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ILLINOIS SUSTAINABLE TECHNOLOGY CENTER
PRAIRIE RESEARCH INSTITUTE

RR-122
March 2013
www.istc.illinois.edu

RR-122

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Submitted to the
Illinois Sustainable Technology Center
Prairie Research Institute
University of Illinois at Urbana-Champaign
www.istc.illinois.edu

The report is available on-line at:
http://www.istc.illinois.edu/info/library_docs/RR/RR122.pdf

Printed by the Authority of the State of Illinois
Patrick J. Quinn, Governor

ACKNOWLEDGEMENTS

This study was funded by the Illinois Sustainable Technology Center (ISTC), a division of the Prairie Research Institute at University of Illinois Urbana-Champaign (UIUC) (Grant no. HWR04188). John Scott (ISTC), Heidi Garbe and Erik Chatroop (UIUC), Allison Klement, Guillermo Arce, John Shaw, Joan Esarey, Marshall McDaniel, Nina Hill, and Molly Tranel (all of Illinois Natural History Survey [INHS]) provided technical assistance throughout this project.

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LIST OF ABBREVIATIONS

A	Aquatic
a-BHC	alpha-hexachlorocyclohexane
Al	Aluminum
ASE	Accelerated Solvent Extractor
Ba	Barium
b-BHC	beta-hexachlorocyclohexane
Ben. Ins.	Benthic Insects
BM	Big Marsh
Bol. Ins.	Bolus insects
BZ	Ballschmiter-Zell
Cd	Cadmium
Cr	Chromium
Cu	Copper
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlordiphenyltrichloroethane
ECD	Electron Capture Detector
EPA	Environmental Protection Agency
EPT	Ephemeroptera, Trichoptera, and Plecoptera
ERA	Ecological risk assessment
g-BHC	gamma-hexachlorocyclohexane or Lindane
GC	Gas Chromatograph
Hg	Mercury
HPX	Heptachlor epoxide (isomer B)
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
IR	Indian Ridge
IRM	Indian Ridge Marsh
J.D.	Julian Date
LCC	Lake Calumet Cluster

LIST OF ABBREVIATIONS (continued)

nm	Not measured
OXC	Oxychlordane
PAH	Polycyclic Aromatic Hydrocarbon
Pb	Lead
PBDE	Polybrominated diphenyl ether
PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzo-p-dioxins
PCDF	Polychlorinated dibenzofurans
PFE	Pressurized Fluid Extraction
PL	Powderhorn Lake
Se	Selenium
T	Terrestrial
USACE	United State Army Corps of Engineers
USEPA	United States Environmental Protection Agency
Zn	Zinc

ABSTRACT

The highly industrialized Grand Calumet River basin includes an extensive wetlands complex that has been severely degraded through heavy industrial activity, sewage and industrial discharges, landfills, and hazardous waste storage/disposal. Sediments and other environmental media in this area are contaminated with heavy metals and organic compounds. Our objective was to empirically quantify risks to insectivorous birds in the Lake Calumet wetlands region from contaminated sediments via ingestion of aquatic insects using tree swallows (*Tachycineta bicolor*) as a model organism. To accomplish this objective, we completed the following tasks: (1) assessed organic contaminant transfer (polychlorinated biphenyls [PCBs], organochlorine pesticides, polybrominated diphenylethers [PBDEs]) from an aquatic ecosystem (sediment and benthic macroinvertebrates) to a terrestrial food chain (tree swallows feeding on emergent aquatic insects), (2) quantified elemental contaminants in these locales and biota, (3) evaluated ecological effects these contaminants may have on tree swallows, comparing mercury loads and nesting ecology data at different sites as a case study, and (4) assessed the value of stable isotope data in determining how food chain length and food source (aquatic versus terrestrial, location of origin) affects contaminant loads in tree swallows nesting at Lake Calumet wetlands. With the exception of timing of nest initiation and other variables that are dependent on nest initiation timing (e.g., clutch size, and nestling mass), we observed no differences among sites in tree swallow nesting ecology endpoints. A variety of inorganic and organic contaminants were accumulated by nestlings via their insect diets, but concentrations of nearly all the contaminants were at the lower end of ranges in the literature. The exception to this trend was dichlorodiphenyldichloroethylene (DDE) concentrations in eggs and nestlings at Big Marsh which were among the higher reported values. To our knowledge, this paper is the first report of PBDEs concentrations in tree swallow nestlings. Our stable isotope analysis suggested a terrestrial origin for many of the contaminants as has been suggested by others.

INTRODUCTION

The highly industrialized Grand Calumet River basin includes an extensive wetlands complex that has been severely degraded through heavy industrial activity, sewage and industrial discharges, landfills, and hazardous waste storage/disposal. Sediments and other environmental media in this area are contaminated with heavy metals and organic compounds. For example, an ecological risk assessment (ERA) of the Lake Calumet Cluster (LCC) sites conducted by the U.S. Environmental Protection Agency (USEPA) revealed that a number of wetlands in that area contained sediments contaminated with high levels of polychlorinated biphenyls (PCBs); dichlordiphenyltrichloroethane (DDT) breakdown products such as dichlorodiphenyldichloroethylene (DDE); and metals, including aluminum (Al), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), selenium (Se), and zinc (Zn) (Sprenger et al., 2001). In an earlier study, Sinars (1999) also found that sediments at the LCC site had elevated concentrations of several metals and polycyclic aromatic hydrocarbons (PAHs). Based on contaminant levels in sediment, soils, and terrestrial invertebrates, Sprenger et al. (2001) determined that a number of assessment endpoints in the LCC site were at risk from PCBs and metals, including benthic macroinvertebrate communities, amphibian populations, insectivorous birds, and omnivorous waterfowl. Sediments at the adjacent Indian Ridge Marsh (IRM) also were found to contain high levels of contaminants including various metals and PAHs (USACE and Tetra Tech, 2001). Furthermore, green sunfish (*Lepomis cyanellus*), fathead minnow (*Pimephales promelas*), and crayfish (Malacostraca, Decapoda) collected from IRM and the adjacent Dead Stick Pond had elevated concentrations of metals, PCBs, and DDE (Levengood et al., 2004).

Spatial distributions of contaminants can be heterogeneous depending on environmental factors such as ultraviolet light (Peterson et al., 1990), oxygen concentration (Johnson, 1998), or sediment carbon content (Moermond, 2006), to list a few. Moreover, when (e.g., relative to a precipitation event), where (e.g., stream-side, or highly sloped landscape), or how compounds were originally used may impact the spatial heterogeneity of contaminants in the environment (e.g., PCBs) (Eisler, 2000). Furthermore, compounds that are more prevalent in specific systems may be transferred to others by biological activity; for example, sediment bound contaminants may be transferred to terrestrial ecosystems via emergent insects (Custer et al., 1998; Custer et al., 2003; Custer et al., 2005; Echols et al., 2004). Emergent adult insects make up a large part of the diet of many aerial insectivores, and it is well known that aquatic stages of insects (e.g., midge larvae, mayfly nymphs, etc.) accumulate sediment-bound contaminants, including PCBs, organochlorine pesticides, metals (Cd and Hg), and PAHs, and then transport the contaminants into terrestrial food webs upon emergence (Ciborowski and Corkum, 1988; Corkum et al., 1997; Dukerschein et al., 1992; Kovats and Ciborowski, 1989; Larsson, 1984; Mauck and Olson, 1977; Menzie, 1980; Reinhold et al., 1999). In fact, Larsson (1984) exposed chironomid larvae to sediment containing PCBs in the laboratory and found that while larval tissue concentrations were approximately an order of magnitude higher than sediment concentrations, PCBs in emergent adult chironomids were even higher than in larvae. Therefore, aquatic insects that emerge as adults from polluted wetlands may present a substantial risk of contaminant exposure to aerial-feeding insectivores such as bats and birds. We investigated this mode of transfer (at Lake Calumet wetlands) using insectivorous tree swallows (*Tachycineta bicolor*).

Tree swallows are commonly used as monitors due in part to their preference for feeding on insects over open water near their nesting site (McCarty, 1997; Mengelkoch et al., 2004; McCarty and Winkler, 1999) and their utilization of nest boxes (Kuerzi, 1941; Graber et. al., 1972). The natural history and ecology of tree swallows is relatively well understood (Kuerzi, 1941; Robertson et. al., 1992), and there has been much work using them to assess contamination throughout their range. Exposure to organochlorines such as PCBs (Custer et al., 1998; Bishop et al., 1999; Secord et al., 1999); DDE (Custer et al., 1998; Bishop et al., 1999); PAHs (Custer et al., 2001; Harris and Elliot, 2000); and polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) (Harris and Elliot, 2000; Custer et al., 2002) has been documented in tree swallows nesting near areas of contaminated sediments. Nest abandonment, reduced egg hatchability, supernormal clutches (McCarty and Secord, 1999a); changes in physiological parameters (Wayland et al., 1998; Bishop et al., 1999); aberrant nest construction (McCarty and Secord, 1999b); and plumage anomalies (McCarty and Secord, 2000) have been observed in tree swallows experiencing increased exposure to environmental contaminants. In addition, suggestions of increased nest abandonment (Wayland et al., 1998; Harris and Elliot, 2000) and decreased hatching and fledging success (Bishop et al., 1999) have been observed in other studies.

Polybrominated diphenyl ethers (PBDEs) are flame-retardants that are widely used in household electrical appliances, foams, bedding material and textiles. Congeners with fewer bromines have been found to be highly persistent and bioaccumulative. In 2004 the penta-brominated formations were banned in the European Union. Recent studies are providing evidence that higher brominated congeners break down to lower brominated forms (Kierkegaard et al., 1999; Van den Steen, 2006). Voorspoels et al. (2007) demonstrated that PBDEs move through simple terrestrial food chains, with measurable concentrations in some common bird species including great tits (*Parus major*), common buzzards (*Buteao buteao*), and sparrowhawks (*Accipiter nisus*). However, to date, no one has investigated PBDE accumulation in North American insectivorous birds like tree swallows.

While tree swallow nestlings have been shown to accumulate high levels of organic pollutants, only relatively recently have investigators used tree swallows to investigate the movement and effects of mercury in the environment (Bishop et al., 1995; Custer et al., 2007; Gerrard & St. Louis, 2001; Longcore et al., 2007a, b). Methyl-mercury is a known teratogen, carcinogen and mutagen in a variety of species and has been shown to adversely affect growth and reproduction in both terrestrial and aquatic organisms (Eisler, 2000). Mercury biomagnifies and bioaccumulates in animals, and tree swallows are primarily exposed through diet and maternal transfer to the eggs (Eisler, 2000). In birds, mercury is mainly depurated through molting, and Braune & Gaskin (1987) indicated that new feathers in gulls contain up to 93% of the mercury body burden after molting.

Tree swallows are both migratory and opportunistic insectivores, so it is imperative to understand the proportion of contaminant burden that is maternally transferred to nestlings from sources outside the study site, as well as the fraction of their diet that is terrestrial or aquatic in origin (Custer et al., 1998; Maul et al., 2006). The stress of migration may increase the bioavailability of existing maternal contaminants and this fact should be considered when determining the proportion of the contaminant burden that is local and the proportion originating from sources outside the study site (Maul et al., 2006; McCarty and Secord, 2000). Tree swallow eggs often are used to help determine the nestling contaminant burden that is maternally transferred and from unknown sources, but many studies have failed to clearly determine whether the contaminants seen in tree swallows originate in aquatic systems. This reality is perhaps all the more relevant because Law et al. (2003), Lindberg et al. (2004) and Jaspers et al. (2006) suggest that terrestrial feeding birds may be exposed to greater levels of higher brominated congeners than aquatic feeding birds. A similar result was reported for mercury by Cristol et al. (2008), who observed that birds that fed on spiders had higher mercury concentrations than aquatic feeding birds.

Stomach content analyses can provide a picture of food web structure, but they are only snapshots of feeding habits at a particular point in time. In contrast, stable isotope analysis provides a longer-term history of diet and trophic position (Power et al., 2002). The development of stable isotope analysis has improved understanding of food webs and trophic relationships in aquatic systems (Peterson and Fry, 1987), and has been shown to provide insight into sources of contaminant uptake (e.g., Power et al., 2002; Ruus et al., 2002; Burger, 2002). Carbon stable isotope ratios provide information on the carbon sources in an individual's diet, but do not change with increasing trophic level. Different types of aquatic and terrestrial plants have predictably different ^{13}C:^{12}C ratios, so information about what a particular organism is consuming can be obtained by comparing its ^{13}C:^{12}C ratio to those of various autochthonous and allochthonous carbon sources (primary producers) and consumers in lower trophic positions in an ecosystem (Cabana and Rasmussen, 1994). On the other hand, the ratio of ^{15}N to ^{14}N increases by an average of 3.4 parts per thousand with each increase in trophic level because ^{14}N is more easily excreted than ^{15}N (Cabana and Rasmussen, 1994). Thus, a fish will have a predictably higher ^{15}N to ^{14}N ratio than an invertebrate it feeds upon. Biomagnified contaminants increase with trophic level, and studies have revealed strong linear relationships between trophic position (as determined by ^{15}N:^{14}N ratio) and some contaminants. For example, Power et al. (2002) found that stable nitrogen isotope ratios (presented as $\delta^{15}N$: the ratio of ^{15}N to ^{14}N in the medium sampled, normalized to the ratio in air) explained 56% of the variation in mercury (Hg) concentrations in the tissues of various fish species in Canadian sub-Arctic lakes. In addition, for individual species such as lake trout (*Salvelinus namaycush*), $\delta^{13}C$ values explained ~59% of the variation from individual to individual in tissue Hg concentrations, which indicated in that case that Hg levels increased with increasing dependence on pelagic carbon (Power et al., 2002).

Our objective was to empirically quantify risks to insectivorous birds in the Lake Calumet wetlands region from contaminated sediments via ingestion of aquatic insects. To accomplish this objective we completed the following tasks: (1) assessed organic contaminant transfer (PCBs, organochlorine pesticides, PBDEs) from an aquatic ecosystem (sediment and benthic macroinvertebrates) to a terrestrial food chain (tree swallows feeding on emergent aquatic insects); (2) quantified elemental contaminants in these locales and biota; (3) evaluated ecological effects these contaminants may have on tree swallows, comparing mercury loads and nesting ecology data at different sites as a case study; and (4) assessed the value of stable isotope data in determining how food chain length and food source (aquatic versus terrestrial, location of origin) affects contaminant loads in tree swallows nesting at Lake Calumet wetlands.

MATERIALS AND METHODS

Experimental design/study sites

Concentrations of PCBs, PBDEs, DDT, DDD, DDE and ten other chlorinated pesticides, as well as mercury and 11 other elements, were measured in tree swallow eggs and nestlings, sediment, and insects at two contaminated sites and one reference site in the Lake Calumet region of south Chicago, IL. To better understand the source of any nestling contamination, diet samples were collected and identified to the taxonomic level necessary for determining ecosystem origin and trophic level (often genus or family level). Eggs and nestlings were collected from the same boxes to quantify the maternally transferred and locally accumulated contaminant burden.

Three study sites were selected to provide a spectrum of contamination around a localized region in the Lake Calumet area of Illinois. Big Marsh (41° 41' 30", 87° 34' 24") and Indian Ridge Marsh (41° 40' 51", 87° 33' 50" hereafter referred to as Indian Ridge) were chosen for their suspected high levels of contamination, and Powderhorn Lake (41° 38' 53", 87° 31' 37") was selected as a reference site due to no known sediment contamination. In March of 2004, thirty tree swallow nest boxes of 9.5 by 14 by 20 cm interior size, having a 3.8 cm diameter entry hole, were placed on posts at approximately 2 meters in height and a minimum of 15 meters apart at each of the sites. Boxes with no obscuring vegetation between them were placed at greater distances to avoid intra-specific aggression. Three-ft long segments of polyvinyl chloride (PVC) pipe filled with expandable foam insulation (Great Stuff© by Dow Chemical) were installed below each box to reduce predator access. All boxes were placed within 20 meters of water, with the vast majority of boxes within 5 meters of water. Several physical habitat measurements were made at each nest box. Percentage of area with a closed canopy was measured at each site using a spherical densitometer. The distance to open water and the distance to nearest woody tree taller than 2 meters were determined for each box as well.

