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Research Article

Polychlorinated biphenyls, mercury, and potential endocrine disruption in fish from the Hudson River, New York, USA

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Abstract. Tissue residues of total mercury (Hg), total polychlorinated biphenyls (PCBs), and lipid-based PCBs; plasma concentrations of endocrine biomarkers; and reproductive and histologic biomarkers were assessed in 460 carp (*Cyprinus carpio*), bass (*Micropterus salmoides* and *Micropterus dolomieui*), and bullhead (*Ameiurus nebulosus*) collected from eight sites across the Hudson River Basin in the spring of 1998 to determine if endocrine disruption was evident in resident fish species and to evaluate contaminant-biomarker interrelations. Total PCBs in bed sediments (maximum 2,500 µg kg^{-1}) could explain 64 to 90 % of the variability in lipid-based PCB residues in tissues (maximum 1,250 µg PCB g-$lipid^{-1}$) of the four species. The 17β-estradiol to 11-ketotestosterone ratio, typically less than 1.0 in male fish and greater than 1.0 in females, exceeded 1.4 in all male largemouth bass and 35 % of male carp and bullhead at one site 21 km downstream from a major PCB source. Endocrine biomarkers were significantly correlated with total Hg in female smallmouth bass and carp, and with lipid-based PCBs in males of all four species. Empirical evidence of endocrine modulation in blood plasma of male and female fish from sites with and without high PCB residues in bed sediments and fish tissues suggest that PCBs, Hg, or other contaminants may disrupt normal endocrine function in fish of the Hudson River.

Key words. Endocrine disruption; Hudson River; fish; mercury; PCBs.

Introduction

A large number of contaminants affect the quality of water, suspended sediments, and bed sediments throughout parts of the Hudson River, NY (Fig. 1) and its tributaries (Sloan et al., 1984; Phillips et al., 1997). Some contaminants bioconcentrate in fish tissues and may affect their reproductive systems (Keith, 1997). Polychlorinated bi-

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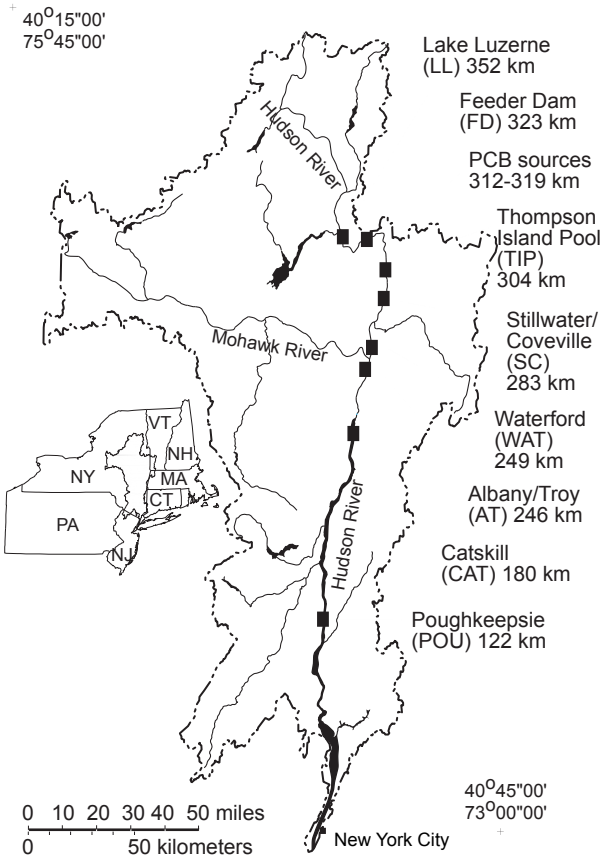

Figure 1. Locations and distance of Hudson River sampling sites (in km) from mouth at Battery Park in New York City.

phenyls (PCBs) and mercury (Hg) are of particular interest to the New York State Department of Environmental Conservation (DEC) (Sloan et al., 1984; Sloan, 2000) for several reasons. First, PCBs in fish, sediment, and water from the Hudson River far exceed baseline (10 µg g^{-1}) or background (< 1 µg g^{-1}) levels (Skinner et al., 1996; Sloan et al., 2002; 2005) and have resulted in fishery restrictions, no-fish-consumption advisories, and declaration of more than 300 km of the river as a U.S. Environmental Protection Agency Superfund site (HRTC, 1997). Second, planned dredging of bed sediments to remove PCBs is controversial and the effects of cleanup activities on PCB concentrations in fish, sediment, and waters are unknown. The effect that PCBs could have on the health of individual fish, fish populations, and entire aquatic communities in the basin is of great concern. Third, the DEC has monitored PCB residues in fish for more than 25 years, but information relating PCB concentrations in fish to potential effects on their endocrine system is limited (McDonald et al., 2000; Monosson, 2000). Fourth, mercury is a widespread human-health issue, but mercury was found to exceed safe-consumption thresholds (1,000 ng g^{-1}) only in smallmouth bass (*Micropterus dolomieui*) from the upper Hudson River (NYSDOH, 2005) and in striped bass (*Morone saxatilis*) from the New York harbor, Hudson River estuary (Skinner et al., 1996). Mercury, like PCBs, has been shown to affect the endocrine systems and reproductive success of several fish species (USEPA, 1997; Friedmann et al., 2002).

The endocrine system of vertebrates produces enzymes and hormones that are secreted into the bloodstream to regulate normal physiological processes, such as reproduction, development, and digestion (Kime, 1998). Many enzymes and hormones can be used as biomarkers to quantify endocrine function. Modulation of endocrine biomarkers in blood plasma has been correlated with environmental contaminants and potential adverse effects on reproductive capacity (USEPA, 1997; Kime, 1998; Schmitt and Dethloff, 2000). The sex-steroid hormone, 17β-estradiol (E2) regulates oogenesis in female fish and 11-ketotestosterone (11KT) controls spermatogenesis in males of many fish species (Mylonas et al., 1997; Cavaco et al., 1999; Todo et al., 1999), even though testosterone is the most important androgen in female of most fish species (Lokman et al., 2002) and some fish species have negative 11KT males (Borg, 1994). The concentrations of both hormones fluctuate widely throughout the year; E2 normally occurs at higher levels in females than in males, whereas 11KT normally occurs at higher levels in males than in females of the same species (Kime, 1998; Schmitt and Dethloff, 2000). A third biomarker, vitellogenin (VTG), is a phospholipoprotein synthesized in the liver of oviparous vertebrates when induced by elevated concentrations of E2 (USEPA, 1997; Kime, 1998). VTG is normally not detectable in plasma of male fish. Exogenous estrogenic, antiestrogenic, androgenic, or antiandrogenic compounds can directly or indirectly affect each of these three biomarkers or E2:11KT (the ratio of E2 to 11KT) in exposed fish and, thereby, cause their levels to deviate from normal. These modulations, if severe, could disrupt the endocrine systems and affect the gonad histology of individual fish, which, if pervasive, might adversely affect entire species populations.

An environmental endocrine disruptor is defined by the USEPA (1997) as an exogenous agent that interferes with the synthesis, secretion, transport, binding action, or elimination of natural hormones in the body that are responsible for the maintenance of homoeostasis, reproduction, development, and (or) behavior. Severe endocrine disruption in individuals can lead to feminization (or demasculinization) of males or masculinization (or defeminization) of females, with cascading effects within afflicted populations (USEPA, 1997). Dozens, and perhaps hundreds, of synthetic chemicals have been identified as potentially disruptive to the endocrine system of terrestrial and aquatic organisms (Keith, 1997; USEPA, 1997). The mechanisms whereby these contaminants either enhance or inhibit sex-steroid hormone or VTG

concentrations in plasma are difficult to identify because they can affect one or more locations or pathways within the hypothalamus-pituitary-gonadal-liver axis (Monosson 1999). For example, exogenous contaminants can alter plasma concentrations of E2 and VTG in male and female fish by (1) mimicking estrogen, binding to estrogen receptors, and inducing (or blocking) expression of genes (typically increasing VTG production in the liver), (2) decreasing the density or binding capacity of estrogen receptors (decreasing VTG production in the liver), or (3) by increasing synthesis (or decreasing degradation) of E2 (Thomas 1989; Monosson, 1999; 2000). Complicating issues of endocrine disruption in wild fish populations are the paucity of information on the many synthetic chemicals that may (1) act either as an estrogen, antiestrogen, androgen, or antiandrogen in male or female fish of the same species, (2) have differing effects on different species, and (3) act synergistically or antagonistically in combination with other environmental contaminants (USEPA 1997; Kime 1998).

Whether PCBs and Hg cause endocrine disruption in common fish species from the Hudson River is of interest for several reasons. The Hudson River once supported extensive commercial and sport fisheries that declined and are now in various stages of recovery (HRTC 1997). Concentrations of PCBs (and other contaminants) in bed sediments and fish from the Hudson River are among some of the highest concentrations found anywhere in North America (Goodbred et al., 1997; Phillips et al., 1997). Several studies found that PCBs (and mercury) are possible endocrine disruptors that may promote, block, or reduce androgen and estrogen activity in male and (or) female fish (Wiener and Spry 1996; Monosson, 2000; Friedmann et al., 2002). Despite a general lack of evidence to link endocrine disruption with population-level effects (Mills and Chichester, 2005), impacts on fish populations in the Hudson River would be discouraging for a system on the verge or recovery. Two recent studies actually found some evidence for endocrine modulation in common carp (*Cyprinus carpio*) and largemouth bass (*Micropterus salmoides*) from a two locations in the Hudson River (Goodbred et al., 1997; Smith and Muir 1998). Yet, the effects of PCBs and other contaminants on the endocrine systems of most fish species throughout the Hudson River are not well documented.

In 1998, the U.S. Geological Survey (USGS) and the New York State Department of Environmental Conservation (DEC) assessed potential endocrine disruption in common carp, largemouth bass, smallmouth bass, and brown bullhead (*Ameiurus nebulosus*) collected from the Hudson River (Fig. 1). Major objectives of this study were to assess endocrine modulation and the potential for endocrine disruption in four fish species at eight sites spread across the Hudson River. Specific goals were to (1) estimate the magnitude and spatial extent of endocrine modulation in selected species, (2) verify the relations between PCB residues in bed sediment and those in filets of each species, and (3) evaluate the relations of PCBs and Hg in tissues with endocrine biomarkers in blood plasma to determine if those contaminants could account for observed disruption.

Materials and methods

Target species for this study were originally common carp (carp), largemouth bass (largemouth), and brown bullhead (bullhead). Several sites lacked sufficient habitat for largemouth, therefore, smallmouth bass (smallmouth) were substituted when available. A total of 142 carp, 123 bullhead, 91 largemouth, and 94 smallmouth were collected from 1- to 15-km long reaches at eight study sites in the Hudson River from May 18 through June 26, 1998 (Fig. 1), as described below. Daytime water temperatures were near or above 18 °C at all sites during fish collections. Carp and both bass species reach gravid condition near this temperature, and all four species typically maximize spawning activity during May and June in the temperate northeastern USA (Carlander 1969; 1977).

Study area

The Hudson River (Fig. 1) flows more than 450 km southward through New York State from its headwaters in the Adirondack Mountains to its mouth in the Atlantic Ocean at New York City. Nine percent of lands in the 34,680-km^2 basin are classified as urban, industrial or residential, 24 % as agricultural, and 62 % as forested. The urban areas are concentrated within 5 to 10 km of the Mohawk and Hudson Rivers, and agricultural areas are mainly in the Mohawk River Basin (Phillips et al., 1997). The Federal Dam at river kilometer (RK) 247 in Troy, NY, divides the river into upper and lower reaches. The upper reach is entirely fresh water, and the lower Hudson is tidal and contains fresh, brackish, and saline waters. The Hudson River supports about 200 species of fish, representing more than 40 families; these include commercial species such as striped bass, American shad (*Alosa sapidissima*), and Atlantic sturgeon (*Acipenser oxyrhynchus*). Although PCB concentrations in some Hudson River species have declined from their peaks during the 1970s, PCB residues in edible tissues of many species remain above the U.S. Food and Drug Administration (FDA) criteria of 2.0 ppm (2.0 µg g^{-1}) set for fish sold in interstate commerce for human consumption, and the New York State Department of Health continues to issue fish-consumption advisories for certain species and locations (NYSDOH, 2005). The New York State body-burden PCB threshold considered safe for piscivorous wildlife, 0.1 µg g^{-1} (Newell et al., 1987), is much lower

than the FDA threshold for fish consumption by humans.

