	20	56	0	0	0	7	8%

Table I-10
Invertebrate Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#199	83	24	51	0	1	0	7	8%
BZ#200	83	21	55	0	1	0	6	7%
BZ#201	83	1	25	9	45	1	2	2%
BZ#202	83	21	41	1	14	0	6	7%
BZ#203NT	83	25	0	4	0	54	0	0%
BZ#205	83	20	55	0	2	2	4	5%
BZ#206	83	9	30	4	27	9	4	5%
BZ#207	83	25	50	0	1	1	6	7%
BZ#208	83	23	47	1	6	1	5	6%
BZ#209	83	24	48	0	4	3	4	5%
Total	12013	2317	2457	781	4834	1252	372	3%

Table I-11
EPA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#1	120	80	5	17	8	9	1	1%
BZ#2	120	89	10	0	0	0	21	18%
BZ#3	120	102	10	0	0	0	8	7%
BZ#4	120	21	37	0	61	1	0	0%
BZ#5	120	112	8	0	0	0	0	0%
BZ#6	120	23	32	5	3	4	53	44%
BZ#7	120	104	8	0	4	4	0	0%
BZ#8	120	21	59	0	36	4	0	0%
BZ#9	120	63	31	0	23	3	0	0%
BZ#10	120	33	11	0	75	1	0	0%
BZ#12	120	104	6	0	9	0	1	1%
BZ#15	120	13	49	0	58	0	0	0%
BZ#16	120	16	10	0	93	0	1	1%
BZ#17	120	5	26	0	84	5	0	0%
BZ#18	120	1	7	97	14	1	0	0%
BZ#19	120	8	0	12	98	2	0	0%
BZ#20	120	15	14	0	91	0	0	0%
BZ#22	120	3	0	90	23	3	1	1%
BZ#23NT	120	48	0	0	72	0	0	0%
BZ#24NT	120	24	0	0	96	0	0	0%
BZ#25	120	0	0	93	23	4	0	0%
BZ#26	120	0	7	0	113	0	0	0%
BZ#27	120	7	3	0	110	0	0	0%
BZ#28	120	1	3	98	17	1	0	0%
BZ#29	120	112	5	0	0	0	3	3%
BZ#31	120	0	4	10	102	4	0	0%
BZ#32NT	120	7	0	1	0	112	0	0%
BZ#33	120	11	2	0	106	1	0	0%
BZ#34NT	120	16	0	0	0	104	0	0%
BZ#35NT	120	112	0	0	0	8	0	0%
BZ#37	120	3	19	0	98	0	0	0%
BZ#39NT	120	21	0	0	0	99	0	0%
BZ#40	120	25	7	0	88	0	0	0%
BZ#41	120	31	10	0	76	2	1	1%
BZ#42	120	0	8	0	108	4	0	0%
BZ#44	120	0	4	105	11	0	0	0%
BZ#45	120	5	0	91	20	3	1	1%
BZ#46NT	120	106	0	0	10	4	0	0%
BZ#47	120	2	5	0	113	0	0	0%
BZ#48NT	120	44	0	0	76	0	0	0%
BZ#49	120	0	4	5	88	23	0	0%
BZ#51NT	120	9	0	0	0	111	0	0%
BZ#52	120	2	2	96	19	1	0	0%
BZ#53	120	11	0	0	103	6	0	0%
BZ#56	120	0	2	0	113	5	0	0%
BZ#57NT	105	47	0	0	58	0	0	0%

Table I-11
EPA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#58NT	120	79	0	0	0	41	0	0%
BZ#59	120	24	10	13	72	0	1	1%
BZ#60NT	120	2	0	0	105	13	0	0%
BZ#63NT	120	23	0	0	0	97	0	0%
BZ#64NT	120	0	0	0	1	119	0	0%
BZ#66	120	0	3	0	111	6	0	0%
BZ#67NT	120	5	0	0	14	101	0	0%
BZ#69NT	120	120	0	0	0	0	0	0%
BZ#70	120	1	2	98	19	0	0	0%
BZ#72	120	6	11	7	89	7	0	0%
BZ#74	120	1	2	97	20	0	0	0%
BZ#75	120	0	22	0	90	8	0	0%
BZ#77	120	7	19	0	89	5	0	0%
BZ#82	120	0	0	99	14	7	0	0%
BZ#83	120	2	0	88	26	4	0	0%
BZ#84	120	0	0	5	115	0	0	0%
BZ#85	120	0	4	7	101	8	0	0%
BZ#87	120	0	0	3	110	6	1	1%
BZ#88NT	105	74	0	0	0	31	0	0%
BZ#91	120	2	0	2	116	0	0	0%
BZ#92	120	1	0	102	15	2	0	0%
BZ#95	120	1	6	0	111	1	1	1%
BZ#96NT	120	30	0	0	0	90	0	0%
BZ#97	120	0	0	102	9	9	0	0%
BZ#99	120	0	2	95	22	1	0	0%
BZ#100NT	120	46	0	0	0	74	0	0%
BZ#101 with BZ#[9	120	0	0	0	119	1	0	0%
BZ#104NT	120	103	0	0	15	2	0	0%
BZ#105	120	0	5	0	114	1	0	0%
BZ#107	120	1	0	7	104	8	0	0%
BZ#110	120	0	2	0	117	1	0	0%
BZ#114NT	120	5	0	1	0	114	0	0%
BZ#115	120	15	50	0	54	1	0	0%
BZ#118	120	0	2	7	102	9	0	0%
BZ#119	120	4	34	2	64	14	2	2%
BZ#122	120	47	37	0	29	7	0	0%
BZ#123	120	75	5	0	29	7	4	3%
BZ#126	120	59	33	0	26	0	2	2%
BZ#128	120	1	6	89	23	1	0	0%
BZ#129	120	7	32	0	80	0	1	1%
BZ#130NT	120	3	0	0	0	117	0	0%
BZ#131NT	120	17	0	0	15	88	0	0%
BZ#132NT	120	1	0	0	0	119	0	0%
BZ#134NT	120	1	0	0	0	119	0	0%
BZ#135	120	1	7	0	105	7	0	0%
BZ#136	120	1	12	6	100	1	0	0%

Table I-11
EPA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#137	120	0	0	30	34	55	1	1%
BZ#138	120	0	0	7	105	8	0	0%
BZ#140NT	120	41	0	0	0	79	0	0%
BZ#141	120	0	17	11	87	5	0	0%
BZ#143	120	107	3	0	1	0	9	8%
BZ#144NT	120	7	0	0	0	113	0	0%
BZ#146NT	120	0	0	0	0	120	0	0%
BZ#149	120	0	0	0	112	8	0	0%
BZ#151	120	0	7	16	97	0	0	0%
BZ#153	120	0	0	0	120	0	0	0%
BZ#156	120	1	22	13	81	3	0	0%
BZ#157	120	18	50	0	42	8	2	2%
BZ#158	120	0	0	1	119	0	0	0%
BZ#160NT	105	95	0	0	0	10	0	0%
BZ#162NT	120	75	0	0	0	45	0	0%
BZ#165	120	35	73	1	6	0	5	4%
BZ#167	120	2	7	16	92	2	1	1%
BZ#168	120	98	6	0	15	0	1	1%
BZ#169NT	120	119	0	0	0	1	0	0%
BZ#170	120	0	21	7	84	8	0	0%
BZ#171	120	10	37	0	72	0	1	1%
BZ#172NT	120	3	0	0	0	117	0	0%
BZ#173NT	120	103	0	0	0	17	0	0%
BZ#174	120	0	0	7	105	8	0	0%
BZ#175NT	120	37	0	0	0	83	0	0%
BZ#176	120	21	63	0	34	0	2	2%
BZ#177	120	2	2	78	38	0	0	0%
BZ#178	120	23	29	0	68	0	0	0%
BZ#179	120	3	1	56	54	6	0	0%
BZ#180	120	15	4	79	22	0	0	0%
BZ#183	120	0	2	7	103	8	0	0%
BZ#184NT	120	17	0	0	0	103	0	0%
BZ#185	120	6	20	8	81	5	0	0%
BZ#187	120	1	1	98	20	0	0	0%
BZ#189	120	34	19	4	52	11	0	0%
BZ#190	120	0	43	4	73	0	0	0%
BZ#191	120	22	2	3	70	12	11	9%
BZ#193	120	77	5	3	31	4	0	0%
BZ#194	120	10	19	56	30	5	0	0%
BZ#195	120	7	81	10	18	4	0	0%
BZ#196	120	14	34	0	71	0	1	1%
BZ#197NT	120	92	0	0	0	28	0	0%
BZ#198	120	19	101	0	0	0	0	0%
BZ#199	120	40	9	2	63	1	5	4%
BZ#200	120	57	26	0	29	8	0	0%
BZ#201	120	0	11	14	95	0	0	0%

Table I-11
EPA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Rejected Results	Percent Rejected
BZ#202	120	25	21	0	68	5	1	1%
BZ#203NT	120	0	0	0	0	120	0	0%
BZ#205	120	42	70	1	6	0	1	1%
BZ#206	120	11	30	31	28	15	5	4%
BZ#207	120	54	8	1	32	16	9	8%
BZ#208	120	19	35	4	61	1	0	0%
BZ#209	120	51	37	6	20	4	2	2%
Total	17355	3635	1630	2124	6984	2822	160	1%

Table I-12
NOAA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Presumed Present Detects	Rejected Results	Percent Rejected
BZ#1	115	71	0	31	13	0	0	0	0%
BZ#2	115	109	0	0	6	0	0	0	0%
BZ#3	115	102	0	4	9	0	0	0	0%
BZ#4	115	19	0	0	96	0	0	0	0%
BZ#5	115	109	0	0	4	0	0	2	2%
BZ#6	115	21	0	26	68	0	0	0	0%
BZ#7	115	105	0	0	10	0	0	0	0%
BZ#8	115	13	0	0	102	0	0	0	0%
BZ#9	115	65	0	0	50	0	0	0	0%
BZ#10	115	28	0	0	87	0	0	0	0%
BZ#12	115	113	0	0	2	0	0	0	0%
BZ#15	115	14	0	0	101	0	0	0	0%
BZ#16	115	8	0	0	107	0	0	0	0%
BZ#17	115	6	0	0	109	0	0	0	0%
BZ#18	115	6	0	95	14	0	0	0	0%
BZ#19	115	13	0	0	102	0	0	0	0%
BZ#20	115	5	0	0	110	0	0	0	0%
BZ#22	115	4	0	95	16	0	0	0	0%
BZ#23NT	115	16	0	0	99	0	0	0	0%
BZ#24NT	115	20	0	0	95	0	0	0	0%
BZ#25	115	3	0	89	23	0	0	0	0%
BZ#26	115	5	0	0	110	0	0	0	0%
BZ#27	115	6	0	0	109	0	0	0	0%
BZ#28	115	1	0	106	8	0	0	0	0%
BZ#29	115	114	0	0	1	0	0	0	0%
BZ#31	115	6	0	0	109	0	0	0	0%
BZ#32NT	115	8	0	0	0	107	0	0	0%
BZ#33	115	7	0	0	108	0	0	0	0%
BZ#34NT	115	39	0	0	0	76	0	0	0%
BZ#35NT	115	106	0	0	0	9	0	0	0%
BZ#37	115	6	0	0	109	0	0	0	0%
BZ#39NT	115	15	0	0	0	100	0	0	0%
BZ#40	115	45	0	0	67	0	0	3	3%
BZ#41	115	24	0	0	91	0	0	0	0%
BZ#42	115	3	0	0	112	0	0	0	0%
BZ#44	115	4	0	103	8	0	0	0	0%
BZ#45	115	5	0	92	18	0	0	0	0%
BZ#46NT	115	114	0	0	1	0	0	0	0%
BZ#47	115	4	0	0	111	0	0	0	0%
BZ#48NT	115	27	0	0	88	0	0	0	0%
BZ#49	115	3	0	0	111	1	0	0	0%
BZ#51NT	115	11	0	0	0	104	0	0	0%
BZ#52	115	4	0	100	11	0	0	0	0%
BZ#53	115	17	0	0	98	0	0	0	0%
BZ#56	115	3	0	0	112	0	0	0	0%
BZ#57NT	115	56	0	0	59	0	0	0	0%
BZ#58NT	115	75	0	0	0	40	0	0	0%
BZ#59	115	3	0	0	111	0	0	1	1%
BZ#60NT	115	36	0	0	79	0	0	0	0%
BZ#63NT	115	5	0	0	0	110	0	0	0%
BZ#64NT	115	1	0	0	0	114	0	0	0%
BZ#66	115	4	0	0	111	0	0	0	0%
BZ#67NT	115	12	0	0	0	103	0	0	0%
BZ#69NT	115	115	0	0	0	0	0	0	0%

Table I-12
NOAA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Presumed Present Detects	Rejected Results	Percent Rejected
BZ#70	115	4	0	105	6	0	0	0	0%
BZ#72	115	4	0	0	111	0	0	0	0%
BZ#74	115	3	0	104	8	0	0	0	0%
BZ#75	115	2	0	0	113	0	0	0	0%
BZ#77	115	7	0	0	108	0	0	0	0%
BZ#82	115	2	0	84	29	0	0	0	0%
BZ#83	115	5	0	83	27	0	0	0	0%
BZ#84	115	5	0	0	110	0	0	0	0%
BZ#85	115	4	0	0	111	0	0	0	0%
BZ#87	115	8	0	0	107	0	0	0	0%
BZ#88NT	115	75	0	0	0	40	0	0	0%
BZ#91	115	4	0	0	111	0	0	0	0%
BZ#92	115	5	0	91	19	0	0	0	0%
BZ#95	115	7	0	0	108	0	0	0	0%
BZ#96NT	115	22	0	0	0	93	0	0	0%
BZ#97	115	4	0	104	7	0	0	0	0%
BZ#99	115	4	0	105	6	0	0	0	0%
BZ#100NT	115	33	0	0	0	82	0	0	0%
BZ#101 with BZ#[9	115	4	0	0	111	0	0	0	0%
BZ#104NT	115	101	0	0	0	14	0	0	0%
BZ#105	115	4	0	0	111	0	0	0	0%
BZ#107	115	3	0	0	112	0	0	0	0%
BZ#110	115	4	0	0	111	0	0	0	0%
BZ#114NT	115	20	0	0	0	95	0	0	0%
BZ#115	115	2	0	0	113	0	0	0	0%
BZ#118	115	4	0	0	111	0	0	0	0%
BZ#119	115	11	0	0	104	0	0	0	0%
BZ#122	115	81	0	0	34	0	0	0	0%
BZ#123	115	87	0	0	24	0	0	4	3%
BZ#126	115	58	0	0	56	0	0	1	1%
BZ#128	115	4	0	78	33	0	0	0	0%
BZ#129	115	3	0	0	112	0	0	0	0%
BZ#130NT	115	10	0	0	0	105	0	0	0%
BZ#131NT	115	30	0	0	47	38	0	0	0%
BZ#132NT	115	0	0	0	0	115	0	0	0%
BZ#134NT	115	7	0	0	0	108	0	0	0%
BZ#135	115	5	0	0	110	0	0	0	0%
BZ#136	115	33	0	0	80	0	0	2	2%
BZ#137	115	4	0	29	82	0	0	0	0%
BZ#138	115	3	0	0	112	0	0	0	0%
BZ#140NT	115	77	0	0	0	38	0	0	0%
BZ#141	115	4	0	0	111	0	0	0	0%
BZ#143	115	112	0	0	1	0	0	2	2%
BZ#144NT	115	16	0	0	0	99	0	0	0%
BZ#146NT	115	1	0	0	0	113	0	1	1%
BZ#149	115	3	0	0	112	0	0	0	0%
BZ#151	115	3	0	0	112	0	0	0	0%
BZ#153	115	3	0	0	112	0	0	0	0%
BZ#156	115	5	0	0	106	4	0	0	0%
BZ#157	115	14	0	0	101	0	0	0	0%
BZ#158	115	4	0	2	109	0	0	0	0%
BZ#160NT	115	112	0	0	0	3	0	0	0%
BZ#162NT	115	59	0	0	0	56	0	0	0%
BZ#165	115	105	0	0	8	0	0	2	2%

Table I-12
NOAA-Funded Fish Sample Summary
Hudson River RI/FS PCB Reassessment

Congener Name	Total Number of Results	Unqual. Non-detects	Estimated Non-detects	Unqual. Detects	Estimated Detects	Est. and Presumed Present Detects	Presumed Present Detects	Rejected Results	Percent Rejected
BZ#167	115	3	0	0	112	0	0	0	0%
BZ#168	115	113	0	0	2	0	0	0	0%
BZ#169NT	115	115	0	0	0	0	0	0	0%
BZ#170	115	4	0	0	111	0	0	0	0%
BZ#171	115	8	0	0	107	0	0	0	0%
BZ#172NT	115	13	0	0	0	102	0	0	0%
BZ#173NT	115	85	0	0	0	30	0	0	0%
BZ#174	115	3	0	0	112	0	0	0	0%
BZ#175NT	115	46	0	0	0	69	0	0	0%
BZ#176	115	18	0	0	97	0	0	0	0%
BZ#177	115	1	0	91	23	0	0	0	0%
BZ#178	115	5	0	0	110	0	0	0	0%
BZ#179	115	4	0	65	46	0	0	0	0%
BZ#180	115	1	0	98	16	0	0	0	0%
BZ#183	115	7	0	0	108	0	0	0	0%
BZ#184NT	115	13	0	0	0	102	0	0	0%
BZ#185	115	8	0	0	107	0	0	0	0%
BZ#187	115	0	0	107	8	0	0	0	0%
BZ#189	115	35	0	6	74	0	0	0	0%
BZ#190	115	4	0	0	111	0	0	0	0%
BZ#191	115	31	0	7	77	0	0	0	0%
BZ#193	115	64	0	5	46	0	0	0	0%
BZ#194	115	6	0	72	37	0	0	0	0%
BZ#195	115	7	0	0	107	0	0	1	1%
BZ#196	115	2	0	0	113	0	0	0	0%
BZ#197NT	115	114	0	0	0	1	0	0	0%
BZ#198	115	21	0	0	94	0	0	0	0%
BZ#199	115	34	0	0	81	0	0	0	0%
BZ#200	115	81	0	0	34	0	0	0	0%
BZ#201	115	4	0	0	111	0	0	0	0%
BZ#202	115	10	0	0	105	0	0	0	0%
BZ#203NT	115	2	0	0	0	113	0	0	0%
BZ#205	115	80	0	0	35	0	0	0	0%
BZ#206	115	9	0	63	43	0	0	0	0%
BZ#207	115	45	0	0	70	0	0	0	0%
BZ#208	115	13	0	0	102	0	0	0	0%
BZ#209	115	34	0	3	72	4	2	0	0%
Total	16675	3971	0	2043	8452	2188	2	19	0%

PHASE 2 - BASELINE ECOLOGICAL RISK ASSESSMENT

HUDSON RIVER PCBS REASSESSMENT RI/FS

APPENDIX J

DATA SUPPORTING TEQ ANALYSIS

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DATA SUPPORTING TEQ ANALYSIS

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APPENDIX J

DATA SUPPORTING TEQ ANALYSIS

J.1 Introduction

This appendix provides the results of an analysis using the United States Environmental Protection Agency (USEPA) Phase 2 data and the United States Fish and Wildlife Service (USFWS) tree swallow data to evaluate the proportion of each Toxic Equivalents (TEQ) congener in the TEQ total. The analysis is conducted using both fish-based Toxicity Equivalency Factor (TEF) as well as avian-based TEF. Since the Phase 2 dataset did not quantitate BZ#81, this exploratory analysis was conducted to determine the effect of using BZ#126 at the detection level in fish as a surrogate for both BZ#126 and BZ#81.

J.2 Proportion of TEQ

To evaluate the impact of using BZ#126 at the detection level and using BZ#126 as a surrogate for BZ#81, the following analysis was conducted. First, all the TEQ-based fish concentrations were compiled and the individual fish-based TEF applied (setting all non-detects equal to the detection level). These values were then summed and each individual congener expressed as a proportion of the TEQ sum for that sample. The results for each individual sample are presented in Table J-1. Since the USFWS tree swallow dataset quantitated BZ#81, this same procedure was again followed using this dataset (only 1995 was used because the 1994 dataset did not quantitate as many congeners) and again applying the fish-based TEF. Table J-2 in Appendix J presents the results obtained by applying the fish-based TEF to the tree swallow TEQ congener concentrations and expressing the results as proportions of the total TEQ for each individual sample.

Table J-3 shows the results of the TEQ proportions when the avian-based TEF were used. Individual tree swallow congener concentrations were multiplied by the appropriate TEF. The result was then expressed as a proportion of the total TEQ and is shown in Table J-3.

Table J-4 shows the comparison of the TEQ-proportion for each individual congener on an average basis from the fish-based analysis using the Phase 2 dataset (USEPA and NOAA fish data) and the USFWS data. The results presented in this table demonstrate that on a TEQ basis, BZ#77, BZ#81, BZ#105, BZ#118 and BZ#126 comprise nearly 97% of the total TEQ concentration. For the fish-based results, the proportion of BZ#126 (even at the detection level) is much higher than the USFWS-based results, and in fact roughly equal the sum of BZ#126 and BZ#81 from the USFWS dataset. This analysis shows that is a reasonable assumption to use the Phase 2 dataset in evaluating TEQ-based exposures.

TABLE J-1
PROPORTION OF TEQ CONGENERS IN FISH USING PHASE 2 DATASET

Species	River Mile	BZ#77	BZ#105	BZ#114	BZ#118	BZ#123	BZ#126	BZ#156	BZ#157	BZ#167	BZ#169	BZ#189
WP	25.8	0.11	0.04	0.00	0.10	0.00	0.70	0.01	0.00	0.01	0.02	0.00
WP	25.8	0.13	0.03	0.00	0.12	0.00	0.64	0.01	0.00	0.01	0.04	0.00
WP	25.8	0.10	0.03	0.00	0.11	0.00	0.70	0.01	0.00	0.01	0.03	0.00
WP	25.8	0.09	0.02	0.00	0.09	0.00	0.74	0.01	0.00	0.01	0.03	0.00
WP	25.8	0.12	0.04	0.00	0.14	0.00	0.60	0.02	0.00	0.02	0.04	0.00
WP	47.3	0.02	0.00	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00
WP	47.3	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
WP	47.3	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
WP	47.3	0.03	0.01	0.00	0.03	0.00	0.92	0.00	0.00	0.00	0.01	0.00
WP	47.3	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
SPOT	58.7	0.02	0.00	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
SPOT	58.7	0.03	0.01	0.00	0.03	0.00	0.92	0.00	0.00	0.00	0.01	0.00
SPOT	58.7	0.02	0.00	0.00	0.02	0.00	0.95	0.00	0.00	0.00	0.01	0.00
WP	58.7	0.06	0.02	0.00	0.05	0.00	0.85	0.01	0.00	0.00	0.01	0.00
WP	58.7	0.03	0.01	0.00	0.03	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	58.7	0.03	0.01	0.00	0.03	0.00	0.93	0.00	0.00	0.00	0.01	0.00
WP	58.7	0.13	0.03	0.00	0.12	0.00	0.63	0.01	0.00	0.01	0.05	0.00
WP	58.7	0.03	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
BB	88.9	0.01	0.06	0.01	0.17	0.00	0.67	0.02	0.01	0.03	0.02	0.00
BB	88.9	0.01	0.07	0.01	0.22	0.00	0.57	0.03	0.00	0.03	0.05	0.00
BB	88.9	0.01	0.08	0.01	0.22	0.00	0.59	0.02	0.00	0.03	0.04	0.00
LMB	88.9	0.02	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
LMB	88.9	0.02	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.01	0.01	0.00
LMB	88.9	0.03	0.00	0.00	0.01	0.00	0.94	0.00	0.00	0.00	0.01	0.00
PKSD	88.9	0.01	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
PKSD	88.9	0.02	0.00	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
PKSD	88.9	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
PKSD	88.9	0.02	0.00	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00
SPOT	88.9	0.02	0.00	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00
SPOT	88.9	0.11	0.03	0.00	0.08	0.01	0.70	0.01	0.00	0.00	0.06	0.00
SPOT	88.9	0.11	0.02	0.00	0.06	0.01	0.73	0.01	0.01	0.00	0.05	0.00
STB	88.9	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
STB	88.9	0.03	0.01	0.00	0.02	0.00	0.92	0.00	0.00	0.00	0.01	0.00
STB	88.9	0.02	0.00	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00

TABLE J-1
PROPORTION OF TEQ CONGENERS IN FISH USING PHASE 2 DATASET

Species	River Mile	BZ#77	BZ#105	BZ#114	BZ#118	BZ#123	BZ#126	BZ#156	BZ#157	BZ#167	BZ#169	BZ#189
WP	88.9	0.09	0.03	0.01	0.08	0.00	0.78	0.01	0.00	0.01	0.01	0.00
WP	88.9	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
WP	88.9	0.08	0.05	0.01	0.18	0.01	0.58	0.02	0.00	0.02	0.05	0.00
YP	88.9	0.05	0.01	0.00	0.03	0.00	0.89	0.00	0.00	0.00	0.01	0.00
YP	88.9	0.09	0.03	0.00	0.10	0.00	0.70	0.01	0.00	0.01	0.05	0.00
YP	88.9	0.01	0.01	0.00	0.03	0.00	0.94	0.00	0.00	0.00	0.01	0.00
YP	88.9	0.08	0.04	0.00	0.13	0.00	0.69	0.01	0.00	0.01	0.04	0.00
SPOT	100	0.02	0.01	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00
SPOT	100	0.02	0.01	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00
SPOT	100	0.03	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
SPOT	113.8	0.02	0.01	0.00	0.01	0.00	0.94	0.00	0.00	0.00	0.01	0.00
SPOT	113.8	0.02	0.01	0.00	0.01	0.00	0.94	0.00	0.00	0.00	0.01	0.00
SPOT	113.8	0.02	0.01	0.00	0.01	0.00	0.94	0.00	0.00	0.00	0.01	0.00
WP	113.8	0.12	0.03	0.01	0.08	0.00	0.74	0.01	0.00	0.00	0.01	0.00
WP	113.8	0.11	0.03	0.01	0.09	0.00	0.74	0.01	0.00	0.01	0.01	0.00
WP	113.8	0.03	0.01	0.00	0.03	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	113.8	0.03	0.01	0.00	0.02	0.00	0.92	0.00	0.00	0.00	0.01	0.00
WP	113.8	0.02	0.01	0.00	0.03	0.00	0.91	0.00	0.00	0.00	0.01	0.00
SPOT	122.4	0.01	0.00	0.00	0.01	0.00	0.96	0.00	0.00	0.00	0.01	0.00
SPOT	122.4	0.02	0.00	0.00	0.01	0.00	0.95	0.00	0.00	0.00	0.01	0.00
SPOT	122.4	0.03	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
WP	122.4	0.04	0.01	0.00	0.02	0.00	0.92	0.00	0.00	0.00	0.01	0.00
WP	122.4	0.00	0.00	0.00	0.00	0.00	0.99	0.00	0.00	0.00	0.00	0.00
WP	122.4	0.01	0.00	0.00	0.01	0.00	0.97	0.00	0.00	0.00	0.00	0.00
WP	122.4	0.03	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
WP	122.4	0.04	0.01	0.00	0.03	0.00	0.90	0.00	0.00	0.00	0.01	0.00
YP	122.4	0.11	0.04	0.01	0.08	0.00	0.74	0.01	0.00	0.01	0.01	0.00
YP	122.4	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
YP	122.4	0.02	0.01	0.00	0.01	0.00	0.94	0.00	0.00	0.00	0.01	0.00
SPOT	137.2	0.03	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
SPOT	137.2	0.02	0.01	0.00	0.02	0.00	0.94	0.00	0.00	0.00	0.01	0.00
SPOT	137.2	0.04	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	137.2	0.05	0.01	0.00	0.03	0.00	0.89	0.00	0.00	0.00	0.01	0.00
WP	137.2	0.34	0.08	0.01	0.14	0.00	0.41	0.01	0.00	0.00	0.00	0.00

TABLE J-1
PROPORTION OF TEQ CONGENERS IN FISH USING PHASE 2 DATASET

Species	River Mile	BZ#77	BZ#105	BZ#114	BZ#118	BZ#123	BZ#126	BZ#156	BZ#157	BZ#167	BZ#169	BZ#189
WP	137.2	0.04	0.01	0.00	0.03	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	137.2	0.03	0.01	0.00	0.03	0.00	0.92	0.00	0.00	0.00	0.01	0.00
PKSD	143.5	0.05	0.01	0.00	0.03	0.00	0.89	0.00	0.00	0.00	0.01	0.00
PKSD	143.5	0.04	0.01	0.00	0.02	0.00	0.93	0.00	0.00	0.00	0.01	0.00
PKSD	143.5	0.08	0.01	0.00	0.03	0.00	0.86	0.00	0.00	0.01	0.01	0.00
PKSD	143.5	0.04	0.01	0.00	0.03	0.00	0.91	0.00	0.00	0.00	0.01	0.00
SPOT	143.5	0.22	0.04	0.00	0.09	0.00	0.59	0.01	0.00	0.00	0.04	0.00
SPOT	143.5	0.06	0.01	0.00	0.03	0.00	0.88	0.00	0.00	0.00	0.01	0.00
SPOT	143.5	0.25	0.05	0.01	0.12	0.00	0.52	0.01	0.00	0.01	0.03	0.00
STB	143.5	0.16	0.03	0.00	0.07	0.00	0.69	0.01	0.00	0.00	0.03	0.00
STB	143.5	0.05	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
STB	143.5	0.04	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	143.5	0.05	0.01	0.00	0.02	0.00	0.90	0.00	0.00	0.00	0.01	0.00
WP	143.5	0.04	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	143.5	0.05	0.01	0.00	0.02	0.00	0.90	0.00	0.00	0.00	0.01	0.00
WP	143.5	0.04	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
WP	143.5	0.05	0.01	0.00	0.02	0.00	0.90	0.00	0.00	0.00	0.01	0.00
YP	143.5	0.18	0.05	0.01	0.11	0.00	0.62	0.01	0.00	0.01	0.01	0.00
YP	143.5	0.04	0.02	0.00	0.04	0.00	0.89	0.00	0.00	0.00	0.01	0.00
YP	143.5	0.04	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
SPOT	159	0.05	0.01	0.00	0.03	0.00	0.90	0.00	0.00	0.00	0.01	0.00
SPOT	159	0.05	0.01	0.00	0.02	0.00	0.90	0.00	0.00	0.00	0.01	0.00
SPOT	159	0.27	0.08	0.01	0.16	0.00	0.42	0.01	0.00	0.01	0.04	0.00
LMB	169.5	0.18	0.06	0.01	0.11	0.00	0.61	0.01	0.00	0.00	0.02	0.00
LMB	169.5	0.16	0.05	0.01	0.12	0.00	0.62	0.01	0.00	0.01	0.02	0.00
LMB	169.5	0.14	0.04	0.01	0.09	0.00	0.67	0.01	0.00	0.00	0.03	0.00
PKSD	169.5	0.19	0.03	0.00	0.07	0.00	0.64	0.01	0.00	0.00	0.05	0.00
PKSD	169.5	0.06	0.01	0.00	0.02	0.00	0.89	0.00	0.00	0.00	0.01	0.00
PKSD	169.5	0.37	0.07	0.01	0.14	0.00	0.40	0.01	0.00	0.00	0.00	0.00
PKSD	169.5	0.06	0.01	0.00	0.03	0.00	0.89	0.00	0.00	0.00	0.01	0.00
PKSD	169.5	0.06	0.01	0.00	0.03	0.00	0.88	0.00	0.00	0.00	0.01	0.00
SPOT	169.5	0.04	0.01	0.00	0.03	0.00	0.91	0.00	0.00	0.00	0.01	0.00
SPOT	169.5	0.04	0.01	0.00	0.02	0.00	0.91	0.00	0.00	0.00	0.01	0.00
SPOT	169.5	0.06	0.02	0.00	0.04	0.00	0.87	0.00	0.00	0.00	0.01	0.00

TABLE J-1
PROPORTION OF TEQ CONGENERS IN FISH USING PHASE 2 DATASET

Species	River Mile	BZ#77	BZ#105	BZ#114	BZ#118	BZ#123	BZ#126	BZ#156	BZ#157	BZ#167	BZ#169	BZ#189
YP	169.5	0.28	0.08	0.02	0.17	0.01	0.43	0.01	0.00	0.01	0.00	0.00
YP	169.5	0.05	0.02	0.00	0.03	0.00	0.88	0.00	0.00	0.00	0.01	0.00
YP	169.5	0.31	0.08	0.01	0.16	0.00	0.38	0.01	0.00	0.01	0.04	0.00
YP	169.5	0.38	0.07	0.02	0.15	0.01	0.35	0.01	0.00	0.01	0.00	0.00
YP	169.5	0.32	0.07	0.01	0.14	0.00	0.45	0.01	0.00	0.00	0.00	0.00
LMB	189.5	0.37	0.06	0.01	0.10	0.00	0.45	0.01	0.00	0.00	0.00	0.00
LMB	189.5	0.35	0.06	0.01	0.10	0.00	0.46	0.01	0.00	0.00	0.00	0.00
LMB	189.5	0.34	0.06	0.01	0.12	0.00	0.45	0.01	0.00	0.00	0.00	0.00
PKSD	189.5	0.47	0.05	0.01	0.11	0.00	0.35	0.01	0.00	0.00	0.00	0.00
PKSD	189.5	0.40	0.06	0.01	0.12	0.00	0.40	0.01	0.00	0.00	0.01	0.00
PKSD	189.5	0.42	0.05	0.01	0.10	0.00	0.42	0.00	0.00	0.00	0.00	0.00
PKSD	189.5	0.47	0.05	0.01	0.10	0.00	0.35	0.01	0.00	0.00	0.00	0.00
PKSD	189.5	0.46	0.05	0.01	0.10	0.00	0.37	0.00	0.00	0.00	0.00	0.00
SPOT	189.5	0.41	0.06	0.01	0.10	0.00	0.41	0.01	0.00	0.00	0.00	0.00
SPOT	189.5	0.32	0.05	0.01	0.10	0.00	0.48	0.01	0.00	0.00	0.02	0.00
SPOT	189.5	0.32	0.05	0.01	0.09	0.00	0.49	0.01	0.00	0.00	0.02	0.00
YP	189.5	0.47	0.09	0.02	0.18	0.01	0.21	0.01	0.00	0.01	0.00	0.00
YP	189.5	0.30	0.06	0.01	0.14	0.00	0.47	0.01	0.00	0.01	0.01	0.00
YP	189.5	0.39	0.08	0.02	0.16	0.01	0.32	0.01	0.00	0.01	0.00	0.00
YP	189.5	0.40	0.06	0.01	0.11	0.00	0.40	0.01	0.00	0.00	0.00	0.00
YP	189.5	0.24	0.05	0.01	0.10	0.00	0.56	0.01	0.00	0.00	0.02	0.00
PKSD	191.5	0.40	0.06	0.01	0.13	0.00	0.39	0.01	0.00	0.00	0.00	0.00
PKSD	191.5	0.37	0.06	0.01	0.12	0.00	0.44	0.01	0.00	0.00	0.00	0.00
PKSD	191.5	0.26	0.05	0.00	0.10	0.00	0.55	0.01	0.00	0.00	0.03	0.00
PKSD	191.5	0.35	0.06	0.01	0.14	0.00	0.42	0.01	0.00	0.00	0.00	0.00
PKSD	191.5	0.24	0.05	0.01	0.10	0.00	0.59	0.01	0.00	0.00	0.00	0.00
PKSD	191.5	0.26	0.05	0.00	0.11	0.00	0.54	0.01	0.00	0.00	0.02	0.00
SPOT	191.5	0.30	0.07	0.01	0.15	0.00	0.42	0.01	0.00	0.01	0.03	0.00
SPOT	191.5	0.26	0.09	0.01	0.17	0.00	0.43	0.01	0.00	0.01	0.03	0.00
SPOT	191.5	0.24	0.08	0.01	0.15	0.00	0.46	0.01	0.00	0.01	0.04	0.00
YP	191.5	0.03	0.01	0.00	0.03	0.00	0.92	0.00	0.00	0.00	0.01	0.00
YP	191.5	0.36	0.07	0.01	0.12	0.00	0.42	0.01	0.00	0.00	0.00	0.00
YP	191.5	0.33	0.08	0.01	0.14	0.00	0.43	0.01	0.00	0.00	0.00	0.00
YP	191.5	0.35	0.07	0.01	0.13	0.00	0.44	0.00	0.00	0.00	0.00	0.00

TABLE J-1
PROPORTION OF TEQ CONGENERS IN FISH USING PHASE 2 DATASET

Species	River Mile	BZ#77	BZ#105	BZ#114	BZ#118	BZ#123	BZ#126	BZ#156	BZ#157	BZ#167	BZ#169	BZ#189
YP	191.5	0.33	0.07	0.01	0.13	0.00	0.43	0.01	0.00	0.00	0.00	0.00
LMB	194.1	0.35	0.07	0.01	0.15	0.00	0.39	0.01	0.00	0.01	0.00	0.00
LMB	194.1	0.11	0.02	0.00	0.03	0.00	0.82	0.00	0.00	0.00	0.01	0.00
LMB	194.1	0.10	0.02	0.00	0.03	0.00	0.83	0.00	0.00	0.00	0.01	0.00
PKSD	194.1	0.22	0.02	0.00	0.03	0.00	0.72	0.00	0.00	0.00	0.00	0.00
PKSD	194.1	0.44	0.06	0.01	0.13	0.00	0.34	0.01	0.00	0.00	0.00	0.00
SPOT	194.1	0.37	0.06	0.01	0.12	0.00	0.42	0.01	0.00	0.00	0.00	0.00
SPOT	194.1	0.35	0.08	0.01	0.15	0.00	0.40	0.01	0.00	0.00	0.01	0.00
SPOT	194.1	0.38	0.06	0.01	0.13	0.00	0.40	0.01	0.00	0.00	0.00	0.00
YP	194.1	0.53	0.09	0.01	0.16	0.01	0.18	0.01	0.00	0.01	0.00	0.00
YP	194.1	0.40	0.07	0.01	0.13	0.00	0.37	0.01	0.00	0.00	0.00	0.00
YP	194.1	0.22	0.19	0.00	0.36	0.00	0.22	0.00	0.00	0.00	0.00	0.00
YP	194.1	0.52	0.09	0.01	0.16	0.01	0.20	0.01	0.00	0.00	0.00	0.00
YP	194.1	0.47	0.07	0.01	0.14	0.00	0.29	0.01	0.00	0.00	0.00	0.00
BB	196.9	0.18	0.08	0.01	0.18	0.00	0.50	0.01	0.00	0.01	0.02	0.00
Whole River Avg		0.15	0.03	0.00	0.07	0.00	0.71	0.01	0.00	0.00	0.01	0.00
Whole River Stdev		0.15	0.03	0.00	0.06	0.00	0.23	0.00	0.00	0.00	0.01	0.00
Upper River Avg		0.28	0.06	0.01	0.11	0.00	0.52	0.01	0.00	0.00	0.01	0.00
Upper River Stdev		0.14	0.03	0.00	0.06	0.00	0.21	0.00	0.00	0.00	0.01	0.00
Lower River Avg		0.05	0.02	0.00	0.05	0.00	0.85	0.00	0.00	0.00	0.01	0.00
Lower River Stdev		0.06	0.02	0.00	0.05	0.00	0.13	0.01	0.00	0.01	0.01	0.00

(BB = Brown Bullhead; LMB = Large Mouth Bass; PKSD = Pumpkinseed; SPOT = Spottailed Shiner; STB = Striped Bass; WP = White Perch; YP = Yellow Perch)

TABLE J-2

PROPORTION OF FISH-BASED TEQ CONGENERS CONTRIBUTING TO TREE SWALLOW CHICK AND EGG, DUCK, AND INSECT BURDENS

Congener Number	Egg RM195	Chick RM195	Chick RM195	Chick RM195	Egg RM195	Chick RM195	Chick RM195	Chick RM195	Odonates RM195	Insects RM195	Chick RM193
BZ#105	**0.01**	**0.04**	**0.01**	**0.05**	**0.04**	**0.02**	**0.05**	**0.04**	**0.03**	**0.05**	**0.02**
BZ#114	0.00	0.00	0.01	0.01	0.00	0.00	0.01	0.00	0.01	0.00	0.01
BZ#118	**0.05**	**0.07**	**0.07**	**0.08**	**0.06**	**0.05**	**0.09**	**0.07**	**0.04**	**0.05**	**0.05**
BZ#123;149	**0.03**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.01**	**0.02**	**0.03**
BZ#126	**0.40**	**0.31**	**0.30**	**0.26**	**0.39**	**0.33**	**0.29**	**0.24**	**0.45**	**0.29**	**0.39**
BZ#156	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.01
BZ#157;201	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#167	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#169	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#189	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#77	**0.36**	**0.43**	**0.40**	**0.42**	**0.34**	**0.44**	**0.38**	**0.43**	**0.35**	**0.43**	**0.39**
BZ#81	**0.14**	**0.12**	**0.18**	**0.16**	**0.14**	**0.13**	**0.15**	**0.20**	**0.10**	**0.14**	**0.10**

Congener Number	Chick RM193	Egg RM193	Chick RM193	Egg RM193	Chick RM193	Chick RM193	Chick RM193	Chick RM193	Duck RM193	Odonate RM193	Insect RM193
BZ#105	**0.05**	**0.04**	**0.01**	**0.04**	**0.05**	**0.03**	**0.05**	**0.02**	**0.06**	**0.03**	**0.06**
BZ#114	0.01	0.00	0.00	0.01	0.01	0.01	0.01	0.00	0.01	0.01	0.01
BZ#118	**0.10**	**0.06**	**0.07**	**0.07**	**0.10**	**0.08**	**0.08**	**0.07**	**0.11**	**0.05**	**0.11**
BZ#123;149	**0.02**	**0.02**	**0.02**	**0.02**	**0.02**	**0.03**	**0.02**	**0.02**	**0.02**	**0.01**	**0.02**
BZ#126	**0.32**	**0.35**	**0.33**	**0.38**	**0.31**	**0.30**	**0.37**	**0.34**	**0.39**	**0.54**	**0.26**
BZ#156	0.01	0.00	0.01	0.01	0.01	0.01	0.01	0.00	0.01	0.01	0.01
BZ#157;201	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#167	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#169	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#189	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#77	**0.38**	**0.39**	**0.43**	**0.36**	**0.37**	**0.45**	**0.32**	**0.39**	**0.33**	**0.31**	**0.42**
BZ#81	**0.12**	**0.12**	**0.13**	**0.11**	**0.14**	**0.10**	**0.14**	**0.15**	**0.07**	**0.03**	**0.11**

TABLE J-2

PROPORTION OF FISH-BASED TEQ CONGENERS CONTRIBUTING TO TREE SWALLOW CHICK AND EGG, DUCK, AND INSECT BURDENS

Congener Number	Wood Duck Egg RM190	Wood Duck Adult RM190	Wood Duck Adult RM184	Mallard Egg RM173	Mallard Adult RM 173	Tree Swallow Egg RM173	Tree Swallow Chick RM173	Tree Swallow Egg RM173	Tree Swallow Chick RM173	Tree Swallow Egg RM173	Tree Swallow Chick RM173	Odonate RM173	Insect RM173
BZ#105	**0.04**	**0.05**	**0.02**	**0.07**	**0.04**	**0.05**	**0.05**	**0.04**	**0.05**	**0.06**	**0.05**	**0.03**	0.00
BZ#114	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.00	0.00
BZ#118	**0.10**	**0.08**	**0.06**	**0.08**	**0.10**	**0.10**	**0.10**	**0.09**	**0.06**	**0.06**	**0.10**	**0.06**	0.00
BZ#123;149	**0.01**	**0.01**	**0.01**	**0.01**	**0.01**	**0.05**	**0.05**	**0.04**	**0.07**	**0.05**	**0.05**	**0.02**	0.00
BZ#126	**0.36**	**0.54**	**0.42**	**0.43**	**0.47**	**0.43**	**0.39**	**0.47**	**0.39**	**0.40**	**0.41**	**0.47**	**0.71**
BZ#156	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.00
BZ#157;201	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#167	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#169	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#189	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#77	**0.34**	**0.18**	**0.40**	**0.28**	**0.21**	**0.25**	**0.30**	**0.25**	**0.31**	**0.31**	**0.27**	**0.36**	**0.17**
BZ#81	**0.12**	**0.11**	**0.07**	**0.10**	**0.15**	**0.09**	**0.10**	**0.09**	**0.10**	**0.11**	**0.09**	**0.03**	**0.09**

TABLE J-3

PROPORTION OF TEQ CONGENERS CONTRIBUTING TO TREE SWALLOW CHICK AND EGG, DUCK, AND INSECT BURDENS

Congener Number	Egg RM195	Chick RM195	Chick RM195	Chick RM195	Egg RM195	Chick RM195	Chick RM195	Chick RM195	Odonates RM195	Insects RM195	Chick RM193
BZ#105	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#114	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#118	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#123;149	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#126	**0.04**	**0.03**	**0.02**	**0.02**	**0.04**	**0.03**	**0.03**	**0.02**	**0.04**	**0.02**	**0.04**
BZ#156	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#157;201	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#167	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#169	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#189	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#77	**0.83**	**0.87**	**0.82**	**0.85**	**0.82**	**0.87**	**0.83**	**0.83**	**0.85**	**0.86**	**0.87**
BZ#81	**0.13**	**0.10**	**0.15**	**0.13**	**0.13**	**0.10**	**0.13**	**0.15**	**0.10**	**0.11**	**0.09**

Congener Number	Chick RM193	Egg RM193	Chick RM193	Egg RM193	Chick RM193	Chick RM193	Chick RM193	Chick RM193	Duck RM193	Odonate RM193	Insect RM193
BZ#105	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00
BZ#114	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#118	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#123;149	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#126	**0.03**	**0.03**	**0.03**	**0.04**	**0.03**	**0.02**	**0.04**	**0.03**	**0.04**	**0.06**	**0.02**
BZ#156	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#157;201	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#167	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#169	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#189	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#77	**0.86**	**0.86**	**0.87**	**0.85**	**0.84**	**0.89**	**0.82**	**0.84**	**0.87**	**0.89**	**0.88**
BZ#81	**0.11**	**0.10**	**0.11**	**0.10**	**0.13**	**0.08**	**0.14**	**0.13**	**0.08**	**0.04**	**0.09**

TABLE J-3
PROPORTION OF TEQ CONGENERS CONTRIBUTING TO TREE SWALLOW CHICK AND EGG, DUCK, AND INSECT BURDENS

Congener Number	Wood Duck Egg RM190	Wood Duck Adult RM190	Wood Duck Adult RM184	Mallard Egg RM173	Mallard Adult RM 173	Tree Swallow Egg RM173	Tree Swallow Chick RM173	Tree Swallow Egg RM173	Tree Swallow Chick RM173	Tree Swallow Egg RM173	Tree Swallow Chick RM173	Odonate RM173	Insect RM173
BZ#105	0.00	0.01	0.00	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.00	0.00
BZ#114	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#118	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#123;149	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#126	**0.04**	**0.09**	**0.04**	**0.05**	**0.06**	**0.06**	**0.04**	**0.06**	**0.04**	**0.04**	**0.05**	**0.05**	**0.12**
BZ#156	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#157;201	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#167	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#169	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#189	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BZ#77	**0.84**	**0.72**	**0.90**	**0.82**	**0.72**	**0.82**	**0.84**	**0.82**	**0.84**	**0.83**	**0.83**	**0.92**	**0.72**
BZ#81	**0.12**	**0.18**	**0.06**	**0.12**	**0.20**	**0.11**	**0.11**	**0.11**	**0.11**	**0.12**	**0.11**	**0.03**	**0.15**

TABLE J-4
AVERAGE PROPORTION OF FISH-BASED TEQ CONGENERS USING EPA 1993 DATASET AND USFWS 1995 DATASET

	BZ#77	**BZ#81**	**BZ#105**	BZ#114	**BZ#118**	BZ#123	**BZ#126**	BZ#156	BZ#157	BZ#167	BZ#169	BZ#189
Upper River Mean	**0.28**	NA	**0.06**	0.01	**0.11**	0.00	**0.52**	0.01	0.00	0.00	0.01	0.00
Lower River Mean	**0.05**	NA	**0.02**	0.00	**0.05**	0.00	**0.85**	0.00	0.00	0.00	0.01	0.00
Whole River Mean	**0.15**	NA	**0.03**	0.00	**0.07**	0.00	**0.71**	0.01	0.00	0.00	0.01	0.00
Egg Mean	**0.32**	**0.11**	**0.04**	0.01	**0.07**	0.03	**0.40**	0.01	0.00	0.00	0.00	0.00
Chick Mean	**0.38**	**0.13**	**0.04**	0.01	**0.08**	0.03	**0.33**	0.01	0.00	0.00	0.00	0.00
Odonate Mean	**0.34**	**0.05**	**0.03**	0.00	**0.05**	0.01	**0.49**	0.01	0.00	0.00	0.00	0.00
Insect Mean	**0.34**	**0.11**	**0.04**	0.01	**0.05**	0.02	**0.42**	0.00	0.00	0.00	0.00	0.00

Source: TAMS/Gradient Database Release 4.1b
NA = not available

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APPENDIX K
EXAMINATION OF EXPOSURE PATHWAYS BASED ON CONGENER PATTERNS

K.1 Introduction

As part of the ecological risk assessment, an examination of the fish exposures to PCBs in the environment was completed. Because fish exposures to PCBs from sediment, water and through the food chain all contribute to the fish body burden, it is useful to examine fish body burdens relative to these sources. Specifically, the congener pattern of a fish's body burden reflects, to varying degrees, the nature and history of its exposure. Thus an examination of the congener patterns in fish and other matrices may provide useful clues in designating the main PCB sources to the fish. If the congener "fingerprint" remains unaltered from source to the fish, this analysis can directly link the source(s) to the fish body burden. Information linking fish body burdens to their sources is clearly useful in selecting effective remedial actions. However, as will be shown, the links between fish body burden and source are not straightforward.

