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Field Estimates of Net Trophic Transfer of PCBs from Prey Fishes to Lake Michigan Salmonids

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Dietary exposure is the principal source of polychlorinated biphenyls (PCBs) to Great Lakes salmonids. PCB transfer efficiency has typically been determined with laboratory studies using high food PCB concentrations and clean animals, making extrapolations to the field uncertain. Here, we present estimates of net PCB transfer efficiencies for Lake Michigan coho salmon, chinook salmon, and lake trout based on 15 years of PCB concentrations in these fish and their prey. Between 1975 and 1990, large changes in Lake Michigan's pelagic food web led to changes in coho and chinook salmon diets and chinook salmon growth efficiency. Early in this period, PCB transfer efficiencies increased in lake trout but decreased in chinook and coho salmon. These early trends carry relatively high uncertainty due to high variability in prey and predator PCB concentrations and are probably not ecologically significant. After 1983, PCB transfer efficiencies remained relatively constant at ~0.55 for lake trout and at ~0.60 for chinook salmon and increased slightly to ~0.50 for coho salmon. PCB transfer efficiencies appear to be little affected by changes in prey PCB concentration, shifts in prey type, and shifts in predator gross growth efficiency. Use of a constant PCB transfer efficiency in models that consider PCB accumulation by piscivorous fishes is therefore appropriate.

Introduction

Fish acquire polychlorinated biphenyls (PCBs) by bioconcentration across the gills and bioaccumulation by dietary extraction from ingested food. Dietary exposure accounts for most PCB accumulation in Great Lakes salmonids (1-4). Some efforts to model PCB accumulation in fishes have incorporated explicit terms for gill exchange (5, 6). However, other models that have ignored gill exchange produce realistic results (7-10). The ability to model PCB accumulation in fish continues to be of interest for predicting risk to wildlife and humans that consume the fish (11, 12).

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Models that examine PCB dynamics in food webs typically assume that PCB transfer efficiency is constant. Whether or not changes in growth efficiency, diet, or PCB congener composition affect PCB transfer efficiencies is unknown. The assumption of a constant PCB transfer efficiency, if false, could have serious consequences for the ability of models to predict PCBs for food webs undergoing change.

A predator's PCB concentration ($[PCB]_j$) is a function of total mass of prey consumed (B_j), prey PCB concentration ($[PCB]_i$); PCB transfer efficiency, corrected for loss terms due to, for example, excretion, egestion, and gill exchange (γ_j); and predator mass (B_j):

$$[PCB]_j = \frac{B_i [PCB]_i \gamma_j}{B_j} \quad (1)$$

Total prey consumed can be calculated from predator mass and average gross growth efficiency (α_j) over the predator's lifetime:

$$B_i = \frac{B_j}{\alpha_j} \quad (2)$$

Equation 1 can be rewritten to express the PCB concentration of a predator as

$$[PCB]_j = \frac{[PCB]_i \gamma_j}{\alpha_j} \quad (3)$$

Thus, predator PCB concentration is inversely related to α_j and directly related to γ_j and prey PCB concentration (Figure 1).

PCB gross assimilation efficiency has typically been determined in the laboratory by measuring fish PCB concentrations following exposure to a diet of known PCB concentration. A related trophic transfer efficiency, γ_j , which represents the net trophic transfer of PCBs after accounting for all sources of depuration and transformations, can also be calculated from field measurements of prey and predator PCB concentrations and α_j , which reflects natural conditions experienced by fishes. Equation 3 can be rearranged to estimate γ_j from such field data:

$$\gamma_j = \frac{[PCB]_j \alpha_j}{[PCB]_i} \quad (4)$$

We estimated γ_j of lake trout (*Salvelinus namaycush*), chinook salmon (*Oncorhynchus tshawytscha*), and coho salmon (*Oncorhynchus kisutch*) from Lake Michigan using eq 4. We coupled 15 years (1975-1990) of PCB concentration data in these salmonids and their prey from contaminant surveys to diet composition and α_j of the predators. During this period, the pelagic food web experienced a marked decline in alewife (*Alosa pseudoharengus*), which led to a diet shift from large alewife to smaller alewife and alternative prey (13) [primarily rainbow smelt (*Osmerus mordax*) but also juvenile yellow perch (*Perca flavescens*), juvenile bloater chub (*Coregonus hoyi*), and slimy sculpin (*Cottus cognatus*)] by chinook and coho salmon but not lake trout. Chinook salmon also experienced a 20% decline in α_j (13) during this period. We show that PCB trophic

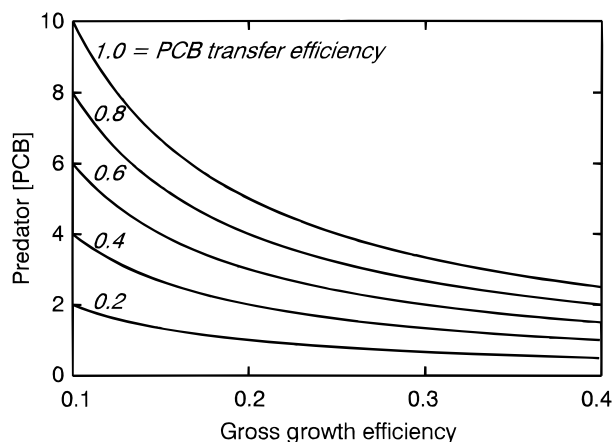


FIGURE 1. PCB concentrations (mg kg^{-1}) of a predator for a range of gross growth efficiencies (α_i) and PCB transfer efficiencies (γ_i) as determined from eq 3. PCB transfer efficiencies are noted above their curves. We have used a constant prey PCB concentration of 1 mg kg^{-1} for this example; higher or lower prey concentrations would shift the predator concentration isopleths up or down, respectively.

transfer efficiency is independent of growth efficiency and diet composition.

