DRAFT: 12.6.2010

Title: Persistent organic pollutant concentrations in fat, blubber, blood and eggs of leatherback turtles with confirmation of maternal transfer.

Authors

Kelly R. Stewart[1,2], Jennifer M. Keller[3], Ryan Templeton[4], John R. Kucklick[3], Chris Johnson[5]

Author Affiliations:

[1] Duke University Center for Marine Conservation, Nicholas School of the Environment, Duke University Marine Laboratory, 135 Duke Marine Lab Rd., Beaufort, NC 28516 USA

[2] Protected Resources Division, Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, 3333 N. Torrey Pines Ct., La Jolla, CA 92037, USA

[3] National Institute of Standards and Technology, Hollings Marine Laboratory, 331 Fort Johnson Road, Charleston, SC 29412, USA

[4] Grice Marine Laboratory, College of Charleston, 205 Fort Johnson Road, Charleston, SC 29412, USA

[5] Loggerhead Marinelife Center, 14200 US Hwy. One, Juno Beach, FL 33408 USA

Corresponding author:

Kelly Stewart

Phone: 858.334.2850

Fax: 858.546.7003

Email: kelly.stewart@noaa.gov

Correspondence Address:

Kelly Stewart

Protected Resources Division, Southwest Fisheries Science Center, 3333 N. Torrey Pines Ct., La Jolla, CA 92037, USA

Abstract

Of published contaminant studies in reptiles, sea turtles are poorly represented. To assess threats to these endangered species, it is critical to establish baseline values for contaminant concentrations that may have detrimental consequences at the individual or population level. The purpose of this study was to measure and then evaluate the relationship between contaminant concentrations in blubber and fat samples from seven dead stranded leatherbacks along the beaches of the Southeastern USA. In addition, we aimed to establish baseline measurements of organohalogen contaminants (PCBs, pesticides, PBDEs, and toxaphenes) in nesting leatherback turtles from Florida and to investigate whether these contaminants are passed on to eggs using paired blood and egg samples from six turtles. In the fat and blubber samples, the five predominant PCBs were 153+132, 187+182, 138+163, 118, and 180+193. These five contaminants were also present in female blood and egg samples, but at different proportions than for the fat and blubber samples. Contaminant concentrations (total PCBs, 4,4’-DDE, total PBDEs and total chlordanes) were positively correlated between the blood and egg samples from nesting turtles suggesting that these contaminants are passed to the eggs during lipid deposition. This relationship was also evident between the fat and blubber concentrations for stranded leatherbacks, but total PBDEs were not significant ($p = 0.13$). There was evidence that less lipophilic PCB congeners were more readily transferred from the females to their eggs than more lipophilic compounds. PBDE profiles in the four tissues were similar to what has been documented for other wildlife populations, but different from some other turtle studies. Although the contaminant concentrations that we measured in leatherbacks were much lower than those shown to have toxic or lethal effects in other aquatic turtles and reptiles, evidence from other sea turtle studies show that our measured concentrations may have sub-lethal effects related to hatchling body condition and health parameters; unfortunately, we could not address this aspect in this study. We recommend that hatchling health and survival related to contaminant concentration be investigated in more detail in future studies, paying close attention to differences across populations, especially those that may be most vulnerable to extirpation.

Key words: leatherback, contaminants, blood, fat, maternal transfer, PBDE

1. Introduction

In the contaminants literature, research that focuses attention on reptiles represents a small proportion of all studies (Sparling et al., 2010). Among reptile papers, sea turtle research is even more severely underrepresented. While there is a growing body of literature on contaminants in loggerhead turtles (*Caretta caretta*), baseline values of contaminants for the other species are rather limited. Persistent organic pollutants (POPs) for which baseline levels have been established in loggerheads include polychlorinated biphenyls (PCBs) (Corsolini et al., 2000; Alam and Brim, 2000; Keller et al., 2004a), organochlorine pesticides (OCPs) (Storelli et al., 2007) and polybrominated diphenyl ethers (PBDEs) (Keller et al., 2005). There is now some evidence that these persistent contaminants are correlated with sub-lethal effects related to general health and immune response in loggerhead turtles (Keller et al., 2004b). Additionally, flame retardants (PBDEs) are contaminants of growing concern particularly because they are still present in a wide variety of consumer products such as electronics, plastics, and fabric, etc. (Ross et al., 2009). These persistent lipophilic compounds have been detected in a variety of wildlife species, principally in birds and in marine mammals (Law et al., 2003). Detrimental health effects of PBDEs include thyroid disruption and neurodevelopmental defects (Ross et al., 2009).

Of the sea turtle contaminant studies to date, only a few investigations have focused on leatherback turtles (*Dermochelys coriacea*) (Mckenzie et al., 1999; Deem et al., 2006; Guirlet et al., 2008; Guirlet et al., 2010). Leatherbacks nest on tropical and subtropical beaches worldwide, with individuals nesting every 2 to 3 years. Within a single nesting season, they lay multiple nests about 10 days apart with clutch sizes ranging from 65 to 86 eggs (Stewart and Johnson, 2006), depending on the population. From nesting grounds they make long distance migrations to foraging areas in cold waters at northern and southern latitudes (James et al., 2005a; Shillinger et al., 2008). Leatherbacks eat primarily gelatinous zooplankton, focusing their attention mainly on large species such as the Lion's Mane jelly (*Cyanea capillata*) (James and Herman, 2001); they also eat cannonball jellies (*Stomolophus meleagris*) in the Southeastern US (Grant et al., 1996). Fat and blubber play an important role in leatherbacks, providing both energetic resources for migratory behavior and vitellogenesis, as well as insulating them against the cold waters in which they forage. Leatherbacks are able to maintain their body temperature up to 8.2 °C above ambient water temperatures (James and Mrosovsky, 2004), and they weigh up to 33% more in foraging areas than in nesting areas (based on similar carapace lengths) (James et al., 2005b).

To evaluate the potential threat to this critically endangered species from contamination, it is necessary to establish baseline concentrations for these reptiles in the wild and to identify potential toxic effects. Baselines allow us to compare threats across nesting populations and to prioritize conservation actions accordingly. To date, these studies have included the investigation of maternal transfer of trace elements (Se and Cd) and chlorinated contaminants (OCPs, DDTs, HCHs and PCBs) in leatherbacks in French Guiana (Guirlet et al., 2008; Guirlet et al., 2010), organochlorine contaminants in nesting turtles in Gabon (Deem et al., 2006) and stranded turtles in the Mediterranean, UK (Mckenzie et al., 1999), and Canary Islands (Oros et al., 2009).

The purpose of this study was to measure POP concentrations (PCBs, OCPs, and PBDEs) in fat and blubber from stranded leatherback turtles and blood and eggs from nesting leatherback turtles from the southeast coast of the U.S. Relationships between the tissues were assessed to better understand the distribution of POPs in leatherback turtles (blubber vs. fat) and the possibility that POPs are passed from nesting females to their eggs. With evidence that POPs indeed maternally transfer to eggs of three turtle species (Kelly et al., 2008; van de Merwe et al., 2010b; Guirlet et al., 2010), it is essential to know baseline POP concentrations in leatherback eggs so as to begin to determine if POPs are a threat to their reproductive success and overall lifetime reproductive output. This is especially important in long-lived species such as sea turtles. Understanding the effects of contaminants on the viability of eggs has long-term consequences on the viability of sea turtle populations that are already threatened or endangered. The baseline concentrations reported in this study provide a foundation for future research that should assess toxic effects.

2. Materials and methods

2.1. Sample collection and processing

Fat and blubber samples were collected post mortem from seven leatherback turtles (*Dermochelys coriacea*) found stranded on the beaches of North Carolina and South Carolina (Table 1). The specimens (six females and one male) ranged in size from 132 cm to 176 cm curved carapace length (CCL); five had major trauma possibly caused by boat strikes. During necropsy, hexane-rinsed scalpels and forceps were used to collect tissues, which were stored in hexane-rinsed foil at -80 ^{o}C until analysis.

Female leatherback turtles were encountered while they were nesting at Juno Beach, Florida (26.9 °N, 80.1 °W) in April and May 2003, as part of a long-term study. Individual turtles were identified using flipper tags (Inconel Style 681, National Band and Tag Co., Newport, KY) and Passive Integrated Transponder (PIT) tags (125 kHz; Digital Angel Corporation, St. Paul, MN). Nesting turtles were approached when oviposition began, checked for identifying tags, and then sampled (Table 2). Blood was drawn from the femoral venous plexus in the rear flipper of six nesting turtles using double-ended needles (1.5”, 20 gauge) directly into glass Vacutainer tubes containing sodium heparin (Becton Dickinson, Franklin

Lakes, NJ). Within 2 h of sampling, whole blood (in the original sampling tubes) was frozen at -20 ºC until analysis. The precise location of each nest was recorded using a Trimble Pro XL differential global positioning system (Trimble Navigation Limited, Sunnyvale, CA). One wooden stake was placed near the egg chamber and another was placed at the vegetated dune line as a secondary method of marking the nest.

Following hatchling emergence, marked nests were excavated, and contents of the nest were evaluated to count hatched and unhatched eggs. Eggs that failed to hatch were collected (up to six eggs per nest) from eight nests; six of these corresponded to the female turtles that were sampled for blood. Of the remaining two nests sampled, one was a second clutch laid later in the season by one of the known females (595AJ) that had been sampled previously for blood, and the other was collected from a female that was not sampled for blood (943SI). Eggs were frozen whole at -20 ºC and thawed on the day of analysis. The shells were removed and total egg contents (yolk and albumen) were pooled from up to six eggs per nest that either had no development or very early stage development. All nests had at least three eggs to pool, except for one nest there was only one egg available (456RE). We decided to include this nest in the analysis, because prior studies showed that POP concentrations are far less variable among individual eggs within a nest than among different nests laid on the same beach (van de Merwe, 2010b) and good agreement among individual sea turtle egg samples within a nest (at least for eggs with no, early, or middle development) was demonstrated with an average relative standard deviation of only 14% for total POPs (calculated from data reported in Alava et al., 2006). Homogenization was performed with a spatula in a beaker.

2.2. Extraction

Methods were modified from Keller et al. (2004c). Briefly, eggs (≈14 g), full-depth blubber (≈1 g), and fat (≈2 g) samples were mixed with Na_2SO_4, transferred to pressurized fluid extraction (PFE) cells (Dionex, Sunnyvale, CA), spiked with an internal standard solution in *iso*-octane containing PCB 103, PCB 198, 4,4'-DDE-d_8, 4,4'-DDD-d_8, 4,4'-DDT-d_8, and endosulfan-I-d_4, and extracted with dichloromethane (DCM) as described by Kucklick et al. (2002). One to three procedural blanks and a five to seven point calibration curve were processed alongside each batch of samples. Blanks were simply PFE cells packed with Na_2SO_4. The calibration standards used for the calibration curves contained National Institute of Standards and Technology (NIST) Standard Reference Materials (SRMs): SRM 2261 Chlorinated Pesticides in Hexane, SRM 2262 Chlorinated Biphenyl Congeners in 2,2,4-Trimethylpentane, SRM 2274 PCB Congener Solution-II in Isooctane, and SRM 2275 Chlorinated Pesticides Solution-II in Isooctane along with a solution containing 14 additional PCB congeners and a solution containing 14 PBDE congeners (Cambridge Isotope Laboratories, Andover, MA). The calibration curve used for the blubber and fat samples ranged from 0.4 ng to 230 ng of each compound. The calibration curve used for the egg samples also contained a solution of 31 additional PCB congeners and pentachlorobenzene and ranged from 0.1 ng to 400 ng of each compound. A separate, non-extracted, calibration curve was prepared to measure 4 toxaphene congeners in the eggs. This curve ranged from 0.5 ng to 0.03 ng of 2-endo, 3-exo, 5-endo,6-exo,8,8,10,10-octachlorobornane (Parlar 26), 2-endo, 3-exo, 5-endo, 6-exo, 8, 8, 9, 10, 10-nonachlorobornane (Parlar 50), 2, 2, 5, 5, 8, 9, 9, 10, 10-nonachlorobornane (Parlar 62), and 2,2, 5-endo, 6-exo, 8,9,10-heptachlorobornane (Parlar 32). One replicate for each of the following control materials was also analyzed: SRM 1945 Organics in Whale Blubber, SRM 1946 Lake Superior Fish Tissue, and an in-house cryohomogenized composite of loggerhead egg yolk (Alava et al., 2006).