Capacitor-manufacturing facilities located near or upstream of Fort Edward (RK 315) released an estimated 589,670 kg of PCBs (primarily Aroclors 1242, 1254, 1221, and 1016) into the Hudson River between the 1940s and 1977 (HRTC, 2002). Various smaller, but appreciable point sources for PCBs have been identified on the Mohawk River at the municipal landfill at Utica, Griffiss Air Force Base at Rome, and Chicago Pneumatic at Frankfort (Armstrong and Sloan, 1988; Preddice et al., 1996).

The present study designated Lake Luzerne (LL) near RK 352 and the Feeder Dam (FD), near RK 323 as upstream "reference" sites (Fig. 1), where PCB concentrations in sediments and fish tissues were generally near or below instrument detection limits (Skinner et al., 1996; Sloan and Field, 1996; USEPA, 1999). The major source of PCBs in the upper Hudson River is near RK 317 (Hudson Falls), which is 13 km upstream from the Thompson Island Pool (TIP) site (RK 304) (USEPA, 1999). Historically, PCB concentrations were high in bed sediments and fish tissues at reaches downstream from this PCB source (USEPA, 1999); three sites in this region, TIP at RK304, Stillwater/Coveville (SC) at RK 283, and Waterford (WAT) at RK 249, were designated by this study as "impacted". Three "less impacted" sites were downstream from the confluence of the Hudson and Mohawk Rivers and the Federal Dam (RK 247); these included Albany/Troy (AT) at RK 246, Catskill (CAT) at RK 180, and Poughkeepsie (POU) at RK 122. Concentrations of PCBs in sediments and fish have generally been lower at the less-impacted sites than at sites upstream from the Federal Dam (upstream of RK 247) (Armstrong and Sloan, 1988; Preddice et al., 1996).

Fish sampling and analyses

Fish from each site were collected with a pulsed DC electroshocking boat over a 1- to 2-day period. All fish were held in live wells on the boat or at streamside, generally for less than three hours before being individually tagged and processed. Ten individual males and females of each fish species was the minimum target sample for each site. To ensure that mature fish were sampled, minimum total lengths for initial processing were set at 350 mm for carp and 250 mm for bass and bullhead (Panek, 1987; Goodbred et al., 1997). Some bullheads, shorter than 250 mm, were collected at LL and WAT where larger individuals were scarce.

Each fish was identified, tagged with a unique number, then processed by methods described in Schmitt et al. (1999) and Schmitt and Dethloff (2000). Every fish was weighed to the nearest 0.1 g, and total length was measured to the nearest millimeter. Scales used to determine carp ages were taken from above the lateral line, below the dorsal fin, and slightly anterior to the middle of the body. Bass scales were taken from below the lateral line, near the tip of the pectoral fin. One pectoral spine was collected from each bullhead for age determinations. Ages of each bass and carp were determined through techniques described by Frie (1982). Ages of bullhead were determined by techniques described in Blouin and Hall (1990). All scales were read by two individuals to confirm age determinations and to assess variability.

Blood was collected from the caudal vein with a 5-cc syringe and 18 or 21-gauge needle. Each blood sample was transferred to a vacutainer, chilled on ice, and centrifuged in the field for 10 minutes at 3,500 rpm. Clear plasma was pipetted into two 2-ml cryovials, immediately frozen on dry ice, and shipped to the University of Florida and the Florida Caribbean Science Center for biomarker analyses. All samples were stored at -80 °C until analyzed.

Necropsies were performed on each fish, whereby external and internal anomalies were noted, gonads weighed, and gonad tissue collected. Histology subsamples from the gonads from each male and female fish were fixed in the field with 10 % buffered formalin. These samples were shipped to the Leetown Science Center, Leetown, VA, for histopathology analyses. Carcasses were individually sealed in food-grade plastic bags, placed on ice in the field, and then frozen at 0 °C. Standard filets were later removed from each carcass and shipped to the ENCHEM laboratory in Madison, WI for PCB, lipid, and Hg analyses following Sloan (2000). PCB and mercury (Hg) residues and lipid content in standard filets from each fish were determined using standard New York State Department of Environmental Conservation operating procedures (Sloan, 2000). In general, each tissue sample was ground and homogenized, and concentrations of Aroclors 1248, 1254, and 1260 were measured by Soxhlet extraction, gel permeation and adsorption chromatographic fractionation, and analysis by dual capillary-column gas chromatography (GC) with electron-capture detection (ECD). Total PCB residues were estimated as the sum of Aroclors 1248, 1254, and 1260. Percent lipid in each filet was estimated from a dried aliquot of the Soxhlet-extracted tissue. Mercury concentrations were measured from aliquots of homogenized filets by cold-vapor atomic absorption spectrophotometry. Total PCB and Hg residues in filets were reported in $\mu g\ g^{-1}$ or parts per million (ppm) wet weight.

Plasma samples from all fish were analyzed for 17β-estradiol (E2) and 11-ketotestosterone (11KT) through standard radioimmunoassay (RIA) procedures (Schmitt and Dethloff, 2000). The ratio E2:11KT was evaluated as a fourth endocrine biomarker because it is generally predictive of gender and, thus, standardizes the two sex hormones to each other, the opposite sex, and different stages of maturation. Males typically have a ratio less 1.0

and females have a ratio greater than 1.0; though some overlap is likely (Bevans et al., 1996; Goodbred et al., 1997; Sepulveda et al., 2004; Spano et al., 2004). A ratio of 1.4 was selected as a conservative threshold for normal E2:11KT in male fish; higher values indicate abnormal plasma concentrations of one or both hormones. A ratio of 0.8 was selected as a conservative threshold for females; lower values indicate abnormal plasma concentrations of one or both of hormones. Vitellogenin concentrations in plasma samples were assayed and quantified by capture Enzyme-Linked Immunosorbent Assay (ELISA) (Schmitt and Dethloff, 2000). The ELISA assay used in this study resulted in a sensitivity of about 0.001 mg ml^{-1} for VTG in blood plasma. The VTG data reported herein include non-detects (zero values) that were converted to one-half the detection limit (0.0005 mg ml^{-1}) for analysis purposes.

Maturation stages for all fish were identified through standard methods described by Schmitt and Dethloff (2000). In brief, gonad aliquots were embedded in paraffin, sectioned to 5 µm, and stained with hematoxylin and eosin; individual cells were then examined under a microscope. Ovary maturation in all female fish was classified as one of six stages on the basis of size and developmental status of the oocytes as follows: (0) undeveloped, (1) early development, (2) mid-development, (3) late development, (4) late development/hydrated, or (5) postovulatory. Testes of carp and bass were classified as one of four maturity stages on the basis of size and developmental status of spermatozoa as follows: (0) undeveloped/immature, (1) early spermatogenic, (2) mid-spermatogenic, and (3) late spermatogenic (Schmitt and Dethloff, 2000). A fifth maturity stage (4) was used for male bullhead to denote a high percentage of post and prespermatogenic cells in the testes (V.S. Blazer, U.S. Geological Survey, Kearneysville, WV, personal communication); this stage may be related to a fractional reproductive strategy (repeated maturation and release of gametes over many months). The percentage of reabsorbing oocytes (atresia) in female ovaries and the presence of oocytes (intersex) in male fish were also recorded during histologic examinations.

Sediment sampling and analyses

River-bed sediments were collected at five of the study sites (CAT, AT, SC, TIP, and LL), to evaluate site-to-site differences in potential endocrine-disrupting contaminants and to evaluate the sources of variability in PCB concentrations in fish tissues. Surficial sediments were collected during low-flow conditions on October 29 and 30, 1998, from depositional areas within fish-collection reaches by standard methods (Shelton and Capel, 1994). Samples were collected from the upper 15 to 20 cm of the sediment surface with a standard Eckman dredge. Five samples were collected from different parts of each study reach, combined and thoroughly homogenized in a stainless-steel bucket, then sieved through a 2-mm stainless-steel sieve. Samples were frozen and sent to the USGS National Water Quality Laboratory (NWQL), in Denver, CO, for analyses of selected contaminants. Sediment PCB data from a 1997 sample were used for the FD site.

Total carbon, organic carbon, inorganic carbon, Aroclors 1242, 1254, and 1260, total PCBs, organochlorine insecticides, polyaromatic hydrocarbons (PAH), and other potential endocrine-disrupting compounds (e.g., 17β-estradiol) were measured in sediment samples following standard extraction, cleanup, and fractionation techniques (Foreman et al., 1995; Furlong et al., 1996). Some procedures were slightly modified by the USGS NWQL to determine estradiol and cholesterol. The general procedures were as follows: (1) parent and alkyl PAH isomers were measured by gas chromatography coupled to mass spectrometry (GC/MS) using selected ion monitoring (SIM); (2) organochlorine (OC) insecticides and Aroclors 1242, 1254, and 1260 (summed for total PCBs) were measured by gas chromatography with electron-capture detection (GC/ECD); and (3) estradiol and cholesterol were measured by full scan GC/MS of extracts used for the PAH analyses. All concentrations were reported as µg kg^{-1} (ppb) sediment, and detection limits ranged from 0.5 to 5.0 µg kg^{-1} for each analyte. Total carbon was measured by thermal conductivity (Wershaw et al., 1987); inorganic carbon was measured by colorimetric titration (Arbogast, 1990); and organic carbon was estimated by subtracting inorganic from total fractions. Carbon data were reported as g kg^{-1} of sample weight.

Data handling and analyses

Relations between (1) total PCBs in bed sediment and lipid-based PCB residues in fish tissue, (2) lipid-based PCB residues in fish tissue and endocrine biomarkers, (3) Hg residues in fish tissue and endocrine biomarkers, and (4) endocrine biomarkers and histologic biomarkers, were analyzed through correlation and regression analyses to evaluate hypotheses that:

1. PCBs in bed sediments are related to PCB residues in fish tissues,
2. PCBs cause modulation of endocrine biomarkers in fish,
3. Hg causes modulation of endocrine biomarkers in fish, and
4. altered endocrine biomarkers (E2, 11-KT, and VTG) correlate with changes in gonad histology or the GSI in one or more fish species.

A single estimate of total PCB concentration in bed sediments was determined for each site sampled in 1998 (CAT, AT, SC, TIP, and LL) and for FD sampled in 1997. Thus, relations between PCB concentrations in sediment and fish tissues were based on single PCB values in sedi-

ments from each site and multiple tissue values for each fish species across the six sites. Statistical significance was based on a $p \leq 0.05$ unless otherwise noted.

Total PCB residue data for individual fish were adjusted to percent lipid content because lipid content and PCB residues varied with species, gender, size, and age. Log_{10} transformations were applied to both the lipid-based PCB data and most biomarker data to distribute the values more normally.

One-way analysis of variance (ANOVA) tests were used to determine if mean and median Hg concentrations, lipid-based PCB residues, endocrine biomarkers (VTG, E2, 11KT and the E2:11KT), and histologic biomarkers (percent atresia and the Gonadal Somatic Index – GSI) differed among sites. Correlations and regression analyses were used to identify possible relations and to determine how well empirical models defined variation in dependent variables. All analyses were limited to fish of comparable maturation stages to minimize variability and bias that immature or spawned fish might impart on hormone-contaminant relations. Only middle-to-late, and hydrated oocyte stages (2, 3, and 4) were used for female fish of all species; early-to-late spermatogenic stages (1, 2, and 3) were used for male carp and bass; and stages 1 through 4 were used for male bullhead. Site-to-site differences in fish maturity and PCB- and Hg-tissue residues were evaluated through an ANOVA and Fisher's least significant difference (LSD) multiple range tests.

Results and discussion

The data gathered herein provides only some of the information needed to completely assess the effects that PCBs and Hg may have on the endocrine systems of selected fish species in the Hudson River. The large number of significant correlations between contaminant concentrations in tissue and endocrine biomarkers, and between endocrine and histopathology biomarkers, is noteworthy because the associations could be confounded by many factors. These factors include: (1) the wide, yet patchy spatial distribution of many potential endocrine-disrupting contaminants within the basin and within each study reach; (2) the variable exposure of individual fish to certain contaminants and differing concentrations in each reach; (3) the numerous ways that disruptors can affect the hypothalamic-pituitary-gonadal-liver axis; (4) the potential synergistic and (or) antagonistic capacity of certain contaminants (or contaminant mixtures) that can affect the endocrine system; (5) the physiological changes that different fish species undergo during asynchronous annual reproductive cycles; and (6) the large number of environmental factors (maturation cues) that affect reproductive cycles in the four fish species. Many of these factors were undefined in the Hudson River and, thus, not directly addressed. Additional study limitations and important information gaps are discussed in the last section on "study limitations and emerging issues".