Collection of sediment samples

Grab samples of sediments were collected in acid-washed, 500 ml glass jars from the sediment surface at three or four locations within each wetland during the summer of 2004. The three to four samples from each site were analyzed for contaminants separately. Grab samples were transferred from the sample jars to large, labeled Pyrex dishes. Samples were frozen, and then air-dried. At intervals during the drying process, the sediments were stirred and the larger pieces were broken up to expedite drying. When a stable weight was achieved, large rocks and pieces of plant matter were manually removed. The air-dried samples were then ground using a tabletop Retsch Laboratory Mortar Grinder. A sub-sample was removed from each air-dried sediment sample, oven dried at 105°C, and percent moisture determined on the oven dried sediment. Approximately 10-g portions of the air-dried sediment samples were weighed for sample preparation.

Conductivity, pH, dissolved oxygen, and temperature of the surface water were measured in the field. A Yellow Springs Instruments (YSI, RDP, Dayton, OH, USA) model 55 meter was used for dissolved oxygen measurements. Specific conductivity levels were determined with a YSI model 30 conductivity/salinity/dissolved solids meter, and pH was determined with an Accumet® (Fisher Scientific, Pittsburgh, PA, USA) model AP62 meter equipped with an Accumet® gel-filled combination electrode.

Sediment toxicity

To further determine the effects of sediment-bound contaminants on food resources for insectivorous birds, direct toxic effects of sediments on aquatic macroinvertebrates were determined in the laboratory. We conducted whole sediment toxicity bioassays using Method 100.1 (USEPA, 2000), "*Hyalella azteca* 10-d Survival and Growth Test for Sediments". Organisms used for testing were reared in the laboratory according to methods described by USEPA (2000).

Benthic macroinvertebrate sampling/adult insect collection

Aquatic insects were collected in both years from each site by kick netting. Larger genera were collected disproportionately to ensure there was enough mass for contaminant analysis, and genera not regularly seen in bolus samples were minimally included. All insects were frozen, identified to order or family, sorted and rinsed in de-ionized water before submission for contaminant analysis.

Five 1-ft^2 emergence traps were placed on the surface of the water, at each site on May 20 in 2004 and May 3 in 2005. The traps were checked and the insects collected every other day until the traps were removed on June 30. Insects were frozen within 48 hours of emergence, identified to order or family, sorted and rinsed in de-ionized water before submission for contaminant analysis. All insect samples for contaminant analysis were ground under liquid argon and sample aliquots were stored at -20ºC.

Tree swallow eggs and nestlings

After placement, boxes were checked once per week to document start of nest building and clutch initiation. Once nest building was initiated, those nests were checked every other day until complete. In 2004 the first 2 eggs in each of ten nests from each site were marked and collected once there were 4 or more eggs in the nest. In 2005 the largest two eggs in each of ten nests at each site were collected because studies by others did not indicate a laying effect on organic contaminant transfer (e.g., Van den Steen et al., 2006), and having the additional mass for chemical analyses was important. Whole eggs were placed in clean, individual, pre-weighed scintillation vials with chlorine-free padding and stored at 4ºC. To prepare for analysis, the eggs were allowed to warm to room temperature, opened, and the contents of eggs from the same nests were combined in clean glass vials to increase sample mass for analysis and homogenized with an Omni ES Mixer. The homogenates were sub-sampled for metals analysis and isotopic analysis. All sample aliquots were stored at -20ºC.

The nestling with the greatest mass was collected from each of the same nests from which eggs were collected. The nestlings were decapitated with a sharp scissors and the carcasses were placed in individual, clean 500 ml glass jars and frozen. The carcasses (less the excised digestive track) were removed from the freezer, homogenized using liquid argon and a blender, and returned to storage at -20ºC. At the time of analysis, the homogenates were allowed to come to room temperature and aliquots were weighed for sample preparation.

Reproductive endpoints measured included the following: Julian date (J.D.) of first egg at a nest, J.D. of first hatch at a nest, J.D. of first fledge at a nest, number of eggs laid, hatch success (calculated as # of nests with at least one egg hatching/# of nests with at least one egg at a given site [BM, IR, or PL]), fledge success (calculated as # of nests with at least one juvenile fledging/# of nests with at least one egg hatching at a given site), and nest success (calculated as # of nests with at least one juvenile fledging/# of nests with at least one egg at a given site). The latter definitions vary from common usage definitions of these metrics due to the fact that we collected two eggs and a nestling at a number of the boxes for chemical analysis. We did not employ the Mayfield method (1975), because nest boxes were checked every other day. Several nest boxes were vandalized or had other occurrences that resulted in unnatural nest failure, and these were excluded from any of the above analyses. Nests that initially failed and then had successful renests were not included in the analyses either.

Boluses were collected from 12- or 14-day-old tree swallow nestlings in 2004 and from 6- and 10-day-old nestlings in 2005. We increased the frequency to collect more boluses and adjusted the age of birds to further avoid the potential for ligatured birds to fledge. Boluses were collected using the ligature method (Quinney and Ankney, 1985; Eisler, 2000; Orians, 1966; Orians and Horn, 1969) with 89mm, black Ty-Rap© zip ties. Ligatures were left on for 30 to 45 minutes. Nests were checked for insects before ligatures were applied and again when ligatures were removed, and all insects were collected. Boluses were preserved with 95% ethanol in 2004, but were frozen in 2005. Boluses were cleaned of mucus, identified to family or lowest relevant taxa that allowed determination of trophic level, dried, and the 2005 insects were submitted for contaminant analysis. This project was approved by the University of Illinois' Institutional Animal Care and Use Committee (Protocol # 03028).

Stable isotope analysis

All samples that were tested for contaminants had small aliquots of the whole homogenized carcass removed for dual analyses of naturally-occurring carbon isotopes 12 and 13, and nitrogen isotopes 14 and 15. Additionally, sestonic algae and vegetative detritus, as well as aquatic, emergent aquatic, and terrestrial macrophytes were collected. These samples were dried at 30°C, and then pulverized. The detrital material was further treated with 95% hydrochloric acid vapors for six hours to remove carbonates. Stable isotope analyses were conducted at the University of California at Davis stable isotope facility by continuous flow Isotope Ratio Mass Spectrometer.

Analytical procedures for tissues/sediments

Trace elements measured included arsenic (As), barium (Ba), cadmium (Cd), chromium (Cr), cobalt (Co), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni), selenium (Se), silver (Ag), and zinc (Zn). Cobalt was not measured in sediments or eggs. Analytical findings for Hg will be reported separately from those for the other elements. Concentrations of 31 polychlorinated biphenyls (PCBs), 15 polybromodiphenyl ethers (PBDEs), dichloro-diphenyl-trichloroethane (DDT) and its break-down products (DDD, DDE) and ten other chlorinated pesticides were measured in tree swallow nestling carcasses, eggs, sediment and insects at the three sites. Sum PCBs were calculated from the total of all analyzed PCB congeners where more than half of the samples registered above the reporting or detection limits. The congeners measured include: 5&8, 18, 28, 31, 33, 44, 49, 52, 66, 70, 74, 77, 84&101, 95, 99, 105, 110, 118, 128, 138&163, 149, 153, 180, 183, 187, 194, 200/201 using the Ballschmiter–Zell (BZ) numbering system with international union of pure and applied chemistry (IUPAC) recommendations. Numbers joined with an ampersand were co-eluted. The PBDE congeners measured included the following: TrBDE #17, TrBDE #28, TeBDE #49, TeBDE #71, TeBDE #47, TeBDE #66, PeBDE #100, PeBDE #99, PeBDE #85, HxBDE #154, HxBDE #153, HxBDE #138, HpBDE #183, HpBDE #190, DeBDE #209. Sum PBDE values were calculated from the total of these congeners that had more than half of the sample results above the detection or reporting limits. The organochlorine pesticides measured include: alpha-chlordane, beta-chlordane, transnonachlor, alpha-hexachlorocyclohexane (a-BHC), beta-hexachlorocyclohexane (b-BHC), gamma-hexachlorocyclohexane (g-BHC or Lindane), Heptachlor epoxide, Isomer B (HPX), Oxychlordane (OXC), dieldrin, heptachlor, DDD, DDE and DDT.

For mercury and other elements, nitric acid microwave digestion procedures equivalent to USEPA Method 3051 (USEPA, 2003b) for eggs, USEPA Method 3052 (USEPA, 2003c) for nestlings and all insects, and a modified version of USEPA Method 3051 (USEPA, 2003b) using only nitric acid for sediments were used prior to analysis. In addition, quality control samples were prepared with each type of samples in each of the digestion batches. Mercury analyses of all media were conducted using atomic fluorescence. Elemental results from eggs, nestlings and insects were obtained by Inductively Coupled Plasma Mass Spectrometry (ICP-MS) using scandium, niobium, rhodium, lanthanum, and thorium as internal standards. Sediment results were obtained by ICP-MS using scandium, yttrium and thorium as internal standards. Mercury analyses of all media were conducted using atomic fluorescence. Rigorous quality control and instrument performance standards and procedures were observed (USEPA 1980).
For organics analysis, aliquots of the prepared samples were extracted using a Dionex Accelerated Solvent Extractor (ASE) following a modified version of USEPA SW-846 Method 3545, Pressurized Fluid Extraction (PFE) (USEPA, 2007). Following the ASE extraction, the samples were taken through two cleanup procedures. The first was a modified version of USEPA SW-846 Method 3640A, Gel Permeation Cleanup (USEPA, 1994), and the second followed a modified version of USEPA SW-846 Method 3630, Silica Gel Cleanup (USEPA, 1996). Two silica gel fractions were collected for instrumental analysis, the first contained the PCBs, 4,4'-DDE, Heptachlor, and a small portion of transnonachlor and the second contained the remainder of the chlorinated pesticides. Polybrominated diphenyl ether (PBDE) flame retardants were contained in both fractions.

The sample fractions were analyzed for PCBs and chlorinated pesticides using a Varian 3400 gas chromatograph (GC) equipped with an electron capture detector (GC/ECD) and a Restek Rtx®-5 Integra guard column. The column was 30 m x 0.25 mm ID with a 0.25-µm df coating of 5% diphenyl-95% dimethyl polysiloxane. The instrumental analysis followed a modification of

USEPA SW-846 Method 8081A, Organochlorine Pesticides by Gas Chromatography (USEPA, 2003d), combined with USEPA SW-846 Method 8082, Polychlorinated Biphenyls (PCBs) by Gas Chromatography (USEPA 2004a). The same fractions were then analyzed on a Varian 3800 GC with Saturn 2000 ion trap mass spectrometer using a modified version of USEPA SW-846 Method 8270C, Semivolatile Organic Compounds by Gas Chromatography/Mass Spectrometry (GC/MS) (USEPA, 2004b). The same type of GC column was used in the GC/MS analysis. The GC/MS was used to confirm the GC/ECD results, and in some cases, to differentiate and quantitate some analytes that co-eluted on the GC/ECD. Both silica gel fractions were analyzed for PBDEs using a Micromass Autospec NT High Resolution mass spectrometer equipped with a Hewlett-Packard 6890 gas chromatograph. The GC/MS was set up following a modification of USEPA Draft Method 1614, Brominated Diphenyl Ethers in Water, Soil, Sediment and Tissue by HRGC/HRMS (USEPA, 2003a). A Restek Stx®-500 (15 m x 0.25 mm ID x 0.15 µm df) column was used in the PBDE analyses. Identification and confirmation of the individual PBDE congeners was accomplished by using peak retention time and the abundance ratios of selected ion fragments.

Statistical analyses

Nesting ecology - All statistical analyses were conducted using Sigma Stat 3.1. To determine if there were significant differences among sites in date of first egg, date of first hatch, date of first fledge, and # of eggs, we used Kruskal-Wallis one-way ANOVA on ranks, because all data were ordinal rather than continuous. Julian dates (J.D.) were used for the egg, hatch and fledge data. Data from the two different years were analyzed separately. If significant differences were observed among sites, Dunn's method of pair-wise multiple comparison was used to determine which sites were different from each other. Hatch, fledge, and nest success were not statistically analyzed because they were single values for each site in each year. In addition, Spearman rank order correlation was used to determine if there was a significant relationship between # of eggs per nest and date of first egg. Data from all three sites were combined and years were analyzed separately.

To analyze nestling mass data, we considered each bird a replicate, and data from the two different years were analyzed separately. Data were tested for normality and homogeneity of variance. If the data passed normality and variance tests, a one-way ANOVA was conducted to test for differences in mean values among sites, and the Holm-Sidak test was used for post-hoc pair-wise comparisons. Non-normal data were analyzed using the Kruskal-Wallis one-way ANOVA on ranks with the Dunn's test being used for post-hoc pair-wise comparisons.

Physical habitat - Physical habitat data (% canopy cover, distance to open water, and distance to nearest woody tree) were tested for normality and homogeneity of variance. If the data passed both, a one-way ANOVA was conducted to test for differences in mean values among sites (measurements were made in 2004 only). Non-normal data were analyzed using the Kruskal-Wallis one-way ANOVA on ranks. Post-hoc pair-wise comparisons were made as described above.

Food availability - To determine availability of insects among sites in 2005 (data from 2004 were not collected in a manner appropriate for statistical analysis), we conducted a Kruskal-Wallis one-way ANOVA on ranks comparing the number of odonates collected per trap, and the number of all other emergent insects collected per trap. We also compared mean total dry mass collected per trap among the three sites using one-way ANOVA and the Holm-Sidak method for post-hoc pair-wise comparisons.

Sediment contaminants - To determine if mean contaminant concentrations in sediments were significantly different among sites, we conducted one-way ANOVAs with site (Big Marsh, Indian Ridge, or Powderhorn Lake) as the treatment. Data were first tested for normality with the Shapiro-Wilks test. If data were non-normal, a Kruskal-Wallis ANOVA on ranks was conducted. For the organics, only sum PCBs, DDD, DDE, and DDT were tested. The pesticides and PBDEs had too many values below detection limits to make comparisons meaningful, and individual congeners of PCBs were not analyzed statistically.

Egg and nestling contaminants - To determine if mean contaminant concentrations in tree swallow eggs and nestlings were significantly different between years and among sites, we conducted two-way ANOVAs with year (2004 or 2005) and site (Big Marsh, Indian Ridge, or Powderhorn Lake) as factors. Data were first tested for normality with the Shapiro-Wilks test. If data were non-normal, they were log-transformed. In the case of Mn and Co concentrations in nestling carcasses, normality and homogeneity of variance tests failed with both untransformed and log transformed data. For the reason that the values for these elements were so low, further statistical analysis was not pursued. If significant differences among years or sites were observed, the Holm-Sidak post-hoc test was used for pair-wise comparisons.

Differences among sites in PCB and PBDE congener suites were investigated for each media type using analysis of similarity (ANOSIM) on generated Euclidean distance matrices using PRIMER (V5.2.4, 2001, Primer-E Ltd). Congener suites included the same individual congeners that were used for sum values. ANOSIM analyzes multiple congeners with a single comparison. Using ANOSIM allowed us to determine site or year differences among the suite of congeners without having to create a 'sum' variable of the individual congeners.

Using correlation analysis, we compared concentrations of various PCB congeners in nestlings at a given site with those in eggs, aquatic and terrestrial bolus insects, benthic insects, and sediments. To do this analysis, we combined mean concentrations for each year to generate an overall mean concentration for each congener for each media type at each site. Congeners not present in a particular media type were excluded from analysis.

Contaminant accumulation - To obtain a rough estimate of total accumulation of various contaminants by nestlings at the three sites, we calculated the mass of a particular contaminant in eggs at a given nest (concentration in egg multiplied by 1 g for the mass of the egg) and the mass of the contaminant in the nestling from the same nest (concentration in nestling carcass multiplied by 20 g for the mass of the nestling carcass). The difference between the nestling and the egg was determined to be the total mass of contaminant accumulated. Any value that resulted in a negative value (mass in egg greater than mass in carcass) was changed to zero. We made this calculation for each nest for which we had measureable contaminant concentrations for both eggs and a nestling carcass. We conducted these calculations for Hg, sum PCBs, DDD, DDE, and sum PBDEs. To determine if there were statistical differences in total accumulation among sites, we first tested for normality and homogeneity of variance as described above. If the data passed normality, two-way ANOVA was conducted with year and site as factors, with the Holm-Sidak test used to make post-hoc pair-wise comparisons. If normality failed, log transformed data were test. If log transformed data failed normality tests, we conducted a Mann-Whitney U test on ranks to determine if there were differences between years, and then proceeded with the Kruskal-Wallis one-way ANOVA on ranks with either both years combined or separately as appropriate (determined by results of Mann-Whitney U test). Dunn's test was used in this case to make post-hoc pair-wise comparisons.

