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PRODUCTIVITY OF AMERICAN ROBINS EXPOSED TO POLYCHLORINATED BIPHENYLS, HOUSATONIC RIVER, MASSACHUSETTS, USA

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?1 **Abstract**—American robins (*Turdus migratorius*) breeding in the Housatonic River (MA, USA) watershed were studied in the field in 2001 to determine whether productivity was adversely affected by exposure to polychlorinated biphenyls (PCBs), as would be suggested by extrapolation from laboratory studies on other avian species. The study involved identifying nests within the Housatonic River floodplain (target area) and in reference areas beyond foraging distance of the floodplain, monitoring clutch size and number hatched and fledged, collecting eggs and nestlings for analysis for PCBs, and testing for differences in productivity between populations. One hundred and six active robin nests were monitored. Although concentrations of PCBs in target specimens were more than two orders of magnitude greater than in reference specimens, the only statistically significant differences in productivity were inconsistent with an exposure-related effect. First-generation productivity of exposed robins was within the range of natural background variation. Bioequivalence tests confirmed that first-generation productivity was statistically and biologically equivalent in target and reference robins. These findings contrast with extrapolations from laboratory studies of other avian species.

Keywords—Robins Polychlorinated biphenyls Reproduction Housatonic River

INTRODUCTION

Polychlorinated biphenyls (PCBs) are nearly ubiquitous environmental contaminants that are widely reported to impair hatching success in birds [1]. Much of the research on reproductive effects of PCBs on birds has been conducted in the laboratory. Laboratory toxicity tests, however, cannot replicate exposures and population-level responses in a complete ecological context in which population dynamics are also a function of competition, predation, and other natural stressors. Such a population-level perspective is desirable when conducting ecological risk assessments for sites contaminated with PCBs [2]. Interspecific variability in toxicological sensitivity to PCBs also confounds interpretation of laboratory studies, which generally employ either domesticated species or wild species that can be raised in captivity (e.g., mallards [*Anas platyrhynchos*] and ring-necked pheasants [*Phasianus colchicus*] [3,4]). Furthermore, PCB mixtures and dosing regimes used in laboratory tests may not reflect environmentally relevant exposures. For all these reasons, many avian toxicological studies are of limited use for ecological risk assessments because they cannot accurately translate individual-level effects observed in the laboratory to population-level responses that might occur in wild species in a natural setting.

Several field studies have been conducted with the objective of characterizing the ecological risks posed by PCBs to birds. These studies have generally focused on hatching success of piscivorous species, usually in systems with multiple chemicals present. For example, Hoffman et al. [5] reported adverse effects on hatching success and hatchling weight in Forster's terns (*Sterna forsteri*) exposed to PCBs, dioxins, and other synthetic organic compounds. Working with the same colonies of Forster's terns, Kubiak et al. [6] also observed effects on incubation period, hatching success, and hatchling weight. Tillett et al. [7] observed a significant correlation between reproductive outcome and total concentrations of PCBs in composites of eggs of double-crested cormorants (*Phalacrocorax auritus*) collected from the Great Lakes. Becker et al. [8] observed that concentrations of higher-chlorinated PCBs in common tern (*Sterna hirundo*) eggs that failed to hatch were 20% higher than in randomly selected eggs. Tree swallows (*Tachycineta bicolor*) are one of the few passerine species that have been evaluated in field studies of PCB effects, but findings have been equivocal [9,10]. We identified just two field studies on the effects of PCBs on birds exposed via the soil-to-terrestrial invertebrate food chain, neither of which reported concentrations of PCBs in eggs, nestlings, or adults [11,12].

Despite the paucity of information on the effects of PCBs on passerines exposed via the soil-to-terrestrial invertebrate food chain, ecological risk assessments often include American robins (*Turdus migratorias*) as a receptor of interest. In such cases, risks to robins are usually estimated on the basis of interspecific extrapolations from laboratory experiments from species, such as mallards and ring-necked pheasant, for which no-observed-adverse-effects levels or lowest-observed-adverse-effects levels are available [13,14]. Despite the uncertainties associated with such extrapolations, risk management and remediation decisions may be influenced by their conclusions. For example, remediation of floodplain soils at the Sheboygan River and Harbor Site in Sheboygan, Wisconsin, USA, was justified on the basis of theoretical risks posed by PCBs

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to robins, as predicted by extrapolations from the literature on impaired hatchability and malformations in other avian species [15].

