REPRODUCTIVE SUCCESS AND EGG CONTAMINANT CONCENTRATIONS OF SOUTHERN NEW JERSEY PEREGRINE FALCONS

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Endangered and Nongame Species Program
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PREFACE

The information presented in this report documents a contaminants evaluation of non-viable peregrine falcon eggs collected from eyries on the Delaware River and Atlantic coast of southern New Jersey. The study was initiated to evaluate the potential reproductive effects of environmental contaminants in this species. This study was a cooperative project between the U.S. Fish and Wildlife Service and the New Jersey Division of Fish, Game and Wildlife.

Questions, comments, and suggestions related to this report are encouraged and should be submitted in writing to the following address:

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

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ABSTRACT

Peregrine falcon (*Falco peregrinus*) eyries in the State of New Jersey averaged 1.38 fledglings per active nest from 1979 to 1988. The Edwin B. Forsythe National Wildlife Refuge's (NWR) Barnegat eyrie failed to produce a single fledgling during this period, despite consistent nesting attempts. Poor reproductive performance at the Barnegat eyrie prompted the U.S. Fish and Wildlife Service and the New Jersey Division of Fish, Game and Wildlife to investigate potential interaction between contaminants and peregrine falcon reproductive performance.

The objective of this study was to test the hypothesis that TCDD equivalent concentrations (TEQs) of planar PCB congeners, PCDDs, and PCDFs are higher in eggs from eyries exhibiting low productivity. In addition, other organochlorines and inorganic contaminants were examined as potential contributors to reproductive risks. Non-viable eggs from the Edwin B. Forsythe NWR and seven other New Jersey eyries were analyzed for planar PCB congeners, PCDDs, PCDFs, inorganics, and organochlorine pesticides. Reproductive data were summarized to evaluate the relationship between the levels of contaminants in peregrine falcon eggs and productivity.

Non-ortho PCBs (specifically PCB 77 and PCB 126) were present at elevated levels in Atlantic coast eggs relative to those of the Delaware River. Based on TEQs, non-ortho PCBs (PCB 77 and PCB 126) were the most likely contributors to observed hatching failure on the Atlantic coast. Based on a raptor-specific reference toxicity value, levels of PCB 126 alone were sufficient to impact hatching success in all Atlantic coast nests, but not Delaware River nests. Total TEQ concentrations on the Atlantic coast of New Jersey were among the highest ever documented in eggs of this species (318 to 757 pg/g fresh weight). These findings contrast with those of the Delaware River eyries where total TEQ concentrations (90 to 175 pg/g) were generally below levels of concern. It is evident that the congener profiles in the Atlantic coast eggs were substantially different from those of the Delaware River.

Concentrations of mercury and DDE were also detected in some eggs at levels which may adversely affect peregrine falcon reproductive success. The mean mercury concentration at Atlantic coast eyries (0.38 ug/g fresh weight) was more than 30 times greater than that at Delaware River eyries (0.012 ug/g). The DDE concentration at some eyries indicated that DDE may continue to be a reproductive hazard for some of these birds (maximum of 24.2 ug/g at the Tuckahoe nest).

While detected concentrations of PCBs, PCDDs, PCDFs, DDE, and mercury in many eggs were theoretically sufficient to impact peregrine falcon reproduction, the productivity and eggshell thinning data do not support a conclusion of reproductive impairment in New Jersey peregrine falcons due to contaminants. Multiple regression analysis revealed no correlation between concentrations of DDE, PCB 77, PCB 126, total PCBs, and total TEQs in eggs and eggshell thickness data from the same eggs. If contaminants are impacting reproduction, the effect is minor when compared to influences such as nest

location (bridge vs. tower), disturbance, predation, etc., and would probably be undetectable at the sample sizes involved in this study. Further congener-specific organochlorine and inorganic monitoring of peregrines, their prey and supporting food chain is recommended to identify exposure pathways and elucidate reasons for the large differences in exposure between Atlantic coast and Delaware River eyries.

I. INTRODUCTION

The peregrine falcon (*Falco peregrinus*) is listed as an endangered species by the federal government and the States of New Jersey and Pennsylvania. Since the extirpation of the eastern subpopulation of this species in the mid 1960s, efforts to reestablish breeding populations of peregrine falcons along the eastern seaboard have resulted in nesting productivity levels above the minimum necessary to sustain a viable population. Eyries in the State of New Jersey averaged 1.38 fledglings per active nest from 1979 to 1988 (New Jersey Division of Fish, Game and Wildlife 1995). Some eyries, however, experienced much poorer reproductive success during that period. Most notable of these is the eyrie at Edwin B. Forsythe National Wildlife Refuge's (NWR) Barnegat Division (Barnegat), which failed to produce a single fledgling during a nine-year period (1984-92) marked by consistent nesting attempts and egg laying (New Jersey Division of Fish, Game and Wildlife 1995).

Bioaccumulation of polychlorinated biphenyls (PCBs), 1,1,1-trichloro-2,2-bis [p-chlorophenyl]ethane (DDT) metabolites (especially 1,1-dichloro-2,2-bis (p-chlorophenyl)ethylene (DDE)), and mercury by the peregrine falcons has been suggested as contributing to the reproductive impairment of some birds (U. S. Fish and Wildlife Service 1991a, 1991b). Substantial PCB, DDE and mercury levels have been measured in carcasses of adult peregrine falcons and in peregrine eggs in New Jersey. Analysis of eggs from Forsythe NWR's Brigantine eyrie have indicated DDE levels among the highest recorded on the East Coast (Steidl 1990).

In 1989, the U.S. Fish and Wildlife Service's New Jersey Field Office (NJFO) conducted a survey of prey species local to the Forsythe NWR eyries in an attempt to assess the organochlorine and elemental contaminant contribution to reproductive impairment (U.S. Fish and Wildlife Service 1991b). While total PCB, DDE, and mercury levels were found to be sufficient to elicit concern for reproductive health, the residue levels were similar at the various collection sites. Data failed to explain the differences in productivity among the eyries. It was hypothesized that these differences might be attributable to differences in levels of planar halogenated hydrocarbons (PHHs), which were not analyzed in the above study. The PHHs include the planar PCB congeners, polychlorinated dibenzo-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs). PHHs are aromatic compounds that contain more than one phenyl ring in a planar configuration. The molecular size and three-dimensional structure of PHH compounds allow them to bind to the aromatic hydrocarbon (Ah) receptor, a cellular protein (Nebert 1989). This receptor-mediated mode of action is thought to cause altered biochemical homeostasis, embryotoxicity, and teratogenesis in wildfowl populations (Poland and Knutson 1982, as cited in Tillitt et al. 1992).

The compound 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD or dioxin) has been demonstrated to have the greatest Ah receptor affinity and is the most potent mixed function oxidase system inducer (Ahlborg et al. 1992). Its isosteres, such as the mono-ortho-chloro-substituted (one chlorine substituted in the ortho-position of the phenyl rings) and non-ortho-chloro-substituted PCBs (which are commonly referred to as the dioxin-like PCBs), also elicit AHH induction and toxicity in direct correlation to their planarity (Kafafi et al.

1993, as cited in Barron et al. 1995). The common mechanism of action of PHH compounds has enabled researchers to examine TCDD isosteres by observing their potency relative to TCDD. Observations of AHH induction in numerous *in vivo* and *in vitro* toxicity studies have permitted assignment of TCDD toxic equivalency factors (TEFs) which represent the potency of each PHH relative to TCDD. TEFs are used to convert individual detected PHH concentrations to their TCDD equivalent concentrations. The sum of all TCDD equivalent concentrations is the total TCDD equivalence (TEQ) (Safe 1990).

In addition to DDE, which entered the environment as the result of extensive DDT application, PHHs may impact peregrine reproduction because of their prevalence in the industrialized centers of northern New Jersey and Philadelphia. Many of these compounds, such as the PCBs, have been used extensively as dielectric fluids, flame retardants, plasticizers, and lubricants. Others, such as the PCDDs and PCDFs, were formed as by-products in industrial processes such as chlorophenol and herbicide production. These compounds are also formed during municipal and hospital waste incineration and are present in fly ash and former fly ash disposal areas (Safe 1990, Bosveld and Van Den Berg 1994).

It has been suggested that an interaction of PHHs and DDE in New Jersey peregrine eggs may cause altered reproductive behavior and eggshell thinning (U. S. Fish and Wildlife Service 1991a). The objective of this U.S. Fish and Wildlife Service study was to test the hypothesis that concentrations of planar PCB congeners, PCDDs, and PCDFs (expressed as 2,3,7,8-TCDD equivalents) are higher in eggs from eyries exhibiting low productivity. This study additionally examined levels of organochlorine pesticides and elemental contaminants in peregrine falcon eggs.

II. METHODS

A. SAMPLE COLLECTION, STORAGE AND TRANSPORTATION

Non-viable peregrine falcon eggs were collected at nesting sites in New Jersey (including bridges between New Jersey and Pennsylvania) during the 1990 and 1991 nesting seasons (general locations are presented in Figure 1). A total of 17 eggs were collected from nine eyries (ten nesting events) as soon as possible after known nesting failures or during routine visits within the nesting season (Table 1). Whole eggs were frozen or refrigerated until processed. Whole egg mass, length, and breadth were measured prior to harvest into tared, chemically clean jars. The mass of egg contents was then recorded. Shells were washed with water, and air dried. Shell thickness was measured at several points near the equator of each egg with a dial-gauge micrometer graduated in units of 0.01 millimeter. The reported eggshell thicknesses are averages of a minimum of five replicate measurements on each egg (in accordance with the procedures identified by Wiemeyer et al. 1988). Samples were frozen, shipped, and received on dry ice and maintained in their frozen state until the initiation of the analytical process.

B. ANALYTICAL METHODS

Chemical analyses were performed by the National Fisheries Contaminant Research Center, U.S. Fish and Wildlife Service, Columbia, Missouri (now the Midwest Science Center, Biological Resources Division, U.S. Geological Survey).

1. Inorganics

Analyses for cadmium, copper, and lead were conducted with the technique of Zeeman graphite furnace atomic absorption spectroscopy. Zinc content was determined by flame atomic absorption spectroscopy. Mercury content was determined by cold vapor absorption spectroscopy. Method detection limits (MDLs) are reported in Appendix A.

2. Organics

Sample extraction, lipid determination, cleanup and fractionation, and instrumental procedures have been described in detail elsewhere (Schmitt et al. 1990, Schwartz and Stalling 1991, Smith et al. 1990, Feltz et al. 1995, Keith et al. 1983). Analytes were determined by high resolution gas chromatography-electron capture detection following the methods of Schwartz and Stalling (1991) for congener-specific analysis, and Smith et al. (1990) for planar PCB congeners. Non-ortho PCBs were determined by capillary gas chromatography-high resolution mass spectrometry. Dioxins and furans were determined by gas chromatography-quadrupole mass spectrometry, monitoring sequential mass windows of 12 selected ions during the chromatographic separation.