PCBs and other contaminants in bed sediments

Total PCB residues in bed sediments ranged from 0 µg kg^{-1} (or 7.5 µg kg^{-1}: sum of half the detection limits for the three major Aroclor™ distributions) at LL to 2,480 µg kg^{-1} at TIP (Table 1), which is immediately downstream from the PCB source between RK 312 and 319. Concentrations of total PCBs in sediments at SC and AT were moderate (720–731 µg kg^{-1}) (Table 1). Sediment concentrations of total PCBs were not assessed at WAT and

Table 1. Concentrations of selected contaminants and carbon in composite bed sediments from five locations in the Hudson River, 1998. [Carbon concentrations are in g kg^{-1}; all others are in µg kg^{-1}. Site locations are shown in Fig. 1.]

Compound	Lake Luzerne (LL)	Thompson Island Pool (TIP)	Stillwater/ Coveville (SC)	Albany/Troy (AT)	Catskill (CAT)
p,p'– DDE	0.52	11	7.4	4.1	3.9
p,p'– DDD	< 0.5	1.8	1.8	1.8	1.8
p,p'– DDT	< 0.5	1.8	2.1	0.4	1.9
Aroclor 1242	< 5.0	1,700	365	520	150
Aroclor 1254	< 5.0	590	295	150	130
Aroclor 1260	< 5.0	190	71	50	25
Total PCBs	7.5[a]	2,480	731	720	305
Total PAHs	840	2,852	212	3,732	2,652
Total organochlorines (excluding PCBs)	112	151	171	185	152
ρ-cresol	100	7,800	6,500	580	410
Caffeine	< 0.080	< 0.080	< 0.080	< 0.080	< 0.080
17β-estradiol	< 0.500	< 0.500	< 0.500	< 0.500	< 0.500
Cholesterol	640	< 1.000	< 1.000	580	30
Inorganic carbon	< 200	400	1,300	3,900	1,500
Organic carbon	29,000	74,000	58,000	14,000	26,000
Total carbon	29,000	74,400	59,300	17,900	27,500

[a] Sum of ½ detection limits for the three Aroclors.

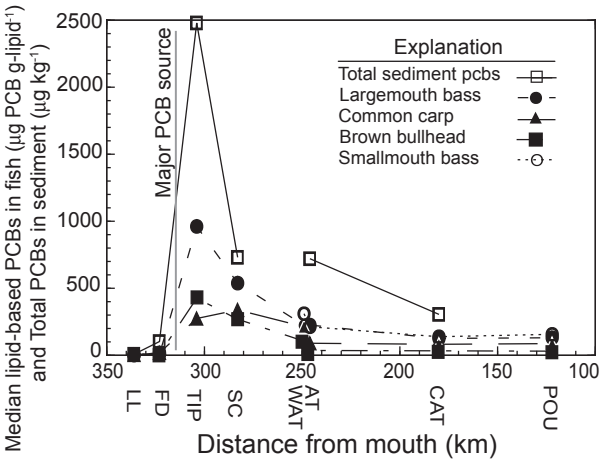

Figure 2. Total PCB concentrations in bed sediment from six sites, and median lipid-normal PCB residues in tissues of four fish species (sexes combined) at eight sites in the Hudson River, 1998. [The total sediment PCB value at FD (RK 323) was obtained from a 1997 NYSDEC sample. Site locations are shown in Fig. 1.]

POU in 1997 or 1998, but were 1,700 and 730 µg kg^{-1} in 1993 (Phillips et al., 1997). Total PCB residues in sediments at CAT were relatively low (305 µg kg^{-1}) in 1998 (Table 1), as were those at FD (102 µg kg^{-1}) in 1997 (Fig. 2).

Other contaminants, such as DDT, DDE, ρ-cresol (a phenol), and total PAHs, were detected in bed sediments at three sites (Table 1). Concentrations of ρ-cresol (synonyms: 4-cresol, 4-methylphenol, 1-hydroxy-4-methylbenzene), a byproduct of automobile emissions, coal-tar refining, wood pulping, and manufacturing and refining of metals (Howard, 1989), were extremely high (from 6,500 to 7,800 µg kg^{-1}) at SC and TIP. The highest total PAH concentrations (2,852 µg kg^{-1}) were measured in sediments from AT. Except for cholesterol, concentrations of constituents that are indicative of municipal wastewater inputs were generally not detected at any site. Though cholesterol may be derived from a variety of sources (Seguel et al., 2001), concentrations from 580 to 640 µg kg^{-1} in sediments at LL and AT, indicate these reaches might be affected by municipal wastewaters. Low or non-detectable concentrations of estradiol (<0.5 µg kg^{-1}) in sediments at all five sites, however, suggest that sewage effluents are well treated and (or) diluted, thus, associated estrogenic compounds may also be below concentrations capable of affecting the endocrine systems of resident fish.

PCB residues in fish tissues

Mean concentrations of lipid-based PCBs in standard filets of male and female fish (genders pooled) ranged from 3 to 315 µg g^{-1} in smallmouth (Fig. 3A), from 5 to 350 µg g^{-1} in carp (Fig. 3B), from 5 to 400 µg g^{-1} in bullhead (Fig. 3C), and from 2 to 1,200 µg g^{-1} in largemouth (Fig. 3D). Median lipid-based PCB residues segregated by gender (Appendix A) correspond closely to mean values pooled by gender (Fig. 3). Both Analysis of Variance (ANOVA) and Kruskal-Wallis tests identified no significant differences between PCB residues in tissues of males and those of females of the same species, but showed significant differences among species and sites. These differences are reflected in the mean and 95% LSD confidence intervals (Fig. 3), which indicate that lipid-based PCBs were consistently lowest at the reference sites (LL and FD) and highest at sites downstream from PCB sources (TIP and SC) for each species. PCB residues in fish generally decreased at the four sites downstream from SC (WAT, AT, CAT, and POU) (Fig. 3).

Findings from the recent literature indicate that current concentrations of PCBs in Hudson River largemouth, carp, and bullhead may be high enough to affect their reproductive systems. PCBs have been linked to changes in metabolism of sex-steroids, growth and development of testes and ovaries, and production of gonadotropins and VTG in mature individuals of some fish species (Monosson, 1999). Threshold concentrations of Aroclor 1254 in livers of adult fish that generally affect their reproductive systems range from 25 to 100 µg g^{-1} (Monosson, 1999; 2000). Concentrations of Aroclor 1254 (and highly chlorinated 1260) in fish livers were not determined by the present study, but could be estimated from total PCB residues measured in filets and liver-filet ratios derived from data in two earlier investigations. Concentrations of total PCBs and Aroclor 1242, 1248, 1254, and 1260 were determined in livers and standard filets (or whole bodies for small fish) from 70 largemouth, smallmouth, bullhead, yellow perch (*Perca flavescens*), white perch (*Morone americana*), and striped bass collected in 1999 from the Hudson River (Sloan et al., 2002). In that study, large variations were evident among species and sites, but the ratio of highly chlorinated Aroclors (1254 and 1260) to total PCBs in standard filets averaged 0.5, and the ratio of total PCBs in liver to that in standard filets averaged 1.9 in all six species (Sloan et al., 2002). These ratios were comparable to those from Monosson (2000); the ratio of Aroclor 1254 to total PCBs in standard filets averaged 0.6, and total PCB residues in livers were twice that in filets. The median total PCB residues in filets of male largemouth, female bullhead, and male and female carp at TIP (21.6, 22.2, 23.8, and 21.8 µg g^{-1}, respectively), and male and female carp at SC (40.1 and 61.0 µg g^{-1}) multiplied by 0.95 or 1.2 (using both ratios from each report), indicate that Aroclor 1254 and 1260 concentrations in livers of the three species may approach or exceed the reproductive-effect threshold of 25 µg g^{-1} in about half of the fish. This may potentially affect reproductive health of the three species at both sites.

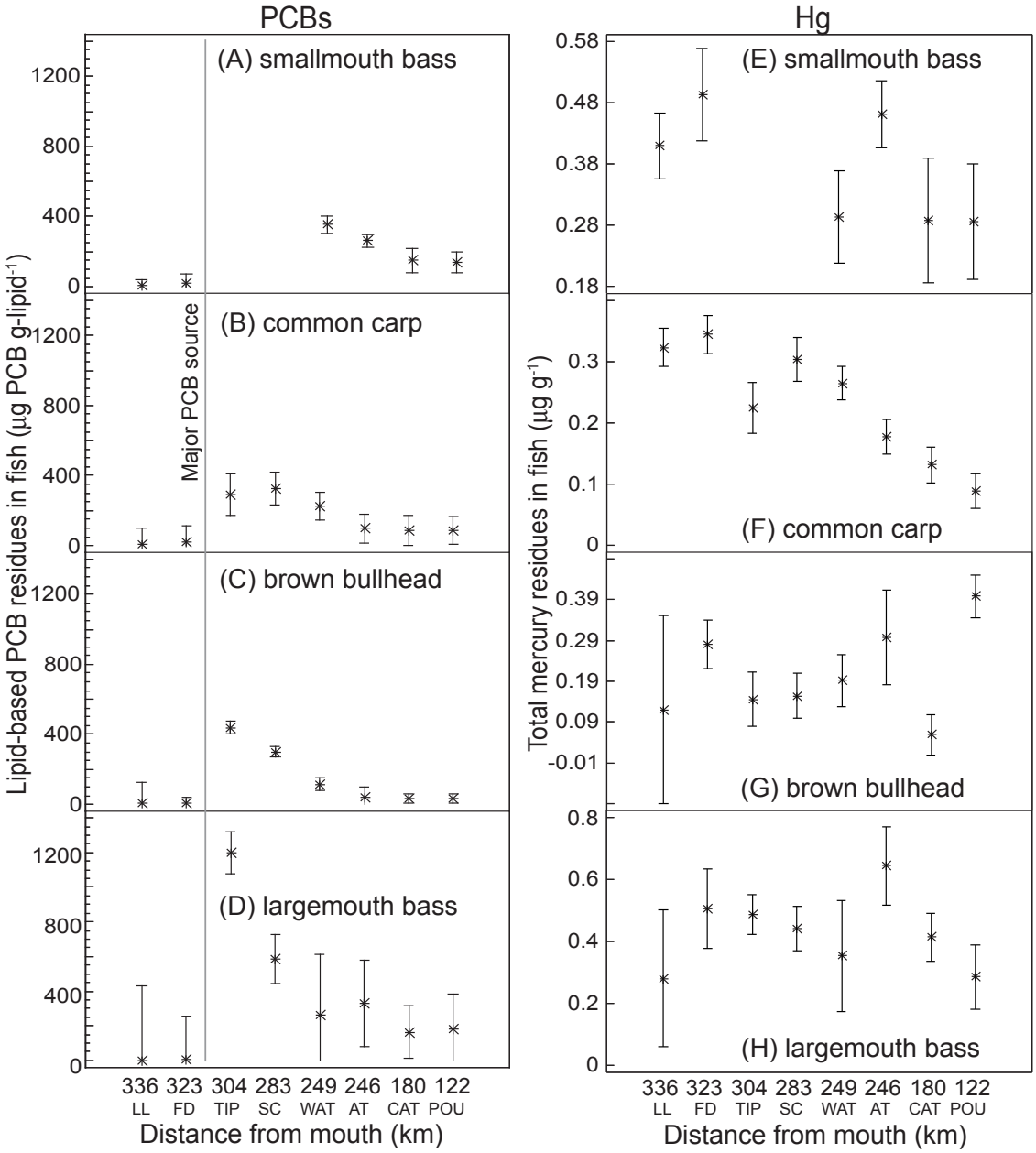

Figure 3. Mean (*) and 95 % least-standard-deviation confidence intervals for lipid-based PCB (left) and total Hg (right) residues in tissues of smallmouth bass (A, E), common carp (B, F), brown bullhead (C, G), and largemouth bass (D, H) at eight study sites in the Hudson River, 1998. [Smallmouth bass were not collected at TIP (RK 304) or SC (RK 283) sites. Site locations are shown in Fig. 1.]

Relations between PCBs in fish tissues and PCBs in bed sediment

Lipid-based PCB residues in fish tissue closely parallel and were significantly correlated with total PCB concentrations in bed sediments at the six sites (Table 2, Fig. 2). Total PCB concentrations in sediments could explain from 64 to 90 % of the variation in lipid-based PCBs in tissues of male and female fish (Table 2, Fig. 4). Total PCB concentrations in bed sediments were generally more strongly correlated with lipid-based PCB residues in tissues of female fish than in male fish (Table 2). Though dissimilar concentrations of PCBs have been found in filets of male and female striped bass and other species (Sloan et al., 2002; 2005), there is little information to assess the underlying behavioral or physiological reason for gender differences.