Stable isotope data - Finally, to attempt to determine potential sources of contaminants, we conducted regression analysis, comparing concentrations of elements, pesticides, and PCB and PBDE congeners in nestling carcasses with stable nitrogen and carbon isotope ratios in the same nestlings. Sites were analyzed separately and data from both years were combined. If significant correlations were observed, the relationship between the contaminant and the isotope profile in the nestling was compared to the trend for the particular isotope in aquatic or terrestrial bolus insects from the same site. For example, if birds with higher Hg concentrations also had higher nitrogen isotope ratios at a given site, we would determine if aquatic or terrestrial bolus insects at that site had a higher nitrogen isotope ratio and conclude that that food source resulted in greater Hg exposure.

RESULTS

Nesting ecology

In 2004, there were significant differences ($p < 0.05$) among sites in median date of first egg, date of first hatch and date of first fledge (Table 1). For each metric, the median value at Big Marsh was significantly earlier than that at Powderhorn, while the values at Indian Ridge were not significantly different from either of the other sites. In 2005, no significant differences existed among sites in first hatch and first fledge, but date of first egg at Indian Ridge was significantly earlier than that at Powderhorn, while the value at Big Marsh was not significantly different from that of either site (Table 1). In both years, the swallows at Big Marsh laid significantly more eggs per nest than those at Powderhorn ($p = 0.018$ in 2004 and $p = 0.024$ in 2005), with the mean number of eggs per nest at Indian Ridge being intermediate and not significantly different from either site (Table 1). Combining data from all three sites, the date of first egg was negatively correlated with the number of eggs laid per nest in both years, i.e., nests with first eggs on later dates tended to have fewer eggs ($r = -0.546$, $p < 0.001$ for 2004; $r = -0.377$, $p = 0.00266$ for 2005). However, in both years, nestlings at Big Marsh and Indian Ridge were significantly lighter in mass than those at Powderhorn (Table 1). In 2004, means for all three sites were significantly different from each other with Powderhorn having the highest mean weight, followed by Indian Ridge, and Big Marsh had the lowest mean weight. In 2005, Powderhorn again had significantly higher weights, but Big Marsh and Indian Ridge were not significantly different from each other.

There were significant differences among sites in physical habitat. Powderhorn had a greater mean percentage of area above nest boxes with a closed canopy than the other two sites ($p = 0.0180$; Big Marsh 29%, Indian Ridge 21% and Powderhorn 40%). Moreover, there was a difference among sites in the distance to open water ($p<0.0001$). Swallows at Indian Ridge Marsh had to travel approximately 45 meters further on average than the other two sites to access open water due to a ring of cattails and other emergent aquatic vegetation. There were no statistical differences among sites in mean distance to nearest woody tree taller than 2 meters.

Table 1. Median values for several tree swallow nesting ecology parameters at three wetlands in the Lake Calumet region, IL. Numbers in parentheses = ($25^{th}/75^{th}$ percentile). For nestling mass, values in parentheses are standard deviation. Different capital letters after values indicate means are significantly different ($p < 0.05$); comparisons were made for each year separately. n = sample size. J.D. = Julian Date.

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
J.D. first egg	134 (131/135)A n = 18	132 (130/135)A n = 23	137 (133/141)AB n = 14	130 (129/133) A n = 21	139 (136/143)B n = 10	134 (132/147)A n = 10
J.D. first hatch	152 (152/155)A n = 16	151 (150/156)A n = 21	155 (153/157)AB n = 11	150 (149/155)A n = 20	159 (157/169)B n = 8	151 (150/155)A n = 10
J.D. first fledge	170 (170/172)A n = 13	172 (170/174)A n = 20	173 (169/177)AB n = 10	170 (168/175)A n = 17	177 (175/187)B n = 8	171 (168/177)A n = 9
# eggs/ nest	6 (5/6)A n = 18	5 (5/6)A n = 23	6 (4/6)A n = 14	5 (5/6)A n = 21	5 (4/5)A n = 10	5 (5/5)A n = 10
hatch success	89%	100%	79%	95%	80%	100%
fledge success	81%	95%	91%	85%	100%	90%
nest success	72%	95%	71%	81%	80%	90%
nestling mass	21.33 (1.73)C n = 67	20.41 (2.40)B n = 94	22.26 (1.94)B n = 38	20.04 (2.50)B n = 77	23.24 (1.19)A n = 21	21.64 (1.72)A n = 21

Swallow diet

There were significant differences among sites in the number of emergent aquatic insects trapped in 2005 ($p < 0.0001$ for odonates, and $p < 0.0001$ for all other emergent insects), with Powderhorn having more odonates (mean = 5.6 per day versus 0.7 and 0.4 for Big Marsh and Indian Ridge, respectively) and Indian Ridge having greater number of all other insect orders (mean = 40 per day versus 17 and 22 for Big Marsh and Powderhorn, respectively). Moreover, Powderhorn had a greater total emergence trap insect mass for the 2005 season than did Indian Ridge and Big Marsh ($p = 0.009$, total weights of 2.1g, 1.8g, and 0.9g dry weight, respectively). Sample quantification was different in 2004, so we do not have comparable numbers, but the total mass collected for the year was highest at Indian Ridge, followed by Big Marsh and then Powderhorn (4.1g, 3.8g, and 2.9g dry weight, respectively). Year differences were likely due to the fact that 2004 was exceptionally wet and 2005 was especially dry, considerably reducing the size of wetland habitat by the end of the 2005 season.

Examining the composition of food boluses collected from nestlings revealed that swallows were generalist insectivores with boluses containing aquatic and terrestrial flies (Diptera), beetles (Coleoptera), bees, wasps, and ants (Hymenoptera), true bugs and aphids (Homoptera/Hemiptera), spiders (Araneida), dragonflies and damselflies (Odonata), and other minor taxa, including a zebra mussel shell. At Powderhorn, the boluses were dominated in mass by odonates, whereas at both Big Marsh and Indian Ridge, aquatic and terrestrial dipterans dominated (Table 2).

The insects comprising food boluses were further sorted according to aquatic or terrestrial origin. Based on the composition of these boluses, the swallow diets at our sites were only one half to two thirds (by dry mass) of aquatic origin (Fig. 1).

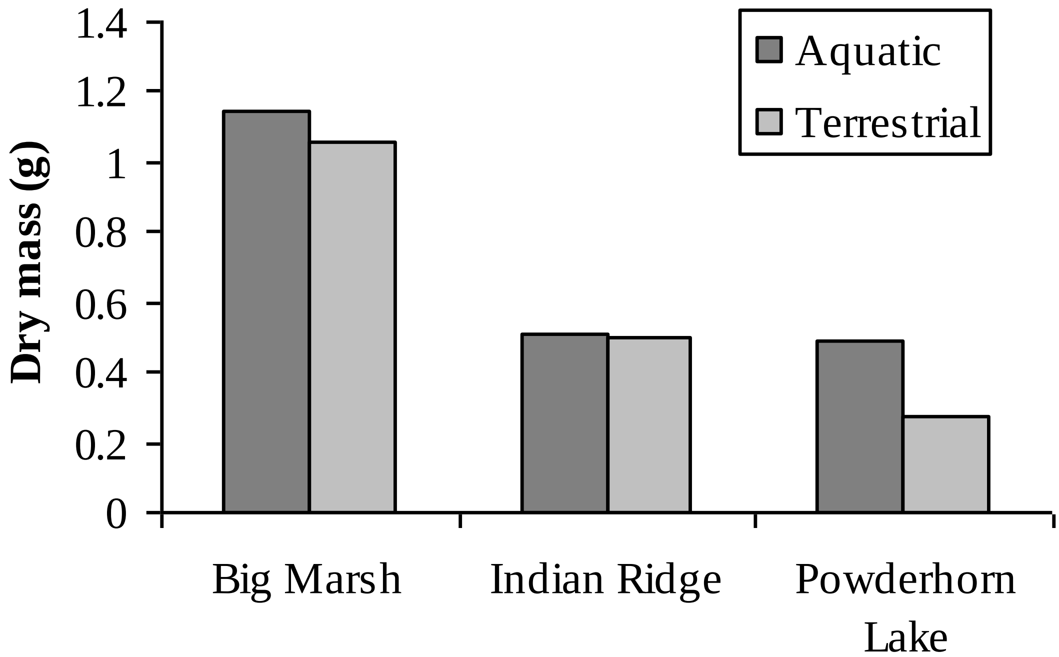


Figure 1. Total mass of aquatic or terrestrial arthropods represented in the 2005 boluses. N= 61, 26, 25 for Big Marsh, Indian Ridge and Powderhorn, respectively.

Table 2. Proportion of arthropod orders by mass, in 2005 boluses by site. n= 61, 26, 25 boluses for Big Marsh, Indian Ridge and Powderhorn, respectively. EPT = Ephemeroptera, Trichoptera and Plecoptera.

	Big Marsh		Indian Ridge		Powderhorn	
	Aquatic	Terrestrial	Aquatic	Terrestrial	Aquatic	Terrestrial
Odonata	21.3	-	8.8	-	50.1	-
Diptera	30.6	25.6	41.5	32.6	13.9	14.3
EPT	0.1	-	0.3	-	0	-
Coleoptera	0.1	4.0	0	5.8	0	3.8
Hemiptera/Homoptera	-	9.2	-	8.5	-	6.4
Hymenoptera	-	7.0	-	2.2	-	11.3
Other	0	2.1	0	0.2	0	0.3

Table 3. Proportion of individuals from various insect orders found in boluses collected from tree swallow nestlings in 2004 and 2005 from three wetlands in the Lake Calumet region, IL.

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
Number of boluses	19	61	6	26	18	25
Odonata	5.8	2.7	1.4	2.2	31.8	17.1
Diptera	24.2	51.6	12.0	67.0	27.6	47.6
Hemiptera	7.2	12.9	5.6	8.1	3.9	15.4
Coleoptera	58.3	3.0	74.6	6.1	35.0	5.6
Hymenoptera	4.4	23.3	6.0	13.9	1.4	9.1
Other	0	6.5	0.4	2.8	0.4	5.2

Sediment toxicity

In ten-day laboratory sediment toxicity tests with *Hyalella azteca* following the USEPA procedures (2000), 10-day survivorship was not significantly different among sites using the Kruskal-Wallis one-way ANOVA on ranks with means (±S.D.) of 83.8 (13.0), 91.3 (3.5), 87.5 (4.6), and 83.8 (19.2) for Big Marsh, Indian Ridge, Powderhorn, and the sand control, respectively. Change in mass between the start and the end of the experiment was evaluated as well with the Kruskal-Wallis one-way ANOVA on ranks indicating that there was a significant difference in growth among groups. Testing this difference further with Tukey's pair-wise multiple comparison procedure determined the differences to be due to the low growth of the sand control (3.6 ± 5.1 mg/individual). These differences were likely due to the fact that the sand control lacked organic matter contained in the field-collected sediments that provided additional nutrients for *H. azteca*. There were no significant growth differences among the three sediments (38.7 ± 13.3, 33.2 ± 16.9, and 48.1 ± 14.6 mg/individual for Big Marsh, Indian Ridge, and Powderhorn, respectively).

Mercury

Mean Hg concentrations in sediment grab samples from Big Marsh, Indian Ridge, and Powderhorn were 191, 83 and 102 μg/kg (dw), respectively. There were no significant differences among the sites (p=0.431), but low sample sizes make the lack of statistical difference suspect. In general, Hg concentrations in insects were about an order of magnitude lower than those in sediments, with benthic insects having values in the same range as those from boluses and emergence traps (Table 4). Because only one sample of each type (benthic Anisoptera, benthic Zygoptera, composite benthic samples, aquatic bolus insects, terrestrial bolus insects) was analyzed due to small sample mass, no meaningful comparisons can be made.

Mercury recovery in bird material was good and all samples except for two eggs had total mercury concentrations above the detection limit. Mercury in the two remaining egg samples was below the lowest concentration standard and thus below the reporting limit, so for analyses these were given values of half the reporting limit value. Mean Hg concentrations in eggs ranged from 113 to 226 μg/kg dry weight (Table 4) and were not significantly different between years ($p = 0.478$, Table 5). Two-way ANOVA indicated there were significant differences among sites in total mercury concentrations in eggs ($p < 0.001$), with both Big Marsh having the lowest concentrations, and Indian Ridge having significantly higher concentrations than Big Marsh but significantly lower concentrations than those at Powderhorn (Table 5).

Mercury concentrations in swallow nestlings were similar to those in eggs (Table 4). Two-way ANOVA indicated that there was no difference in mean concentrations between the two years ($p = 0.798$, Table 5), but that there were significant differences among sites in the combined data set ($p = 0.005$, Table 5). Nestlings from Big Marsh had significantly lower concentrations than those at Powderhorn, and those at Indian Ridge had intermediate levels, not being significantly different from the other two sites.

Table 4. Mean ± standard deviation total mercury concentrations (µg/kg dry weight) at three wetlands in the Lake Calumet region, IL. Numbers in parentheses = sample size. Benthic insect composite samples include Odonata, Diptera, Ephemeroptera. T = terrestrial, A = aquatic. nm = not measured.

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
eggs	113 ± 18 (10)	122 ± 56 (13)	210 ± 33 (10)	163 ± 38 (12)	226 ± 64 (10)	220 ± 53 (11)
nestlings	113 ± 31 (11)	202 ± 301(9)	155 ± 049(11)	107 ± 23 (10)	116 ± 23 (8)	178 ± 42 (9)
benthic Anisoptera	nm	13 (1)	nm	15 (1)	nm	40 (1)
benthic Zygoptera	nm	10 (1)	nm	11 (1)	nm	28 (1)
benthic insects (composite)	33 (1)	38 (1)	62 (1)	25 (1)	54 (1)	35 (1)
bolus insects (T)	nm	28 (1)	nm	11 (1)	nm	17 (1)
bolus insects (A)	nm	37 (1)	nm	25 (1)	nm	20 (1)
emergent insects	18 (1)	nm	42 (1)	nm	34 (1)	nm

Table 5. Two-way ANOVA results for mercury in tree swallow eggs and nestling carcasses. Different capital letters indicate site means are significantly different.

	p -value			site differences		
Medium	year	site	interaction	BM	IR	PL
Eggs	0.478	<0.001	0.243	C	B	A
Nestling carcasses	0.798	0.005	<0.001	B	AB	A

Other elements

Of the 10 priority elemental pollutants measured in sediments collected in 2004 (Ag, As, Ba, Cd, Cr, Cu, Pb, Ni, Se, Zn), seven were nominally highest in concentration at Big Marsh (Table 6). Cobalt was not measured Selenium was nominally highest at Indian Ridge and As and Cd were nominally highest at Powderhorn. However, there were no significant differences among the sites, likely due to the high variability and relatively low sample size.

In 2004, only enough emergent and benthic insect material was collected to analyze one composite sample per site for elements (Table 7). As was the case with sediments in 2004, element concentrations in emergent insects were most often nominally highest at Big Marsh (seven of 11). Barium, Cu, and Zn were the elements found in the highest concentrations in both emergent and benthic insects, while Cd and Ag were the lowest. Because only one sample of each type was analyzed, no meaningful comparisons can be made.

In 2005, three benthic insect samples were analyzed per site for elements, but this sample size was still too low for meaningful statistical analysis (Table 8). The same list of elements was analyzed and Ag was below detection in all samples. Barium, Cu, and Zn were again the elements found in the highest concentrations in benthic insects with Cd being the lowest of the detected elements.

Food boluses collected from nestlings were analyzed for 11 elements in 2005 (Table 9). Samples were first split into aquatic and terrestrial insects. For the most part, aquatic bolus insects had similar element concentrations to the benthic insect samples collected in 2005 (Table 8) with the exceptions of As, Pb, and Ni, which were approximately an order of magnitude lower in the bolus insects than they were in the benthic insects. Comparing aquatic and terrestrial bolus insects, concentrations were similar form most elements at all sites with the exception of Cd, which was an order of magnitude higher in terrestrial samples than in aquatic samples at both Big Marsh and Indian Ridge (Table 9).

In tree swallow eggs (Table 10), concentrations of As, Pb, Ni, Ag, and Cd were lower than the detection limit in over half of the samples and thus are not reported here. Cobalt was not measured. There were no significant differences among sites in the remaining elements (Ba, Cr, Cu, Se, and Zn, Table 12) and there was no apparent trend in which site had the nominally highest concentrations (Table 10). Two-way ANOVA did indicate that for Ba, Cr, and Zn there were differences in mean concentrations between the two years with 2005 being higher than 2004 for Ba and Zn and 2004 being higher than 2005 for Cr (Table 12).