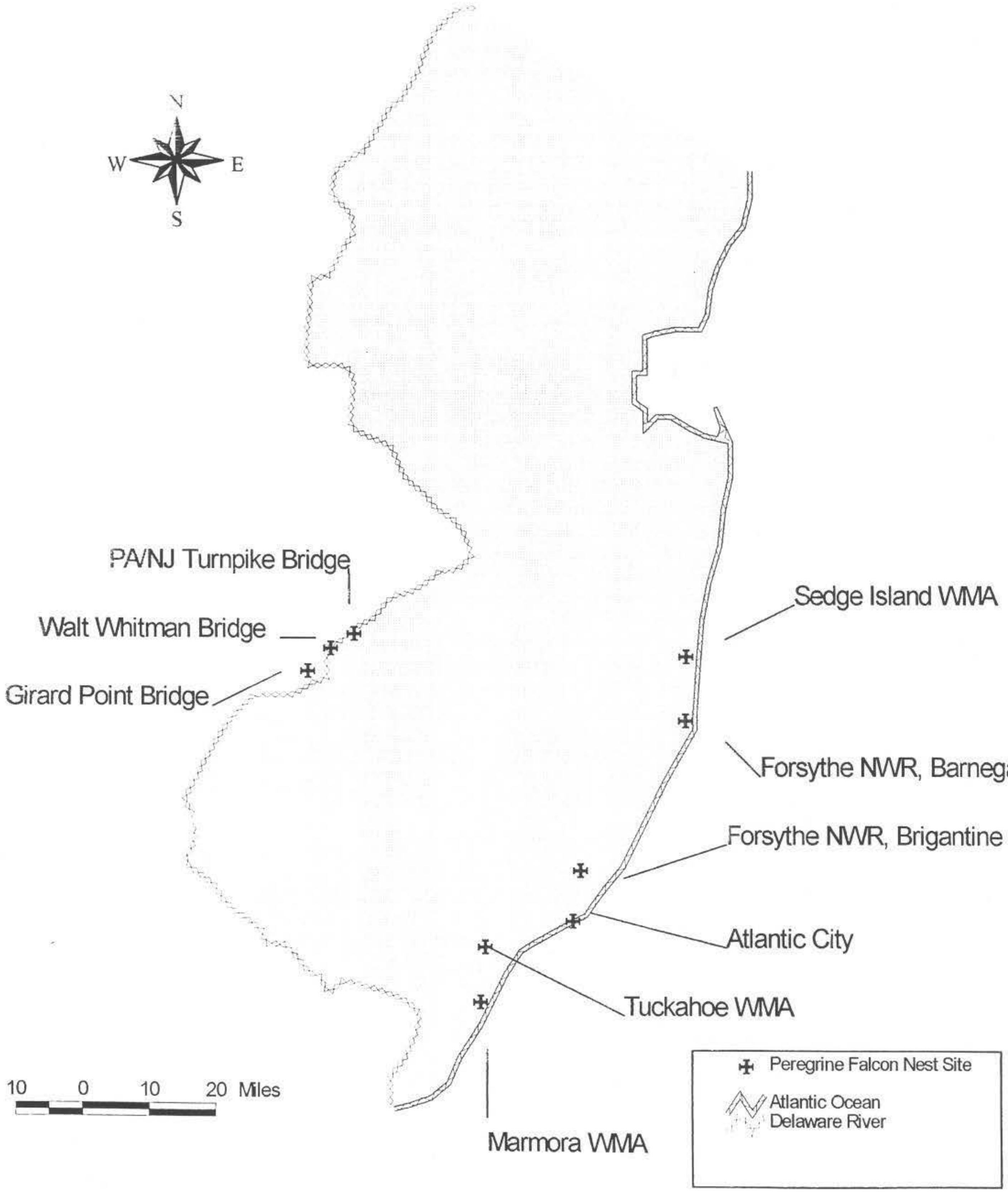

Figure 1. Peregrine falcon egg collection sites, 1990-1991.

Table 1. Peregrine Falcon Eggs Collected in New Jersey, 1990 - 1991.

Nest site	Sample ID	Date collected	Length (mm)	Width (mm)	Whole egg weight (g)	Content weight (g)	Calculated fresh weight (g)
Atlantic Coast							
Forsythe NWR, Barnegat Div.	FBARN90-1	4/23/90	49.1	40.6	43.2	35.8	41.3
" "	FBARN90-2	4/23/90	48.0	38.6	37.7	31.1	36.5
" "	FBARN90-3	4/23/90	49.5	39.3	41.2	34.4	39.0
" "	FBARN90-4	4/23/90	48.9	40.4	39.8	34.3	40.7
" "	FBARN91	5/16/91	NA	NA	NA	35*	44*
Forsythe NWR, Brigantine Div	FBRIG	5/29/91	NA	NA	NA	35*	44*
Tuckahoe WMA	TWMA-1	4/29/91	NA	NA	NA	35*	44*
" "	TWMA-2	4/29/91	NA	NA	NA	35*	44*
Marmora WMA	MWMA	6/14/90	57.5	37.3	36.6	30.9	40.8
Sedge Island WMA	SWMA	5/10/91	54.3	42.1	47.2	42.0	49.0
Atlantic City	AC-1	5/10/90	55.6	42.1	41.6	35.6	50.3
" "	AC-2	5/10/90	54.2	42.0	45.1	38.1	48.8
Delaware River							
Girard Point Bridge	GPB	5/17/91	54.5	43.1	NA	38.7	51.6
Walt Whitman Bridge	WWB	4/23/91	50.7	41.9	39.6	34.8	45.4
PA / NJ Turnpike Bridge	PTB-1	6/13/91	50.1	38.7	32.7	26.6	38.3
" "	PTB-2	6/13/91	NA	NA	NA	35*	44*
" "	PTB-3	6/13/91	NA	NA	NA	35*	44*

*Estimated, based on mean of all eggs.

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sheet 5

Method and procedure blanks were processed with the samples to verify that solvents, reagents, apparatuses and instrumentation did not add detectable positive biases to the residue concentrations. The method lower limits of detection (MLLDs) and lower limits of quantitation (MLLQs) for each contaminant were determined according to the method of Keith et al. (1983). In general, the MLLDs and MLLQs were similar to those reported previously (Smith et al. 1990, Feltz et al. 1995).

C. DATA CONVERSIONS

Fresh weights of eggs (weight of egg contents adjusted to account for dehydration and lipid utilization) were calculated using the relationship identified by Stickel et al. (1973). Due to shell breakage and absence of measurement data for six samples (PTB-2, PTB-3, FBARN91, FBRIG, TWMA-1 and TWMA-2), estimates of fresh weight concentration for these samples were determined by using the average of the fresh egg weights of the other 11 eggs. Details of the calculation of fresh weight and fresh weight concentration are provided in Appendix B.

Toxic equivalency factors (TEFs) for PCB numbers 77, 105, 118, 126, 156 and 169 are taken from Hoffman et al. (1996). These TEFs were selected because they are based on avian embryomortality from egg injection studies, and thus represent the most appropriate values for the present study. The TEF selected for PCB 77 was also used for PCB 81 because the two compounds are approximate stereoisomers (L. Williams, Pers. Comm.). The TEFs used for PCDDs / PCDFs are the widely-accepted U.S. Environmental Protection Agency / International values, as cited in Ahlborg et al. (1992). The TEFs for the remaining PCBs (114, 123, 157, 167 and 189) are based on those calculated by Ahlborg et al. (1994). However, it should be pointed out that the latter are based on ingestion scenarios using mammalian systems. There are some data suggesting that TEFs for mammalian systems may not be applicable for birds (Ahlborg et al. 1994). The TEFs selected for use in the present study are presented in Table 2.

After individual PHH concentrations were converted to TCDD equivalent concentrations, the latter were summed to determine total TCDD equivalent concentrations (TEQs). The calculated TEQs were compared to a toxicity reference value for bird eggs to estimate the potential combined toxicological impact of detected PHH concentrations.

Table 2. Toxicity Equivalence Factors (TEFs) for Coplanar PCBs, PCDDs and PCDFs.

Congener	TEF	Ref.	Congener	TEF	Ref.
2,3,7,8-TCDD	1.0	(1)	PCB 77	0.02	(2)
1,2,3,7,8-PeCDD	0.5	(1)	PCB 81	0.02	-
1,2,3,6,7,8-HxCDD	0.1	(1)	PCB 126	0.05	(2)
1,2,3,7,8,9-HxCDD	0.1	(1)	PCB 169	0.001	(2)
1,2,3,4,7,8-HxCDD	0.1	(1)	PCB 105	0.00007	(2)
1,2,3,4,6,7,8-HpCDD	0.01	(1)	PCB 114	0.0005	(3)
OCDD	0.001	(1)	PCB 118	0.00003	(2)
2,3,7,8-TCDF	0.1	(1)	PCB 123	0.0001	(3)
2,3,4,7,8-PeCDF	0.5	(1)	PCB 156	0.0001	(2)
1,2,3,7,8-PeCDF	0.05	(1)	PCB 157	0.0005	(3)
1,2,3,4,7,8-HxCDF	0.1	(1)	PCB 167	0.00001	(3)
2,3,4,6,7,8-HxCDF	0.1	(1)	PCB 189	0.0001	(3)
1,2,3,6,7,8-HxCDF	0.1	(1)			
1,2,3,7,8,9-HxCDF	0.1	(1)			
1,2,3,4,6,7,8-HpCDF	0.01	(1)			
1,2,3,4,7,8,9-HpCDF	0.01	(1)			
OCDF	0.001	(1)			

References: (1) Ahlborg et al. 1992; (2) Hoffman et al. 1996; (3) Ahlborg et al. 1994.

III. RESULTS

A. REPRODUCTIVE SUCCESS AND EGGSHELL THICKNESS

Eggshell thicknesses were determined to be 2.4 to 24.3 percent below pre-DDT levels (Table 3) (Anderson and Hickey 1972). Multiple regression analysis was preformed with eggshell thickness as the dependent variable, and DDE, log DDE, PCB 77, PCB 126, total PCB, TEQ, and mercury as the independent variables. Contrary to expectation, no correlations were identified between contaminant concentrations and eggshell thickness.

The 5-year average productivity (1990-94) for peregrine falcon nests included in this study varied from 0.2 young per year at Walt Whitman Bridge to 3.2 young per year at the Forsythe National Wildlife Refuge (NWR), Brigantine Unit. Atlantic coast nests averaged 2.0 young per nest over the same time period, while nests on Delaware River bridges produced only 0.8 young per year (Table 3). As was the case with eggshell thickness, no relationship between levels of any contaminant and peregrine falcon productivity could be found. On the contrary, nests with the highest productivity (Sedge Island, Brigantine and Marmora) had some of the highest contaminant concentrations.

B. ANALYTICAL RESULTS

1. Metals scan

Average mercury concentrations at eyries on the Atlantic coast were over 30 times the mercury concentrations at Delaware River eyries (Table 4). The highest mercury concentrations were found at the Sedge Island, Brigantine, and Barnegat nests (Figure 2). Concentrations of other metals are presented in Appendix A.

2. Organochlorine scan

Mean total PCB concentrations were only slightly higher in Atlantic coast eyries than in Delaware River eyries (Table 4). Total PCBs were detected at all eyries at concentrations ranging from 7.2 μg/g to 27.9 μg/g (Figure 3; Appendix A). The lowest total PCB concentration was detected at the Girard Point Bridge eyrie, while the highest was found at the Atlantic City and Brigantine nests.

The compound DDE was detected at all eyries. DDE concentrations were highest in eggs from the Tuckahoe WMA (Figure 3). The average DDE concentrations in Atlantic coast and Delaware River eyries were 8.2 ug/g and 4.7 ug/g, respectively (Table 4). Analytical results for 12 other organochlorine pesticides and their metabolites are presented in Appendix A.

Table 3. Peregrine Falcon Productivity and Eggshell Thickness Reductions (percent of pre-DDT) at Selected Nests in New Jersey.