The strong correlations of PCBs in bed sediments with PCBs in bass, carp and bullhead at six Hudson River sites confirm that those species accumulate PCBs in proportion to concentrations in their immediate surround-

Table 2. Regression coefficients (R^2) for the relations of lipid-based PCB residues in tissues from males and females of four fish species with total PCB concentration in bed sediment at six sites in Hudson River, 1998. [All relations are significant at $p < 0.05$. n, total number of observations.]

Dependent variable	Log_{10} total PCB concentration in bed sediments							
	Smallmouth bass		Common carp		Brown bullhead		Largemouth bass	
	Male	Female	Male	Female	Male	Female	Male	Female
Log_{10} lipid-based PCBs in fish tissue	0.88 (n=41)	0.88 (n=30)	0.64 (n=59)	0.77 (n=51)	0.76 (n=29)	0.90 (n=54)	0.69 (n=38)	0.86 (n=39)

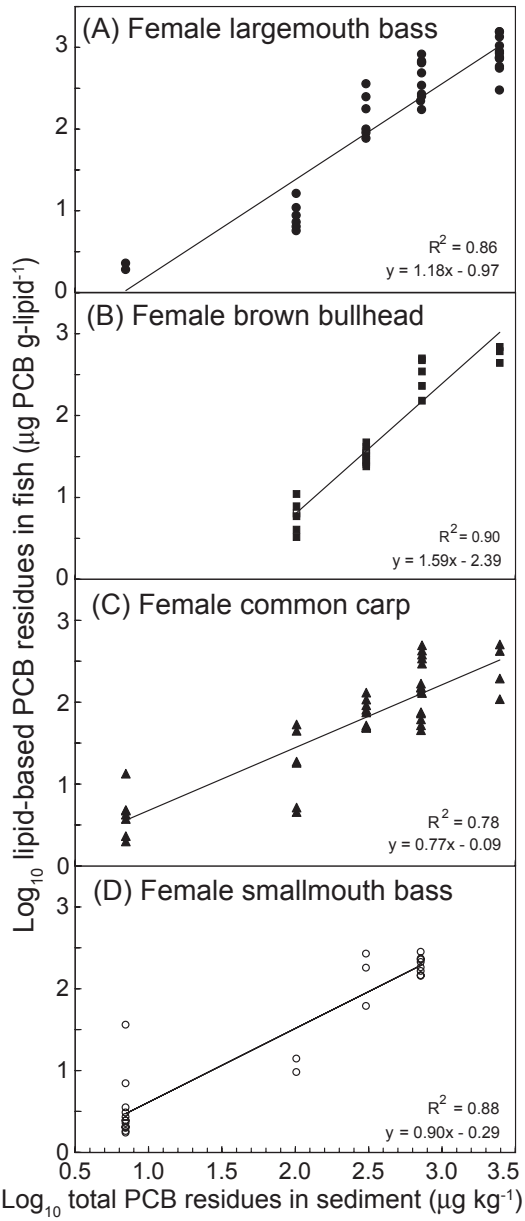

Figure 4. Log_{10} lipid-based PCB residues, as a function of log_{10}-transformed total PCBs in bed sediments, in tissues of female largemouth bass (A), brown bullhead (B), common carp (C), and smallmouth bass (D). [Sediment PCB concentrations from sites SC and AT overlap (2.86 mg PCB kg) so they appear as one site.]

ings. The highest PCB concentrations in fish tissues and bed sediments were found just downstream from the major PCB source areas (sample sites at RK 312 and 319); PCB residues in sediments and fish tissues were lower at other sites located further downstream. Several locks and dams in the upper Hudson River (upstream from RK 247) ensure that most fish collected at each site had been subjected primarily to conditions within the corresponding study reach even though PCB exposures could be highly variable within each reach. Strong associations between PCB residues in water, sediment, and fish from the Hudson River have also been observed in other studies (Sloan et al., 1984; Sloan et al., 2002; 2005; Parsons, 2006), and leave little doubt that PCB residues in fish accumulate from their contaminated environment.

Mercury in fish tissues

Mean total mercury (Hg) concentrations in tissues of smallmouth (pooled males and females) ranged from approximately 0.28 µg g^{-1} at WAT, CAT, and POU to 0.47 µg g^{-1} at FD, and were significantly higher at FD and AT (0.49 µg g^{-1}) than at WAT and the two downstream sites (CAT and POU) (Fig. 3E, Appendix A). Mean total Hg residues in tissues of largemouth at all sites, except AT ranged from 0.28 to 0.50 µg g^{-1}; concentrations in largemouth from AT were significantly higher (0.64 µg g^{-1}) than at most other sites (Fig. 3H). Mean Hg residues in carp decreased significantly between the furthest upstream sites (except at TIP) and the downstream sites; tissue residues at LL and FD averaged from 0.26 and 0.34 µg g^{-1} and at the three downstream sites (AT, CAT, and POU), they averaged 0.09 to 0.18 µg g^{-1}. Total Hg residues in carp at TIP were lower (0.22 µg g^{-1}) than at the two upstream reference sites (Fig. 3F). Mean Hg residues in bullhead ranged from 0.12 and 0.40 µg g^{-1} and, except for two sites (FD and CAT), increased between upstream and the downstream sites (Fig. 3G). Mercury residues, however, were significantly higher in bullhead at FD (0.28 µg g^{-1}) than at TIP, SC, and CAT, and significantly lower at CAT (0.06 µg g^{-1}) than at most other sites (Fig. 3G).

Total Hg concentrations in filets of both bass species and carp generally showed similar patterns with maximum values at FD or LL and at AT, and decreasing with downstream distance. The patterns of Hg-tissue concentrations for bullhead were partly reversed; like both bass species, tissue-Hg concentrations in bullhead were higher at FD and AT than at most downstream sites, but unlike the bass pattern, the highest Hg residues were at POU and increased with distance upstream. Mercury residues were typically higher in female than in male fish of the same species except at POU where the order was reversed (Appendix A). Mercury residues in bed sediment were not analyzed, thus, relations between sediment and tissue concentrations could not be assessed.

The low to intermediate concentrations of total Hg in fish filets have some important implications. First, median Hg residues in all four fish species approach or exceed the recently lowered federal criterion of 0.30 µg Hg g^{-1} wet weight for human health at one or more sites in the basin (USEPA, 2001) (Fig. 3, Appendix A). The median Hg tissue residues for largemouth, smallmouth, and bullhead also exceed the recently superseded threshold of 0.50 µg Hg g^{-1} wet weight at several sites. These findings indicate that Hg concentrations in these species may be an important human health issue. Second, Hg concentrations in filets fall one to two orders of magnitude below the levels that had been found to produce acute or chronic toxicity (Wiener and Spry, 1996); thus, no fish mortality is anticipated. Third, Hg concentrations in filets from Hudson River fish were similar to, or somewhat lower than, the concentrations associated with increased 11-KT blood-serum concentrations in male largemouth from three reservoirs in New Jersey (Friedmann et al., 2002). Female largemouth were not sampled in that study, but their data suggest that Hg could have androgenic effects on Hudson River black bass of both sexes.

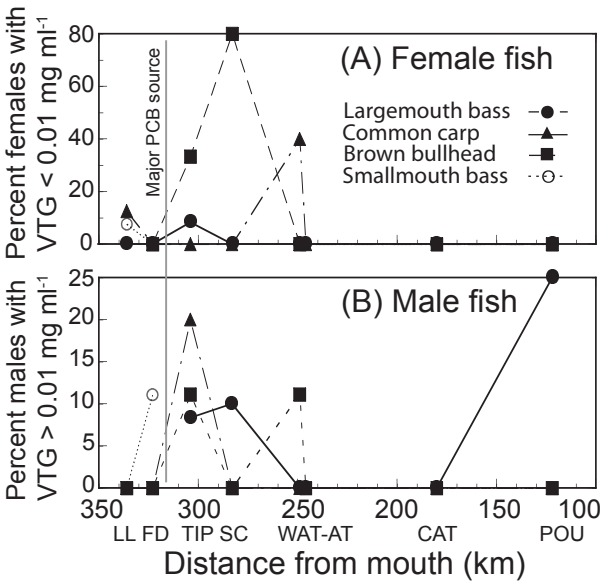

Figure 5. Percentage of (A) female fish with less than 0.01 mg ml^{-1} vitellogenin concentration in plasma and (B) male fish with greater than 0.01 mg ml^{-1} vitellogenin concentration in plasma, at eight sites in the Hudson River, 1998. [Site locations are shown in Fig. 1.]

Vitellogenin in plasma

Median concentrations of VTG in plasma ranged from 0.06 to 7.3 mg ml^{-1} in mature female carp, bass, and bullhead at the eight study sites (Appendix A). Vitellogenin concentrations lower than 0.01 mg ml^{-1} were found in 35 to 80 % of female bullhead at TIP and SC and in 40 % of female carp at WAT (Fig. 5). The median VTG concentration in males for all species was 0 mg ml^{-1} (Appendix A), but concentrations of 0.01 mg ml^{-1} (5 fold higher than background levels of 0.005 mg ml^{-1}) or higher were detected in plasma from 20 % of male carp at TIP and from 25 % of male largemouth at POU (Fig. 5). With several exceptions, site-to-site differences in maturation stage were not significant (Fig. 6). Maturity stages for male carp at TIP ranged from 1 to 2 and were significantly lower than at all other sites where stages ranged from 2 to 3 (Fig. 6). Maturity stages of male largemouth at POU were comparable to those at TIP, but significantly lower than those from AT and moderately lower than largemouth at most other sites (Fig. 6).

Low concentrations of VTG are possible in female fish at different maturation stages, however, detectable levels of VTG in plasma of male fish indicate their endocrine system could be disturbed (Folmar et al., 1996). Vitellogenin is a precursor protein that contributes to egg development in female ovaries, and is synthesized by the liver in response to elevated plasma concentrations of the sex steroid, 17β-estradiol, or in response to other estrogenic compounds or mimics. Vitellogenin concentrations in plasma of female fish are generally highest near the time of egg maturation and, thus, can fluctuate widely throughout the year (Schmitt and Dethloff, 2000). Pre- or post-spawn condition could not account for the low VTG concentrations in plasma of female bullhead and carp at SC, TIP, and WAT because their maturity stages were similar to those observed at most other sites even where VTG concentrations were high (Fig. 6). The low VTG concentrations in plasma of female carp and bullhead suggest that the two species may have been exposed to antiestrogens.

Male fish possess the gene for production of VTG in their liver but typically they do not produce the VTG protein (USEPA, 1997). This gene can be expressed (and VTG can be synthesized), however, when male fish are exposed to exogenous or endogenous estrogens or mimics (USEPA, 1997). Smith et al. (2002) suggests that very low concentrations (<0.01 mg ml^{-1}) of VTG in male largemouth might be normal, regardless of the presence