This paper presents a field study conducted at another PCB site, the Housatonic River in Massachusetts, USA, with the objective of comparing field observations of first-generation productivity in robins exposed to PCBs to predictions based on extrapolations from the toxicological literature on other species. Between the 1930s and 1977, PCBs were used in manufacturing processes at the General Electric Company facility in Pittsfield (MA, USA). Prior to 1977, releases of PCBs were conveyed to the Housatonic River and subsequently deposited in downstream sediments. During periodic flooding, river sediments containing PCBs were resuspended and deposited on soils throughout the 10-year floodplain [16]. Figure 1 illustrates the study site, including the location of the 1-mg/kg-PCB isopleth, which approximately matches the 10-year floodplain. Within this 16-km reach of the river, concentrations of PCBs in surface soils of the floodplain range from below detection to 334 mg/kg (mean = 15 mg/kg, standard error [SE] = 0.60). Concentrations of PCBs in earthworms range from 1.1 to 27 mg/kg (mean = 13 mg/kg, SE = 1.7), while concentrations in soil litter invertebrates range from 1.4 to 4.9 mg/kg (mean = 3.4 mg/kg, SE = 0.44) [16].

Screening-level risks posed by PCBs to robins may be estimated by combining the available data on concentrations of PCBs in prey with the dose equation and exposure parameter values for robins [13]. Based on default literature-based assumptions, adult robins inhabiting the Housatonic River floodplain would be predicted to ingest approximately 7.8 mg/kg/d PCBs. Reference toxicity values (safe doses) generated from the avian toxicological literature for first-generation reproductive effects of PCB Aroclor® 1254 (Monsanto, St. Louis, MO, USA; a highly chlorinated commercial PCB mixture similar to the types of PCBs originally released at this site) range from 0.18 mg/kg/d (based on pheasants) [3] to 1.4 mg/kg/d (based on mallards) [4]. Because the estimated dose (7.8 mg/kg/d) exceeds the range of reference toxicity values (0.18–1.4 mg/kg/d), such a literature-based extrapolation would predict that Housatonic River robins are at risk because of exposure to PCBs. We conducted this study to assess whether robins in the Housatonic have high levels of PCBs and whether PCB exposure translates to reduced productivity.

Several characteristics of robins facilitate evaluation of their productivity in the field. Sufficient sample sizes can be obtained because robins are common, they inhabit a variety of habitats, and their nests are not highly camouflaged. The source of exposure to robins can be well defined because of their limited foraging range (300 m from the nest [17]) and diet (predominantly terrestrial invertebrates) [13]. It is technically feasible to monitor productivity in robins based on hatching success, fledging success, incubation period, nestling period, Mayfield nest success [18], and rates of abandonment and depredation. These reproductive endpoints are meaningful in that they closely parallel the effects evaluated in the avian toxicological studies most often used as a source of toxicity reference values. Although it is possible that PCBs may also impair thyroid function as well as growth, survival, and productivity of offspring [19–22], it is far less feasible to evaluate these endpoints through a field study and within the time constraints typical of ecological risk assessments.