Nest Location	Year sampled	No. eggs measured	Eggshell thickness reduction (%)	Young produced in sample year	5-yr. avg. productivity (1990-94)
Atlantic Coast					
Forsythe NWR, Barnegat Div.	90	4	13.3*	0	0.8
Forsythe NWR, Barnegat Div.	91	1	24.3	0	"
Forsythe NWR, Brigantine Div.	91	1	17.9	3	3.2
Tuckahoe WMA	91	2	14.1*	1	1.0
Marmora WMA	90	1	22.9	1	2.3
Sedge Island WMA	91	1	8.5	3	3.0
Atlantic City	90	2	15.5*	2	2.0
mean			15.5	1.4	2.0
Delaware River					
Girard Point Bridge	91	1	2.4	2	1.2
Walt Whitman Bridge	91	1	15.5	1	0.2
PA / NJ Turnpike Bridge	91	3	12.9*	0	1.0
mean			11.4	1.0	0.8

*Mean is given if more than one egg was measured.

Source of productivity data: NJDFGW, 1995.

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sheet 9

Table 4. Concentrations of Selected Reproductive Toxins in New Jersey Peregrine Falcon Eggs from Sites on the Atlantic Coast and Delaware River (Mean(Range)).

Location	Mercury	p,p'-DDE (ug/g fresh wt.)	Total PCBs	Mono-ortho PCBs	Non-ortho PCBs (TEQs-pg/g fresh wt.)	PCDDs and PCDFs	**Total TEQs**
Atlantic coast (n=12)	0.38 (0.19-0.58)	8.2 (1.6-24.2)	19.2 (11.1-27.9)	141.1 (77-217)	309.8 (201-539)	45.0 (20-75)	496.0 (318-757)
Delaware River (n=5)	0.012 (.004-.036)	4.7 (2.0-8.0)	14.7 (7.2-21.0)	80.4 (46-106)	48.0 (37-63)	12.1 (7.3-14)	140.5 (90-175)

TEQs=2,3,7,8-TCDD equivalents.

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sheet 6

Figure 2. Mercury concentrations in New Jersey peregrine falcon eggs.

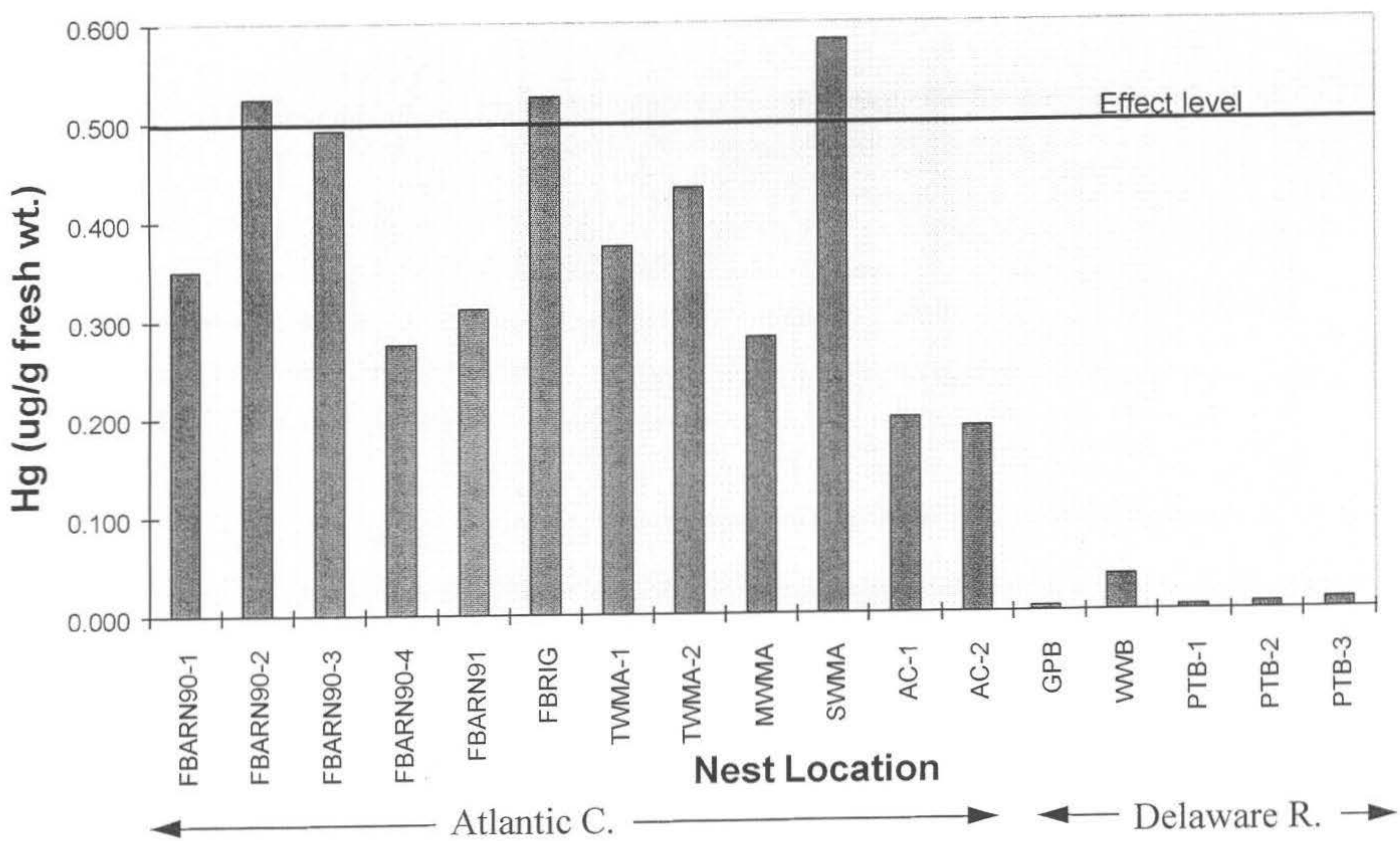
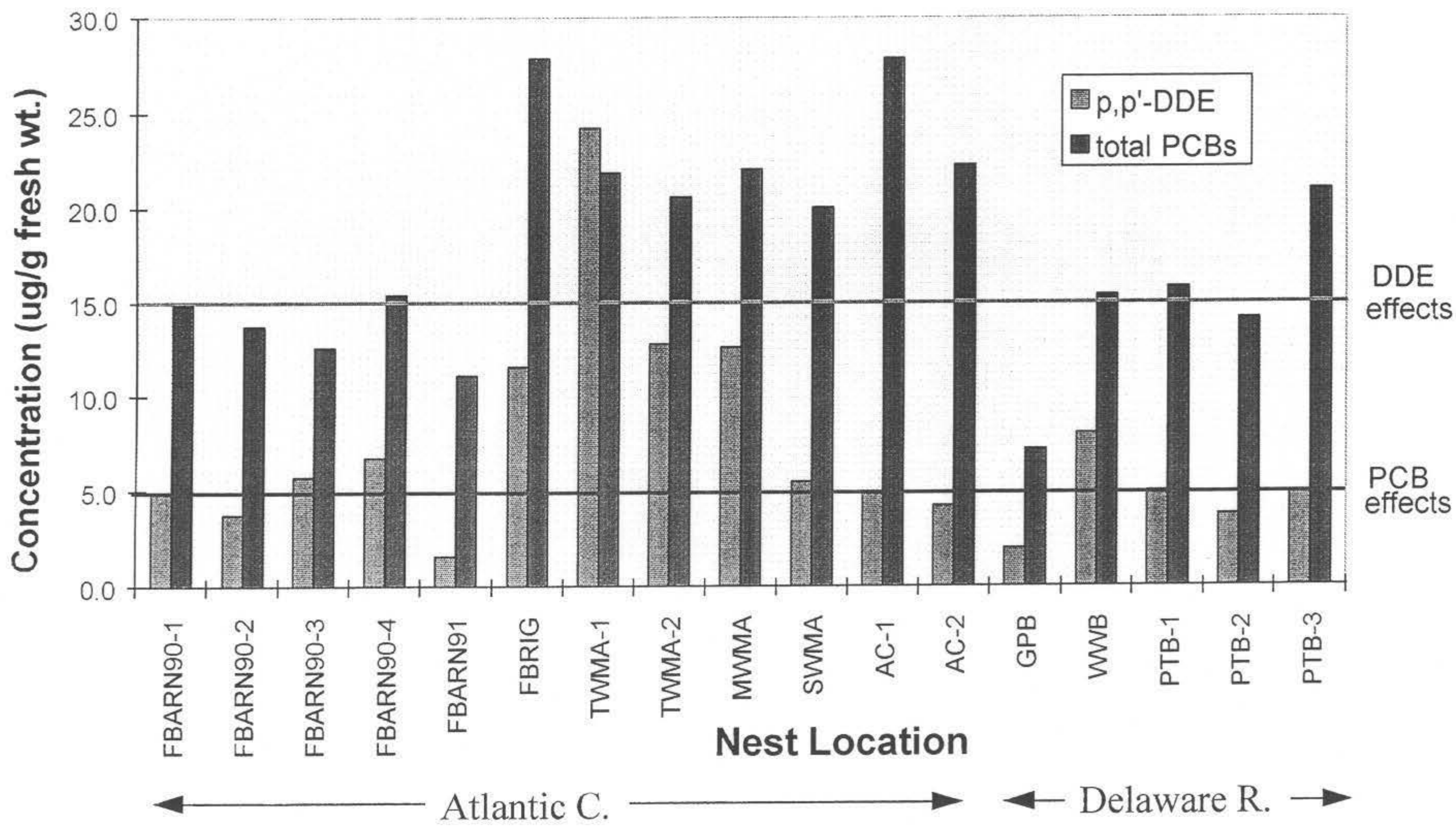


Figure 3. DDE and total PCB concentrations in New Jersey peregrine falcon eggs.



3. Coplanar PCBs

The mono-ortho-PCBs contributed over 99 percent of the total PHH concentration in all eggs. The PCB congener 118 was by far the most abundant in the mono-ortho class (Appendix A). Mean total mono-ortho PCB concentrations in Atlantic coast eyries were approximately four times higher than in Delaware River eyries. Mono-ortho PBCs contributed about one-third to one-half of total TEQs (Figure 4).

Among the dioxin-like, non-ortho PCBs, the potent PCB congeners 3,3',4,4'-tetrachlorobiphenyl (PCB 77) and 3,3',4,4',5-pentachlorobiphenyl (PCB 126) were the most prevalent (Appendix A). All non-ortho PCBs were detected in concentrations which were elevated in Atlantic coast eyries relative to those from the Delaware River. The concentrations of PCB 77 and PCB 126 were 12- and 7-fold higher, respectively, at Atlantic Coast eyries. The highest concentrations of these congeners were found at the Sedge Island and Brigantine nests (Figure 5). Non-ortho PCBs contributed about two-thirds of total TEQs at Atlantic coast nests, and about one-third at Delaware River nests (Figure 4).

4. PCDDs and PCDFs

The compound TCDD was detected in eggs from all eyries except for the Pennsylvania / New Jersey Turnpike Bridge. Detected TCDD concentrations ranged from 0.3 pg/g at the Girard Point Bridge to 24.3 pg/g at the Marmora WMA (Appendix A). Mean TEQs due to PCDDs and PCDFs were approximately four-fold higher in eggs from Atlantic coast nests as compared to the Delaware River nests (Table 4). In general, PCDDs / PCDFs contributed only a small fraction (about 10 percent or less) of total TEQs in peregrine eggs (Figure 4).

Figure 4. Total TCDD-equivalent concentrations (TEQs) in peregrine falcon eggs in New Jersey.