Patterns of PCB contamination in fish and benthic invertebrates are examined in this appendix using the congener-specific PCB data from the 1993 United States Environmental Protection Agency (USEPA) Phase 2 ecological investigation, the 1993 National Oceanic and Atmospheric Administration (NOAA) fish analyses, and the 1995 NOAA fish analyses. Additionally, the long-term monitoring records for fish obtained by New York State Department of Environmental Conservation (NYSDEC) are examined along with United States Geological Survey (USGS) water column data to establish current trends between PCB body burden and water column concentrations for several fish species. This subsection represents the biological extension of the geochemical analysis presented in the Data Evaluation and Interpretation Report (DEIR) (USEPA, 1997) and the Low Resolution Sediment Coring Report (LRC) (USEPA, 1998), examining the correlations among fish and invertebrate body burdens, sediment, and water column conditions.

Also in this analysis, the congener patterns contained in fish are examined from the context of classifying the mixture to assist in the selection of toxicity reference values (TRVs). This examination addresses, to a limited extent, the "best" basis for quantifying current fish body burdens in terms of Aroclor-based analyses and standards. This issue arises from the historical analytical protocols that characterized fish body burdens in terms of Aroclors 1248 and 1254, despite the documented presence of a predominantly Aroclor 1242-based source throughout the freshwater Hudson River (USEPA, 1997).

K.2 Summary of the 1997 Report by the National Oceanic and Atmospheric Administration (NOAA)

Prior to the analysis completed for this ERA, NOAA (1997) prepared a report on the congener-based fish results collected by USEPA, NOAA, and NYSDEC in 1993, as well as a subsequent 1995 sampling study carried out by NOAA and NYSDEC. The NOAA report provides an initial basis for the analyses performed as part of this ERA. It should be noted that the NOAA report dealt exclusively with fish congener data, while the analyses completed for the ERA examine fish, sediment, surface water, and invertebrate data. Nonetheless, the NOAA report reached several important conclusions that provide the basis for the analyses described here. These conclusions and some of the supporting results are presented below.

NOAA used a statistical technique involving the calculation of Euclidean distances between samples based on the congener mass fractions contained in Hudson River fish samples. The calculation of the Euclidean distances was based on a subset of the 107 congeners reported by the laboratory for the 1995 data. The congener selection criteria included frequency of detection in the fish (congeners had to be present in at least 75% of the fish samples) and importance to the fish total PCB mass (any individual congener had to contribute at least 1% to the total PCB mass in at least one fish sample). Based on these criteria, 46 congeners were selected (see Table K-1) These congeners represented between 82% to 94% of the total PCB mass in any one fish sample. The analytical technique used to quantitate the congeners in 1995 was the same as that used by USEPA and NOAA in 1993, enabling direct comparison between the sampling periods.

Based on their analysis, NOAA reached several important conclusions, summarized below:

1. Total PCB concentrations in fish generally declined downstream of the Thompson Island (TI) Pool, with the largest change occurring between the Upper Hudson and the Lower Hudson, presumably due to dilution from the Mohawk River discharge.

2. Congener concentrations among fish changed in a regular way moving from upstream to downstream locations, with downstream locations exhibiting higher fractions of the heavier homologues despite an overall decrease in fish body burden with distance downstream.

3. Congener patterns among various fish species collected from a single station were more similar than the same species at different locations. This indicated that fish patterns (and by inference, exposures) were more closely governed by local conditions than by trophic level or species type.

4. Seasonal variations in congener pattern were evident in several species of fish, including striped bass, white perch, and yellow perch. Spring body burdens contained a larger proportion of the pentachloro- and hexachloro-homologues relative to the trichloro- and tetrachlorohomologues with the reverse being true in fall. These results suggest that the fish body burdens can respond rapidly, perhaps as a result of seasonal changes in their exposure pathways.

The conclusions of a second study (Field and Sloan, 1996) involving the same data set were:

5. The congener patterns of samples from the same species at a given location were very similar even though the total PCB concentrations could vary by a factor of two or more on a wet-weight or lipid-normalized basis.

6. The shift toward a more-chlorinated fish body burden with decreasing river mile results primarily from a more rapid decrease in the body burdens of tri and tetrachlorinated congeners. The body burdens of higher chlorinated congeners decrease at a much slower rate with river mile.

Supporting evidence for several of these conclusions is shown in Figures K-1 to K-4. In Figure K-1, congener patterns between two fish (i.e., largemouth bass and spot tail shiner) collected in 1993 at the same station are compared on a mass fraction basis in the upper diagram. If the patterns were identical, a regression line would yield a slope of unity, an intercept of zero and a R^2 value of unity. The comparison between largemouth bass and spot tail shiner yields a regression coefficient which is substantially closer to unity (0.89) than that given in the lower diagram (0.50). In the lower diagram, two samples of the same species, i.e., largemouth bass, are compared between two stations. Thus, this figure documents some of the evidence supporting the third conclusion above. Note that the largemouth bass from RM 89.4 was a juvenile 95 cm in length while the spottail shiner was 64 cm long. Given the comparable size of these fish, it is unlikely that adult spot tail shiners were a food item for the largemouth bass. The largemouth bass from RM 190 was also a juvenile, at 98 cm in length. PCB concentrations at RM 89.4 were 0.8 and 1.5 mg/kg wet weight for the largemouth bass and spottail shiner, respectively. In contrast, the largemouth bass at RM 190 was 23.4 mg/kg wet weight.

Figures K-2 to K-4 present the mean homologue patterns for several fish species collected on a seasonal basis in 1995. Typical sample collection for the NOAA 1995 fish collection program consisted of 5 separate samples per species per location. Because congener patterns among fish species collected from a single station were more similar than those for a single species at different locations, this report will use selected fish samples which are representative of the congener pattern at a given station along the Hudson River to illustrate variations with river mile. In each instance, these samples are carefully chosen using visual inspection after graphing most or all fish samples at the station. In this manner, the pattern presented in this report can be considered the median pattern for the species at the given location.

Figures K-2 and K-3 present the homologue patterns for fish collected exclusively from the Lower Hudson, respectively striped bass and white perch. Although a gradual variation in the homologue pattern with river mile is evident in each figure, a more distinct seasonal shift can be seen for the samples collected at RM 152. In each instance, a shift in homologue pattern from a pentachlorohomologue-dominated spectra in spring to a tetrachlorohomologue-dominated pattern in fall is evident for both species at this location. This shift is also evident in Figure K-4 which presents representative congener patterns for yellow perch. All three figures also show a gradual increase toward a heavier homologue suite with increasing distance downstream. Notably, the white perch and yellow perch at RM 152 are more similar to each other than to samples of the same species at the adjacent upstream and downstream stations. Thus, these figures document some of the evidence supporting conclusions 2, 3 and 4 above.

Extensive discussion and supporting evidence for these conclusions can be found in NOAA (1997). Supporting evidence for conclusions 1, 5 and 6 will be presented later in this chapter. Because congener patterns in fish collected from a single station were more similar than the same species at different locations, this report will use selected fish samples which represent the congener pattern at a given station along the Hudson River to illustrate various phenomena. These samples are carefully chosen using visual inspection after graphing most or all fish samples at the station.

K.3 An Examination of Multiple Matrices via Principal Components Analysis

The analysis prepared by NOAA, as well as those of the DEIR and LRC, demonstrated the complexities of the PCB congener patterns in the Hudson River among the various matrices (i.e., sediments, water, fish and benthic invertebrates). In order to capture and reflect these complexities in the data analysis, a principal components analysis (PCA) was undertaken. Effectively, PCA reduces the data set and its associated variables into a minimum number of variables which can then be used to examine the data. This PCA analysis provides a means of showing the appropriateness of using toxicity reference values (TRVs) based on Aroclor 1254 and will explore the ability to trace the source of PCBs in fish.

The first principal component is constructed as a linear combination of the original variables so as to encompass (or “explain”) the greatest amount of the variance for the original data set. Subsequent principal components encompass the largest amount of the remaining variance of the data set while being uncorrelated (orthogonal) to all previously constructed principal components.

K.3.1 Selection of the Congener Variables for the Analysis

To perform a PCA on the four matrices of interest, a set of congeners common to all matrices was established with an emphasis on congeners of greatest importance to fish. Specifically, the following matrices were examined:

- Sediments - including ecological sediment samples representing 0 to 5 cm and the individual high resolution core samples from 0 to 8 cm which were sliced into 2 and 4 cm subsamples. High resolution core results were included with the ecological sediments in this analysis based on the fact that they were frequently obtained from similar areas of the river. Taken together, these samples were considered to represent some of the possible surficial sediment congener patterns in a given area although it is unclear whether they are truly representative of the mean surficial sediment conditions for fish exposure at each location;

- Fish - both USEPA and NOAA samples for 1993. Since nearly contemporaneous samples were used for the selection of congeners in all matrices, NOAA fish samples taken in 1995 were excluded. In addition, the remedial actions taken between 1993 and 1995 in the Bakers Falls region had a major impact on water column concentrations and thus could have had impacts on the concentration and pattern of the contaminants in the fish. This might confound the comparison among matrices. subsection K.6 contains a comparison of the 1993 and 1995 fish samples;

- Water - on a whole water sample basis (dissolved plus suspended fractions) excluding samples above RM 197 as well as all transect 2 samples, transect 1- station 4 and transect 5 - station 4, due to quantitation issues with some congeners (These issues are discussed in the DEIR, (USEPA, 1997), the Responsiveness Summary for Volumes A, B and C (USEPA, 1998), and the Responsiveness Summary for the LRC (USEPA, 1999);

- Benthic Invertebrates – all benthic invertebrate samples collected from the Hudson downstream of RM 197. Each sample consisted of invertebrates found in the sediments between 0 and 5 cm depth. Most samples consisted of an unsorted mixture of invertebrates from each location. Occassionally, a single species or taxa was found in sufficient abundance and was analyzed separately. These samples were also included in the PCA. Epibenthic samples, collected at the sediment-water interface were included in this matrix as well. (A discussion of the benthic invertebrate results can be found in subsection K.5.)

The primary focus in this analysis was to examine the relationships among the matrices downstream of the GE releases and therefore background samples for all matrices were excluded from the analyses. This was done since the well-documented

large differences between the background and mainstem Hudson River conditions might reduce the ability to resolve some of the more subtle variations among the downstream stations.

In the NOAA (1997) report, simple frequency and mass fraction criteria were established since only a single matrix was involved. This procedure would not work for the examination of four matrices since it was clear from an initial review that not all congeners were equally prevalent in all matrices. A more sophisticated procedure was also needed since many congeners remained undetected in one or more matrices.

For this reason, an optimization scheme was developed wherein a PCA was performed on congener results for each matrix individually (i.e., principal components analyses were performed on the set of mainstem fish samples, benthic invertebrate samples, etc.). The initial analysis utilized all 126 congeners as reported in the USEPA database (USEPA, 1998b). In this analysis, congener contributions were normalized to the total PCB concentration represented by the 126 congeners. These results were then standardized for each individual congener so that each congener was equally weighted in the analysis. From this initial analysis, a subset of congeners was selected based on the following criteria:

1. Any congener with a loading factor greater than 0.25 in the first three principal components for fish was kept in all other matrices for the subsequent iteration.

2. The loading factor for any single congener to the first three principal components in any matrix had to be greater than 0.1. Any congener with loading factors less than 0.1 in all media were dropped from subsequent iterations.

The subset of congeners obtained via these criteria were then renormalized to their sum and restandarized on an individual congener basis. The principal components analyses were repeated for the individual matrices and screened using the criteria given above resulting in a still smaller subset of congeners for the next iteration.

Five iterations were performed in all, converging on the list of 29 congeners given in Table K-2. While this list and the original list prepared by NOAA share many congeners, there are some important differences. In particular, the USEPA list includes 2 dichloro-congeners and has near equal numbers of tri, tetra, penta, and hexa congeners (6:6:5:7) versus the NOAA list which more heavily weights the tetra and penta congeners relative to the tri and hexa congeners (8:11:13:6). The NOAA list also includes more hepta congeners (5 vs. 3) as well as representative octa and nona congeners. Nonetheless, as discussed later, the general data separations obtained by the USEPA congener list for fish data and those obtained from the NOAA list are quite similar.

K.3.2 Initial Results of the Principal Components Analysis

The optimized congener list was used to perform a PCA on the entire suite of matrices (i.e., sediment, surface water, fish, and benthic invertebrates). The analysis yielded a set of principal components which could be used to examine the entire PCB congener data set for the mainstem Hudson. Figure K-5 presents the loadings or scaling factors for the first two principal components. These factors serve to transform the congener mass fraction data into the orthogonal components which are then used for data analysis and charting. Essentially, these factors represent the importance of the individual congeners to each principal component. The greater the absolute value of the load factor, the more important the congener is to the principal component. A positive load factor indicates that the principal component value tends to increase as the specific congener mass fraction increases. A negative factor indicates that the principal component value tends to decrease as the specific congener mass fraction increases. The load factors are used to linearly combine the original 29 congener variables (as mass fractions) into the two principal components.

The first component, plotted in the upper diagram of Figure K-5 represents the ratio of the mass fraction of the tetrachloro and higher congeners to that of the di and trichloro congeners indicated that this component is probably related to the molecular weight of the sample. Component 2 represents the ratio of the dechlorination products to the middle portion of the congener set. It also represents the ratio of the highest chlorinated congeners to the main portion of the congener set. Thus, this component measures the degree of dechlorination in sediment as well as a portion of any enhanced bioaccumulation of the heaviest congeners.

Figure K-6 presents the entire 1993 mainstem Hudson River plotted as a function of component 1 and component 2. The data clearly fall into a “V”-shaped pattern with sediments and water on the left and fish samples on the right. Invertebrates tend to fall near convergence of the “V” and to the right. The PCA can clearly separate the various media based on their congener pattern alone. Also plotted on the diagram are large markers designating several Aroclor standards measured by the same analytical technique and transformed into the two components. Note that the Aroclors were not included in the PCA itself, but are simply plotted here to show the relationship between them and the Hudson River samples in terms of the two components.

It is clear from the diagram that Aroclor 1248 most closely matches the lower end of the fish domain. This is the result of the close match between the molecular weight of Aroclor 1248 and that of many fish samples. The upper end of the fish domain trends toward Aroclor 1260. This is partly a function of molecular weight and partly the exclusion of the heaviest congeners (primarily present in Aroclor 1260) from the principal components analysis.

As noted previously and by NOAA (1997), fish congener patterns appear to change as a function of river mile. However, based on this PCA and the seasonal changes noted by NOAA, the relationship between the congener patterns of the fish and those of

the exposure media (i.e., sediment and water) is not a direct one. That is, it is unlikely that a simple relationship such as a congener pair ratio (e.g., BZ#4/BZ#52) determined from the sediments, water and fish can be directly linked so as to be able to predict the fish congener patterns given the ratios of the sediment and water. In subsection K.9, the inability to predict fish congener patterns directly from sediment conditions (i.e., without sophisticated modeling) is further explored.

Unlike the fish, the sediments and water column samples resemble Aroclors 1242 and 1016, as might be expected, given their prevalence in the GE discharges. The sediment and water data trend toward a different end member, a value to the upper left, indicating lower molecular weight as measured by component 1 and a higher dechlorination fraction as measured by component 2. For illustrative purposes, a theoretical, fully-dechlorinated sediment sample is plotted in the lower diagram of Figure K-6. A line is drawn from this point to the lower end of the sediment data. The sediment data tend to fall along or just above this line, supporting this interpretation of the two principal components with regard to sediments. Further interpretation of the data is possible when the data are examined as a function of river mile.

Figures K-7 to K-11 represent the sediment, water, fish, and benthic invertebrate data grouped into five river sections based on river mile, beginning with the Upper Hudson from the TI Pool to Stillwater (RM 195 to 175) and ending with the saline portion of the Lower Hudson (RM 60 to 0). These sections represent major Hudson River regions, based on proximity to the GE source area and natural or man-made barriers. The division of the freshwater Lower Hudson at RM 100 was simply based on the long extent of this region and the available data. In these diagrams, the progression of the sediments from upper left to lower center is matched by the progression of the fish results from lower center to upper right. Thus, the matched sediment-fish pairs by location remain relatively far apart. Fewer data are available for the water column, but these tend to remain along the upper left branch of the graph. Benthic invertebrates tend to fall in an intermediate location in the diagram, between the sediment and fish, as might be expected based on their direct contact with sediments.

The movement of the sediment data along the left branch of the “V” is commensurate with the declining importance of dechlorination as a means of modifying the sediment inventory as a function of downstream distance from the GE facilities. It is important to note how little the sediment varies between the two freshwater sections of the Lower Hudson River. These results record the absence of important changes to the congener pattern of the sediments in this region and thus the absence of important additional loads to the river. This information coupled with the decline in concentration described in the DEIR (USEPA, 1997) is evidence for the absence of significant additional PCB loading to the Hudson River. Not until the saline region of the river does the range of the sediment data change, reflecting the more chlorinated congener patterns of the inputs associated with the New York City metropolitan area. The DEIR (USEPA, 1997) provides an extensive discussion of the congener patterns found in the lower river.

These results can be contrasted with the fish results, which show a trend toward heavier molecular weight (increase in both components with river mile) on a more continuous basis (i.e., throughout the Lower Hudson). While the increase in the two principal components in the saline Lower Hudson is undoubtedly due in part to the local New York City influence, the cause of the increase in these components for the region between RM 100 and 60 is less clear. There is clearly no evidence for a substantive change in the PCB loads to the river, as recorded by the high resolution sediment cores. Yet, the values for the two components continue to rise through this section of the river. Note as well that this shift to higher molecular weight is not accompanied by an increase in the total PCB body burdens in the fish, as will be discussed in the next section.

K.3.3 Interpretation of the Principal Components Analysis

The PCA suggests a strong similarity between the fish body burdens and Aroclor 1248. This is largely due to the bioaccumulation of the tetrachlorocongeners which are most prevalent in this Aroclor. As is suggested by the loading to components 1 and 2, this PCA strongly reflects the molecular weight of the congener mixture and emphasizes its importance in examining the congener data.

The agreement between Aroclor 1248 and the body burden for upper river fish is demonstrated by comparing upper river fish samples to Aroclor standards on a mass fraction basis. Figure K-12 presents several regressions between a typical upper river 1993 largemouth bass sample from RM 190 versus several Aroclors standards on a mass fraction basis. The regressions represent double hit pairs only, that is congeners which were detected in both sediment and the Aroclor. Although agreement is best for Aroclor 1248, the result is not a true line and several congener proportions fall well away from line. This analysis was repeated using a typical lower river white perch sample from RM 26 and is shown in Figure K-13. Based on the previous principal components analysis, fish in the Lower Hudson appeared to approach Aroclor 1260. However, when all congeners are considered via regressions such as those in Figure K-13, the best regressions are obtained against Aroclor 1254. For the lower river fish sample shown in the figure, the best fit is achieved against Aroclor 1254 with a regression coefficient of 0.65 that is relatively close to the regression coefficient of 0.7 for the upper river fish sample against Aroclor 1248. The fact that the regression coefficients are highest for two different Aroclors is simply indicative of the shift in molecular weight of the fish PCB body burden while moving downstream.

Component 1 itself was examined as a function of river mile for both sediment and fish (see Figure K-14). Though the variance observed is nontrivial, trends in the data are evident. The more pronounced rise in the value of component 1 for the fish data relative to the sediment data is clearly in evidence. In the figure, the lines represent a weighted average of the data. While the fish data appear to rise relatively steadily, the sediment results show several distinct features, including a marked drop in the Upper Hudson River, a near-plateau level in the freshwater Lower Hudson River and finally a sharp rise near the salt front at RM 60. The plateau value of the freshwater Lower Hudson River is directly contrasted against the rising fish component 1 levels in Figure

K-15. The consistency of the component 1 value in the sediments versus the rising values in the fish may indicate a change in the absorption and retention of PCBs in fish in this region of the river because an additional, substantive, higher molecular weight PCB load to this region is not in evidence (USEPA, 1997). Alternatively, this may be attributable to a change in the PCB exposures to the fish resulting from the loss of the lighter congeners from the water column during transport downstream. This would yield fish body burdens which had higher molecular weight but lower total PCB mass.

Component 1 appears to closely match molecular weight. Note the similarity in the trends of component 1 versus molecular weight in fish and sediments as function of river mile (see Figure K-14 and the top diagram in Figure K-16). As in Figure K-14, the lines in Figure K-16 represent weighted averages and are used to simply illustrate general trends. Both component 1 and molecular weight show a gradual rise from the TI Pool to New York City harbor with a plateau in the freshwater Lower Hudson for sediments but not for fish. As shown in the lower diagram in Figure K-16 an enlargement of RM 155 to 75, this rise in molecular weight in fish is paralleled only by a rise in the molecular weight of the water-column dissolved-phase PCB fraction. Note the similar slope values as well as the high R^2 values relative to the other two matrices plotted.

The reason for the parallel trends in the fish and water column dissolved phase matrices in this region is unclear because, in general, the dissolved phase contains a higher proportion of less chlorinated congeners due to partitioning while the congeners in fish are more chlorinated. Most likely, the molecular weight increase in the dissolved phase is due to gas exchange plus degradative losses of the lighter dissolved congeners as well as the possible partial replenishment via the resuspension of less dechlorinated, higher molecular weight PCBs from the sediments of the Lower Hudson River, similar to interpretation of the water column data of the Upper Hudson River presented in Appendix C of the Responsiveness Summary for the LRC (USEPA, 1999a). To the extent that water column exposure to fish is important, the increase in the molecular weight of the dissolved phase combined with its absolute decline in concentration may produce the observed trends in fish body burden. Alternatively, the simple decline in water column concentrations alone with river mile would serve to decrease the overall fish exposure (resulting in lower body burdens) while raising the mean molecular weight of the mixture to which the fish are exposed (resulting in higher molecular weights).

K.4 Examination of Fish Body Burden with River Mile

The actual trends in fish body burden with river mile are explored in Figures K-17 to K-19. In these figures, 1993 fish samples are classified according to feeding guild and examined as a function of river mile. The feeding guilds designated in subchapter 2.4.2 of the ERA are used in this analysis. The classifications are forager, omnivore, semi-piscivorous, and piscivorous. It is acknowledged that feeding habits of fish may change with age and size and therefore a single species may occur in several guilds during its life history. While these divisions do not definitively categorize the fish, this classification provides a general estimate of PCB bioaccumulation potential. Table K-3 lists the

assignment of fish species obtained from the Upper and Lower Hudson River in 1993 to one of the above four feeding guilds.

The fish data were examined on both wet weight and lipid normalized total PCB bases. The trend of total PCB concentration with river mile for each of these guilds is plotted in Figure K-17 on a wet weight basis. Note that the data shown in the figure were not normalized for lipid content There are several declines in concentration along the river with a marked decline in body burden between RM 170 and 143. This parallels the substantial decline in sediment and water column concentrations that result from the introduction of the relatively clean waters and sediments of the Mohawk River to the mainstem Hudson at RM 156. The data for each guild are fitted with an exponential decay curve, yielding straight lines on the semilog plot as well as R^2 values above 0.3 for three of the four guilds. These curve fits provide a basis for comparison.

The occurrence of biomagnification of the PCB concentration across feeding guilds in not evident in these data since piscivores have average body burdens which are comparable to the omnivores and foragers. Semi-piscivores (comprised of white and yellow perch) had the highest average body burdens, between 2 and 3 times greater than the other feeding guilds. These results have potentially important implications for subsequent ecological impacts, since the data indicate that the PCB exposure to piscivorous birds and mammals will vary by as much as a factor of three depending upon the prey species. The results also show that the fish body burdens decline by more than an order of magnitude over the study area.

Although the curve fits express the general trend of declining body burden with river mile, the fits fail to capture the trend to a nearly constant average value for each feeding guild in the Lower Hudson. The one exception to the trend to nearly constant body burden in the Lower Hudson is the omnivore feeding guild. However, this may be due to lack of sufficient stations for this group in the Lower Hudson.

The differences among the guilds are largely removed by normalization of the PCB body burden to lipid content, as shown in Figure K-18. Lipid-normalization brings the four guilds into close agreement, indicating that most of the differences in body burden in Figure K-17 were the result of lipid differences among the species and not PCB exposure. Regression results for two of the four guilds improve substantively (to 0.88 and 0.64 for omnivores and semi-piscivores, respectively) while the forager group regression remains largely unchanged. The suggestion of a concentration plateau in the Lower Hudson is less evident in this diagram and its occurrence appears further downstream in the river. Again, like the wet weight data, the lipid-normalized PCB concentration itself appears to decline more than an order of magnitude across the area of study.

The consistency of lipid-normalized body burdens across feeding guilds is also seen in the molecular weight results which show a similar consistency across all of the guilds except for the omnivores. The molecular weight for the omnivores (which are represented solely by brown bullhead and white catfish) is distinctly higher than the molecular weight for the other guilds. This is illustrated in Figure K-19, which shows a

strong correlation between river mile and molecular weight for all four groups as well as close agreement among individual guilds.

The decline in fish body burden with river mile is generally consistent with the decline in the ecological and high resolution sediments obtained by the Phase 2 sampling effort (see Figure K-20) and supports the first NOAA conclusion in subsection K.2. For both the fish and the sediment, the majority of the decline occurs between RM 190 and 143. Although the sediment samples do not reflect all surface sediment concentrations in a region (the sample set is too small), they provide a general measure of the trends in fine-grained sediments with river mile. The sediment results indicate an approximate order-of-magnitude decline in the surface sediments across the study area, similar to that seen in the fish. Notably, fish body burdens and sediment concentrations appear to decline much more slowly with river mile in the region below RM 143. The occurrence of an approximately constant or slowly declining sediment PCB concentration below RM 143 is partially attributed to the lack of substantial additional PCB or flow loadings below this point. In particular, additional substantive flow would serve to dilute the sediment concentrations with clean materials from the watershed. The correlation between the decline in sediment PCB inventory and river mile was discussed at length in the DEIR (USEPA, 1997). It is simply noted here that the decline when normalized for cesium-137, a geochemical tracer (or marker) for fine-grained sediments, is proportional to the drainage area below Stillwater (RM 177).

The consistency of the molecular weight results across the feeding guilds is a corollary of the conclusion by NOAA regarding the similarity of congener pattern across species at a given station. This observation suggests that there is little biomagnification of heavier congeners via trophic level. Rather biomagnification of heavier congeners occurs independent of trophic level such that PCB body burdens in all fish have substantially higher molecular weights than the media (water and sediment) they are exposed to. Hence, fish body burdens more closely resemble Aroclors 1248 and 1254 in terms of molecular weight even though the mixtures they have been exposed to are more Aroclor 1242-like in nature.

Similarly, when fish body burdens are normalized for lipid, the differences among groups disappear, thus most of the group differences are related to guild or species-dependent lipid content and not the guild feeding preference itself. The homogeneous nature of congener patterns at a given location plus the consistency of lipid-normalized body burden across the feeding guilds suggest that fish exposure is probably the integration of both water and sediment exposures. This in turn suggests that all species are affected equally at each location by all exposure media and that their exposures do not depend strictly on position in the food web. Alternatively, PCB contamination may be so ubiquitous that fractionation within the food web is prevented. Both suggestions are consistent with the bivariate model assumptions discussed in the BMR (USEPA, 1999b). The results presented here are also consistent with the third NOAA conclusion discussed in subsection K.3.

Before leaving the topic of the variation of the fish body burdens with river mile, it is useful to examine some of the actual homologue distributions themselves. For fish, homologue patterns from the Fall 1993 collection were examined as a function of river mile. Figures K-21 to K-25 represent two typical homologue patterns from representative stations throughout the Hudson. Also shown on each figure are the homologue patterns for Aroclors 1242, 1248, and 1254. Figures K-21 to K-24 show the freshwater Hudson fish samples, all dominated by tetra homologues and yielding homologue relationships similar to those seen in Aroclor 1248. These plots also provide support for the agreement across species for each station, as concluded by NOAA (conclusion 3). These plots also show similar relationships among the homologue groups as was subsequently found by NOAA for Fall 1995 (i.e., tetrachlorohomologue dominance, followed by tri and penta groups) and also shown in Figures K-2 to K-4. Note that the fall patterns found in both 1993 and 1995 are distinct from the patterns found in Spring 1995. Spring patterns are dominated by the pentachlorohomologue group, rather than the tetra and thus have even higher molecular weights (examined via PCA later in this chapter).

It should be noted that NOAA discerned some differences among the congener patterns (as compared to the homologue patterns presented above) for 1993 and 1995 for some fish species and stations. These results are not directly supported by the principal components analyses presented later in this report, but this may be due to limitations in the number and type of data available for comparison.

The Fall 1993 homologue pattern below the salt front is distinctly different from that seen upriver. Specifically, these patterns are pentachoro-dominated, with a more significant hexachloro component and a heptachloro fraction equal to that of the trichloro fraction. Note that the white perch pattern shown is nearly identical to the mean fall pattern measured by NOAA in 1995 (see Figure K-3). Also worth noting is the fact that the Atlantic silverside, representing forager species, also has a pentachloro-dominated spectrum, unlike the forager species upstream. As noted previously, it is likely that these samples reflect the impact of additional PCB input associated with the New York City metropolitan area, also noted in the coring analysis reported in the DEIR (USEPA, 1997). Unlike the samples upstream, the samples from this region more closely resemble Aroclor 1254 than 1248, due to their higher molecular weight and greater proportion of higher chlorinated congeners.

While tetra-dominated homologue patterns were clearly in the majority in the freshwater Hudson River, a notable exception was found for catfish species (i.e., brown bullhead and white catfish) at RM 89.4, Esopus Meadows. Catfish were not typically caught at other sampling locations for comparison, but these samples, representing the omnivore feeding guild, exhibited a distinctly heavier pattern than other fish at this location (compare Figure K-26 with Figure K-24). This may be due to the nature of their feeding habits, typically associated with a sediment-based pathway. Alternatively, the penta-dominated spectra may result from differences in their metabolics rates and pathways (Yuan et al., 1997) relative to other fish caught at this location.

Figure K-27 shows the change in homologue pattern for white perch and striped bass for the entire suite of lower river stations. The increase in penta through octa chlorinated homologues relative to tri and tetra chlorinated homologues as a function of river mile is clearly evident, with the largest change occurring between RM 47.3 and RM 25.8, attributed to metropolitan New York-related PCB discharges. It is important to note here as well that migratory behavior of these fish species may be partially responsible for some of the increase in molecular weight seen in the body burdens, since time spent in New York harbor serves to expose the fish to a higher molecular weight mixture than that of the freshwater Lower Hudson.

K.4.1 Summary

Fish body burdens were shown to decline with river mile to about the same degree as the changes in the sediment PCB concentration. Similarly, molecular weight in fish samples increased with distance from the Upper Hudson River source areas. Differences in total PCB concentration among species was shown to be significant based on feeding guild (i.e., food source). However, when normalized to lipid content, the interspecies differences disappeared and the largest changes in PCB concentration coincided with river mile. Similarly, the molecular weight of the PCB body burdens in fish was not found to vary by feeding guild but simply by river mile. These results indicate that PCB uptake and biomagnification of individual congeners in fish is largely related to distance downstream of the GE facilities and not to trophic level. In addition, the reason for the increase in molecular weight with distance downstream was not known but may be attributed one or more several possible causes including decreasing importance of water column exposure for fish due to declining water column concentrations, particularly for lighter congeners. Alternatively, water column concentrations may simply become higher in molecular weight due to replenishment from less-dechlorinated, Lower Hudson sediments, yielding a higher molecular weight for water-based exposure. Lastly, metropolitan New York discharges present higher molecular weight mixtures for fish exposure in the saline portion of the lower Hudson

K.5 Examination of the Results for Benthic Invertebrates

The Fall 1993 benthic invertebrate samples collected for congener analysis were examined in a manner similar to that used for the fish data. As noted in the PCA presented in subsection K.3, benthic invertebrate results typically fell midway between the sediment and fish results at any given river mile. This was expected due to the close contact between the benthic invertebrates and the sediments, as well as the generally lower trophic level of the benthic invertebrates relative to fish. Figures K-6 to K-11 illustrate this relationship based on the principal components analysis.

The benthic invertebrate sample set was in some ways more limited, but in other ways more useful, than the results obtained for fish. Specifically, benthic invertebrates were obtained from a fewer number of stations than fish samples and therefore fewer data points are available. However, due to the nature of intimate contact between the sediments and invertebrates, the invertebrate data could be directly compared and

correlated with the sediment data, unlike the fish, whose relationship to the sediments is less well defined.

As an initial analysis, PCB body burdens in benthic invertebrates were examined as a function of river mile (see Figure K-28). The upper diagram in this figure shows the same general decline in benthic invertebrate body burden as was noted for the fish, although the decline appears to be closer to 1.5 orders of magnitude for the benthic invertebrates. Due to the relatively few distinct species/taxa analyses available for the lower river, these results are presented as general benthic invertebrate samples in this diagram. Sufficient species -specific results were obtained for the TI Pool which are plotted in the lower diagram of Figure K-28. Although many individual species were noted, no clear trend among the benthic invertebrates is evident in this diagram. Note that unlike some fish results, the benthic invertebrate results are not presented on a lipid-normalized basis. This is because in most instances the regressions did not improve. The lack of improvement was attributed to the difficulties in determining the percent lipid on what were typically very small samples.

A similar analysis was performed using the molecular weight of the benthic invertebrate samples and is presented in Figure K-29. These results also parallel the fish results, with higher molecular weights found at lower river miles. A regression between the benthic invertebrate results and river mile yields an R^2 of 0.424, indicating a significant correlation between the parameters, but not a predictive relationship. There is some suggestion in the data that the molecular weight is constant in the freshwater portion of the Lower Hudson River, but there are too few data to clearly resolve this. Again an analysis of the TI Pool on a species-specific basis does not appear to yield a discernable relationship among the samples.

Figure K-30 presents the results of a regression between the total PCB concentration in sediments and that found in the benthic invertebrates. There is clearly a parallel decline in both sediment and benthic invertebrate concentrations but the correlation remains weak. Table K-4 provides summary statistics for this comparison and shows that the stations associated with sediment PCB concentrations less than 2,000 µg/kg have statistically lower benthic invertebrate PCB concentrations relative to sediments greater than 8,000 µg/kg, based on both the arithmetic and geometric means of the two groups. Table K-4 also presents the ratio of the mean benthic invertebrate body burden to the associated sediment samples for both the Upper and Lower Hudson. Although it was not found to be statistically significant, the data suggest that the ratio of benthic invertebrate body burden to sediment is lower in the Lower Hudson relative to the Upper Hudson. This may suggest a change in the bioavailability of the PCBs downstream or else the importance of another exposure route, such as a water column-based exposure. Alternatively, differences in benthic invertebrate species or feeding guilds may also be partly responsible for the Upper Hudson-to-Lower Hudson difference.

The best correlation between the benthic invertebrate data and other parameters was found for molecular weight in benthic invertebrates and their associated sediments. Specifically, the arithmetic mean of the ecological sediment samples associated with each

station was plotted against the individual benthic invertebrate sample results. This is shown in Figure K-31. This relationship yielded an R^2 of 0.639, indicating a fairly strong correlation. Notably, the regression line falls well above a line of matching molecular weight, indicating that a molecular weight shift takes place between the sediments and the benthic invertebrates. Specifically, the results suggest that the molecular weight of the benthic invertebrates is about 20% higher than the sediments they were associated with, indicating preferential absorption or retention of the higher molecular weight congeners. The notable exception to these results was found at Station 5 at RM 189, whose data points fell about and below the match line, indicating little or no enhancement of heavier congeners at this site. As will be shown below, this site was characterized by a congener mixture which had undergone a substantive level of dechlorination. Thus, higher molecular weight congeners represented a much smaller fraction of the sediment total PCB mass.

The last comparisons concerning benthic invertebrates were based on the differences between benthic and epibenthic invertebrate samples. Epibenthic invertebrates live on or near the sediment-water interface, as compared to the benthic infaunal invertebrates which live within the sediment itself. The distinction between epibenthic and benthic infaunal is based on the location of the invertebrates within the sediments, not the species of invertebrate. Because epibenthic invertebrates do not dwell within the sediment, it might be expected that their body burdens and molecular weights would be different from other benthic invertebrates at a given sampling location. In Figure K-32, the relationship between the two invertebrate types for total PCB body burden and river mile is examined. The results suggest that epibenthic invertebrates tend to have a lower body burden than nearby benthic invertebrates, both in the TI Pool as well as in downstream locations. However, when the molecular weight results are examined, no clear difference is apparent (see Figure K-33). Note that these conclusions are based on a small set of epibenthic samples relative to the number of benthic samples.

This finding is consistent with the results obtained for the fish data, which show little apparent difference in body burden molecular weight at individual stations. This also affirms the conclusion by NOAA (1997) regarding the similarity of congener pattern across species at a given station. It should be noted that the benthic invertebrate molecular weights are typically lower than the fish from the same station, suggesting the occurrence of some biomagnification between invertebrates and fish. Alternatively, these differences may represent differences in biochemistry rather than a biomagnification process. The differences in molecular weight can be inferred from the Figures K-7 to K-11 which show a lower component 1 value for the benthic invertebrates relative to fish in all regions except the saline Lower Hudson River where the component 1 values are about the same.

Homologue distributions for the benthic invertebrates were prepared for representative stations from throughout the Hudson River, as was done for the fish results. These distributions are presented in Figures K-34 to K-37. In the first three figures, the upper diagram represents the typical pattern for the stretch of river indicated (RM 195 to 175, RM 156 to 100 or RM 100 to 60, not merely the station from which the

samples shown were taken) while the lower diagram represents one of the more atypical patterns found at this section of the river. The atypical patterns were not consistently found in any specific invertebrate classification and in fact both the typical and atypical samples consisted primarily of unsorted invertebrate samples. In these three figures representing the freshwater Hudson, the typical benthic invertebrate homologue distribution is dominated by the tetrachlorohomologue group, similar to that seen in the fish. However, the spectra tend to be shifted to slightly lower homologues, with a reduced importance for the hexachloro and higher fractions relative to the fish.

Homologue distributions for Station 18 in the saline Lower Hudson did not yield any typical patterns and thus all 5 results are presented. These distributions are clearly heavier than those seen upstream, with two spectra clearly dominated by the hexachlorohomologue group. These results are typically heavier than for the fish from the same region (see Figure K-25), suggesting that the PCB releases associated with the New York City metropolitan area may impact the sediment-based biota more directly than the upstream PCB loads generated by GE and the Upper Hudson sediments. This might be expected since the heavier congeners associated with the New York City metropolitan area would have substantially higher partition coefficients and thus produce little increase in a dissolved PCB fraction relative to the upstream loads (USEPA, 1997).

Figure K-38 compares two homologue distributions from Station 5 in the TI Pool at RM 189. This is the station whose data points lay below the equal molecular weight line in Figure K-31. At this station, both the sediments and the benthic invertebrates are characterized by a highly dechlorinated homologue spectra, typical of many sediments found in this region. In this instance there appears to be little shift in molecular weight, perhaps because little bioaccumulation occurs for the lighter congeners present in these sediments. In this instance, the body burden for the benthic invertebrates would simply reflect the sediments directly, with no modification of the congener pattern. This congener pattern cannot be explained by the species collected because none of the species sampled at this station were unique.

K.5.1 Summary

In this subsection, benthic invertebrate data were examined and shown to be similar to the results for fish for much of the Hudson River. Benthic invertebrates in the freshwater Hudson River typically have lower molecular weights than the fish from the same location, but have higher molecular weights than the sediments in which they live. Benthic invertebrate body burdens decline with river mile. Benthic invertebrates in the saline Lower Hudson distinctly show the impact of the New York City metropolitan area inputs. These invertebrates have a substantially higher molecular weight than that of the Upper Hudson River. Epibenthic invertebrates appeared to have lower body burdens but similar molecular weights relative to other benthic invertebrates collected from the same station. This suggests that the bioaccumulation process may be dependent on PCB congener type or perhaps molecular weight.

K.6 Comparison of the 1993 and 1995 Hudson Fish PCB Congener Results

The 1993 and 1995 data sets represent the most detailed congener-specific fish data available for the Hudson River. During the years between the two sampling surveys, several changes occurred at the General Electric (GE) facilities, resulting in marked declines in the PCB loads originating from these locations. With this reduction in water column concentration and annual transport of PCBs between 1993 and 1995, changes to the expected average congener composition as well as body burden in the fish were likely. To the extent that these reductions were important to the body burdens present in fish, there should be a resulting change in congener pattern between the 1993 and 1995 fall sampling surveys. For this reason, a comparison was made between the 1993 and 1995 fish results based on the 29 congeners selected via the optimization process described in subsection K.3. Specifically, a PCA was performed to discern differences in the congener patterns between the two data sets. All 1993 mainstem Hudson fish samples were included in the analysis. The 1995 data used all NOAA samples except the 11 samples excluded by NOAA based on their outlier analysis. As in the principal component analysis described in subsection K.3, the results for the 29 congeners were normalized to the total sample mass and then standardized by congener. This PCA yielded a different set of principal components relative to the ones discussed previously since it was based on a different data set. Since the basis for the analysis was simply to discern differences between the two data sets as a possible measure of trends in the fish contamination, no interpretation of the principal components themselves was performed.

Figure K-39 presents three diagrams of components 1 and 2 color coded to show molecular weight, river mile, and sampling time. The data themselves form an inverted "V" pattern. In the first two diagrams, banding of the data based on the third variable (i.e., molecular weight or river mile) is clearly evident, reflecting the ability of these components to recognize congener pattern variations which correlate with these important variables. In the third diagram, some separation is evident among the three sampling periods, although many samples coincide, suggesting that the differences among the sampling periods may not be substantive. There does appear to be a large portion of the Fall 1993 data set (i.e., the left branch of the data set) which is not matched by conditions in Fall 1995. However, when the second diagram in the figure is reviewed so as to define the nature of the samples in this region, the reason for the lack of overlap in this region in Fall 1995 becomes clear. No Fall 1995 samples were collected above RM 152. Hence, the samples on the left branch, representing Fall 1993, have no point of comparison in Fall 1995. Nonetheless, there does appear to be some differences between the fall and spring conditions in the Lower Hudson River, as mapped onto the right branch of the "V." In particular, the Fall 1995 and Spring 1995 results appear to fall farther down the right branch (i.e., higher molecular weight for the same river mile) relative to Fall 1993. However, this difference appears to be a rather subtle one given the degree of scatter evident in the diagram. For this area, a subsequent comparison was performed.

In Figure K-40, four diagrams representing four different species are plotted using color coding for the sampling period. In each instance there appears to be some differences which might be attributed to sampling period, perhaps representing temporal change. The actual temporal differences become clearer when life stage is considered, as shown in Figure K-41. Specifically, for largemouth bass, represented by the first diagram in both Figures K-40 and K-41, it appears that the apparent differences between the Fall 1993 and Spring 1995 samples are confounded by the near exact coincidence of life-stage, such that the differences noted by the PCA might be due to either factor. In light of the likely impact of life-stage on PCB absorption and retention, it is unclear which factor is of greater importance in this instance.

Similarly, for the second diagram in Figures K-40 and K-41, representing striped bass, life-stage is again coincident with much of the temporal difference noted. In this instance, there may actually be a discernable difference between the Fall 1993 results and the Fall 1995 results since there is a portion of both data sets based on the same life-stage which appear to be different. The Spring 1995 to Fall 1995 data also appear to be slightly different for the striped bass adults although it is unclear if the difference will be important based on this analysis.

Lastly, both the yellow perch and the white perch PCB analyses were performed on the same life-stage in all sampling periods, thus removing this confounding factor from the data for these species. For white perch, no discernable difference is apparent based on these principal components. Similarly, for yellow perch, there does not appear to be any consistent difference among the various sampling events.

K.6.1 Summary

Combining the results of Figures K-39, K-40 and K-41, there appears to be a minor shift toward higher molecular weights (i.e., heavier congeners) from Fall 1993 to Fall 1995 and Spring 1995.The shift appears to be much greater for the Fall 1993 to Spring 1995 sampling than from Fall 1993 to Fall 1995. Based on the last diagram in Figure K-39, the Spring 1995 results also appear to have a higher molecular weight than that for Fall 1995. These general trends were also noted in the NOAA report (1997) based on several individual congeners. However, these conclusions must be tempered by the confounding factor of life-stage which was also shown to coincide with changes in molecular weight. Based on these results plus the direct homologue comparisons provided in Figures K-2 to K-4, it appears likely that seasonal variation in fish body burden does occur, with heavier molecular weights coinciding with the spring. On the other hand, there does not appear to be a systematic change in the fall conditions in 1995 relative to Fall 1993. There may be some decline in a few specific congeners, but as shown later, some of these congeners may reflect a complexity in their biogeochemistry which precludes their use as simple markers for recently released PCBs.