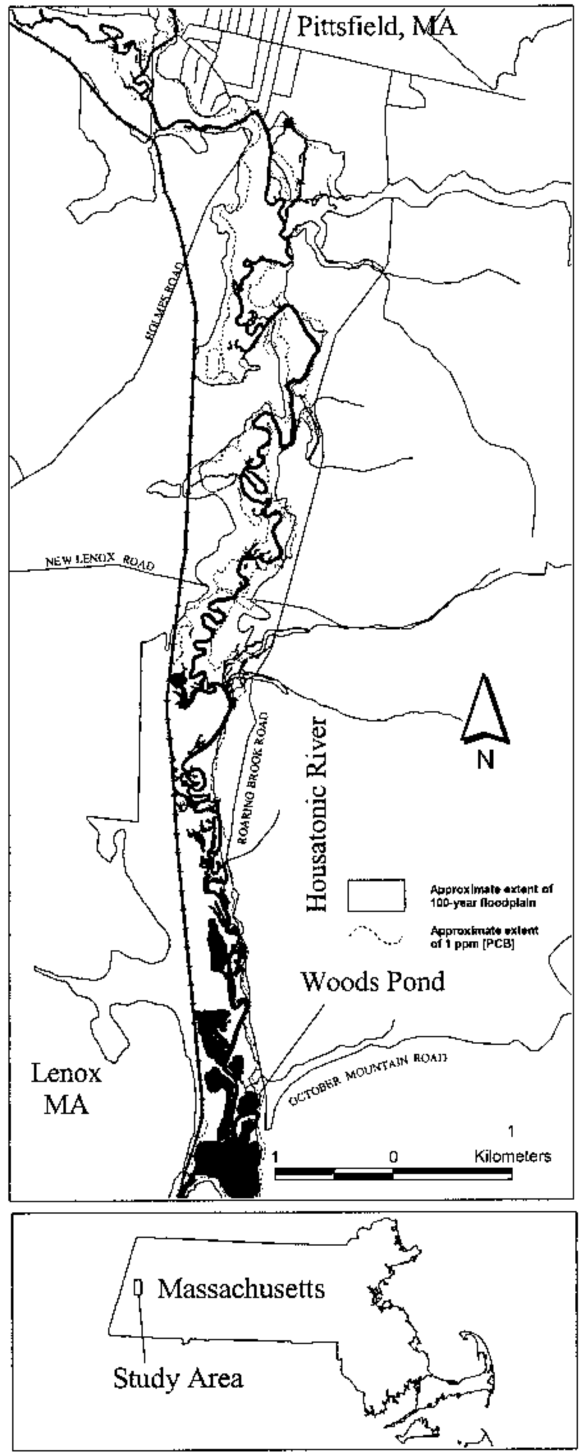

Fig. 1. Robin study area. Housatonic River between Pittsfield and Woods Pond (MA, USA).

METHODS

This study was conducted during the 2001 breeding season and involved identifying as many robin nests as possible both within the Housatonic River floodplain and in reference areas beyond foraging distance of the floodplain; monitoring clutch size, number hatched, and number fledged in each of those nests; collecting one egg and one 7-d-old nestling from each nest when possible and analyzing those samples to determine concentrations of PCBs; and evaluating data for differences in reproductive performance in the exposed (target) and unexposed (reference) populations. Nest outcome, fertility, survival to hatching, development, and survival to fledging were evaluated as reproductive endpoints. Parental care was indirectly evaluated on the basis of rates of depredation and abandonment.

The study area encompassed the Housatonic River watershed in Berkshire County (MA, USA). The target area was restricted to the 10-year floodplain of the Housatonic River, from the confluence of the East Branch and West Branch of the Housatonic River to Woods Pond Dam. All reference areas were public lands within the Housatonic River watershed that were located well over 300 m from the 10-year floodplain of the Housatonic River.

In order to avoid confounding effects of decreased productivity later in the season and with second broods [23], only broods assumed to be first broods were included in the study. Broods were assumed to be second broods if they were observed in nests where broods had already been completed in 2001 or if they were initiated in July or August, which is the period during which clutch size declines most dramatically [23].

Active nests were monitored following the methodology described by Martin and Geupel [24]. Each nest was visited approximately every 3 d, at which time the numbers of eggs and nestlings present were recorded. Nest outcome was also recorded to document instances when nests failed because of abandonment or depredation. Nests were classified as depredated if all eggs or nestlings disappeared before the young were old enough to fledge. If young were absent but old enough to fledge, nests were classified as successful.