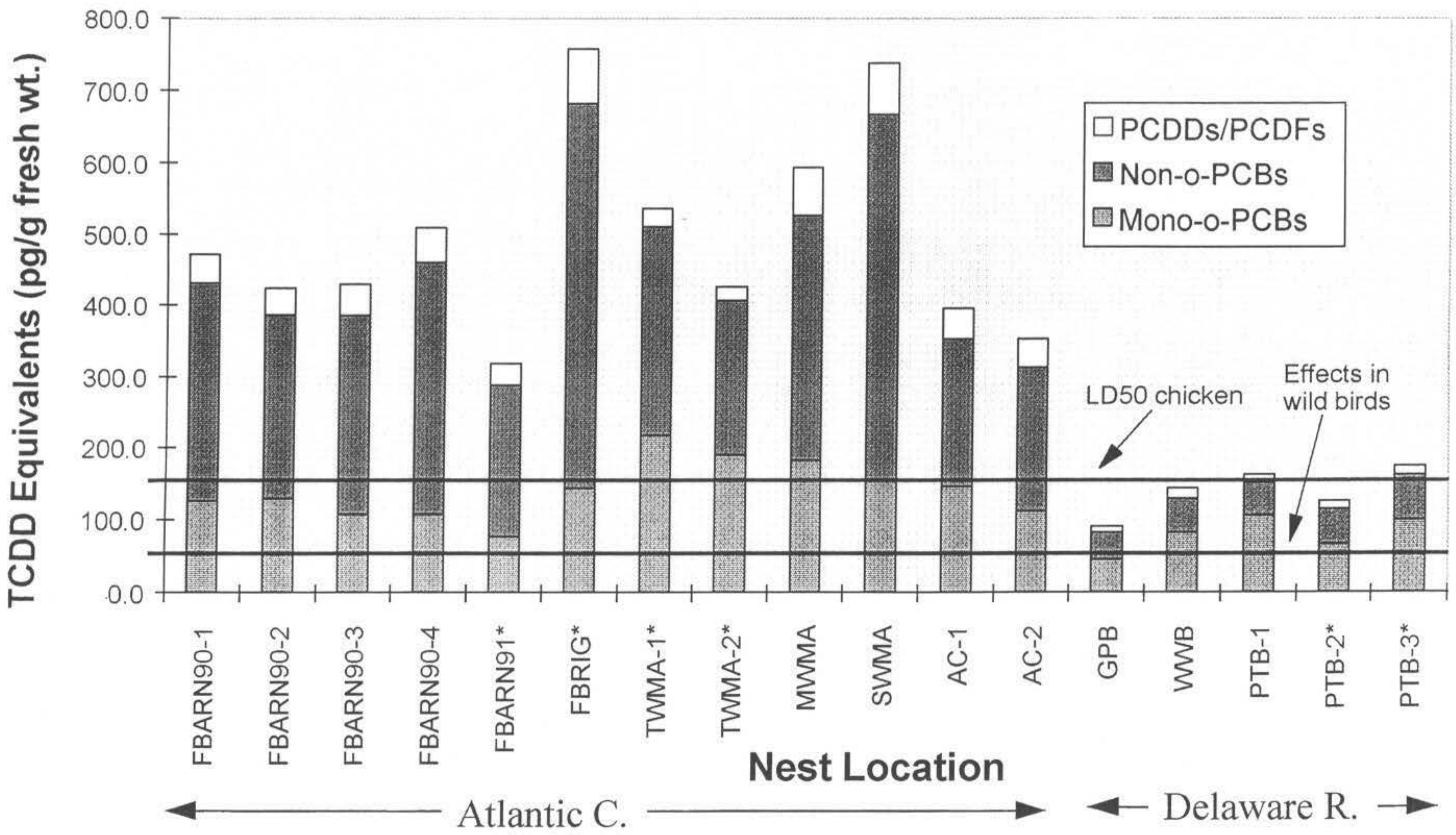
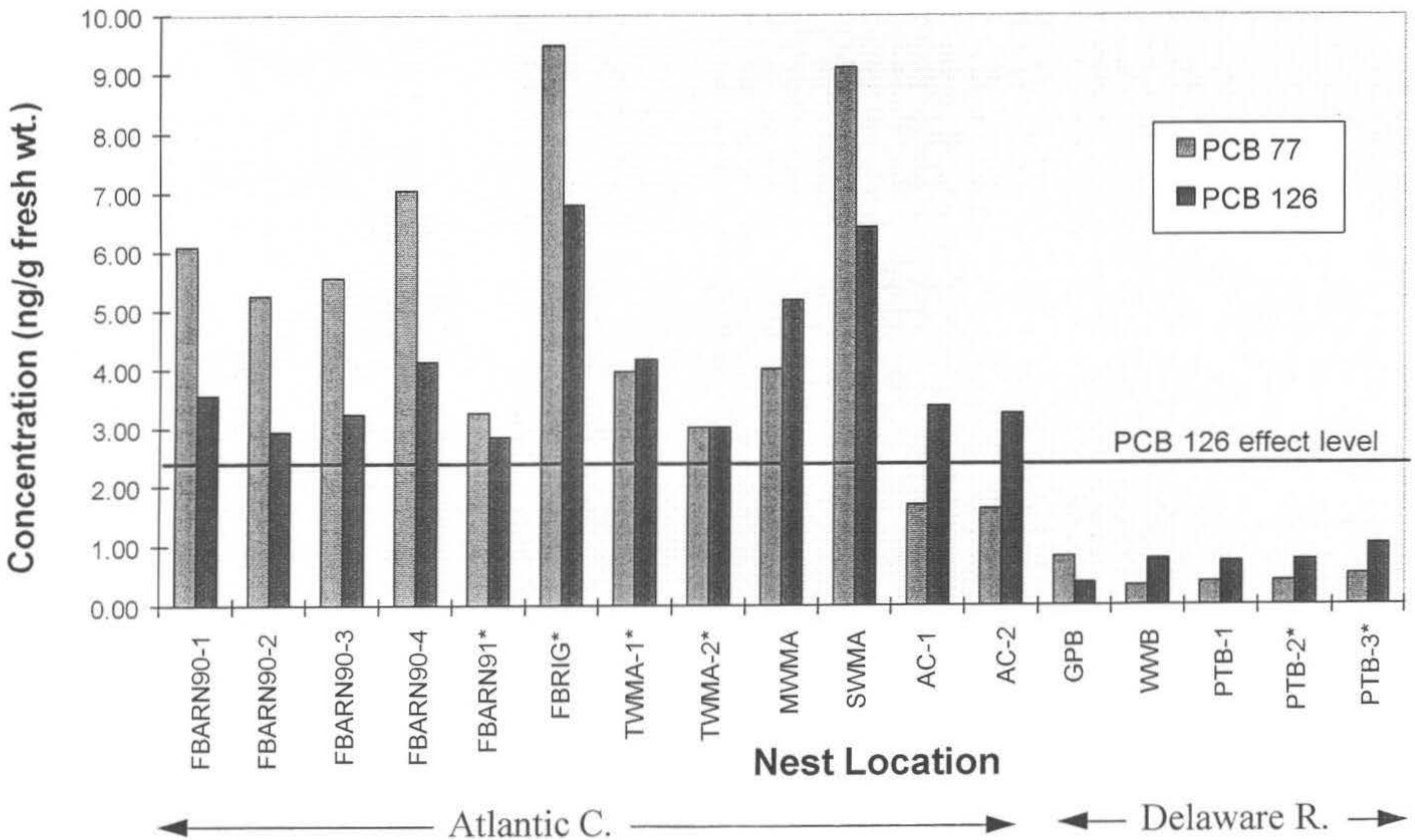


Figure 5. PCB 77 and PCB 126 concentrations in New Jersey peregrine falcon eggs.



15

IV. DISCUSSION

A. PCBS, PCDDS AND PCDFS

Reproductive risks are discussed separately below for total TCDD equivalent concentrations (TEQs), non-ortho PCB congeners, and total PCBs. This approach provides multiple independent measures of potential impact to peregrine falcons using the available data. Independent evaluations were made because: (1) the U.S. Environmental Protection Agency accepts evaluations of total PCBs, but has not yet accepted the TEQ approach for wildlife evaluation (L. Williams, Pers. Comm.); (2) only part of the toxicity of polychlorinated compounds is described by the TEQ approach; (3) toxicity reference values corresponding to non-ortho PCB congeners and total PCBs are available; and, (4) the evaluation of total PCBs provides data which may be compared with historical data and may be useful to determine the value of future total PCB monitoring.

1. TCDD Equivalents

a. Limitations of the TEQ approach

It is important to review the uncertainties associated with the TEQ approach before presenting the analysis. First, the TEQ approach is largely based on the assumption that AHH induction is the only important mechanism of toxicity of PHH compounds. However, PCDD, PCDF, and PCB mixtures may also elicit non-Ah receptor-mediated responses, such as long-term behavioral dysfunctions and neurotoxicity, which are not described by analysis of the dioxin-like activity of the compounds (Ahlborg et al. 1992). Limited information characterizing toxicological endpoints such as behavioral aberrations or neurological toxicity contributes to this uncertainty. For example, the quantitative toxicity data documenting behavioral effects on birds is limited to only a few studies.

Second, the TEQ approach used herein is based on TEFs derived from a variety of different species, including birds, mammals and *in vitro* studies. Given the well characterized differences in dioxin susceptibility between species, TEFs, based on a variety of different species and then applied to yet another species, may not be valid.

Third, the TEQ approach assumes that AHH active congeners are strictly additive. It does not take into account synergistic or antagonistic effects of mixtures of PCBs, PCDDs, and PCDFs. Interactions among Ah-active congeners or among Ah-active and non-Ah-active congeners and other toxic synthetic halogenated compounds are not accounted for. Additive, synergistic, and / or antagonistic interactions have been demonstrated to occur (Banister et al. 1987). The U.S. Environmental Protection Agency

(1991) and Davis and Safe (1989) demonstrated that comparisons of TEQs in different mixtures using the TEF approach significantly over-estimated observed induction and immunotoxicity (Ahlborg et al. 1992). Due to their different chemical structures, the individual PHHs exhibit a wide range of chemical properties and toxicological potencies which cannot be adequately described in a single TEF value. Safe (1990) cited this limitation to the TEQ approach, stating that the observed potency of a specific PHH or PHH mixture is dependent on the PHH tested and its concentration relative to other PHHs.

In addition to inherent uncertainties with the TEQ approach, implementation of the approach by different investigators has led to a body of literature which is not readily comparable. There has been little consensus regarding the selection of specific TEF values for individual PHHs (Bosveld and Van Den Berg 1994). The TEFs for PCBs have proved to be particularly problematic. There is no international agreement on the TEFs for coplanar (non-ortho and mono-ortho) PCBs. Proposed TEF values range considerably between authors. For example, TEFs for PCB 126, which is commonly accepted as the most potent PCB congener, range from 0.01 to 0.05 (Safe 1990, Hoffman et al. 1996). Selection of different TEFs by different researchers may create wide disparity between estimates of TEQ toxicity.

b. <u>TCDD toxicity in bird eggs</u>

The following discussion provides a profile of TCDD toxicity and selects the three toxicity reference values which will be compared to calculated TEQs. The compound TCDD is a potent reproductive and developmental toxicant that bioaccumulates in wildlife. This compound is translocated to eggs where it has the potential to cause embryotoxicity (Nosek et al. 1992, 1993). Many avian species have been shown to be sensitive to embryonic mortality due to TCDD exposure. In laboratory studies, chickens (*Gallus domesticus*) have been shown to be the most sensitive species. Verrett (1970, as cited by Hoffman et al. 1996) demonstrated embryonic mortality, edema, and teratogenic effects in chickens at 10 to 20 pg/g TCDD in eggs. Cheung et al. (1981) reported a dose-dependent relationship between TCDD concentrations and increased cardiovascular malformations. Ventricular septal defects, aortic arch anomalies, and conotruncal malformations in chick embryos were noted. A dose of 6 pg/g of TCDD in eggs caused a 20 percent increase in malformations; at 65 pg/g malformations doubled (Cheung et al. 1981 as cited in Kubiak et al. 1989). Progressive impairment of avian ventricular development has recently been identified to be the result of TCDD interference with intracellular calcium processing (Rifkind 1993). In chicken embryos, this interference impairs the ability of the sarcoplasmic reticulum to sequester calcium causing an evolving sequence of contractile defects (Rifkind 1993).