In nestlings (Table 11), there were significant differences among sites in whole body Se concentration, with greater concentrations at Big Marsh compared to Indian Ridge and Powderhorn ($p = 0.001$, Table 12). Barium also was different among sites with Indian Ridge having significantly higher concentrations than the other two sites ($p = 0.019$). There were differences in mean Fe and Zn concentrations between the two years, with Fe being highest in 2005, and Zn being highest in 2004 (Table 12). The latter point is in contrast to the trend for higher Zn in eggs in 2005.

Table 6. Mean (±SD) elemental concentrations (mg/kg dry weight) in sediment from three sites in the Lake Calumet area in 2004. Range is indicated below the means. No statistical differences among sites were observed. Cobalt was not measured in sediments.

	Big Marsh	Indian Ridge	Powderhorn
Arsenic	12.63 ± 4.15	7.13 ± 4.11	15.17 ± 8.39
	(7.9 – 18.0)	(3.2 – 14.0)	(4.5 – 25.0)
Barium	176.33 ± 39.62	131.50 ± 88.69	157.33 ± 45.33
	(127 – 224)	(38 – 263)	(101 – 212)
Cadmium	2.97 ± 2.13	0.90 ± 0.80	3.70 ± 0.99
	(0.9 – 5.9)	(0.2 – 2.2)	(2.6 – 5.0)
Chromium	93.67 ± 44.50	50.55 ± 25.04	63.67 ± 27.76
	(31 – 130)	(9.2 – 76.0)	(30 – 98)
Copper	107.33 ± 20.14	72.00 ± 64.03	79.33 ± 33.09
	(79 – 124)	(15 – 179)	(38 – 119)
Lead	400.67 ± 257.05	87.00 ± 51.67	250.09 ± 49.44
	(150 – 754)	(30 – 168)	(217 – 320)
Nickel	50.00 ± 21.65	26.25 ± 14.60	31.33 ± 8.34
	(27.0 – 79.0)	(8.0 – 48.0)	(24 – 43)
Selenium	1.17 ± 0.52	1.73 ± 0.59	1.47 ± 0.70
	(0.7 – 1.9)	(1.2 – 2.7)	(0.7 – 2.4)
Silver	0.50 ± 0.29	0.29 ± 0.23	0.27 ± 0.12
	(0.2 – 0.9)	(0.1 – 0.66)	(0.1 – 0.4)
Zinc	2081 ± 2366	278.00 ± 225.53	361.67 ± 83.20
	(162 – 5415)	(45 – 538)	(255 – 458)

Table 7. Elemental concentrations (mg/kg dry weight) in aquatic insect samples from three sites in the Lake Calumet area in 2004. One sample was analyzed at each site. Nd = greater than half of the values are “not detected” or < the lowest standard.

	Big Marsh Emergent	Indian Ridge Emergent	Powderhorn Emergent	Big Marsh Benthic	Indian Ridge Benthic	Powderhorn Benthic
Arsenic	3.10	0.51	0.94	1.52	2.10	2.39
Barium	14.00	4.30	8.70	18.96	6.32	11.46
Cadmium	0.090	0.057	0.210	0.16	0.06	0.28
Chromium	2.60	0.90	1.50	2.36	1.14	1.95
Cobalt	0.52	0.10	0.13	0.60	0.31	0.22
Copper	21.00	19.00	30.00	29.78	23.44	26.70
Lead	9.10	1.50	7.90	8.27	2.39	3.93
Nickel	6.90	1.00	3.50	2.18	5.74	0.99
Selenium	1.90	1.40	1.90	1.79	1.20	1.23
Silver	0.12	0.20	0.07	nd	nd	nd
Zinc	130.00	110.00	120.00	130.17	98.10	102.66

Table 8. Mean (±SD) elemental concentrations (mg/kg dry weight) in benthic insect samples from three sites in the Lake Calumet area in 2005. Range is indicated below the means. Measurements for silver had more than half the values below the limit of detection and so are not shown. Three samples were analyzed at each site.

	Big Marsh Benthic	Indian Ridge Benthic	Powderhorn Benthic
Arsenic	2.34±1.05	1.22±0.37	3.62±1.40
	(1.30 – 3.78)	(0.83 – 1.71)	(2.15 – 5.51)
Barium	8.14±1.05	5.72±3.17	22.41±23.38
	(6.80 – 9.36)	(2.52 – 10.04)	(5.02 – 55.46)
Cadmium	0.05±0.02	0.04±0.01	0.26±0.13
	(0.03 – 0.07)	(0.03- 0.05)	(0.14 – 0.44)
Chromium	1.67±0.26	1.61±0.66	2.55±1.19
	(1.34 – 1.99)	(0.90 – 2.49)	(1.12 – 4.03)
Cobalt	0.25±0.04	0.21±0.10	0.30±0.10
	(0.20 – 0.30)	(0.12 – 0.34)	(0.19 – 0.44)
Copper	18.04±3.71	14.54±2.35	28.18±9.36
	(13.64 – 22.71)	(11.23 – 16.31)	(16.22 – 39.07)
Lead	3.49±0.36	2.64±1.75	9.74±4.58
	(2.97 – 3.79)	(1.27 – 5.11)	(4.79 – 15.84)
Nickel	1.11±0.21	1.07±0.31	1.58±0.82
	(0.92 – 1.40)	(0.74 – 1.49)	(0.76 – 2.71)
Selenium	2.57±1.72	1.42±0.38	1.37±0.16
	(1.15 – 4.98)	(0.88 – 1.74)	(1.20 – 1.58)
Zinc	86.88±11.26	82.66±15.81	93.03±12.37
	(76.56 – 102.54)	(66.88 – 104.27)	(80.85 – 110.00)

Table 9. Elemental concentrations (mg/kg dry weight) in bolus samples from three sites in the Lake Calumet area in 2005. * = value is < the lowest standard. n = 1 for all categories.

	Big Marsh		Indian Ridge		Powderhorn	
	Aquatic	Terrestrial	Aquatic	Terrestrial	Aquatic	Terrestrial
Arsenic	0.20	0.22	0.16	0.14	0.46	0.25
Barium	3.68	2.32	11.75	2.10	2.79	10.04
Cadmium	0.26	1.80	0.08	0.28	0.40	0.14
Chromium	0.52	1.05	0.60	1.08	1.73	1.01
Cobalt	*	0.06	*	*	*	*
Copper	21.40	26.56	13.64	14.22	25.05	18.52
Lead	0.49	0.85	0.52	0.75	0.62	0.58
Nickel	0.23	0.64	0.85	0.39	1.39	0.32
Selenium	1.21	1.10	2.13	0.98	0.79	1.08
Silver	*	0.20	0.24	*	*	*
Zinc	120.00	130.00	78.26	90.62	110.00	88.85

Table 10. Mean (±SD) elemental concentrations (mg/kg dry weight) in tree swallow eggs from three sites in the Lake Calumet area in 2004 and 2005. Range is indicated below the means. Values for arsenic, lead, nickel, silver and cadmium all had more than half the values below the limit of detection. Cobalt was not measured. n = number of samples analyzed.

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
n	10	13	15	12	12	11
Barium	1.60 ± 1.00	1.97 ± 0.86	1.63 ± 0.64	2.91 ± 1.06	2.39 ± 1.39	2.44 ± 1.13
	(0.6 - 3.4)	(0.8 - 3.9)	(0.6 - 2.3)	(1.5 - 4.7)	(0.8 - 4.9)	(1.2 - 5.4)
Chromium	4.06 ± 2.38	1.53 ± 0.37	4.13 ± 2.11	1.88 ± 0.90	3.29 ± 1.01	2.11 ± 1.54
	(1.6 - 10.4)	(1.1 - 2.3)	(1.6 - 6.7)	(1.3 - 4.6)	(1.6 - 4.6)	(1.2 - 6.9)
Copper	3.81 ± 1.03	4.77 ± 1.37	5.46 ± 2.44	4.12 ± 1.40	5.29 ± 2.09	5.00 ± 3.06
	(2.9 - 6.3)	(2.1 - 7.7)	(2.7 – 10.0)	(2.7 - 7.2)	(2.7 - 8.9)	(2.9 - 14.3)
Selenium	2.39 ± 0.25	2.28 ± 0.38	2.39 ± 0.34	2.19 ± 0.48	2.69 ± 0.72	2.48 ± 0.56
	(1.9 - 2.8)	(1.7 - 2.9)	(1.9 - 2.9)	(1.3 - 2.6)	(2.0 - 4.3)	(1.7 - 3.6)
Zinc	48.10 ± 12.36	58.62 ± 9.53	45.25 ± 11.73	54.42 ± 7.45	42.71 ± 16.32	56.00 ± 8.21
	(18 - 65)	(35 - 74)	(19 - 61)	(43 - 68)	(19 - 61)	(43 - 71)

Table 11. Mean (±SD) elemental concentrations (mg/kg dry weight) in tree swallow nestlings from three sites in the Lake Calumet area in 2004 and 2005. Range is indicated below the means. Values for arsenic, nickel, silver and cadmium all had more than half the values below the limit of detection. nd = greater than half of the values are ND or < the lowest standard. n = number of samples analyzed.

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
n	11	9	11	10	8	9
Barium	2.55 ± 0.71	2.61 ± 0.99	3.12 ± 1.11	4.33 ± 2.11	2.34 ± 0.83	2.66 ± 1.26
	(1.0 - 3.6)	(1.4 - 4.4)	(1.1 - 5.4)	(2.0 - 9.0)	(1.4 - 3.9)	(1.0 - 4.4)
Chromium	0.94 ± 56	1.06 ± 0.89	2.52 ± 1.81	0.86 ± 0.18	1.56 ± 0.29)	0.85 ± 0.37
	(0.4 - 2.0)	(0.6 - 3.5)	(1.6 - 7.6)	(0.6 - 1.1)	(1.2 - 2.2)	(0.4 - 1.5)
Cobalt	0.08 ± 0.01	nd	0.08 ± 0.02	nd	0.06 ± 0.01	nd
	(0.05 - 0.09)		(0.05 - 0.10)		(<.05 - 0.08)	
Copper	8.00 ± 1.77	7.49 ± 1.13	7.40 ± 1.18	6.87 ± 1.19	8.22 ± 1.28	7.68 ± 2.10
	(6.0 – 13.0)	(5.2 - 8.9)	(5.6 - 9.4)	(5.6 - 8.9)	(6.3 – 10.0)	(3.8 – 12.0)
Iron	155 ± 37	247 ± 58	134 ± 25	256 ± 41	131 ± 33	286 ±119
	(110 – 210)	(150 – 350)	(110 – 200)	(170 – 310)	(80 – 190)	(150 – 580)
Lead	0.20 ± 0.06	nd	0.35 ± 0.48	nd	0.94 ± 1.90	nd
	(0.08 - 0.25)		(0.08 - 0.38)		(0.09 - 6.00)	
Selenium	2.70 ± 0.87	2.24 ± 0.7	2.09 ± 0.39	1.75 ± 0.53	1.78 ± 0.37	1.43 ± 0.77
	(1.7 - 4.4)	(1.8 - 5.0)	(1.5 - 2.9)	(1.1 - 2.8)	(1.3 - 2.4)	(0.5 - 2.6)
Zinc	85.05 ± 14.07	69.12 ± 10.48	78.12 ± 10.66	63.07 ± 6.76	73.97 ± 8.37	66.91 ± 9.48
	(63 – 120)	(51 – 83)	(59 – 98)	(52 – 72)	(60 – 87)	(48 – 82)

Table 12. Statistically significant two-way ANOVA results for elemental contaminants in tree swallow eggs and nestling carcasses. Different capital letters indicate site means are significantly different.

	Eggs[a]					
	p -value			year differences		
Contaminant	year	site	interaction			
Ba	0.015	0.136	0.379	2005 > 2004		
Cr	<0.001	0.862	0.492	2004 > 2005		
Zn	<0.001	0.469	0.858	2005 > 2004		
	Nestling carcasses[b]					
	p -value			site differences		
Contaminant	year	site	interaction	BM	IR	PL
Ba	0.308	0.019	0.569	B	A	B
Se	0.037	0.001	0.958	A	B	B
	p -value			site differences		
Fe	<0.001	0.858	0.262	2005 > 2004		
Zn	<0.001	0.107	0.416	2004 > 2005		

[a]As, Pb, Ni, Ag, and Cd all had more than half the values below the limit of detection. No significant differences were observed for Cu, Mn, and Se. Co was not measured.
[b]As, Ni, Ag, and Cd all had more than half the values below the limit of detection. No significant differences were observed for Cu or Pb.

PCBs

Sum PCBs in sediments were highly variable (Table 13). While Indian Ridge had the highest nominal mean value, this value was strongly driven by one sample that had a sum PCB concentration of 3,224 ng/g, and no statistical differences were observed among sites. The remaining samples had concentrations less than 100 ng/g. In benthic insect samples, mean sum PCB concentrations ranged from 17 to 112 ng/g, the same approximate range of concentrations found in sediments (Table 13). Bolus insects were analyzed in 2005 only and were approximately an order of magnitude higher on average than the benthic insect samples, ranging from 138.5 ng/g at Big Marsh to 272 ng/g at Indian Ridge (Table 13).

Mean sum PCB concentrations in tree swallow eggs were substantially higher than those in insects or sediments (Table 13). Based on two-way ANOVA, there were no significant differences in mean sum PCBs between the two years, but there were differences among sites after combining the years (Table 14). Powderhorn had a significantly higher mean than Indian Ridge and Big Marsh had an intermediate mean that was not significantly different from either site. However, the interaction term was significant as well, indicating that within 2004 there were no differences among sites, but in 2005, Powderhorn had significantly higher concentrations than both Big Marsh and Indian Ridge (Table 14).

Sum PCBs in nestlings were approximately an order of magnitude lower than values in eggs (Table 13), and there was an overall significant difference between the means for the two years (Table 14). In both years, concentrations at Big Marsh were significantly higher than those at the other two sites.

We detected 22 PCB congeners in sediment samples at all three sites in 2004 (Fig. 2). Congeners 5/8, 18, 33, 77, 126, and 200/201 were below detection limits in more than half of the samples and thus were excluded from analysis. As expected, there was a lot of within-site variability in congener concentrations, and ANOSIM indicated that there were no differences among sites in the assemblages of concentration of sum PCBs ($R=-0.027$, $p=0.508$). Surprisingly, twelve of the congeners, including many of the higher molecular weight ones, were nominally highest in mean concentration at Powderhorn. Congeners with the highest mean concentrations included 84/101, 138/163, 149, and 153.

We measured PCB congeners in benthic insects in 2005 only. Ten congeners (the same listed above for sediments plus 28, 31, 66, and 194) were below detection limits in over half of the samples. Because of the mass required for sample analysis, replication was not possible. Therefore, only nominal trends are reported here. In general, concentrations in insects were higher than those observed in sediments (Fig. 3). With the exception of two congeners (74 and 105) the highest concentrations were consistently found in Big Marsh composite samples (all insects combined). This detail was in contrast to trends observed in sediments, where Powderhorn most often had the highest concentrations. As was the case with sediment samples, congeners 138/163 and 153 tended to be the highest in benthic insects.

In both 2004 and 2005, PCB concentrations in tree swallow eggs were much higher than those in insect samples, in many cases, two orders of magnitude higher (Fig. 4 a & b). Four congeners (5/8, 18, 33, and 77) were infrequently detected in 2004, and three (5/8 and 18) in 2005. Again congeners 138/163 and 153 tended to be the highest, and 84/101 concentrations were high in both years as was observed in sediments but not insects. According to ANOSIM, the suite of lipid-normalized PCB congeners in eggs were not significantly different among years ($R=0.003$, $p=0.323$), but were significantly different among sites ($R=0.062$, $p=0.003$) with both Big Marsh and Indian Ridge differing from Powderhorn ($R=0.08$, $p=0.008$; $R=0.067$, $p=0.016$, respectively). In 2005, Powderhorn had the highest concentration of most congeners.