One randomly selected viable egg was collected after approximately 10 d of incubation from accessible nests containing four or more eggs. One 7-d-old nestling was randomly collected from each nest containing three or more nestlings whenever possible. External anatomy was evaluated for deformities, and samples were then transferred to Northeast Analytical Environmental Lab Services of Schenectady (NY, USA), for chemical analysis. Feathers, beaks, and legs of nestling samples were removed before analysis. All samples were analyzed for PCBs using SW-846 Method 8082 (http://www.epa.gov/epaoswer/hazwaste/test/sw846.htm [25]). The practical quantitation limits for individual Aroclors ranged from 0.139 to 12.8 mg/kg for target eggs, from 0.0512 to 1.61 mg/kg for target nestlings, from 0.135 to 0.209 mg/kg for reference eggs, and from 0.0510 to 0.0554 mg/kg for reference nestlings (all concentrations given as wet wt). In cases where the practical quantitation limits were elevated (e.g., target eggs), results were well above the practical quantitation limits. The PCB analytical results that were below the laboratory's instrument detection limit (U-qualified) were assigned a concentration equal to one-half the practical quantitation limit for purposes of describing that specimen's assumed exposure.

The number of successful nests was calculated as the total number of nests that fledged at least one young. Although this approach does not account for when the individual nests were found, as the Mayfield method [18] does, it offers a simple measure of overall performance that is necessary for comparisons with other published studies that did not apply the Mayfield method [18]. Contrast [26] was used to test Mayfield's [18] index of nest success, which is based on the daily predation and survival rates of nests. Hatching success was calculated as the number of young hatched divided by the number of eggs present just before hatching [18]. Because this definition does not account for the collected viable eggs that would have hatched had they not been collected, it underestimates what the hatching success would have been in the absence of egg collections. Fledging success was calculated as the ratio of young fledged to young hatched for all successful nests. Again, the outcome of collected eggs and nestlings was ignored, likely biasing this measure low.

The number of nestlings hatched per successful nest was calculated in two ways. First, the actual number hatched was counted, ignoring the likelihood that most viable eggs that were collected from successful nests would have hatched had they not been collected; this was referred to as the range-low number. Second, the number of nestlings that would have hatched had the eggs not been collected from successful nests was estimated; this was referred to as the range-high number. In this case, it was assumed that the collected viable eggs would have hatched. Because the former value may be biased low and the latter biased high, the two values define the range of nestlings hatched per successful nest. In calculating the number of nestlings fledged per successful nest, range-low and range-high numbers fledged were calculated using the same approach.

Statistical analyses were conducted using Power and Sample Size for Windows (Pass) (Ver 2000 NCSS, Kaysville, UT, USA). Student's t tests were performed to determine if statistically significant differences were observed between reference and target data for factors related to exposure and effects. For comparisons involving heterogeneous variances and sample sizes differing by more than 10%, Welch's approximate t was used to calculate the significance of the comparison. Fisher's exact tests were used to test for differences in measures based on proportional data. Statistical comparisons were considered significant at an alpha level (α) of 0.05.

Bioequivalence tests of means were conducted for the most ecologically relevant endpoints, using the methodology described by Hintze [27] and Blackwelder [28]. Under conventional hypothesis testing, the absence of statistically significant differences does not imply that the means are equivalent. In a bioequivalence test, if statistical significance is achieved ($p < 0.05$), it indicates that means are not distinguishable within the range defined for the test (δ). The null hypotheses tested through the bioequivalence approach were that the target area mean numbers of young hatched or fledged per nest were more than one-half of a nestling less than the reference area mean and that the mean hatching success and fledging success for the target area were more than 20% lower than that of the reference area. The δ of 0.5 is within the range of natural variability (as discussed later). The δ of 20% is consistent with the level of effect considered potentially significant at the Oak Ridge Reservation as well as under several regulatory auspices, including the Chronic National Ambient Water Quality Criteria, the National Pollution Discharge Elimination System, the rapid bioassessment procedure (see [29]). No other de minimis ecological effect level has been proposed in the literature, regulations, or guidance.