Whole blood samples were extracted using a liquid:liquid extraction technique modified from technique A described in Keller et al. (2004a). Briefly, aliquots of blood ranging from 3 g to 19 g were spiked with a solution of the internal standards listed above, but this solution was in acetone rather than *iso*-octane so that it would mix with the samples. Samples were equilibrated for 2 h at room temperature and then treated with formic acid and extracted three times by vortex mixing with 1:1 (v/v) methyl-*tert*-butyl ether (MTBE):hexane. The calibration solutions used to prepare a calibration curve for blood analysis were extracted alongside the samples and ranged from 0.03 ng to 30 ng of each compound in SRMs 2261, 2262, 2274, and 2275 plus the two additional PCB solutions mentioned above, a solution of 28 PBDE congeners, and octachlorostyrene. The extracts were screened for toxaphenes using the additional non-extracted calibration solutions mentioned above. One replicate of SRM 1589a PCBs, Pesticides, PBDEs, and Dioxins/Furans in Human Serum and one replicate of an in-house pool of loggerhead plasma were analyzed as control materials.

2.3. Total extractable organic (TEO) content determination and extract clean-up

Total extractable organic (TEO) content was determined as a proxy for percent lipid content from all sample extracts gravimetrically as described by Keller et al. (2004a). High molecular mass interfering compounds in the blubber, fat, and egg extracts were removed using size exclusion chromatography (SEC) as described in Kucklick et al. (2002). Blubber and fat extracts were further cleaned up on 1 g Florisil columns (deactivated with 1.2% deionized H_2O; mass fraction) eluted with 12 mL 1:1 (v/v) DCM:hexane. Further clean-up of the egg extracts was achieved with 1.8 g alumina columns (deactivated with 5% deionized H_2O; mass fraction) eluted with 9 mL 35:65 (v/v) DCM:hexane. Blubber, fat and egg extracts were fractionated into relatively lower and higher polarity fractions (F1 and F2, respectively) using 1 g silica gel columns (deactivated with 2.5% deionized water; mass fraction). After conditioning the columns and loading the samples, 5 mL of hexane was collected into fraction 1 (F1). Fraction 2 (F2) was then collected using 6 mL of 1:1 (v/v) DCM:hexane. Compounds found in each fraction are listed in Alava et al. (2006).

Blood extracts were cleaned up using alumina columns followed by SEC. For SEC, extracts were injected onto a semi-preparatory scale (7.5 mm x 300 mm, 10 µm particle size with 100 Å diameter pores) PLGel column (Polymer Labs, Amherst, MA) with a flow rate of 1 mL/min of DCM. The fraction containing the compounds of interest was collected between 7.5 min and 13 min. The extracts were then fractionated with silica columns as described above, evaporated to approximately 0.2 mL, and PCB 14 was spiked into each of the fractions as a recovery standard.

2.4. Quantification

Each fraction of blubber and fat samples was reduced to approximately 1.0 mL for analysis by gas chromatography with electron capture detection (GC-ECD) according to Kucklick et al. (2002). 4,4'-DDT and *trans*-nonachlor were split between the two fractions, so these were quantified by a second GC-ECD injection after recombining both fractions. The recombined extracts were evaporated to approximately 0.2 mL and analyzed for PBDEs with a GC/mass spectrometer (GC/MS; Agilent 6890N/5973 inert, Palo Alto, CA) operated in negative chemical ionization (NCI) mode using a 2 µL cool on-column injection onto a 5 m x 0.25 mm Siltek guard column (Restek, Bellefonte, PA) connected to a 10 m x 0.18 mm x 0.18 µm film thickness DB-5MS capillary column (Agilent, Palo Alto, CA). The oven, column, and source parameters were similar to those used in Method 3 of Stapleton et al. (2007) to monitor the following compounds (m/z): PBDEs (79, 81), PBDE 209 (409, 487), PCB 198 (427.7, 429.7), and endosulfan-I-d_4 (376, 410).

Egg and blood extracts were fractionated initially to analyze them using GC-ECD; however, they were recombined, evaporated to approximately 0.2 mL, and analyzed using the GC/MS instead. Extracts (20 µL) were injected three separate times using a programmable temperature vaporization (PTV) inlet as described in Moss et al. (2009) onto either a 60 m x 0.25 mm x 0.25 µm film thickness DB-5MS capillary column or a 15 m x 0.25 mm x 0.25 µm film thickness DB-5MS capillary column (Agilent Technologies, Santa Clara CA). The first injection was used to measure PCBs and certain pesticides, such as the DDT compounds, from a 60 m capillary column using electron impact (EI) mode. The second injection used a 60 m column with NCI mode for toxaphenes, chlordanes, HCHs, HCB, and endosulfans. The third injection used a 15 m column with NCI mode to measure the PBDEs. The inlet, oven, column, and source parameters were similar to those used in Keller et al. (2009) and Stapleton et al. (2007).

The amount of each compound was calculated relative to an appropriate internal standard and the slope and intercept of linear regression using at least a three-point calibration curve that bracketed the peak area ratios of the samples. The reporting limit (RL) for all compounds was established as the ng in the lowest detectable calibration solution divided by the extracted sample mass. When multiple blanks were analyzed in a batch, an RL was also calculated as the average plus three times the standard deviation of ng measured in the blank noise divided by the extracted sample mass. The maximum of the two calculated RLs was used.

2.5. Statistical Analysis

To estimate totals for a contaminant class, only detected compounds were summed. Total PCBs was the sum of 44 congeners for the blubber and fat and 77 congeners from eggs and blood. Total DDTs was the sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT. The total chlordane concentration was the sum of heptachlor epoxide, oxychlordane, *trans*- and *cis*-chlordane, and *trans*- and *cis*-nonachlor. Descriptive statistics were performed using the statistics program R version 2.11.1 (R Development Core Team, Vienna, Austria) using the "NADA" package, which can handle left-censored datasets, such as those with concentrations below the RL as recommended by Helsel (2005). Mean, standard deviation, and median were calculated with either Kaplan-Meier or Regression on Order (ROS) models. The choice between the two was based on sample size and detection frequency as outlined in Helsel (2005). The Kendall's Tau correlation was used to assess the relationship of lipid-normalized concentrations (i.e., ng/g lipid) between blubber and fat of the stranded animals and between the maternal blood and egg samples for total PCBs, 4,4'-DDE, and total chlordanes using JMP 5.1 (SAS Institute Inc., Cary, NC) and for total PBDEs (which contained non-detects) using the "NADA" package with R. As suggested by Swarthout et al. (2010), concern over accuracy was considered for PBDE concentrations when the recovery of internal standards was below 40%. This was noted for only one blood sample (693PH), which was therefore completely excluded from the descriptive statistics and correlation analysis for PBDEs. Only the first of the two clutches of eggs obtained from one female (595AJ) was used in the descriptive statistics and correlation analysis between blood and egg samples, because it was the egg sample paired with the blood for this turtle. In addition, concentrations in eggs collected from the clutch of a turtle (943SI) not sampled for blood were not included in the means. All individual data, however, can be found in Supporting Material. Paired t-tests were performed to determine if POP patterns differed between blubber and fat and between eggs and blood. These t-tests evaluated the percent of total PCBs, percent of total chlordanes, and percent of total PBDEs data for individual congeners or compounds paired for individual turtles.

3. Results and discussion

3.1. Quality control analyses

The concentration of compounds and the TEO content measured in the single replicate of three different SRMs (1945, 1946, and 1589a) differed from certified values listed on the Certificates of Analysis on average for all compounds by only -8%, -9%, and -5%, respectively. Rarely compounds were >30% different from certified values with maximum deviations being -47%, -59%, and -46%, respectively. The measured concentrations in the single replicates of two loggerhead control materials (eggs and plasma), which better match the matrices measured from the leatherback turtles, agreed with previous measurements in these materials (within 30% of consensus values). This good agreement with the certified and consensus concentrations indicates that the leatherback data are of high quality.

3.2. Total extractable organic (TEO) content and contaminant concentrations in fat and blubber

TEO content (%) and concentrations of the predominant PCBs, OCPs and PBDEs are presented for leatherback fat and blubber samples in Table 3. Values for individual turtles may be found in the Supporting Material (SM, Table 1). PCBs were the primary contaminant class in fat and blubber, with the predominant PCB in both tissues being PCB 153+132 (Fig. 1). The contribution of PCB 153+132 to total PCBs was significantly greater in blubber than in fat. The predominant chlordane was *trans*-nonachlor (Fig. 2). HCB, mirex and dieldrin were detectable in both fat and blubber, with dieldrin having the highest concentration of those three contaminants in both tissues. Of the DDT metabolites, 4,4'-DDE was the primary contributor, comprising over 80% of the total in each tissue (SM, Table 1). Other than PCB 153+132, fat and blubber did not differ in their POP patterns (Figs. 1-3), indicating minimal to no compound-specific distribution of POPs between these two lipid-rich tissues.

Several PBDE congeners were detected in both fat and blubber with PBDE 47 having the greatest contribution (Fig. 3). The patterns seen here with PBDE 47 being the predominant congener followed by four others (PBDE 99, 100, 153 and 154) is the typical pattern observed for biota in the majority of the literature (Hites, 2004) and in some, but not all, turtle studies. This typical pattern was noted for blood samples of Kemp's ridley turtles (*Lepidochelys kempii*) from the Gulf of Mexico, green turtles (*Chelonia mydas*) from nearshore South Carolina to Florida (Swarthout et al., 2010), green turtles and a flatback turtle (*Natator depressus*) from Australia (Hermanussen et al., 2008), and loggerhead sea turtles from South Carolina to Florida (Keller et al., 2005). However, loggerhead turtle plasma samples from North Carolina are dominated by PBDEs 100 and 154 (Carlson, 2006). These congeners are about equal in proportion to PBDE 47 in loggerhead eggs from North Carolina (Keller et al., 2005). Atypical patterns of proportionally higher PBDEs 100 and 154 were noted in the southern hemisphere as well, in green turtle tissues and a hawksbill (*Eretmochelys imbricata*) blood sample from Australia (Hermanussen et al., 2008), and PBDE 47 makes up less of the total PBDEs than PBDEs 99 and 153 in green turtle eggs, as well as maternal and hatchling blood from Malaysia (van de Merwe et al., 2010b). Likewise, atypical patterns have been seen in plasma from freshwater turtles (*Sternotherus odoratus* and *Trachemys scripta troosti*) from Tennessee (Moss et al., 2009) and plasma and fat from diamondback terrapins (*Malaclemys terrapin*) from New

Jersey (Basile et al., 2011). Reasons for these atypical patterns have been postulated to be due to biotransformation or elimination of certain PBDEs, but this appears to be location-specific rather than species-specific (Basile et al., 2011).

TEO content and contaminant concentrations were fairly comparable between the two tissues for each stranded turtle, except Turtle Dc-TP-99-06-09 had much higher contaminant concentrations in the blubber than in the fat on a wet-mass basis. Since the fat tissue of this animal had such low TEO content (less than 1%), when concentrations were lipid-normalized the two tissue concentrations were fairly similar. Without this turtle, the mean blubber concentrations of total PCBs, 4,4'-DDE, total chlordanes, and total PBDEs were 48.1 ng/g wet mass, 17.5 ng/g wet mass, 17.0 ng/g wet mass, and 10.5 ng/g wet mass; these are similar to the mean concentrations in fat. The samples from this particular turtle were measured previously by Keller et al. (2004c), who hypothesized that POPs might accumulate at higher concentrations in leatherback blubber than fat. This hypothesis, however, was proven incorrect by the current study's larger sample size, highlighting the importance of having a robust sample size and considering the TEO content and body condition of individuals sampled. The TEO content of this turtle's (Dc-TP-99-06-09) blubber and fat were respectively 4 times and 80 times lower than the mean of the other six turtles (59.0% TEO in blubber and 56.0% TEO in fat) (SM Table 1). This suggests that she had utilized a significant amount of her fat stores, and not surprisingly lipids are more easily mobilized from fat depots than the structural blubber layer of the carapace. The large difference in wet mass POP concentrations between her blubber and fat samples also suggests that POPs are mobilized differently between these two tissues. No studies are currently available that address lipid and concurrent POP mobilization in reptiles; however, mobilization of both from blubber of marine mammals into blood (Yordy et al., 2010b) as well as stratification of POPs by blubber depth (Krahn et al., 2004) have been investigated. Similar studies with leatherback blubber and blood samples are warranted because they could provide insight as to how POPs are distributed and mobilized throughout this reptile. Comparing the concentrations in fat samples found in this study to those from other leatherback studies that examined fat, the concentrations of total PCB were similar to those reported by Oros et al. (2009) but lower than reported by Godley et al. (1998) and McKenzie et al. (1999) (Table 5).