Neurotoxicity is another disturbing and difficult to quantify result of TCDD exposure. The compound TCDD and other PHHs have been found in various avian studies to affect brain chemistry and behavior. As cited in Bosveld and Van den Berg (1994), Henshel et al. (1993) observed an increase in interhemispheric asymmetry in great blue heron (*Ardea herodias*) hatchlings from a highly contaminated colony, as compared to a reference colony. The threshold level for this effect on brain symmetry was 50 pg/g to 60 pg/g in egg. In these birds, the area of the brain which is involved in olfactory discrimination and the limbic system (which controls emotion and motivational behavior) had morphological abnormalities (increased cell density and overall medial-to-lateral width in the pyriform cortex). Fox et al. (1978) and Peakall et al. (1980) demonstrated that exposure to PHH contaminants may play a role in parental inattentiveness and reduced reproductive success in herring gulls. Total PHH content of eggs was found to be related to the time that eggs were unattended by the parent.

Developing embryos and young are more sensitive than adults to TCDD exposure (Nosek et al. 1993). Translocation of TCDD to eggs provides an important route of elimination for adult birds (in ring-necked pheasants, 1 percent of cumulative dose is passed to each of the first 15 eggs laid) at the expense of the developing embryo. In chickens, Nosek et al. (1992) reported the LD_{50} in the embryo to be 0.25 μg/kg in egg; 100 times less than the dose producing mortality in adults (25 μg/kg body weight). Ring-necked pheasant (*Phasianus colchicus*) embryos and adults evaluated by Nosek et al. (1992, 1993) experienced 50 and 75 percent mortality at 1.3 μg/kg in egg and 25 μg/kg body weight, respectively.

Dioxin and dioxin-like compounds express toxicity in a species-specific manner. Nosek et al. (1993) illustrated differential toxicity between species for both lethal and sublethal effects. In fertilized chicken eggs, TCDD and related compounds cause hydropericardium, subcutaneous edema, liver lesions, inhibition of lymphoid development, microphthalmia, beak deformities, and cardiovascular malformations (Mckinney et al. 1976). Turkeys (*Meleagris gallopavo*) show similar deformities following exposure, but ringed-necked pheasant, mallards (*Anas platyrhynchos*), goldeneyes (*Bucephala clangula*), and blackheaded gulls (*Larus ridibundus*) are not sensitive to these expressions of toxicity (Brunstrom 1988, Brunstrom and Reutergardh 1986).

Embryonic mortality in eastern bluebird (*Sialia sialis*) has been demonstrated at LD_{50}s ranging between 1 μg/kg and 10 μg/kg TCDD in egg (Thiel et al. 1988, as cited in Nosek et al. 1993). In ring-necked pheasants, Nosek et al. (1993) concluded that a cumulative dose of TCDD at 10 μg/kg body weight (1.0 μg/kg per week for 10 weeks) caused reduced egg production and hatchability. Elliott et al. (1989) observed total reproductive failure in a colony of great blue herons at TEQs ranging from 65 to 496 (mean 230) pg/g in eggs, although the authors felt that predation had also played a role. Using international TEFs, White and Seginak (1994) evaluated the relationship between TEQs and reproductive success in wood ducks (*Aix sponsa*). The threshold for reproductive impairment in wood ducks eggs was 20 pg/g to 50 pg/g. Similarly, TEQ concentrations of 90 to 339 pg/g in eggs have been correlated to embryotoxicity, congenital deformities, and poor hatching success in Forster's Tern (*Sterna forsteri*) (Tillitt et al. 1993). Using the rat hepatoma cell bioassay (H4IIE) to quantify TCDD equivalents in double-crested cormorant (*Phalacrocorax auritus*) eggs collected from the field, Tillitt et al. (1992) found increased egg mortality beginning at TEQ concentrations of approximately 85 pg/g.

Three toxicity threshold concentrations were used to evaluate the toxicity of detected PHHs in examined eggs. Threshold egg concentrations (toxicity reference values) were selected to evaluate the potential for: (1) lowest observed adverse effect level (LOAEL) in sensitive avian species; (2) documented effects in populations of wild birds; and, (3) 50 percent embryonic mortality. A value of 10 pg/g was selected as the LOAEL for TCDD. This value has been demonstrated to cause embryonic mortality, edema, and teratogenic effects in the most sensitive avian species (i.e., chicken) (Verrett 1970). A value of 10 pg/g was also used in the Great Lakes Initiative to evaluate the sublethal effects of TCDD (USEPA 1995). TEQ levels exceeding the 10 pg/g toxicity reference value would be predictive of the above toxicological effects in sensitive species. The TEQ concentration of 50 pg/g was selected as the toxicity reference value for documented effects in wild birds. Eggs containing PHH compounds in concentrations sufficient to produce TEQ levels exceeding the 50 pg/g toxicity reference value have been associated with effects on reproduction or development in great blue herons, wood ducks, Forster's terns, and double-crested cormorants. An LD_{50} value of 147 pg/g in chicken eggs (Verrett 1976), which is the foundation of the calculated TEFs for the non-ortho congeners (Hoffman et al. 1996), was selected as the third reference toxicity value, representing 50 percent embryonic mortality.

c. <u>Comparison of effect levels with observed levels</u>

Calculated total TEQ concentrations for all eggs (Figure 4; Appendix A) and mean TEQs for both the Delaware River and Atlantic coast (Table 4) exceed levels which would be expected to have adverse effects on peregrine falcons. Total PHH TEQs were highest at Brigantine (757 pg/g) followed by Sedge Island (737 pg/g) and Marmora (594 pg/g). The average TEQ concentration of eggs from Atlantic coast eyries (496 pg/g) was 3.5 times greater than that of eyries on the Delaware River (140 pg/g). Peregrine falcons exposed to PHHs at these levels would be expected to be at risk of PHH-induced toxicity at all eyries; the LOAEL for TCDD toxicity (10 pg/g) was exceeded by a factor ranging from 9- to 75-fold. The observed levels in eggs were 2- to 15-fold higher than levels which have been documented to impact wild birds (50 pg/g). Notably, the TEQ LD_{50} reference value (147 pg/g) was exceeded at all Atlantic coast eyries by a factor of 2- to 5-fold. Unless peregrines are much less sensitive than other species, these results suggest that New Jersey peregrines should be experiencing reproductive impacts due to dioxin and dioxin-like compounds (especially nests on the Atlantic coast).

2. Non-ortho PCBs

a. Toxicity in bird eggs

The non-ortho, coplanar PCBs are considered to be the most toxic of the PCBs. Non-ortho PCBs have been shown to cause embryo mortality in wildlife species such as: ringed-necked pheasant, mallard, goldeneye, and blackheaded gull (Brunstrom 1988; Brunstrom and Reutergardh 1986). A clear picture of congener-specific toxicity has not yet emerged. The potency of individual congeners is highly dependent on both the test species and the measured response (Hoffman et al. 1996). However, congeners 77 and 126 appear to be the most potent of the non-ortho congener class. Calculations using the TEQ approach (Appendix A) indicated that collectively, PCBs 77 and 126 were likely to contribute about 30 to 60 percent of total PHH toxicity. Fortunately, raptor-specific toxicity reference values for these potent congeners are available. Use of raptor-specific toxicity data for PCBs 77 and 126 circumvents the necessity of extrapolating toxicity data between avian species that are not closely-related (which is a primary source of uncertainty in the TEQ approach).

Numerous studies have documented great variability between species in response to PCB exposure. Depending on the species examined, the toxicity associated with non-ortho congeners has been shown to range over several orders of magnitude. In six breeds of chicken exposed to PCB 77, Brunstrom (1988) demonstrated 70 to 100 percent embryonic mortality at doses of 20 μg/kg in each egg. Ducks and herring gulls appear to be much more tolerant to PCB 77. At doses of 1000 μg/kg or more in eggs of PCB 77, ducks and herring gulls experienced no ill effects (Brunstrom 1988). Exposure to PCB 77 caused microphthalmia and beak deformities in turkeys at 800 μg/kg egg (Brunstrom and Lund 1988). Other effects of PCB 77, including liver lesions, edema, and thymic hypoplasia noted in chickens, have not been evident in turkeys (Brunstrom 1989). Thus, toxicity reference doses based on other species, as was done in the TEQ approach, may not adequately represent peregrine falcons.

Hoffman et al. (1998) provide the only laboratory data on non-ortho PCBs for raptors. American kestrel (*Falco sparverius*) eggs injected with PCB 77 at a dose of 100 $\mu g/kg$ experienced a 40 percent reduction in hatching success. As compared to controls, egg injections of congener 126 at concentrations of 2.3 $\mu g/kg$ and 23 $\mu g/kg$ resulted in 10 and 24 percent reductions in hatching success, respectively (Hoffman et al. 1998). Although the reductions in hatching success were not statistically significant, there were significant differences in hatchling weight, liver weight, and number of chicks with edema or malformations at both doses. The LD_{50} of PCB 126 in kestrel eggs was 65 $\mu g/kg$. The availability of toxicity data for raptors provides the opportunity to evaluate dioxin-like effects on peregrine falcons using a closely-related species, providing the highest level of certainty presently available.

b. Comparison of effect levels with observed levels

Based on the above information, the reference toxicity values chosen for PCB 77 and PCB 126 were 100 $\mu g/kg$ and 2.3 $\mu g/kg$, representing increases in embryonic mortality of approximately 40 and 10 percent, respectively. None of the peregrine falcon eggs analyzed in this study exceeded the reference value for PCB 77. For PCB 126, all eggs from the Atlantic coast nests exceeded the reference value (Figure 5 and Appendix A). These findings indicate an unacceptable risk of embryonic mortality in peregrine falcons living on the Atlantic coast due exclusively to congener 126. The confidence in these findings may be considered high as they are based on congener-specific analysis of a closely-related species and they are not encumbered by subjective interspecies extrapolation or the use of uncertainty or modifying factors. Modification of existing plans for monitoring contaminant concentrations to specify levels of these important congeners is indicated by these findings.

3. Total PCBs

a. Toxicity in bird eggs

Total polychlorinated biphenyls at levels greater than 5 $\mu g/g$ have been shown to reduce egg hatchability and cause embryotoxicity such as edema, growth retardation, and deformities in laboratory birds (Platonow and Reinhart 1973). Tumasonis et al. (1973) established a lowest observed effect level (LOEL) for reduced hatchability at 4 $\mu g/g$ in chicken eggs. Kubiak et al. (1989) found that hatching success was 50 percent below normal in a Forster's tern colony where the eggs had a total PCB concentration ranging from 6 to 26 $\mu g/g$. The reproductive impairment in the contaminated colony was attributed to both intrinsic chemical factors in the egg as well as extrinsic factors such as parental inattentiveness. Similarly, Peakall and Peakall (1973) demonstrated that diets containing 10 μg/g Aroclor 1254 caused mean egg concentrations of 16 μg/g wet weight in ringed turtledoves (*Streptopelia risoria*). These egg concentrations were associated with decreased parental attentiveness and increased embryo mortality. Kubiak and Best (1991) and Helander et al. (1982) have independently determined that a level of 5 to 10 μg/g or less in eggs was necessary for "healthy" reproduction in eagles (as cited in Hoffman et al. 1996). In contrast, Peakall et al. (1990) reported that, in the American kestrel, a dose of Aroclor 1254 sufficient to produce 40 μg/g in eggs had no adverse effects on reproduction. Based on the effect levels described above for both raptors and laboratory birds, a reference toxicity value of 5 μg/g was selected.

b. <u>Comparison of effect levels with observed levels</u>

All eggs contained total PCB concentrations above the reference toxicity value. The highest concentrations were found at the Atlantic City and Brigantine eyries (both 27.9 μg/g) (Figure 3). Means for the Atlantic coast (19.2 μg/g) and Delaware River (14.7 μg/g) were both well above the reference level (Table 4). These data suggest that total PCB contamination would be expected to impair reproductive success of peregrine falcons throughout the State. However, there are indications that members of the genus *Falco* may be less sensitive to total PCBs than other avian species. Peakall et al. (1990) suggested that a level greater than 40 μg/g would be necessary to reach a critical level in peregrine falcons, based on experiments with American kestrels. The results of this study support the relative insensitivity of peregrine falcons to total PCBs.