Patterns of PCBs concentrations in nestlings were in contrast to those in eggs (Fig. 5 a & b). The suite of lipid-normalized PCBs in nestlings were significantly different between years ($R=0.425$, $p<0.001$). The suite of congeners and concentrations in nestlings were also significantly different among sites in both years (2004 $R=0.209$, $p<0.001$; 2005 $R=0.139$, $p=0.005$). In both 2004 and 2005 the suite of lipid normalized PCB congener concentrations in nestlings at Big Marsh was significantly different from that at Powderhorn ($R= 0.323$, $p= 0.001$ and $R=0.213$, $p=0.005$, respectively) with congener concentrations at Big Marsh being higher in most cases. In 2004 the suite of lipid normalized PCB congener concentrations in nestlings at Indian Ridge was also significantly different from that at Powderhorn ($R=0.262$, $p=0.001$), with concentrations at Indian Ridge being higher than those at Powderhorn.

Comparing concentrations of various PCB congeners in different media types, the strongest correlations were between nestlings and eggs, with *r*-values approaching 1.0 at all three sites (Table 15). This suggests that congener concentration patterns were very similar in nestlings and eggs. At Big Marsh, strong correlations also were observed between nestlings and aquatic bolus insects as well as benthic insects. Weaker but still significant correlations were observed between nestling profiles and those in terrestrial bolus insects and sediments. At Indian Ridge, weak relationships were observed with the only significant correlation (aside from that between nestlings and eggs) being that between nestlings and benthic insects. At Powderhorn, nestling congener profiles were strongly correlated with sediment profiles and to a lesser extent, aquatic bolus insects and benthic insects.

Table 13. Mean (± SD when available) sum PCB concentrations (ng/g dry weight) in various media types at three wetlands in the Lake Calumet region, IL, for 2004 and 2005. nm = not measured. Ben. Ins = benthic insects. Bol. Ins. = bolus insects.

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
Sediment	44.5 ± 17.0	nm	832.9 ± 1,594.3	nm	56.9 ± 57.5	nm
Ben. Ins.	nm	112.4	36.9	17.2 ± 15.2	66.7	46.0 ± 18.1
Bol. Ins.	nm	138.5 ± 50 6	nm	272.85 ± 47.3	nm	246.9
Eggs	3553.2 ± 1807.9	2729.8 ± 724.5	2753.0 ± 1681.7	2453.7 ± 1192.6	3177.5 ± 1389.5	6149.2 ± 4247.5
(max)	6,869.9	4,056.9	6,958.3	5,419.9	5,439.3	15,502.6
Birds	656.8 ± 151.6	580.4 ± 196.9	480.9 ± 100.3	336.3 ± 69.5	369.4 ± 99.4	398.3 ± 183.8
(max)	879.9	986.4	590.6	439.0	502.3	801.3

Table 14. Two-way ANOVA results for sum PCB concentrations in tree swallow eggs and nestling carcasses. Different capital letters indicate site means are significantly different.

	p -value			site differences 2004			site differences in 2005		
Medium	year	site	interaction	BM	IR	PL	BM	IR	PL
Eggs	0.354	0.026	0.022	A	A	A	B	B	A
Nestling carcasses	0.028	<0.001	0.085	A	B	B	A	B	B

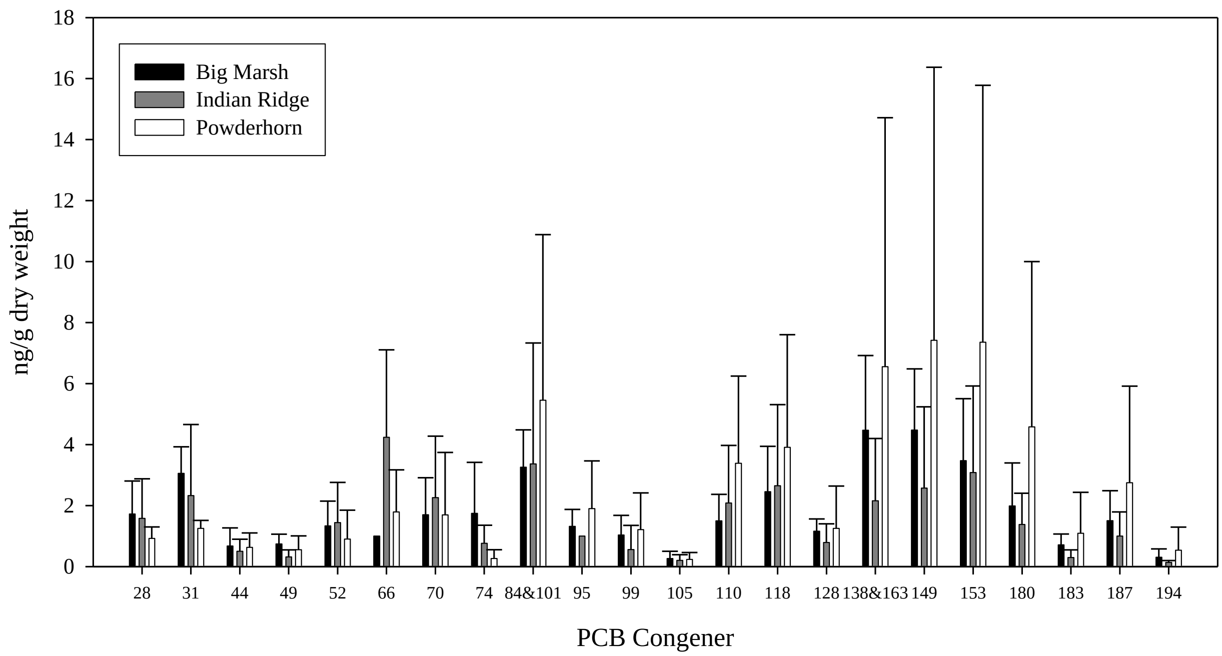


Figure 2. PCBs in 2004 Sediments. N=3, one outlier was removed from Indian Ridge. Congeners 5&8, 18, 33, 77, 126, and 200/201 were excluded as more than half the results were no-detects. Remaining non-detects and values less than the lowest standard were assigned a value of half of the detection limit or lowest standard.

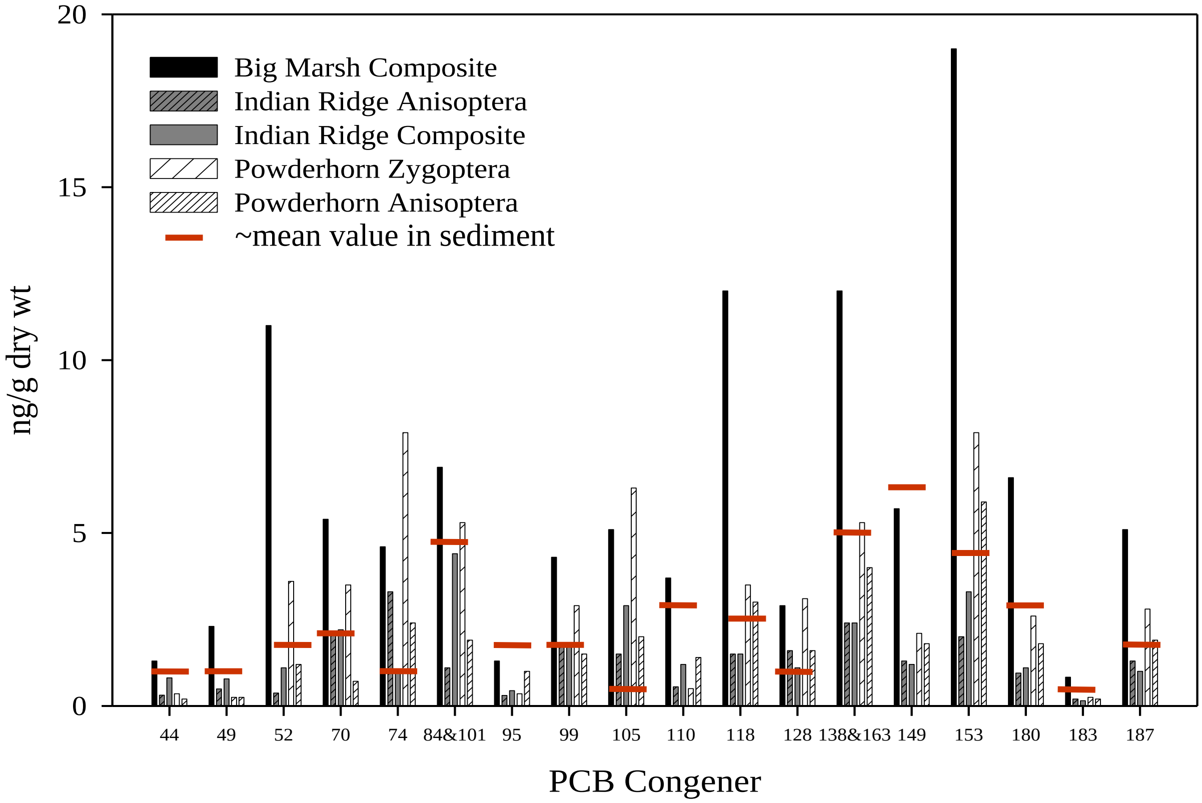
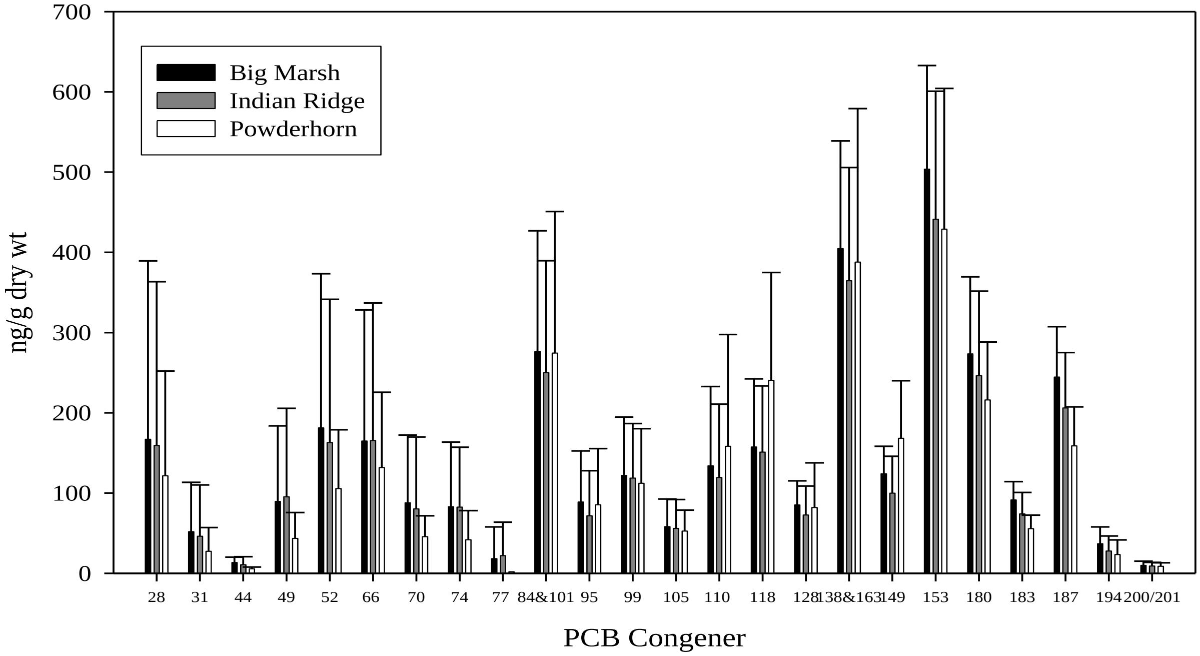


Figure 3. PCB profile of 2005 benthic insects. PCB congeners 5&8, 18, 28, 31, 33, 66, 77, 126, 194 and 200/201 are excluded as more than half the values are non-detects. Remaining non-detects and values less than the lowest standard were assigned a value of half of the detection limit or lowest standard. Mean value of sediments at all the sites is indicated for context.

a.



b.

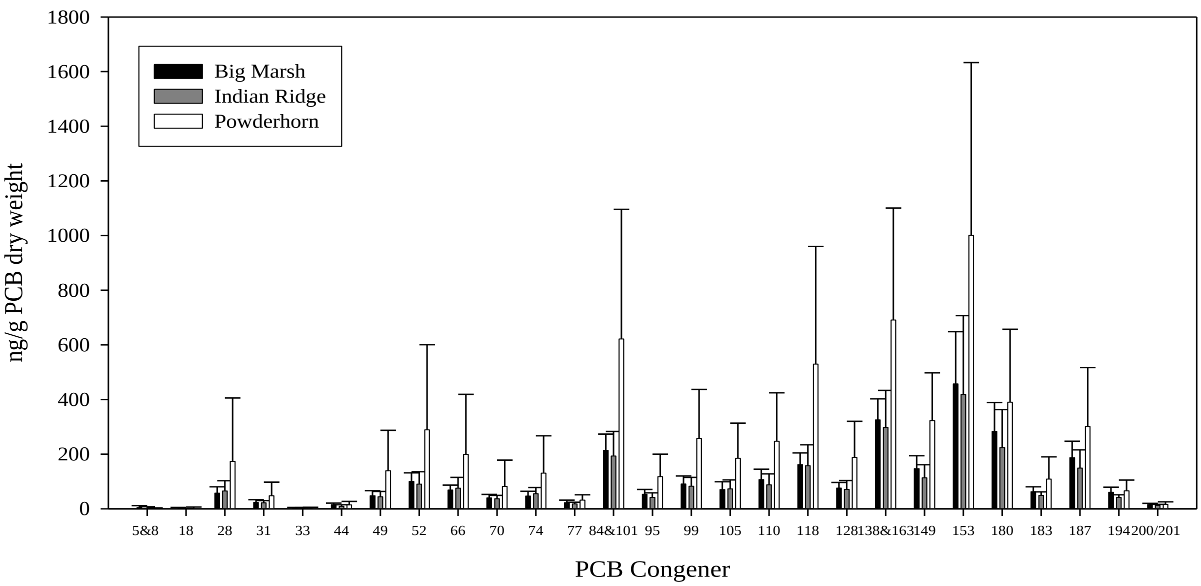
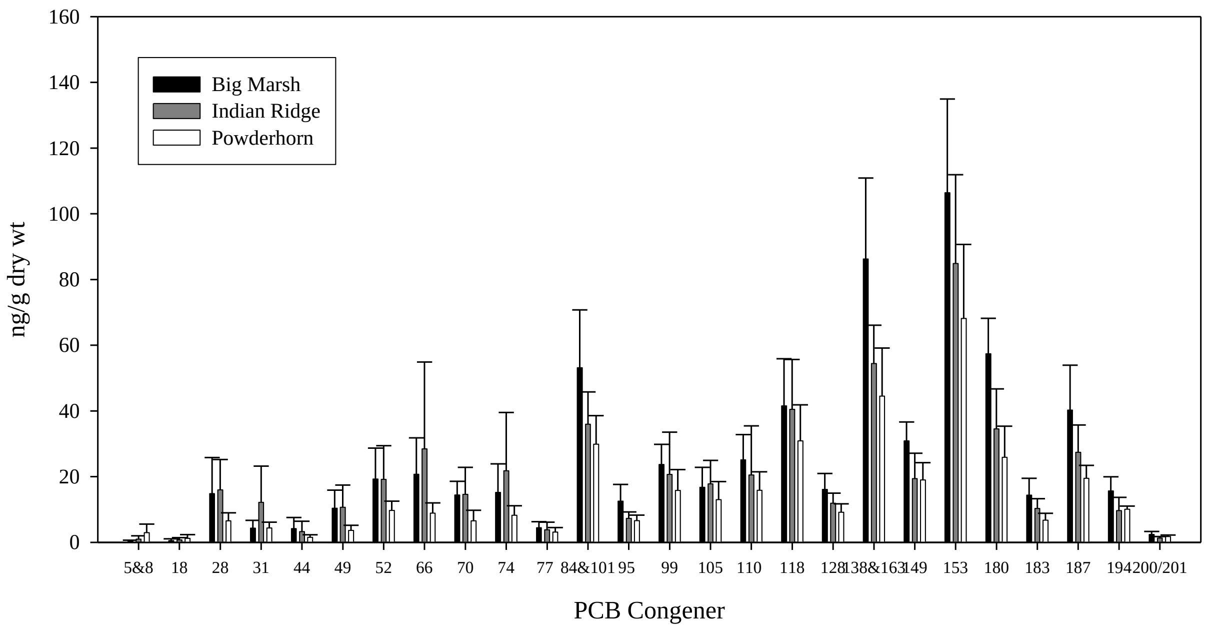


Figure 4 a & b. PCB profiles in 2004 (a) and 2005 (b) eggs. PCB congeners 5/8, 18, 33, and 77 are excluded in 2004 as the majority of the values were less than the detection limit. In 2005, congeners 5&8 (considered one congener), and 18 were not included due to majority of non-detections. Non-detects and values less than the lowest standard are given a value of ½ the detection limit or lowest standard, respectively.

a.



b.