3.3. TEO content and contaminant concentrations in blood and eggs

Concentrations of PCBs, OCPs including toxaphenes, PBDEs and TEO content are summarized for leatherback blood and egg samples in Table 4. Concentrations in individual turtles may be found in the Supporting Material (SM, Table 2). All six nesting turtles had detectable levels of DDT-related compounds and PCBs in their blood, while all but one had detectable levels of PBDEs.

Sea turtles undergo vitellogenesis or development of follicles, which later each become the egg yolk of an individual egg, during the two or more years between nesting seasons (Miller 1997). During this time, they deposit lipids, proteins, and other essential nutrients, as well as POPs that they are consuming on the foraging grounds or mobilizing from their lipid stores into the follicles (Miller 1997). By the time they have reached the nesting beach, a leatherback sea turtle has prepared approximately 480 follicles ready to become her 6 or so clutches of about 80 eggs each that nesting season. Because of this reproductive strategy, eggs from a single female within a nesting season are expected to have similar POP concentrations regardless of whether it was her first or last clutch. The same logic extends to eggs within a clutch, and two studies have shown evidence of low variability in contaminant concentrations between eggs of the same clutch compared to variability among nests from different females on the same beach (Alava et al., 2006; van de Merwe et al., 2010b). In the current study, eggs from two clutches from the same turtle (eggs collected on 7/9/2003 and 7/31/2003 from nests laid by 595AJ) had quite similar contaminant concentrations (average percent difference for all lipid-normalized compounds of the 2^{nd} nest from the 1^{st} was 4.5%; range = -34.0% to 65.6%) (SM, Table 2). However, Guirlet et al. (2010) found with a larger sample size that although PCB and DDT concentrations in blood of nesting turtles in French Guiana remained constant over the nesting season, those contaminants decreased over the season in eggs from successive clutches, even after lipid normalization of the sample concentrations (lipid content decreased significantly in eggs over the season), so our sample size on this aspect may have been insufficient to pick up seasonal differences in egg contaminant concentrations.

The main PCB congener found in blood and egg samples in our study was PCB 153+132 (Fig. 1); this was similar to the results for leatherback blood and egg samples from French Guiana of Guirlet et al. (2010), who found that PCB 153 and PCB 153+105 predominated in blood and eggs, respectively. The chlordane compound found at the highest concentration in blood and eggs, which was the same as that found for fat and blubber, was *trans*-nonachlor (Fig. 2). Also similar to the fat and blubber samples, HCB, mirex and dieldrin were detected in blood and eggs, with dieldrin again having the greatest concentrations in both tissues. 4,4'-DDE was the only DDT metabolite detected in blood; several other DDT-related compounds were detected in the eggs (SM, Table 2). The primary PBDE congener in both blood and eggs was PBDE 47 (Fig. 3), and the PBDE profile was typical of that seen in wildlife and humans (Hites, 2004). Although toxaphenes were not detectable in blood, there were three primary congeners found in the eggs (Parlar 26, 50, and 62), with each congener contributing fairly equal proportions.

Comparing our results with those of Guirlet et al. (2010), in general our total PCB and total DDT means for blood were slightly higher (1.3 to 2-fold higher; Table 5). For egg samples, again our total PCB and total DDT concentrations were marginally higher. Previous studies have also found higher concentrations of contaminants in turtles that forage farther north than south in the Atlantic Ocean (O'Connell et al., 2010; Ragland et al., in review).

In contrast with this study and that of Guirlet et al. (2010), Deem et al. (2006) found no detectable OCP or PCB contaminants in plasma samples from female leatherbacks nesting along the coast of Gabon. They suggested that the gelatinous zooplankton diet of leatherbacks might have been the cause for the negative results; however, we found that OCPs and PCBs were detectable in leatherbacks nesting in Florida. One major difference between our study and that of Deem et al. (2006) was the reporting limit (or limit of detection) of the compounds of interest. Deem et al. (2006) had reporting limits of approximately 20 ng/g wet mass for major contaminant compounds, while our reporting limits ranged from 0.002 ng/g wet mass to 0.237 ng/g wet mass. It is possible that turtles nesting in Gabon may have had contaminants of interest, but unfortunately they may not have been detected because of high reporting limits. Leatherbacks nesting in the Western (Florida) and Eastern Atlantic (Gabon) Ocean are likely to be foraging on similar prey species, even though the foraging grounds for these populations may differ spatially (Northwestern and Southwestern Atlantic, respectively). Examining inter-population differences in contaminant burdens would be very informative.

3.4. Tissue contaminant relationships

Paired blubber and fat concentrations of total PCBs, 4,4'-DDE, and total chlordanes were positively and significantly correlated ($p < 0.039$), and an insignificant but positive relationship

(Kendall’s tau = 0.60, $p = 0.133$) was observed for total PBDEs between the tissues (Fig. 4). When the turtle with extremely high contaminant concentrations (Dc-TP-99-06-09) was included in the correlations, all four contaminant classes were significantly correlated between tissues (Kendall’s tau > 0.71, $p < 0.03$; data not shown). These correlations, the slopes of nearly 1.0 (except 0.55 for total PCBs), and the similar concentrations between fat and blubber suggest that POPs are distributed equally between tissues of similar high lipid content.

Paired maternal blood and egg concentrations of total PCBs, total PBDEs, 4,4’-DDE and total chlordanes were all positively correlated ($p < 0.05$; Fig. 5). In the study by Guirlet et al. (2010), there was a positive correlation between blood and eggs for 4,4’-DDE but not for total PCBs or total HCHs (although PCB 153+105, PCB 180 and PCB 118 taken alone did show positive correlations between blood and eggs). The positive correlations between blood and egg values in this study provide strong evidence for maternal transfer of organohalogen contaminants into the egg. Maternal transfer has been shown previously in freshwater turtles (Dabrowska et al., 2006; Kelly et al., 2008) and in green sea turtles (van de Merwe et al. 2010b), but it was previously questionable in leatherbacks based on the few significant correlations observed by Guirlet et al. (2010). In addition, the current correlations provide evidence that measuring contaminant load in unhatched eggs gives a valid approximation of the contaminant load in the nesting turtle. This method is particularly attractive because of the non-destructive nature of the sampling, which is a primary consideration in species of conservation concern. A few unhatched eggs are easily collected during nest inventory, and the adult turtle does not need to be sampled while egg-laying. More importantly, viable eggs do not need to be sacrificed. Most monitoring programs already conduct nest inventories and collecting unhatched eggs would be a practical way to evaluate contaminants in the population. Van de Merwe et al. (2010b) suggest exploring whether contaminant concentrations vary in nonviable eggs over the course of incubation while they are decomposing, and NIST is currently undertaking a study of this nature. Until that is known, the strong correlations seen in the current study suggest that even slightly decomposed, unhatched eggs represent POP concentrations from the mother rather than external beach sources.

3.5. Congener-specific maternal transfer in leatherbacks

Based on the PCB congener profile observed in this study (Fig. 1), congener-specific transfer may be occurring between the maternal blood and the eggs. Compounds that are more lipophilic (i.e., PCB 170 to 206) made up a higher proportion in the blood than in the eggs, while compounds that are less lipophilic (i.e., PCB 66 to 138+163 and 153+132) were found at a higher proportion in the eggs. Significant differences were observed between blood and eggs for PCBs 66, 99, 138+163, 153+132, 170, and 187-182 (Fig. 1). This trend has been noted in green turtles (van de Merwe et al., 2010b) and in another population of leatherback turtles (Guirlet et al., 2010). This pattern is also consistent with the findings of Yordy et al. (2010a) that showed that less lipophilic compounds were more readily transferred into milk from maternal blood in bottlenose dolphins (*Tursiops truncatus*). The mechanism that deposits POPs into eggs or milk is not well understood, but it would be interesting to compare the reptilian and mammalian models for similarities.

3.6. Differences in contamination levels in marine turtles

To our knowledge, this study is only the second to examine maternal transfer of organochlorine contaminants in leatherbacks and one of the first to examine PBDEs and toxaphenes in any sea turtle species. It is also the first to compare POP concentrations among tissues from adult leatherbacks, focusing on lipid-rich tissues as a storage depot of POPs. Table 5 contains a summary of organohalogen contaminants measured to date in various tissues of sea turtles. Beginning with blood, which has become a common tissue studied in several species, leatherbacks have higher concentrations of the major classes of contaminants (PCBs, DDTs, PBDEs) compared to green turtles, but far lower concentrations than loggerhead turtles. Of course, age, size, foraging area and sex may have confounding effects on contaminant concentrations in turtles, so these comparisons are relative. Species differences in contaminant concentrations should reflect trophic status, with green turtles being lowest (mainly herbivores), leatherbacks intermediate and loggerheads and Kemp's ridleys being at higher trophic levels (omnivores feeding on benthic organisms). Egg concentrations from each of the three species studied do reflect trophic status with loggerheads having the highest concentrations, followed by leatherbacks and then green turtles (Table 5). Compared to other reptiles, our leatherback PCB and OCP concentrations are extremely low. Deleterious effects related to embryo development and hatchling deformities were demonstrated in snapping turtle eggs (*Chelydra serpentina serpentina*) at DDE and PCB concentrations of 389 ng/g wet mass to 3575 ng/g wet mass, respectively (Bishop et al., 1998), which are 200-400 times higher than the DDE and PCB egg concentrations (1.59 ng/g wet mass and 8.45 ng/g wet mass, respectively) reported in this study. In alligators (*Alligator mississippiensis*), Rauschenberger et al. (2004) found that females dosed with organohalogen contaminants had lower clutch success and higher mortality of embryos than control females; although again the concentrations that caused these effects were orders of magnitude higher than our concentrations.

Measuring contaminants and monitoring spatial and temporal trends between populations are important; however, there is a need for studies that demonstrate the physiological effects these toxicants might have on sea turtle species. Although the contaminant concentrations measured in the current study are lower than concentrations shown to have deleterious effects in other reptiles, there is reason to believe that sub-lethal effects could be occurring in this species. Van de Merwe et al. (2010b) found a significant negative relationship between green turtle total POP concentrations (wet mass basis) in eggs and hatchling body condition (mass: straight carapace length ratio). This correlation might support the authors' interpretation that turtle hatchlings exposed to higher POPs during embryonic development have a lower chance of survival, because they may have lower yolk reserves available to supply critical energy stores during the frenzy period, when they enter the sea and swim offshore to developmental areas (Wyneken and Salmon, 1992). The faster a hatchling can escape nearshore predators, the better its chance of survival (Stewart and Wyneken, 2004). However, the correlation should be considered cautiously, because van de Merwe et al. (2010b) did not report correlative results for body condition vs. lipid-normalized POP concentrations, body condition vs. percent lipid, nor egg mass vs. POP concentrations, all of which would have provided insight as to whether the hatchlings with poorer body condition were more a result of less lipid available for growth, more contaminants present, or smaller egg mass from the beginning. Nevertheless, the PCB concentrations measured in eggs in the van de Merwe et al. (2010b) study (Table 5) were lower than total PCBs in eggs from our study suggesting that similar body condition impacts should be assessed in leatherbacks. Unfortunately we were not able to assess this aspect of hatchling health. In another study, Keller et al. (2004b) found correlations between concentrations of organochlorine compounds and poor health indicators in blood chemistries for loggerheads in foraging areas in North Carolina. The total PCBs in that study were twice as high as the current study, but total chlordane and total DDT concentrations in that study were only slightly higher than what we observed in this study. Together these previous sea turtle studies showed that even low concentrations of organochlorine compounds may have sub-lethal effects on juvenile and hatchling sea turtles and that it is important for these threats to be properly evaluated.