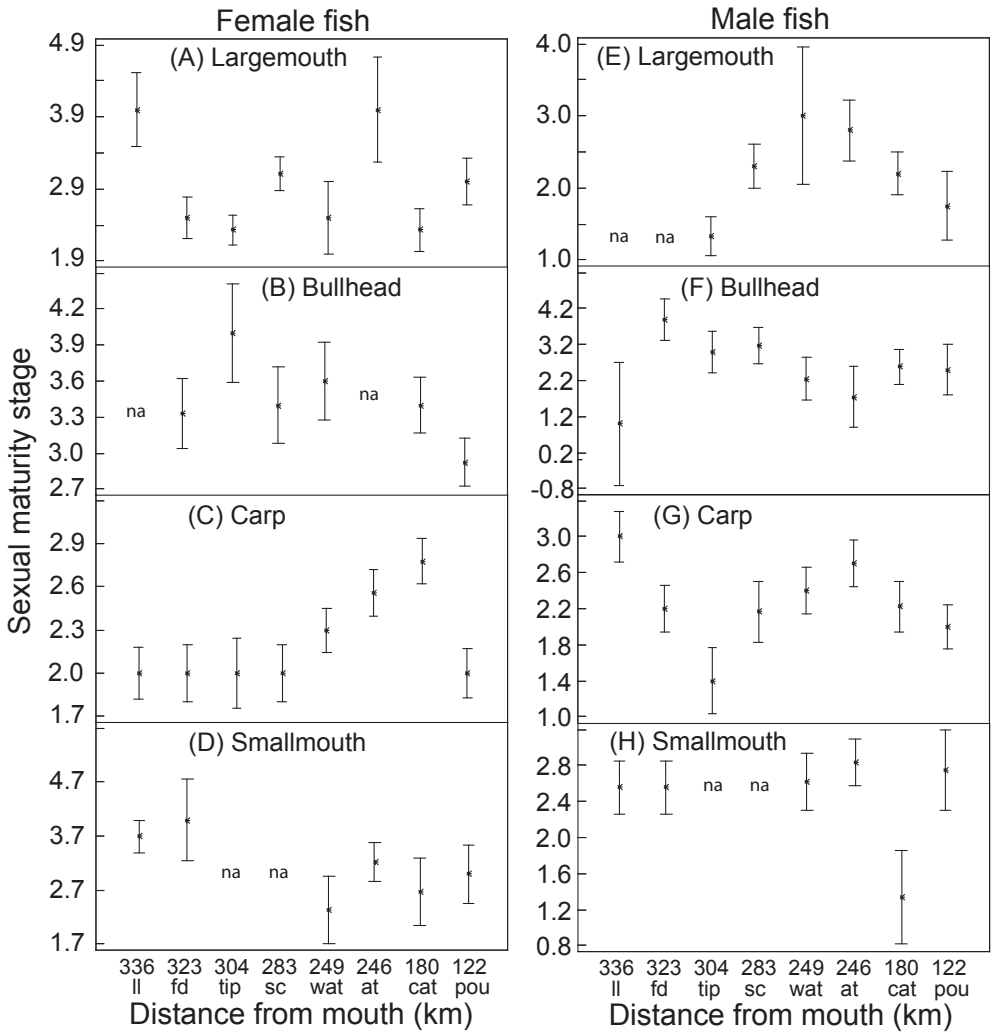

Figure 6. Mean maturation stage and 95 % least-standard-deviation confidence intervals for male and female fish from eight sites in the Hudson River that were used for analysis of biomarker and contaminant relations. [Site locations are shown in Fig. 1.]

or concentration of exogenous contaminants. The significance of VTG in males, thus, is complicated and needs additional study (Schmitt and Dethloff, 2000). The predominance of early maturity stages in male largemouth at POU and TIP and in male carp at TIP, and detection of VTG in moderate numbers of males of both species at these sites, indicate that males are being exposed to estrogenic contaminants. Whether the source of estrogenicity is PCBs, Hg, unusual thermal regimes or other localized contaminants, detectable VTG in male largemouth and carp indicate that their endocrine systems may be adversely affected in parts of the basin.

17β-estradiol in plasma

Median concentrations of 17β-estradiol (E2) ranged from 158 to 3,178 pg ml^{-1} in plasma of mature female carp, bass, and bullhead, and from 68 to 690 pg ml^{-1} in mature male fish at all eight sites (Appendix A). The E2 concentrations were typically lower in male fish than in female fish of the same species at most sites, however, median concentrations were higher in male largemouth than in female largemouth at SC and AT, and in smallmouth at the FD and CAT sites (Appendix A). Like VTG, male fish typically have no E2 in their plasma or their concentrations are much lower than in female fish of the same species at a given time and location (Kime, 1998; Schmitt and Dethloff, 2000). Nevertheless, the low concentrations of E2 in males can sometimes exceed those in females because E2 concentrations in females fluctuate widely between reproductive and nonreproductive seasons. Higher E2 concentrations in largemouth and smallmouth males than in females suggest that males may have been exposed to an estrogen, and (or) females were exposed to an antiestrogen, at some sites.

11-ketotestosterone in plasma

Median concentrations of 11-ketotestosterone (11KT) ranged from 125 to 1,027 pg ml^{-1} in mature females of all species and from 192 to 3,246 pg ml^{-1} in mature males from the eight sites (Appendix A). Mean and median 11KT concentrations were generally lower in female fish than in male fish of the same species. Concentrations of 11KT in female largemouth, however, were higher than in male largemouth at LL, SC, AT, and POU (Appendix A). Concentrations of 11KT were also higher in female smallmouth at FD and in female bullhead at TIP than in males from the respective sites (Appendix A). The concentrations of 11KT are often the opposite of E2 in each gender – 11KT concentrations are generally lower in plasma of female fish than in male fish of the same species at a given location (Kime, 1998; Schmitt and Dethloff, 2000). High 11KT concentrations in plasma of female fish and low 11KT concentrations in plasma of male fish indicate males were possibly exposed to an antiandrogen, and (or) female fish were exposed to androgens, at some sites.

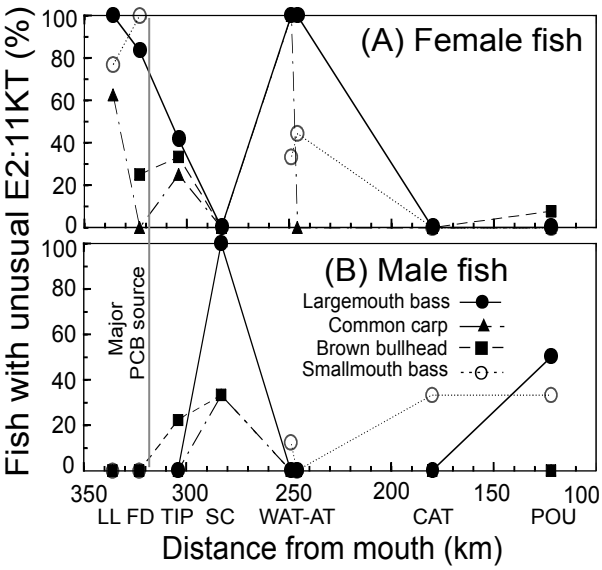

Figure 8. Percentage of mature (A) female fish with E2:11KT less than 0.8 and (B) male fish with E2:11KT greater than 1.4 at eight sites in the Hudson River, 1998. [Site locations are shown in Fig. 1.]

E2:11KT in plasma

The E2:11KT ratios were generally greater than 1 for female fish at most sites on the Hudson River (Fig. 7, Appendix A). The median E2:11KT was, however, less than 0.8 for female carp, smallmouth, and largemouth at LL, for female smallmouth and largemouth at FD, and for female carp, bullhead, and largemouth at WAT and AT (Fig. 7). The ratio was less than 0.8 for nearly 65 % of female carp at LL and between 35 and 100 % of female largemouth and smallmouth at LL, FD, WAT, and AT (Fig. 8). These results generally reflect 11KT concentrations in females that approach or exceed the 11KT concentrations in male fish (Appendix A) and indicate that female fish are exposed to, and are responding to, an androgen at some locations in the river. The low E2:11KT for female largemouth, carp, and smallmouth at LL results primarily from elevated 11KT concentrations, which suggests LL may not be the best control site for gauging the effects of PCBs, Hg, and other contaminants on sex-steroid hormones in fish across the basin.

The E2:11KT ratios were also generally less than 1 in male fish at all sites, but the median ratio exceeded 1.4 for males of all species at one or more sites (Fig. 7, Appendix A). The ratio exceeded 1.4 in 100 % of male largemouth and in 35 % of male carp and bullhead at SC, where the ratio was higher in largemouth males (3.5) than in females (2.3) (Fig. 8). About 50 % of male largemouth at POU, and 35 % of male smallmouth at CAT, also had an E2:11KT greater than 1.4 (Fig. 8). The high median E2:11KT for male largemouth at SC reflects low concentrations of 11KT and high concentrations of E2 in plasma of male fish. These findings indicate that the endocrine systems of male largemouth at SC (and other sites), as well as male carp and bullhead at SC were exposed to an estrogen and (or) antiandrogen.

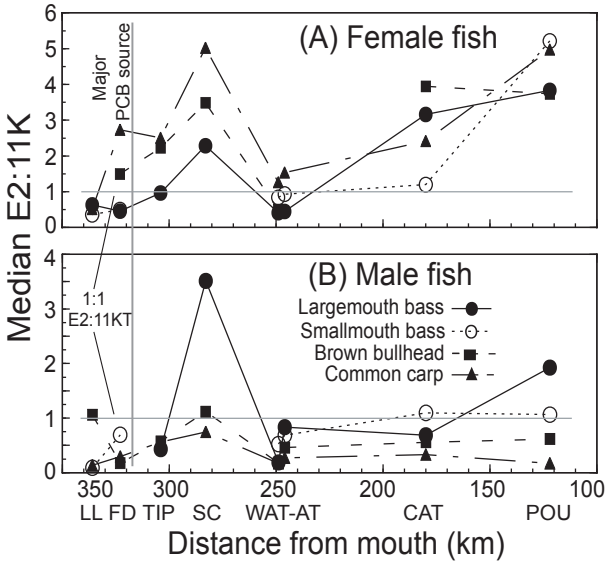

Figure 7. Median E2:11KT values in mature (A) female and (B) male fish collected at eight sites in the Hudson River, 1998. [Site locations are shown in Fig. 1.]

Relations of endocrine biomarkers to total Hg residues in fish

Vitellogenin concentrations were inversely correlated with total Hg tissue residues in male carp (Table 3A). Low VTG levels in male carp might not negatively affect the reproductive ability of individuals in Hg-contaminated reaches because VTG has no critical function in

Table 3. Correlation (r) of biomarkers in plasma with (A) total Hg and (B) lipid-based PCB residues in tissues from individual male and female fish from six to eight sites in the Hudson River, 1998. [Coefficients of correlation are significant to $p < 0.05$. Boldface values indicate correlation with potential adverse effects on reproductive health of the species or gender. Approximate number of biomarkers assessed for each species and gender are given in Appendix A. na = not applicable; ns = not significant.]

Dependent variable	Smallmouth bass		Common carp		Brown bullhead		Largemouth bass	
	Male	Female	Male	Female	Male	Female	Male	Female
A. Mercury concentrations in fish tissues								
Log_{10} 17β-estradiol	ns	ns	ns	ns	ns	ns	ns	ns
Log_{10} 11-ketotestosterone	ns	**0.39**	ns	**0.40**	ns	ns	ns	ns
Log_{10} (E2:11KT ratio+1)	ns	**–0.33**	ns	**–0.40**	ns	ns	ns	ns
Log_{10} vitellogenin	ns	**–0.39**	–0.79[a]	ns	na	ns	ns	ns
B. Log_{10} lipid-based PCB concentrations in fish tissues								
Log_{10} 17β-estradiol	**0.51**	ns	ns	ns	ns	ns	**0.31**	ns
Log_{10} 11-ketotestosterone	**–0.48**	ns	**–0.32**	–0.42	ns	ns	0.32	–0.38
Log_{10} (E2:11KT ratio+1)	**0.56**	0.33	**0.25**	0.38	**0.36**	ns	ns	ns
Log_{10} vitellogenin	ns	ns	**0.62**[a]	ns	na	ns	**0.54**	0.44

[a] Many zero values and fewer than 7 nonzero values were assessed.

males. The 11KT concentrations in female carp and smallmouth, however, were positively correlated with Hg, and the E2:11KT was inversely correlated with total Hg residues (Table 3A). Vitellogenin concentrations were inversely correlated with Hg residues in female smallmouth (Table 3A). The significant correlations between Hg and selected biomarkers indicate that the endocrine systems of female carp and smallmouth may be affected by exposure to total Hg or, perhaps, to the more biologically available methylmercury (Kime, 1998) and that Hg may have an androgenic affect on their endocrine systems mainly at sites in the upper Hudson River.

The likelihood that Hg produces androgenic and (or) antiestrogenic effects in female carp, bullhead, and bass from the Hudson River is supported mainly by evidence from studies of other species of oviparous fish and amphibians. Friedmann et al. (2002) found that plasma concentrations of 11KT were positively, but not significantly, correlated with Hg residues in male largemouth. This study only examined males, but the result is consistent with the positive relations found between Hg and 11KT in female smallmouth and carp, and the inverse relation between Hg and VTG (and E2:11KT) in female smallmouth observed in this study (Table 3A). No studies relating total Hg and (or) methylmercury in tissues to plasma concentrations of sex-steroid hormones or VTG in carp, bullhead, or smallmouth could be found. Several studies report that Hg can inhibit or reduce gonad development, gametogenesis, fertilization success, number of eggs, the gonadosomatic index (GSI), and the viability of eggs mainly in female fish (Wiener and Spry, 1996; Kime, 1998). For example, exposure of female catfish (*Clarias batrachus*) to methylmercuric chloride and a mercurial fungicide at 0.03 to 0.05 mg Hg L^{-1} completely inhibited oocyte development and sexual maturation (Kirubagaran and Joy, 1988). Exposure of female murrel (*Channa punctatus*) to $HgCl_2$ at 0.017 mg Hg L^{-1} also decreased the size and number of stage II and III oocytes, increased atresia, and lowered the GSI (Dey and Bhattacharya, 1989).