B. DDE

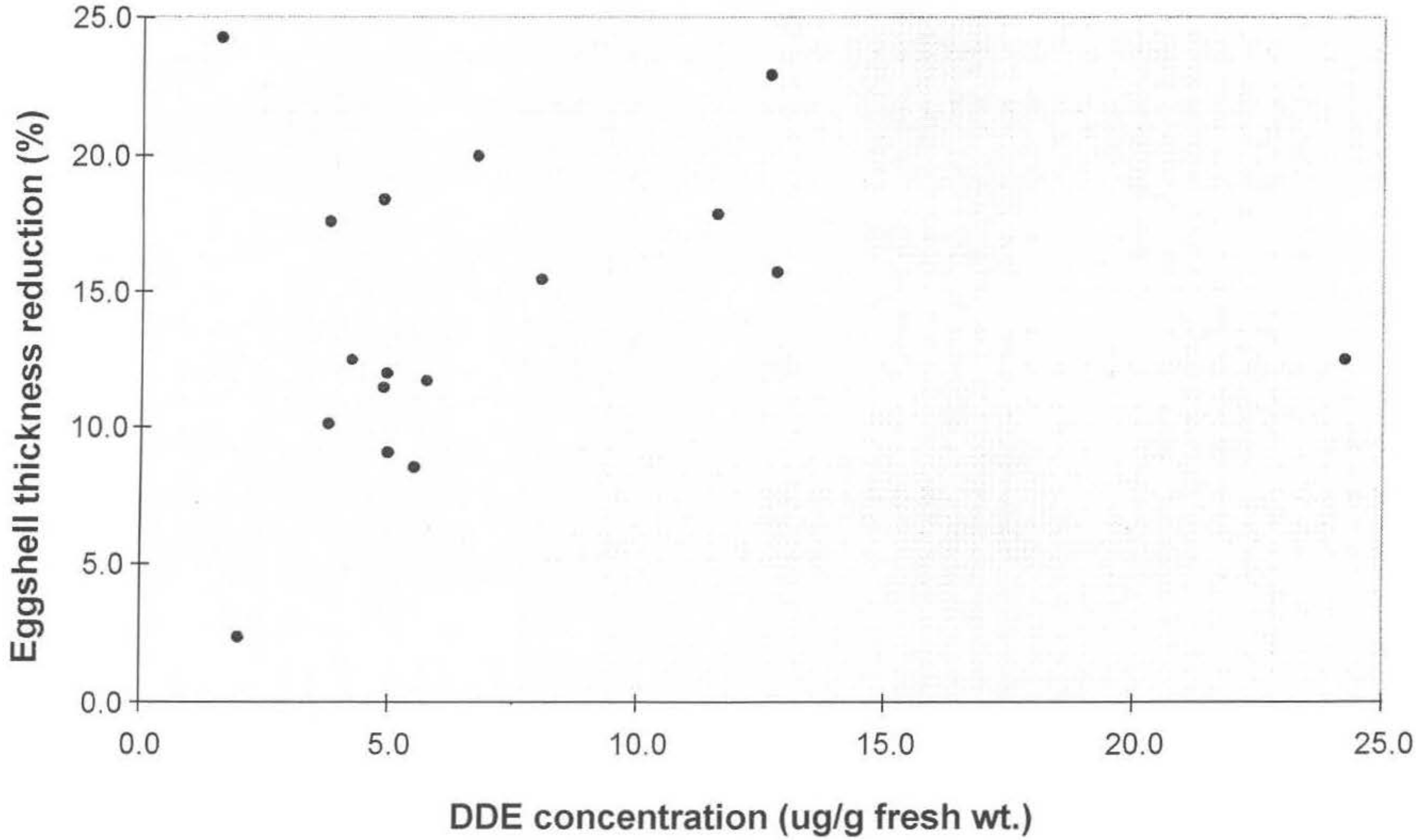
With the exception of DDE, organochlorine pesticide concentrations (Appendix A) appear to be of minimal toxicological significance when compared with available toxicological data and findings reported in other peregrine falcon studies (Ellis et al. 1989, Jarman et al. 1993a, Jarman et al. 1993b, Henny et al. 1994). The compound DDE, a metabolite of DDT, causes eggshell thinning, shell breakage, and depressed reproduction (Cooke 1975, Risebrough and Anderson 1975). The literature associating past declines of peregrine falcon populations with exposure to DDT / DDE is extensive and conclusive. Failure of peregrine eggs has been reported to be associated with egg DDE concentrations of 15 to 20 μg/g wet weight (Peakall and Kiff 1988, Peakall et al. 1990) and greater than 30 μg/g wet weight (Ambrose et al. 1988, Blus 1996). Mean reductions in eggshell thickness of 17 to 18 percent of pre-DDT levels are thought to be the threshold for impaired reproductive success for peregrine falcons (Blus 1996, Peakall and Kiff 1988). A peregrine population with an average eggshell thickness reduction above 17 to 18 percent would be expected to display reduced productivity and would not be expected to maintain itself (Peakall and Kiff 1988). A reference toxicity value of 15 μg/g was selected as an effect level for DDE in peregrine falcon eggs.

The present data provide only limited evidence suggesting a relationship between DDE residues and peregrine falcon productivity in New Jersey. Eggshell thickness in Atlantic coast samples averaged 15.5 percent below pre-DDT levels (Table 3), which is below the threshold for reduced productivity cited above. This finding is in concurrence with eggshell thickness reductions of 16.9 percent reported by Steidl et al. (1991) in eggs from the southern Atlantic Coast of New Jersey. Reductions in eggshell thickness in Delaware River samples averaged 11.4 percent below pre-DDT levels, which is below the 15.9 percent reported by Steidl et al. (1991) on the Delaware Bay and Delaware River, and is also below the threshold for reproductive effects. Thickness reductions in individual eggs ranged from 2.4 to 24.3 percent below pre-DDT levels. The corresponding DDE concentrations (2.0 to 24.2 $\mu g/g$) were mostly below the range of DDE concentrations associated in the scientific literature with Peregrine falcon egg failure. Regression analysis of these data revealed no relationship between eggshell thickness and DDE concentration (Figure 6). This apparent lack of influence of DDE on nesting success generally correlates with a study of Alaskan peregrines, showing similar productivity (1.8 and 1.7 young/site) between sites with eggs having less than 15 ppm and greater than 15 ppm DDE, respectively (Ambrose et al. 1988).

Cracked eggs in the present study (53 percent; 9 of 17) provide circumstantial evidence of a possible impairment of eggshell formation. Notably, however, six of the nine cracked eggs had thickness reductions of less than fourteen percent (mean = 10.7 percent). It is unknown to what extent other non-contaminant related impacts may have contributed to the high incidence of cracked eggs in the study. It is recognized that addled eggs collected in this study were likely subjected to adverse environmental conditions and long-term parental manipulation which may have resulted in a high incidence of egg cracking.

Using 15 $\mu g/g$ as the reference toxicity value, an increased risk of embryonic mortality due to DDE would be predicted for only one eyrie. Eggshell thickness and productivity data support this conclusion. The highest DDE levels were found at the Tuckahoe, Marmora, and Brigantine eyries. Reproduction at two of these nests (Marmora and Brigantine) has been excellent over the past 5 years; eggs from both of these nests contained less than 15 ppm DDE. In contrast, the Tuckahoe nest, which had over 24 ppm DDE in one egg, has been among the poorest producing nests on the Atlantic Coast of New Jersey (Table 3 and Figure 3). An earlier study of New Jersey peregrine eggs indicated that Marmora, Tuckahoe and Barnegat eggs had DDE levels ranging between 12 ppm and 18 ppm in 1984 (Gilroy and Barclay 1988), when the nesting females were probably different individuals than in 1990-91. The detection of similar levels at Marmora and Tuckahoe in this study suggests continuing availability of organochlorine contaminants at coastal eyries. In summary, while detected DDE concentrations and eggshell thinning data do not correspond to expectations, it appears that the level of DDE contamination in eggs from at least one eyrie (Tuckahoe) may be sufficient to explain the consistently poor productivity there.

Figure 6. DDE concentration and eggshell thickness reduction in New Jersey peregrine falcon eggs.



C. MERCURY

Metal concentrations, with the exception of mercury, were below known effect levels and were generally comparable to concentrations judged to have limited bearing on raptor success (Negro et al. 1993, Hernandez et al. 1988). For example, Negro et al. (1993) found no detrimental effects in lesser kestrels (*Falco naumanni*) having egg concentrations of cadmium, copper, lead, and zinc that were similar in magnitude to those in the present study.

While increases in the number of soft-shelled eggs in chicken and pheasants have been reported to be the result of oral exposure to methyl mercury, elevated mercury concentrations do not appear to be related to reduced eggshell thickness in American kestrels or prairie falcons (*Falco mexicanus*) (Peakall and Lincer 1972, Fimreite et al. 1970). Fimreite et al. (1970) found no correlation between eggshell thickness and mercury levels in 59 prairie falcon eggs with mercury concentrations of 0.019 μg/g to 1.71 μg/g. Heinz (1979) found that mallards fed a diet containing 0.5 μg/g of mercury produced eggs with mercury concentrations of 0.79 μg/g to 0.86 μg/g. These mallards displayed lower rates of productivity. Fimreite (1971) reported that ring-necked pheasant eggs containing 0.5 μg/g to 1.5 μg/g mercury had significantly reduced hatchability. High embryo mortality and decreased egg laying were observed in ring-necked pheasants fed dietary mercury to levels which corresponded with 0.9 μg/g to 3.1 μg/g in eggs (Span et al. 1972). In general, 0.5 μg/g of mercury in eggs is considered the level which may result in adverse effects on reproduction in raptors (Fimreite 1979, Wiemeyer et al. 1984).

Mean mercury concentrations in eggs for the Delaware River and Atlantic Coast were 0.012 μg/g and 0.38 μg/g, respectively. These findings suggest that the mercury content of some Atlantic coast eggs approached the avian toxicity threshold for reproductive effects. The highest mercury concentrations were detected in eggs from the Sedge Island, Brigantine, and Barnegat nests, all of which slightly exceeded 0.5 μg/g. While definitive conclusions cannot be made, detected mercury concentrations may have contributed to hatching failure in some eggs. However, the nest with the highest mercury concentration (Sedge Island) has also had the best long-term productivity.