3.7. Sources and consequences of contaminants in leatherback turtles

This study provides important baseline data for contaminant concentrations in tissues of dead stranded leatherbacks as well as for nesting leatherback turtles. Turtles nesting in Florida generally forage in the Western North Atlantic, where as capital breeders, they return to nesting beaches once they have accumulated the required reserves necessary for egg-laying. During foraging periods and migration, they may sample a wide range of habitats, thus consuming prey with varying contaminant loads. Following a nesting season, leatherback foraging would presumably be quite intense considering that they have no adipose reserves from the beginning of the nesting season (Guirlet et al., 2010). Foraging would be especially intense during the summer and early fall months following nesting when water temperatures are high and jellyfish production is at a maximum. Guirlet et al. (2010) discuss the effect of the remigration interval on levels of contaminants in nesting females, suggesting that turtles may differ in terms of their contaminant load based on how long they are foraging before returning to nest (usually every 2 years or 3 years), with those turtles that spend longer at foraging areas having higher burdens. One other interesting factor may be the specific foraging area for individual turtles, as leatherbacks are known to forage in many locations around the North Atlantic. They suggest that total contaminant load for nesting turtles may be the result of interplay between these two factors. Unfortunately we were not able to assess differences based on 2-year or 3-year remigration intervals as it was only the third season of our long-term study, however we do have some information on foraging locations for turtles nesting in Florida.

Although leatherbacks are generally considered pelagic, they are frequently observed in coastal waters of the Southeastern US, particularly during summer months (Ernst and Gilroy, 1979; Eckert et al., 2006; TEWG, 2007). Evidence from telemetry studies of Florida nesting leatherbacks revealed that individuals from this population spent significant amounts of time near estuaries and bays in the Southeastern US (Eckert et al., 2006). Of 10 turtles tracked from Florida nesting beaches, 4 remained in the coastal water areas of Georgia, South Carolina, North Carolina, Virginia and Maryland. Bays and estuaries (Brunswick and Savannah, GA, Charleston, SC and the Chesapeake Bay, MD) in this region receive major inputs of POPs from land-based sources such as silviculture, agriculture and urban runoff (Lee and Maruya, 2006). This may make the Florida population particularly susceptible to contaminant exposure while in these waters.

4. Conclusions

Results from this study demonstrate that not only are POPs present in the tissues of leatherback turtles, but that contaminants are also passed on to eggs. This study provides strong evidence that POPs are maternally transferred in leatherback turtles and provides an important baseline of PCB, OCP and PBDE concentrations for nesting and stranded leatherbacks in the Southeastern USA, while laying the foundation for studies to be undertaken that examine the source of these contaminants, and more importantly, to determine the population-level effects of these compounds on this endangered species. Although contaminant concentrations measured in our study were substantially lower than concentrations in other reptile studies that demonstrated toxic effects, it is possible that sub-lethal effects may be occurring in this species. We recommend that studies of hatchling development, body condition and relevant health parameters in relation to contaminant loads should be undertaken.

Disclaimer

Certain commercial equipment or instruments are identified in the paper to specify adequately the experimental procedures. Such identification does not imply recommendations or endorsement by the NIST nor does it imply that the equipment or instruments are the best available for the purpose.

Acknowledgements

For assistance with blubber and fat sample collection, we'd like to thank R. Boettcher, J. Wyneken, S. Clusella, W. Cluse, M. Godfrey, S. Murphy, T. Murphy, and S. Johnson. Sampling in North Carolina was conducted under Duke University IACUC protocols A299-98-7, A351-99-07-3, A351-99-7-2, and A206-01-07, NC Wildlife Resources Commission Permit #s 99ST70, 01ST45, 02ST57, and under US Fish and Wildlife Service Permit #s PRT-676379 and TE-676379-2. Field work and sample collection in Florida was conducted under Marine Turtle Permits #077 and #157 (Florida Fish and Wildlife Conservation Commission). For training in blood sampling, we thank K. Frutchey; as well as S. Phelan and N. Desjardin for assistance with nesting turtles. Thanks to D. Sobel (Sea Turtle Conservation League of Singer Island) for collecting unhatched eggs. Funding and in-kind support for field work was provided by the Oak Foundation, Duke University Marine Lab, Florida Marine Turtle Grants Program, MacArthur

Beach State Park, A. Simler and B. Geyer. Additionally, this paper was based upon work supported by the National Science Foundation Award DBI-0552828 and the Department of Defense ASSURE Program. We'd like to thank R. Day, N. Kellar and D. Weller for their critical reviews of a previous draft of this manuscript as well as two anonymous reviewers for their excellent comments that improved the quality of our manuscript.

Supporting Material

Tables that supply supplementary data for this paper may be found in the Supporting Material.

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Table 1. Biometric information for stranded leatherback turtles sampled for blubber and fat.

Turtle ID	Date (Month/Day/Year)	Tissue	Stranding Latitude & Longitude	General Stranding Location	CCL / CCW (cm)	Sex	Cause of death
Dc-TP-99-06-09	6/9/1999	blubber, fat	33° 54.7' N 78° 13.1' W	Long Beach, NC	176 / 124	female	propeller wound, euthanized
Dc-RB-99-06-24	6/24/1999	blubber, fat	34° 42.3' N 76° 33.9' W	Harker's Island, NC	147.5 / 116.2	female	5 holes in head, euthanized
Dc-SC-01-05-19-01	5/19/2001	blubber, fat	34° 37.4' N 76° 32.5' W	Cape Lookout Bight, NC	147 / NA	male	propeller wound
Dc-SAJ-01-05-31-01	5/31/2001	blubber, fat	34° 41.7' N 76° 42.6' W	Atlantic Beach, NC	161 / 104	female	propeller wound
Dc-WMC-02-05-16-01	5/16/2002	blubber, fat	34° 41.8' N 76° 46.4' W	Atlantic Beach, NC	132 / 87	immature female	unknown
Dc-MG-02-05-26-01	5/26/2002	blubber, yellow fat, brown fat	34° 41.4' N 76° 40.2' W	Atlantic Beach, NC	143 / 100	female	propeller wound
Dc-03-05-30-01 SC	5/30/2003	blubber, fat	32° 45.9' N 79° 52.0' W	Charleston, SC	175 / 126	female	propeller wound

Table 2. Information on nesting leatherback turtles sampled for whole blood and eggs from particular clutches. All turtles were encountered nesting at Juno Beach, FL. The number of eggs pooled from each clutch is indicated in parentheses. Of the final two clutches of eggs sampled, one was from a known turtle (595AJ) and one (943SI) was from an unknown female.

Turtle ID	**CCL / CCW (cm)**	**Blood collected**	**Eggs collected (#)**
595AJ	148.1 / 108.7	4/15/2003	7/9/2003 (5)
456RE	158.4 / 109.0	4/19/2003	7/21/2003 (1)
617MA	140.1 / 104.9	4/19/2003	7/29/2003 (4)
693PH	162.9 / 113.2	5/11/2003	8/16/2003 (5)
622CL	149.8 / 108.4	6/13/2003	8/22/2003 (5)
567CO	145.0 / 104.5	4/10/2003	6/10/2003 (3)
595AJ (2nd clutch)	na	na	7/31/2003 (5)
943SI	na	na	9/4/2003 (6)

na = not available

Table 3. Organohalogen contaminant concentrations (ng/g wet mass) and percent total extractable organics (TEO) measured in fat and blubber of 7 stranded leatherback turtles (6 females, 1 male). The number of samples for each congener above the reporting limit (RL) is given (n > RL).

Tissue	Fat						Blubber					
	n > RL						n > RL					
Compound	Total n = 7	Mean	SD	Median	Min	Max	Total n = 7	Mean	SD	Median	Min	Max
PCB 66	6	1.39	1.40	1.57	<0.100	3.88	6	2.44	4.73	0.460	<0.087	12.9
PCB 101+90	7	2.28	1.74	2.01	0.353	5.48	6	2.50	3.35	1.68	<0.126	9.71
PCB 99	7	2.95	1.88	2.88	0.292	5.42	7	3.67	5.29	1.82	0.134	15.3
PCB 149	7	2.13	1.05	2.01	0.422	3.38	6	2.72	2.53	1.64	<0.411	8.19
PCB 107[b]	5	0.821	0.690	0.609	<0.256	2.08	3	2.59	6.08	0.006	<0.412	16.3
PCB 118	7	6.78	5.49	7.06	0.172	15.7	6	18.3	37.1	5.52	<0.234	101
PCB 146	6	3.87	2.31	3.63	<0.267	7.59	7	8.52	16.8	3.33	0.027	46.4
PCB 153+132	7	20.5	14.9	18.0	1.31	45.6	7	53.8	109	19.5	<0.557	299
PCB 105	7	2.05	1.69	2.06	0.017	4.47	7	5.59	11.5	1.56	0.042	31.6
PCB 138+163	7	10.1	7.28	10.1	0.234	20.2	6	22.7	44.1	9.53	<0.439	121
PCB 187+182	7	10.3	5.74	10.9	0.483	17.7	6	20.8	38.6	8.44	<0.421	107
PCB 183	7	2.54	1.58	2.10	0.473	5.17	7	5.66	11.1	2.03	0.077	30.7
PCB 128	6	1.36	1.06	1.17	<0.145	2.79	5	3.60	7.42	1.05	<0.356	19.8
PCB 201	6	1.38	0.895	1.87	<0.244	2.34	4	2.88	6.59	0.472	<0.412	16.8
PCB 180+193	7	5.71	4.16	4.76	0.648	13.2	6	14.2	29.9	4.20	<0.426	80.9
PCB 170	6	2.05	1.80	1.47	<0.256	5.45	5	6.22	14.6	1.09	<0.103	38.1
PCB 194	6	1.67	1.45	1.36	<0.278	4.44	4	4.13	10.5	0.150	<0.150	26.4
PCB 206	7	1.59	1.28	1.20	0.007	3.98	5	3.38	6.63	0.818	<0.411	17.9
Total PCBs	7	90.1	65.9	75.1	4.87	188	7	193	384	66.9	1.52	1061
Heptachlor epoxide[ab]	3	0.929	0.716	0.430	<0.225	2.29	3	1.73	2.79	0.170	<0.349	7.80
trans-Chlordane	0				<0.232	<0.262	0				<0.360	<0.440
cis-Chlordane	0				<0.225	<0.254	0				<0.348	<0.425
trans-Nonachlor	7	16.1	10.5	12.6	2.00	34.0	7	37.9	70.1	14.5	1.02	196
cis-Nonachlor	7	2.66	1.28	2.48	1.14	5.16	6	4.68	3.83	2.96	<0.394	12.3
Oxychlordane	5	3.58	1.86	2.62	<0.244	6.51	4	9.52	17.6	2.98	<0.393	46.8
Total chlordanes	7	22.4	14.4	19.6	3.14	47.3	7	52.2	93.6	23.3	3.11	263
HCB	7	0.628	0.363	0.743	0.121	1.18	7	0.700	0.335	0.657	0.323	1.11
Mirex	5	0.379	0.404	0.164	<0.046	0.944	1				<0.352	7.60
Dieldrin	7	4.41	1.92	4.71	2.16	7.92	7	8.39	10.3	4.67	2.92	31.6
4,4'-DDE	7	19.7	10.9	20.0	5.14	35.7	7	41.5	64.4	22.7	4.80	185
2,4'-DDD	0				<0.225	<0.254	0				<0.349	<0.425
4,4'-DDD	4	2.09	0.359	1.88	<0.227	2.59	5	2.61	1.12	3.00	<0.388	4.28
4,4'-DDT	7	1.12	0.724	0.934	0.227	2.50	7	1.07	0.420	1.21	0.421	1.62
Total DDTs	7	24.1	13.9	23.4	6.55	47.2	7	49.5	76.1	30.3	6.01	220
PBDE 47	6	5.06	2.08	4.50	<0.656	9.15	6	7.78	8.51	4.79	<1.08	26.5
PBDE 85[b]	5	0.863	0.228	0.826	<0.578	1.14	3	1.07	1.45	0.347	<0.844	4.24
PBDE 99	6	2.49	0.862	2.06	<1.37	4.26	4	3.65	3.39	2.10	<2.10	10.7
PBDE 100	6	2.19	0.762	2.10	<0.920	3.50	6	4.29	6.04	1.99	<1.51	17.8
PBDE 153	6	1.43	1.20	0.965	<0.588	3.94	4	3.95	7.78	1.12	<0.934	20.4
PBDE 154[b]	6	2.68	1.21	2.68	<1.43	4.61	3	4.93	10.1	0.106	<2.07	27.5
Total PBDEs	6	15.4	6.67	13.2	<1.67	26.0	6	25.7	41.1	9.99	<2.94	116
TEO content (%)	7	48.1	23.9	60.6	0.697	67.9	7	52.7	16.9	58.5	14.9	63.2

Notes: Summary statistics calculated by Kaplan–Meier (K-M) methods except for those congeners denoted by [a] (fat) and [b] (blubber), which were calculated using Regression on Order models (ROS). Values for individual turtles are reported in the Supporting Material (SM, Table S1).