RESULTS

A total of 106 active robin nests were located and monitored during the 2001 breeding season. Of these, 44 were located in the reference area and 62 in the target area. The nests monitored likely represent a substantial proportion of the actual robin population breeding within the study area in light of the

Table 1. Bioequivalence test results

	Biologically relevant difference (δ)	Probability level (p)	Power
Range-low number of nestlings hatched	0.5	0.027	0.62
Range-high number of nestlings hatched	0.5	0.000070	1.0
Range-low number of nestlings fledged	0.5	0.00029	0.98
Range-high number of nestlings fledged	0.5	0.000019	1.0
Hatching success	20%	0.00068	0.96
Fledging success	20%	0.0014	0.97

intensity of the nest searching effort and the experience of the field biologists. The concentration of PCBs in target eggs (mean = 84 mg/kg wet wt, SE = 22, n = 9) was significantly greater than in reference eggs (mean = 0.15 mg/kg wet wt, SE = 0.085, n = 2) (t = 3.8, df = 9, p = 0.0021). Similarly, the concentration of PCBs in target nestlings (mean = 12 mg/kg wet wt, SE = 4.7, n = 11) was significantly greater than in reference nestlings (mean = 0.037 mg/kg wet wt, SE = 0.0062, n = 6) (t = 2.54, df = 15, p = 0.011).

The proportion of target nests that were successful was 29% (n = 62), whereas the proportion of reference nests that were successful was 25% (n = 44), a difference that was not statistically significant (Fisher's exact test, p = 0.67). The proportion of target nests that fledged at least two nestlings was 27% (n = 62), whereas the proportion of reference nests that fledged at least two nestlings was 20% (n = 44), a difference that also was not significant (Fisher's exact test, p = 0.50). The proportions of nests that were abandoned in the target area (1.6%, n = 62) and the reference area (6.8%, n = 44) were not significantly different (Fisher's exact test, p = 0.30). The proportions of depredated nests in the target area (69%, n = 62) and the reference area (68%, n = 44) also were not significantly different (Fisher's exact test, p = 0.83). Lower numbers of young were either abandoned or depredated per target nest (mean = 0.64, SE = 0.14, n = 22 nests) than per reference nest (mean = 1.8, SE = 0.42, n = 19 nests), a difference that was significant (Welch's approximate t = 2.6, df = 32, p = 0.013) but opposite that which would be predicted if PCBs were assumed to impair food availability or parental care.

The overall mean Mayfield daily predation rate for the reference area was 7.7% (SE = 1.4%, n = 338.5 exposure days), which corresponds to an overall nest success or survival rate of 11%. The overall mean daily predation rate for the target area was 4.6% (SE = 0.80%, n = 711 exposure days), which corresponds to an overall survival rate of 26%. These differences were not significant (Contrast: chi-square = 3.4, df = 1, p = 0.065). Possible seasonal effects were considered by calculating the Mayfield index for the target area nests with the 25 earliest egg dates (defined as early nests) and the 25 latest egg dates (defined as late nests). No significant differences were observed between early target nests (mean = 5.7%, SE = 1.5%, n = 229.5 exposure days) and late target nests (mean = 4.2%, SE = 0.91%, n = 481.5 exposure days) (Contrast: chi-square = 0.63, df = 1, p = 0.43). However, daily predation rates of late target nests (mean = 4.2%, SE = 0.91%, n = 481.5 exposure days) and reference nests (mean = 7.7%, SE = 1.5%, n = 338.5 exposure days) differed significantly (chi-square = 4.3, df = 1, p = 0.039), suggesting that differences in depredation were related to differences in habitat (e.g., quality of cover for robins, quality of habitat for predators) and predator density in the target and reference areas to a greater extent than seasonal differences.

The clutch sizes for target nests (mean = 3.6, SE = 0.13, n = 39 nests) and reference nests (mean = 3.3, SE = 0.10, n = 29 nests) did not differ significantly (t = −1.5, df = 66, p = 0.14). The numbers of nonviable eggs per successful target nest (mean = 0.47, SE = 0.17, n = 17 nests) and per successful reference nest (mean = 0.22, SE = 0.22, n = 9 nests) also did not differ significantly (t = 0.86, df = 24, p = 0.40). The proportion of nonviable eggs per successful target nest (mean = 11%, SE = 4.0%, n = 17 nests) and per successful reference nest (mean = 5.6%, SE = 5.6%, n = 9 nests) did not differ significantly (t = −0.78, df = 24, p = 0.28).