Although results from the present study do not directly address fish reproduction, others have related Hg exposure to decreased reproduction ability in males of various species and some adverse effects on fish populations. For example, gonadal development was completely arrested in female and male murrel (oogenesis in female ovaries and spermatogenesis in male testes) exposed to sublethal concentrations of emisan (methoxy ethyl mercuric chloride) or $HgCl_2$ for six months during normal recrudescence periods (Ram and Sathyanesan, 1983; Ram and Joy, 1988). Growth and gonadal development were inhibited, and testes were atrophied, in male walleye (*Stizostedion vitreum*) fed catfish filets that were injected with 0.1 and 1.0 µg Hg g^{-1} over a six-month period (Friedmann et al., 1996). Experimental populations of mosquito fish (*Gambusia holbrooki*) declined, and the sex ratio, normally female biased in the wild, became male biased in populations that were exposed to 18 µg Hg L^{-1} for 111 d (Mulvey et al., 1995). Plasma concentrations of estradiol in female largemouth were inversely correlated, and the frequency of atretic oocytes in female largemouth and bluegill sunfish were positively correlated, with Hg concentrations in sediments at 11 sites in the Clinch River/Watts Bar Reservoir system that were contaminated by Hg and PCBs (Adams et al., 1999). Despite these examples, evidence for linkages between endocrine disruption and reproductive impairment in individuals and population effects are rare (Mills and Chichester, 2005), thus, it is not possible to determine if the health of wild fish populations in the Hudson River are at risk from

elevated concentrations of Hg, PCBs, and other contaminants.

The higher median Hg residues in carp, smallmouth, and largemouth from FD and TIP (smallmouth were not collected at TIP) than from sites in the middle and lower reaches (Fig. 3) indicate that acidic headwaters may be one source of Hg. The elevated median Hg residues in bullhead, smallmouth, and largemouth from AT (RK 246) indicate that the Mohawk River may be another Hg source; the Mohawk enters the Hudson River near RK 247 (Fig. 1) and, along with industrialized reaches of the Hudson River between AT (RK 246) and WAT (RK 249), might contribute Hg that accumulates in resident fish. Low Hg residues in fish tissues from downstream reaches could reflect lower Hg-methylation rates and methylmercury concentrations in water, sediment, and prey than in the upper reaches. Other studies indicate that dilute acidic waters (and high concentrations of dissolved organic carbon), typical of headwater streams, can produce elevated Hg residues in fish (Simonin et al., 1993).

Relation of endocrine biomarkers to PCB residues

Smallmouth bass. Concentrations of E2 and 11KT, and the E2:11KT, were significantly correlated with lipid-based PCB residues in filets of male smallmouth, and E2:11KT was positively correlated with PCB residues in filets of female smallmouth (Table 3B). Increases in plasma concentrations of E2, or decreases in 11KT, in response to elevated PCB residues, may not necessarily affect the oocyte production in female smallmouth because that condition is not unusual. Correlation and regression analyses of smallmouth data from six sites (Table 3B) indicate that lipid-based PCBs could explain from 23 to 31 % of the variability in the concentrations of E2 and 11KT or E2:11KT in male smallmouth, suggesting that PCBs could have either estrogenic or antiandrogenic effects on male smallmouth. No studies could be located that relate PCB concentrations to the endocrine function or reproductive health of smallmouth.

Largemouth bass. Concentrations of E2, 11KT, and VTG were positively correlated with lipid-based PCB residues in male largemouth (Table 3B). Lipid-based PCB residues were inversely correlated with 11KT concentrations and positively correlated with VTG concentrations in female largemouth (Table 3B). Increased plasma concentrations of E2 and VTG in male largemouth suggest their endocrine systems are affected, yet lipid-based PCB residues could only explain 10 % of the variability in E2 and 29 % of the variability in VTG concentrations. These findings indicate that PCBs could have a moderate estrogenic effect on the endocrine systems of male and female largemouth, but that other factors also could be important.

Several other studies reported that PCBs could have an antiestrogenic or androgenic effect in female fish, whereas, a small number identified estrogenic effects in male fish as we noted in the Hudson River. For example, plasma concentrations of 11KT were lower and E2 concentrations were higher in plasma of male largemouth, and E2 and VTG concentrations were lower in female largemouth from sites contaminated by PCBs (and other pollutants) than in fish from an uncontaminated reference site during the reproductive season in the St. Johns River, Florida (Sepulveda et al., 2002). Exogenous PCBs were found to interact directly with cellular estrogen receptors and, thus, to function as an estrogen in male largemouth and as either an estrogen or an antiestrogen in prespawning female largemouth at sites in a PCB-contaminated reservoir on the South Carolina-Georgia border (Garcia et al., 1997). During spring (reproductive period) surveys, plasma concentrations of VTG were lower in female largemouth from a highly PCB-contaminated site in Woods Pond, Massachusetts, than VTG in fish from an uncontaminated nearby lake; VTG concentrations, however, were similar in male and female largemouth from both systems during fall (non-reproductive period) surveys (Smith et al., 2002). The E2:11KT ratios were generally higher for male largemouth than for females at Woods Pond because of a sex-steroid imbalance in both sexes.

Common carp. Concentrations of 11KT and VTG, and E2:11KT were significantly correlated with lipid-based PCB residues in male carp; only 11KT and the E2:11KT were correlated with PCB residues in female carp (Table 3B). Correlation and regression analyses indicate that lipid-based PCB residues explain 6 to 10 % of the variability in 11KT concentrations and in the E2:11KT in male carp (Table 3B). These findings suggest that PCBs may have small antiandrogenic or estrogenic effects on endocrine systems of male carp in the basin.

Detectable concentrations of VTG in 20 % of male carp at TIP, and altered concentrations of sex steroids and VTG in carp from PCB-contaminated sites in other studies, indicate that PCBs could act as an estrogen and possibly affect endocrine systems of carp in the Hudson River. Vitellogenin concentrations as high as 0.6 mg ml^{-1} have been detected in plasma of male carp from waters across the United States and England that receive either industrial or municipal effluents (Goodbred et al., 1997; Jobling et al., 1998; Smith et al., 2002). Vitellogenin was measured in a high percentage of male carp, and elevated concentrations of VTG and 11KT (and low E2/11KT) were measured in plasma of female carp from Las Vegas Wash and Las Vegas Bay on Lake Mead, Nevada, where sediments and fish were contaminated by PCBs and other organochlorines (Bevans et al., 1996). Median concentrations of E2 in male and female carp were inversely correlated with sediment PCB residues in Woods Pond, Massachusetts (Smith et al., 2002). Antiandrogenic and

antiestrogenic effects were observed in carp after four weeks of intraperitoneal injections of Aroclor 1254; concentrations of estrogens and androgens in blood of female and male were significantly reduced (Sivarajah et al., 1978a); developing oocytes became fragmented and spermatozoa were damaged (Sivarajah et al., 1978b).

Brown bullhead. Only the biomarker E2:11KT was significantly correlated with lipid-based PCB residues in male bullhead (Table 3B). Lipid-based PCB residues could explain from 5 to 13 % of the variability in the 11KT concentration and E2:11KT in male bullhead. These findings indicate that PCBs could have a small antiandrogenic effect on the endocrine system of male bullhead in some parts of the Hudson River. For example, male bullhead with high lipid-based PCB residues tend to have low 11KT concentrations and, therefore, high E2:11KT ratios. No studies could be found that relate PCB concentrations to the endocrine function or reproductive health of bullhead.

General. The effects of PCBs on the endocrine systems of bullhead, carp, smallmouth, and largemouth have not been investigated directly, but PCBs have been reported to induce modulation of endocrine biomarkers and (or) affect the reproductive health of several saltwater and freshwater fish species (USEPA, 1997; Kime, 1998; Monosson, 2000). For example, plasma concentrations of estradiol, VTG, and testosterone were significantly reduced in female Atlantic croaker (*Micropogonias undulatus*) fed PCB Aroclors during ovarian recrudescence (Thomas, 1989). Plasma concentrations of E2 and VTG were also lowered significantly and the number of sexually mature female white perch were suppressed, or sexual maturation delayed, following intraperitoneal injections of a planar PCB (Monosson et al., 1994). Except for the relation of 11KT with PCB residues in largemouth, results from the present study, and from several other investigations, indicate that PCBs are generally estrogenic (positively correlated with E2 and VTG) or antiandrogenic (negatively correlated with 11KT) in male and female fish in the Hudson River.

Relation of endocrine biomarkers to atresia

Oocyte atresia is an involution or resorption of unfertilized eggs by the ovaries (Schmitt and Dethloff, 2000). Atresia is sometimes observed in mature females of many fish species (especially after spawning), but high rates may be indicative of endocrine disruption or reproductive impairment when prevalent during all but the post-ovulatory maturation stages (Cross and Hose, 1988). Though background rates in the Hudson River were not known, median atresia was less than 10 % in female carp and bullhead at all sites, but was near or above 10 % in female largemouth from LL, FD, SC, and AT; median

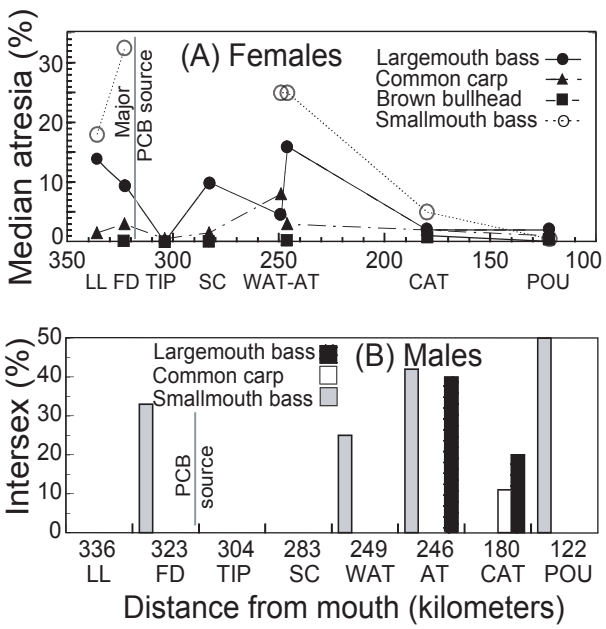

Figure 9. Atresia in female fish and intersex in male fish from eight study sites in the Hudson River, 1998: (A) median percent of atresia in females and (B) percent intersex in males. [Intersex did not occur in male bullhead. Smallmouth bass were not collected at TIP or SC. Site locations are shown in Fig. 1.]

atresia ranged from 18 to 33 % in smallmouth from LL, FD, WAT, and AT (Fig. 9A, Appendix A). High rates of atresia were found in some largemouth at TIP, but the median value was zero because it was documented only in a small number of individuals (Appendix A).

Significant ($p \leq 0.1$) inverse correlations were found between Log_{10}-tranformed and untransformed atresia and concentrations of VTG and E2 and the ratio, E2:11KT in female smallmouth (Table 4). A significant inverse correlation between atresia and VTG concentration was also noted in female largemouth, but not in female carp or bullhead (Table 4). These results indicate that the endocrine systems and ability of female smallmouth and largemouth to produce viable eggs could be adversely affected only if Hg or PCB exposures cause a decrease in E2 and VTG concentrations and in E2:11KT.

Though contaminant concentrations were not directly correlated with atresia in fish from the Hudson River, other studies found that PCB exposure increased oocyte atresia and suppressed oocyte production or development in selected fish species (Monosson et al., 1994; Hontela et al., 1995; Sepulveda et al., 2002). High rates of atresia in smallmouth at most sites in the middle and upper reaches of the Hudson indicate their endocrine systems were either adversely affected by contaminants, females were post-ovulatory or they normally exhibit high rates of atresia. Low rates of atresia in smallmouth collected

Table 4. Correlation (r) of reproductive biomarkers (atresia and the gonadosomatic index-GSI) with endocrine biomarkers in plasma of individual fish collected from six to eight sites in the Hudson River, 1998. [Coefficients of correlation are significant to $p < 0.05$ or 0.10[a]. Boldface values indicate correlations with potential adverse effects on reproductive health of the species or gender. Approximate number of biomarkers assessed for each species and gender are given in Appendix A. na = not applicable; ns = not significant.]