Table 4. Organohalogen contaminant concentrations (ng/g wet mass) and percent total extractable organics (TEO) measured in blood and eggs of 6 nesting leatherback turtles. The sample size above the reporting limit (RL) is given for each compound (n > RL).

Tissue	Blood						Eggs					
	n > RL						n > RL					
Compound	Total n = 6	Mean	SD	Median	Min	Max	Total n = 6	Mean	SD	Median	Min	Max
PCB 66[a]	3	0.017	0.022	0.005	<0.002	0.054	6	0.169	0.137	0.101	0.011	0.321
PCB 101	5	0.030	0.012	0.026	<0.008	0.045	6	0.132	0.097	0.120	0.020	0.243
PCB 99	5	0.073	0.051	0.050	<0.007	0.143	6	0.345	0.278	0.192	0.036	0.684
PCB 107	0				<0.033	<0.237	6	0.098	0.071	0.061	0.017	0.173
PCB 118	6	0.131	0.104	0.083	0.010	0.279	6	0.663	0.570	0.313	0.021	1.37
PCB 146	6	0.090	0.073	0.059	0.009	0.208	5	0.344	0.313	0.141	<0.010	0.733
PCB 153+132	6	0.464	0.426	0.215	0.038	1.20	6	2.11	1.89	0.935	0.091	4.87
PCB 105	5	0.046	0.024	0.033	<0.016	0.085	5	0.210	0.167	0.111	<0.018	0.394
PCB 163	5	0.079	0.045	0.063	<0.019	0.159	6	0.583	0.447	0.338	0.075	1.12
PCB 138	6	0.272	0.209	0.139	0.029	0.591	6	0.974	0.799	0.517	0.069	2.12
PCB 158	6	0.011	0.009	0.006	0.003	0.028	5	0.027	0.017	0.020	<0.006	0.051
PCB 187	6	0.331	0.324	0.125	0.039	0.892	5	0.676	0.717	0.213	<0.017	1.90
PCB 183[b]	5	0.047	0.037	0.029	<0.008	0.119	3	0.091	0.127	0.042	<0.009	0.338
PCB 128	5	0.034	0.017	0.026	<0.013	0.062	4	0.108	0.096	0.028	<0.010	0.227
PCB 177	5	0.057	0.040	0.037	<0.007	0.115	5	0.101	0.078	0.053	<0.007	0.184
PCB 202	4	0.114	0.203	0.019	<0.007	0.505	4	0.146	0.249	0.021	<0.014	0.623
PCB 180+193	6	0.163	0.114	0.096	0.028	0.331	6	0.396	0.281	0.237	0.088	0.846
PCB 170[b]	5	0.056	0.031	0.037	<0.007	0.112	3	0.047	0.059	0.024	<0.006	0.159
PCB 199	4	0.116	0.183	0.033	<0.008	0.469	5	0.147	0.258	0.025	<0.009	0.659
PCB 203+196[a]	3	0.045	0.072	0.016	<0.005	0.188	4	0.128	0.184	0.036	<0.014	0.479
PCB 194	4	0.029	0.020	0.018	<0.004	0.066	4	0.035	0.040	0.012	<0.008	0.110
Total PCBs	6	2.50	2.27	1.62	0.162	6.54	6	8.45	7.59	4.15	0.441	19.9
Heptachlor	0				<0.013	<0.091	0				<0.089	<0.090
Heptachlor epoxide	2				<0.014	<0.098	6	0.219	0.091	0.183	0.096	0.362
trans-Chlordane	5	0.023	0.012	0.020	<0.002	0.036	5	0.148	0.073	0.117	<0.005	0.243
cis-Chlordane	4	0.010	0.009	0.005	<0.004	0.023	6	0.079	0.007	0.080	0.067	0.086
trans-Nonachlor	6	0.196	0.098	0.149	0.053	0.307	6	1.25	1.27	0.635	0.190	3.72
cis-Nonachlor	6	0.043	0.037	0.034	0.007	0.115	6	0.180	0.142	0.109	0.017	0.397
Oxychlordane	6	0.050	0.028	0.044	0.011	0.093	6	0.417	0.272	0.265	0.101	0.825
Total chlordanes	6	0.328	0.158	0.281	0.081	0.507	6	2.28	1.71	1.36	0.562	5.39
HCB[a]	3	0.054	0.042	0.038	<0.022	0.115	6	0.225	0.076	0.207	0.150	0.368
Pentachlorobenzene	0				<0.003	<0.021	0				<0.030	<0.031
Mirex	0				<0.009	<0.062	0				<0.083	<0.084
Dieldrin	6	0.140	0.100	0.097	0.040	0.328	6	0.535	0.347	0.450	0.132	1.16
4,4'-DDE	6	0.424	0.235	0.317	0.211	0.865	6	1.59	0.930	1.14	0.563	3.18
2,4'-DDD	0				<0.011	<0.079	0				<0.027	<0.027
2,4'-DDT+4,4'-DDD	0				<0.015	<0.106	4	0.233	0.050	0.199	<0.182	0.300
4,4'-DDT	0				<0.022	<0.156	6	0.116	0.036	0.115	0.059	0.172
Total DDTs	6	0.424	0.235	0.317	0.211	0.865	6	1.87	1.02	1.53	0.683	3.49
PBDE 47	4[c]	0.103	0.066	0.073	<0.019	0.214	6	0.486	0.327	0.469	0.073	0.804
PBDE 99	3[c]	0.025	0.016	0.022	<0.016	0.050	4	0.077	0.027	0.056	<0.018	0.109
PBDE 100	3[c]	0.040	0.026	0.027	<0.018	0.082	6	0.153	0.112	0.105	0.028	0.295
PBDE 153	1[c]				<0.009	<0.056	5	0.040	0.046	0.016	<0.008	0.126
PBDE 154[b]	3[c]	0.031	0.027	0.021	<0.020	0.076	3	0.090	0.101	0.054	<0.015	0.279
Total PBDEs	4[c]	0.198	0.190	0.155	<0.050	0.510	6	0.845	0.630	0.689	0.121	1.64
Parlar 26	0				<0.002	<0.012	6	0.023	0.009	0.017	0.016	0.035
Parlar 32	0				<0.002	<0.013	0				<0.002	<0.002
Parlar 50	0				<0.002	<0.013	6	0.025	0.007	0.021	0.017	0.035
Parlar 62	0				<0.006	<0.040	6	0.026	0.014	0.020	0.015	0.053
Total Toxaphenes							6	0.074	0.029	0.061	0.048	0.121
TEO content (%)	6	0.562	0.292	0.483	0.224	1.00	6	5.00	0.380	4.89	4.67	5.69

Notes: Summary statistics calculated by Kaplan–Meier (K-M) methods except for those congeners denoted by [a] (blood) and [b](egg), which were calculated using Regression on Order models (ROS). [c] PBDE blood values have a total n = 5 because one turtle (693PH) had low recoveries of these compounds. Values for individual turtles are reported in the Supporting Material (SM, Table 2).

Table 5. Comprehensive summary of organohalogen contaminants measured in fat (adipose), blubber, blood, and eggs of sea turtles as reported in the literature. Values presented are means (in ng/g wet mass unless otherwise noted) with SD (or SE) in parentheses below.

Species	Stage/ Sex	Tissue	Year	Location	Total PCBs	Total DDTs	4,4'-DDE	Total chlordanes	Dieldrin	Total PBDEs	Total toxaphenes	n	Reference
Leatherback turtle													
Dermochelys coriacea	AJMF	Fat	1999-2001	Southeast USA	90.1 (65.9)	24.1 (13.9)	19.7 (10.9)	22.4 (14.4)	4.41 (1.92)	14.1 (8.59)	ND	7	This study
Dermochelys coriacea	AJMF	Blubber	1999-2001	Southeast USA	193 (384)	49.3 (76.3)	41.5 (64.4)	52.2 (93.6)	8.39 (10.3)	24.9 (40.8)	ND	7	This study
Dermochelys coriacea	AF	Blood	2003	Juno Beach, FL, USA	2.50 (2.27)	0.424 (0.235)	0.424 (0.235)	0.328 (0.158)	0.140 (0.100)	0.198 (0.190)	ND	6	This study
Dermochelys coriacea	AF	Eggs	2003	Juno Beach, FL, USA	8.45 (7.59)	1.87 (1.02)	1.59 (0.930)	2.28 (1.71)	0.535 (0.347)	0.845 (0.630)	0.074 (0.029)	6	This study
Dermochelys coriacea	AF	Blood	2006	Yalimapo, French Guiana	1.26[a] (0.71)	0.31[a] (0.22)	0.2[a] (0.20)	NA	NA	ND	ND	44	Guirlet et al., 2010
Dermochelys coriacea	AF	Eggs	2006	Yalimapo, French Guiana	6.98 (5.02)	1.44 (1.26)	1.24 (1.24)	NA	NA	ND	ND	46	Guirlet et al., 2010
Dermochelys coriacea	AF	Fat	2002-2005	Canary Islands, Spain	77.00	NA	NA	NA	NA	ND	ND	1	Orós et al., 2009
Dermochelys coriacea	AF	Blood	2001-2002	Gabon	ND	ND	ND	ND	ND	ND	ND	9	Deem et al., 2006
Dermochelys coriacea	AM	Adipose	1993+1995	Scotland, U.K.	113 (47.0-178)	36.0 (14.0-58.0)	33.5 (10.0-57.0)	17.0 (12.0-22.0)	16.0 (13.0-19.0)	ND	ND	2	McKenzie et al. 1999
Dermochelys coriacea	AM	Adipose	1993-1996	Wales & Scotland, U.K.	152 (94.3)	ND	45.0 (35.8)	ND	23.0 (13.0-33.0)	ND	ND	3	Godley et al. 1998
Green turtle													
Chelonia mydas	JMF	Blood	2001-2002	Gulf of Mexico	0.534 (0.701)	0.128 (0.114)	0.0664* (0.110)	0.011 (0.0205)	0.096	0.158 (0.217)	NA	9	Swarthout et al., 2010
Chelonia mydas	JMF	Blood	2006-2007	Queensland, Australia	0.684 (SE=0.1528)	ND	ND	NA	ND	0.079 (SE=0.0108)	NA	16	van de Merwe et al., 2010a
Chelonia mydas	AF	Blood	2004	Terengganu, Malaysia	0.579 (0.0856)	NA	NA	NA	NA	0.121 (0.0141)	NA	11	van de Merwe et al., 2010b
Chelonia mydas	AF	Eggs	2004	Terengganu, Malaysia	0.554 (0.0546)	NA	NA	NA	NA	0.129 (0.0081)	NA	11	van de Merwe et al., 2010b
Chelonia mydas	J	Blood	2004	Terengganu, Malaysia	0.851 (0.1052)	NA	NA	NA	NA	0.083 (0.0144)	NA	11	van de Merwe et al., 2010b
Chelonia mydas	JF	Fat	2002-2005	Canary Islands, Spain	144.00	NA	NA	NA	NA	NA	NA	1	Orós et al., 2009
Chelonia mydas	AJMF	Blood	2004-2008	Queensland, Australia	NA	NA	NA	NA	NA	0.004	NA	7	Hermanussen et al., 2008
Chelonia mydas agassizii	J	Adipose		Baja California, Mexico	ND-49.5^	ND-12.2^	NA	ND-65.1^	ND	NA	NA	7	Gardner et al., 2003
Chelonia mydas	J	Adipose	1995	Cyprus, Greece	136 (113)	12.4 (9.93)	9.13 (8.73)	ND	ND	ND	ND	3	McKenzie et al., 1999
Chelonia mydas	AF	Eggs	1994-1996	Cyprus, Greece	6.10	4.30	2.30	NA	NA	NA	NA	1	McKenzie et al., 1999

Table 5 continued.