K.7 Characterization of the Fish Body Burdens in the Hudson River

The characterization of the fish body burdens in the Hudson River has been discussed throughout subsection K.3. Many of the figures of this subsection describe both the samples as well as standard Aroclor mixtures as points of reference. In particular, Figures K-6, K-12, and K-21 to K-26 all document the general close agreement between fish tissue congener patterns and those of Aroclor 1248. However, this is not a perfect match and many congeners are present at higher and lower proportions than those found in Aroclor 1248 (Figure K-12). The agreement for the saline Lower Hudson River is even worse, with a relative "best" fit between Aroclor 1254 and the fish samples of this region. However, most of these comparisons were made on a homologue basis or based on the 29 congeners which were optimized for all matrices. This subsection will examine fish congener patterns using PCA and the set of 46 congeners which were selected by NOAA to best represent the congener patterns in fish alone. This analysis will serve to further support the previous assessment as well as more closely examine the nature of the congener patterns found in Hudson River fish.

To emphasize the broad spectrum of congeners represented by the fish samples and support the need to examine a larger set of congeners, Figures K-42 and K-43 were prepared. In each figure, the mass fraction of each congener detected in two typical fish samples from the Upper Hudson and saline Lower Hudson as well as a water column sample taken at Rogers Island during the spring flood is compared with the congener mass fractions present in several Aroclor standards. The water column sample results shown represent the upstream source pattern originating from the GE Hudson Falls Plant Site. Figure K-42 uses a linear scale and provides a basis to assess the center of the various distributions with respect to mass fraction. From this figure, it is evident that the fish of both the Upper and Lower Hudson River represent a far greater range of congeners than can be attributed to a single Aroclor. The apparent agreement between Aroclor 1248 and that of the Upper Hudson River fish has to do primarily with the coincidence on their distributions since may individual congener proportions are different. This issue probably results from several factors, including the release of Aroclors other than Aroclor 1242 from the GE facilities. (The DEIR documents the presence of Aroclor 1254 and 1260 in the congener pattern at Rogers Island.) Additionally, bioaccumulation serves to enhance some congeners relative to others so that the proportions among many congeners will change as a result and mask the original mixture from which they were derived. Thus there is no exact fit. Nonetheless, it is possible to represent the presence of most, if not all, congeners present in the fish by some linear combination of Aroclor standards, if this is desired. Alternatively, the fit based on homologue or mean molecular weight may be sufficient.

Figure K-43 represents the same information as Figure K-42, except on log-scale. This figure emphasizes the broad range of congeners present at low levels in fish throughout the Hudson River. This range is much greater than that seen in any Aroclor and it is even greater than that seen in the congener spectrum at Rogers Island. To some degree, the differences between the congeners present at Rogers Island and those in the

Upper Hudson River largemouth bass are due to the biomagnification of the low concentration, more chlorinated congeners from the sediments and water through the biological uptake process(es). While the uptake process(es) serves to document the presence of these congeners in the Hudson by enhancing their concentrations to detectable levels, it also serves to distort their ratios. Thus, assigning the fish congener pattern to a source based on the observed patterns in fish, sediment and water in a simple, linear fashion (i.e., by proportion or ratio) becomes an extremely difficult, if not impossible task. Since the individual congener toxicities are not well known, it is unclear whether these levels represent important concentrations. The issue of assigning toxicity to the suite of congeners present in fish cannot be directly addressed in this report due to the lack of data. This discussion acknowledges the uncertainty associated with the lack of information.

Results of a third principal components analysis are described below. This analysis used the 46 congeners selected by NOAA on the basis of frequency and mass fraction in fish alone and was performed for the following reasons:

- To further characterize the congener mixtures present in the fish in terms of closest Aroclor;
- To examine the validity of the apparent Aroclor 1260 end member present in Figure K-6; and
- To address the potential concern that the 29 optimized congeners might not completely characterize the congener spectra.

The data for the 46 congeners from both the 1993 and 1995 fish sampling events were included in this analysis. The data were normalized and standardized as described previously in this Appendix. The end result yielded the diagram given in Figure K-44. This diagram has a similar shape to that seen in Figure K-39 which used only the 29 congeners. Also shown on Figure K-44 are the five Aroclor standards originally shown in Figure K-6. Utilizing the 46 congeners, it becomes clear that the fish data do not exactly resemble any Aroclor, that is the fish data do not cluster around any single Aroclor standard. Additionally, the fish results do not trend toward Aroclor 1260 as suggested by Figure K-6 but rather vary between what appears to be a set of biologically-derived mixtures, which have substantively different patterns relative to the original source materials (i.e., sediments and water). Most importantly, the data suggest that the bioaccumulation process serves to create essentially unique end members which result from the preferential absorption and depuration of various congeners. This becomes clearer when results for the Lower Hudson alone are viewed (see Figure K-45). The results for the Lower Hudson represent the right branch of the distribution in Figure K-44. These data appear to trend between two undefined congener pattern end members, neither of which is a good approximation of an Aroclor in this principal component space.

The unique congener composition of the fish samples is emphasized in Figures K-46 and K-47 which show the fish and surface sediment results for the Upper and Lower Hudson, respectively. The sediment data on these diagrams has not been included in the

original PCA analysis itself, but rather, like the Aroclors, simply been translated into the fish-based principal components space. This permits a simple, general comparison of the fish and sediment patterns. In both sections of the river, the fish data are almost completely separate from the sediment data in this principal component space. This serves to emphasize that the uptake and depuration processes within the fish significantly alter the congener composition of the fish body burden relative to the congener pattern in local contamination.

To reaffirm the comparison of the three fish sampling events, Fall 1993, Spring 1995 and Fall 1995 using the 29 congeners, the data were again examined using the 46 congeners. The three fish sampling events are represented by different symbols on Figures K-44 and K-45. These results can be compared to Figure K-39 which was based on the 29 congeners selected by optimization. This comparison will focus on the Lower Hudson since that is where the majority of the 1995 samples were collected and thus provides the best basis for comparison. A review of Figure K-45 suggests that there may be some differences in congener pattern among the sampling events.

Loadings for principal components 1 and 2 developed from the 46 congener data set are presented in Figure K-48. These principal components are developed from the fish data only and are thus different from the principal components presented for the entire 1993 data set although they share some similarities (see Figure K-5). The first principal component developed from the 46 congeners represents the ratio of the tri and tetrachlorocongeners to the penta and higher congeners. This component is related to the mean molecular weight of the sample and is similar to the first principal component presented in Figure K-5. The major difference is the split point, which is between tri and tetra in the first analysis and between tetra and penta in this analysis. The second principal component based on the 46 congeners shows a different emphasis than the one based on the 29. Specifically, the second principal component focuses on several of the heavier congeners which may undergo alteration. Many of these congeners have been identified by Brown et al. (1997) as possible tracers of modified mixtures. Several of these congeners are discussed later in this appendix. This second principal component, based on the 46 congeners in fish alone, differs from the one developed for the 29 congeners for four Hudson matrices. The latter component is focused on the proportions of dechlorination products while the former focuses on congeners which may be lost via that process. Part of this change in focus is the result of the differences in the congener selection itself. This is discussed in subsection K.3.

In order to best discern any differences in congener pattern due to changes in the fish exposure (i.e., river contamination) over time, it is best to isolate other confounding variables such as river mile, species and life stage. In this manner, the likelihood that any measurable differences are due to temporal changes is increased. To this end, only two species of fish were obtained from the same locations during all three sampling events, white perch and yellow perch.

Figure K-49 presents the principal components results for yellow perch in the Lower Hudson between RM 110 and 154. Clearly evident in this figure are the

differences between the samples collected in the spring of 1995 and those of the other two sampling events. Differences between the spring and fall conditions were noted and discussed earlier in this appendix as well as by NOAA (1997). The results for the two fall sampling events appear to occupy the same region of the diagram and thus do not appear to be different. At a minimum, it is clear that the year-to-year difference in yellow perch between the fall sampling conditions was much less than the seasonal differences in 1995. This would indicate that the remedial efforts made by GE between 1993 and 1995 did not greatly affect the nature of PCBs (i.e., the congener pattern) to which fish were exposed in the Lower Hudson up to this point.

Figure K-50 presents the results for white perch at three locations in the Lower Hudson. The first location, represented in the top-most diagram in the figure, matches the yellow perch locations just presented. Here again, there is a large difference between the spring and fall conditions but little between the two fall sampling events. This supports the conclusion drawn from the yellow perch. Farther downstream near RM 59, there is a distinct difference between the two fall sampling events. This change in congener pattern is attributed to the change in fish body burden (normalized to lipids) at this location. The 1995 fish have body burdens on average 73 percent higher than the 1993 results but this difference is not statistically different, largely due to the small number of samples. The last diagram compares white perch in the saline Lower Hudson. In this case, the scatter is quite large and no discernable difference is evident.

Based on these results, it is clear that a substantive change in congener pattern occurs between spring and fall conditions, as discussed in subsection K.2. This change, probably seasonal in nature, is clearly much larger than the temporal difference between fall sampling events. The change at RM 59 for white perch was coincident with a large gain in fish body burden which may simply represent natural variability. Notably the large gain in body burden indicated only a small shift in molecular weight (less than 3 percent), suggesting that the body burden was derived from the same material in both sampling events. Note that although principal component 1 is strongly affected by the molecular weight of the sample, principal component 1 is different for 1993 and 1995 for these samples even though the molecular weights are nearly the same. There is a correlation between principal component 1 and molecular weight, but this is not shown in every case.

The factors responsible for the differences noted were examined on a limited basis. Specifically, the fish data for the region between RM 110 and RM 154 were examined based on molecular weight. These results are presented in Figure K-51. Like the principal components themselves, the molecular weight results show a spring to fall difference but little difference between the two fall sampling events.

K.7.1 Summary

Using the 46 congeners to examine the three sampling events largely confirmed the prior analyses performed by NOAA (1997) as well as the analyses in previous parts of subsection 3.3 of this report. In particular, spring conditions were distinctly different

from those of the fall based in this principal components analysis. Little difference was evident between the two fall sampling events, suggesting that little had occurred (such as GE remediation which substantially lowered the water column concentration of PCBs and reduce or eliminate the water column pathway for the fish) to affect the basic routes of exposure in fish. Alternatively, the lack of difference in fall conditions may be partially the result of the bioaccumulation processes which simply serve to create the same general congener pattern in the fall, so long as exposure routes and congener concentrations are approximately the same. NOAA (1997) did note some differences between the 1993 and 1995 fall sampling events but, based on the principal components analysis presented here, these differences were relatively minor and not resolvable via these techniques.

Congener patterns found in fish do not match identically with any standard Aroclor. Upper Hudson fish body burdens most closely approximate Aroclor 1248, although nearly all fish samples have higher moelecular weight congeners present, indicating the presence of a significant fraction of Aroclor 1254. In the Lower Hudson, fish body burdens show a broader range, with some locations more closely related to Aroclor 1254 than 1248. Nonetheless, it is still these two Aroclors which most closely resemble the fish body burdens.

K.8 Examination of Recent Trends in Fish and Water Column PCB Concentrations

Since the inception of the Phase 2 investigation, PCB releases from the General Electric Hudson Falls facility have increased and decreased by more than an order of magnitude, based on weekly monitoring by General Electric. During this period, fish body burdens responded to the increased concentrations, as might be expected. As a result of the remedial activities undertaken by GE at the Hudson Falls plant, water column loads and concentrations have returned to the levels seen in the late 1980's (when the Reassessment was initiated). As shown in Figures K-52 through K-54, fish concentrations have responded to the decline as well, returning to the conditions seen in the late 1980's. Thus, current fish contamination levels are where they were when the Reassessment began.

In light of these developments, USEPA has reviewed the long term fish monitoring results obtained by NYSDEC and long term water column monitoring results obtained by the USGS. These results are presented in Figure K-55 for three different fish species.

The data have been filtered so that the sampling season for fish and water are concurrent. For pumpkinseed, only the August-September collections are shown (which eliminates 1991-1992 data), eliminating seasonal variability and age related differences from the data trend. For largemouth bass and brown bullhead, only the May-June samples are shown. Further, an attempt has been made to correlate observations with nearly coincident (instead of whole summer) water column concentrations: The May-June data are plotted against the April-June water column concentration average, while the August-

September data are plotted against the July-September water column average. No data on July-September concentrations are available for several years at Stillwater in the 1990's. A thorough discussion of the extent distribution and quality of the fish data is presented in the BMR (USEPA, 1999b).

Each point on the graphs represents an arithmetic average of lipid-normalized total PCB concentration in fish and an arithmetic average of the total PCB concentration in the water column. Pre-1983 fish data are corrected to a consistent 1983-89 basis. Note that some potential inconsistencies remain in the interpretation of analytical methods from the 1990's due to changes in the analytical laboratories used by NYSDEC. The correction of data to a consistent basis and the interpretation of analytical methods are addressed in detail in the Baseline Modeling Report (USEPA, 1999b). Each point is labeled to show location and year; e.g., “88 (red) " indicates RM 189 - 193 for 1988. Spatially, four groups are as defined in the Preliminary Model Calibration Report (PMCR) (USEPA, 1996). Group 1 is the TI Pool (RM 189-193); Group 2 is the Stillwater area (RM 168-175); Group 3 is above Federal Dam near Waterford (RM 160), and Group 4 is below Federal Dam near Troy (RM 142-165). Water column concentrations appropriate to each station were estimated from USGS observations at Stillwater and Waterford with adjustment for dilution. The adjustment procedure is discussed fully in the PMCR (USEPA, 1996).

Concentrations are shown on logarithmic axes, thus, while a trend is evident, the predictive power of the relationship is not very strong (on linear axes the data is scattered). On an arithmetic scale, the relationship approximates an exponential form, with significant “above the line” scatter although in each case the correlation of the mean values was found to be statistically significant. There also appear to be systematic differences by station. For instance, in the plot for largemouth bass, the TI Pool samples almost entirely lie above samples from other stations. It is suspected this may be due to greater exposure to concentrated sediment stores in the TI Pool which may not occur downstream.

Also evident in Figure K-55 is that recent results (1990-present) do not follow a simple extrapolation of the body burden to water column relationship which was apparent in the late 1970's and early 1980's. Rather, the more recent results yield positive deviations (i.e.,higher body burdens than might be expected), possibly suggesting an increased importance of local sediment contamination to current fish body burdens relative to water column-based exposures.

The results demonstrate the relationship between the fish body burdens and the water column concentration. However, further interpretation is limited due to several issues. Specifically, the number of samples in each annual average varies from year to year, the range of values associated with the individual annual means is often large relative to the mean, and the variability of the data is potentially increased by uncertainty in both PCB and lipid quantitations. Water column concentrations are also both imprecise in terms of individual patterns and highly uncertain as averages (due to limited sampling). Additional systematic variability may be caused by the range of ages of individual

samples and the inclusion of both male and female fish in a mean value. For these and other reasons it is not wholly appropriate to place a regression line through the data, nor could a regression line be interpreted to be a reliable quantitative predictive tool.

These issues are assessed and resolved to the extent possible in the bivariate bioaccumulation factor (BAF) analysis presented in Book 3 of the Baseline Modeling Report (USEPA, 1999b). In this analysis, several analytical uncertainties are addressed directly, reducing some of the data variance. Nonetheless, it is clear from the results that both fish and water column concentrations have declined with time in a parallel fashion. Within any of the river sections defined as above, the most recent fish data presented in Figure K-55 (1996) indicate levels similar to those seen in the late 1980's. Subsequent analysis presented in the Baseline Modeling Report shows lipid-normalized concentrations at the lowest levels ever measured for some Upper Hudson species.

K.9 Examination of the "H, H'" Ratios As Potential Markers for Recently Released PCBs

Recent work by scientists at General Electric (Brown et al. 1997) suggested that certain congener ratios (labeled "H, H'" ratios by the authors) could be used as tracers or "fingerprints"for establishing the source of PCBs to Hudson River fish. These ratios might then also be used as model calibration targets, providing additional verification of the models. In this appendix, several of the ratios proposed by General Electric are examined in this context using the ecological and high resolution coring data. These congener ratios are examined for fish, sediments, benthic invertebrates, water column suspended matter and water column dissolved phase PCBs. Although aquatic receptors are exposed to whole water, the suspended and dissolved phase are analyzed separately in an attempt to determine the route of contamination.

In Brown et al. (1997), ratios for tetrachlorocongeners, pentachlorocongeners, and hexachlorocongeners are proposed as possible tracers. In each ratio, a congener from the associated homologue group is identified as minimally affected by in situ alteration processes. This congener forms the denominator in all ratios for the homologue group. Other congeners in the homologue group are identified as subject to these processes and then used in the numerator of the ratio. In this manner these ratios should provide markers for PCBs which were subject to large degrees of dechlorination relative to the fresh releases from the GE facilities. Thus, fish which were exposed to recently released PCBs should have relatively high values for these ratios. Fish exposed to altered mixtures should have lower ratios.

This approach is predicated on an important assumption. Specifically, it assumes that there is little modification of the ratio by other processes including the absorption by the fish itself. Essentially, this assumption requires that the two congeners in the ratio have identical physical parameters, such as the sediment-water partition coefficient and the Henry's law constant. It also assumes that the rates of absorption, metabolism and depuration by the fish are identical for the two congeners. As will be shown later, this assumption may not be valid for all congener ratios.

This examination was limited to four congener ratios, representing tetrachlorohomologues only. These ratios have been discussed by GE in comments on other Phase 2 reports and thus were deemed most appropriate for this examination. These ratios also represent the most important homologue group to the fish in terms of mass fraction, i.e.,tetrachlorohomologue. Specifically the ratios of BZ#56, 60, 66 and 74 to BZ#49 were examined using the fish, benthic invertebrate, sediment, suspended and dissolved phase water column samples. These ratios were calculated for the entire set of mainstem Hudson River samples for the five matrices. Only detected results were used. Samples which were nondetect for BZ#49 were excluded entirely while individual ratios were excluded when the numerator of the ratio was nondetect.

The results for the four ratios in the five matrices are plotted in Figures K-56 to K-59. In each figure, the center diagram represents the 1993 USEPA and NOAA fish results. The sediment diagram represents both the ecological sediment samples (0-5 cm) as well as the high resolution core results (0-8 cm).These sediment samples are considered representative of fine-grained sediments in the region from which they were collected. The samples may not be representative of all sediments in that reach with respect to these ratios. The water column data represent the transect samples only since these are the only samples with true dissolved phase and suspended matter analyses. The flow-averaged water column samples could not be used reliably due to the nature of their handling. (The flow-averaged samples provide a measure of whole water conditions only.) All diagrams in these figures contain a solid line which represents a weighted mean of the data. The figures also contain a dashed line representing the ratio in Aroclor 1242 which is an approximate surrogate for the GE source ratio (USEPA, 1997).

A review of these figures shows several general trends which apply across all the ratios. In each instance, the fish results show a gradual decline with river mile to a very low ratio value. The decline itself appears to be exponential in nature although this was not confirmed. In all instances, the maximum value occurs in the TI Pool while the minimum value occurs in the saline Lower Hudson although the value is frequently nearly constant between RM 90 and RM 27. The high end value associated with the TI Pool is typically at or above any other matrix.

Similarly, the benthic invertebrate results also show a decline with distance downstream, largely paralleling the decline in the fish ratio. The benthic invertebrate results are somewhat limited due to fewer samples but are consistent in this parallel, declining trend. Typically, benthic invertebrate ratios are lower than those seen in fish in the TI Pool while falling at or below the fish values in the areas farthest downstream.

Behavior which parallels that seen in the biota matrices is typically only found in the dissolved phase water column data. Specifically, the ratios in the dissolved phase generally decline with river, yielding values which are close to the ratios seen in the biota. Ratio values for the dissolved phase are consistently lower than those of the suspended matter, an unanticipated result, since these congeners were selected with the intention of minimizing differences among the congeners in terms of partitioning

between particles and water. The observation that the dissolved phase ratio is always less than that observed in the suspended matter implies that the partitioning behavior of the two congeners is not the same. Typically, the ratio in the dissolved phase is about half that seen in the suspended matter, implying that the partition coefficient for BZ#49 is about twice that of the other congeners. Thus the absorption of the two congeners in each ratio from the water to other media will yield significant modification to the ratio, roughly a factor of two. Since these ratios typically only vary about a factor of three or four in range, this ratio modification process may represent a significant portion of the variation.

A related issue in this regard can be seen in the TI Pool results for the four ratios in fish. Specifically, the ratios measured in fish at this location ought to represent a minimal degree of alteration. At a minimum, the upper bound on the ratios seen in fish in the TI Pool ought to represent the unmodified mixture, typically seen at Rogers Island, while other samples may fall below due to local influences, such as altered sediment. A comparison of the water column and fish ratios for the TI Pool shows a great deal of scatter among the ratios for both fish and water. Nonetheless, the mean condition can be used for comparison. In the case of the ratios 56/49 and 60/49, the fish ratios fall between the suspended and dissolved phase ratios, as might be expected since the combination of these two fractions should generate the whole water ratio which would lie between them. However, the remaining two ratios do not have the same relationship. Specifically, the 74/49 ratio in fish closely matches that of the suspended matter while the 66/49 ratio falls above both water phases. These results indicate that processes other than simple absorption are modifying these ratios right from the outset.

In reviewing the two remaining matrices, sediment and suspended matter, the diagrams show only minor variation with river mile and no consistent downward trend. If the sediments were the exclusive source of these congeners to the fish then the ratios in fish should at least parallel the trends seen in the sediments. This is clearly not the case. In the sediments only the ratio 60/49 declines with river mile and the decline is far less than that seen in the fish. Additionally, the fact that the four congener ratios show different trends with river mile is indicative of the fact that each is affected differently by the various biogeochemical processes operating in the river. Thus it becomes unclear which of these ratios is the “correct” one to relate the fish ratios to the original source.

The lack of correspondence of the suggested congener ratios among the various matrices examined precludes their use as simple tracers for the source of PCBs to the fish. While there is no doubt that these ratios reflect the source materials to some degree, subsequent modification by absorption, partitioning and other processes serves to modify these ratios, effectively erasing the “fingerprint” of the source material. This is consistent with the results of the principal components analyses presented earlier in this appendix which show that the fish congener burden does not resemble any of the known source materials. The lack of resemblance was attributed to the modification of the fish congener pattern by the uptake and depuration processes, preferentially absorbing some congeners over others and resulting in congener patterns unique to fish. Presumably, these same processes serve to modify the individual congener ratios discussed above.

It may be possible to develop a model specific to the five congeners examined here so as to understand the various processes which affect them but this model would probably be no less sophisticated than those already developed to simulate total PCBs in the BMR. The complexities of PCB biogeochemistry suggested by the poorly understood variations in these ratios prevents a simple interpretation of the data and limits the usefulness of these ratios in understanding the sources of PCBs to the fish. As part of the modeling analysis, a set of five congeners, representing a wide range of PCB properties will be simulated as an aid to understanding the various biological uptake processes as well as potentially assisting in identification of PCB sources to fish.

K.10 Examination of the Correlation Between Dechlorination and Sediment PCB Concentration

As has been extensively discussed in the DEIR and the LRC, the sediment PCB concentrations of the Hudson River exhibit a characteristic relationship with the degree of dechlorination. This relationship was developed as part of the high resolution core analysis presented in the DEIR. As discussed in the LRC, the low resolution coring data was consistent with, but could not confirm the high resolution coring results due to cross-contamination issues as well as sediment homogenization over large vertical intervals. However, the sediment samples collected as part of the ecological sampling program do not have these same concerns and can be used to provide further confirmation of the high resolution coring results with regards to dechlorination. Specifically, the ecological sediment sampling thickness (5 cm) is only slightly thicker than the high resolution core slicing intervals (2 and 4 cm); the ecological sediment samples are collected from throughout the Hudson River, representing a large range in PCB concentration conditions; and finally, since the ecological sediments were collected from 0 to 5 cm utilizing 2.5 in. manual coring tubes, there is little likelihood of cross-contamination during the sample process.

Given the above, the dechlorination measures used in the high resolution coring analysis were calculated for the ecological sediment samples. These results were then plotted against total PCB concentration as shown in Figure K-60. In the diagram, the slope obtained from the ecological samples agrees with that obtained from the high resolution cores. The agreement indicates the absence of the cross-contamination issues which plagued the low resolution coring efforts as well as supporting the original high resolution finding with a new set of data.

K.11 Conclusions for Appendix K

- The PCB mixture contained in the fish of the Hudson River can be best characterized as an Aroclor 1248-type mixture in the Upper Hudson with a trend toward a heavier mixture (i.e., Aroclor 1254) in fish from the freshwater Lower Hudson and the harbor. These congener mixtures do not imply the increased presence of these Aroclors in the freshwater Lower Hudson but rather are indicative of the enhanced

bioaccumulation of the heavier congeners contained in the mixture released by GE. For the purposes of toxicity assessment, Upper Hudson fish are best classified as containing Aroclor 1248, based on the molecular weight and homologue patterns contained in the fish. Similarly, Lower Hudson fish are best classified as containing a mixture of Aroclors 1248 and 1254.

- The PCB body burden of the benthic invertebrates is intermediate between the sediments and fish body burdens based on congener pattern. These benthic invertebrates are still most similar to Aroclor 1248 although less so than the fish. A principal components analysis showed a slightly closer association of the sediments and benthic invertebrate congener pattern. Similarly, the magnitude of the benthic body burdens is seen to vary with the sediment concentrations, with lower body burdens associated with lower sediment concentrations.

- Examination of fish congener patterns using principal components analysis showed that the fish are distinct from their exposure media, in that a readily discernable molecular weight and congener pattern shift occurs with the accumulation of PCBs. This shift increased with decreasing river mile despite the overall decrease in fish body burden. Specifically, an enhancement of the proportion of heavier congeners (penta- and hexachlorohomologues) occurs at the same time that the fish body burdens decline. This occurs despite a much smaller change in the congener composition of the sediments. Changes in water column concentrations may be partially responsible for the enhanced molecular weight in fish, largely attributed to the loss of the lighter congeners from the water column during transport from the Upper River, and not to the introduction of additional heavier Aroclor mixture to the freshwater Lower Hudson. The principal component analysis also shows that benthic invertebrates results typically lie part way between the fish and sediment domains, as might be expected based on trophic level.

- Fish body burdens decrease downstream of the GE facilities, regardless of species. However, the congener properties do not remain constant and the fraction of higher molecular weight congeners increases with decreasing river mile.

- The ratios of BZ#56, 60, 66 and 70 to 49 were examined for several different matrices with the intent of using these ratios as tracers or “fingerprints” of the PCB sources to the fish. These ratios exhibited a large degree of variation in fish which was not shown to occur in any other media. Additionally, comparison of dissolved and suspended matter ratios suggested that the geochemistries of these congeners are not identical and may be different enough to preclude their usefulness as tracers. Overall, these ratios showed a general decline in fish with distance downstream although the ratios themselves were only somewhat similar to those seen in the dissolved phase water column and were distinctly lower than downriver sediments. These poorly understood variations in the ratios preclude their use as tracers. Essentially, the environmental modifications, particularly those produced by fish, serve to erase the “fingerprint” of the original PCB source material. Ultimately, the ratios found in fish (and benthic invertebrates) were unique to the biota, and provided

little clue as to the nature of the source.

- Fish body burdens show a return to late 1980's conditions, in fact, for a few fish species at some locations the body burdens in 1996 are as low or lower than concentrations in 1988.

- Using two different sets of congeners, principal components analysis was used to compare the 1993 and the 1995 fish congener patterns. Using the larger of the two congener sets (46 congeners), the analysis largely confirmed the prior analyses performed by NOAA (1997) as well as in previous subsections of this report. In particular, spring conditions in 1995 were distinctly different (higher molecular weight in spring) from those of the two fall sampling events. Little difference was evident between the two fall sampling events, suggesting that little had occurred (such as GE remediation of the Hudson Falls releases) to affect the congener patterns and, by inference, the basic routes of exposure in fish. Alternatively, the lack of difference in fall conditions may be partially the result of the bioaccumulation processes which simply serve to create the same general congener pattern in the fall, so long as exposure routes and congener concentrations are approximately the same.

- The ecological sediment data confirm the molecular weight (AMW) vs Total PCB concentration relationship developed from the high resolution cores.

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Table K-1
Congeners Selected by NOAA for Statistical Analysis of 1993 and 1995 Fish Samples
(NOAA,1997)

Congener Number	Homologue Group	Congener
BZ#17	Tri	2,2',4-Trichlorobiphenyl
BZ#18	Tri	2,2',5-Trichlorobiphenyl
BZ#19	Tri	2,2',6-Trichlorobiphenyl
BZ#22	Tri	2,3,4'-Trichlorobiphenyl
BZ#26	Tri	2,3',5-Trichlorobiphenyl
BZ#27	Tri	2,3',6-Trichlorobiphenyl
BZ#28	Tri	2,4,4'-Trichlorobiphenyl
BZ#31	Tri	2,4',5-Trichlorobiphenyl
BZ#42	Tetra	2,2',3,4'-Tetrachlorobiphenyl
BZ#44	Tetra	2,2',3,5'-Tetrachlorobiphenyl
BZ#47	Tetra	2,2',4,4'-Tetrachlorobiphenyl
BZ#49	Tetra	2,2',4,5'-Tetrachlorobiphenyl
BZ#52	Tetra	2,2',5,5'-Tetrachlorobiphenyl
BZ#53	Tetra	2,2',5,6'-Tetrachlorobiphenyl
BZ#56	Tetra	2,3,3',4'-Tetrachlorobiphenyl
BZ#66	Tetra	2,3',4,4'-Tetrachlorobiphenyl
BZ#70	Tetra	2,3',4',5-Tetrachlorobiphenyl
BZ#74	Tetra	2,4,4',5-Tetrachlorobiphenyl
BZ#75	Tetra	2,4,4',6-Tetrachlorobiphenyl
BZ#84	Penta	2,2',3,3',6-Pentachlorobiphenyl
BZ#85	Penta	2,2',3,4,4'-Pentachlorobiphenyl
BZ#87	Penta	2,2',3,4,5'-Pentachlorobiphenyl
BZ#91	Penta	2,2',3,4',6-Pentachlorobiphenyl
BZ#92	Penta	2,2',3,5,5'-Pentachlorobiphenyl
BZ#95	Penta	2,2',3,5',6-Pentachlorobiphenyl
BZ#97	Penta	2,2',3',4,5-Pentachlorobiphenyl
BZ#99	Penta	2,2',4,4',5-Pentachlorobiphenyl
BZ#101 with BZ#90	Penta	2,2',4,5,5'-Pentachlorobiphenyl
BZ#105	Penta	2,3,3',4,4'-Pentachlorobiphenyl
BZ#107	Penta	2,3,3',4',5-Pentachlorobiphenyl
BZ#110	Penta	2,3,3',4,6-Pentachlorobiphenyl
BZ#118	Penta	2,3',4,4',5-Pentachlorobiphenyl
BZ#128	Hexa	2,2',3,3',4,4'-Hexachlorobiphenyl
BZ#135	Hexa	2,2',3,3',5,6'-Hexachlorobiphenyl

Table K-1
Congeners Selected by NOAA for Statistical Analysis of 1993 and 1995 Fish Samples
(NOAA,1997)

Congener Number	Homologue Group	Congener
BZ#138	Hexa	2,2',3,4,4',5'-Hexachlorobiphenyl
BZ#149	Hexa	2,2',3,4',5',6-Hexachlorobiphenyl
BZ#151	Hexa	2,2',3,5,5',6-Hexachlorobiphenyl
BZ#153	Hexa	2,2',4,4',5,5'-Hexachlorobiphenyl
BZ#170	Hepta	2,2',3,3',4,4',5-Heptachlorobiphenyl
BZ#177	Hepta	2,2',3,3',4',5,6-Heptachlorobiphenyl
BZ#180	Hepta	2,2',3,4,4',5,5'-Heptachlorobiphenyl
BZ#183	Hepta	2,2',3,4,4',5',6-Heptachlorobiphenyl
BZ#187	Hepta	2,2',3,4',5,5',6-Heptachlorobiphenyl
BZ#194	Octa	2,2',3,3',4,4',5,5'-Octachlorobiphenyl
BZ#201	Octa	2,2',3,3',4',5,5',6-Octachlorobiphenyl
BZ#206	Nona	2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl

Table K-2
Congeners Selected for Principal Component Analysis Based on Optimization of 1993 Sediment, Water, Benthic Invertebrates and Fish Samples

Congener Number	Homologue Group	Congener	Used by NOAA, 1997
BZ#4	Di	2,2'-Dichlorobiphenyl	
BZ#10	Di	2,6-DiChlorobiphenyl	
BZ#19	Tri	2,2',6-Trichlorobiphenyl	yes
BZ#22	Tri	2,3,4'-Trichlorobiphenyl	yes
BZ#27	Tri	2,3',6-Trichlorobiphenyl	yes
BZ#28	Tri	2,4,4'-Trichlorobiphenyl	yes
BZ#31	Tri	2,4',5-Trichlorobiphenyl	yes
BZ#37	Tri	3,4,4'-Trichlorobiphenyl	
BZ#44	Tetra	2,2',3,5'-Tetrachlorobiphenyl	yes
BZ#47	Tetra	2,2',4,4'-Tetrachlorobiphenyl	yes
BZ#49	Tetra	2,2',4,5'-Tetrachlorobiphenyl	yes
BZ#52	Tetra	2,2',5,5'Tetrachlorobiphenyl	yes
BZ#66	Tetra	2,3',4,4'-'Tetrachlorobiphenyl	yes
BZ#70	Tetra	2,3',4',5-Tetrachlorobiphenyl	yes
BZ#84	Penta	2,2',3,3',6-Pentachlorobiphenyl	yes
BZ#87	Penta	2,2',3,4,5-Pentachlorobiphenyl	yes
BZ#91	Penta	2,2',3,4',6-Pentachlorobiphenyl	yes
BZ#101 with BZ#90	Penta	2,2',4,5,5'-Pentachlorobiphenyl	yes
BZ#105	Penta	2,3,3',4,4'-Pentachlorobiphenyl	yes
BZ#135	Hexa	2,2',3,3',5,6'-Hexachlorobiphenyl	yes
BZ#136	Hexa	2,2',3,3',6,6'-Hexachlorobiphenyl	
BZ#138	Hexa	2,2',3,4,4',5-Hexachlorobiphenyl	yes
BZ#149	Hexa	2,2',3,4',5',6-Hexachlorobiphenyl	yes
BZ#151	Hexa	2,2',3,5,5',6-Hexachlorobiphenyl	yes
BZ#153	Hexa	2,2',4,4',5,5'-Hexachlorobiphenyl	yes
BZ#156	Hexa	2,3,3',4,4',5-Hexachlorobiphenyl	
BZ#170	Hepta	2,2',3,3',4,4',5-Heptachlorobipher	yes
BZ#180	Hepta	2,2',3,4,4',5,5'-Heptachlorobipher	yes
BZ#187	Hepta	2,2',3,4',5,5',6-Heptachlorobipher	yes

Table K-3
Feeding Guild Classification for Hudson River Fish Species

Region	Forager	Semi-Piscivore	Piscivore	Omnivore
Fresh Water Only (RM 196 to 60)	Red Breasted Sunfis Cyprinid Species Tesselated Darter Longnose Dace Sucker Species Pumpkinseed Spot Tail Shiner Brook Silverside	Yellow Perch	Largemouth Bass Rock Bass Smallmouth Bass	Brown Bullhead White Catfish
Fresh to Saline (RM 154 to 26)	Atlantic Silverside	White Perch	Striped Bass	

Table K-4
Summary Statistics for PCB Concentration in Co-located Sediments and Benthic Invertebrates in the TI Pool and Lower Hudson

Location	Sediment Total PCB Concentration (ug/kg)				Benthic Invertebrates Total PCB Concentration (ug/kg)				Ratio of Benthic Invertebrates / Sediment[2]	
	Arithimetic Mean	Range[1]	Geometric Mean	Range[1]	Arithimetic Mean	Range[1]	Geometric Mean	Range[1]	Arithimetic Mean	Geometric Mean
TI Pool (RM 188.5 - 191.5)	16,400	12,960 - 19,800	14,700	12,200 - 17,700	17,700	11,800 - 23,700	9,540	7,130 - 12,800	1.08	0.65
Lower Hudson (RM 25.8 - 122.4)	850	670 - 1030	700	510 - 940	410	260 - 560	290	200 - 430	0.48	0.41

Location	Sediment Total PCB Concentration Normalized to TOC (ug/kg - TOC)				Benthic Invertebrates Total PCB Concentration Normalized to Percent Lipids (ug/kg - lipid)				Ratio of Benthic Invertebrates / Sediment[2]	
	Arithimetic Mean	Range[1]	Geometric Mean	Range[1]	Arithimetic Mean	Range[1]	Geometric Mean	Range[1]	Arithimetic Mean	Geometric Mean
TI Pool (RM 188.5 - 191.5)	420,000	347,000 - 493,000	387,000	327,000 - 458,000	656,000	379,000 - 934,000	328,000	247,000 - 435,000	1.56	0.85
Lower Hudson (RM 25.8 - 122.4)	32,300	25,000 - 39,500	26,400	19,600 - 35,500	19,300	12,000 - 26,700	14,100	9,600 - 20,800	0.60	0.53

Note:

1. Range is the mean plus and minus two standard errors.
2. Ratio represents the simple quotient of the means given in the columns to the left.

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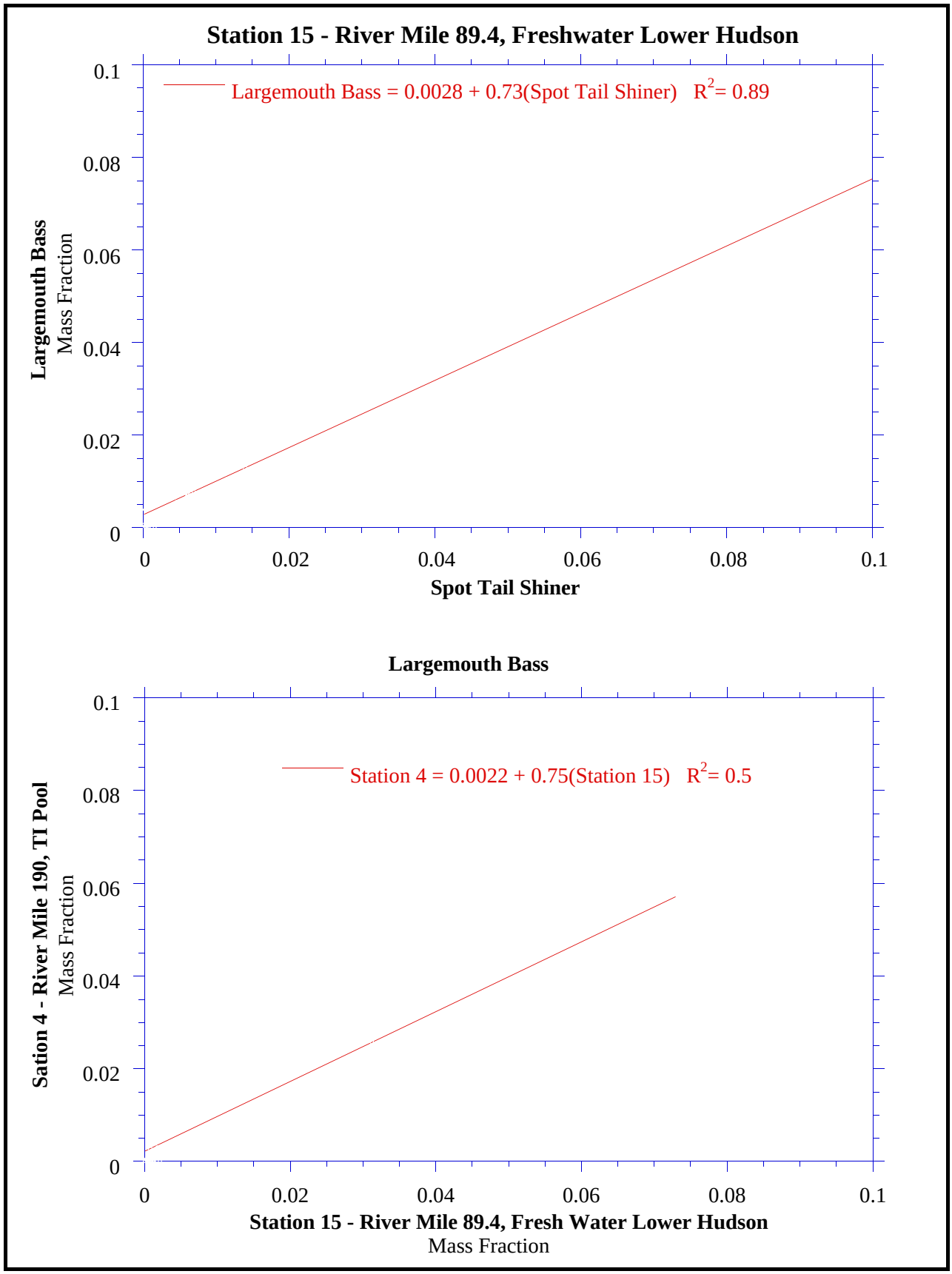

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Figure K-1
Comparisons of Congener Mass Fraction Between Species and Between Stations
1993 USEPA and NOAA Data

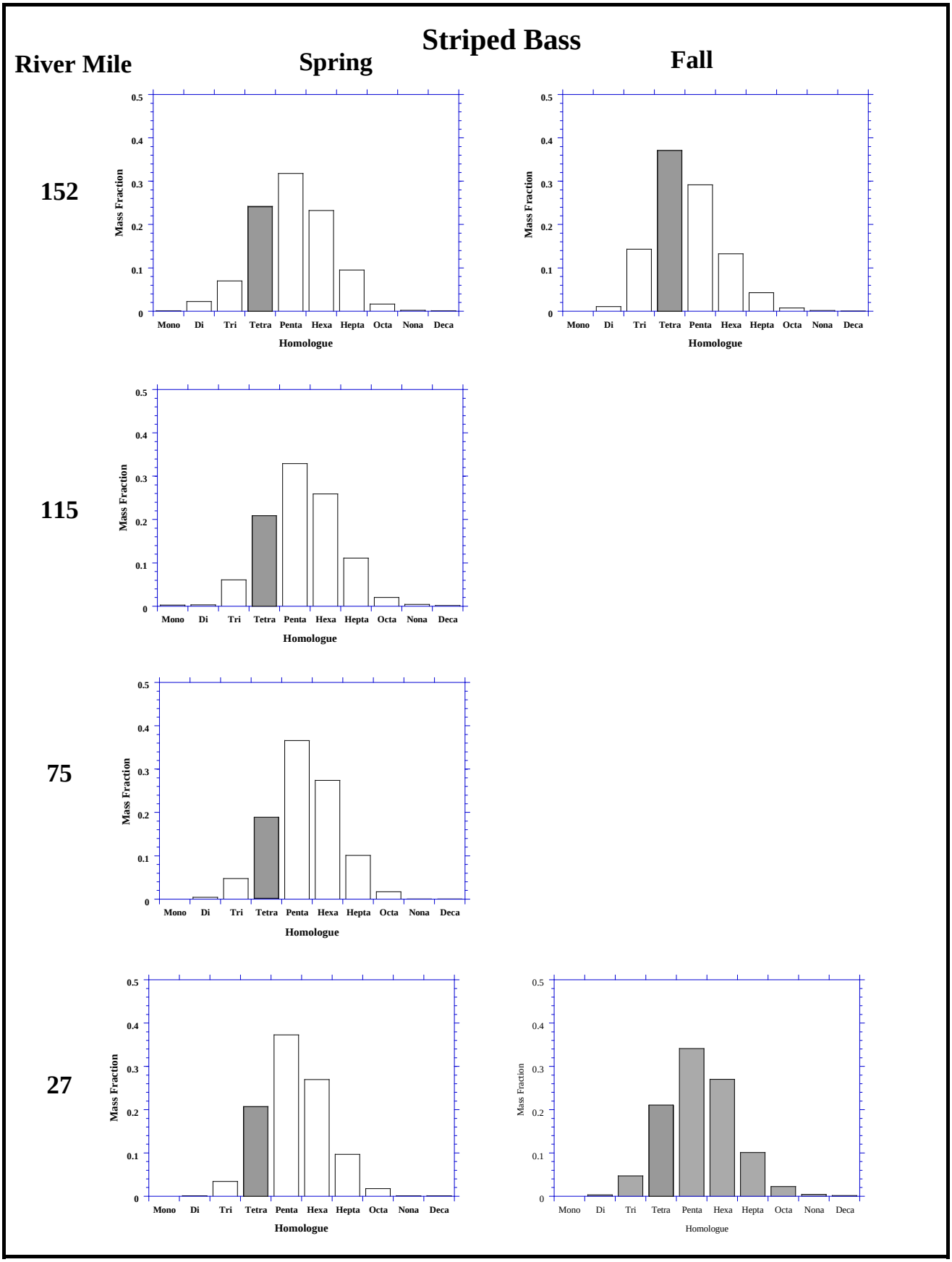

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Figure K-2
A Comparison of Homologue Patterns in Striped Bass
Male Adults as a Function of River Mile
1995 NOAA Data

White Perch

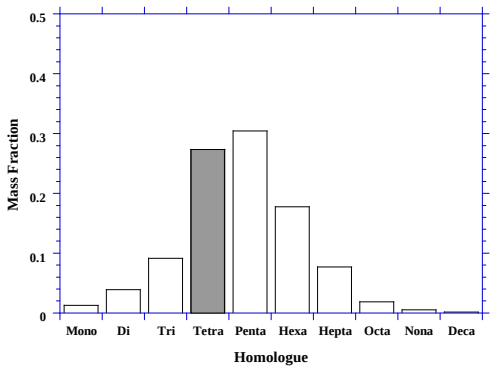
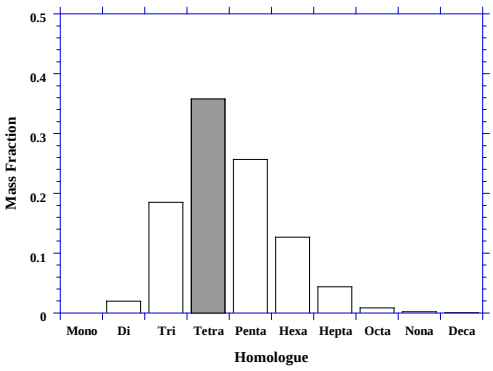
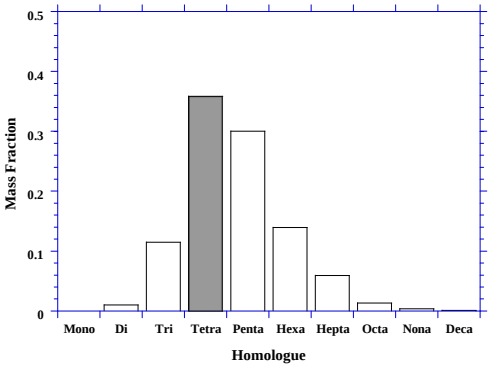
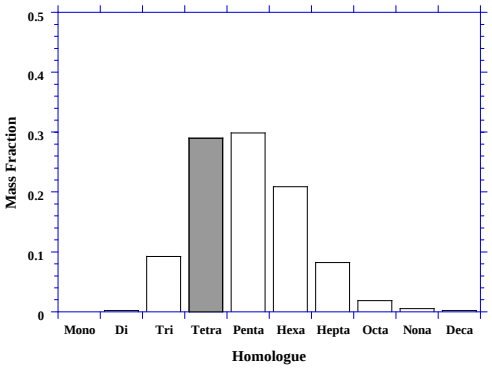
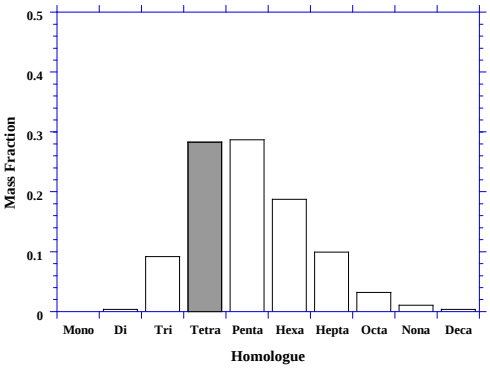
River Mile	Spring	Fall
152		
115		
59		
27		