Dependent variable	Smallmouth bass		Common carp		Brown bullhead		Largemouth bass	
	Male	Female	Male	Female	Male	Female	Male	Female
A. Log_{10}17β-estradiol								
Atresia	na	**–0.32**[a]	na	ns	na	ns	na	ns
Log_{10}GSI	ns	**0.46**	ns	ns	ns	ns	ns	**0.261**
B. Log_{10}11-ketotestosterone								
Atresia	na	ns	na	ns	na	ns	na	ns
Log_{10}GSI	ns	ns	0.34	ns	ns	ns	ns	**–0.221**
C. Log_{10} (E2:11KT ratio+1)								
Atresia	na	**–0.36**	na	ns	na	ns	na	ns
Log_{10}GSI	ns	**0.38**	ns	ns	ns	ns	ns	**0.34**
D. Log_{10}Vitellogenin								
Atresia	na	**–0.60**	na	ns	na	ns	na	**–0.54**
Log_{10}GSI	ns	**0.45**	**–0.95**[b]	–0.44	ns	ns	**–0.452**	0.35

[a] significant to $p < 0.1$
[b] many zero values and 5 to 7 nonzero values were assessed.

from CAT and POU, before and after collection of smallmouth with high rates of atresia from other sites, indicate that high atresia rates are not the normal background condition. Differing degrees of sexual maturation could be another reason for the disparity in atresia rates in fish from upstream and downstream reaches, but this is not likely because maturity (stage) differences were typically not significant, and daytime water temperatures were similar (near 18 °C) at all sites during collections. Spawning of smallmouth in the Northeast normally takes place during June, when water temperatures reach 15–18 °C (Carlander, 1977). Though background levels of atresia are unknown, the moderate-to-high levels found in female largemouth and smallmouth suggest that oocyte production of individuals may be adversely affected by an androgen or antiestrogen (not necessarily PCBs or Hg) in parts of the basin.

Relation of endocrine biomarkers to intersex

Developing oocytes have been discovered in the testes of male fish (intersex) in various species populations with increasing frequency (Burke, 2002). In the present study, oocytes (intersex) were observed in no male bullhead, in 10 % of male carp at CAT, in 25 to 50 % of male smallmouth at FD, WAT, AT, and POU, and in 20 to 40 % of male largemouth at AT and CAT (Fig. 9B, Appendix A). The large percentage of intersex smallmouth and largemouth bass at several sites indicate that an estrogen may affect their endocrine systems in the middle and lower reaches of the basin. The lack of substantial intersex in male largemouth, and the low rates of atresia in female largemouth at the sites with the highest PCB concentrations in sediments and tissues (TIP and SC) suggest that PCBs did not affect reproductive histology in largemouth, even though they had a small effect on two endocrine biomarkers (hormone and VTG concentrations).

High rates of intersex in male bass at downstream sites in the Hudson River may be caused by increased accumulations of estrogenic contaminants from treated sewage that flows into the river along its entire length. Proximity of fish to treated sewage effluent has been correlated with increased rates of intersex in male roach (*Rutilius rutilus*) (Jobling et al., 1998) and in males of other salt- and freshwater species (Vigano et al., 2001; Burke, 2002). Besides having oocytes in their testes, intersex male roach in U.K. rivers that received sewage often had physical malformations that could prevent release of sperm (Jobling et al., 2002). The specific contaminants that affect intersex, the normal or background rates of intersex in male bass, and the potential effects of intersex on the reproductive health of bass in the Hudson River, however, remain unknown.

Relation of endocrine biomarkers to the gonadosomatic index

Adverse effects of endocrine disruption on egg and sperm production may, but not always, be inferred from decreases in the gonadosomatic index (GSI), which is the ratio of ovaries or testes weight to body weight (Murphy and Willis, 1996). Log_{10}-transformed GSIs in fish from the Hudson River were significantly correlated (inversely) only with log_{10}-transformed 11KT and VTG in male

carp and with log_{10}-transformed VTG in male largemouth (Table 4). These correlations were moderately strong, but the small number of non-zero VTG values (5 to 7) indicates that the effect of elevated VTG concentrations on gonad size in male largemouth and carp is speculative. Log_{10}-transformed GSI values were significantly correlated (positively) with E2, E2:11KT, and VTG in female smallmouth and largemouth; inversely correlated with 11KT in female largemouth; and significantly correlated (inversely) with VTG concentrations in female carp (Table 4). The GSI was not correlated with any endocrine biomarkers in male or female bullhead (Table 4). The weak relations between GSI and 11KT in female largemouth, and between GSI and VTG in male carp and largemouth indicate some effect on egg and sperm number or size, but the overall effect on gamete production cannot be quantified directly using the GSI.

Potential endocrine disruption in fish of the Hudson River

Results from the present study indicate that modulation of sex-steroid hormones and VTG concentrations occurs in the four species examined, although the degree of departure from the norm and, thus, the severity and significance of endocrine disruption cannot be fully quantified. The results do implicate Hg as a possible cause for increased 11KT concentrations, decreased E2:11KT, and decreased VTG concentrations in female smallmouth and carp; PCB exposures also may decrease 11KT, increase E2:11KT, and increase VTG concentrations in males of all four fish species studied. The results support the hypotheses that PCBs and Hg contamination and exposures potentially alter endocrine biomarkers in smallmouth, largemouth, carp, and bullhead. Additional information on normal fluctuations in atresia, intersex, sex-steroid hormones, and VTG through annual reproductive cycles in these species from contaminated and uncontaminated locations is needed before the specific causes for endocrine biomarker modulation and the significance of potential endocrine disruption can be evaluated.

Study limitations, information gaps, and emerging issues

Some discussion of study limitations, information gaps, and emerging issues may help place results of the present investigation into better perspective. This study used a continuing NYSDEC sampling program that was designed to assess long-term trends in PCB and Hg residues in fish from selected sites in the Hudson River (Sloan et al., 1984; Sloan, 2000; Sloan et al., 2002; 2005). The number of fish species and sites were increased, but collections were made only once at each of eight sites during a 5-week period in the late spring or early summer (May-June). Sampling intersected the probable spawning periods for the four fish species in the Hudson River (Carlander, 1969; 1977). In addition, only PCBs and Hg residues in fish tissues were analyzed. Bed sediments were collected in 1998 from five sites, (LL, TIP, SC, AT, and CAT) for analysis of important Aroclors and several other potential endocrine-disrupting compounds (some wastewater constituents). A more extensive and intensive sampling design could produce contaminant data needed to more completely quantify the extent, magnitude, causes, and implications of endocrine disruption in fish from the river. The normal range and baseline (background) biomarker concentrations that each species and sex exhibit during their growth and annual maturation cycles in uncontaminated sites in the Hudson River were not known. Limited information for carp is available, but not in temperate rivers of the Northeast. These information gaps preclude a full interpretation of biomarker deviation and potential endocrine disruption in fish from the Hudson River.

The high Hg and 11TK residues (and low E2:11KT) in female bass and carp at LL suggest Hg acts as an androgen and may have affected their endocrine systems, therefore, LL may not have been a good control site. Though a pristine control site may not exist in the Hudson River, endocrine biomarker data from fish with low PCBs and low Hg residues in filets would help quantify background or reference conditions and better gauge the effects of contaminants on the endocrine system of affected fish species.

The absence of smallmouth from the two most highly PCB-contaminated sites in the river (TIP and SC) hampers interpretation of the relations between tissue PCB residues and endocrine biomarkers in smallmouth. Though not initially selected for study, smallmouth were collected at some sites where the minimum number of mature largemouth could not be obtained. This information gap is important because our data indicate the endocrine system of smallmouth could be more responsive to environmental stressors than that of the other three species.

Another important information gap not be addressed by this study, was the direct association between effects (elevated endocrine biomarkers in fish) and specific causes (e.g., high PCB residues in tissues or in sediments). Direct cause-and-effect relations could be determined only by exposing fish to PCBs of Hg while holding all other conditions constant. Though most effectively studied under laboratory conditions, confidence in contaminant and biomarker relations could be strengthened through an assessment of the associations between endocrine biomarkers and tissue residues of a wide range of potential endocrine-disrupting chemicals that contaminate the Hudson River.

Significant modulation of endocrine biomarkers in male and female smallmouth, largemouth, carp, and bullhead at some reaches indicate the endocrine systems of

all four species may be disrupted in parts of the Hudson River; however, additional monitoring and research are needed to more completely characterize and quantify the sources and magnitude of endocrine modulation, the effects on reproduction output (eggs and sperm production) in individuals, and possible impacts of endocrine disruptors on resident fish populations. Such efforts should attempt to: (1) delineate the distribution of potential endocrine-disrupting contaminants (including pharmaceuticals) in water, sediment, and fish tissues, (2) assess specific effects of these contaminants on the endocrine and reproductive systems of local fish species, and (3) define the range and baseline biomarker concentrations throughout normal maturation cycles for selected species. Though laboratory studies indicate that endocrine disruption can affect the reproductive health of selected fish species, actual evidence linking endocrine disruption in individual fish with adverse effects on the health and viability of wild fish populations is rare or nonexistent. Therefore, linkages between estrogenic or androgenic contaminants, endocrine disruption within individuals, and population-level effects are greatly needed to better document and evaluate the threat of these contaminants in aquatic ecosystems.

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Appendix A. Median and standard deviation (SD) for age, maturity stage, length, weight, and concentration of Hg, PCBs, and biomarkers for mature fish collected from eight sites in the Hudson River, 1998. [n, number of samples; GSI, gonadosomatic index; RK, river kilometer (distance of site from mouth, in kilometers); na, not applicable.]

Species, gender, and number of samples (n)	Weight (g)	Length (mm)	Age (years)	Total Hg (µg g^{-1})	VTG (mg ml^{-1})	E2 (pg ml^{-1})	11KT (pg ml^{-1})	E2:11KT ratio	Total PCBs (µg g^{-1})	Lipid PCBs (µg g-$lipid^{-1}$)	GSI	Maturity stage	Atresia – Intersex[a] (%)
Lake Luzerne (LL) – RK 352 (sampled June 15–16, 1998)													
Largemouth bass-female (2)	227	259	4	0.280	0.49	601.0	939.5	0.634	0.050	2.09	0.036	4.0	13.0
SD	67	26	0	0.127	0.19	254.6	357.1	0.030	0.000	0.26	0.000	0.0	8.5
Largemouth bass-male (1)	144	22	4	0.120	0.00	401.0	378.0	1.061	0.050	4.67	0.000	1.0	0
SD	na	na	na	na	na	na	na	na	na	na	na	na	na
Carp-female (7)	2672	549	8	0.330	3.70	399.0	854.0	0.574	0.098	4.21	0.104	2.0	1.0
SD	606	41	1	0.101	1.08	500.2	296.7	0.557	0.176	3.86	0.083	0.0	1.0
Carp-male (9)	2879	571	9	0.270	0.00	271.0	2584.0	0.105	0.228	3.27	0.084	3.0	0
SD	709	54	2	0.049	0.00	222.9	2429.3	0.111	0.294	12.45	0.019	0.0	na
Smallmouth bass-female (13)	270	277	5	0.360	0.60	212.0	564.0	0.359	0.050	2.44	0.028	4.0	18.0
SD	104	31	1	0.221	1.48	259.8	266.3	0.507	0.062	9.37	0.010	0.8	9.8
Smallmouth bass-male (9)	250	281	4	0.370	0.00	68.0	860.0	0.089	0.079	3.10	0.003	3.0	0
SD	143	48	2	0.155	0.00	186.4	333.4	0.160	0.023	2.32	0.004	0.7	na
Feeder Dam (FD) – RK 323 (sampled June 17–18, 1998)													
Largemouth bass-female (6)	707	361	4	0.425	0.95	359.5	788.0	0.457	0.112	8.01	0.020	2.5	9.5
SD	243	41	2	0.144	1.52	482.9	170.8	0.539	0.046	3.93	0.008	0.5	19.4
Brown bullhead-female (6)	413	308	7	0.355	2.76	3178.0	1027.5	2.666	0.139	6.14	0.064	3.0	0.0
SD	72	19	1	0.040	1.35	2296.4	428.3	1.855	0.069	2.78	0.039	0.5	0.8
Brown bullhead-male (9)	518	331	8	0.240	0.00	522.0	3246.0	0.176	0.175	10.49	0.003	4.0	0
SD	61	20	1	0.041	0.00	194.3	2154.6	0.312	0.076	5.46	0.004	0.3	na
Carp-female (6)	8229	800	10	0.290	7.30	1021.5	429.5	2.733	1.173	18.49	0.090	2.0	3.0
SD	2009	37	3	0.116	3.60	570.0	173.7	1.513	0.796	20.41	0.028	0.0	9.1
Carp-male (10)	6753	750	11	0.410	0.00	237.0	846.5	0.297	1.840	15.20	0.050	2.0	0
SD	1213	34	3	0.141	0.00	109.4	372.7	0.229	0.854	16.32	0.016	0.8	na
Smallmouth bass-female (2)	219	264	5	0.495	0.58	278.0	544.0	0.512	0.155	11.75	0.022	4.0	32.5
SD	14	7	0	0.049	0.16	147.1	5.7	0.276	0.097	3.10	0.011	0.0	4.9
Smallmouth bass-male (9)	237	270	5	0.460	0.00	359.0	487.0	0.694	0.210	19.22	0.002	3.0	33.3
SD	70	24	1	0.135	0.16	141.1	77.1	0.281	0.196	16.95	0.001	0.7	na
Thompson Island Pool (TIP) – RK 304 (sampled May 18–19; June 26, 1998)													
Largemouth bass-female (12)	1242	426	7	0.555	6.41	499.0	531.0	0.970	10.800	803.49	0.029	2.0	0.0
SD	564	87	2	0.255	2.51	324.0	187.9	0.512	13.062	397.32	0.010	0.5	7.4
Largemouth bass-male (12)	1097	420	7	0.535	0.00	327.5	855.5	0.431	21.550	1252.85	0.005	1.0	0
SD	404	49	2	0.223	0.01	128.1	625.2	0.218	9.384	872.81	0.012	0.5	na
Brown bullhead-female (3)	564	331	7	0.170	2.63	2199.0	961.0	2.226	22.200	609.89	0.107	4.0	0.0
SD	91	7	3	0.050	2.13	1645.2	58.5	1.899	9.682	128.46	0.055	0.0	1.2
Brown bullhead-male (9)	377	299	5	0.130	0.00	197.0	587.0	0.573	11.900	285.09	0.003	3.0	0
SD	139	35	2	0.057	0.01	447.1	1395.4	0.778	9.627	214.42	0.003	1.2	na
Carp-female (4)	3606	608	6	0.240	4.82	641.0	258.0	2.520	23.750	308.10	0.169	2.0	0.5
SD	3884	169	3	0.105	2.71	279.1	78.3	1.211	27.918	187.11	0.108	0.0	1.4
Carp-male (5)	3111	580	3	0.150	0.00	256.0	659.0	0.563	21.800	274.02	0.033	1.0	0
SD	2038	152	5	0.145	0.05	138.7	912.3	0.274	33.293	108.51	0.037	0.6	na