Species	Stage/ Sex	Tissue	Year	Location	Total PCBs	Total DDTs	4,4'-DDE	Total chlordanes	Dieldrin	Total PBDEs	Total toxaphenes	n	Reference
Loggerhead turtle													
Caretta caretta	JMF	Fat	2002-2005	Canary Islands, Spain	450.00 (1700)	NA	NA	NA	NA	NA	NA	30	Orós et al., 2009
Caretta caretta	AF	Egg yolk	2002	Florida, USA	144.00 (280)	50.20 (92.4)	314.00[b] (485)[b]	25.50 (46.7)	2.53 (2.76)	NA	NA	22	Alava et al., 2006
Caretta caretta	JMF	Plasma	1998-2006	North Carolina, USA	3.780 (5.810)	2.400 (3.610)	NA	0.031 (0.0423)	NA	0.066 (0.0719)	NA	45	Carlson, 2006
Caretta caretta	JMF	Plasma	2003	Southeast USA	2.530	NA	NA	NA	NA	0.131	NA	29	Keller et al., 2005
Caretta caretta	J	Blood	1998-2001	North Carolina, USA	5.140 (3.950)	0.583 (0.307)	0.576 (0.305)	0.260 (0.182)	0.046 (0.0292)	NA	NA	5	Keller et al., 2004a
Caretta caretta	JMF	Blood	2000-2001	North Carolina, USA	5.560 (5.280)	0.649 (0.685)	300.00[b] (578.00)[b]	0.225 (0.201)	0.061 (0.141)	NA	NA	44	Keller et al., 2004b
Caretta caretta	AF	Eggs	1994-1996	Cyprus, Greece	89.00	155.00	154.00	1.80	0.60	NA	NA	1	McKenzie et al., 1999
Caretta caretta	AF	Eggs (membrane)	1993	South Carolina, USA	10100[b] (SE=5466)	NA	NA	NA	NA	NA	NA	16	Cobb and Wood, 1997
Caretta caretta	AF	Eggs	1993	South Carolina, USA	1188[b] (SE=311)	NA	NA	NA	NA	NA	NA	16	Cobb and Wood, 1997
Kemp's ridley turtle													
Lepidochelys kempii	JMF	Blood	2001-2002	Gulf of Mexico	4.27 (3.620)	0.686 (0.656)	0.472* (0.633)	0.113 (0.100)	0.225 (0.119)	0.230 (0.273)	NA	46	Swarthout et al., 2010
Lepidochelys kempii	JMF	Blood	2001-2002	Southeast USA	10.70 (12.220)	1.49 (1.790)	0.733* (1.800)	1.22 (1.490)	0.608 (0.409)	0.148 (0.141)	NA	3	Swarthout et al., 2010
Lepidochelys kempii	JMF	Blood	1999	Massachusetts, USA	4.540 (5.760)	0.793 (0.678)	166.00[b] (147.00)[b]	0.356 (0.376)	0.083 (0.0751)	NA	NA	8	Keller et al., 2004b
Olive ridley turtle													
Lepidochelys olivacea	J	Adipose		Baja California, Mexico	18.40	5.10	NA	8.10	ND	NA	NA	1	Gardner et al., 2003
Flatback turtle													
Natator depressus	AF	Blood	2004-2008	Queensland, Australia	NA	NA	NA	NA	NA	0.006	NA	1	Hermanussen et al., 2008
Hawksbill turtle													
Eretmochelys imbricata	JF	Blood	2004-2008	Queensland, Australia	NA	NA	NA	NA	NA	0.013	NA	1	Hermanussen et al., 2008

Notes: A = adult, J = juvenile, M = male, F = female. NA = not available, ND = not detected. [a] = ng/mL blood, [b] = ng/g lipid, * = geometric mean, ^ = range was reported.

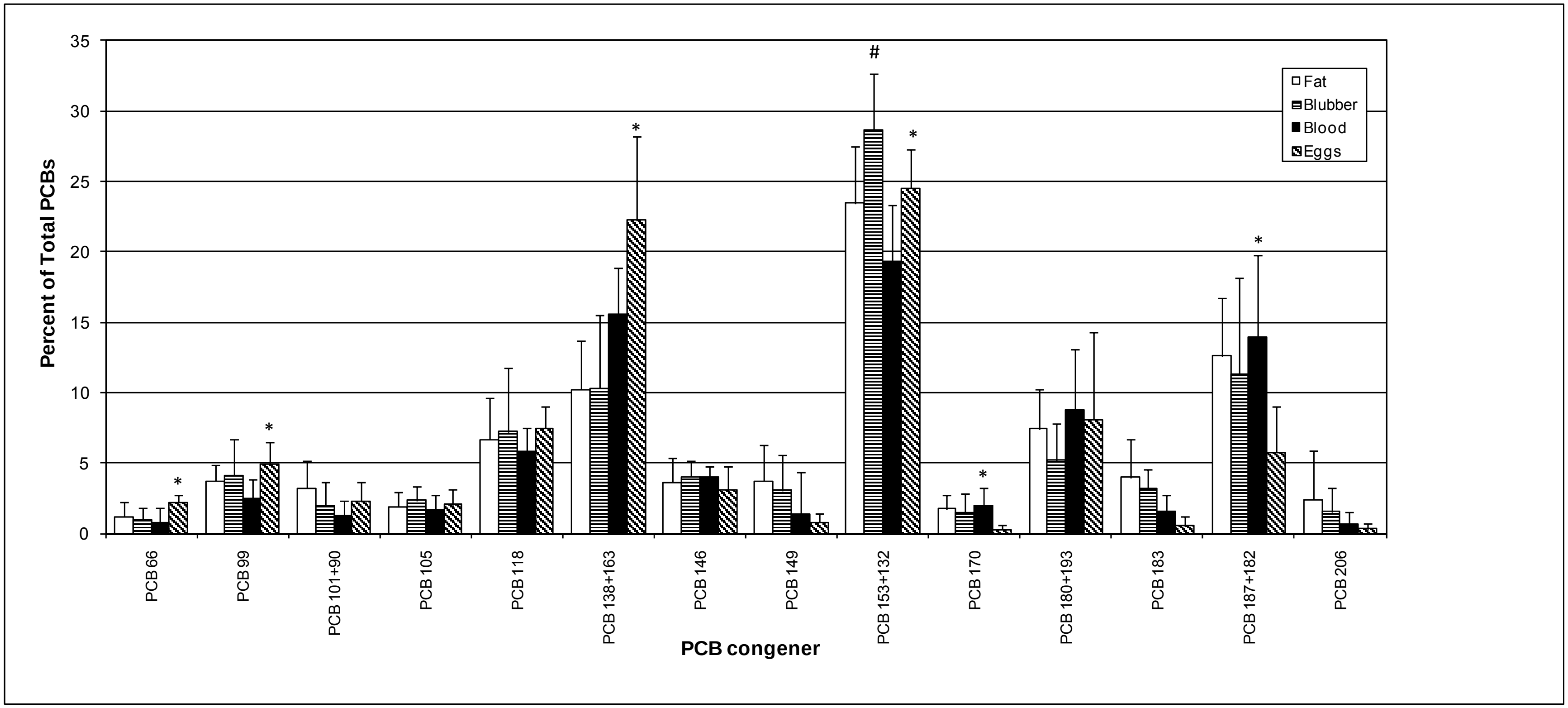

Figure 1. PCB profiles measured in leatherback fat (n = 7), blubber (n = 7), blood (n = 6), and eggs (n = 6). Only those congeners with > 2.0% contribution (in at least one of the tissue types) are shown. Error bars represent one standard deviation. * indicates a significant difference between blood and eggs. # indicates a significant difference between blubber and fat.

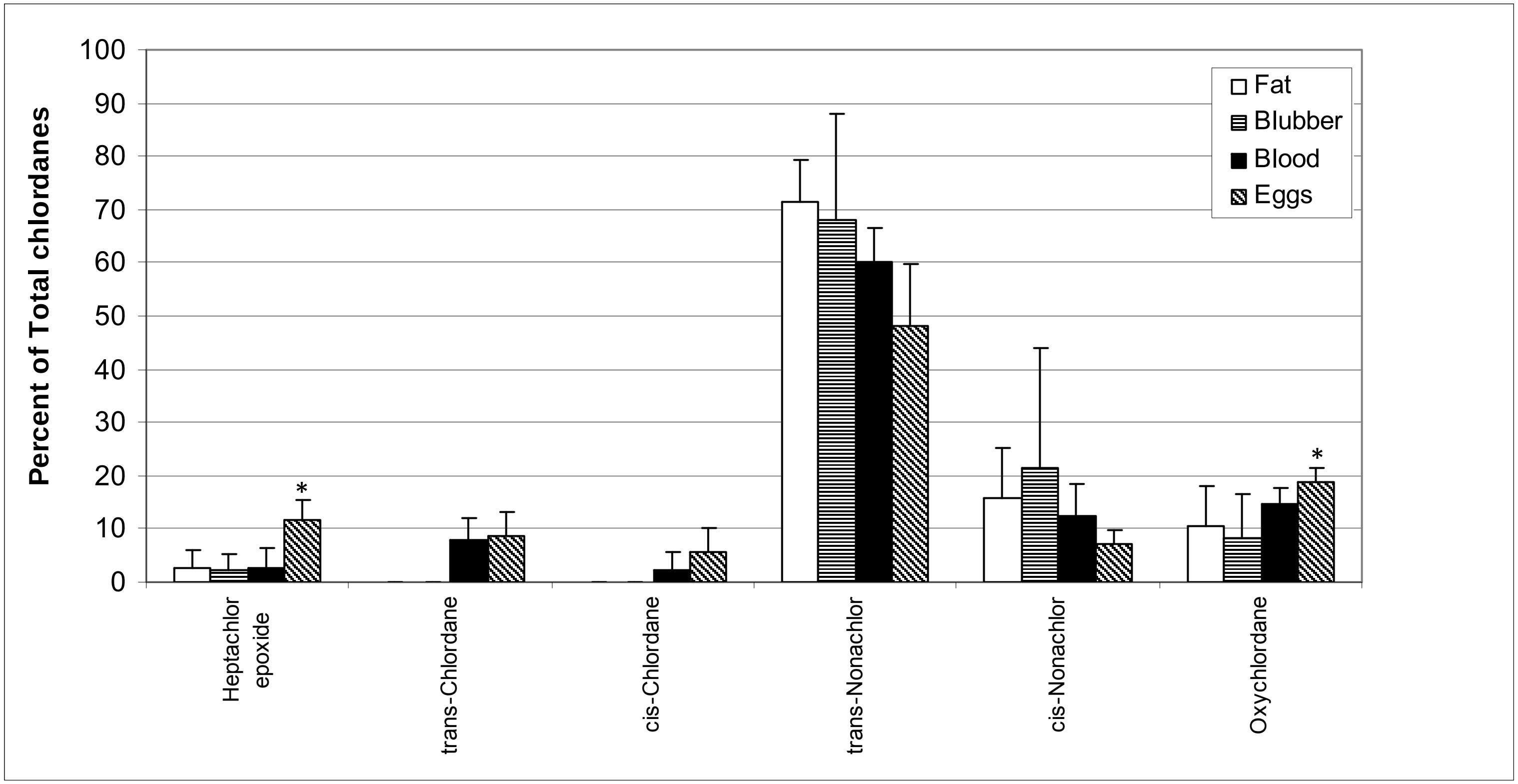

Figure 2. Chlordane profiles measured in leatherback fat (n = 7), blubber (n = 7), blood (n = 6), and eggs (n = 6). Error bars represent one standard deviation. * indicates a significant difference between blood and eggs. No significant differences were observed between blubber and fat.