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Figure K-3
A Comparison of Homologue Patterns in White Perch
Male Adults as a Function of River Mile
1995 NOAA Data

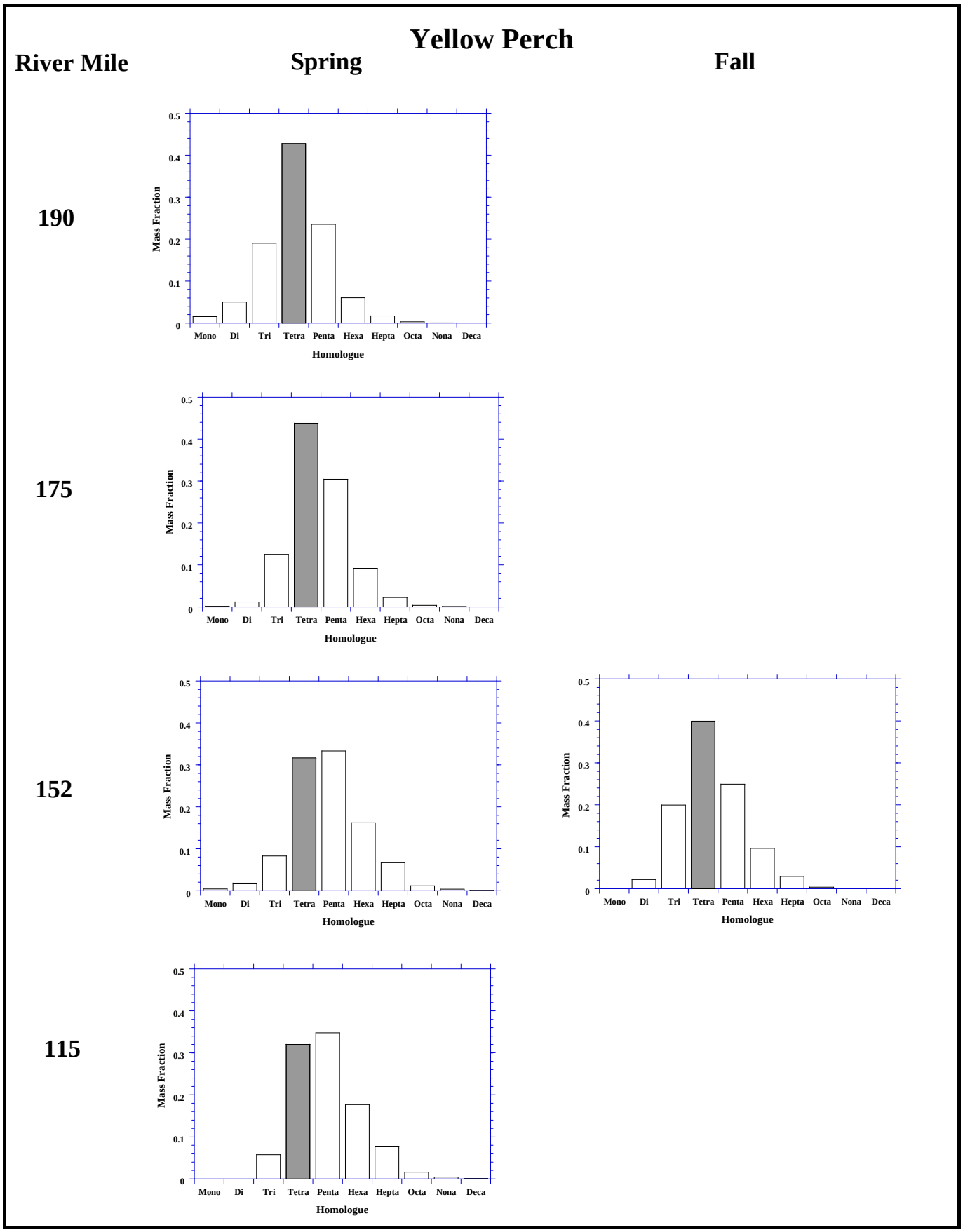

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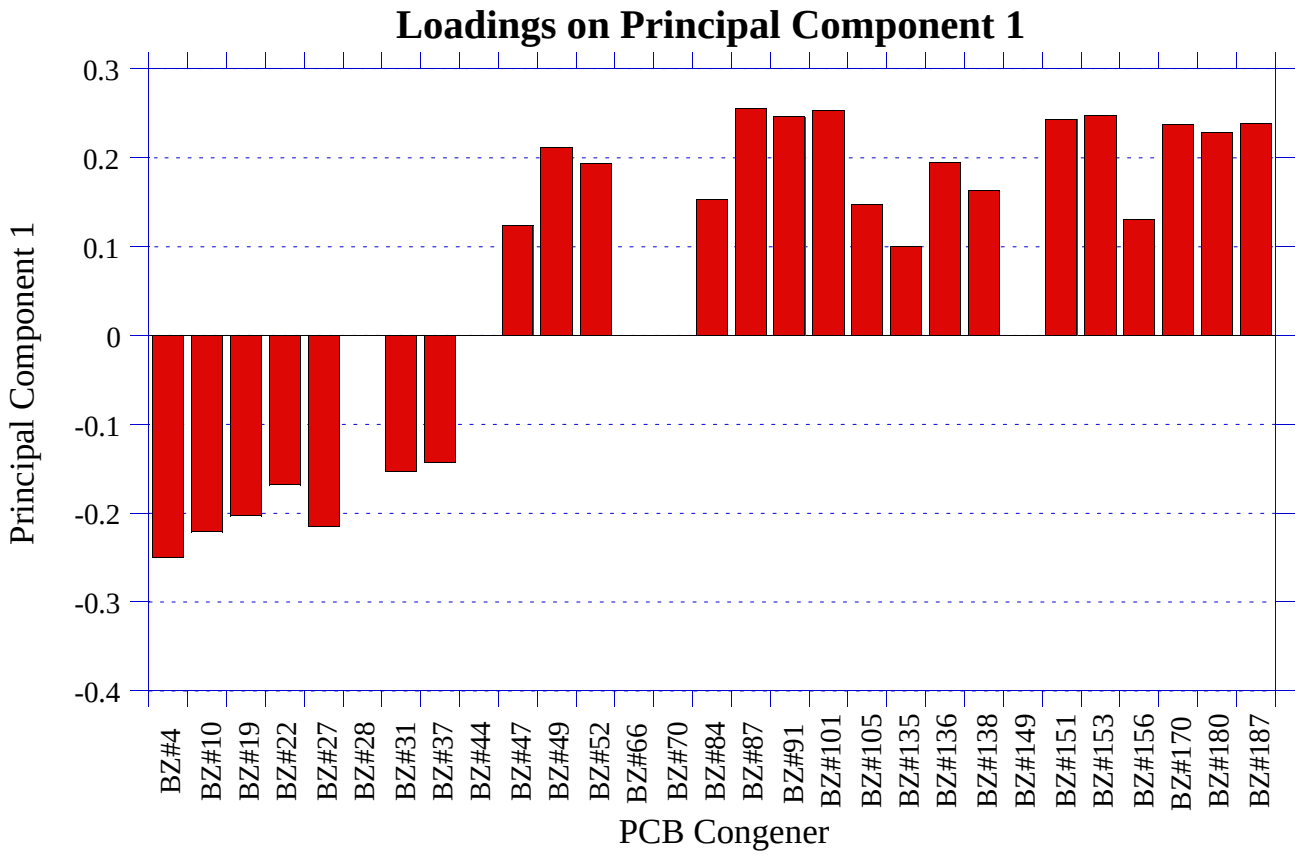
Figure K-4
A Comparison of Homologue Patterns in Yellow Perch Male Adults as a Function of River Mile 1995 NOAA Data



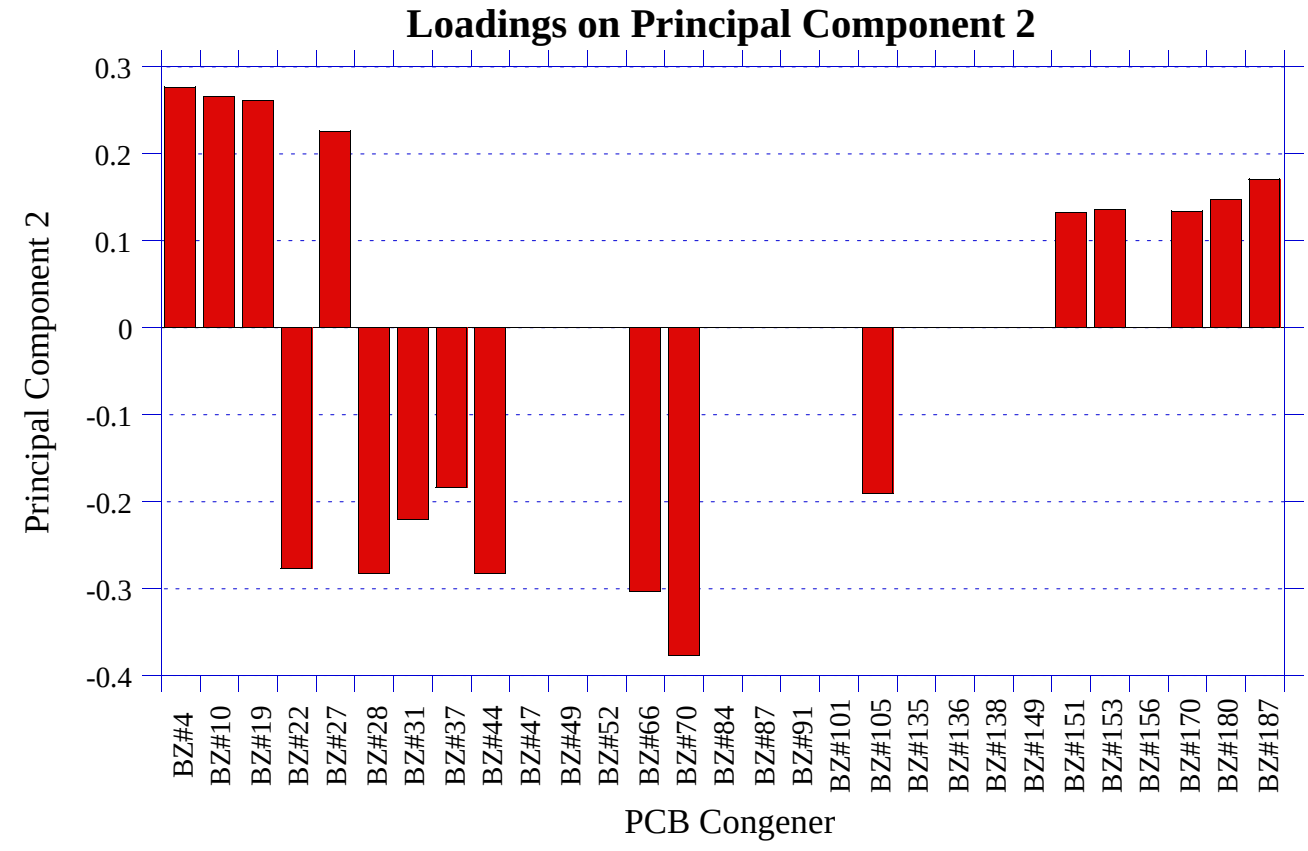


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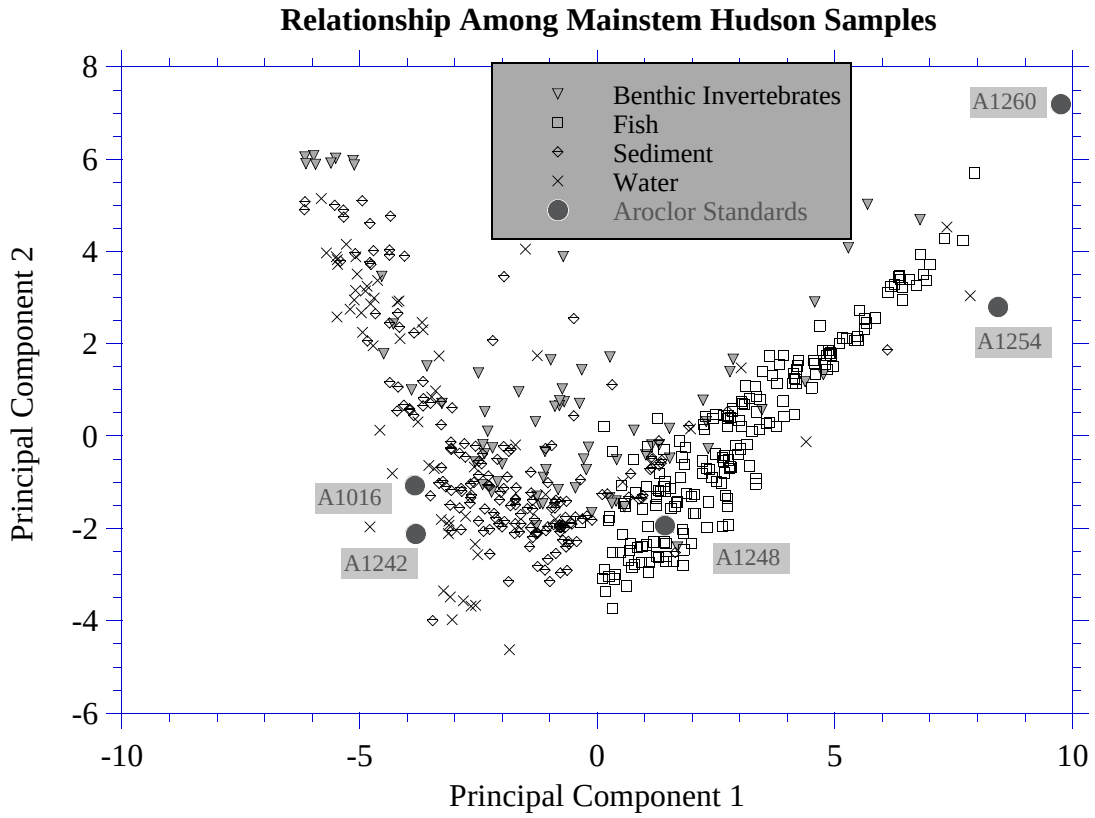
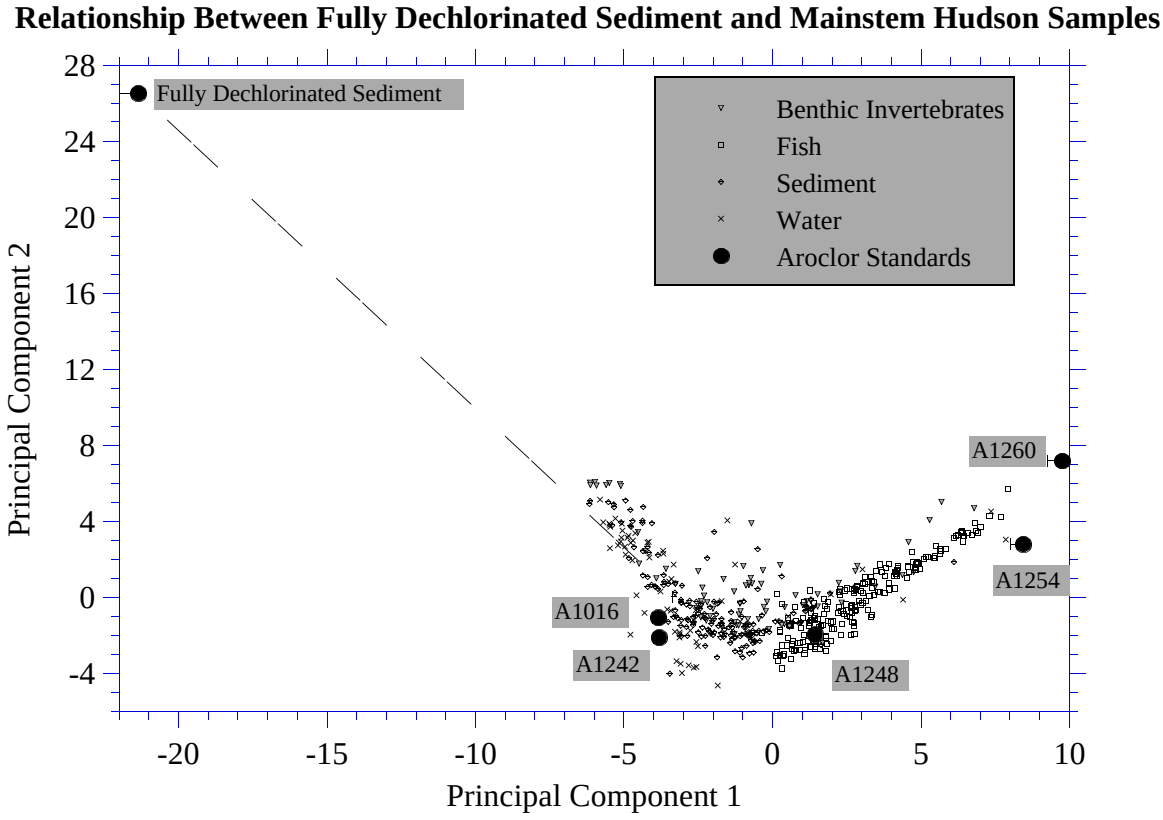
Figure K-5
Congener Loadings for Principal Components 1 and 2
1993 USEPA and NOAA Data for Sediments, Water, Fish and Benthic Invertebrates





Note: 1. Line drawn for illustration purposes only.

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Figure K-6
Principal Component Results for Phase 2 Sample Results:
All Mainstem Hudson Locations

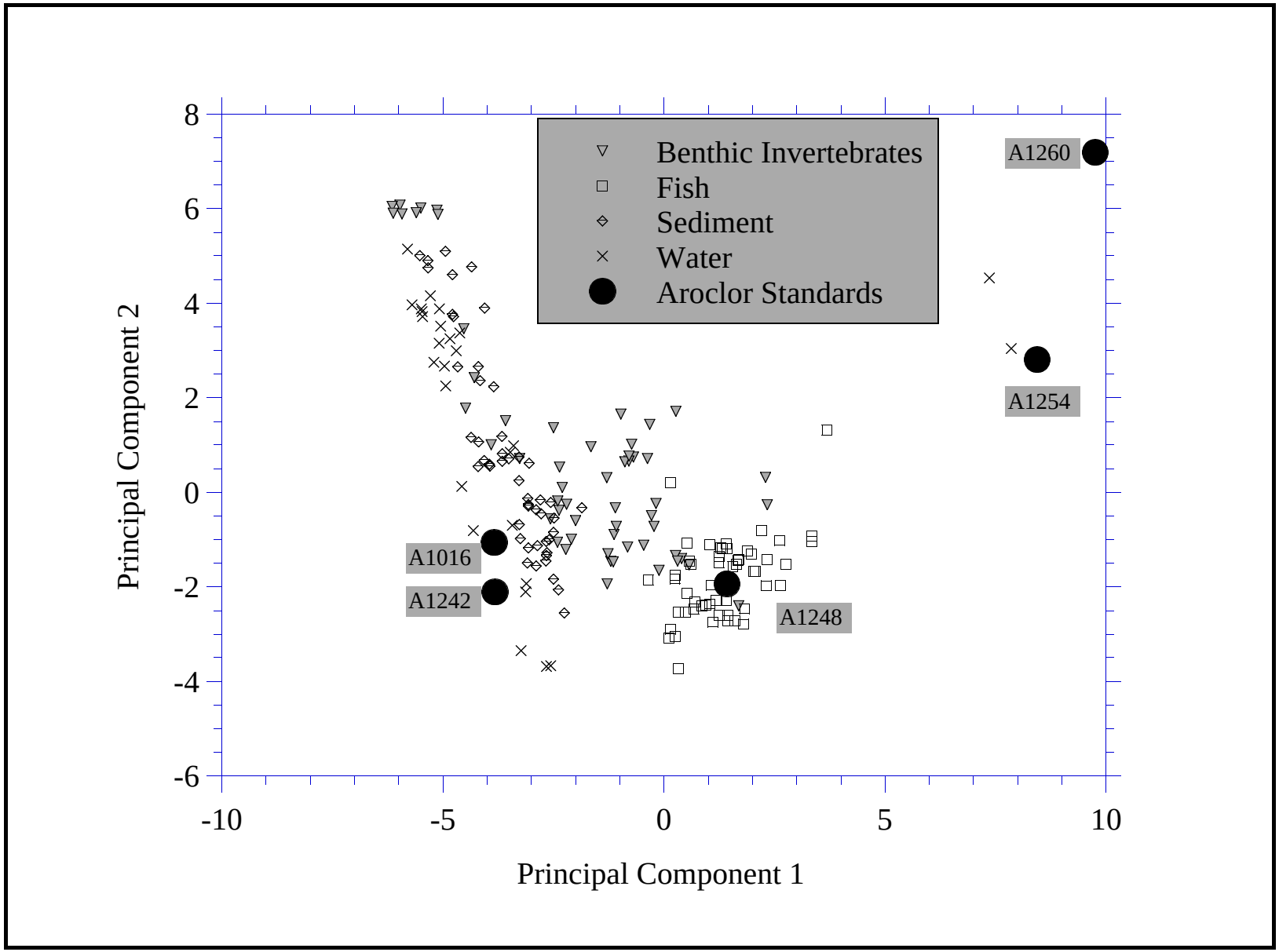

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Figure K-7
Principal Component Results for Hudson River Media
River Miles 195 to 175

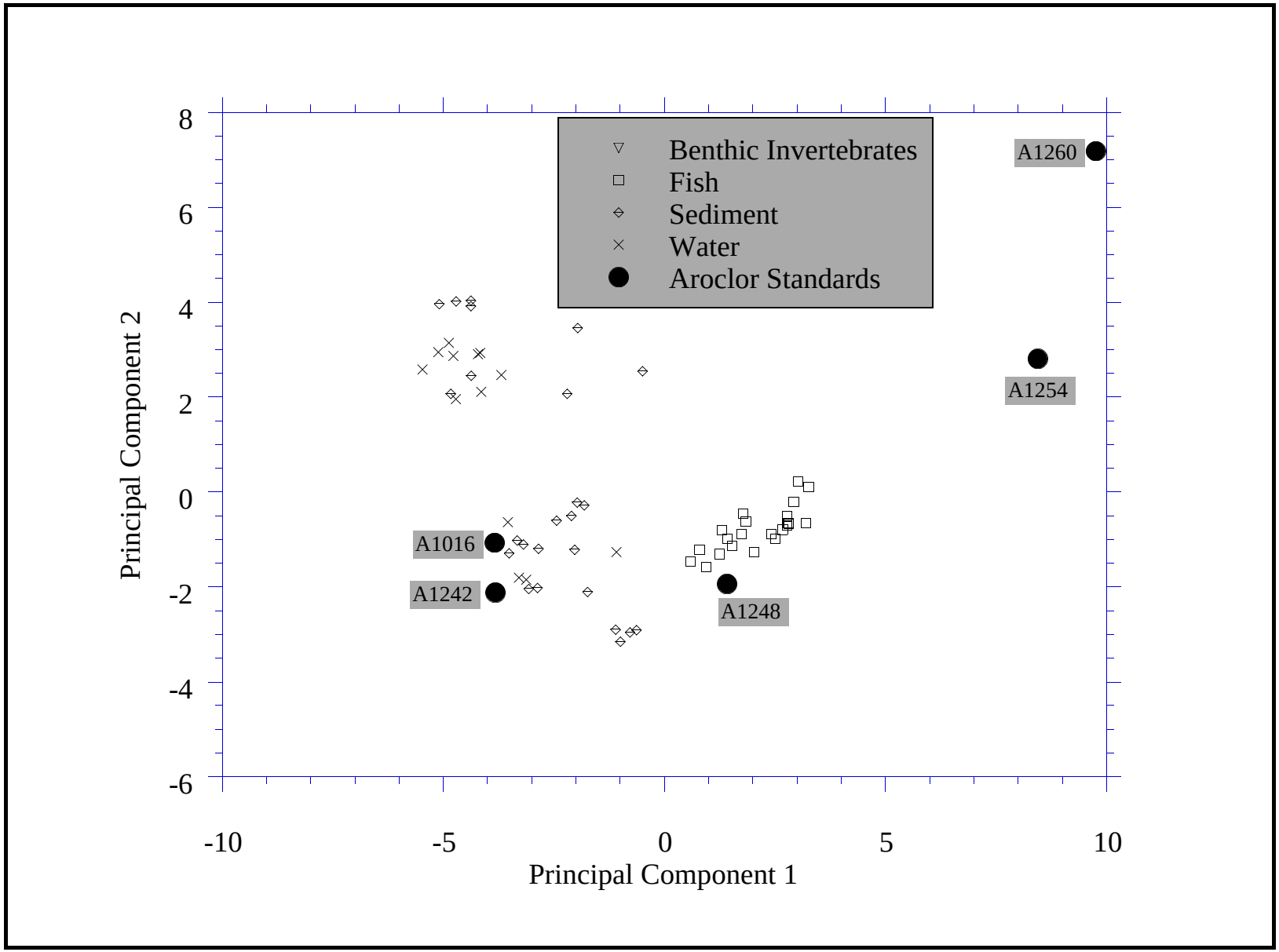

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Figure K-8
Principal Component Results for Hudson River Media
River Miles 175 to 156

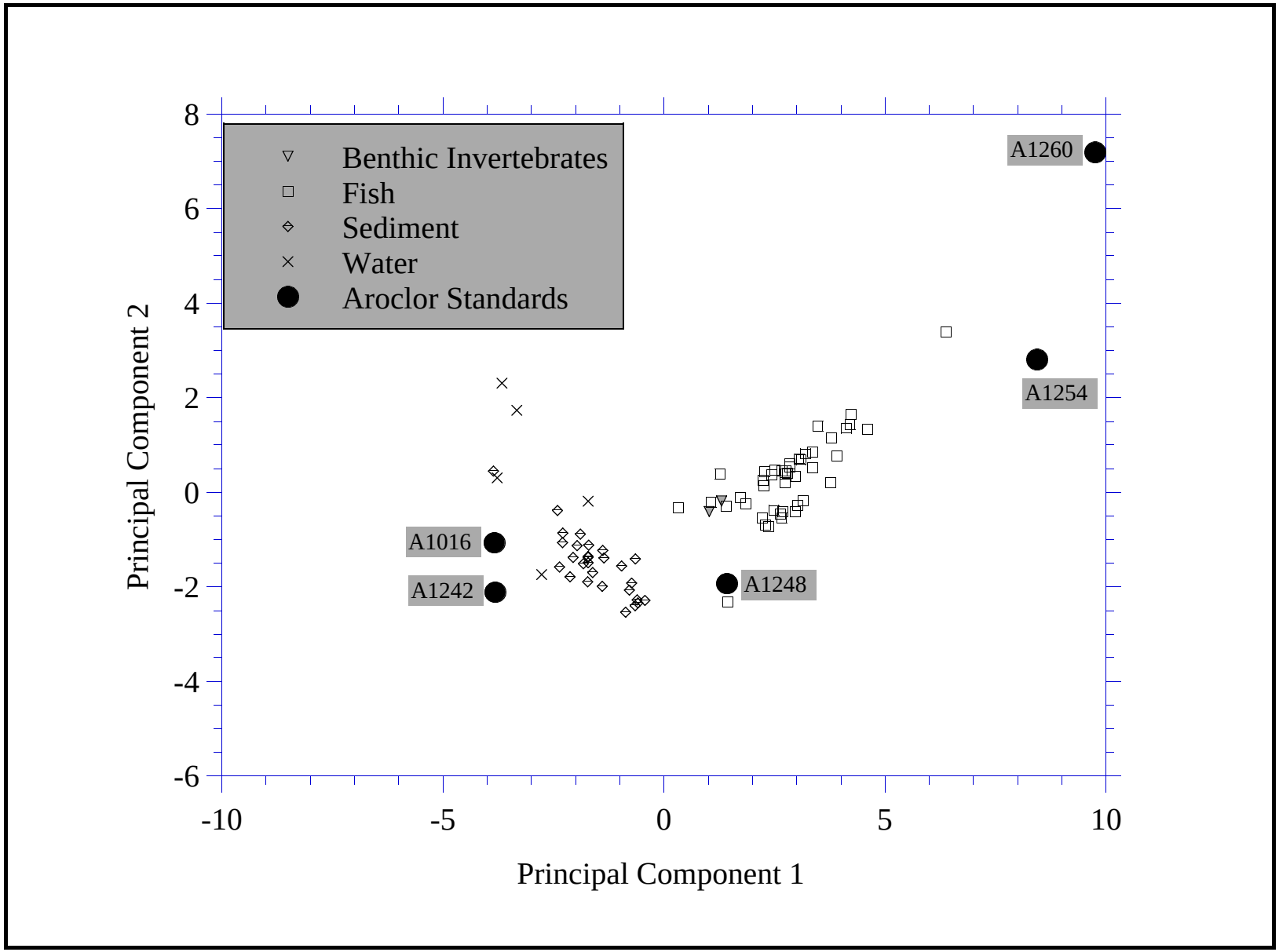

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Figure K-9
Principal Component Results for Hudson River Media
River Miles 156 to 100

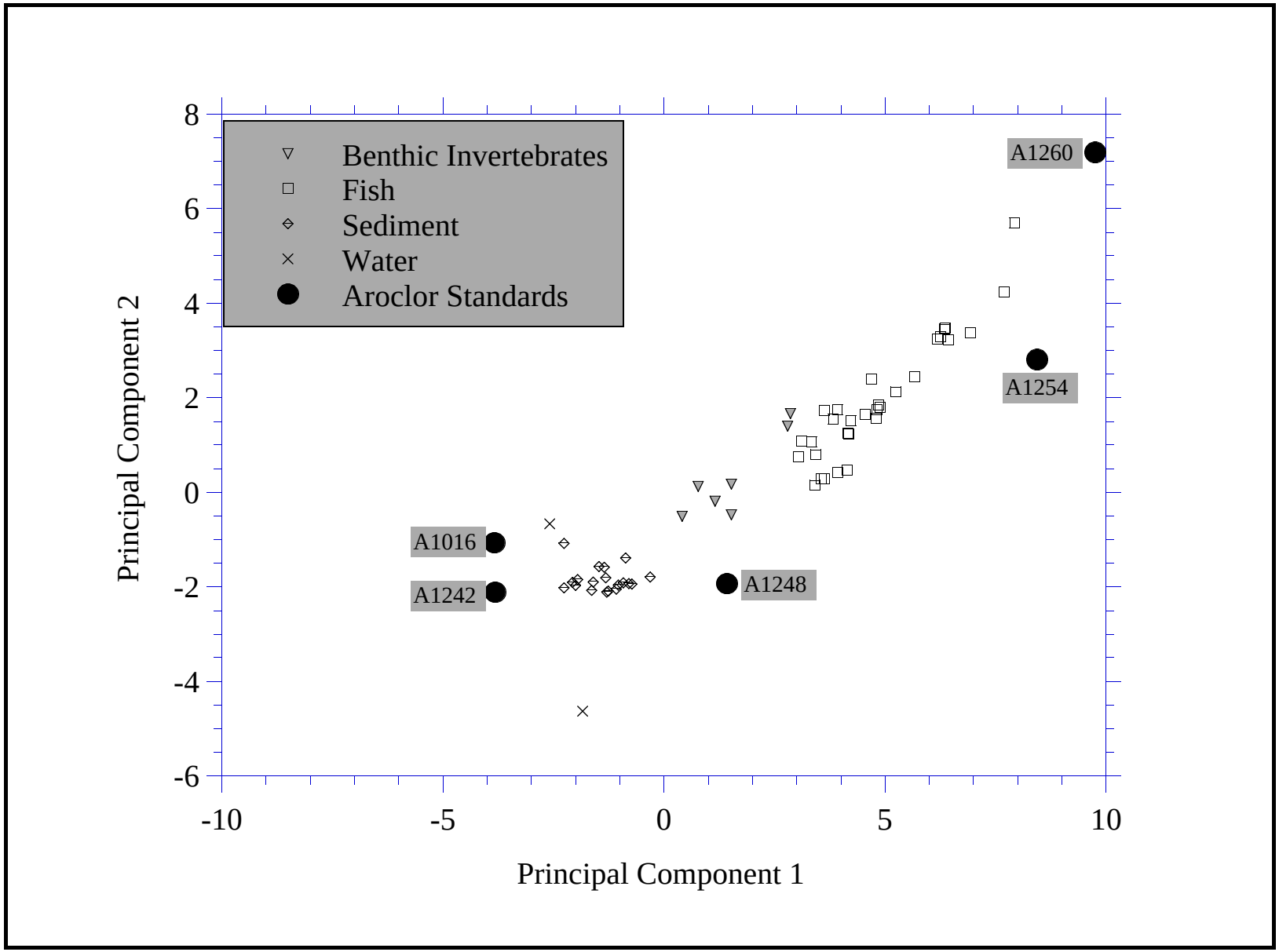

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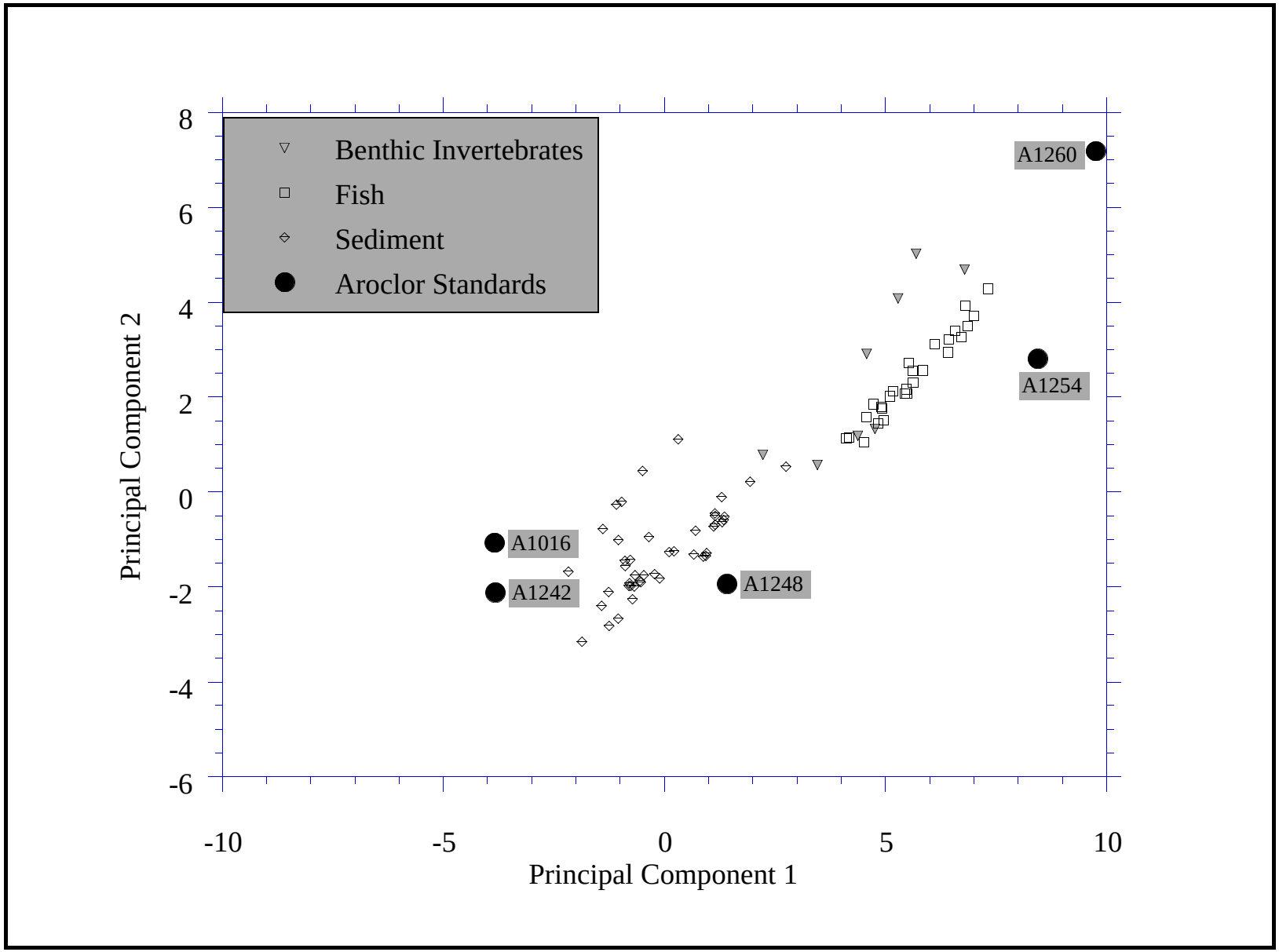
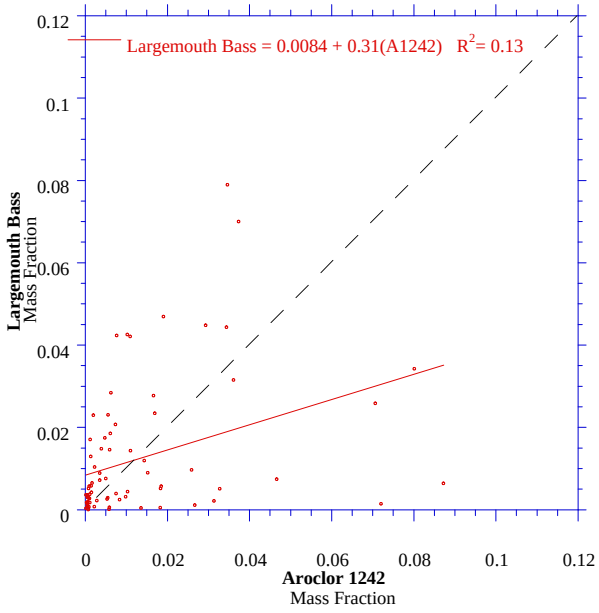
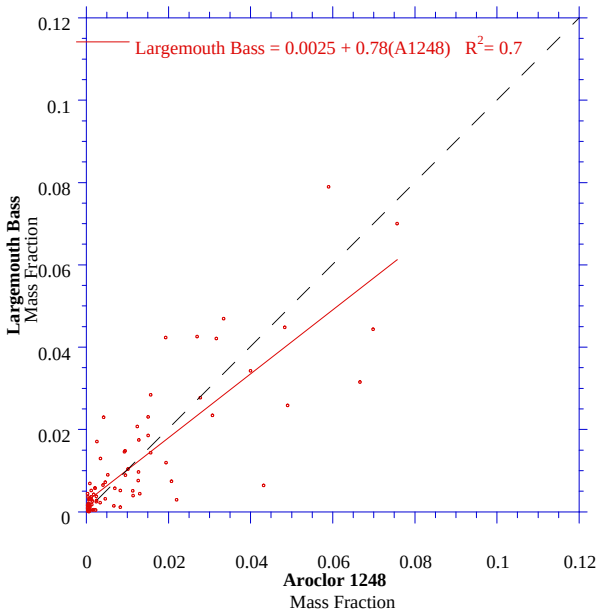
Figure K-10
Principal Component Results for Hudson River Media
River Miles 100 to 60

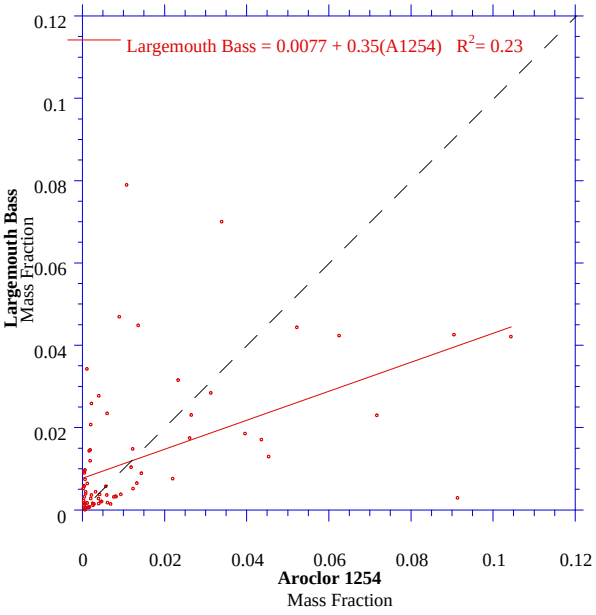


TAMS/MCA

Figure K-11
Principal Component Results for Hudson River Media
River Miles 60 to 0







Upper Hudson
Station 4 (River Mile 190)

Note: Dashed line represents line of perfect agreement.

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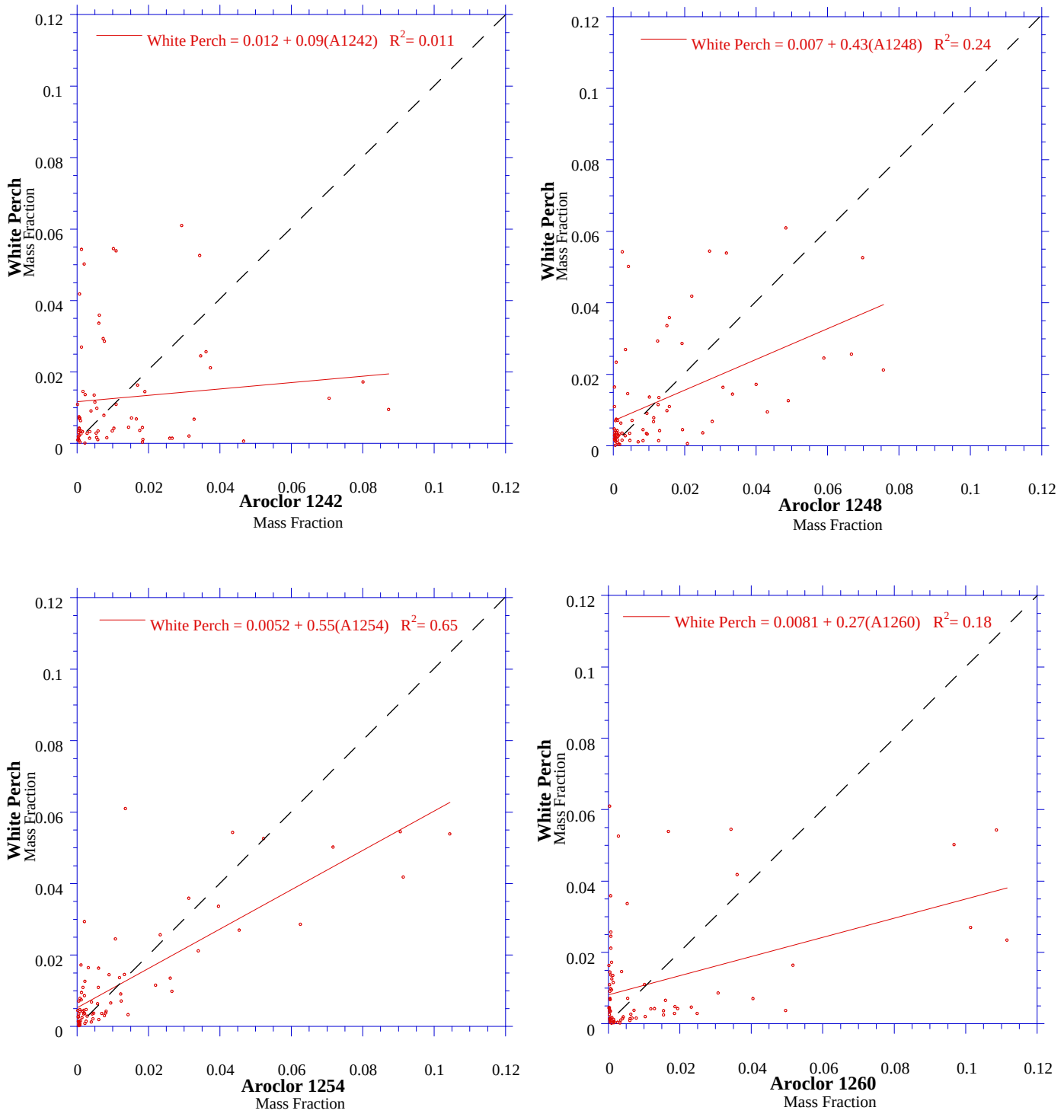
Figure K-12
Comparison of Congener Mass Fraction Between a Large Mouth Bass Sample from RM 190 and Several Aroclor Standards



Lower Hudson Saline
Station 18 (River Mile 26)

Note: Dashed line represents line of perfect agreement.