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Species, gender, and number of samples (n)	Weight (g)	Length (mm)	Age (years)	Total Hg (μg g^{-1})	VTG (mg ml^{-1})	E2 (pg ml^{-1})	11KT (pg ml^{-1})	E2:11KT ratio	Total PCBs (μg g^{-1})	Lipid PCBs (μg g-$lipid^{-1}$)	GSI	Maturity stage	Atresia – Intersex[a] (%)
Largemouth bass-female (9)	511	340	5	0.400	2.66	636.0	251.0	2.288	4.260	486.96	0.033	3.0	10.0
SD	415	70	3	0.167	1.26	263.6	156.9	1.853	5.457	231.64	0.014	0.6	8.6
Largemouth bass-male (10)	905	393	7	0.370	0.00	644.0	191.5	3.522	9.500	566.21	0.004	2.5	0
SD	304	47	3	0.161	0.01	166.3	87.1	1.084	9.938	352.83	0.079	0.8	na
Brown bullhead-female (5)	435	310	6	0.160	1.18	992.0	344.0	3.485	8.300	345.46	0.120	3.0	0.0
SD	161	42	1	0.049	na	665.1	118.6	1.282	5.713	151.07	0.069	0.5	0.0
Brown bullhead-male (12)	688	354	7	0.135	0.00	690.0	570.5	1.118	10.750	268.64	0.003	3.0	0
SD	148	22	2	0.050	0.00	388.6	404.9	0.760	3.656	71.65	0.001	0.8	na
Carp-female (6)	9525	834	11	0.290	3.09	942.0	149.0	5.032	40.150	360.51	na	2.0	1.5
SD	3792	134	3	0.104	0.76	281.2	104.8	2.576	42.841	125.62	na	0.0	2.3
Carp-male (6)	6577	729	10	0.320	0.00	296.5	347.5	0.761	61.000	318.96	0.077	2.0	0
SD	1583	31	3	0.101	0.00	234.4	283.6	0.791	172.818	1166.75	0.011	0.4	na
Waterford (WAT) – RK 249 (sampled June 22–23, 1998)													
Largemouth bass-female (2)	953	404	6	0.335	4.64	199.5	462.5	0.420	2.025	282.32	0.030	2.5	4.5
SD	4	10	2	0.304	3.94	118.1	47.4	0.212	1.322	124.00	0.015	0.7	6.4
Largemouth bass-male (1)	744	365	5	0.390	0.00	160.0	850.0	0.188	7.170	224.77	0.004	3.0	0
SD	na	na	na	na	na	na	na	na	na	na	na	na	na
Brown bullhead-female (5)	488	320	6	0.210	3.35	297.0	895.0	0.352	5.030	65.16	0.118	4.0	0.0
SD	145	30	3	0.071	1.53	1282.7	399.9	1.036	3.633	77.33	0.051	0.9	12.0
Brown bullhead-male (8)	435	324	7	0.190	0.00	237.5	1228.5	0.183	3.100	116.02	0.002	2.0	0
SD	166	34	3	0.130	0.02	181.1	2366.4	0.119	1.574	64.61	0.004	1.4	na
Carp-female (10)	3648	618	7	0.300	4.37	640.0	506.5	1.265	13.650	189.93	0.096	2.0	8.0
SD	1255	68	3	0.062	0.55	209.1	138.4	0.532	18.620	88.34	0.097	0.5	27.2
Carp-male (10)	3228	601	7	0.255	0.00	111.5	759.5	0.172	14.650	265.03	0.052	2.5	0
SD	576	43	2	0.061	0.01	142.7	399.6	0.205	17.374	131.65	0.011	0.7	na
Smallmouth bass-female (3)	606	374	5	0.430	1.07	367.0	429.0	0.840	5.030	253.79	0.017	2.0	25.0
SD	100	7	0	0.050	1.52	88.1	36.0	0.155	2.798	99.61	0.007	0.6	7.0
Smallmouth bass-male (8)	397	318	4	0.235	0.00	248.5	476.0	0.526	5.715	336.71	0.002	3.0	25.0
SD	142	57	1	0.085	0.00	93.0	202.1	0.442	1.703	222.37	0.004	0.7	na
Albany/Troy (AT) – RK 246 (sampled June 3–4, 1998)													
Largemouth bass-female (1)	916	424	5	0.610	0.72	288.0	639.0	0.451	10.900	221.55	0.021	4.0	16.0
SD	na	na	na	na	na	na	na	na	na	na	na	na	na
Largemouth bass-male (5)	934	409	7	0.460	0.00	372.0	492.0	0.835	7.800	227.62	0.005	3.0	40.0
SD	456	75	3	0.474	0.00	108.1	1339.1	0.358	8.931	213.23	0.002	0.5	na
Brown bullhead-male (4)	476	332	9	0.150	0.00	512.5	1111.5	0.462	2.080	36.77	0.002	1.0	0
SD	172	47	2	0.355	0.00	48.1	290.1	0.096	1.736	16.49	0.001	1.5	na
Carp-female (9)	1934	505	3	0.170	4.95	544.0	346.0	1.538	9.300	75.56	0.126	3.0	3.0
SD	2917	132	3	0.078	2.49	190.2	107.3	0.858	8.702	48.93	0.088	0.5	1.6
Carp-male (10)	3539	631	7	0.150	0.00	178.0	624.0	0.274	11.950	92.85	0.055	3.0	0
SD	1424	75	3	0.102	0.01	155.3	557.6	0.207	7.369	48.38	0.029	0.7	na

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Smallmouth bass-female (9)	505	349	5	0.510	0.43	438.0	469.0	0.934	5.200	182.40	0.024	4.0	25.0
SD	250	48	1	0.176	0.49	113.1	165.1	0.520	1.913	46.09	0.010	1.0	12.2
Smallmouth bass-male (12)	529	352	5	0.395	0.00	390.0	551.0	0.684	6.900	274.49	0.003	3.0	41.7
SD	231	55	1	0.211	0.00	86.8	69.2	0.227	2.438	206.59	0.001	0.4	na
Catskill (CAT) – RK 180 (sampled June 1, 1998)													
Largemouth bass-female (6)	1133	416	6	0.590	6.90	863.0	298.5	3.165	2.230	138.59	0.044	2.0	2.0
SD	357	36	1	0.257	7.52	686.0	93.9	3.579	4.843	110.59	0.016	0.5	1.5
Largemouth bass-male (10)	668	371	4	0.295	0.00	167.0	308.5	0.686	3.810	125.19	0.005	2.0	20.0
SD	425	66	2	0.157	0.00	88.0	104.6	0.322	3.435	78.21	0.011	0.8	na
Brown bullhead-female (10)	262	258	4	0.055	3.98	1519.5	328.5	3.952	1.660	32.79	0.086	3.0	1.0
SD	109	29	1	0.018	1.44	1034.7	157.2	2.621	0.684	7.36	0.031	0.5	1.3
Brown bullhead-male (12)	297	270	4	0.055	0.00	269.5	477.0	0.557	1.550	33.60	0.002	3.0	0
SD	109	25	1	0.011	0.00	133.7	333.7	0.237	0.524	6.71	0.002	1.4	na
Carp-female (9)	2280	534	3	0.130	4.47	454.0	188.0	2.415	5.080	75.04	0.105	3.0	2.0
SD	832	72	3	0.061	2.47	473.5	309.5	1.606	2.202	29.51	0.037	0.4	9.1
Carp-male (9)	2102	511	3	0.090	0.00	245.0	777.0	0.332	4.500	88.10	0.047	2.0	11.1
SD	1063	87	3	0.069	0.00	132.6	374.5	0.192	6.419	49.42	0.023	0.8	na
Smallmouth bass-female (3)	652	315	5	0.350	0.90	158.0	136.0	1.198	4.500	180.72	0.028	3.0	5.0
SD	285	41	1	0.135	2.72	52.3	7.6	0.300	2.811	103.53	0.026	0.6	13.1
Smallmouth bass-male (3)	320	290	3	0.220	0.00	211.0	251.0	1.095	3.160	136.54	0.003	1.0	0
SD	83	32	1	0.015	0.00	75.8	78.8	0.322	0.666	18.71	0.001	0.6	na
Poughkeepsie (POU) – RK 122 (sampled May 27–29, 1998)													
Largemouth bass-female (5)	584	345	4	0.230	5.76	1064.0	252.0	3.837	2.660	131.58	0.047	3.0	2.0
SD	547	76	2	0.214	11.74	1094.0	366.9	1.171	2.169	154.25	0.022	0.0	7.9
Largemouth bass-male (4)	717	365	5	0.290	0.00	374.5	200.5	2.018	2.110	170.85	0.006	2.0	0
SD	274	44	1	0.069	0.01	65.6	102.0	1.242	1.348	60.74	0.003	0.5	na
Brown bullhead-female (13)	254	253	6	0.520	3.71	703.0	254.0	3.738	2.040	34.28	0.041	3.0	0.0
SD	44	10	2	0.327	1.34	443.3	237.1	4.681	0.767	12.07	0.081	0.3	1.4
Brown bullhead-male (6)	244	256	5	0.160	0.00	371.5	484.0	0.617	2.320	29.12	0.002	2.5	0
SD	38	9	1	0.333	0.00	191.3	221.4	0.253	0.774	8.27	0.001	1.6	na
Carp-female (8)	2981	568	8	0.060	4.17	792.5	165.5	5.075	2.720	63.21	0.142	2.0	1.0
SD	1774	114	3	0.019	1.83	477.8	78.6	3.110	3.998	40.61	0.085	0.0	1.3
Carp-male (12)	3084	599	7	0.090	0.00	173.5	662.0	0.170	7.095	90.56	0.082	2.0	0
SD	865	44	2	0.056	0.00	68.7	873.8	0.450	2.475	42.22	0.018	0.0	na
Smallmouth bass-female (4)	455	312	4	0.210	2.83	789.5	125.5	5.272	1.585	104.31	0.055	3.0	0.5
SD	148	47	1	0.092	1.43	439.6	91.9	1.517	1.286	70.40	0.048	0.0	17.3
Smallmouth bass-male (4)	412	317	4	0.290	0.00	353.0	332.0	1.063	4.405	156.76	0.006	3.0	50.0
SD	129	32	1	0.304	0.00	73.8	352.3	0.810	1.424	101.68	0.003	0.5	na

[a] atresia in females and intersex in males