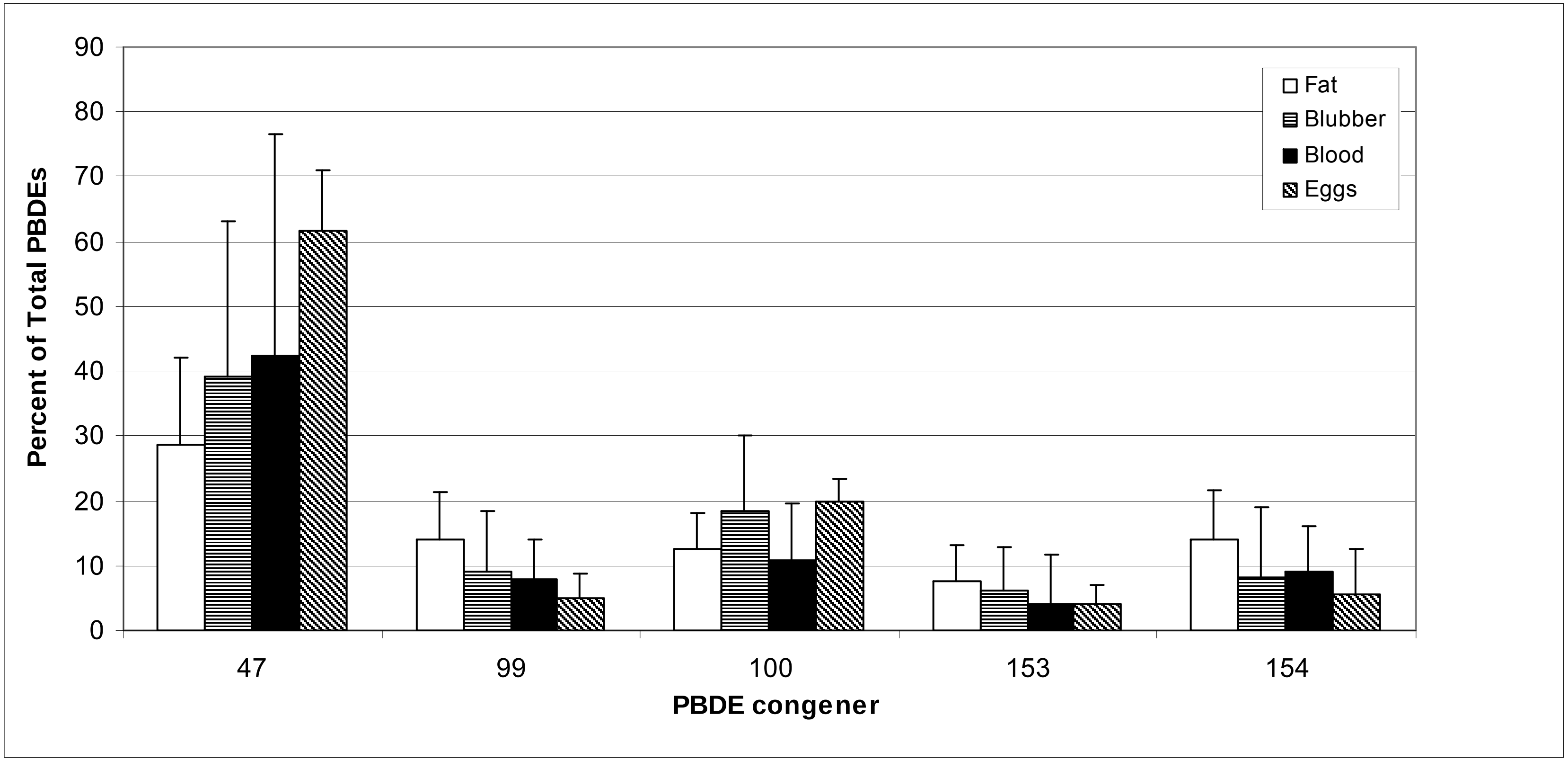

Figure 3. PBDE congener profiles for leatherback fat (n = 7), blubber (n = 7), blood (n = 6), and eggs (n = 6). Error bars represent one standard deviation. No significant differences were observed between blubber and fat or between blood and eggs.

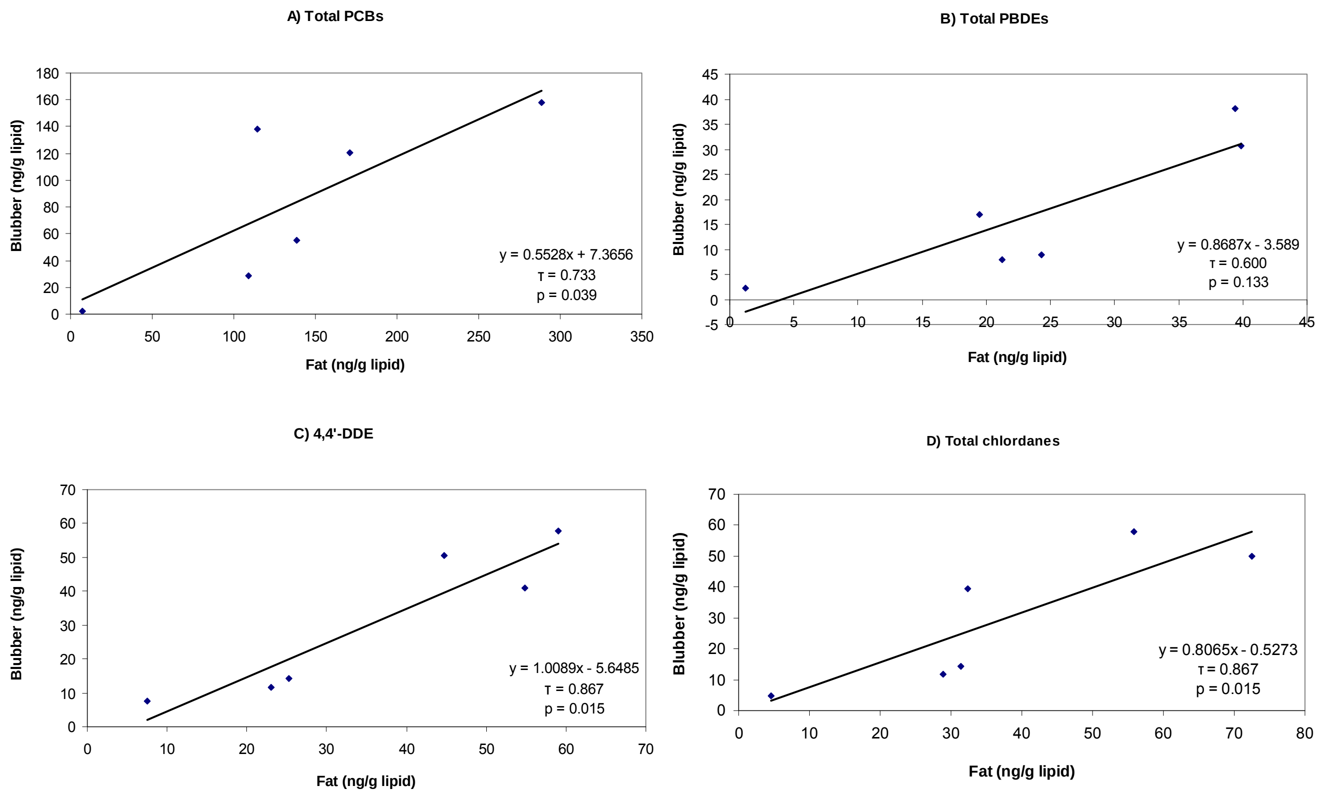

Figure 4. Relationships between fat and blubber samples from stranded leatherbacks for A) total PCBs, B) total PBDEs, C) 4,4'-DDE and D) total chlordanes (n = 6 paired samples; Note that stranded turtle Dc-TP-99-06-09 is not included in the correlations). Τ= Kendall's tau correlation coefficient, p-value, and linear trendline equation are given.

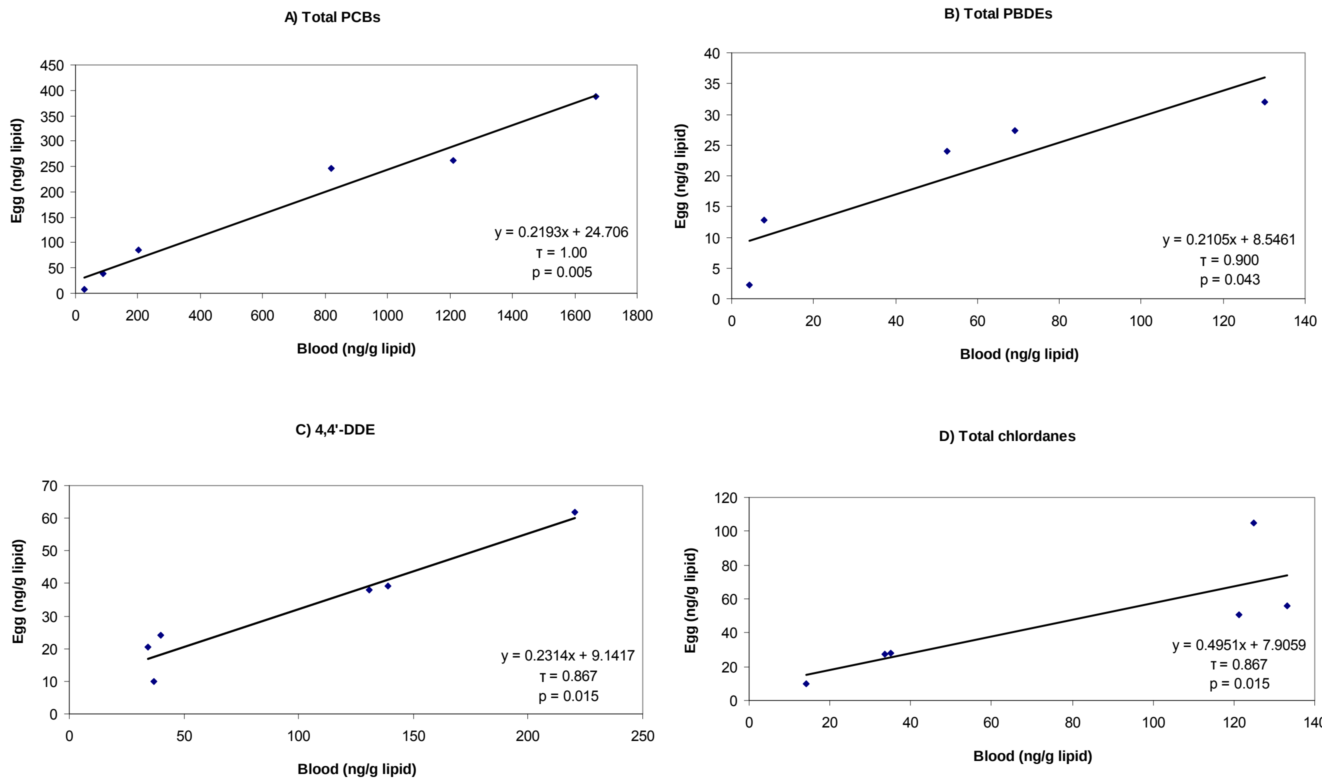

Figure 5. Significant positive correlations between blood from nesting female leatherbacks and their unhatched eggs for A) 4 total PCBs, B) total PBDEs, C) 4,4'-DDE and D) total chlordanes (n = 6 paired samples for PCBs, 4,4'-DDE and chlordanes, n = 5 for PBDEs; Note that Turtle 693PH is not included in the PBDE correlation because of low recovery of the internal standards in her blood sample). Τ= Kendall's tau correlation coefficient, p-value, and linear trendline equation are given.

Supporting Material

SM, Table 1. Organohalogen contaminant concentrations (ng/g wet mass) and percent total extractable organics (TEO) in fat and blubber of individual stranded turtles (n = 7). RL = reporting limit.