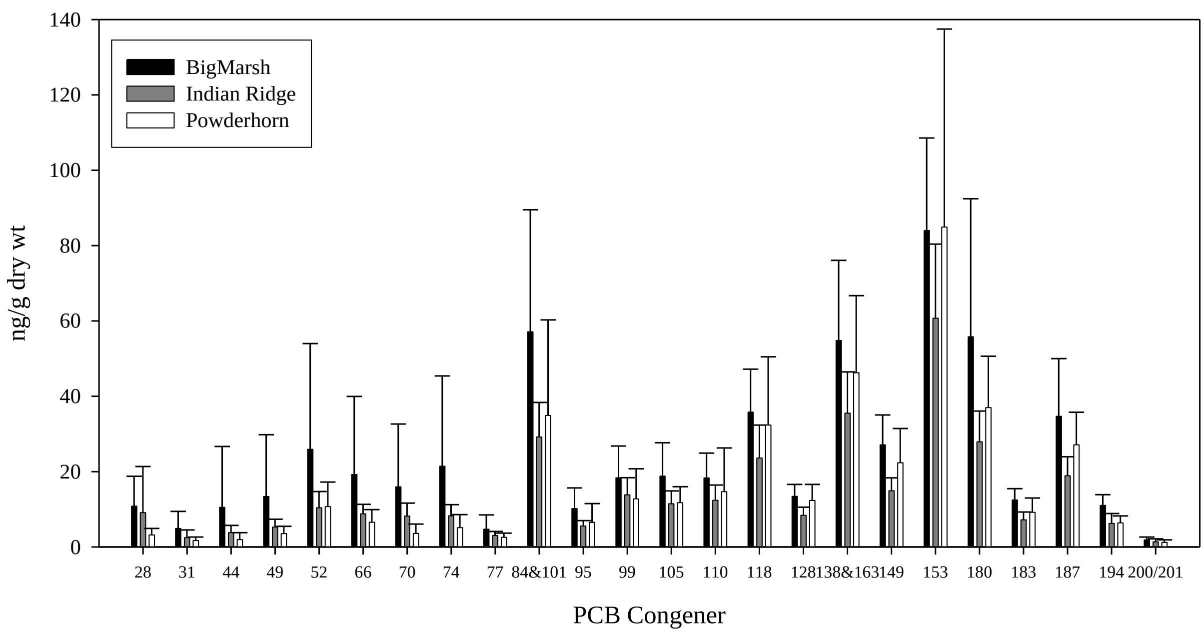


Figure 5 a & b. PCB profiles of 2004 (a) and 2005 (b) nestlings by site. In 2004, congeners 31, 33 and 126 were not tested due to majority of non-detections. Non-detects and values less than the lowest standard are given a value of ½ the detection limit or lowest standard, respectively.

Table 15. Correlation coefficients (*r*) for comparisons of PCB congener profiles in nestlings versus other media (eggs, insects, sediment) at the three study sites. Mean concentrations of each congener for both years used for these comparisons. * indicates relationship is statistically significant ($p < 0.05$). Terr = terrestrial.

Site	eggs	bolus (aquatic)	bolus (terr)	benthic insects	sediment
BM nestlings w/	0.989*	0.752*	0.611*	0.838*	0.732*
IR nestlings w/	0.980*	0.033	0.132	0.576*	0.308
PL nestlings w/	0.970*	0.521*	n/a	0.577*	0.819*

Organochlorine pesticides

Concentrations of DDT and its breakdown products, DDD and DDE, were widely variable in sediments from the three sites with ranges as large as two orders of magnitude (Table 16). No statistical differences were observed among sites in sediment concentrations. DDT was only detected at Powderhorn. DDD was detected at both Powderhorn and Indian Ridge, and DDE was found at all three sites albeit at lower concentrations than were observed for DDT and DDD.

We had sufficient emergent insect material to analyze for DDT and breakdown products on only one composite sample per site in 2004 (Table 16). Both DDD and DDE were detected at all three sites in emergent insects. In benthic insects, DDE concentrations tended to be at least an order of magnitude higher than DDD concentrations. DDD was only detected at Indian Ridge and Powderhorn. As with emergent and benthic insects, small sample sizes of bolus insects prevented meaningful comparisons among sites, but both DDD and DDE were detected in bolus samples at all three sites in 2005 (Table 16).

Both DDD and DDE were detected frequently in tree swallow eggs, while DDT was detected in less than half of the samples at all sites in 2004 and at low concentrations in 2005 (Table 17). In both years, there were wide ranges in concentrations of these pesticides at all of the sites. Two-way ANOVA indicated that there were differences between years in DDD concentrations in eggs, but the interaction between site and year was not significant (Table 18). There was a significant difference among sites in DDD concentrations when years were combined, with Big Marsh having a significantly higher mean than Indian Ridge, and Indian Ridge having a significantly higher mean than Powderhorn (Table 18). The only other significant differences among sites in organochlorine pesticides were for transnonachlor, dieldrin and HPX & OXC, all three of which were significantly higher at Powderhorn than at the other two sites (Table 17 and 18).

In general, concentrations of organochlorine pesticides were substantially lower in nestlings than they were in eggs (Table 17). Fewer significant differences among sites were observed in nestling than were seen in eggs. The mean concentrations of DDD in nestling carcasses at Big Marsh was significantly higher than that at Indian Ridge, which was significantly higher than that at Powderhorn (Table 18). No year effects were observed for DDD. There were significant differences among sites in DDE as well, with Big Marsh having a significantly higher mean nestling carcass concentration than both of the other two sites. There was a difference between the two years as well for DDE with concentrations in 2004 being higher than in 2005 (Table 18). No significant differences among years or sites were observed for any of the other organochlorine pesticides.

Table 16. Mean (±SD) organochlorine pesticide concentrations (ng/g dry weight) in select media from three sites in the Lake Calumet region, IL, in 2004 and 2005. Range is indicated in parentheses below means. Compounds with more than half of the values were non-detect or less than the lowest standard, are either excluded from table or indicated with nd. Non-detects and values below the lowest standard that are included in calculations are given the value of half the detection limit or half the lowest standard, respectively. ns indicates no sample. ^ indicates one sample lost. nm = not measured. n = sample size

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
Sediments (n)	3		4		3	
4,4'-DDD	nd (ND-13.1)	nm	189.7 ± 313.3 (7.5 – 732.4)	nm	113.1 ±127.0 (11.8 – 292.1)	nm
4,4'-DDE	6.3 ± 6.8 (ND – 15.7)	nm	13.15 ± 18.19 (1.0 – 44.6)	nm	20.6 ± 25.7 (1.0 – 56.9)	nm
4,4'-DDT	nd	nm	nd	nm	92.8±70.9 (2.6 – 175.8)	nm
Emergent insects (n)	1		1		1	
4,4'-DDD	110	nm	140	nm	67	nm
4,4'-DDE	270	nm	180	nm	170	nm
Benthic insects (n)	0	1	1	2	1	2
4,4'-DDD	ns	nd	9.0	nd	6.1	nd
4,4'-DDE	ns	160	32	28.5± 19.6 (9– 48)	17	23.0± 6.5 (16 – 29)
Bolus insects (n)		2		2		1
4,4'-DDD	nm	43^	nm	141 ± 79.5 (61 – 220)	nm	30
4,4'-DDE	nm	46 ± 31 (15 – 77)	nm	65 ± 30 (35 – 95)	nm	5.8

Table 17. Mean (±SD) organochlorine pesticides (ng/g dry weight) in select media from three sites in the Lake Calumet region, IL, in 2004 and 2005. Range is indicated below the means. Compounds with more than half of the values were non-detect or less than the lowest standard, are either excluded from table or indicated with nd. Values for a-BHC, heptachlor, and lindane all had more than half the values below the limit of detection for all media. n = sample size

	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
Eggs n =	10	11	11	10	8	11
4,4'-DDD	1182± 936	1236± 750	356± 285	620± 527	90.5± 81.6	418± 457
	(289 – 2890)	(307 – 2900)	(69 – 975)	(230 – 1900)	(15 – 293)	(41 – 1600)
4,4'-DDE	2139± 543	2355± 1021	1528± 558	1480± 689	1041± 323	2088± 1429
	(1110 – 3260)	(1100 – 4900)	(631 – 2610)	(800 – 3300)	(369 – 1620)	(290 – 4800)
4,4'-DDT	nd	7± 6	nd	7± 4	nd	12± 8
		(nd – 18)		(4.1 – 16)		(3.6 – 31)
transnonachlor	46.8± 56.73	66± 62	49.7± 28.25	62± 47	56.1± 24.58	361± 691
	(4 – 211)	(6.3 – 220)	(13 – 110)	(16 – 140)	(24 – 86)	(44 – 2400)
dieldrin	nd	40± 41	28.2± 13.45	40± 36	85.74± 48.09	194± 190
	(nd – 53)	(6.8 – 150)	(9 – 49)	(11 – 130)	(28 – 166)	(34 – 590)
HPX & OXC	112.1± 38.04	168± 101	146.5± 56.25	179± 112	214.0± 99.60	380± 263
	(68 – 197)	(36 – 350)	(62 – 235)	(68 – 380)	(48 – 392)	(160 – 1100)
Nestlings	11	9	11	10	8	9
4,4'-DDD	782± 715	753.3± 524	245± 240	221.1± 144	12.0± 11.58	145.4± 373
	(14 – 2490)	(170 – 2000)	(15 – 753)	(81 – 550)	(2.4 – 37)	(5.7 – 1200)
4,4'-DDE	582.4± 354.8	381.1± 165	258.6± 95.6	205± 59	246.1± 105.7	152.7± 52
	(137 – 1360)	(210 – 730)	(133 – 403)	(120 – 310)	(123 – 403)	(75 – 220)
4,4'-DDT	1.4± 0.6	nd	6.7± 4.0	nd	5.3± 2.12	nd
	(nd – 2.5)		(nd – 11)		(2.2 – 7.6)	
dieldrin	11.09± 3.4	66.7± 125	114.0± 194.5	19.3± 6	22.8± 6.5	43.9± 24
	(7.1 – 16)	(10 – 420)	(nd – 532)	(12 – 30)	(12 – 32)	(18 – 84)
transnonachlor	6.3± 1.9	56.4± 129	24.8± 36.9	12.4± 6	14.6± 4.7	61.5± 127
	(3.1 – 10)	(4.9 – 420)	(2.7 – 114)	(4.5 – 25)	(7 – 21)	(8.9 – 420)

Table 18. Significant two-way ANOVA results for organochlorine pesticide concentrations in tree swallow eggs and nestling carcasses. Different capital letters indicate site means are significantly different.

Eggs

	p -value			site differences		
Contaminant	year	site	interaction	BM	IR	PL
DDD	0.003	<0.001	0.105	A	B	C
t-nonachlor	0.022	0.008	0.270	B	B	A
dieldrin	<0.001	<0.001	0.008	C	B	A
HPX + OXC	0.024	<0.001	0.515	B	B	A

	p -value			year differences
Contaminant	year	site	interaction	
DDD	0.003	<0.001	0.105	2005>2004
t-nonachlor	0.002	0.008	0.270	2005>2004
dieldrin	<0.001	<0.001	0.008	2005>2004
HPX + OXC	0.024	<0.001	0.515	2005>2004

Nestling carcasses

	p -value			site differences		
Contaminant	year	site	interaction	BM	IR	PL
DDD	0.129	<0.001	0.697	A	B	C
DDE	0.009	<0.001	0.688	A	B	B

	p -value			year differences
Contaminant	year	site	interaction	
DDE	0.009	<0.001	0.688	2004>2005

PBDEs

Sum PBDEs in sediments were quite low as a high percentage of the congeners were below detection limits (Table 19). Mean concentrations in sediments ranged from 4.8 to 32.4 ng/g, with the highest average found at Powderhorn; however, there were no significant differences in mean sum PBDE concentrations among sites. PBDE 209 was below detection limits in all but two samples, but those samples had relatively high concentrations: 23 ng/g at Big Marsh and 82 ng/g at Powderhorn.

In benthic insect samples, mean sum PBDE concentrations were in the same general range as those in sediments and ranged from 11 to 51 ng/g (Table 19). Benthic insect samples were analyzed for PBDEs after the samples were evaluated for elements, PCBs and organochlorine pesticides. Measuring PBDEs after all the other analyses left fewer samples that could be run. Only PBDEs 47, 99, and 183 were detected in all benthic insect samples, with mean (maximum) values of 3.3 (7.8), 3.0 (8.0), and 2.0 (7.5) ng/g dry weight, respectively. Half of the samples had concentrations of PBDEs 17, 28, 49, 71, 154, 138, 190, and 209 that were lower than the reporting limit or were non-detects. PBDEs 66, 85, 100, and 153 were reported more frequently.

As was the case with sum PCBs, sum PBDEs in bolus insects were approximately an order of magnitude higher on average than the benthic insect samples, with the exception of the samples from Big Marsh, which were only slightly higher than the benthic insect samples at that site (Table 19). Only PBDEs 71, 47, and 99 were found above detection limits in more than half of the samples.

Mean sum PBDE concentrations in tree swallow eggs were of approximately the same order of magnitude as those in bolus insects (Table 19). There were no significant differences in mean concentrations of sum PBDEs in eggs between years or among sites. The highest single value was 1,227 ng/g at Indian Ridge in 2004. We analyzed tree swallow eggs for 15 different PBDE congeners and had frequent detections for 12 of them in both years (Fig. 6 a & b). Mean concentrations of congeners detected ranged from less than 10 to nearly 200 ng/g dry weight. In both years, the highest concentrations observed were for TeBDE47 and PeBDE99, which were at least an order of magnitude more concentrated than the remaining congeners. According to ANOSIM, there were no significant differences between years ($R=0.024$, $p=0.100$) or among sites ($R=0.010$, $p=0.253$) in the concentrations of the suite of PBDE congeners (Fig. 6 a & b).

Mean sum PBDEs in nestlings were lower than values in eggs (Table 19). Two-way ANOVA indicated that there was a significant difference between the two years with means in 2005 being higher than those in 2004 (Table 20). When accounting for differences between years, there were significant differences among sites as well, with both Big Marsh and Indian Ridge having significantly higher mean concentrations than those in Powderhorn. In 2004, only nine of the fifteen congeners were found with frequency (Fig. 7a). In 2005, 14 of the 15 congeners were found frequently (Fig. 7b), including 209, which has been detected only rarely in environmental samples. According to ANOSIM, the suite of PBDE congener concentrations in nestling carcasses was significantly different between the two years ($R=0.122$, $p<0.001$) and the suite of PBDEs had significantly different concentration profiles among sites in both 2004 and 2005 ($R=0.152$, $p=0.007$; $R=0.243$, $p<0.001$ respectively). In 2004 both Big Marsh and Indian Ridge nestlings had significantly different suites of PBDE congener concentrations than those at Powderhorn ($R=0.208$, $p=0.021$ and $R=0.208$, $p= 0.015$, respectively), and again in 2005 ($R=0.470$, $p<0.001$ and $R= 0.315$, $p=0.003$, respectively).

Examining differences in detection rates between egg and nestlings may provide a picture of what congeners are available at the sites, assuming that egg burdens were accumulated by mothers elsewhere. In both years, TeBDE 71 was detected in nestlings but not in eggs, suggesting that this congener was locally accumulated. This fact was also true of De209 in 2005.

Table 19. Mean (± SD when available) sum PBDE concentrations (ng/g dry weight) in various media types at three wetlands in the Lake Calumet region, IL, for 2004 and 2005. nm = not measured. Ben. Ins = benthic insects. Bol. Ins. = bolus insects.