The incubation periods for target area nests (mean = 14 d, SE = 0.20, n = 10 nests) and reference area nests (mean = 14 d, SE = 0, n = 3 nests) were not significantly different (t = −0.53, df = 11, p = 0.078). The range-low numbers of nestlings hatched per successful target area nest (mean = 2.9, SE = 0.14, n = 18 nests) and per successful reference area nest (mean = 2.6, SE = 0.24, n = 11 nests) were not significantly different (t = −0.98, df = 27, p = 0.34). The range-high number of nestlings hatched per successful target nest (mean = 3.2, SE = 0.13, n = 18 nests) was greater than the number hatched per successful reference nest (mean = 2.7, SE = 0.20, n = 11 nests), a difference that was significant (t = −2.2, df = 27, p = 0.036) but opposite that which would be predicted by an exposure-related effect. Mayfield's hatching success in target area successful nests (mean = 93%, SE = 3.5%, n = 22 nests) and in reference area successful nests (mean = 96%, SE = 3.04%, n = 18 nests) did not differ significantly (t = 0.55, df = 38, p = 0.58). The development of embryos and nestlings was evaluated by examining external anatomy. No morphological abnormalities (n = 28) were observed in any of the specimens examined.

Nestling periods for target nests (mean = 14 d, SE = 0.21, n = 15 nests) and reference nests (mean = 14 d, SE = 0.18, n = 9 nests) were not significantly different (t = −1.3, df = 22, p = 0.22). The range-low numbers of nestlings fledged per successful target area nest (mean = 2.2, SE = 0.13, n = 18 nests) and per successful reference area nest (mean = 1.9, SE = 0.16, n = 11 nests) were not significantly different (t = 1.5, df = 27, p = 0.16). The range-high number of nestlings fledged per successful target nest (mean = 3.2, SE = 0.03, n = 18 nests) was greater than the number fledged per successful reference nest (mean = 2.5, SE = 0.25, n = 11 nests), a difference that was significant (t = 2.8, df = 27, p = 0.0098) but opposite that which would be predicted if PCBs were assumed to impair nestling survival. Fledging success of target nestlings (mean = 98%, SE = 1.83%, n = 18 nests) and

Table 2. Productivity results compared to literature values

Endpoint	Target nests	Other studies	Citation
Proportion successful nests	29	8.3–75	[30,31]
Number nestlings fledged per successful nest	2.2–3.2	1.0–4.2	[30,32]
Proportion nestlings fledged per successful nest	98	62–100	[30,33,34]
Mayfield nest success (%)	26	18–90	[35,36]

reference nestlings (mean = 91%, SE = 6.51%, $n = 11$ nests) was not significantly different ($t = -1.3$, $df = 27$, $p = 0.20$).

Bioequivalence testing [27] was employed to determine whether any potentially biologically relevant differences existed in numbers of nestlings hatched and fledged, hatching success, and fledging success between the target and the reference populations (Table 1). In all cases tested, the null hypothesis (that there is a biologically relevant difference between target and reference populations) was rejected with $p < 0.05$ and relatively high power ($p = 0.62–1.0$). Hence, the sample sizes were sufficient to demonstrate the statistical and biological equivalence of first-generation productivity in target and reference populations.

DISCUSSION

Despite substantial differences in the degree to which target and reference area robins were exposed to PCBs, this study provides no evidence of adverse effects of PCB exposures on first-generation productivity of exposed robins. Exposure to PCBs in the target robin population was more than two orders of magnitude greater than in reference robins. Indeed, the concentration of PCBs in target robin eggs (mean = 84 mg/kg wet wt) may be the highest reported for any passerine species. By way of comparison, the arithmetic mean concentration of PCBs in tree swallow pippers (eggs and newly hatched nestlings) nesting in boxes along the Housatonic River between 1998 and 2000 was 69 mg/kg wet weight (C. Custer, U.S. Geological Survey, La Crosse, WI, USA, unpublished data). Mean concentrations of total PCBs in eggs of tree swallows nesting in multiple grids along the Hudson River ranged from 9.3 to 29.5 mg/kg [9]. Nevertheless, we did not find any adverse effect of the PCB exposure on the productivity endpoints assessed. In addition, based on review of similar productivity endpoints in other studies (which did not involve PCB exposure), measures of first-generation productivity for exposed robins were within the range of natural variability for robins (Table 2).