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Figure K-13
Comparisons of Congener Mass Fraction Between a White Perch Sample from RM 26 and Several Aroclor Standards

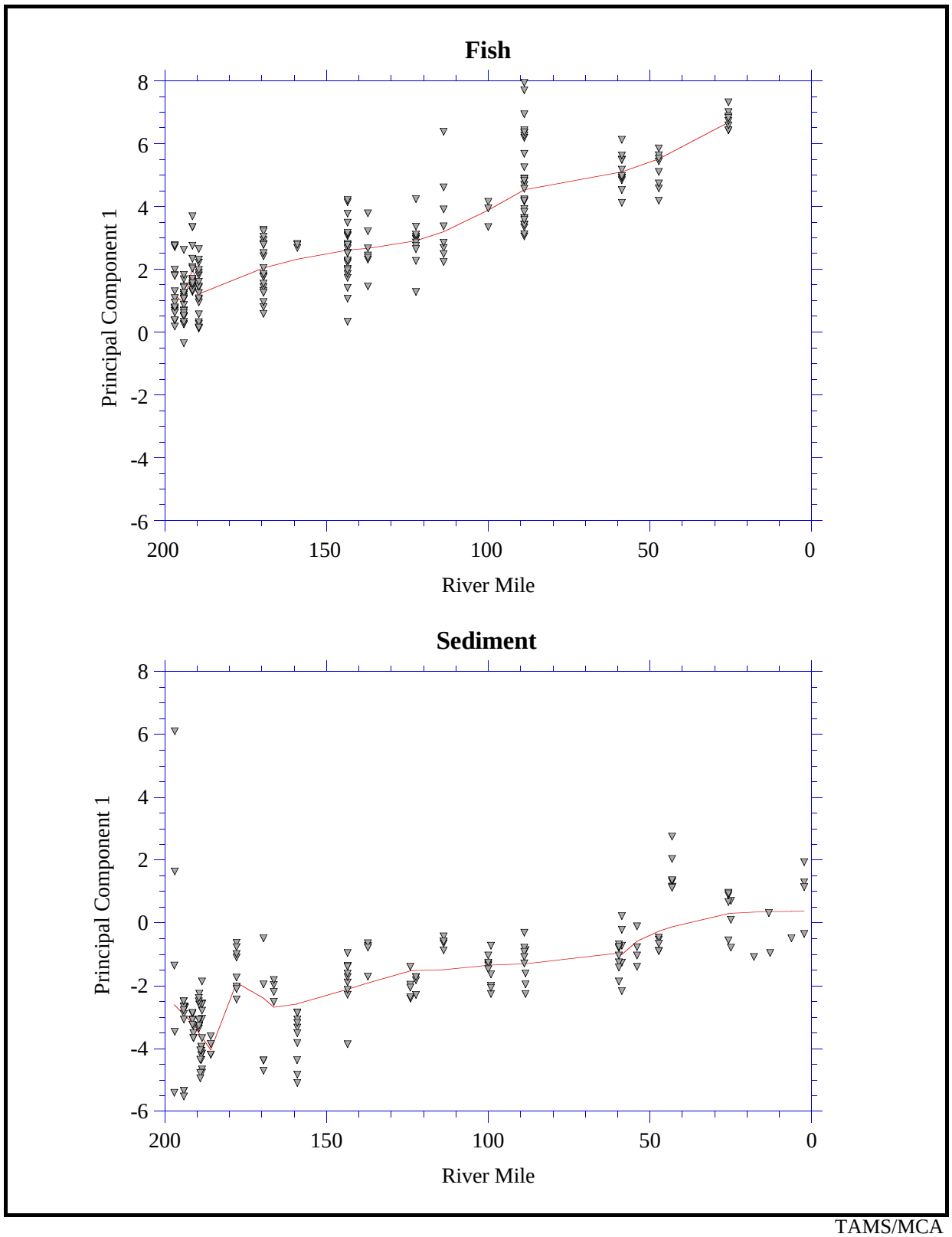

Figure K-14
Variation of Principal Component 1 with River Mile in Fish and Sediment

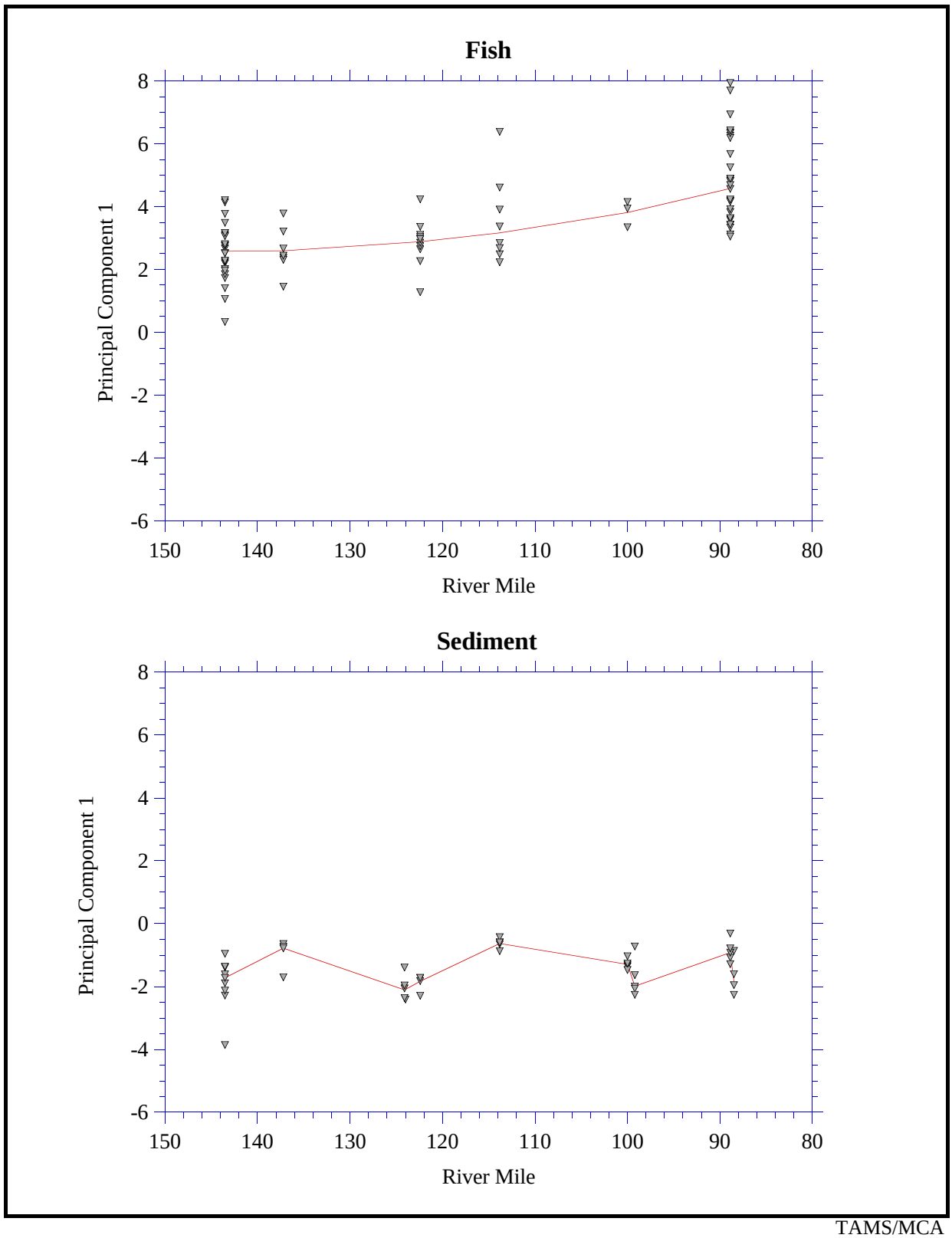

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Figure K-15
Variation of Principal Component 1 with River Mile in Fish and Sediment River Miles 150 to 80

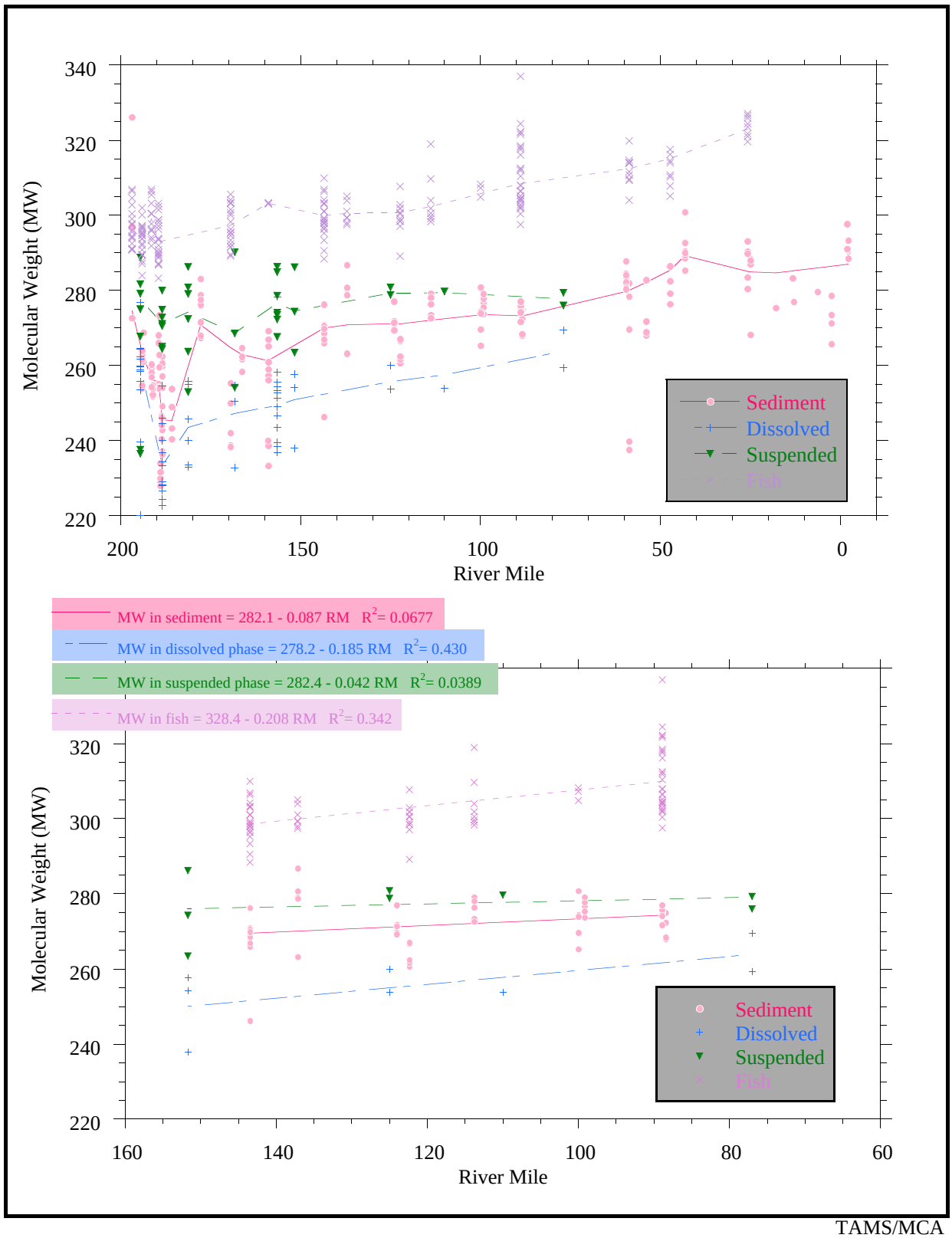

Figure K-16
Relationship Between Molecular Weight and River Mile for 1993 Hudson River Samples

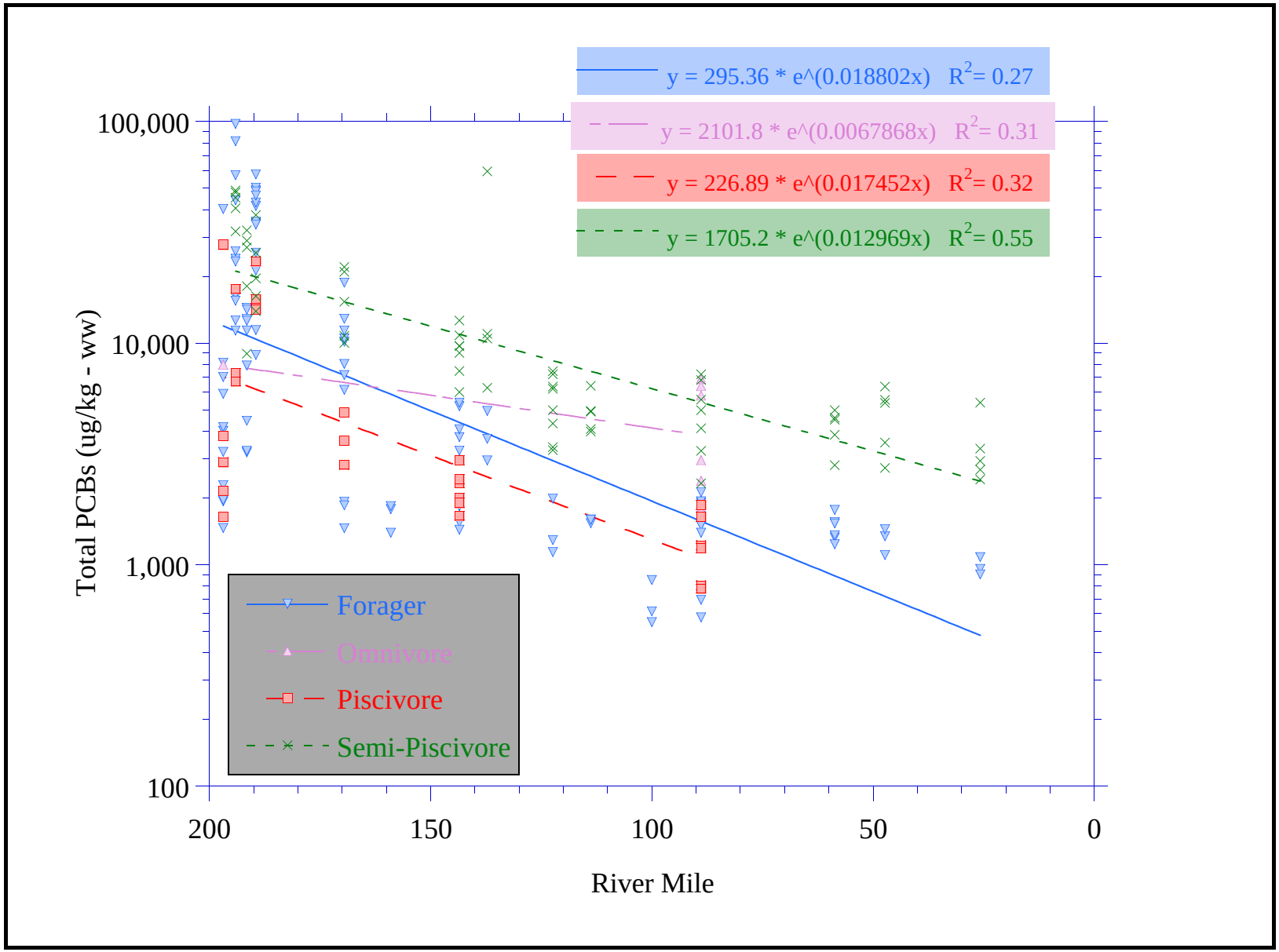

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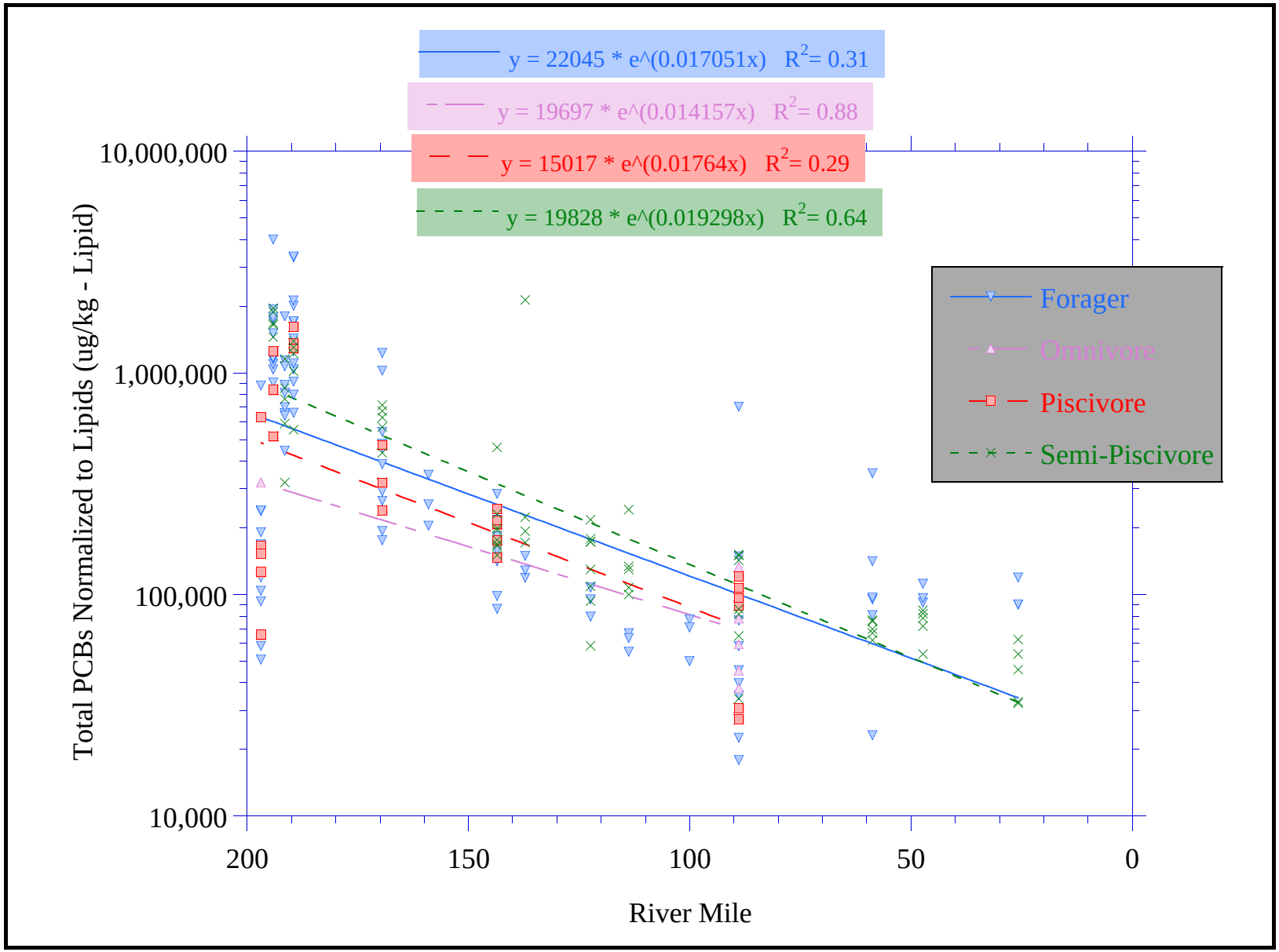
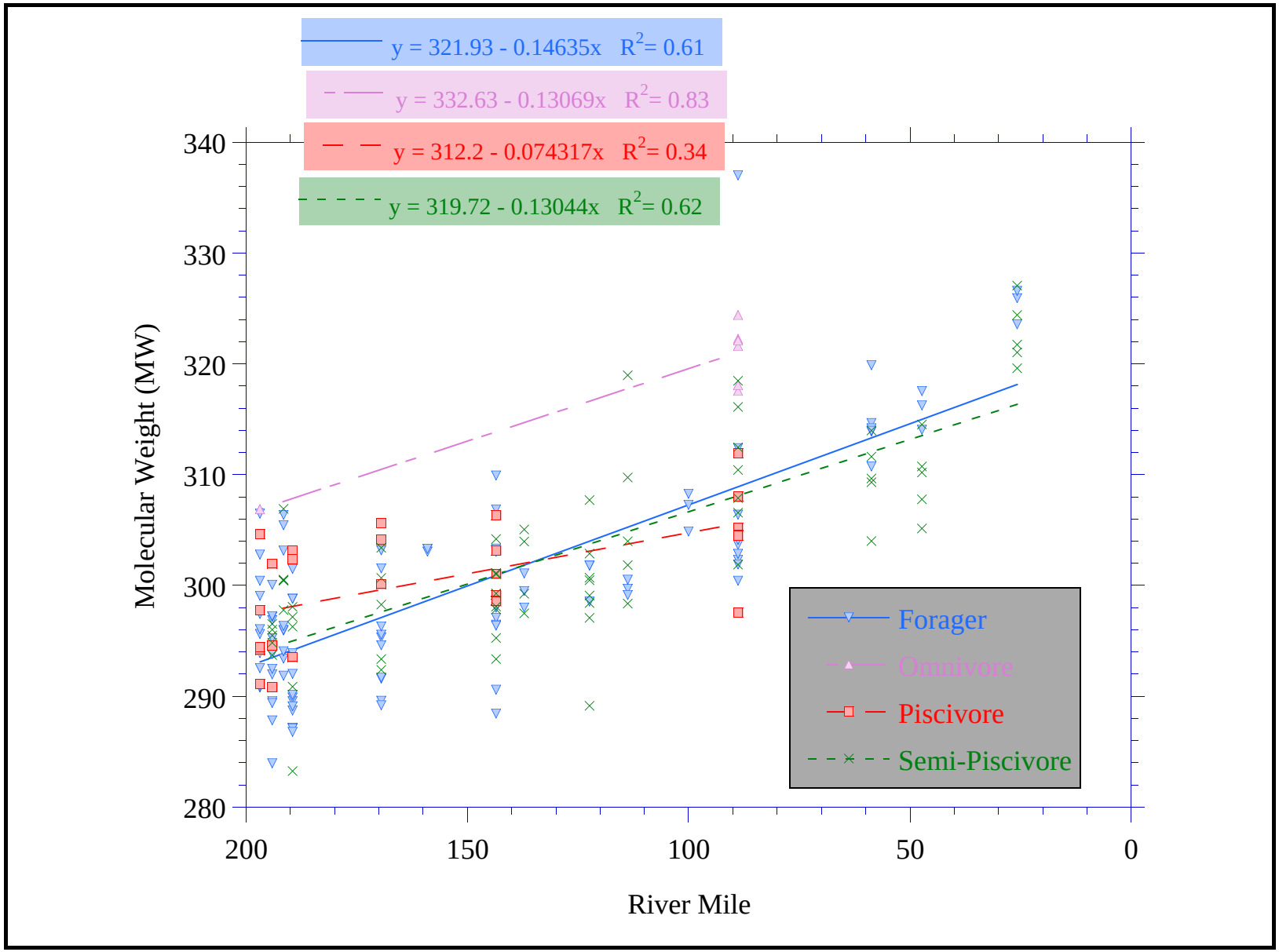
Figure K-17
Total PCBs Versus River Mile for 1993 Fish Data, Classified by Feeding Guild



TAMS/MCA

Figure K-18

Normalized Total PCBs Versus River Mile for 1993 Fish Data, Classified by Feeding Guild



TAMS/MCA

Figure K-19
Molecular Weight Versus River Mile for 1993 Fish Data, Classified by Feeding Guild

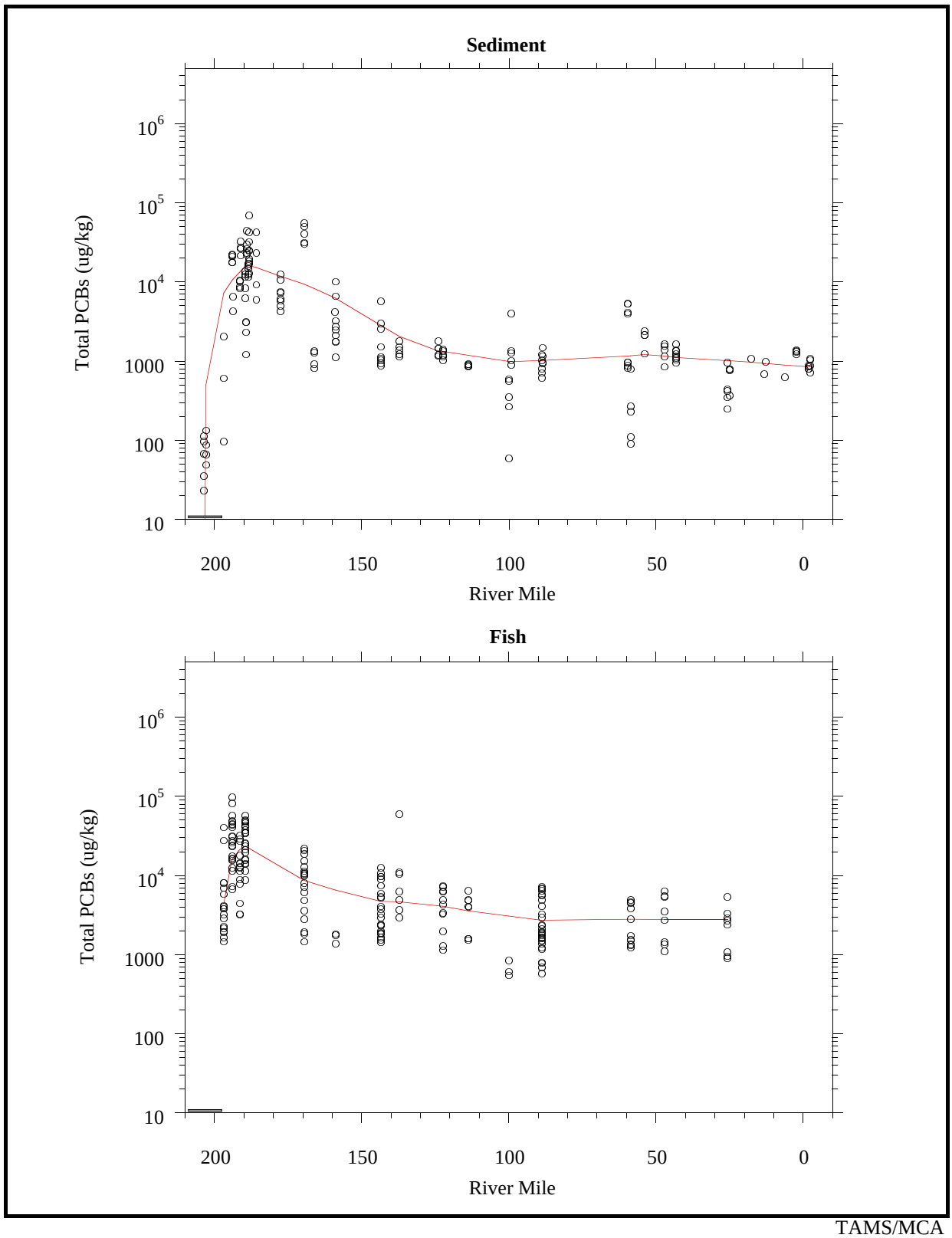
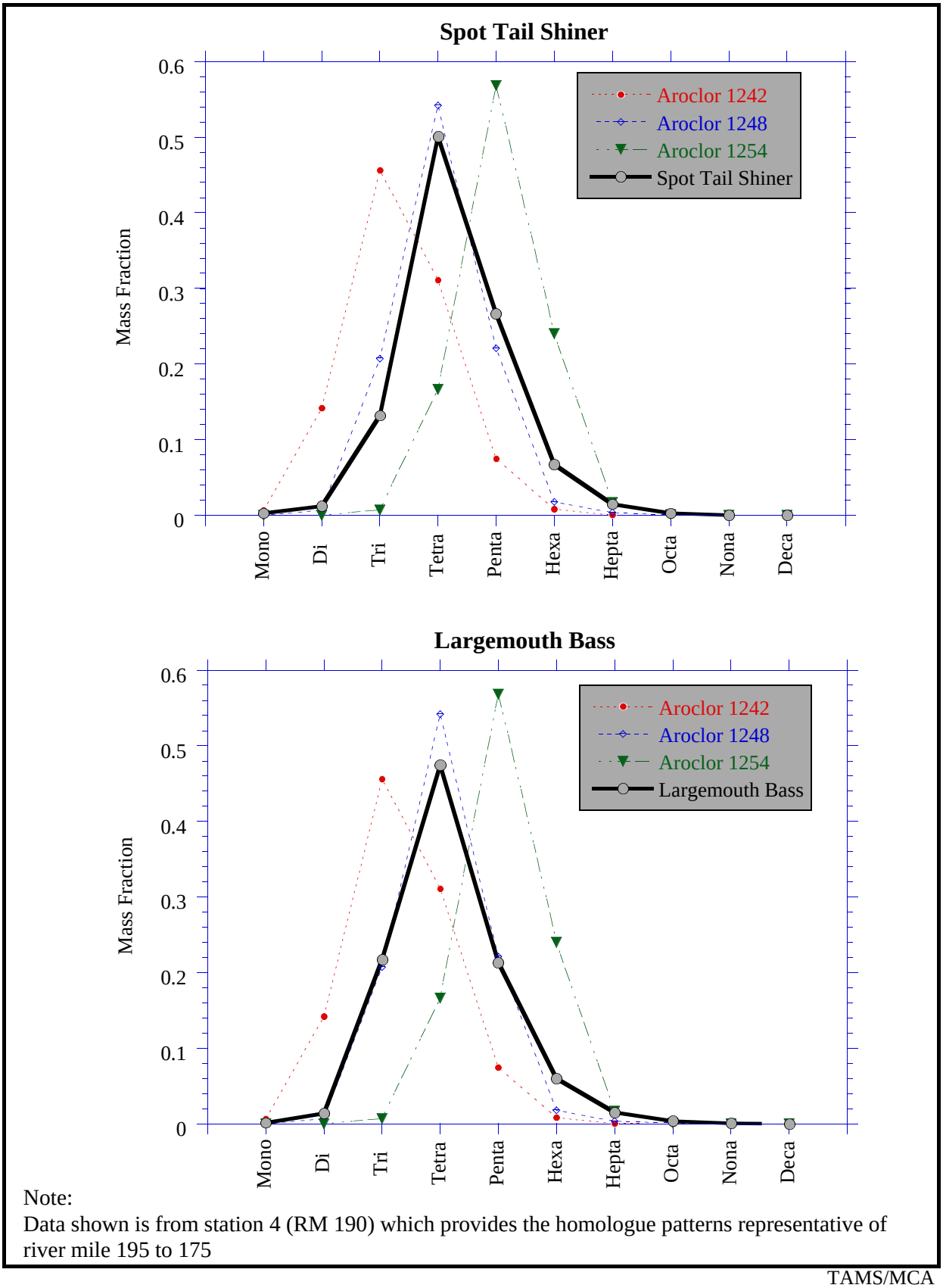


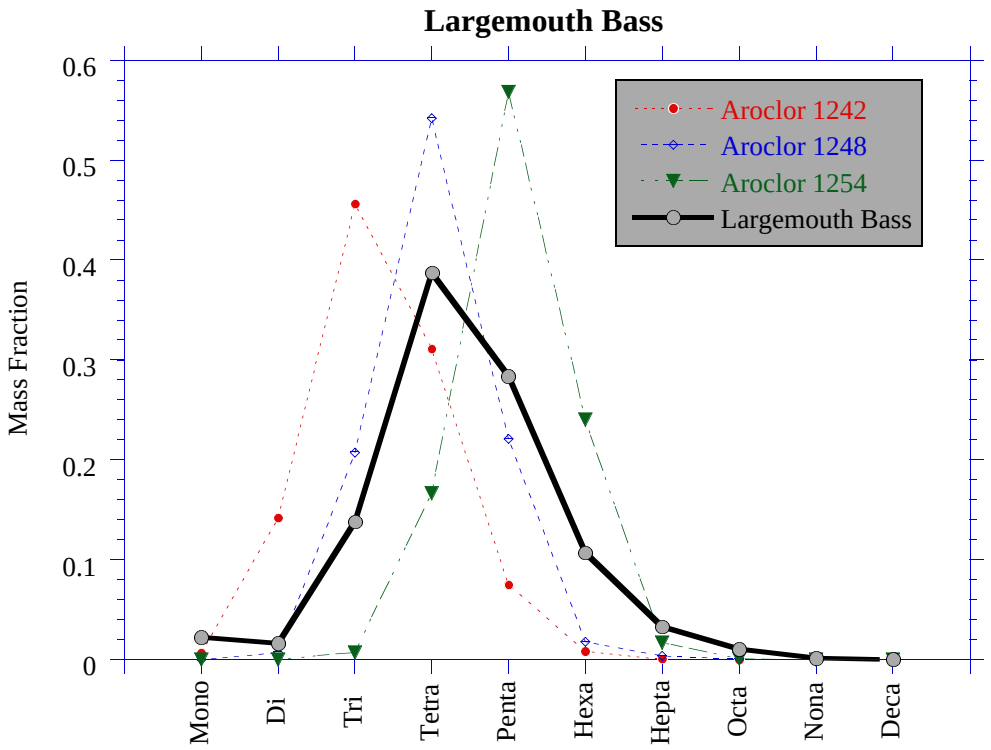
Figure K-20
Variation of Total PCB Concentration with River Mile in Shallow Fine-Grained Sediments and 1993 Fish Data

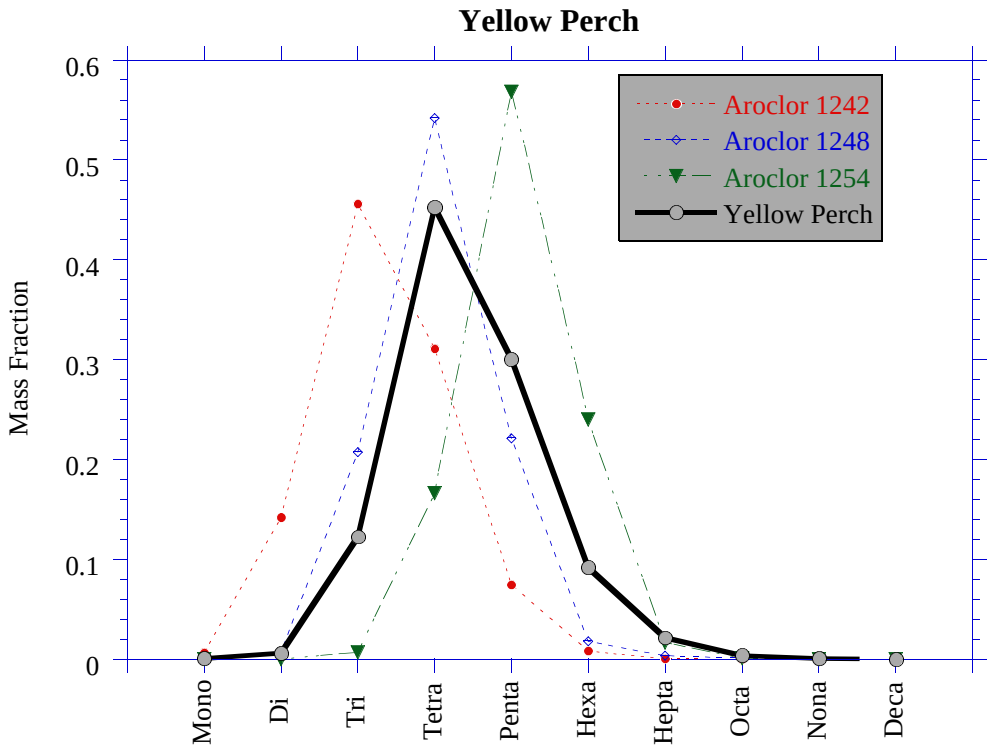


Note:
Data shown is from station 4 (RM 190) which provides the homologue patterns representative of river mile 195 to 175

TAMS/MCA

Figure K-21
A Comparison of Homologue Patterns in Aroclors and Hudson River Fish: River Mile 195 to 175
1993 USEPA and NOAA Data

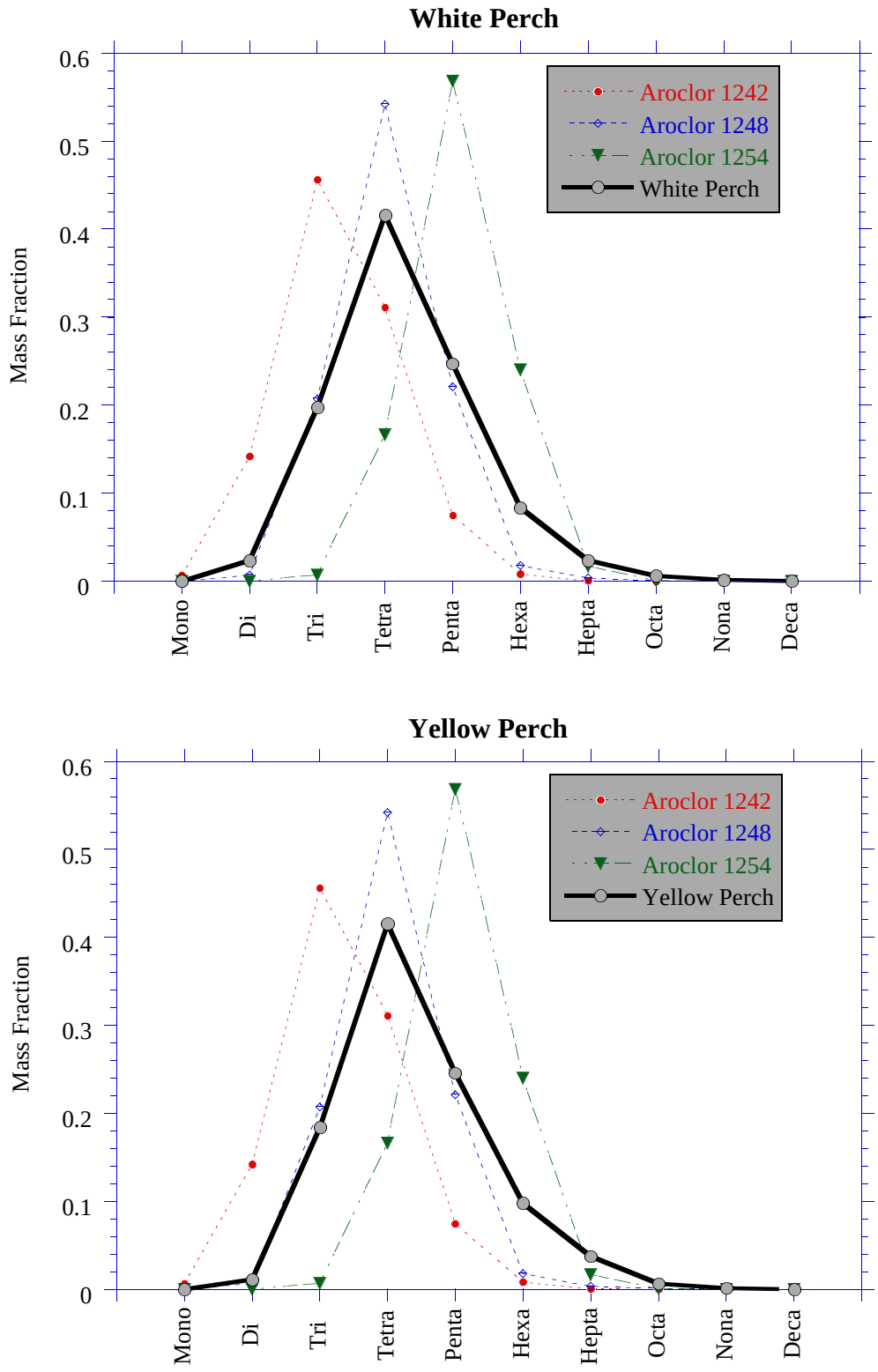




Note:
Data shown is from station 8 (RM 169.5) which provides the homologue patterns representative of river mile 175 to 156.

TAMS/MCA

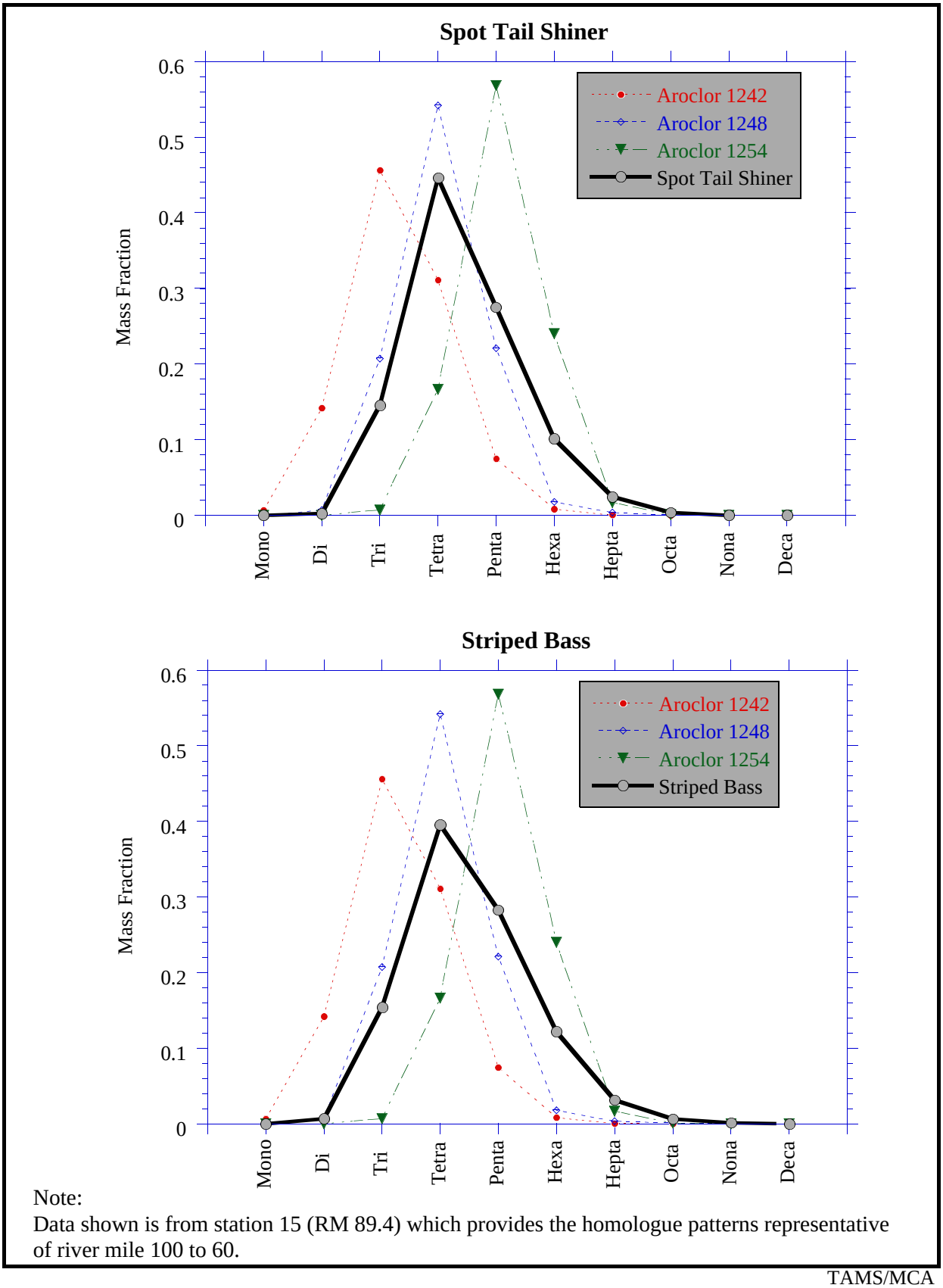
Figure K-22
A Comparison of Homologue Patterns in Aroclors and Hudson River Fish: River Mile 175 to 156
1993 USEPA and NOAA Data



Note:
Data shown is from station 12 (RM 122.7) which provides the homologue patterns representative of river mile 156 to 100.

TAMS/MCA

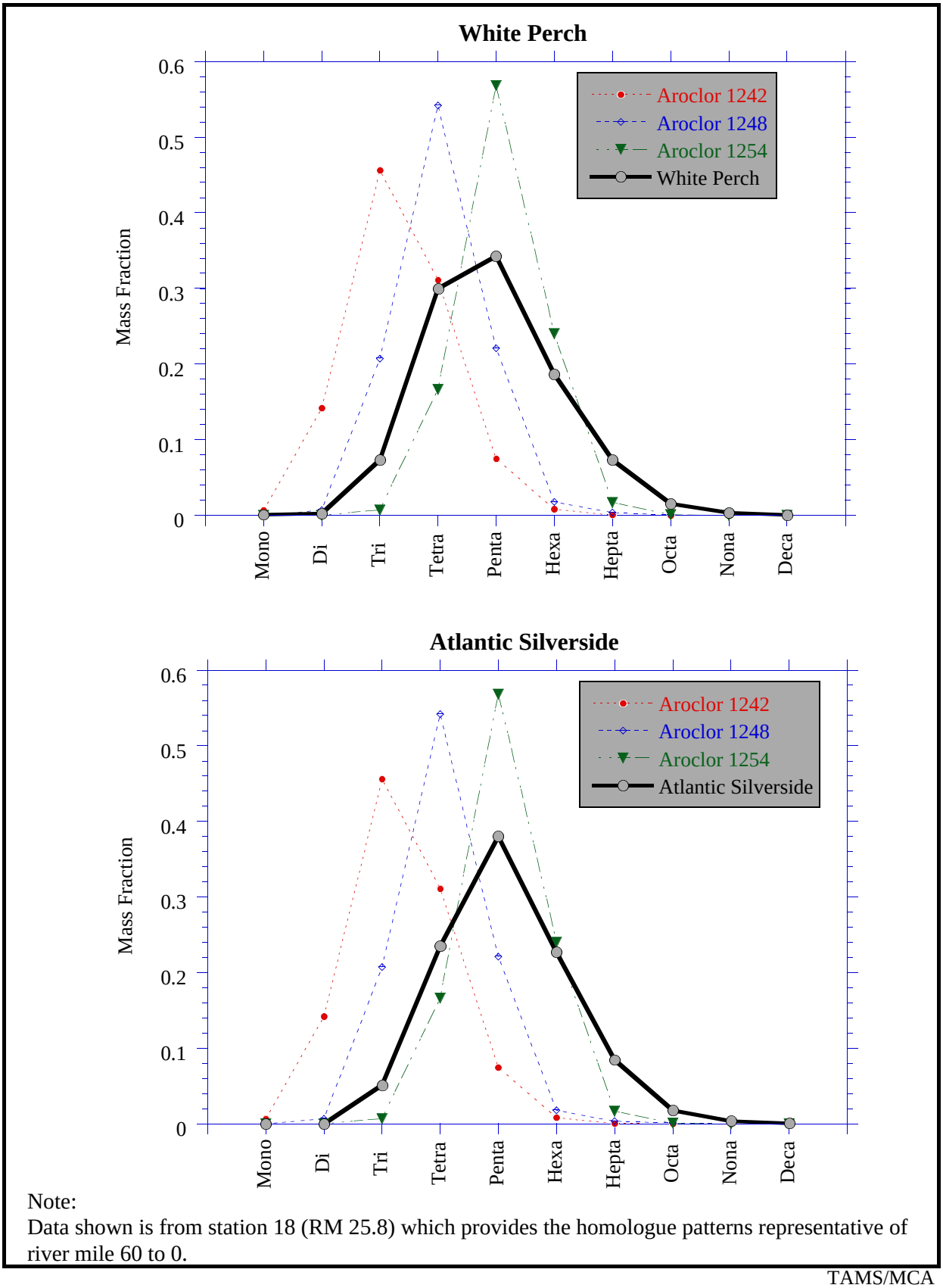
Figure K-23
A Comparison of Homologue Patterns in Aroclors And Hudson River Fish: River Mile 156 to 100
1993 USEPA and NOAA Data



Note:
Data shown is from station 15 (RM 89.4) which provides the homologue patterns representative of river mile 100 to 60.

TAMS/MCA

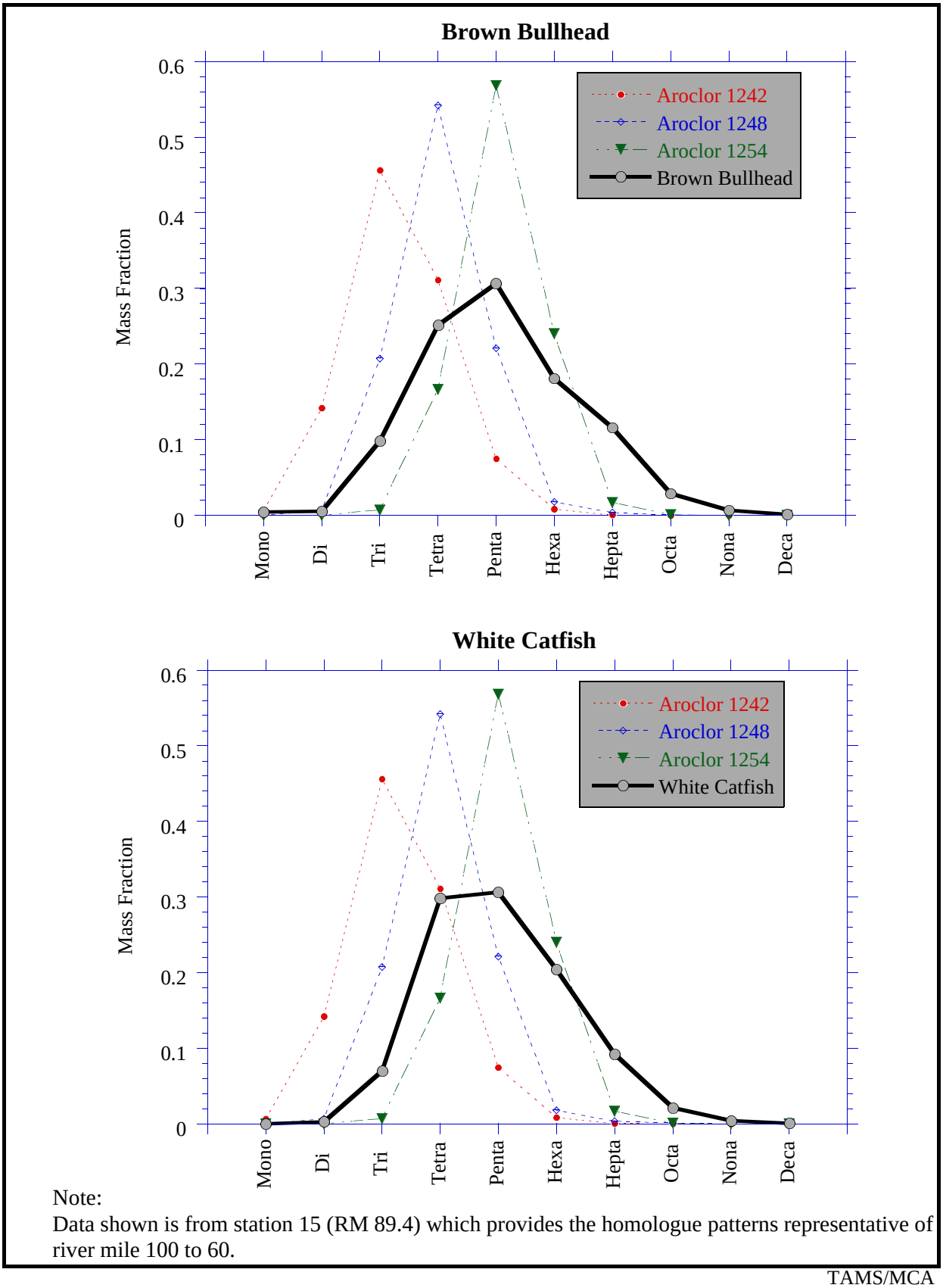
Figure K-24
A Comparison of Homologue Patterns in Aroclors and Hudson River Fish: River Mile 100 to 60
1993 USEPA and NOAA Data



Note:
Data shown is from station 18 (RM 25.8) which provides the homologue patterns representative of river mile 60 to 0.