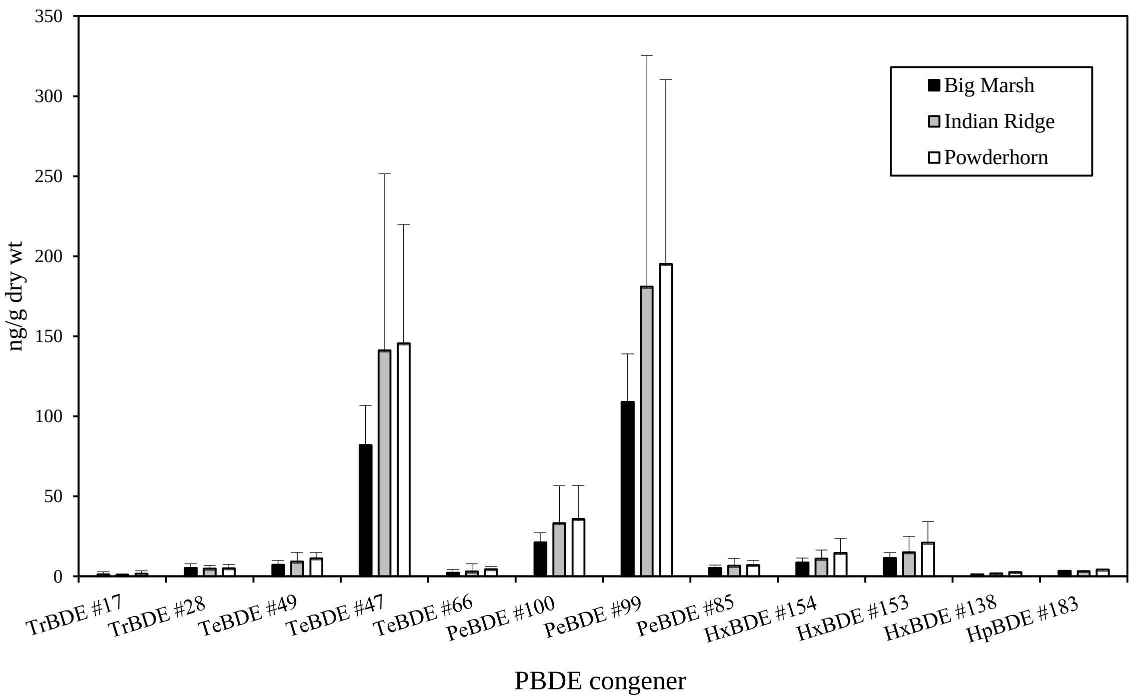
	Big Marsh		Indian Ridge		Powderhorn	
	2004	2005	2004	2005	2004	2005
Sediment	11.6 ± 10.6	nm	4.8 ± 0.2	nm	32.39 ± 37.9	nm
Ben. Ins.	nm	51.7	19.0	11.3 ± 11.2	45.9	21.3 ± 3.8
Bol. Ins.	nm	68.0 ± 1.6	nm	148.5 ± 5.5	nm	139.8
Eggs	255.1 ± 69.5	307.8 ± 241.4	427.1 ± 327.6	361.3 ± 273.4	448.4 ± 257.7	334.9 ± 144.9
(max)	387.6	903.9	1,227.03	930.1	1017.2	591.4
Nestling carcasses	66.8 ± 16.2	162.7 ± 79.8	84.7 ± 38.2	199.9 ± 207.2	65.2 ± 33.3	58.9 ± 37.5
(max)	97.4	322.3	175.4	778.1	147.1	132.3

Table 20. Two-way ANOVA results for sum PBDE concentrations in tree swallow nestling carcasses. Different capital letters indicate site means are significantly different.

	p -value			site differences		
Medium	year	site	interaction	BM	IR	PL
Nestling carcasses	0.005	<0.001	0.214	A	A	B

	p -value			year differences
Medium	year	site	interaction	
Nestling carcasses	0.005	<0.001	0.214	2005>2004

a.



b.

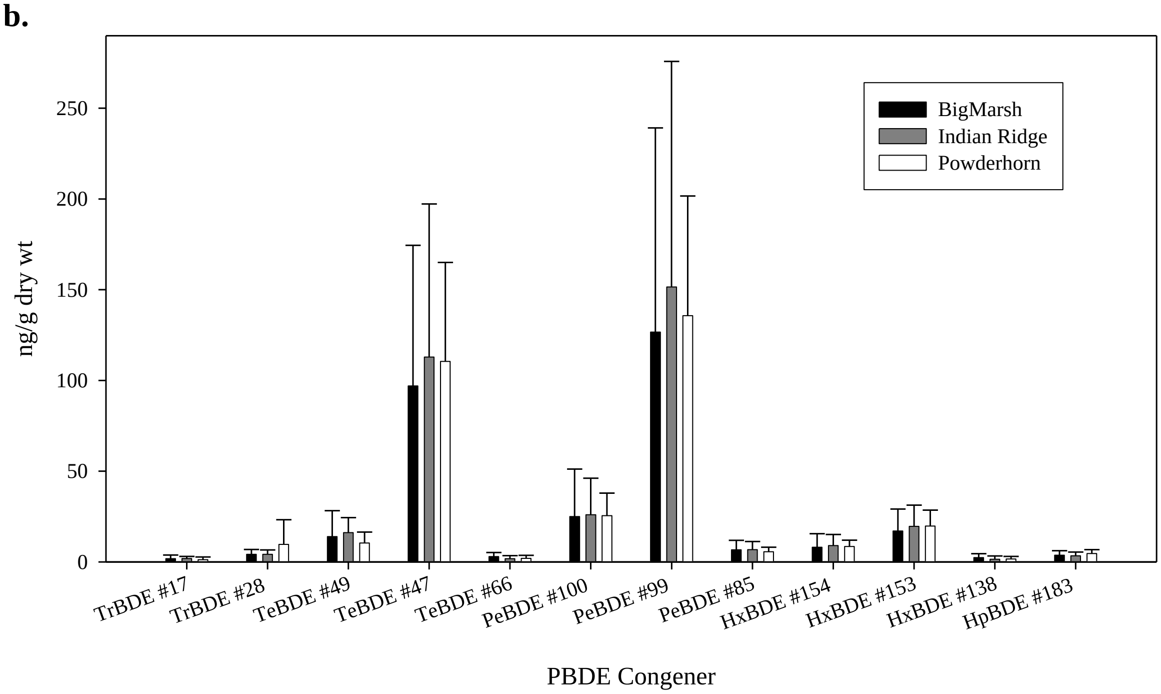
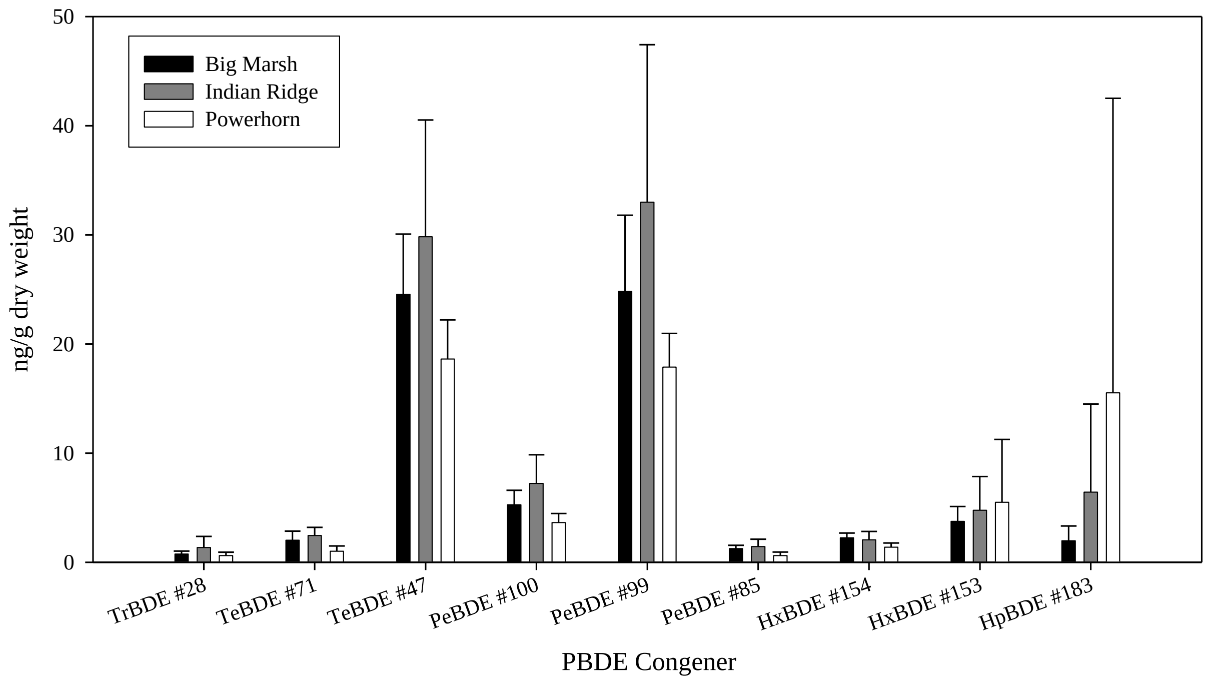


Figure 6 a & b. Egg PBDE profiles for twelve of fifteen congeners tested in 2004 (a) and 2005 (b). Congeners were excluded from analyses if the majority of values were either non-detects values or “less than” values. Non-detects that are included in analyses were given the value of ½ the detection limit and ‘less than’ numbers were valued at ½ the specific ‘less than’ number. Error bars are standard deviation.

a.



b.

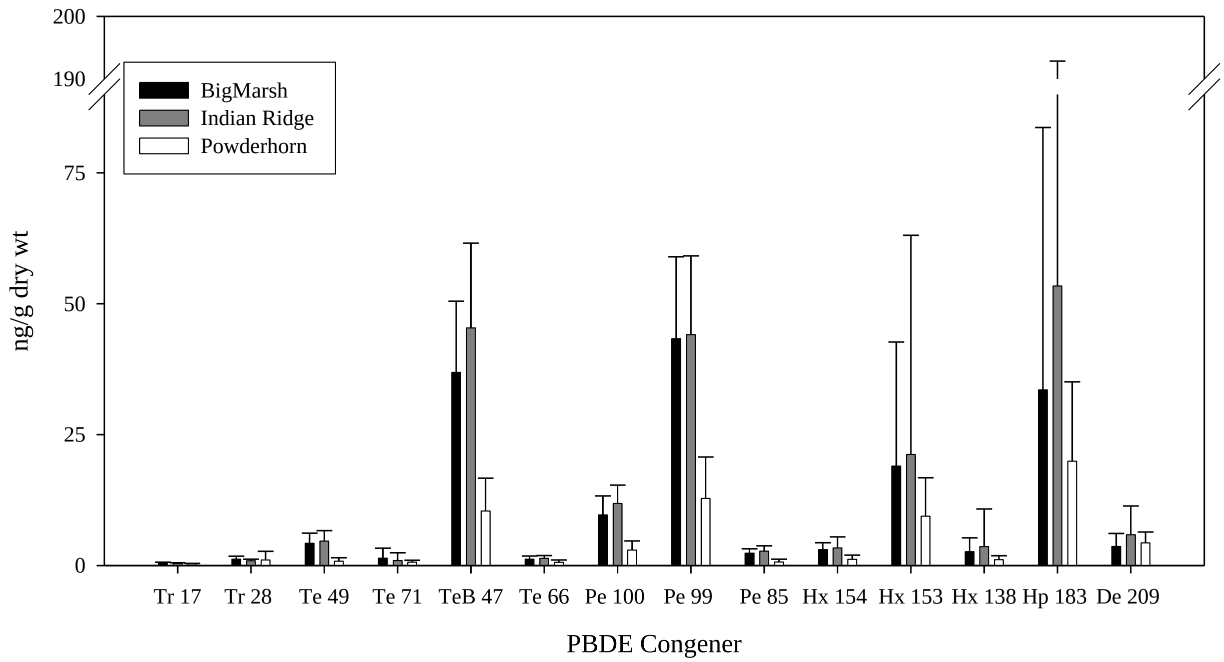


Figure 7 a & b. Nestling PBDE profiles for nine (2004 (a)) and 14 (2005 (b)) of 15 congeners tested. Congeners were excluded from analyses if the majority of values were either non-detects or 'less than' values. Non-detects that are included in analyses were given the value of ½ the detection limit and 'less than' numbers were valued at ½ the specific 'less than' number. Error bars are standard deviation. The error bar for Hp 183 at Indian Ridge ends at approximately 190 ng/g. HpBDE is not included as it was only detected in three birds.

Accumulation of selected contaminants

Comparing total mass accumulation of selected contaminants by normalizing concentrations in eggs and nestling carcasses by mass, there were a number of significant differences among sites (Table 21). Swallows accumulated the least Hg at Powderhorn in 2004, but the most in 2005. For sum PCBS, there were no differences between years, and swallows at Big Marsh accumulated significantly greater mass that at either of the other two sites. Big Marsh and Indian Ridge both accumulated more DDD than Powderhorn. For DDE, there were no differences among sites in 2004, but Big Marsh birds accumulated significantly more DDE than Powderhorn birds in 2005. Finally for sum PBDEs, Big Marsh birds accumulated significantly greater mass than Powderhorn birds in both years, and in 2005, Indian Ridge birds also accumulated significantly more PBDEs. To summarize, with the exception of Hg in 2005, birds at Big Marsh accumulated the greatest mass of each of the selected contaminants below, while in most cases, Powderhorn birds accumulated the least.

Table 21. Comparison of mean (s.d.) mass (ng for all) of various contaminants accumulated by tree swallow nestlings. If years are shown separately, a significant difference between years was observed. Different capital letters after means indicate means are significantly different ($p < 0.05$)

		Mercury	
Year	Big Marsh	Indian Ridge	Powderhorn
2004	2225 (542) A n = 10	2469 (728) A n = 7	1921 (278) B n = 6
2005	1924 (708) B n = 8	1971 (450) B n = 10	3340 (821) A n = 9
		sum PCBs	
	Big Marsh	Indian Ridge	Powderhorn
Both years	9217 (3102) A n = 19	5383 (2107) B n = 17	3499 (2942) B n = 16
		DDD	
	Big Marsh	Indian Ridge	Powderhorn
Both years	13638 (119710) A n = 19	4078 (3656) A n = 17	107 (177) B n = 15
		DDE	
	Big Marsh	Indian Ridge	Powderhorn
2004	8850 (6569) A n = 10	3400 (1984) A n = 8	3788 (1901) A n = 7
2005	5067 (3171) A n = 9	2589 (1411) AB n = 9	1234 (1289) B n = 9
		sum PBDEs	
	Big Marsh	Indian Ridge	Powderhorn
2004	1031 (262) A n = 10	1165 (850) AB n = 9	587 (281) B n = 7
2004	2856 (1616) A n = 8	3874 (4333) A n = 9	938 (627) B n = 8

Stable isotope analysis

Stable isotope profiles in various media from the three sites are provided in Table 22. Comparisons of mean isotope ratios among sites are meaningless because they are a function of the location from which food is obtained. For example the fact that nestlings at Big Marsh and Indian Ridge had higher $\delta^{15}N$ values than those at Powderhorn does not necessarily indicate that those swallows fed at a higher trophic position; the difference may just be a function of Big Marsh and Indian Ridge having anthropogenic nitrogen inputs (e.g., sewage) which would increase the baseline level of $\delta^{15}N$. The values for the various media are shown to provide a basis of comparison for the correlation analyses.

We compared stable isotope profiles of individual birds at a given site with contaminant burdens using regression analysis and generated a number of significant correlations (Table 23). At Big Marsh, nearly all of the significant correlations were suggestive that increased contaminant burdens were associated with profiles reflective of consumption of terrestrial insects. The single exception to this was DDE which was negatively correlated with $\delta^{13}C$, suggesting an aquatic source. A similar pattern was observed at Indian Ridge, where several PCBs, chlordane, dieldrin, and PBDE 28 were negatively correlated with either $\delta^{13}C$ or $\delta^{15}N$, suggesting terrestrial sources. The exception there was for PBDE 47 which was positively correlated with $\delta^{15}N$, suggesting an aquatic source. At Powderhorn, only mercury was associated with aquatic sources whereas a number of PCBs and PBDEs were associated with terrestrial diets.

Table 22. Mean (± S.D. when available) stable carbon and nitrogen profiles for eggs, nestlings and bolus insects at the three study sites. Values in parentheses = sample size.