Literature-based extrapolations from other avian species would predict that exposed robins are at risk in that their estimated doses of PCBs are at least five times greater than toxicity reference values for first-generation reproductive effects. This site-specific field study illustrates the uncertainty associated with such extrapolative approaches, at least in the case of exposure of robins to PCBs. Neither the extrapolative approach nor the field study addressed effects on survival or productivity of offspring. Rather, the discrepancy in findings between the extrapolative approach and the field study may be due to differences in dosing regime and interspecific variability in sensitivity to PCBs. Although it seems intuitive that toxicity reference values derived from other avian species are unlikely to provide useful information on the sensitivity of robins to PCBs, many ecological risk assessments conducted for regulatory purposes depend on such interspecific extrapolations.

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REFERENCES

1. Barron MG, Galbraith H, Beltman D. 1995. Comparative reproductive development toxicology of PCBs in birds. *J Comp Biochem Physiol C Comp Pharmacol Toxicol* 112:1–14.
2. U.S. Environmental Protection Agency. 1999. Issuance of final guidance: Ecological risk assessment and risk management principles for Superfund sites. OSWER Directive 9285.7-28P. Office of Solid Waste and Emergency Response, Washington, DC.
3. Dahlgren RB, Linder RL, Carlson CW. 1972. Polychlorinated biphenyls: Their effects on penned pheasants. *Environ Health Perspect* 1:89–101.
4. Custer TW, Heinz GH. 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor 1254. *Environ Pollut* 21A:313–318.
5. Hoffman DJ, Rattner BA, Sileo L, Docherty D, Kubiak TJ. 1987. Embryotoxicity, teratogenicity, and aryl hydrocarbon hydroxylase activity in Forster's terns on Green Bay, Lake Michigan. *Environ Res* 42:176–184.
6. Kubiak TJ, Harris HJ, Smith LM, Schwartz TR, Stalling DL, Trick JA, Sileo L, Docherty DE, Erdman TC. 1989. Microcontaminants and reproductive impairment of the Forster's tern on Green Bay, Lake Michigan—1983. *Arch Environ Contam Toxicol* 18:706–727.
7. Tillett DE, Ankley GT, Giesy JP, Ludwig JP, Kurita-Matsuba H, Weseloh DV, Ross PS, Bishop CA, Sileo L, Stromborg KL, Larson J, Kubiak TJ. 1992. Polychlorinated biphenyl residues and egg mortality in double-crested cormorants from the Great Lakes. *Environ Toxicol Chem* 11:1281–1288.
8. Becker PH, Shumann S, Koepff C. 1993. Hatching failure in common terns (*Sterna hirundo*) in relation to environmental chemicals. *Environ Pollut* 79:207–213.
9. McCarty JP, Secord AL. 1999. Reproductive ecology of tree swallows (*Tachycineta bicolor*) with high levels of polychlorinated biphenyl contamination. *Environ Toxicol Chem* 18:1433–1439.
10. Stapleton M, Dunn PO, McCarty J, Secord A, Whittingham LA. 2001. Polychlorinated biphenyl contamination and minisatellite DNA mutation rates of tree swallows. *Environ Toxicol Chem* 20: 2263–2267.
11. Henning MH, Ebert ES, Keenan RE, Martin SG, Duncan JW. 1997. Assessment of effects of PCB-contaminated floodplain soils on reproductive success of insectivorous songbirds. *Chemosphere* 34:1121–1137.
12. Arenal CA, Halbrook RS. 1997. PCB and heavy metal contamination and effects in European starlings (*Sturnus vulgris*) at a Superfund site. *Bull Environ Contam Toxicol* 58:254–262.
13. U.S. Environmental Protection Agency. 1993. *Wildlife Exposure Factors Handbook*. EPA/600/R-93/187. Office of Research and Development, Washington, DC.
14. U.S. Environmental Protection Agency. 1997. Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments. Interim Final. EPA 540-R-97-006. Office of Solid Waste and Emergency Response, Edison, NJ.
15. U.S. Environmental Protection Agency. 2000. U.S. EPA Superfund record of decision: Sheboygan River and Harbor, Sheboygan, WI. Region 5, Chicago, IL.
16. Blasland, Bouck and Lee. 2002. RCRA Facility Investigation. Syracuse, NY, USA.
17. Knupp DM, Owen RB Jr, Dimond JB. 1977. Reproductive biology of the American robin in northern Maine. *Auk* 94:80–85.
18. Mayfield HF. 1975. Suggestions for calculating nest success. *Wilson Bull* 87:456–466.