TAMS/MCA

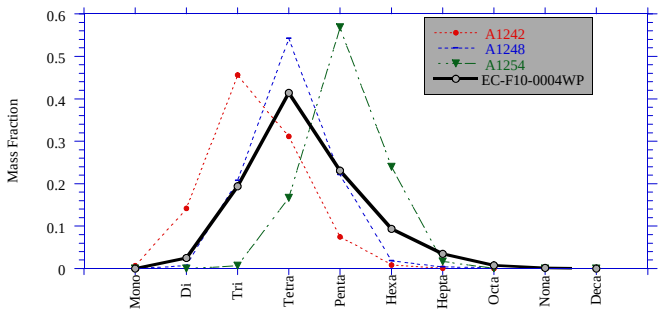
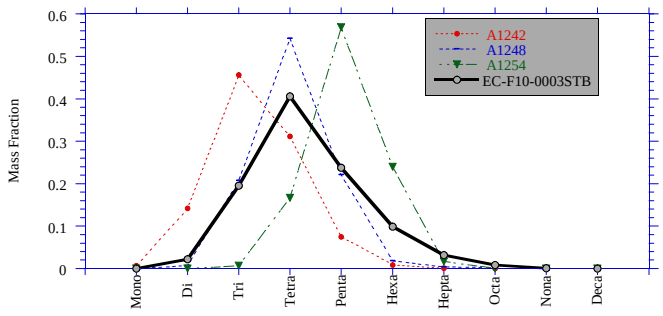
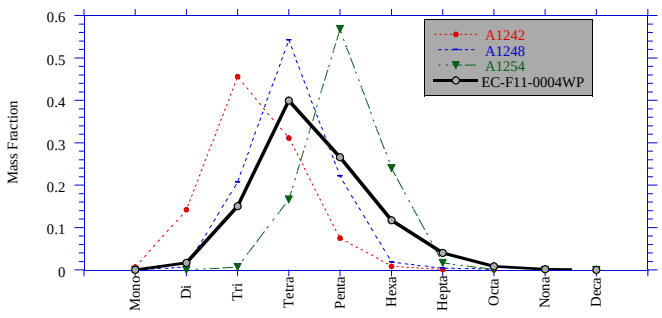
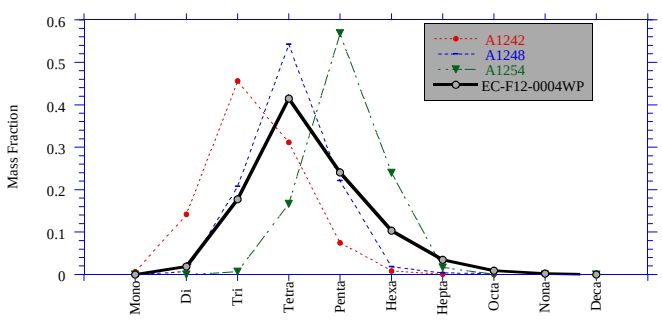
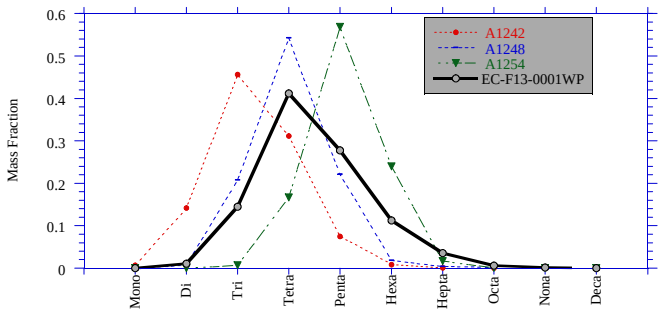
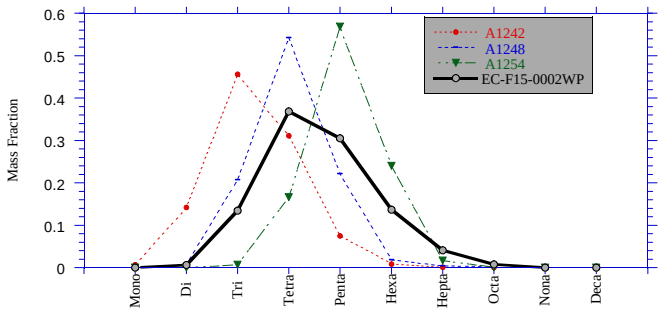
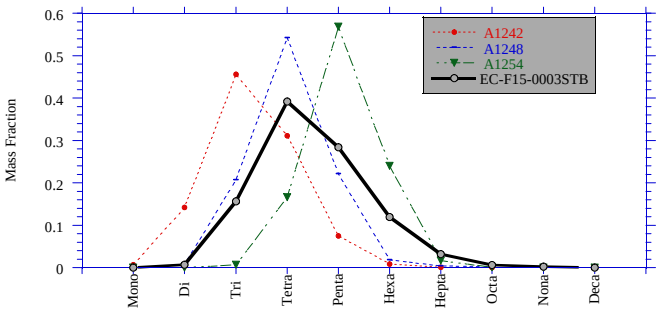
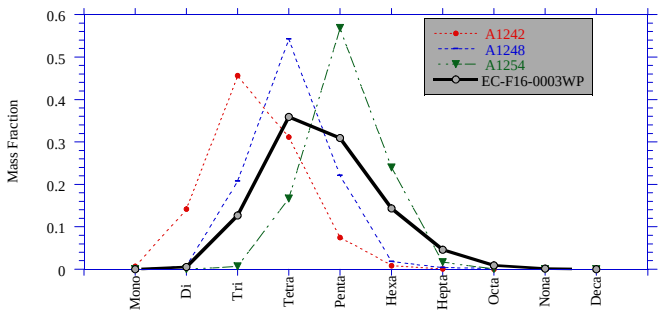
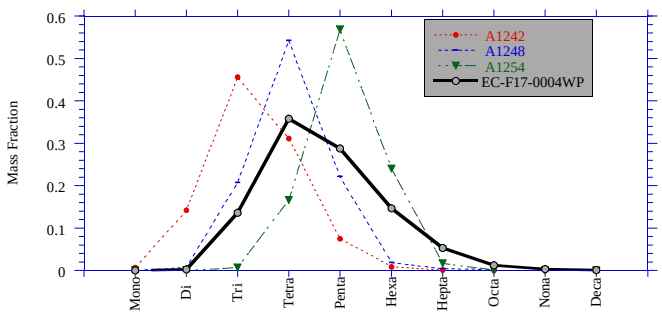
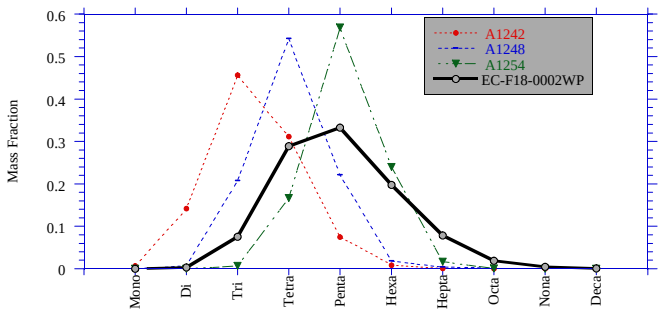
Figure K-25
A Comparison of Homologue Patterns in Aroclors and Hudson River Fish: River Mile 60 to 0
1993 USEPA and NOAA Data



Note:
Data shown is from station 15 (RM 89.4) which provides the homologue patterns representative of river mile 100 to 60.

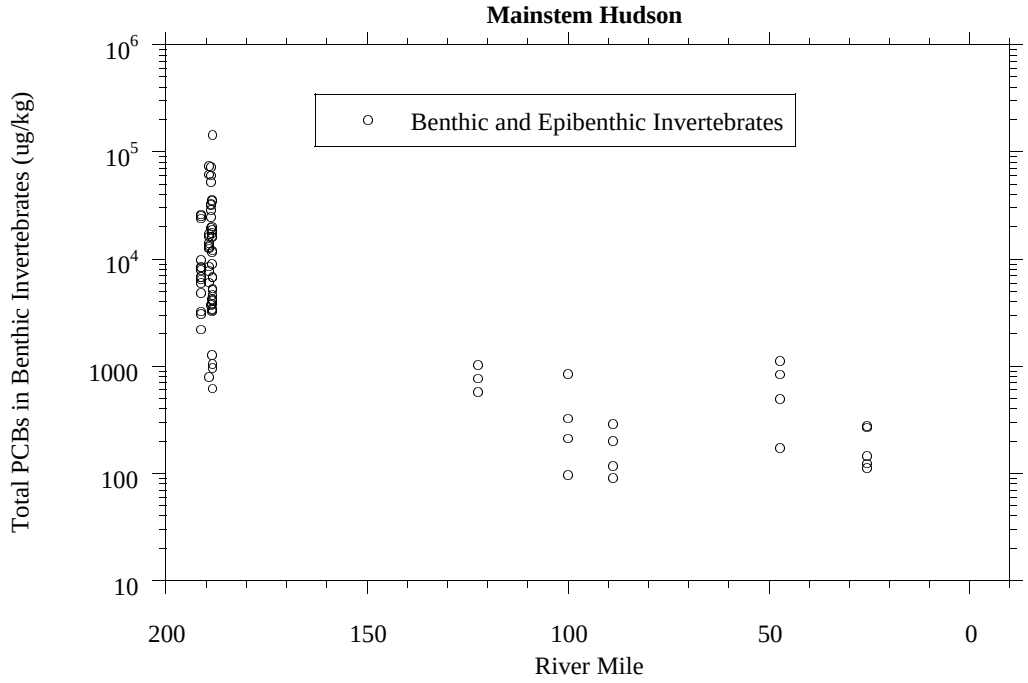
TAMS/MCA

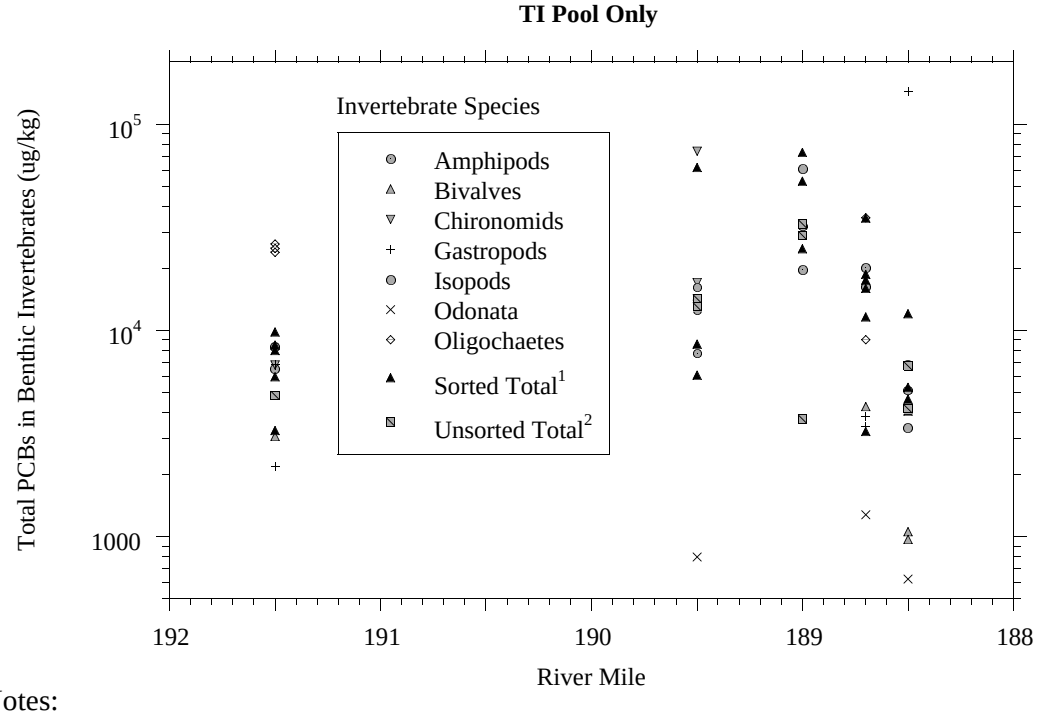
Figure K-26
A Comparison of Homologue Patterns in Aroclors and Catfish:
River Mile 100 to 60
1993 USEPA and NOAA Data

River Mile	White Perch	Striped Bass
143.5		
137.2		
122.4		
113.8		
88.9		
58.7		
47.3		
25.8		

TAMS/MCA

Figure K-27
A Comparison of Homologue Patterns in Aroclors and Lower Hudson River Fish: River Mile 154 to 0
1993 USEPA and NOAA Data

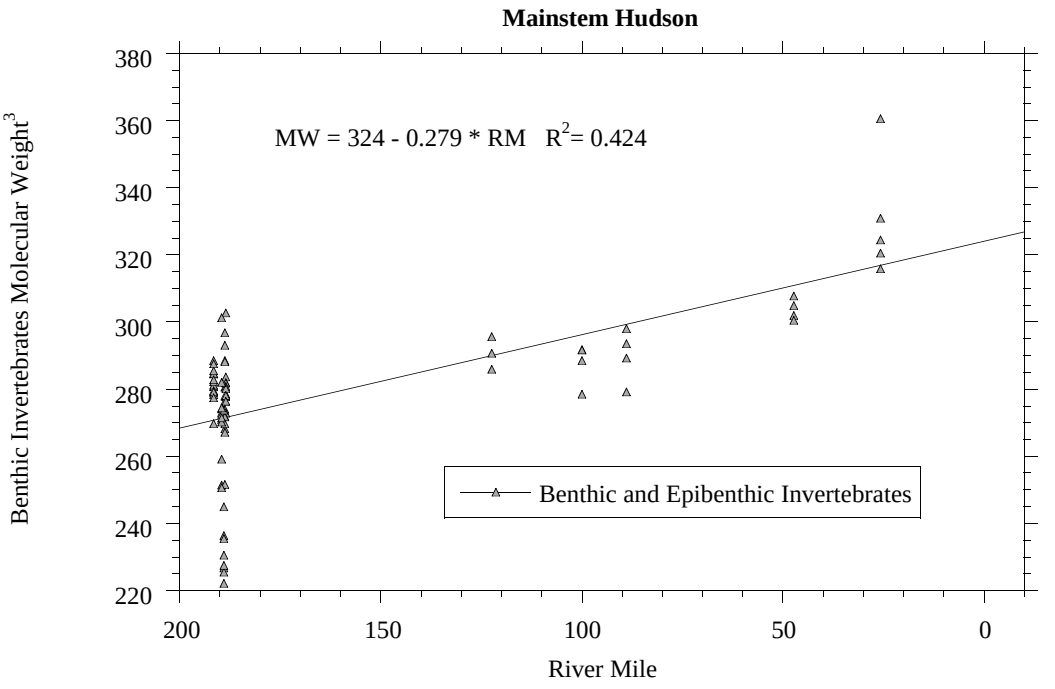


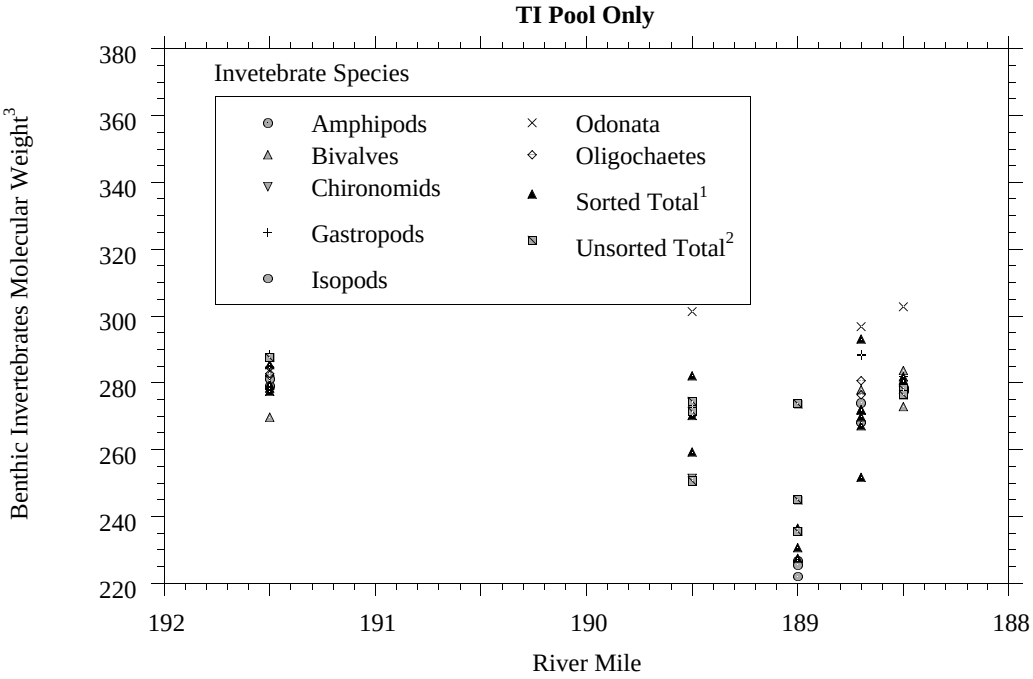


Notes:
1. Sorted total = all invertebrates after removing individual taxa for analysis.
2. Unsorted total = all invertebrates together with none removed for individual taxa analysis.

TAMS/MCA

Figure K-28
Total PCB Concentration
in Benthic Invertebrates vs. River Mile

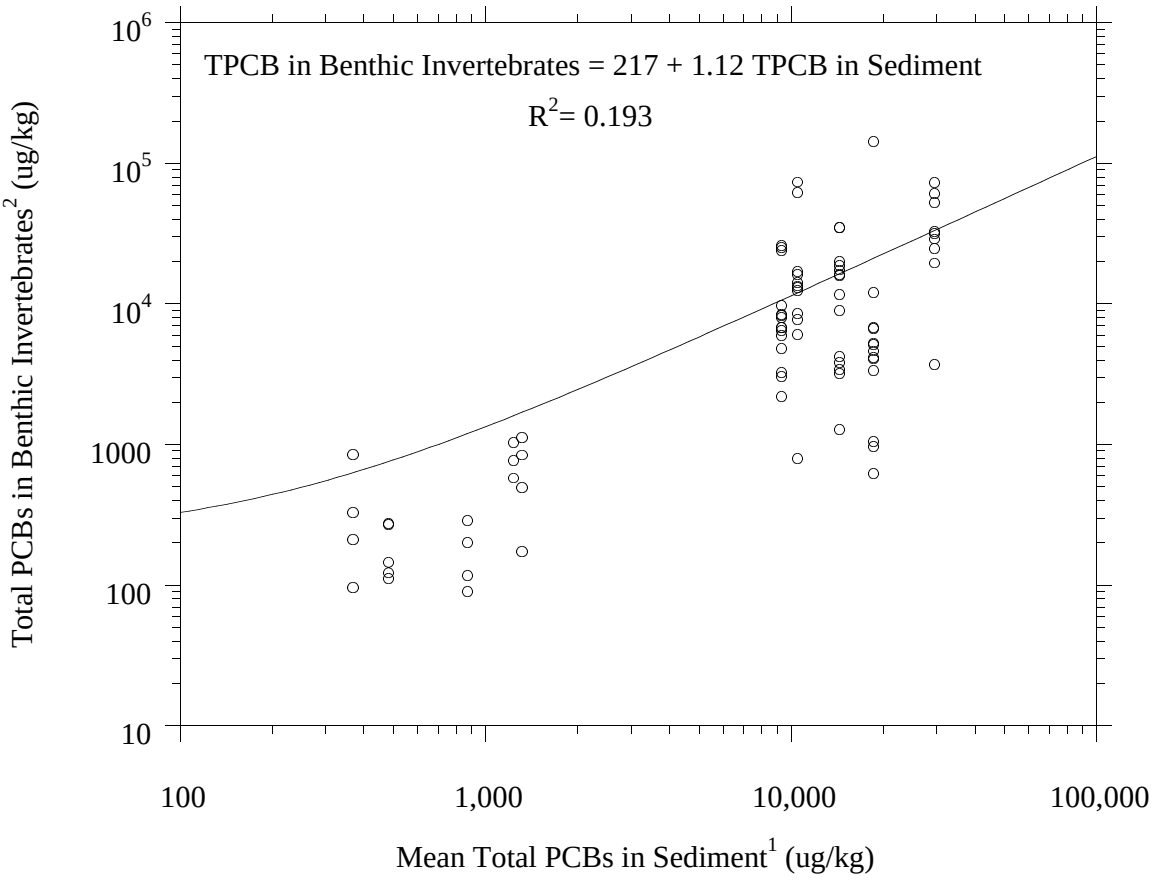




Notes:
1. Sorted total = all invertebrates after removing individual taxa for analysis.
2. Unsorted total = all invertebrates together with none removed for individual taxa analysis.
3. Points represent the individual benthic and epibenthic invertebrates at each location.

TAMS/MCA

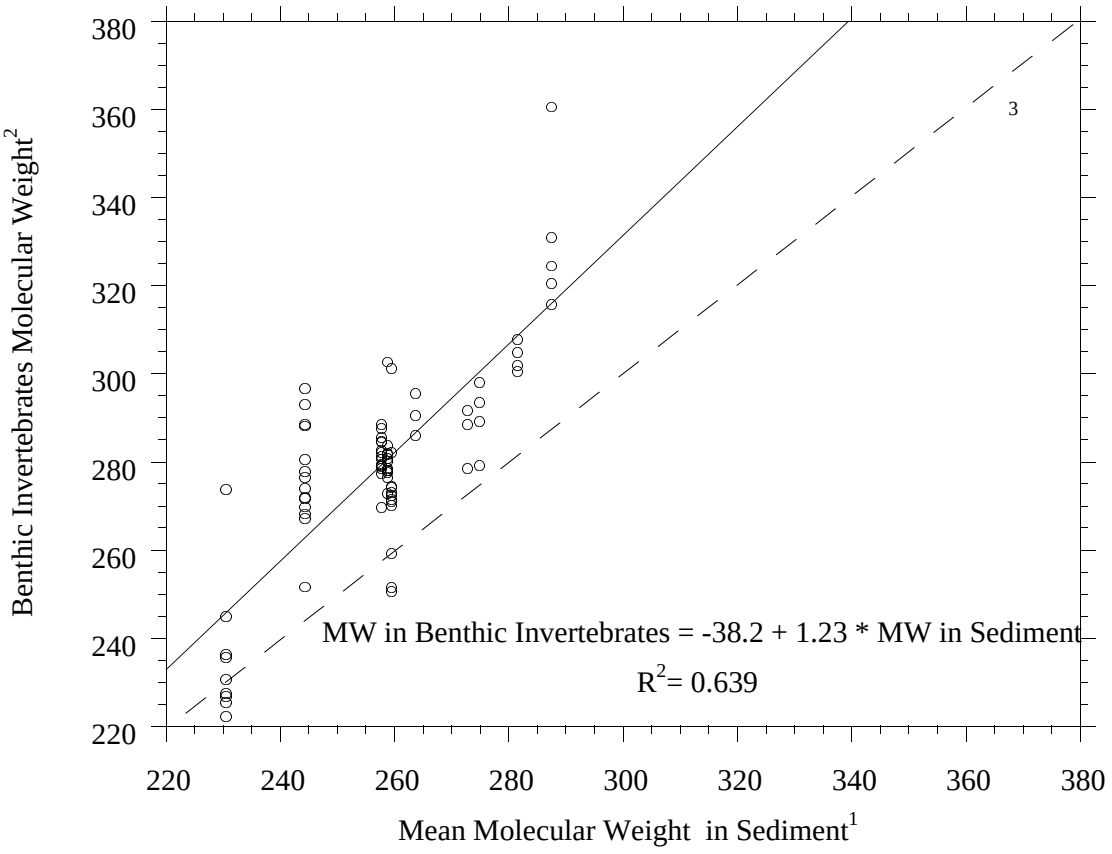
Figure K-29
Molecular Weight of Total PCBs in Benthic Invertebrates vs. River Mile



Notes:
1. Mean Total PCB concentration based on the average of 5 sediment samples collected at each location (0-5cm depth).
2. Points represent the individual benthic and epidbenthic invertebrate samples from each location.

TAMS/MCA

Figure K-30
Relationship of Total PCB Concentration Between Benthic Invertebrates and Sediment

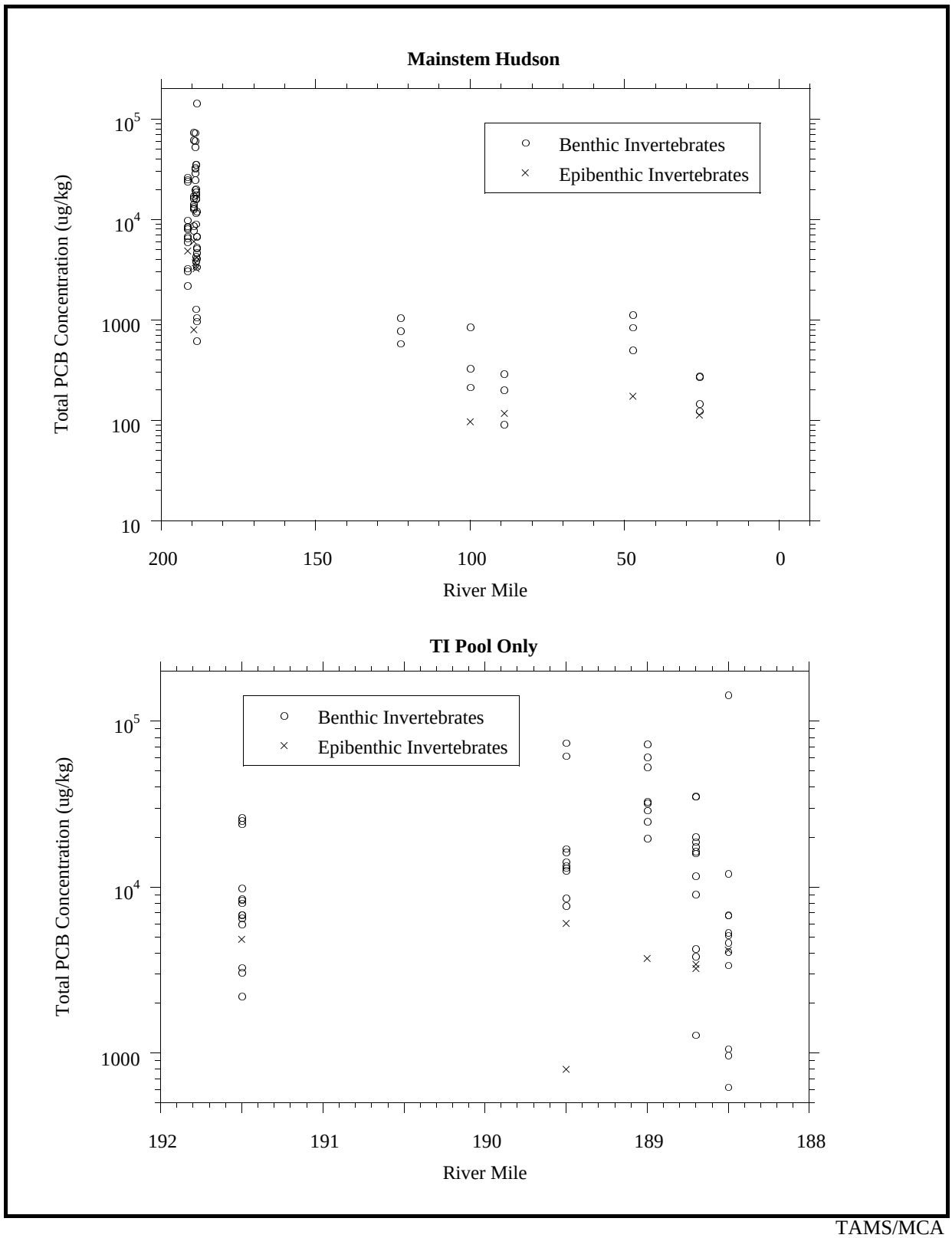


Notes:

1. Total PCB mean molecular weight based on the average of 5 sediment samples collected at each location (0-5cm depth).
2. Points represent the individual benthic and epibenthic invertebrate sample from each location.
3. Dashed line represents identical MW in sediment and benthic invertebrates.

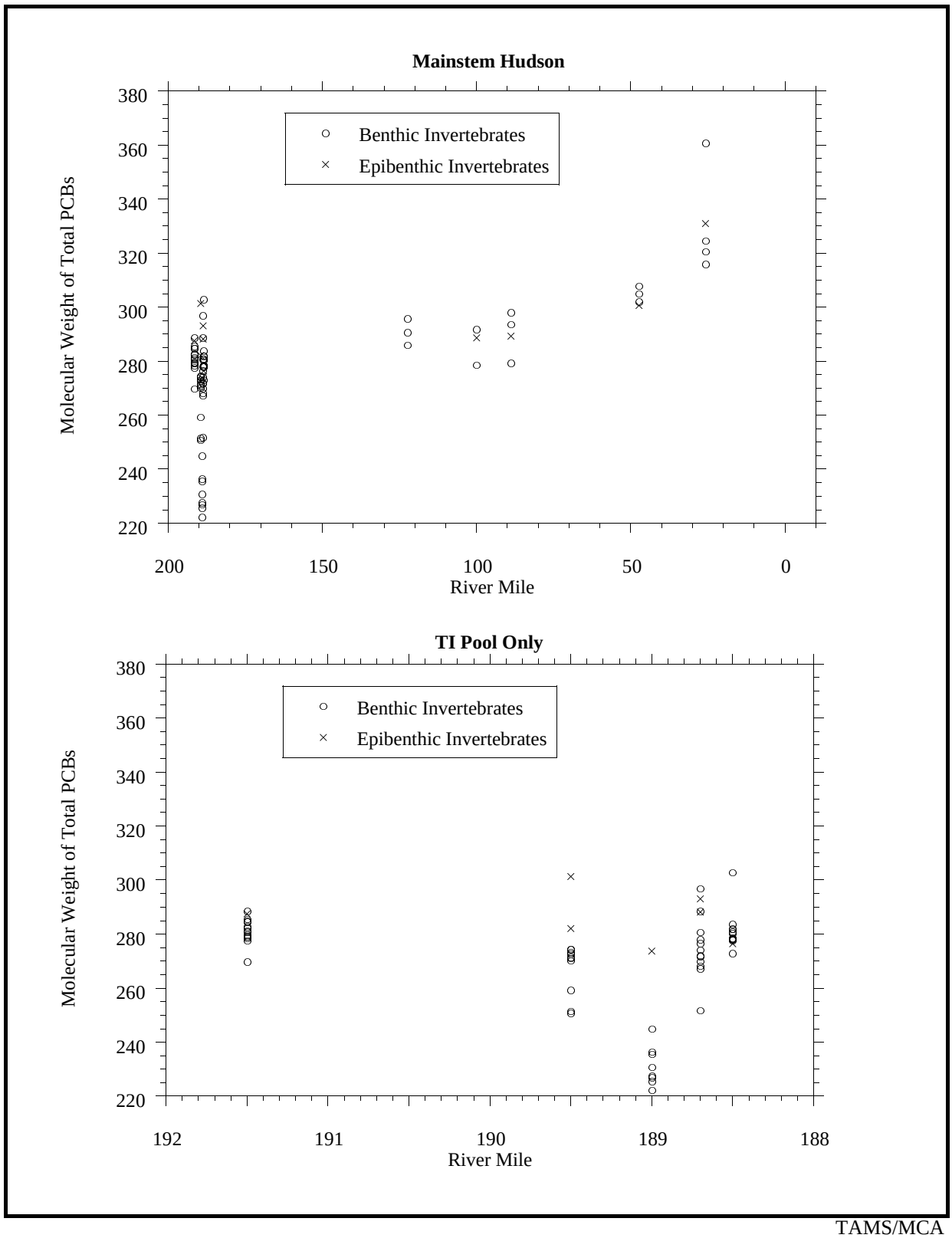
TAMS/MCA

Figure K-31
Relationship of Total PCB Molecular Weight Between Benthic Invertebrates and Sediment



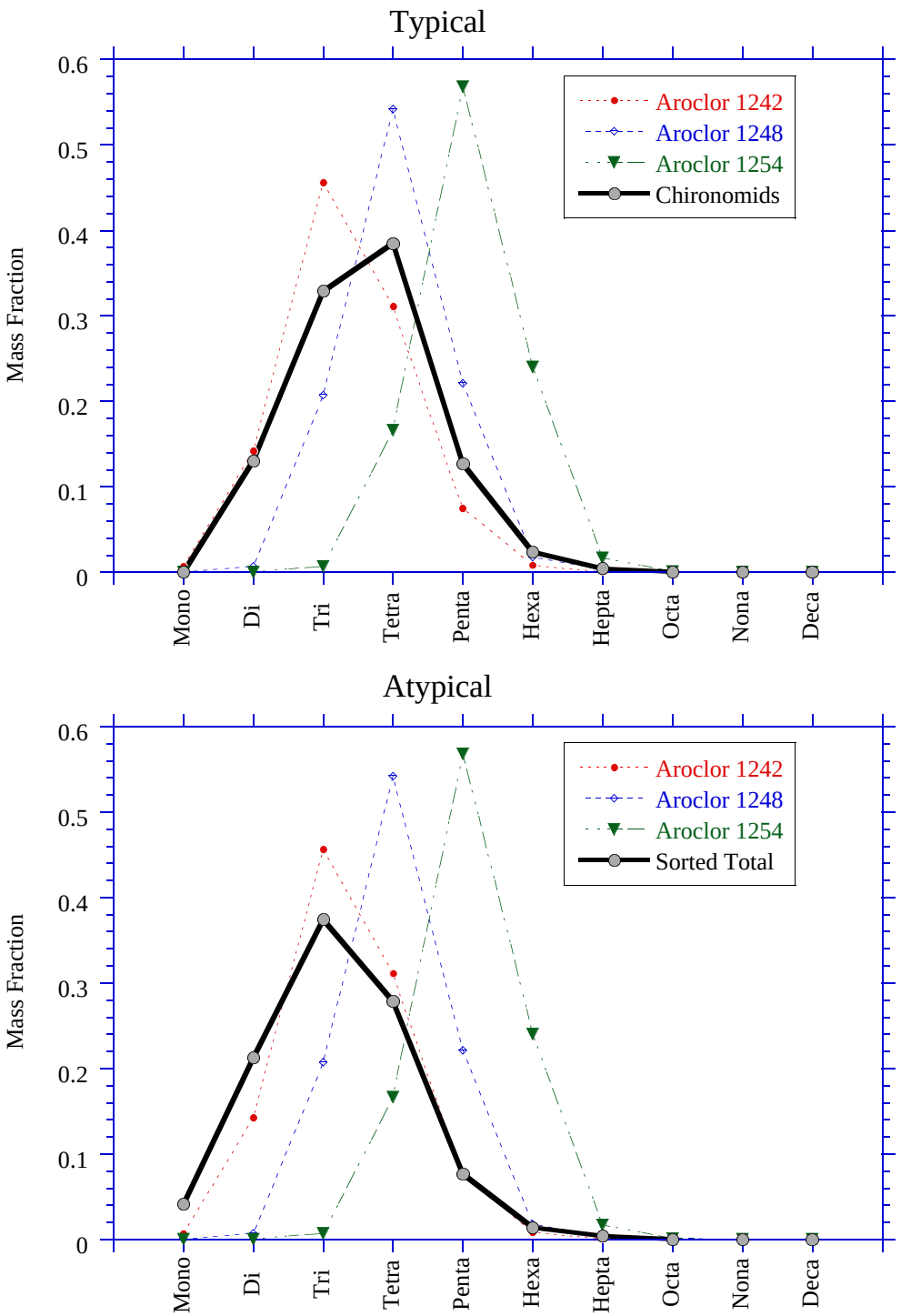
TAMS/MCA

Figure K-32
Comparisons of Total PCB Concentration Between Benthic Invertebrates and Epibenthic Invertebrates as a Function of River Mile



TAMS/MCA

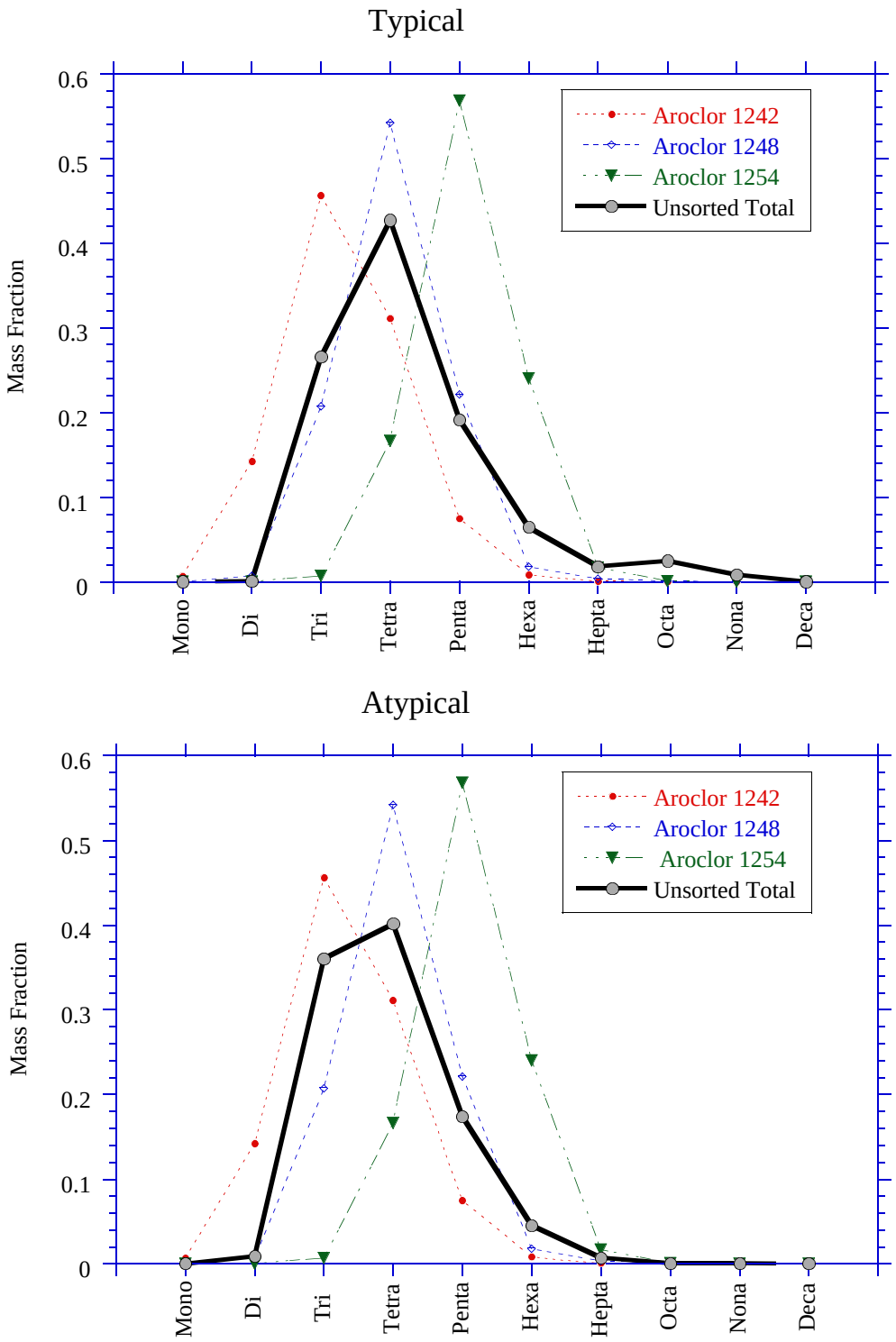
Figure K-33
Comparisons of Total PCB Molecular Weight Between Benthic Invertebrates and Epibenthic Invertebrates as a Function of River Mile



Notes:
1. Data shown is from station 4 (RM 190) which provides typical and atypical homologue patterns representative of river mile 195 to 175.
2. Sorted total = all invertebrates after removing individual taxa for analysis.

TAMS/MCA

Figure K-34
A Comparison of Homologue Patterns in Aroclors and Hudson River Benthic Invertebrates: River Mile 195 to 175
1993 USEPA and NOAA Data

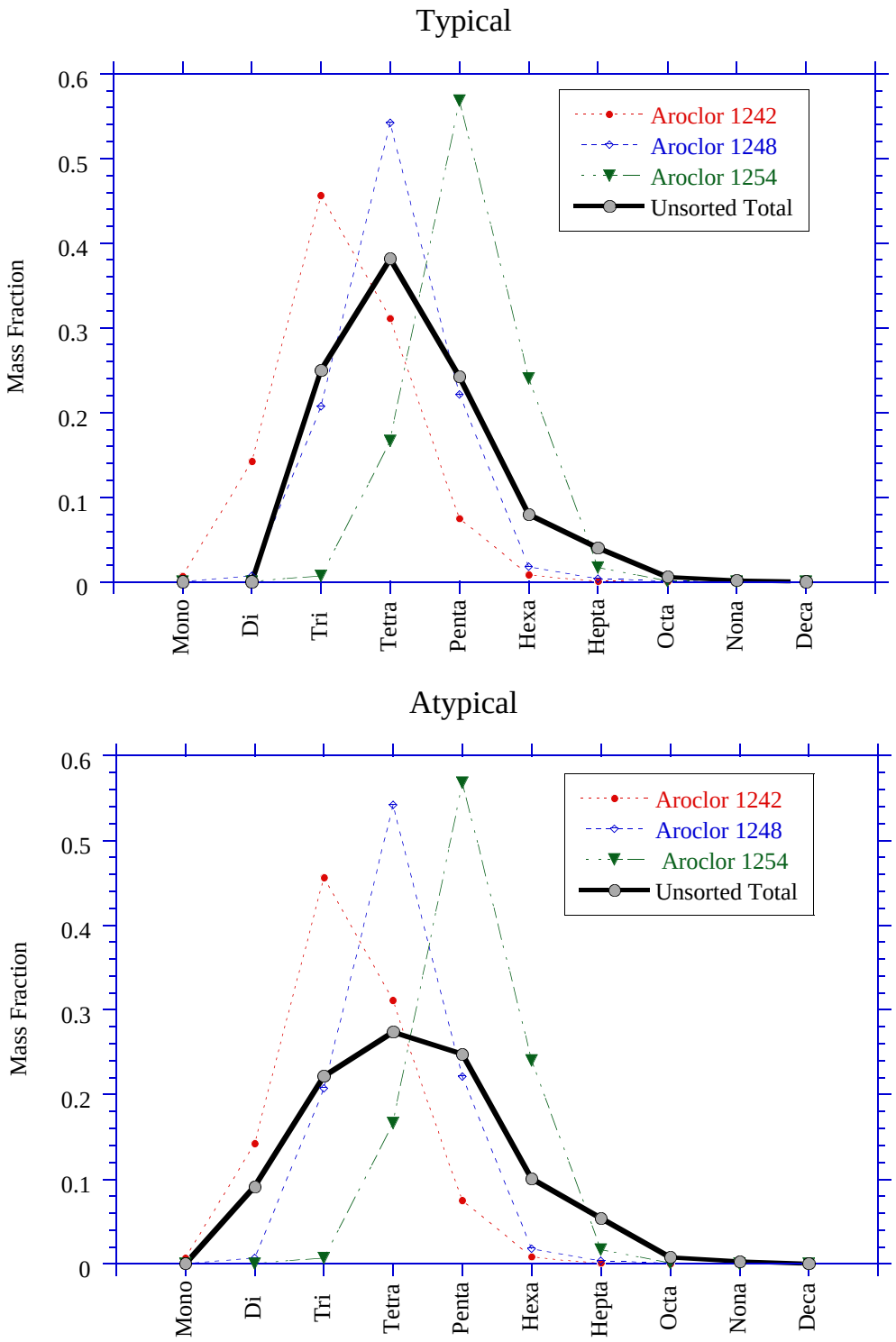


Notes:

1. Data shown is from station 12 (RM 122.7) which provides typical and atypical homologue patterns representative of river mile 156 to 100.
2. Unsorted total = all invertebrates together with none removed for individual taxa analysis.

TAMS/MCA

Figure K-35
A Comparison of Homologue Patterns in Aroclors And Hudson River Benthic Invertebrates: River Mile 156 to 100
1993 USEPA and NOAA Data

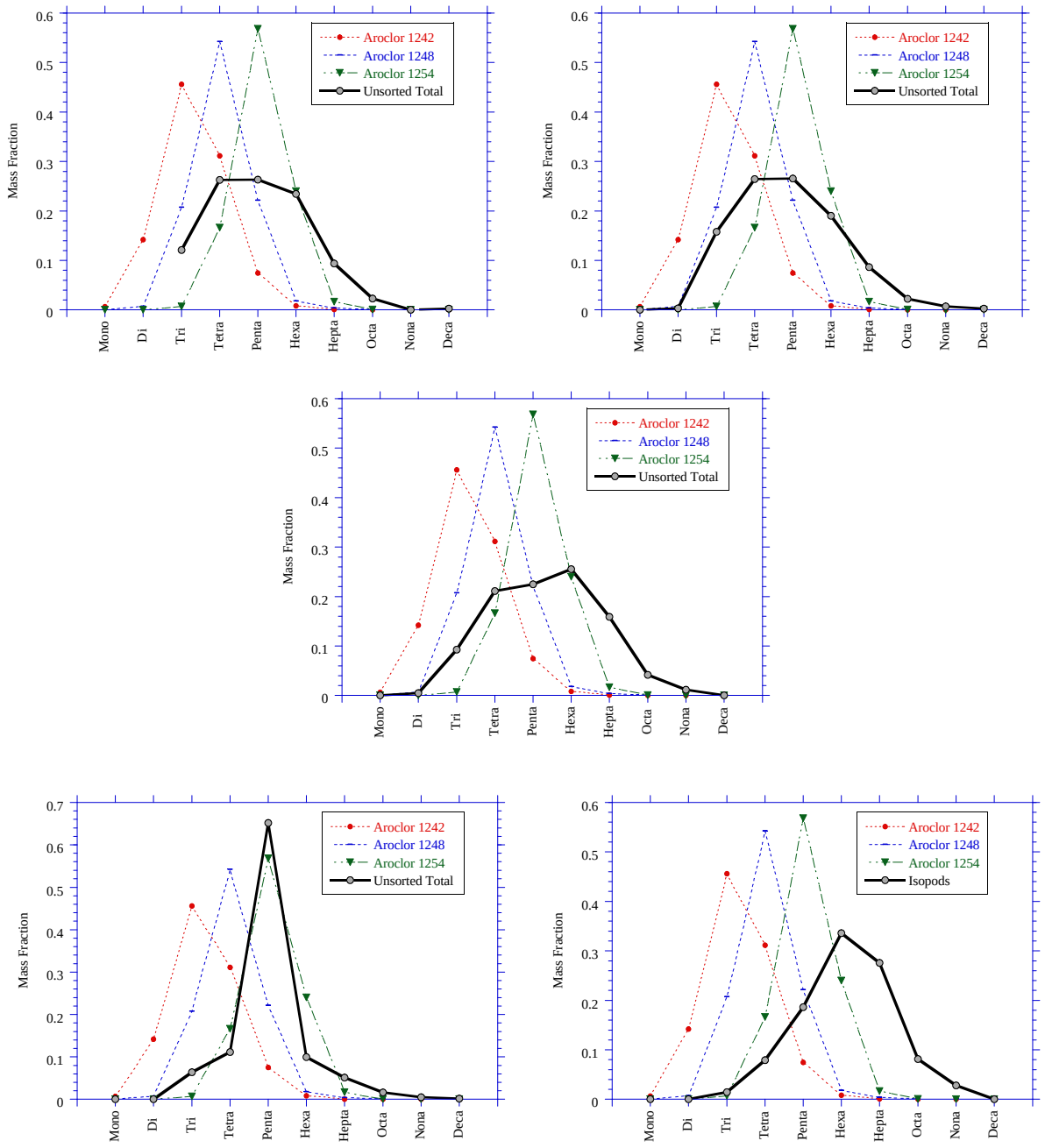


Notes:

1. Data shown is from station 15 (RM 89.4) which provides typical and atypical homologue patterns representative of river mile 100 to 60.
2. Unsorted total = all invertebrates together with none removed for individual taxa analysis.