Several studies of New Jersey peregrines and their prey have indicated a pathway of exposure which may lead to elevated mercury concentrations in coastal peregrine eggs. Willets (*Catoptrophorus semipalmatus*), a peregrine prey species that feeds on aquatic insects, marine worms, small mollusks, and fish fry, were found to have mercury concentrations ranging from 0.524 μg/g to 0.630 μg/g dry weight at three New Jersey eyries (U.S. Fish and Wildlife Service 1991b). These concentrations are above dietary concentrations associated with impaired reproductive success, abnormal egg laying behavior, and hyper-responsiveness to fright stimulus in ducks (Heinz 1979, 1987). The level of mercury (3.55 μg/g whole-body wet weight) in a 6-year-old adult peregrine found dead near the Barnegat eyrie, while below the level associated with toxicity, provides further evidence of exposure risk in coastal peregrines (U.S. Fish and Wildlife Service 1991a).

D. POTENTIAL ADDITIVE RISKS / RISK SUMMARY

The above discussion evaluates the potential risks of individual chemicals and chemical classes to peregrine falcons. Risks have been shown to exist in varying magnitudes for each contaminant and / or contaminant class. No attempt was made to evaluate the potential antagonistic, additive and / or synergistic interactions between the suite of contaminants. While the toxicology of chemical mixtures is poorly understood, it is safe to assume that the total risk from all contaminants present in a single egg is greater than what would be expected from any single contaminant. This is especially true since eggs having high levels of one contaminant tended to have high levels of the others as well.

Based on levels of all contaminants, nests at Brigantine and Sedge Island appear to be at greatest risk of reproductive impacts due to contaminants.

V. CONCLUSIONS / RECOMMENDATIONS

This study was designed to test the hypothesis that TCDD equivalent concentrations (TEQs) of planar PCB congeners, PCDDs, and PCDFs are higher in eggs from eyries exhibiting low productivity. The results do not confirm a correlation between productivity and levels of PHH contaminants or TCDD equivalent concentrations. While actual concentrations and TEQs of planar PCBs, PCDDs, and PCDFs were higher at several eyries (e.g., Brigantine, Sedge Island and Marmora), these same eyries displayed the healthiest long-term reproduction. A large number of confounding factors, such as differences in nesting structure (i.e., towers versus bridges), breeding pair age and experience level, other contaminant exposures (i.e., mercury, DDE), predation, and disturbance, make interpretation difficult. Considerable differences exist between the contaminant levels at Atlantic coast eyries relative to those of the Delaware River. However, the two locations could not be compared with respect to the available reproductive data, due to the extreme differences in nesting conditions (i.e., the extreme limits on reproduction caused by structural conditions at the Delaware River bridge eyries). Additionally, differences in productivity among Atlantic coast eyries do not appear to be explained by differential contaminant burdens in the eggs. The relative hatch rates at Atlantic coast eyries do not correspond to contaminant concentrations in any way.

Based on reference toxicity values determined in other bird species, the measured concentrations of PCBs, PCDDs, and PCDFs in all Atlantic coast eggs were theoretically sufficient to impact the Peregrine falcon population. The TEQ analysis specifically identified non-ortho PCB contamination as the most likely contributor to potential reproductive impacts on the Atlantic coast. Non-ortho PCB congeners contributed 50 to 70 percent of the TEQ concentrations at Atlantic coast eyries. These data contrast with the findings on the Delaware River eyries, where non-ortho PCB congeners contributed only 28 to 40 percent of the TEQ concentrations.

Comparison of total PCB concentrations to congener-specific (TEQ and non-ortho PCBs) evaluations suggests the inadequacy of total PCB analysis for toxicological evaluations. The total PCB data do not distinguish the relative toxicity potential of PHH compounds between eyries. The greater than three-fold difference in mean TEQ levels between Atlantic coast and Delaware River eyries can be attributed primarily to differences in the levels of two non-ortho-PCBs (PCB 77 and PCB 126). Mean concentrations of PCB 77 and PCB 126 were 12 and 7 times greater at Atlantic coast eyries than at Delaware River eyries. In contrast, total PCB levels were about the same at the two locations. It is apparent that differences in reproductive risks at Atlantic coast and Delaware River eyries would not have been revealed by total PCB analysis alone.

Both mercury and DDE were detected at concentrations in some eggs that have the potential to adversely affect peregrine falcon reproductive success. Mercury concentrations were 30 times higher at the Atlantic coast eyries than Delaware River eyries. The DDE concentrations were highest at the Tuckahoe eyrie, which has had one of the poorest reproductive performances of all Atlantic coast nests sampled.

The following recommendations are provided relative to contaminant findings:

1. Because of the high levels of contamination found in this study, further monitoring of non-viable eggs is warranted. It is recommended that all addled peregrine eggs collected and archived since this study (i.e., nesting seasons of 1992-97) be analyzed for congener specific PHHs, mercury, and DDE contamination. Bioassays (H4IIE) should be added to the analytical methods to quantify site-specific TCDD equivalence.

2. The present findings support analysis for non-ortho PCBs in lieu of analysis based only on total PCBs. Specific analysis for these congeners will likely yield the most useful toxicological data and be most cost effective.

3. Design future studies to identify pathways of peregrine falcon exposure to contaminants. These would include:

   (1) Quantitatively determine the species composition of peregrine diets on the Atlantic coast and Delaware River, by identifying prey remains that have been archived during nest monitoring or other methods; and,

   (2) Evaluate dietary exposure of peregrines to bioaccumulating compounds by measuring contaminant concentrations in samples of prey species utilized during the nesting season.

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B. PERSONAL COMMUNICATIONS

Williams, L. 1995. Fish and Wildlife Biologist, U.S. Fish and Wildlife Service, East Lansing, Michigan.

APPENDIX A

Analytical Chemistry Results

Table A1. Metal Concentrations in Peregrine Falcon Eggs in New Jersey (ug/g fresh weight).

Sample ID	% Moisture	As DL=.029	Cd DL=.0019	Cu DL=.04	Hg DL=.005	Pb DL=.025	Se DL=.009	Zn DL=.13
Atlantic Coast								
FBARN90-1	83.7	nd	nd	0.38	0.350	nd	0.346	6.8
FBARN90-2	83.5	nd	nd	0.37	0.524	nd	0.387	5.8
FBARN90-3	82.2	nd	nd	0.41	0.492	0.023	0.394	7.8
FBARN90-4	84.1	nd	nd	0.30	0.275	nd	0.345	5.6
FBARN91	81.1	nd	nd	0.72	0.312	0.060	0.325	5.9
FBRIG	81.5	nd	0.0030	0.53	0.526	nd	0.318	7.9
TWMA-1	80.4	nd	nd	1.80	0.375	nd	0.403	5.6
TWMA-2	81.8	nd	nd	0.50	0.434	nd	0.344	5.5
MWMA	86.0	nd	nd	0.89	0.281	nd	0.310	6.9
SWMA	82.2	nd	nd	0.46	0.583	nd	0.350	6.2
AC-1	83.2	nd	nd	1.95	0.198	nd	0.311	7.7
AC-2	80.3	nd	0.0052	0.40	0.189	nd	0.319	8.0
mean	82.5	-	-	0.73	0.378	-	0.346	6.6
Delaware River								
GPB	80.3	nd	nd	0.38	0.005	0.023	0.279	6.0
WWB	84.0	nd	nd	1.35	0.036	0.064	0.339	7.0
PTB-1	77.2	nd	0.0022	0.56	0.004	0.022	0.251	8.3
PTB-2	77.6	nd	nd	0.44	0.007	nd	0.310	9.0
PTB-3	58.9	nd	nd	0.85	0.010	0.036	0.682	16.1
mean	75.6	-	-	0.72	0.012	-	0.372	9.3

nd=not detected.

DL=detection limit.

Table A2. Organochlorine Pesticides and Total PCB Concentrations in Peregrine Falcon Eggs in New Jersey (ng/g fresh weight).

Sample ID	% Lipid	HCB	Lindane	bBHC	Oxychlor	Hepta Epox	c-Nona	t-Nona	Dieldrin	p,p'-DDE	p,p'-DDD	p,p'-DDT	Mirex	Toxaphene	Total PCBs
Atlantic Coast															
FBARN90-1	3.7	16	nd	7	310	211	68	221	190	4883	14	36	147	312	14901
FBARN90-2	4.3	14	nd	4	247	181	56	172	129	3777	9	31	125	167	13744
FBARN90-3	4.3	20	nd	7	495	243	66	285	139	5734	14	36	188	170	12623
FBARN90-4	4.0	17	nd	6	380	203	68	217	224	6753	18	39	170	170	15419
FBARN91	4.1	18	nd	6	517	249	56	342	386	1568	64	nd	134	224	11124
FBRIG	5.9	14	nd	10	321	188	44	204	157	11586	36	68	136	161	27856
TWMA-1	3.5	22	nd	7	841	368	84	539	542	24230	57	5	515	264	21868
TWMA-2	5.1	16	nd	6	685	339	77	492	391	12793	41	33	204	248	20586
MWMA	2.3	20	nd	9	384	224	74	315	259	12646	205	nd	405	262	22061
SWMA	4.5	22	nd	5	195	166	21	113	297	5497	46	28	151	63	20068
AC-1	2.6	17	nd	8	207	134	25	84	101	4860	6	21	159	91	27850
AC-2	5.0	19	nd	8	335	137	23	101	107	4246	16	22	194	101	22264
mean	4.1	18.0	-	7.0	409.8	220.2	55.1	257.1	243.3	8214.4	44.0	31.9	210.7	186.1	19196.9
Delaware River															
GPB	4.6	nd	nd	2	74	62	4	14	21	1984	5	11	104	23	7212
WWB	2.1	5	nd	3	437	332	50	224	224	8045	101	5	54	57	15409
PTB-1	5.7	3	66	nd	298	213	24	97	161	4974	38	17	22	108	15812
PTB-2	6.5	6	76	3	325	259	25	116	209	3782	37	7	53	58	14177
PTB-3	6.5	8	99	nd	300	306	18	130	154	4948	26	18	70	47	20991
mean	5.1	5.6	80.3	2.6	286.6	234.5	24.2	116.2	153.7	4746.6	41.4	11.4	60.4	58.4	14720.4

nd=not detected.

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sheet 1

Table A3. Mono-ortho Substituted PCB Concentrations (ng/g fresh weight) and TEQs (pg/g fresh weight) in Peregrine Falcon Eggs in New Jersey.