		Big Marsh		
Isotope ($^{o}/_{oo}$)	eggs (20)	nestlings (20)	bolus (aquatic)	bolus (terrestrial)
$\delta^{15}N$	13.19 ± 0.87	15.16 ± 1.78	13.65	14.60
$\delta^{13}C$	-27.36 ± 1.08	-25.86 ± 0.85	-27.65	-25.77
		Indian Ridge		
Site	eggs (18)	nestlings (20)	bolus (aquatic)	bolus (terrestrial)
$\delta^{15}N$	11.85 ± 1.20	16.10 ± 1.28	20.46	8.64
$\delta^{13}C$	-26.15 ± 0.59	-24.72 ± 0.85	-25.07	-25.72
		Powderhorn		
Site	eggs (20)	nestlings (17)	bolus (aquatic)	bolus (terrestrial)
$\delta^{15}N$	9.53 ± 0.87	8.61 ± 0.24	4.48	5.44
$\delta^{13}C$	-26.29 ± 1.24	-24.46 ± 0.66	-26.06	-25.65

Table 23. Summary of relationships between stable isotope profiles and contaminant concentrations in tree swallow nestlings. Only significant relationships ($p < 0.05$) are included. The "correlation" column indicates the direction of the correlation between the given contaminant and $\delta^{15}N$ or $\delta^{13}C$ and the "associated w/" column indicates whether aquatic or terrestrial bolus insects were positively correlated with contaminant concentrations (based on their isotope profiles)

Site	correlation	associated w/
Big Marsh		
PCBS 44, 52 and 105	higher $\delta^{15}N$, less negative $\delta^{13}C$	terrestrial
PCB 70	less negative $\delta^{13}C$	terrestrial
PCB 194	lower $\delta^{15}N$	aquatic
DDE	more negative $\delta^{13}C$	aquatic
PBDE 49, 47, 85, 99, 100	higher $\delta^{15}N$	terrestrial
Indian Ridge		
PCB 52	lower $\delta^{15}N$	terrestrial
PCB 77	lower $\delta^{15}N$, more negative $\delta^{13}C$	terrestrial
PCB 95, 153	more negative $\delta^{13}C$	terrestrial
Chlordane and Dieldrin	more negative $\delta^{13}C$	terrestrial
PBDE 28	more negative $\delta^{13}C$	terrestrial
PBDE 47	higher $\delta^{15}N$	aquatic
Powderhorn		
Hg	lower $\delta^{15}N$	aquatic
PCB 31, 70, 95, 99, 110, 194, 201	higher $\delta^{15}N$	terrestrial
PBDE 71, 47, 99, 85, 154	less negative $\delta^{13}C$	terrestrial

DISCUSSION

Nesting ecology

Although there were differences among sites in dates of first egg laying and as a result, date of first hatch and fledge, there did not appear to be strong differences in hatch, fledge, or nest success. The differences in egg laying and hatching dates were only present in 2004 and were likely related to habitat preferences and not contamination at the sites. Later arriving birds were left with less favorable habitat, which at Powderhorn meant a more closed canopy. At Indian Ridge, swallows had to travel farther from nest boxes to get to open water, but they still selected this site earlier than Powderhorn. The slightly (but not significantly) reduced number of eggs per nest at Powderhorn relative to Big Marsh in 2004 may have been related to the later mean date of nest completion, as has been shown by others (Stutchbury and Robertson, 1988). In addition, the greater mean nestling mass at Powderhorn may have been related to clutch size as well, but higher contaminant accumulation at Big Marsh and Indian Ridge may have also played a role.

Swallow diet

We observed differences among sites in the number and mass of insects caught in emergence traps, suggesting a difference in insect availability. Powderhorn had more odonates and Indian Ridge had more beetles and flies. There were also differences among sites in the composition of boluses, again with Powderhorn tending to have more odonates and the other two sites being dominated (in mass) by flies. However, the emergence trap data probably do not adequately describe food availability at the tree sites because, as our bolus composition results indicate, approximately 40 to 50 percent of the diet (by mass) at all three sites consisted of terrestrial insects. Others have observed a wide range of relative proportions of aquatic and terrestrial insects in tree swallow boluses; for example, Menglekoch et al. (2004) found samples to be 95% aquatic, while Johnson and Lombardo (2000) found boluses to be 85% terrestrial by count. Clearly these relative proportions are site specific and may have important implications for contaminant uptake. This hypothesis will be discussed further.

Sediment toxicity

In 10-d laboratory toxicity bioassays with sediments collected from the three sites and a sand control, survivorship of the amphipod *Hyalella* was not significantly different among treatments, and growth in all three field collected sediments was greater than in the sand control. Ingersoll et al. (2002) conducted similar bioassays using *Hyalella* with sediments collected from 30 different sites within the Grand Calumet River and the Indiana Harbor Canal and found that survivorship was significantly reduced in 15 of the 30 samples. These toxic samples were dominated by PAHs and PCBs, and most of the samples had total PCB concentrations that were an order of magnitude higher than the highest average sum PCB value in sediments from this study (Indian Ridge). In addition, many others had high PAH concentrations (Ingersoll et al., 2002). The disparity between our sediment toxicity results and those of Ingersoll et al. (2002) indicates that some areas of the Lake Calumet region are far more severely polluted than others and that the sediments from the wetlands pose a far lower risk to sediment dwelling invertebrates than do those from the Indiana Harbor and the Grand Calumet River.

Mercury

A number of recent studies have documented mercury levels in swallow eggs, livers, and nestlings at various sites in North America (C. M. Custer et al., 2006 and 2007; T. W. Custer et al., 2006 and 2008; Brasso and Cristol, 2008). In most cases (e.g., in North Dakota, Louisiana [barn swallows in this case], and Minnesota), concentrations in eggs were in the range of 200 μg/kg d.w. (C. M. Custer et al., 2006; T. W. Custer et al., 2006 and 2008); however, in Nevada, Custer et al. (2007) obtained a geometric mean of 7,340 μg/kg. None of the studies by C. M. Custer et al. (2006 and 2007) and T. W. Custer et al. (2006 and 2008) were able to find a relationship between egg mercury loads and hatching success or other toxic effects. In our study, the highest mean mercury concentration we observed (226 μg/kg) was at the low end of the range documented in these previous studies. We found no differences in hatching success among sites. These same previous studies also measured mercury in livers of nestling tree swallows, and with the exception of the site in Nevada (liver concentration = 3,790 μg/kg d.w.) that had the highest egg mercury concentration (Custer et al., 2007), concentrations typically ranged from 140 to 240 μg/kg d.w. (C. M. Custer et al., 2006; T. W. Custer et al., 2006 and 2008). Three of our mean values for total body mercury concentrations in nestlings fell into the above stated range for mercury in livers (Indian Ridge in 2004, and Powderhorn and Big Marsh in 2005); the other mean values were below this range. In these previous studies and ours, no apparent toxic effects were associated with these mercury loads. In contrast to these other studies, toxic effects of mercury exposure were observed in swallows inhabiting the headwaters of the Shenandoah River in Virginia (Brasso and Cristol, 2008). Colonies associated with blood mercury concentrations in the range of 2 to 4 ppm produced fewer nestlings relative to those at reference areas. Two of these previous studies also measured mercury in tree swallow diets; insects in North Dakota had concentrations within the range we observed (47 μg/kg, Custer et al., 2008), while the site in Nevada had higher concentrations than were observed in our study (1,170 μg/kg, Custer et al., 2007).

Comparing mean masses of Hg accumulated by nestlings at the tree sites suggests that there were differences among the sites, but the differences were not consistent in both years. In 2004 Powderhorn nestlings accumulated the least Hg, whereas in 2005, they accumulated the most. Birds at Powderhorn accumulated the lowest levels of all of the other contaminants analyzed in this way. There were no significant differences among sites in Hg in sediments (although sample sizes were low), and stable isotope analysis suggested that at Powderhorn, Hg in nestlings were indicative of an aquatic source. Powderhorn emergence traps and aquatic bolus samples were dominated by odonates, which are large predatory insects. Perhaps these were the source of Hg at Powderhorn.

Metals and other elements

Aside from mercury, only five elements (Ba, Cr, Cu, Se, and Zn) were consistently detected in tree swallow eggs at our sites. The Ba concentrations in eggs at our sites (means of 1.6 to 2.9 mg/kg) were generally lower than those observed in other studies (2.8 – 5.6 mg/kg, C. M. Custer et al., 2006 and 2007; T. W. Custer et al., 2006 and 2008), while our Cr concentrations were higher than those in previous studies (none of which detected this element). Our mean Cu concentrations also were on average higher than those in the studies by C. M. Custer et al. (2006 and 2007) and T. W. Custer et al. (2006 and 2008), while our Se and Zn concentrations fit within the same range as previously documented. No toxic effects were observed in these studies (C. M. Custer et al., 2006 and 2007; T. W. Custer et al., 2006 and 2008). We also detected Pb in nestlings albeit, only in 2004, whereas only one of the previous studies detected Pb in livers. As was observed in the eggs, Se and Zn concentrations in nestlings in our study were similar to those previously reported, with concentrations of Se being highest in nestlings from Big Marsh.

PCBs

Numerous studies have documented total PCB concentrations in tree swallow eggs and nestlings at various sites throughout eastern North America. Many of these studies report sum PCB concentrations in terms of wet weight, while ours are reported in dry weight. In order to make comparisons, we used conversion factors for eggs and nestlings from our study to estimate dry weight concentrations from reported data, and all further concentrations stated are in terms of dry weight. Reported egg total PCB concentrations in the literature range from 627 to 168,150 ng/g, with the highest concentrations being reported in the Hudson and St. Lawrence River drainages in New York (Bishop et al., 1999; Secord et al., 1999) and in barn swallows in Louisiana (Custer et al., 2006). The mean sum PCB concentrations in eggs from our study, ranging from 2,453 to 6,149 ng/g, fall into the lower 50th percentile of reported values and are comparable to concentrations reported in the Wisconsin River, WI (Custer et al., 2002); the Saginaw River, MI (Nichols et al., 1995); and other sites in the Great Lakes and St. Lawrence River drainages (Bishop et al., 1999). Most of these studies either did not measure or did not observe effects on hatching success, although Custer et al. (2003) determined that hatching success decreased when sum PCB concentrations in "pippers" (eggs or newly hatched nestlings) approached approximately 100,000 ng/g. As expected, based on our sum PCB concentrations, we saw no effects on hatching success in our study.

Reported sum PCB concentrations in tree swallow nestlings range from 132 to 205,260 ng/g (d.w.), with the highest values observed at the Hudson and St. Lawrence Rivers (Bishop et al., 1999; Secord et al., 1999); Crab Orchard National Wildlife Refuge in southern Illinois (Maul et al., 2006); and the Housatonic River in Massachusetts (Custer et al., 2003). Our mean values ranged from 336 to 656 ng/g, falling into the lower 25th percentile of reported values. In fact, our concentrations were comparable to those reported for "reference" sites in Ontario (Bishop et al., 1999); the Fox River, WI (Custer et al., 1998); and the Saginaw River, MI (Nichols et al., 1995). As was the case with Hg, the eggs at Powderhorn (with the latest mean nest initiation) tended to have higher sum PCBs than eggs from the other two sites. Based on accumulation calculations, we estimated that tree swallow nestlings at our Lake Calumet sites accumulated mean total PCB loads from a low of 3,499 ng at Powderhorn (in 2005) to a high of 9,217 ng at Big Marsh, or approximately 250 to 658 ng/d, respectively. These accumulation rates are relatively low considering that Secord et al. (1999) calculated daily accumulation rates at a reference site of 100 to 1700 ng/d, and at contaminated sites, had rates ranging from 4,400 to 15,800 ng/d. The strong relationship between egg and nestling PCB profiles ($r \geq 0.97$ at all three sites) supports the contention that PCBs in eggs accounted for much of the burden in nestlings. At Big Marsh, where the concentrations of PCBs in nestlings were highest, the nestlings accumulated only three times the mass of PCBs that already existed in the eggs (9,217 ng).

Organochlorine pesticides

In contrast to other contaminants, the concentrations of DDE we observed in eggs were in the top 50 percent of previously reported values, and the highest mean value in nestlings (582 ng/g at Big Marsh) is among the highest reported. Concentrations of DDE of as high as 950 to 4,800 ng/g have been reported in tree swallow nestlings at sites in Ontario on the Lake Erie shore (Bishop et al., 1999). However, most reported values for nestlings are between ~10 and 500 ng/g (Bishop et al., 1999; Custer et al., 1999, 2002, 2003, 2006). Mayne et al. (2005) observed altered thyroid function in tree swallow embryos exposed to DDE. The concentrations of DDE in eggs were in the 100 ng/g range. Concentrations of DDD in tree swallows are rarely reported. Custer et al. (2002) measured DDD in hooded mergansers (*Lophodytes cucullatus*), wood ducks (*Aix sponsa*), and tree swallows and only hooded mergansers had measurable concentrations. In our study, concentrations of DDD in nestlings were higher at Big Marsh, were similar at Indian Ridge and were lower at Powderhorn, as compared to concentrations of DDE. In our estimation, nestlings at Big Marsh accumulated more DDD than DDE. However, concentrations apparently were still low enough that hatch, fledge, and nest success at Big Marsh were comparable to values for Powderhorn where accumulation of these contaminants were lower.

PBDEs

To our knowledge, no published studies have documented PBDE concentrations or effects in tree swallows. A number of studies have documented effects on predatory birds (e.g., Drouillard et al., 2007; Elliot et al., 2005; Fernie et al., 2005; Jaspers et al., 2005; Naert et al., 2007; Voorspoels et al., 2004), but there have been a limited number of studies with passerine birds (Dauwe et al., 2005; Van den Steen et al., 2006). PBDE concentrations in eggs fell within, albeit at the lower end, the range of concentrations observed in great blue herons (*Ardea herodias*), double-crested cormorants (*Phalacrocorax auritus*), and osprey (*Pandion haliaetus*). The values in our study were lower than concentrations (~500 ng/g dw converted from ww) observed to cause altered thyroid function and oxidative stress in American kestrels (*Falco sparverius*). In our study, sum PBDE concentrations in both eggs and nestlings were generally an order of magnitude lower than sum PCBs and DDE, and accumulation rates were lower as well. Tree swallow nestlings at Indian Ridge had the greatest accumulation at 3,864 ng in 2005. The most prevalent congeners in eggs were 47 and 99, while in nestlings, higher levels of 183 and to a lesser extent 153 were also observed. Congeners 47, 99 and 183 were those most commonly measured in benthic insects.

Stable isotope analysis

These nominal differences among sites in egg nitrogen and carbon isotope profiles may suggest that the mother tree swallows are from different populations feeding on different carbon and nitrogen sources, but it is more likely a reflection of local conditions, given that N and C turnover rates in eggs are approximately 6 days (Hobson, 1995) and the mothers may inhabit their nesting territory for two weeks before laying eggs. This latter hypothesis is supported by the fact that at Powderhorn the insect $\delta^{15}N$ values were much lower than those at the other two sites, and so were the values in eggs at that site.

Others have suggested that terrestrial-feeding predatory birds will have different PCB and PBDE profiles than those feeding in aquatic environments (Law et al., 2003; Lindberg et al., 2004; Jaspers et al., 2006). For example, Jaspers et al. (2006) found that lower brominated congeners such as BDE 47 were prominent in herons and grebes (fish-eating birds), while BDEs 99 and 153 were at higher concentrations in terrestrial species. Likewise, aquatic birds had higher concentrations of PCBs 28 and 74 than terrestrial birds (Jaspers et al., 2006). In our study, tree swallows ate insects of both aquatic and terrestrial origin. At Big Marsh and Indian Ridge, the proportions of aquatic and terrestrial insects in boluses were approximately equal, whereas at Powderhorn, swallow diets were 60% aquatic. Comparing stable isotope profiles of individual birds to their contaminant profiles, we observed that in many cases, higher contaminant concentrations were associated with isotope profiles more indicative of terrestrial diets. Exceptions included PBDE 47 at Indian Ridge, which was associated with aquatic diets, in agreement with the findings of Jaspers et al. (2006). In contrast, at Big Marsh and Powderhorn higher concentrations of PBDE 47 were associated with terrestrial diets. The only other contaminants that were positively correlated with aquatic diets were DDE at Big Marsh and mercury at Powderhorn. Most of the higher PCBs and PBDEs were associated with terrestrial diets.

CONCLUSIONS

With the exception of timing of nest initiation and other variables that are dependent on nest initiation timing (e.g., clutch size and nestling mass), we observed no differences among sites in tree swallow nesting ecology parameters. Nestlings accumulated a variety of inorganic and organic contaminants via their insect diets, but concentrations of nearly all the contaminants were at the lower end of ranges reported in the literature. The exception to this rule was concentrations of DDE in eggs and nestlings at Big Marsh, which were among the higher reported values. To our knowledge, this paper is the first report of PBDE concentrations in tree swallow nestlings. Our stable isotope analysis suggested a terrestrial origin for many of the contaminants, as has been suggested previously by others.

While our nesting ecology data suggest that the contaminants accumulated by tree swallows at these sites are not having short-term effects on reproduction, longer-term studies are required to determine the full extent of the risks.

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