TAMS/MCA

Figure K-36
A Comparison of Homologue Patterns in Aroclors and Hudson River Benthic Invertebrates: River Mile 100 to 60
1993 USEPA and NOAA Data

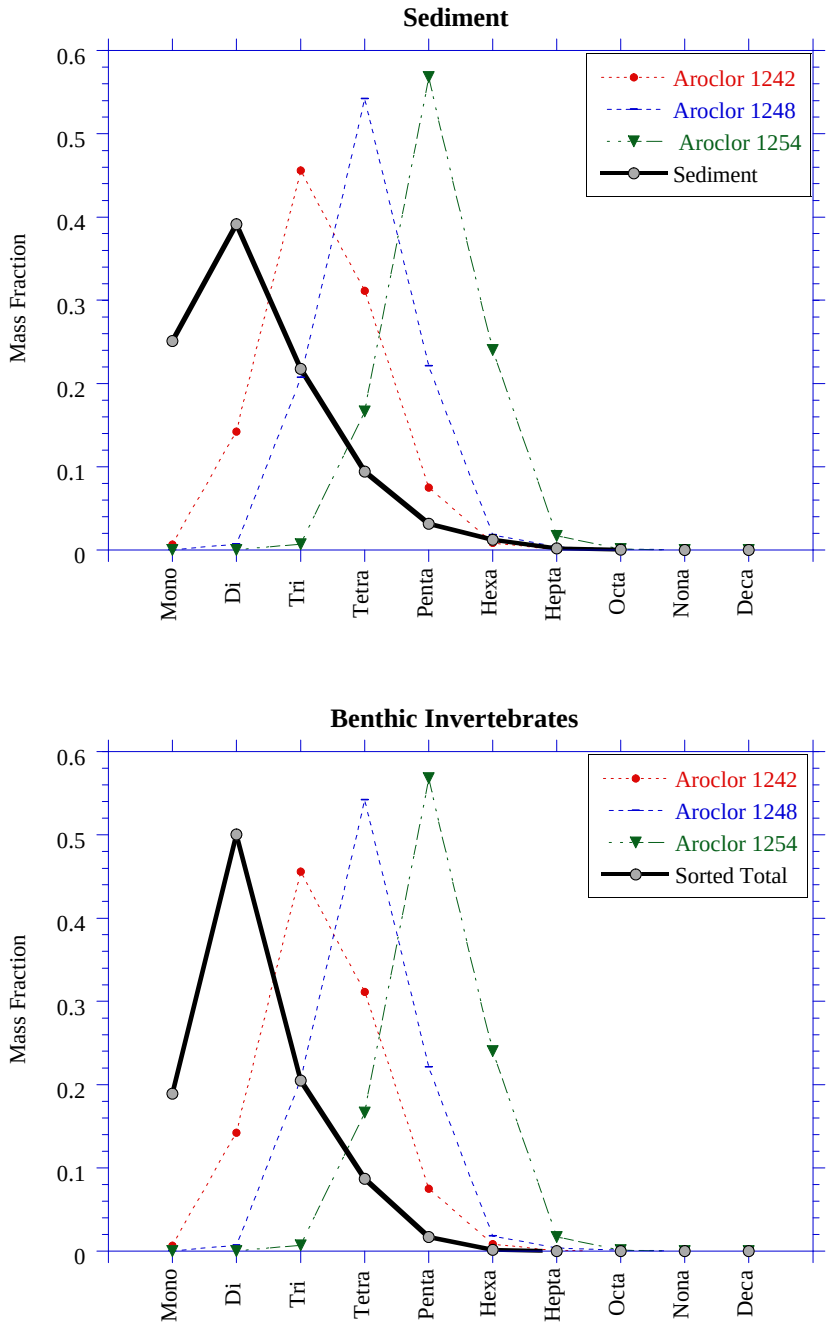


Note: All five samples were collected on August 26, 1993.

Station 18 (RM 25.8)

TAMS/MCA

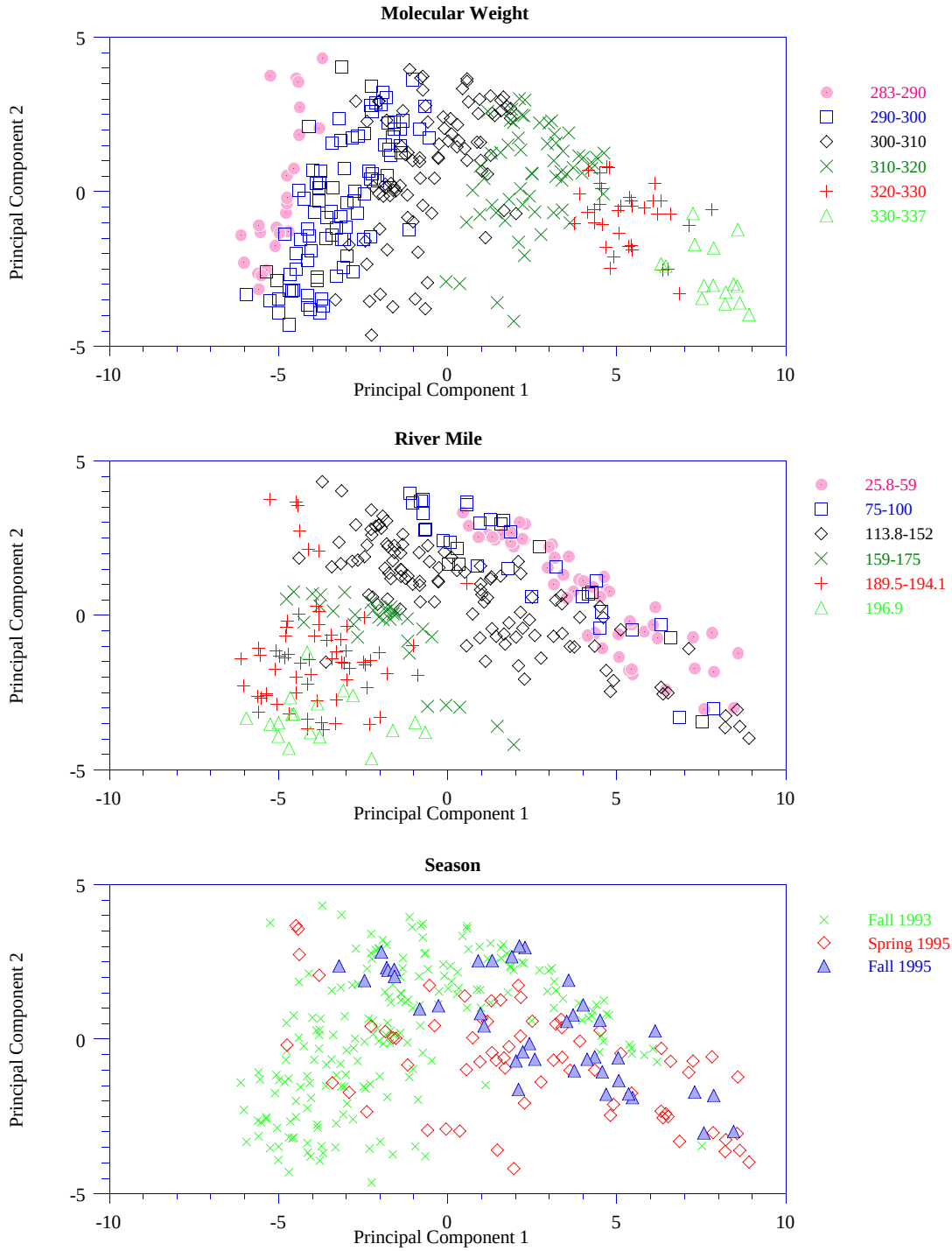
Figure K-37
A Comparison of Homologue Patterns in Aroclors and Hudson River Benthic Invertebrates: River Mile 60 to 0
1993 USEPA and NOAA Data



Station 8 (RM 169.5)

TAMS/MCA

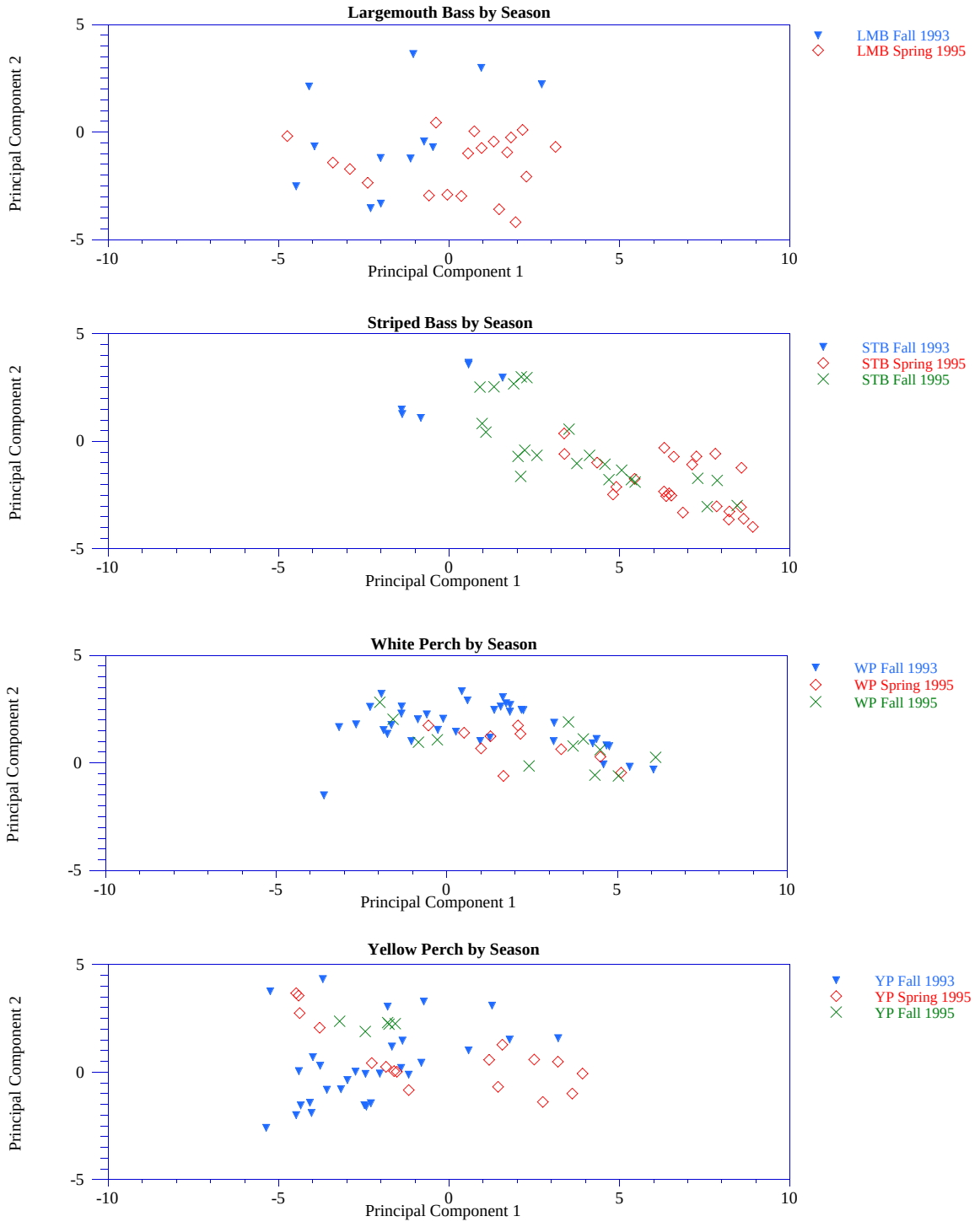
Figure K-38
A Comparison of Homologue Patterns Between Sediment and Benthic Invertebrates at Station 5 in TI Pool
1993 USEPA and NOAA Data



Note: 1993 USEPA and NOAA Fish Data and 1995 NOAA Fish Data
316 Samples (Eleven 1995 Samples Omitted, NOAA, 1997)

TAMS/MCA

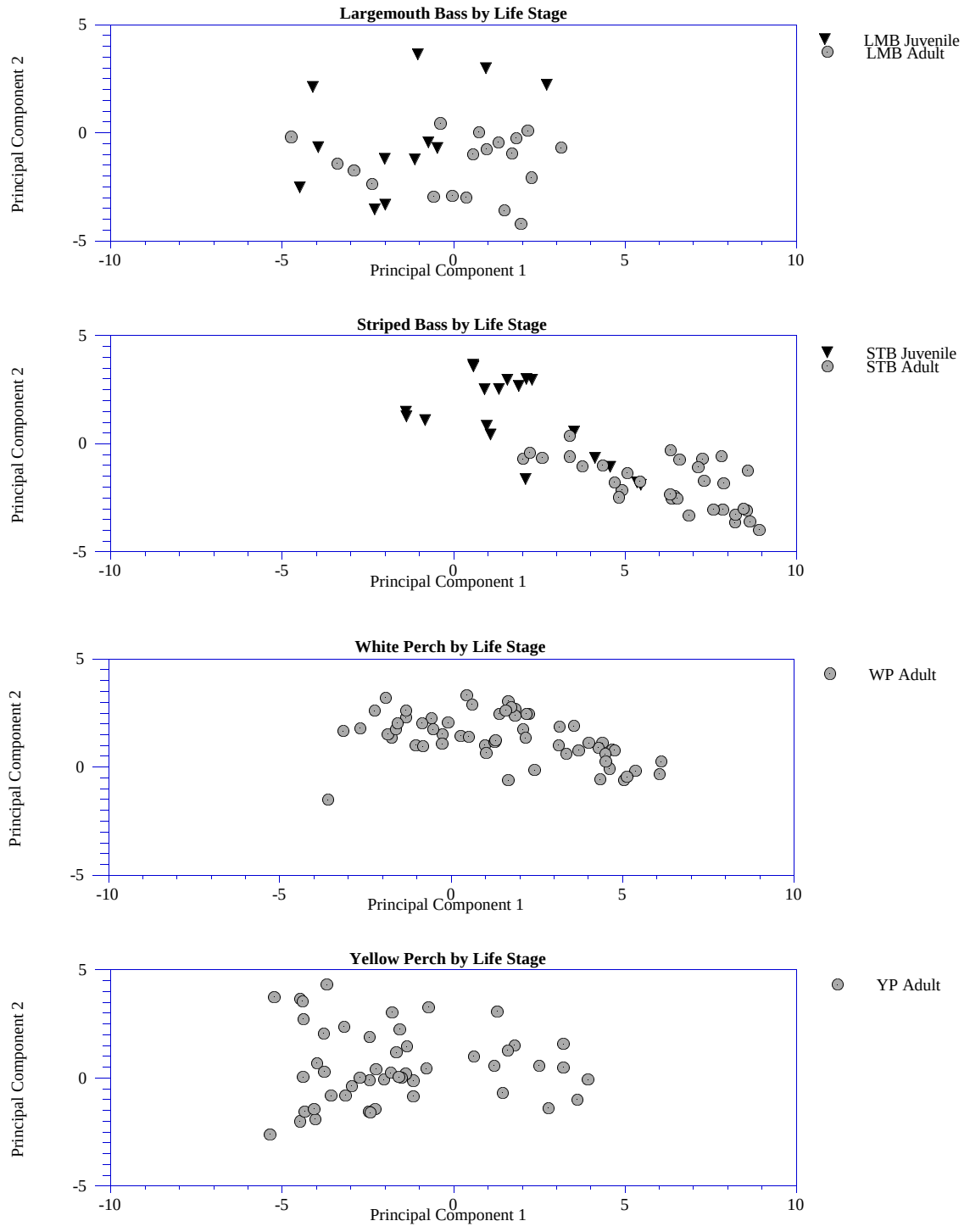
Figure K-39
Principal Component Results for 1993 and 1995
Fish Data Based on 29 Congeners



Note: 1993 USEPA and NOAA Fish Data and 1995 NOAA Fish Data
316 Samples (Eleven 1995 Samples Omitted, NOAA, 1997)

TAMS/MCA

Figure K-40
Principal Component Results for 1993 and 1995
Fish Samples by Species and Season
(Based on 29 Congeners)



Note: 1993 USEPA and NOAA Fish Data and 1995 NOAA Fish Data
316 Samples (Eleven 1995 Samples Omitted, NOAA, 1997)

TAMS/MCA

Figure K-41
Principal Component Results for 1993 and 1995
Fish Samples by Life Stage
(Based on 29 Congeners)

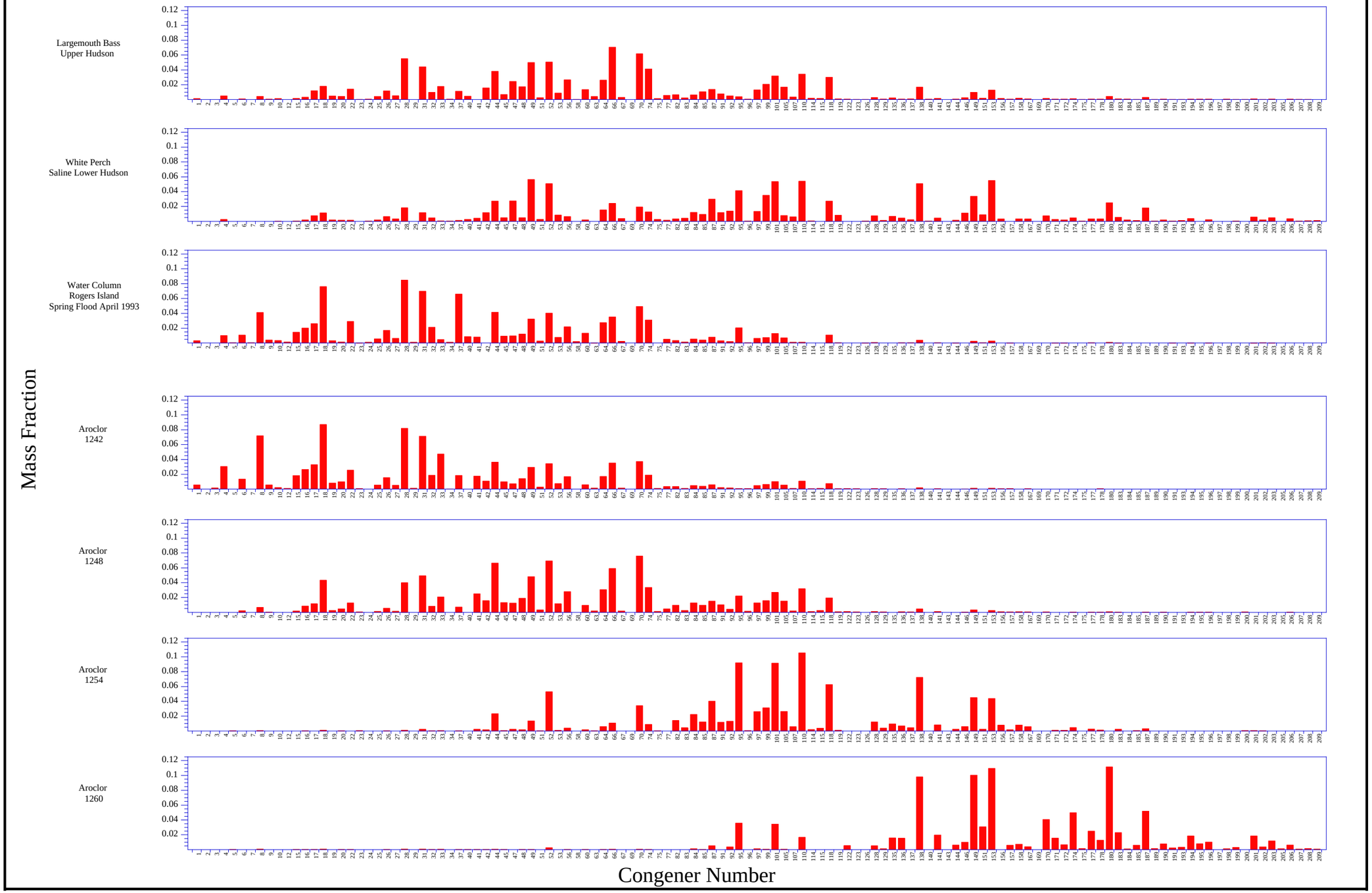


Figure K-42
Comparison of Congener Mass Fraction in Hudson River Fish and Several Aroclor Standards: Linear Scale

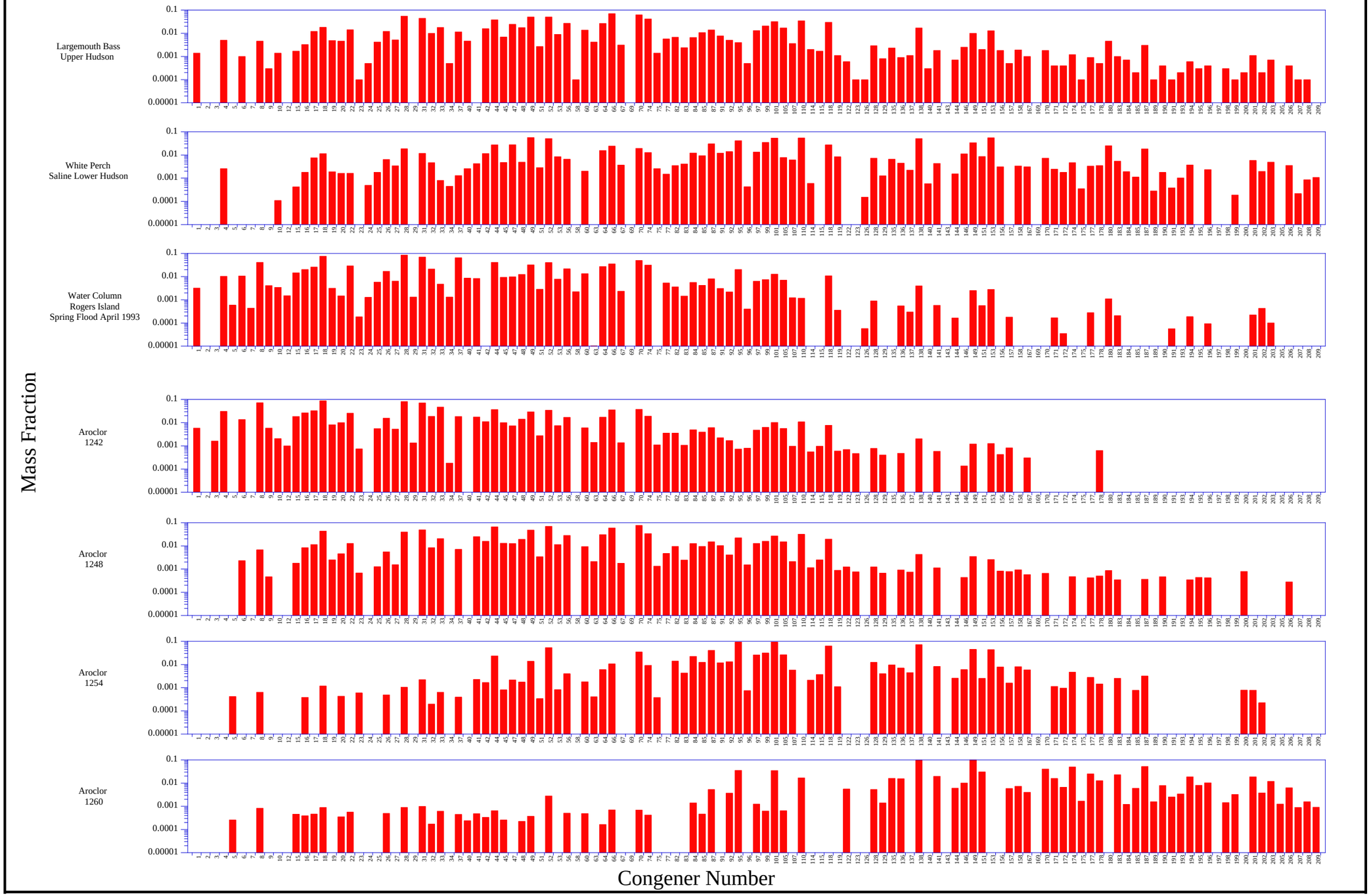
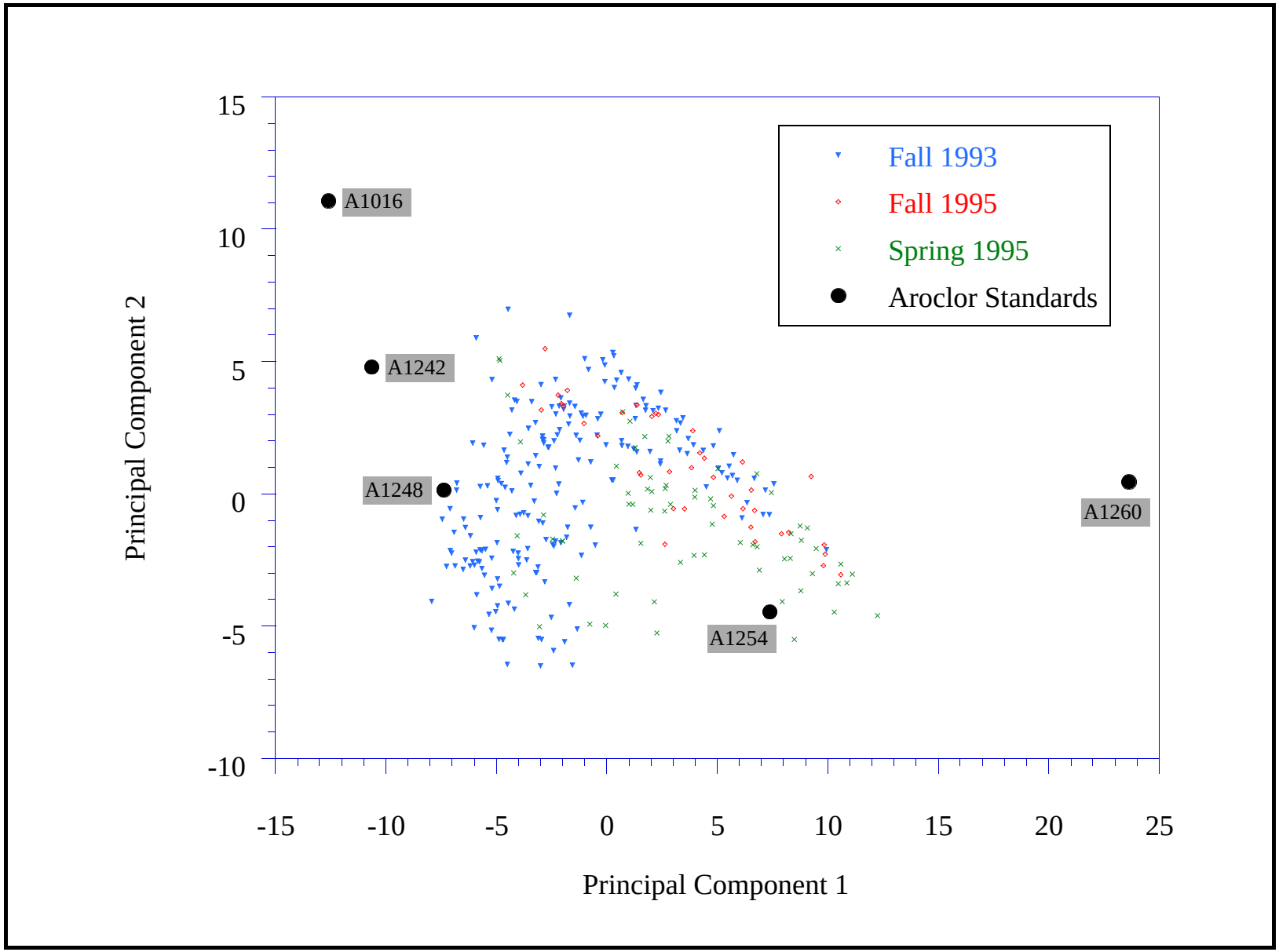
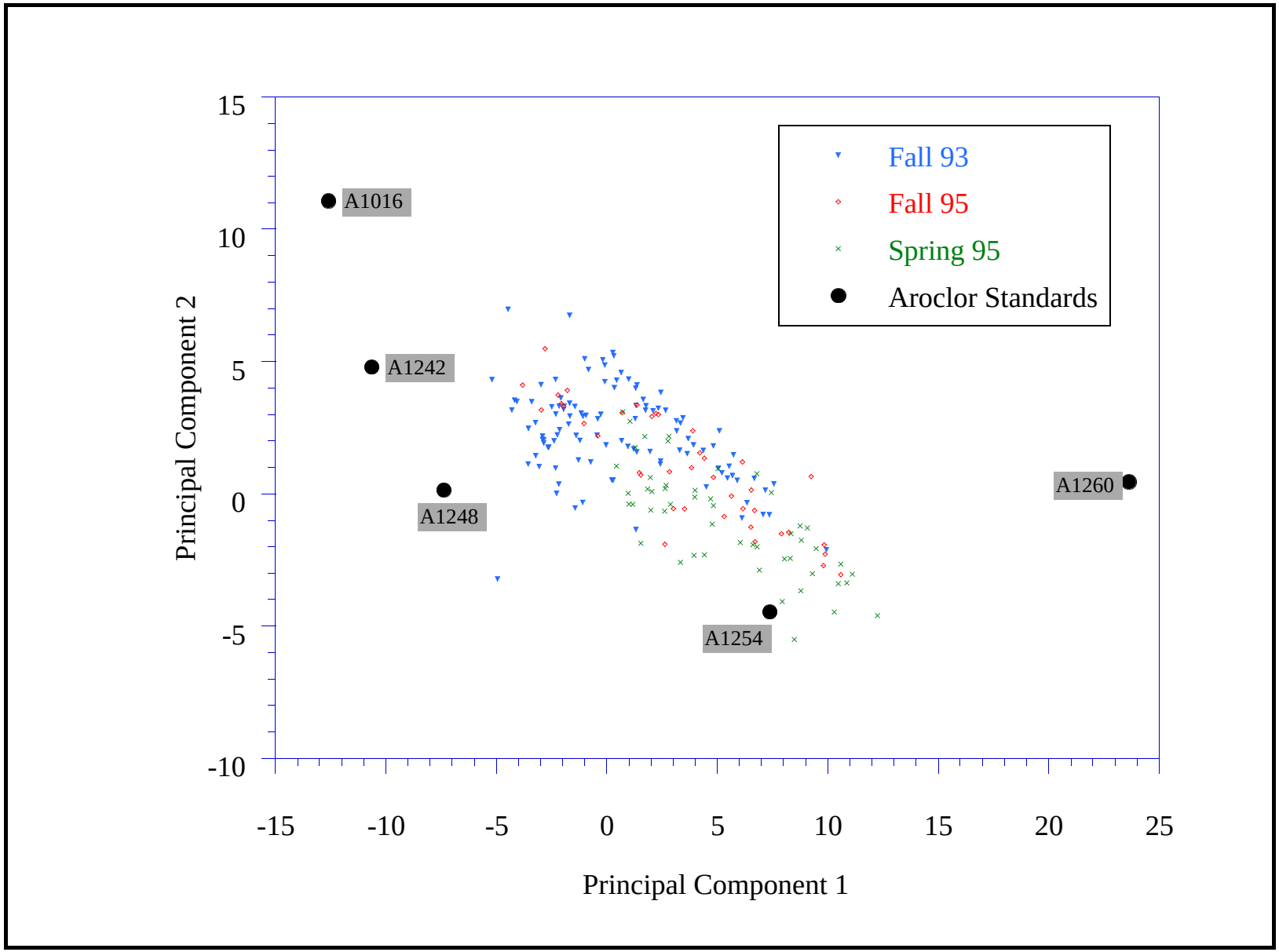


Figure K-43
Comparison of Congener Mass Fraction in Hudson River Fish and Several Aroclor Standards: Semilogarithmic Scale



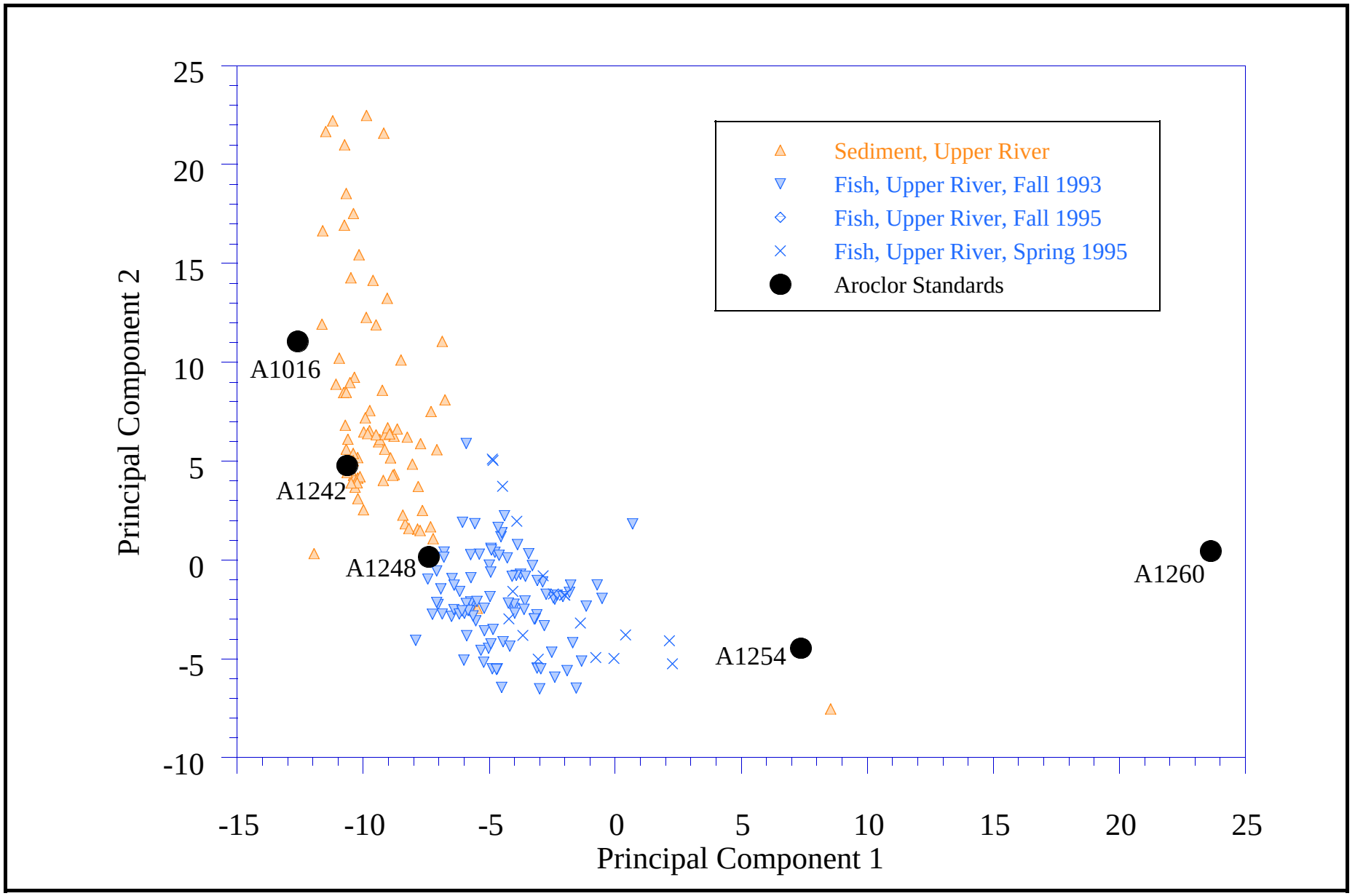
TAMS/MCA

Figure K-44
Principal Component Results for 1993 and 1995 Fish Data Based on 46 Congeners
(Congener Selection from NOAA, 1997)



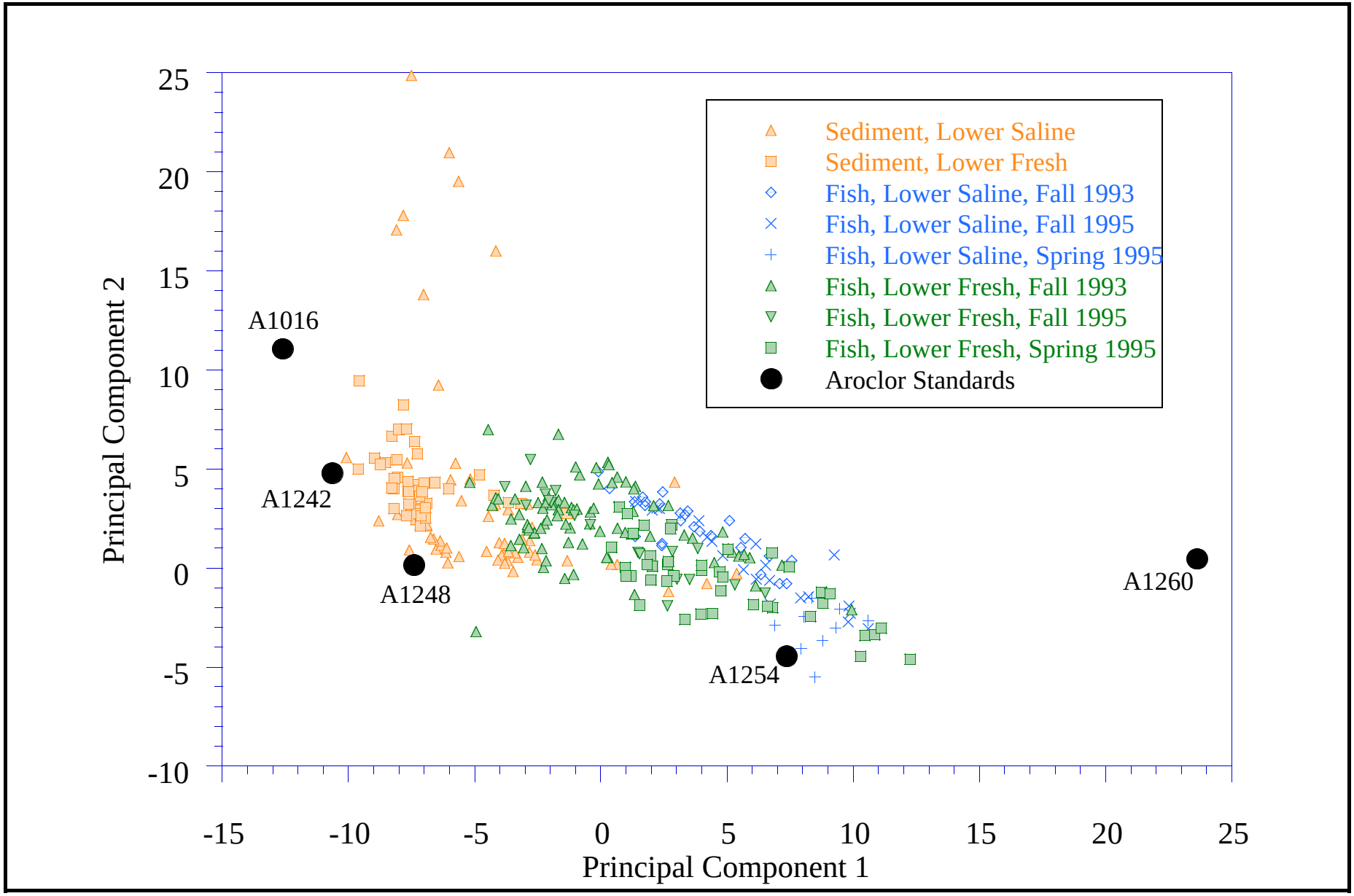
TAMS/MCA

Figure K-45
Principal Component Results for 1993 and 1995 Fish Data
Based on 46 Congeners - Lower Hudson Only
(Congener Selection from NOAA, 1997)



TAMS/MCA

Figure K-46
Principal Component Results for 1993 and 1995 Fish and Sediment Data Based on 46 Congeners - Upper Hudson Only
(Congeners Selection from NOAA, 1997)



TAMS/MCA

Figure K-47
Principal Component Results for 1993 and 1995 Fish and Sediment Data Based on 46 Congeners - Lower Hudson Only
(Congeners Selection from NOAA, 1997)

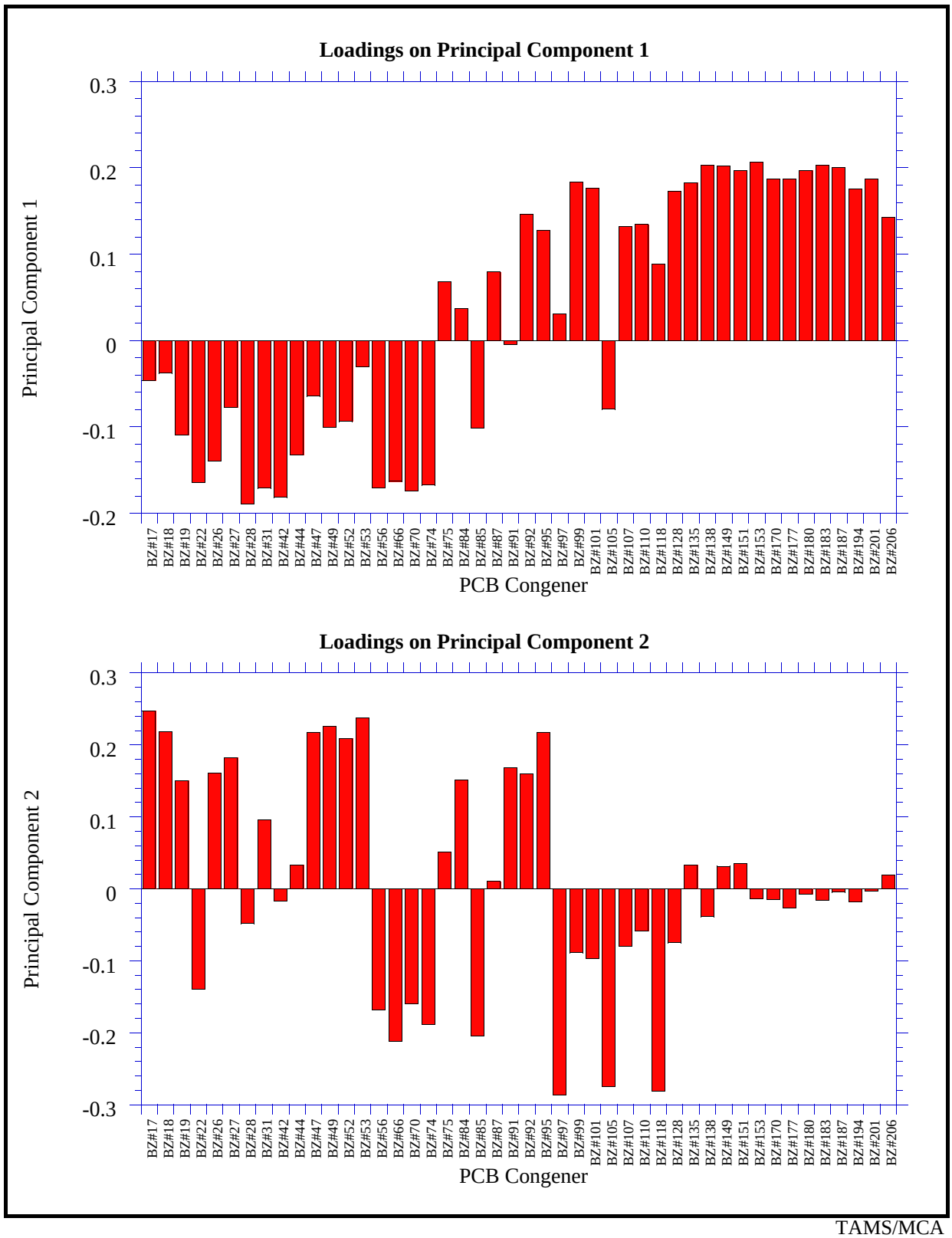


Figure K-48
Congener Loadings for Principal Components 1 and 2
Based on 46 Congeners
1993 USEPA, 1993 NOAA and 1995 NOAA Data for Fish