19. Koval PJ, Peterle TJ, Harder JD. 1987. Effects of polychlorinated biphenyls in mourning dove reproduction and circulating progesterone levels. *Bull Environ Contam Toxicol* 39:663–670.
20. Fernie KJ, Smits JE, Bortolotti GR, Bird DM. 2001. Reproductive success of American kestrels exposed to dietary polychlorinated biphenyls. *Environ Toxicol Chem* 20:776–781.
21. Fernie KJ, Smits JE, Bortolotti GR, Bird DM. 2001. In ovo exposure to polychlorinated biphenyls: Reproductive effects on second-generation American kestrels. *Arch Environ Contam Toxicol* 40:544–550.
22. Fernie K, Smits J, Bortolotti G. 2003. Developmental toxicity of in ovo exposure to polychlorinated biphenyls: I. Immediate and subsequent effects of first-generation nestling American kestrels (*Falco sparverius*). *Environ Toxicol Chem* 22:554–560.
23. Howard DV. 1967. Variation in the breeding season and clutch-size of the robin in the Northeastern United States and the Maritime Provinces of Canada. *Wilson Bull* 79:432–440.
24. Martin TE, Geupel GR. 1993. Nest-monitoring plot methods for locating nests and monitoring nest success. *J Field Ornithol* 64:507–519.
25. U.S. Environmental Protection Agency. 1998. Test methods for evaluating solid waste, physical/chemical methods. SW-846. Office of Solid Waste, Washington, DC.
26. Hines JF, Sauer JR. 1989. Program Contrast—A general program for the analysis of several survival or recovery rate estimates. Technical Report 24. U.S. Fish and Wildlife Service, Washington, DC.
27. Hintze JL. 2000. Power analysis and sample size for Windows. NCSS, Haysville, UT, USA.
28. Blackwelder WC. 1982. Proving the null hypothesis in clinical trials. *Control Clin Trials* 3:345–353.
29. Suter GW II, Efroymson RA, Sample BE, Jones DS. 2000. *Ecological Risk Assessment for Contaminated Sites.* Lewis, Boca Raton, FL, USA.
30. Ortega CP, Ortega JC, Rapp CA, Vorisek S, Backensto SA, Palmer DW. 1997. Effect of research activity on the success of American robin nests. *J Wildl Manag* 61:948–952.
31. Brehmer PM, Anderson RK. 1992. Effects of urban pesticide application on nesting success of songbirds. *Bull Environ Contam Toxicol* 48:352–359.
32. Sallabanks R, James FC. 1999. American robin (*Turdus migratorius*). In Poole A, Gill F, eds, *The Birds of North America,* 462. The Birds of North America, Philadelphia, PA, USA, pp 1–28.
33. Gill H, Wilson LK, Cheng KM, Trudeau S, Elliott JE. 2000. Effects of Azinphos-methyl on American robins breeding in fruit orchards. *Bull Environ Contam Toxicol* 65:756–763.
34. Kemper DL, Taylor JM. 1981. Seasonal reproductive changes in the American robin (*Turdus migratorius* L.) of the Pacific Northwest. *Can J Zool* 59:212–217.
35. Knupp DM, Owen RB Jr, Dimond JB. 1977. Reproductive biology of the American robin in northern Maine. *Auk* 94:80–85.
36. Morneau F, Lepine C, Decarie R, Villard M, DesGranges J. 1995. Reproduction of American robin (*Turdus migratorius*) in a suburban environment. *Landscape and Urban Planning* 32:55–62.