	PCB 105 TEF[1]=0.00007		PCB 114 TEF=0.0005		PCB 118 TEF=0.00003		PCB 123 TEF=0.0001		PCB 156 TEF=0.0001		PCB 157 TEF=0.0005		PCB 167 TEF=0.00001		PCB 189 TEF=0.0001		Total mono-o-PCBs	
Sample ID	Conc.	TEQs[2]	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	Total TEQs
Atlantic Coast																		
FBARN90-1	208	14.6	28	14.1	1724	51.7	12	1.2	227	22.7	36	17.8	140	1.4	22	2.2	2397	125.8
FBARN90-2	218	15.3	29	14.7	1835	55.0	12	1.2	224	22.4	34	17.0	136	1.4	23	2.3	2510	129.4
FBARN90-3	192	13.4	23	11.6	1539	46.2	9	0.9	179	17.9	28	13.9	121	1.2	20	2.0	2110	107.1
FBARN90-4	192	13.4	23	11.5	1535	46.1	9	0.9	175	17.5	29	14.6	123	1.2	20	2.0	2107	107.1
FBARN91*	120	8.4	18	9.2	1022	30.7	8	0.8	151	15.1	20	10.2	90	0.9	16	1.6	1446	76.9
FBRIG*	305	21.4	27	13.4	1886	56.6	41	4.1	220	22.0	43	21.3	139	1.4	36	3.6	2699	143.8
TWMA-1*	430	30.1	46	23.0	2665	80.0	19	1.9	419	41.9	61	30.3	298	3.0	69	6.9	4011	217.0
TWMA-2*	369	25.8	44	21.9	2296	68.9	18	1.8	348	34.8	57	28.7	257	2.6	56	5.6	3443	190.0
MWMA	373	26.1	35	17.6	2275	68.3	17	1.7	347	34.7	52	26.0	238	2.4	54	5.4	3389	182.0
SWMA	358	25.1	31	15.7	2132	64.0	97	9.7	181	18.1	40	20.2	139	1.4	21	2.1	3001	156.1
AC-1	271	19.0	29	14.6	1700	51.0	14	1.4	301	30.1	45	22.4	148	1.5	62	6.2	2568	146.1
AC-2	224	15.7	24	11.8	1320	39.6	12	1.2	210	21.0	33	16.3	123	1.2	53	5.3	1998	112.2
mean	272	19.0	29.9	14.9	1827	54.8	22.4	2.2	248	24.8	39.8	19.9	163	1.6	38	3.8	2640	141.1
Delaware River																		
GPB	61	4.3	13	6.4	463	13.9	nq	-	125	12.5	13	6.4	41	0.4	20	2.0	517	45.9
WWB	131	9.2	22	10.9	917	27.5	10	1.0	188	18.8	26	13.1	67	0.7	17	1.7	948	82.8
PTB-1	161	11.2	30	15.2	1040	31.2	9	0.9	265	26.5	37	18.4	63	0.6	20	2.0	747	106.0
PTB-2*	99	7.0	18	9.0	624	18.7	5	0.5	176	17.6	25	12.4	43	0.4	14	1.4	444	67.0
PTB-3*	142	9.9	27	13.6	954	28.6	9	0.9	260	26.0	36	18.1	57	0.6	18	1.8	498	99.5
mean	118.8	8.3	22.0	11.0	800	24.0	8.1	0.8	203	20.3	27.4	13.7	54.3	0.5	17.7	1.8	630.8	80.4

[1]TEF=toxicity equivalency factor.

[2]TEQs=2,3,7,8-TCDD equivalents.

*Concentrations based on 44 g average fresh weight.

nq=not quantifiable.

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sheet 7

Table A4. Non-ortho Substituted PCB Concentrations and TEQs in Peregrine Falcon Eggs in New Jersey (pg/g fresh weight).

	3,4,4',5-TCB (PCB 81) TEF[1]=0.02		3,3',4,4'-TCB (PCB 77) TEF=0.02		3,3',4,4',5-PeCB (PCB 126) TEF=0.05		3,3',4,4',5,5'-HxCB (PCB 169) TEF=0.001		
Sample ID	Conc.	TEQs[2]	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Total TEQs
Atlantic Coast									
FBARN90-1	208	4.2	6090	121.8	3565	178.2	579	0.58	304.8
FBARN90-2	201	4.0	5259	105.2	2939	146.9	495	0.49	256.6
FBARN90-3	201	4.0	5562	111.2	3244	162.2	603	0.60	278.1
FBARN90-4	260	5.2	7048	141.0	4137	206.8	674	0.67	353.7
FBARN91*	143	2.9	3263	65.3	2855	142.8	490	0.49	211.4
FBRIG*	395	7.9	9485	189.7	6798	339.9	1107	1.11	538.6
TWMA-1*	250	5.0	3965	79.3	4174	208.7	877	0.88	293.9
TWMA-2*	212	4.2	3022	60.4	3022	151.1	725	0.73	216.5
MWMA	201	4.0	4018	80.4	5190	259.5	1122	1.12	345.0
SWMA	358	7.2	9109	182.2	6421	321.0	836	0.84	511.2
AC-1	137	2.7	1696	33.9	3391	169.6	771	0.77	207.0
AC-2	252	5.0	1629	32.6	3258	162.9	696	0.70	201.2
mean	235	4.7	5012	100.2	4083	204.1	748	0.75	309.8
Delaware River									
GPB	47	0.94	825	16.5	384	19.2	156	0.16	36.8
WWB	38	0.76	334	6.7	785	39.2	305	0.31	47.0
PTB-1	41	0.82	391	7.8	738	36.9	130	0.13	45.7
PTB-2*	52	1.04	407	8.1	768	38.4	127	0.13	47.7
PTB-3*	60	1.21	513	10.3	1027	51.3	166	0.17	63.0
mean	47.7	0.95	494	9.9	740	37.0	177	0.18	48.0

[1]TEF=toxicity equivalency factor.

[2]TEQs=2,3,7,8-TCDD equivalents.

*Concentrations based on 44 g average fresh weight.

Table A6. PCDF Concentrations and TEQs inPeregrine Falcon Eggs in New Jersey (pg/g fresh weight).

	2,3,7,8-TCDF TEF[1]=0.1		1,2,3,7,8-PeCDF TEF=0.05		2,3,4,7,8-PeCDF TEF=0.5		1,2,3,4,7,8-HxCDF TEF=0.1		1,2,3,6,7,8-HxCDF TEF=0.1		1,2,3,4,6,7,8-HpCDF TEF=0.01		1,2,3,4,7,8,9-HpCDF TEF=0.01		OCDF TEF=0.001		
Sample ID	Conc.	TEQs[2]	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	Conc.	TEQs	**Total TEQs**
Atlantic Coast																	
FBARN90-1	71.1	7.1	7.6	0.4	21.8	10.9	3.7	0.4	2.8	0.3	2.2	0.0	0.6	0.0	2.4	0.0	19.1
FBARN90-2	65.1	6.5	6.5	0.3	19.8	9.9	3.2	0.3	2.5	0.2	nq	-	nd	-	1.5	0.0	17.3
FBARN90-3	78.6	7.9	8.3	0.4	23.0	11.5	3.7	0.4	2.8	0.3	nq	-	nd	-	1.5	0.0	20.4
FBARN90-4	82.9	8.3	7.8	0.4	25.4	12.7	4.0	0.4	3.2	0.3	nq	-	nq	-	nq	-	22.1
FBARN91*	30.2	3.0	3.7	0.2	16.9	8.5	3.9	0.4	2.7	0.3	nq	-	nd	-	0.8	0.0	12.3
FBRIG*	87.6	8.8	11.2	0.6	60.5	30.3	6.3	0.6	3.5	0.3	nq	-	nq	-	1.4	0.0	40.6
TWMA-1*	26.7	2.7	2.3	0.1	17.1	8.6	1.7	0.2	1.5	0.1	nq	-	1.0	0.0	1.7	0.0	11.7
TWMA-2*	21.0	2.1	2.3	0.1	13.9	7.0	1.7	0.2	1.5	0.2	nq	-	0.9	0.0	2.0	0.0	9.5
MWMA	40.5	4.1	4.2	0.2	35.0	17.5	3.7	0.4	2.2	0.2	nq	-	0.7	0.0	nq	-	22.3
SWMA	89.4	8.9	11.2	0.6	44.6	22.3	4.6	0.5	3.3	0.3	nq	-	nq	-	nq	-	32.6
AC-1	12.6	1.3	nq	-	25.4	12.7	2.8	0.3	1.8	0.2	nq	-	0.2	0.0	nq	-	14.4
AC-2	11.4	1.1	nq	-	25.2	12.6	2.7	0.3	1.3	0.1	nq	-	nq	-	nq	-	14.1
mean	51.4	5.1	6.5	0.3	27.4	13.7	3.5	0.3	2.4	0.2	-	-	-	-	-	-	19.8
Delaware River																	
GPB	2.7	0.3	0.3	0.0	8.1	4.1	1.3	0.1	0.7	0.1	nq	-	nd	-	nd	-	4.5
WWB	2.2	0.2	nq	-	11.2	5.6	1.9	0.2	1.6	0.2	nq	-	nq	-	nd	-	6.2
PTB-1	1.1	0.1	nq	-	7.8	3.9	2.4	0.2	1.5	0.2	nq	-	nd	-	nd	-	4.4
PTB-2*	1.4	0.1	nq	-	7.5	3.7	2.5	0.2	1.4	0.1	nq	-	nd	-	nd	-	4.2
PTB-3*	1.2	0.1	nd	-	8.5	4.2	3.0	0.3	1.5	0.2	nq	-	nd	-	nd	-	4.8
mean	1.7	0.2	0.3	0.0	8.6	4.3	2.2	0.2	1.3	0.1	-	-	-	-	-	-	4.8

[1]TEF=toxicity equivalency factor.

[2]TEQs=2,3,7,8-TCDD equivalents, rounded to nearest 0.1 pg/g.

*Concentrations based on 44 g average fresh weight.

nq=not quantifiable; nd=not detected. Congeners not listed were either nd or nq in all samples.

Table A7. Total TEQ Concentrations Due to Coplanar PCBs, PCDDs and PCDFs in Peregrine Falcon Eggs in New Jersey.

Nest location	Mono-*o*-PCBs	Non-*o*-PCBs	PCDDs	PCDFs	PCDDs+PCDFs	**Total**
Atlantic Coast		TEQs (pg/g fresh weight)				**TEQs**
FBARN90-1	125.8	304.8	22.3	19.1	41.4	471.9
FBARN90-2	129.4	256.6	20.2	17.3	37.5	423.6
FBARN90-3	107.1	278.1	23.9	20.4	44.3	429.5
FBARN90-4	107.1	353.7	26.3	22.1	48.5	509.3
FBARN91*	76.9	211.4	17.7	12.3	30.0	318.3
FBRIG*	143.8	538.6	33.9	40.6	74.5	756.9
TWMA-1*	217.0	293.9	14.0	11.7	25.7	536.6
TWMA-2*	190.0	216.5	10.0	9.5	19.5	426.0
MWMA	182.0	345.0	44.2	22.3	66.5	593.6
SWMA	156.1	511.2	37.0	32.6	69.6	737.0
AC-1	146.1	207.0	27.6	14.4	42.0	395.1
AC-2	112.2	201.2	25.8	14.1	39.9	353.3
mean	141.1	309.8	25.2	19.8	45.0	496.0
Delaware River						
GPB	45.9	36.8	2.8	4.5	7.3	90.0
WWB	82.8	47.0	7.6	6.2	13.7	143.6
PTB-1	106.0	45.7	6.4	4.4	10.8	162.5
PTB-2*	67.0	47.7	6.5	4.2	10.8	125.5
PTB-3*	99.5	63.0	7.6	4.8	12.4	175.0
mean	80.4	48.0	7.2	4.8	12.1	140.5

[2]TEQs=2,3,7,8-TCDD equivalents, rounded to nearest 0.1 pg/g.

*Concentrations based on 44 g average fresh weight.

Appendix B

Methodology for Estimation of Fresh Weight Concentrations

Appendix B. Methodology for Estimation of Fresh Weight Concentrations.

Fresh weight of eggs (weight of egg contents adjusted to account for dehydration and lipid utilization) were calculated using the relationship identified by Stickel et al. (1973):

Egg volume = Fresh weight = $0.51 \times L \times B^2$

where:

V	=	Egg volume (ml)
FW	=	Fresh weight (g)
L	=	Length (cm)
B	=	Breadth (cm)

The reported fresh weight concentrations were then calculated using the relationship defined below:

$$FWC = \frac{ECW \times WWC}{FW}$$

where:

FWC	=	Fresh weight concentration ($\mu g/g$)
ECW	=	Egg content weight (harvested mass without shell (g)
WWC	=	Wet weight concentration ($\mu g/g$)

Fresh weights of whole egg samples were averaged to estimate fresh weights for samples for which no measurement data were available (PTB-2, PTB-3, FBARN91, FBRIG, TWMA-1, and TWMA-2).