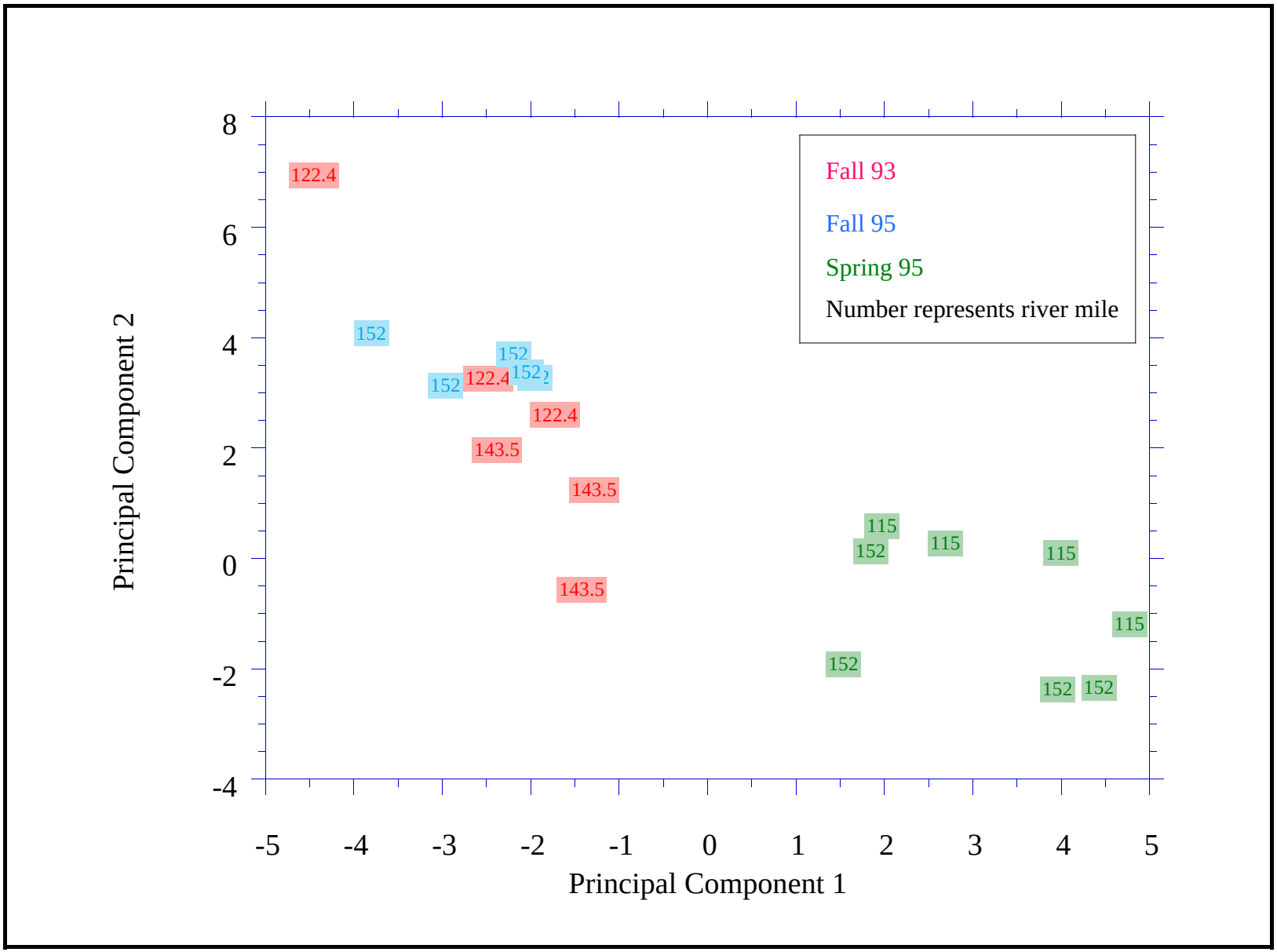


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Figure K-49
Principal Component Results by Time of Collection and River Mile for 1993 and 1995 Yellow Perch Data
Based on 46 Congeners - Freshwater Lower Hudson River (RM110 - 154)

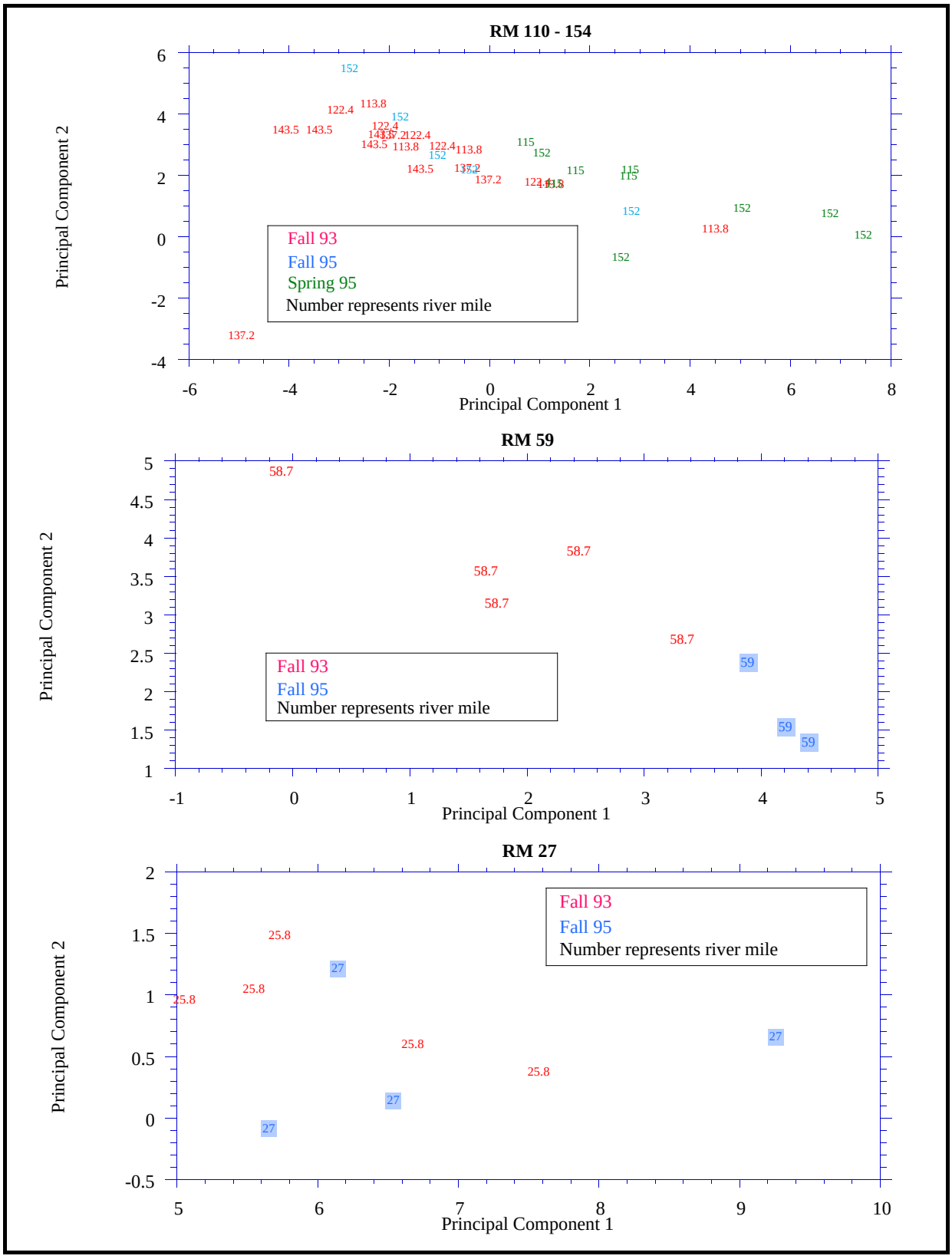

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Figure K-50
Principal Component Results by Time of Collection and River Mile for 1993 and 1995 White Perch Data
Based on 46 Congeners

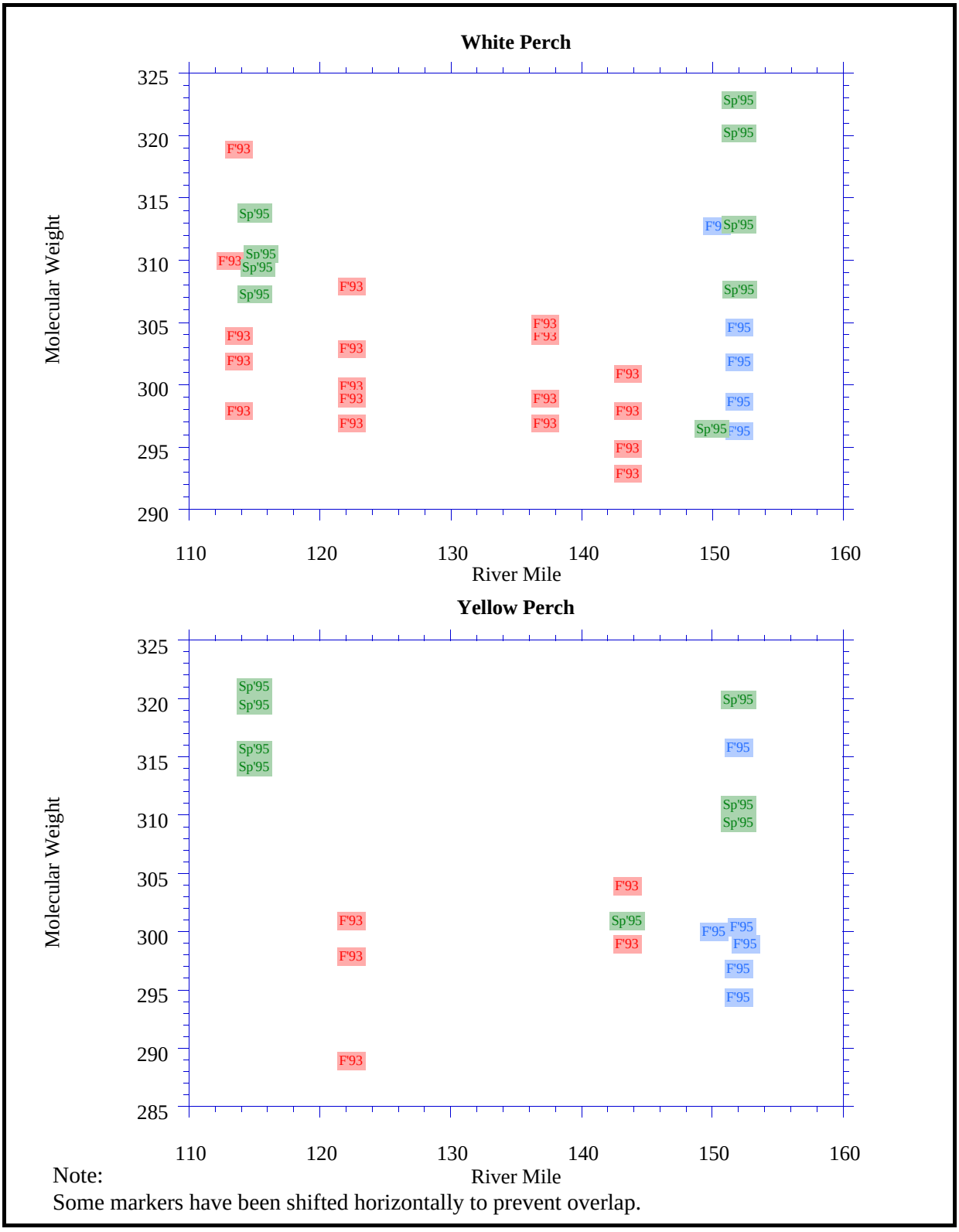

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Figure K-51
Total PCB Molecular Weight in Fish vs. River Mile
Fresh Lower Hudson River (RM 110 - 154)

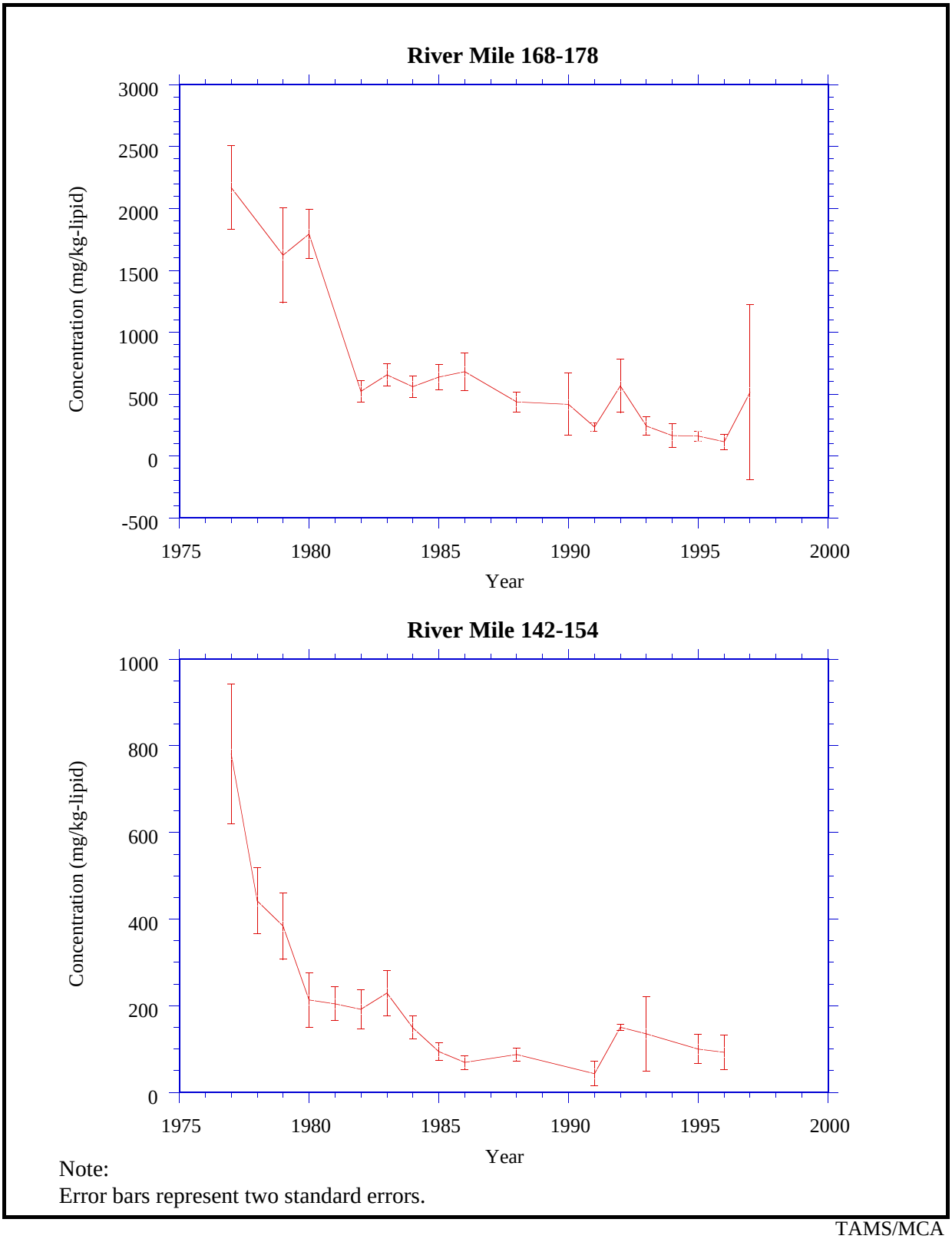
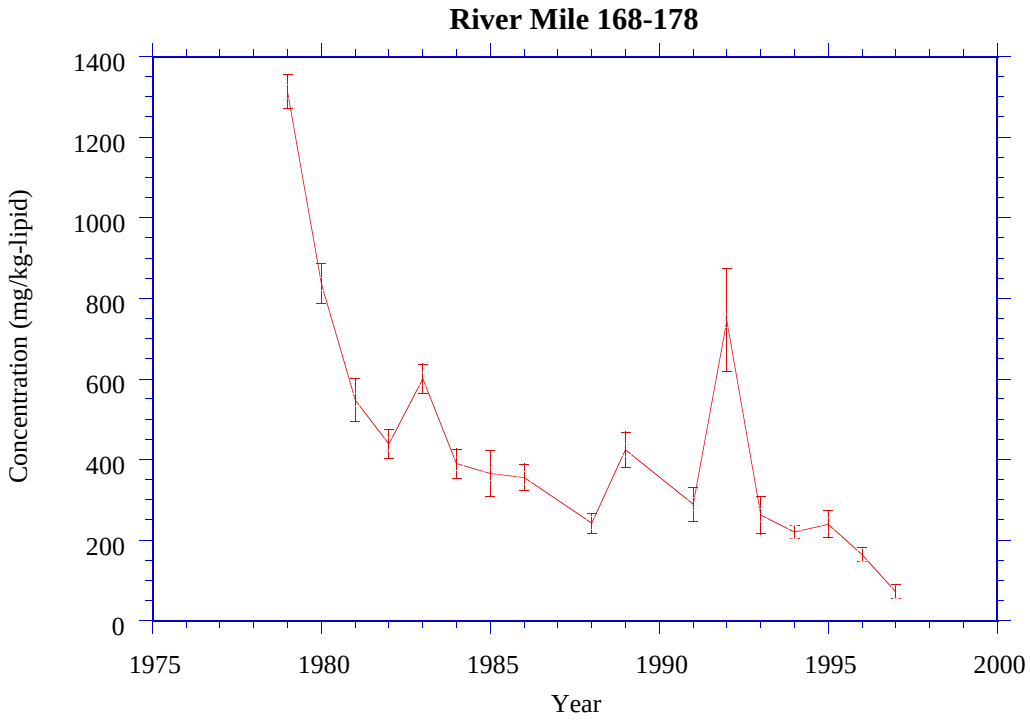
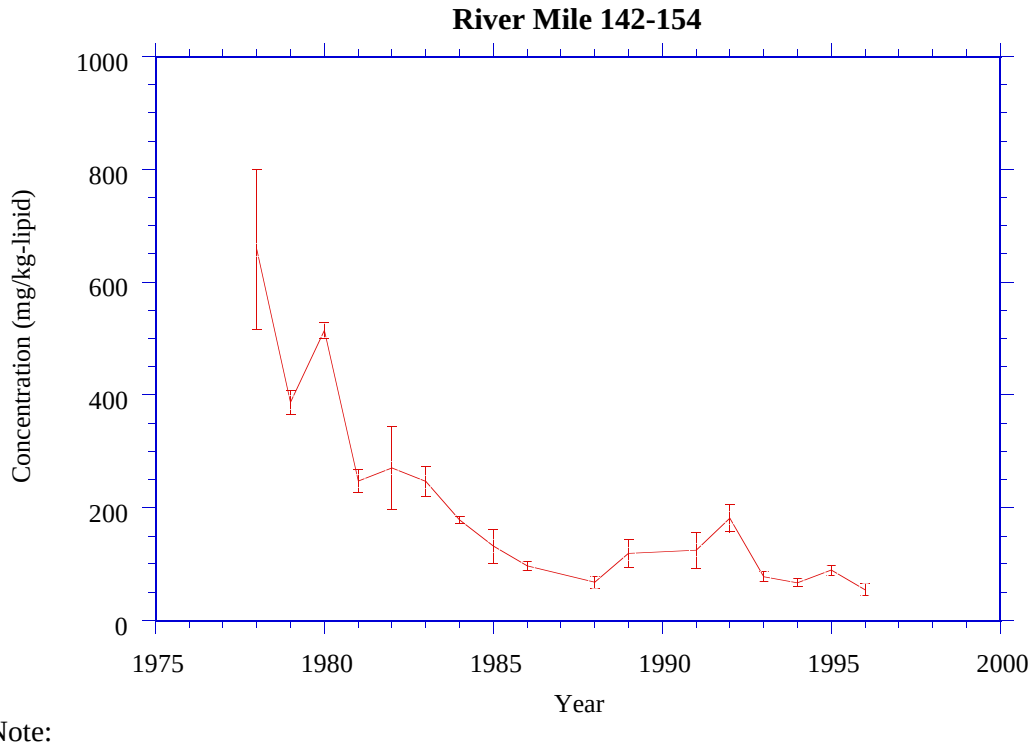


Figure K-52
Observations of Mean Summer Body Burden of Tri+ PCBs in Brown Bullhead





Note:
Error bars represent two standard errors.

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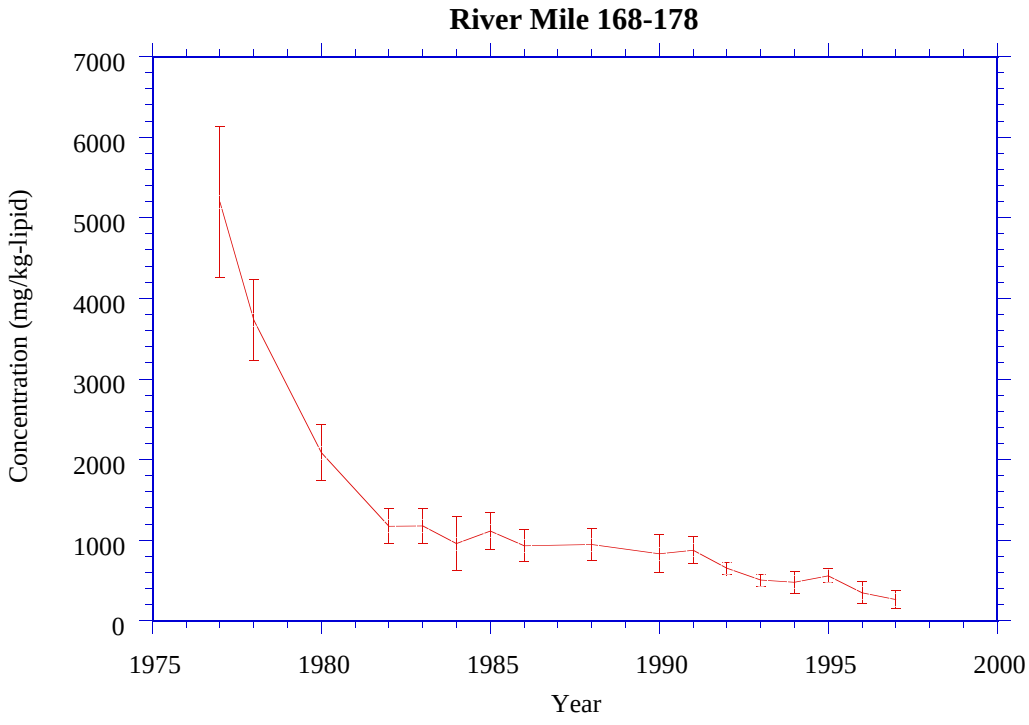
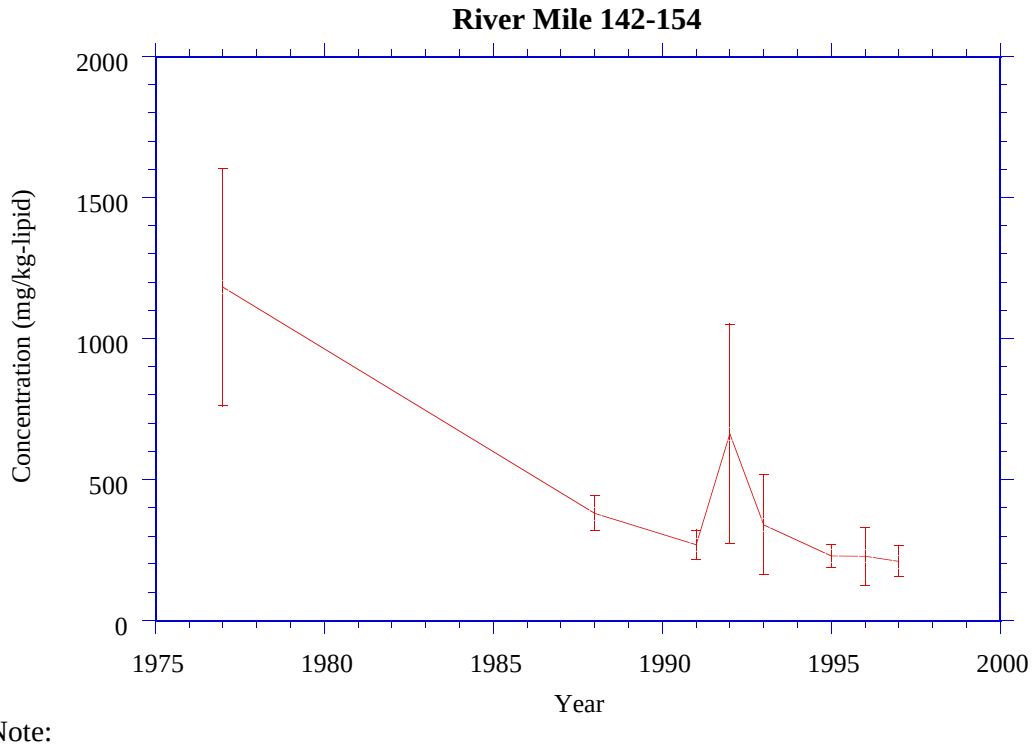
Figure K-53
Observations of Mean Summer Body Burden of Tri+ PCBs in Pumpkinseed

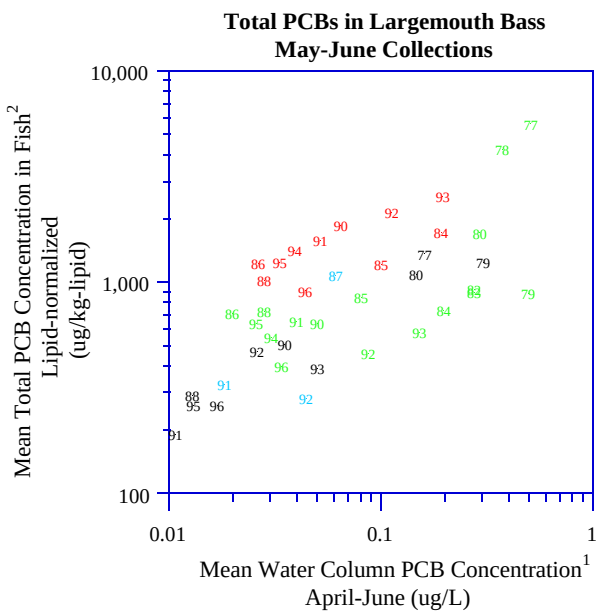


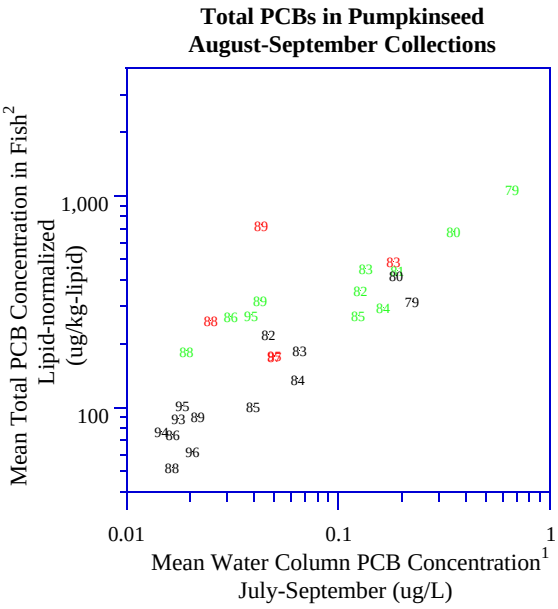


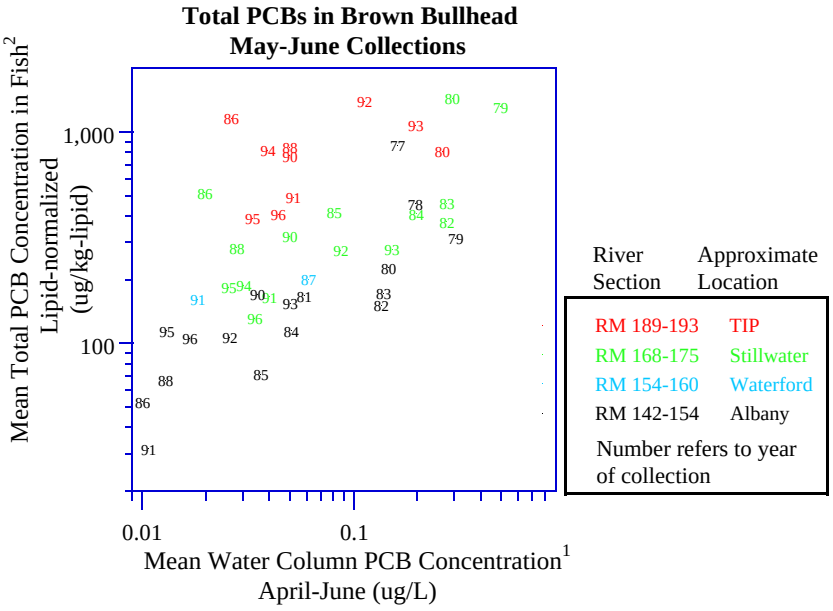
Note:
Error bars represent two standard errors.

TAMS/MCA

Figure K-54
Observations of Mean Summer Body Burden of Tri+ PCBs in Largemouth Bass



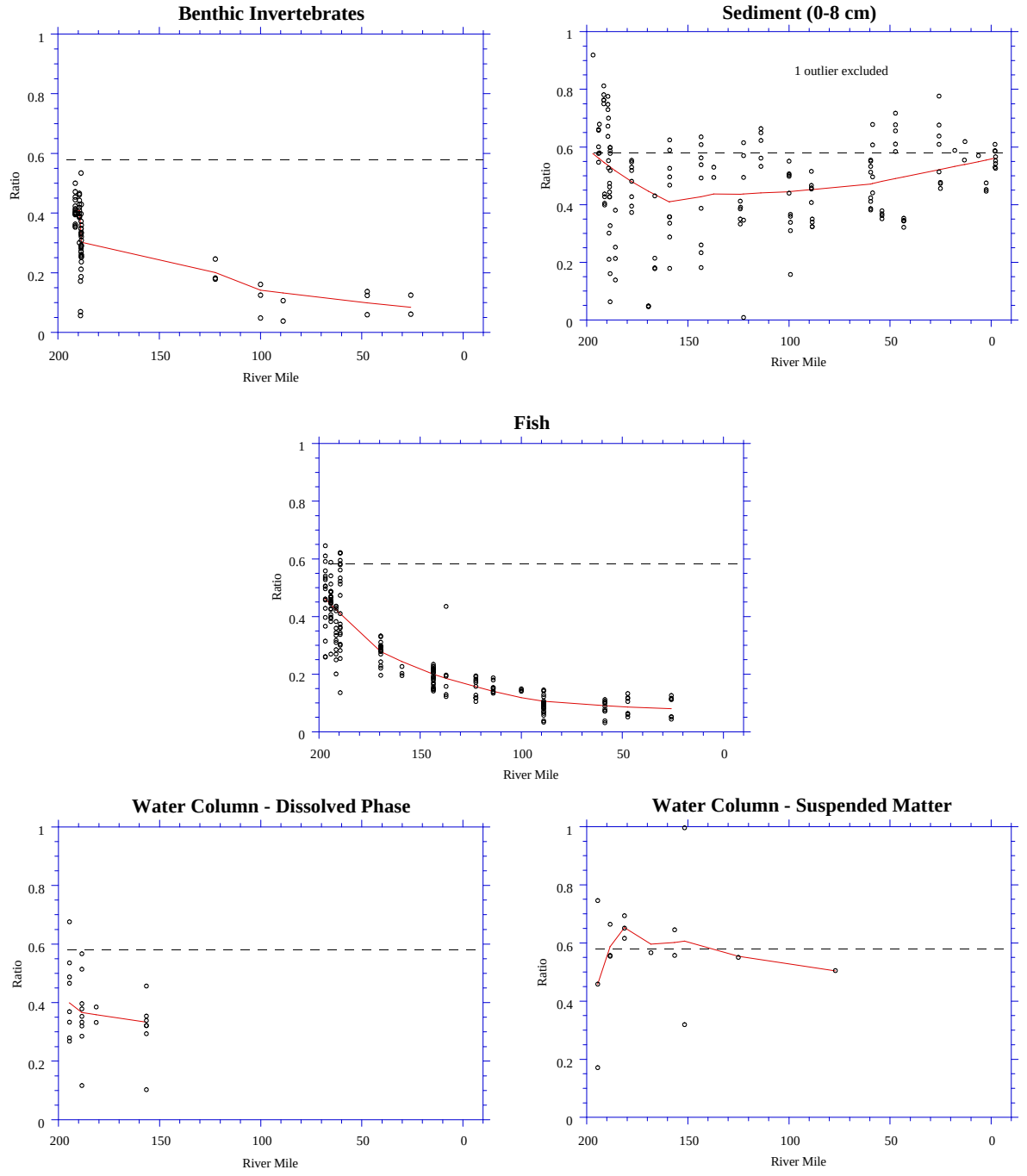




Note:
1. USGS Water Column Monitoring Data
2. NYSDEC Fish Monitoring Data

TAMS/MCA

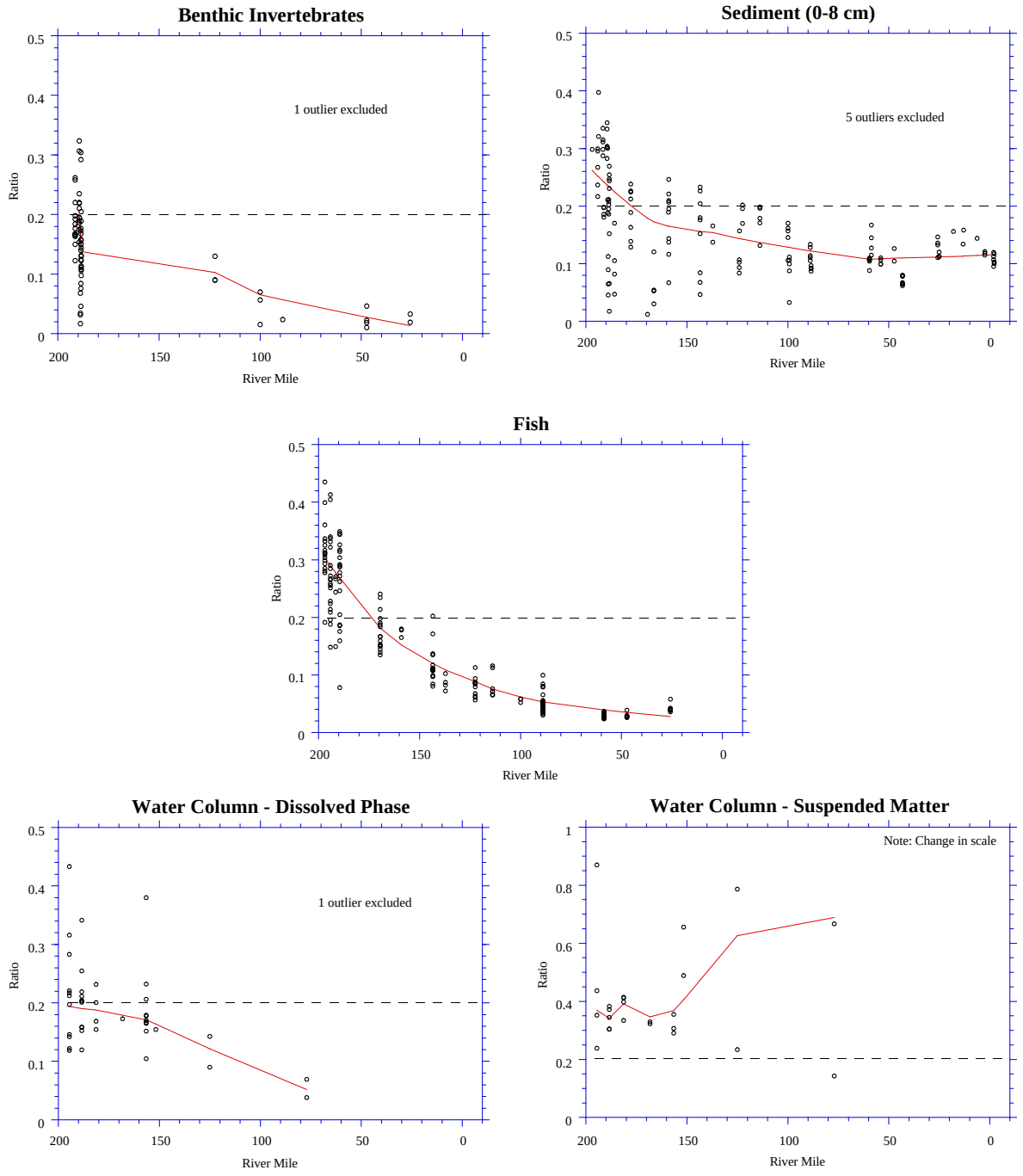
Figure K-55
Total PCB Concentration in Water vs. Lipid-normalized PCB Concentration in Fish



Note: Solid line represents weighted average trend with river mile and dashed line represents Aroclor 1242.

TAMS/MCA

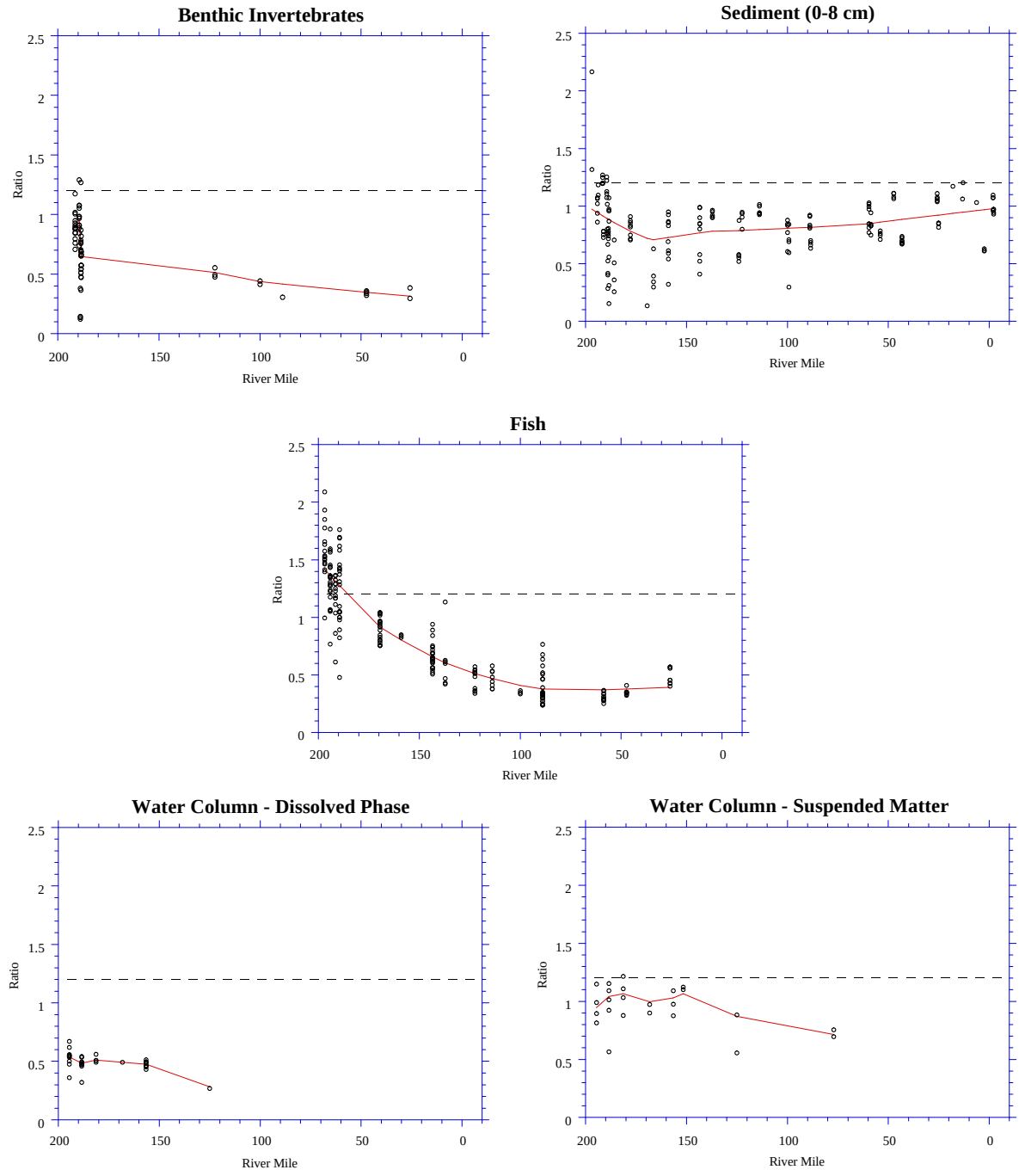
Figure K-56
A Comparison of Congener Ratio 56/49 for 1993 Hudson River Samples
1993 USEPA and NOAA Data



Note: Solid line represents weighted average trend with river mile and dashed line represents Aroclor 1242.

TAMS/MCA

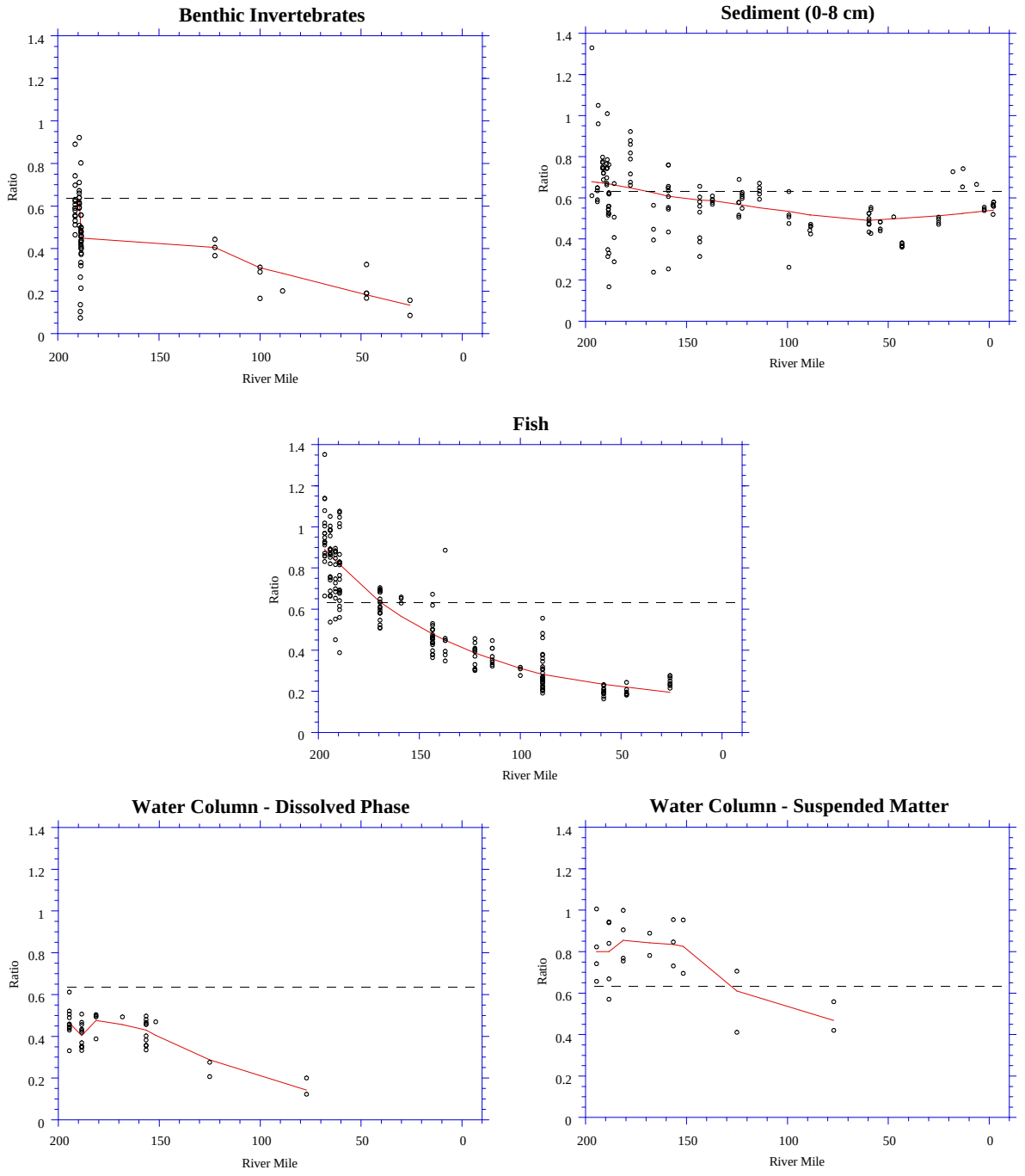
Figure K-57
A Comparison of Congener Ratio 60/49 for 1993 Hudson River Samples
1993 USEPA and NOAA Data



Note: Solid line represents weighted average trend with river mile and dashed line represents Aroclor 1242.

TAMS/MCA

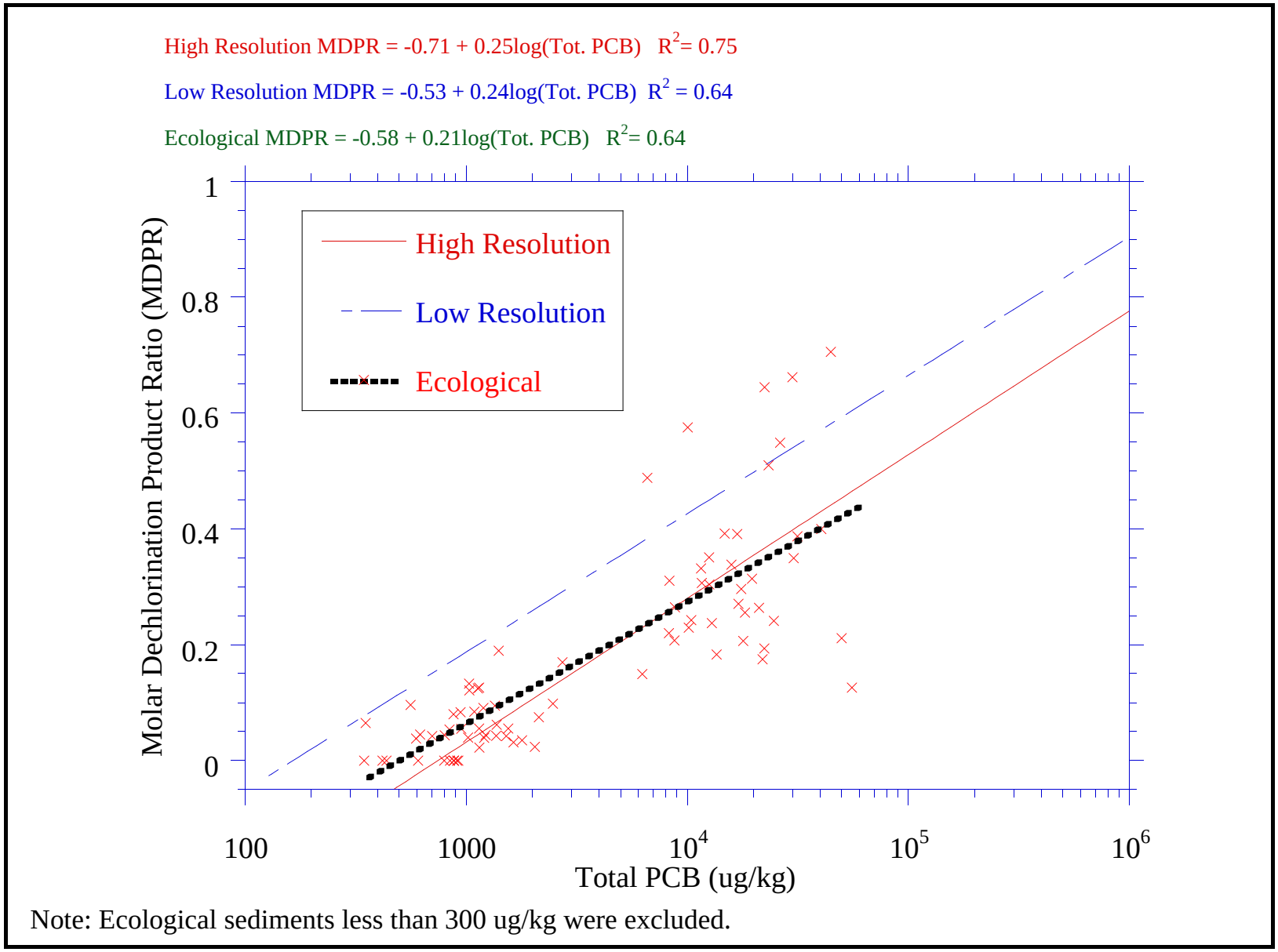
Figure K-58
A Comparison of Congener Ratio 66/49 for 1993 Hudson River Samples
1993 USEPA and NOAA Data



Note: Solid line represents weighted average trend with river mile and dashed line represents Aroclor 1242.

TAMS/MCA

Figure K-59
A Comparison of Congener Ratio 61/49 for 1993
Hudson River Samples
1993 USEPA and NOAA Data



Note: Ecological sediments less than 300 ug/kg were excluded.

TAMS/MCA

Figure K-60

Relationship Between MDPR and Total PCBs in Ecological Sediments

Comparison to High Resolution and Low Resolution Sediment Coring Results