Turtle ID	Dc-TP-99-06-09		Dc-RB-99-06-24		Dc-SC-01-05-19-01		Dc-SAJ-01-05-31-01		Dc-WMC-02-05-16-01		Dc-MG-02-05-26-01		Dc-03-05-30-01 SC	
Tissue	Fat	Blubber	Fat	Blubber	Fat	Blubber	Fat	Blubber	Fat	Blubber	Fat	Blubber	Fat	Blubber
Compound														
PCB 66	1.58	12.9	3.88	1.84	<RL	<RL	1.57	1.60	2.22	0.116	0.100	0.087	0.219	0.460
PCB 101+90	2.16	9.71	5.48	2.33	0.353	<RL	2.01	2.19	3.47	1.68	0.732	0.125	1.77	1.32
PCB 99	4.92	15.3	5.42	3.75	0.292	0.134	2.88	3.06	3.40	1.82	0.967	0.654	2.80	0.991
PCB 149	1.61	8.19	3.38	2.41	0.422	<RL	1.57	1.56	2.83	1.64	2.01	1.16	3.13	2.93
PCB 107	2.08	16.3	1.35	0.637	<RL	<RL	0.767	1.15	0.609	<RL	<RL	<RL	0.32	<RL
PCB 118	15.7	101	11.3	9.02	0.172	<RL	7.39	8.70	7.06	3.63	1.00	0.234	4.87	5.52
PCB 146	7.59	46.4	5.77	3.93	<RL	0.027	3.63	3.84	4.26	1.60	1.31	0.550	3.24	3.33
PCB 153	45.6	299	32.7	22.2	1.31	0.557	18.0	20.3	21.1	10.2	7.22	4.45	17.8	19.5
PCB 105	4.47	31.6	3.85	2.65	0.017	0.042	2.29	2.45	2.06	0.703	0.223	0.051	1.41	1.56
PCB 138+163	20.2	121	17.8	11.7	0.234	<RL	10.1	9.53	10.9	4.80	2.63	1.23	8.61	9.73
PCB 187+182	17.7	107	16.1	9.45	0.483	<RL	7.60	8.44	11.0	3.70	8.32	3.85	10.9	9.60
PCB 183	5.17	30.7	3.93	3.58	0.473	0.077	1.38	2.03	2.10	0.261	1.97	0.57	2.75	2.45
PCB 128	2.79	19.8	2.53	1.66	<RL	<RL	1.17	1.63	1.61	0.356	0.145	<RL	1.13	1.05
PCB 201	2.34	16.8	2.10	0.647	<RL	<RL	0.244	0.792	0.967	<RL	1.88	<RL	1.87	0.472
PCB 180	12.9	78.6	8.53	5.93	0.648	<RL	3.98	5.22	4.67	1.28	2.83	0.890	5.46	4.20
PCB 193	0.24	2.32	0.450	<RL	<RL	<RL	0.106	0.133	0.094	<RL	<RL	<RL	0.064	<RL
PCB 170	5.5	38.1	3.42	1.88	<RL	<RL	1.27	2.16	1.47	0.103	0.585	<RL	1.56	1.09
PCB 194	4.44	26.4	2.42	0.728	<RL	<RL	0.404	1.20	0.813	<RL	1.88	<RL	1.36	0.150
PCB 206	1.16	17.9	2.27	1.06	0.007	<RL	0.695	1.45	1.20	<RL	3.98	0.818	1.82	0.815
Total PCBs	167	1061	188	87.7	4.87	1.52	69.5	81.5	86.0	34.3	39.6	16.6	75.1	66.9
Heptachlor epoxide	<RL	7.80	2.29	1.96	<RL	<RL	1.35	<RL	1.15	<RL	<RL	<RL	<RL	1.67
trans-Chlordane	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
cis-Chlordane	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
trans-Nonachlor	23.7	196	34.0	18.8	2.00	1.02	12.6	14.5	11.4	5.33	9.65	8.46	19.2	21.2
cis-Nonachlor	2.48	12.3	5.16	2.96	1.14	2.09	2.26	4.94	3.10	2.02	1.70	<RL	2.76	6.36
Oxychlordane	6.51	46.8	5.77	4.04	<RL	<RL	3.35	3.87	2.28	<RL	<RL	<RL	2.62	2.98
Total chlordanes	32.7	263	47.3	27.7	3.14	3.11	19.6	23.3	17.9	7.35	11.4	8.46	24.5	32.2
HCB	0.121	1.11	1.18	0.907	0.796	1.08	0.743	0.657	0.835	0.397	0.309	0.420	0.413	0.323
mirex	0.89	7.60	0.944	<RL	<RL	<RL	<RL	<RL	0.164	<RL	0.046	<RL	0.397	<RL
dieldrin	4.87	31.6	7.92	6.42	2.42	2.92	4.71	4.67	3.93	3.33	2.16	3.15	4.89	6.68
4,4'-DDE	20.0	185	35.7	22.7	5.14	4.80	27.1	29.8	15.7	8.78	8.34	6.80	25.9	32.1
2,4'-DDD	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
4,4'-DDD	<RL	4.28	2.59	3.00	<RL	<RL	1.94	1.54	2.57	3.08	<RL	<RL	1.88	3.26
4,4'-DDT	0.23	0.579	2.50	1.27	1.37	1.21	0.73	0.42	1.33	1.62	0.744	1.23	0.934	1.17
Total DDTs	21.3	220	47.2	30.3	6.55	6.01	31.9	32.2	23.4	13.5	9.09	6.80	29.0	36.6
PBDE 47	5.00	26.5	9.15	5.68	<RL	<RL	4.50	4.79	4.01	3.33	3.33	3.45	6.13	7.40
PBDE 85	1.08	4.24	1.05	0.931	<RL	<RL	0.651	<RL	1.14	<RL	<RL	<RL	0.826	1.22
PBDE 99	2.47	10.7	4.26	2.82	<RL	<RL	1.89	2.10	2.06	<RL	2.05	<RL	2.81	3.60
PBDE 100	2.85	17.8	3.50	2.31	<RL	<RL	1.86	1.99	2.10	1.62	1.45	1.73	2.17	3.02
PBDE 153	3.94	20.4	1.91	1.39	<RL	<RL	0.883	1.12	1.07	<RL	0.631	<RL	0.965	1.37
PBDE 154	4.61	27.5	3.80	2.98	<RL	<RL	1.99	<RL	2.80	<RL	0.720	<RL	2.68	3.56
Total PBDEs	21.7	116	26.0	17.0	<RL	<RL	11.8	9.99	13.2	4.94	8.81	5.18	17.3	21.2
TEO content (%)	0.697	14.9	65.2	55.4	67.9	63.2	60.6	59.0	62.1	62.3	36.2	58.5	43.9	55.5

SM, Table 2. Organohalogen contaminant concentrations (ng/g wet mass) and percent total extractable organics (TEO) in blood and eggs of individual nesting leatherbacks (n = 6). RL = reporting limit.

Turtle ID	595AJ			456RE		617MA		693PH		622CL		567CO		943SI
Tissue	Blood	Eggs (1)	Eggs (2)	Blood	Eggs	Blood	Eggs	Blood	Eggs	Blood	Eggs	Blood	Eggs	Eggs
Compound														
PCB 66	0.036	0.101	0.116	<RL	0.234	<RL	0.011	<RL	0.038	0.054	0.321	0.006	0.310	0.047
PCB 101	0.045	0.120	0.139	0.026	0.137	<RL	0.020	0.017	0.030	0.037	0.242	0.037	0.243	0.035
PCB 99	0.050	0.192	0.238	0.143	0.684	<RL	0.036	0.027	0.076	0.069	0.563	0.125	0.522	0.093
PCB 107	<RL	0.061	0.070	<RL	0.173	<RL	0.017	<RL	0.026	<RL	0.157	<RL	0.154	0.032
PCB 118	0.083	0.313	0.416	0.279	1.37	0.010	0.021	0.071	0.147	0.108	1.11	0.234	1.03	0.173
PCB 146	0.059	0.141	0.184	0.208	0.733	0.009	<RL	0.032	0.040	0.104	0.593	0.129	0.518	0.054
PCB 153+132	0.215	0.935	1.23	1.20	4.87	0.038	0.091	0.195	0.503	0.450	3.34	0.685	2.92	0.628
PCB 105	0.039	0.111	0.132	0.085	0.394	<RL	<RL	0.027	0.040	0.033	0.348	0.065	0.326	0.047
PCB 163	0.063	0.338	0.356	0.159	1.12	<RL	0.075	0.041	0.157	0.073	0.947	0.097	0.858	0.203
PCB 138	0.139	0.517	0.684	0.591	2.12	0.029	0.069	0.132	0.301	0.395	1.46	0.346	1.37	0.365
PCB 158	0.028	0.020	0.021	0.006	0.051	0.009	<RL	0.012	0.010	0.003	0.034	0.006	0.037	0.011
PCB 187	0.125	0.213	0.305	0.892	1.90	0.039	<RL	0.082	0.101	0.451	0.940	0.398	0.799	0.128
PCB 183	0.037	<RL	0.016	0.119	0.338	<RL	<RL	0.029	<RL	0.028	0.103	0.038	0.074	<RL
PCB 128	0.036	0.028	0.054	0.062	0.227	<RL	<RL	0.019	<RL	0.026	0.180	0.042	0.160	<RL
PCB 177	0.037	0.053	0.070	0.115	0.184	<RL	<RL	0.017	0.021	0.078	0.171	0.078	0.152	0.025
PCB 202	0.019	0.021	0.027	0.505	0.623	<RL	<RL	<RL	<RL	0.063	0.106	0.057	0.087	<RL
PCB 180+193	0.096	0.237	0.260	0.331	0.846	0.028	0.088	0.080	0.185	0.216	0.544	0.228	0.479	0.237
PCB 170	0.037	<RL	<RL	0.112	0.159	<RL	<RL	0.035	<RL	0.051	0.063	0.068	0.040	<RL
PCB 199	0.033	0.025	0.036	0.469	0.659	<RL	<RL	<RL	0.016	0.073	0.090	0.052	0.077	0.018
PCB 203+196	0.045	0.036	0.036	0.188	0.479	<RL	<RL	<RL	<RL	<RL	0.100	0.029	0.081	0.017
PCB 194	0.032	0.012	0.016	0.066	0.110	<RL	<RL	0.022	<RL	<RL	0.035	0.018	0.030	<RL
Total PCBs	1.62	4.15	4.92	6.54	19.9	0.162	0.441	0.878	1.77	2.70	12.9	3.12	11.5	2.24
Heptachlor	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
Heptachlor epoxide	<RL	0.183	0.181	<RL	0.362	<RL	0.096	<RL	0.175	0.025	0.241	0.030	0.259	0.224
trans-Chlordane	0.033	0.145	0.151	<RL	<RL	0.009	0.078	0.029	0.117	0.020	0.223	0.036	0.243	0.143
cis-Chlordane	0.023	0.086	0.081	0.018	0.080	<RL	0.080	<RL	0.067	0.004	0.079	0.005	0.085	0.073
trans-Nonachlor	0.149	0.573	0.803	0.292	3.72	0.053	0.190	0.229	0.635	0.145	1.15	0.307	1.22	0.791
cis-Nonachlor	0.041	0.109	0.144	0.115	0.397	0.007	0.017	0.034	0.067	0.026	0.233	0.036	0.259	0.081
Oxychlordane	0.034	0.265	0.289	0.066	0.825	0.011	0.101	0.044	0.205	0.051	0.559	0.093	0.546	0.274
Total chlordanes	0.281	1.36	1.65	0.490	5.39	0.081	0.562	0.336	1.27	0.271	2.48	0.507	2.61	1.59
HCB	<RL	0.207	0.219	0.022	0.150	<RL	0.215	0.112	0.368	0.038	0.181	0.068	0.229	0.431
Pentachlorobenzene	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
Mirex	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
Dieldrin	0.097	0.339	0.488	0.328	1.16	0.040	0.132	0.144	0.450	0.091	0.578	0.141	0.549	0.697
4,4'-DDE	0.317	1.14	1.43	0.865	3.18	0.211	0.563	0.343	0.952	0.311	1.92	0.498	1.79	1.27
2,4'-DDD	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
2,4'-DDT+4,4'-DDD	<RL	0.213	0.241	<RL	0.199	<RL	<RL	<RL	<RL	<RL	0.286	<RL	0.300	<RL
4,4'-DDT	<RL	0.172	0.132	<RL	0.115	<RL	0.120	<RL	0.059	<RL	0.106	<RL	0.127	0.121
Total DDTs	0.317	1.53	1.80	0.865	3.49	0.211	0.683	0.343	1.01	0.311	2.31	0.498	2.22	1.39
PBDE 47	0.064	0.469	0.462	0.214	0.789	<RL	0.073		0.114	0.073	0.804	0.099	0.667	0.153
PBDE 99	<RL	0.056	0.066	0.050	0.109	<RL	<RL		<RL	0.016	0.107	0.022	0.077	<RL
PBDE 100	<RL	0.105	0.112	0.082	0.295	<RL	0.028		0.038	0.027	0.239	0.038	0.214	0.045
PBDE 153	<RL	0.016	<RL	0.035	0.126	<RL	0.008		<RL	<RL	0.040	<RL	0.043	<RL
PBDE 154	<RL	<RL	<RL	0.076	0.279	<RL	<RL		<RL	0.020	0.113	0.021	0.088	<RL
Total PBDEs	0.064	0.689	0.655	0.510	1.64	<RL	0.121		0.151	0.155	1.34	0.200	1.13	0.198
Parlar 26	<RL	0.017	0.025	<RL	0.035	<RL	0.019	<RL	0.033	<RL	0.016	<RL	0.017	0.041
Parlar 32	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL	<RL
Parlar 50	<RL	0.023	0.029	<RL	0.032	<RL	0.021	<RL	0.035	<RL	0.017	<RL	0.020	0.043
Parlar 62	<RL	0.021	0.031	<RL	0.029	<RL	0.020	<RL	0.053	<RL	0.015	<RL	0.016	0.056
Total Toxaphenes	<RL	0.061	0.085	<RL	0.096	<RL	0.061	<RL	0.121	<RL	0.048	<RL	0.054	0.140
TEO content (%)	0.798	4.89	5.68	0.392	5.13	0.574	5.69	1.00	4.67	0.224	4.90	0.381	4.69	5.37

Notes: 595AJ had 2 successive clutches, and only the first clutch, which was paired with the only blood sample from this turtle, was included in the means in Table 4 and in the correlations for blood vs. eggs. 943SI represents a clutch that was not paired with a blood sample, and this clutch was not included in the summary data shown in Table 4.