Methods

We took α_j for 1978–1988 from a study that examined salmonid growth and diet composition in Lake Michigan (13, 14). We assumed α_j from 1975 to 1977 was the same as 1978 and that 1989 and 1990 were identical to 1988 for each species (Figure 2A) because we are aware of no other data that suggest otherwise. We calculated prey consumption by the average mass predator of the contaminant surveys for each year. There was no significant trend in fish mass over the 15 years; therefore, we simulated fish growth to match the long-term average mass for each species (chinook salmon, 5.0 kg; lake trout, 2.14 kg; coho salmon, 2.03 kg).

We used Lake Michigan salmonid diet data (13, 14) to calculate the mass of invertebrates, large alewife, small alewife, and other forage fishes (smelt and sculpin) consumed by each salmonid species over its lifetime. The age-specific diet proportions were weighted by the proportion of total consumption that occurred during that year of life of the predator (13, 14). To determine the typical prey PCB concentration experienced by a predator, we weighted prey PCB concentration by the proportion of total consumption that prey category represented to the predator's lifetime consumption.

We used previously published time trends for PCB concentrations in salmonids and prey fishes (15) to estimate their median concentrations from 1975 to 1990 (Figure 2B). The average mass of alewife surveyed was 45 g. We decreased large alewife PCB concentrations (Figure 2C) to account for differences between the size of adult alewife eaten by lake trout (16) and coho and chinook salmon (17) and those captured by the DNR trawls (i.e., to account for differential size selective predation by the salmonids). The median PCB concentration (mg kg^{-1}) of the salmonids (Figure 2B) and alewife (Figure 2C), their principal forage, decreased during this 15-year period. We estimated PCB concentrations of small (average mass 5.9 g) alewives consumed with a PCB accumulation model previously developed for Lake Michigan alewives (8). Sufficient data

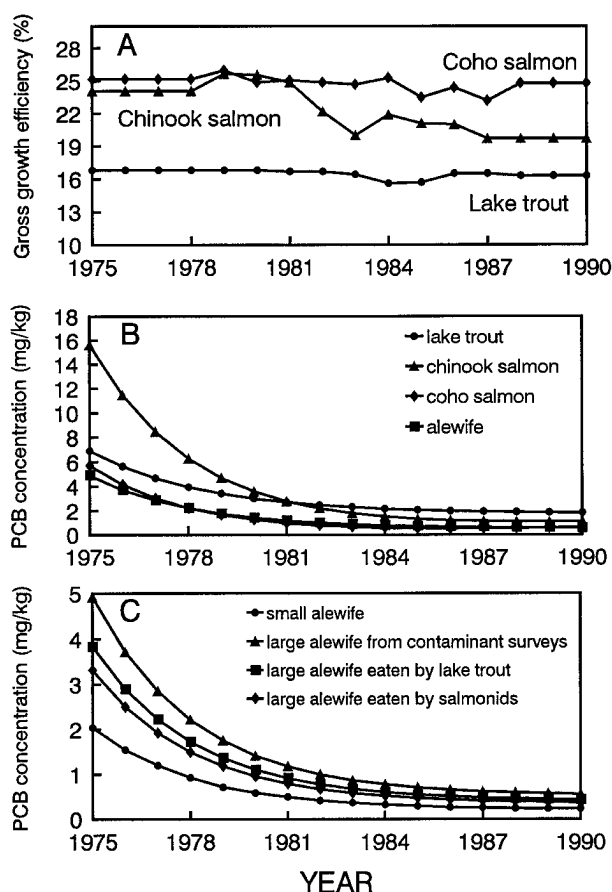


FIGURE 2. Salmonid gross growth efficiencies (A), PCB concentrations (B), and PCB concentrations of different size alewives from Lake Michigan between 1975 and 1990. Growth efficiencies from 1978 to 1988 are from ref 13. We assumed that growth efficiencies for 1975–1977 and 1989–1990 were identical to 1978 and 1988 values, respectively. Median values for the salmonids and alewife represent lines of best fit (24) from contaminant survey data. PCB concentrations of alewife eaten by the salmonid species were corrected for size based on diet information according to ref 13. Small alewives were assumed to be 5.9 g (13), and their PCB concentrations were determined with the alewife model of ref 8.

existed to estimate PCB concentration trends for the salmonids and alewife, but sample sizes for smelt ($n = 14$) and sculpin ($n = 4$) were too small. We examined three scenarios of forage fish contaminant concentration trends by assuming that (1) only alewife PCB concentrations declined following 1975 but smelt and sculpin remained constant [as has been observed in Lake Ontario (18)]; (2) alewife and smelt but not sculpin concentrations declined; and (3) concentrations of each forage species declined over the 15-year period. Smelt and sculpin concentration trends were estimated with an averaged relationship for alewife and bloater chub (*C. hoyi*) as a first approximation to a generalized prey fish model:

$$\text{PCB} = 5.16e^{-0.26t} + 0.50 \quad (5)$$

where PCB is the smelt or sculpin concentration (mg kg^{-1}) and t is the number of years since 1974. The smelt and sculpin data are primarily from the mid-1980s. We adjusted the asymptote by -0.44 (smelt) and -0.04 (sculpin) to force the estimated median values of eq 5 through the median smelt and sculpin values in 1985. *Mysis relicta* and *Diporeia hoyi* PCB concentration estimates over the time series (8) were averaged to provide a single invertebrate PCB concentration estimate for each year.

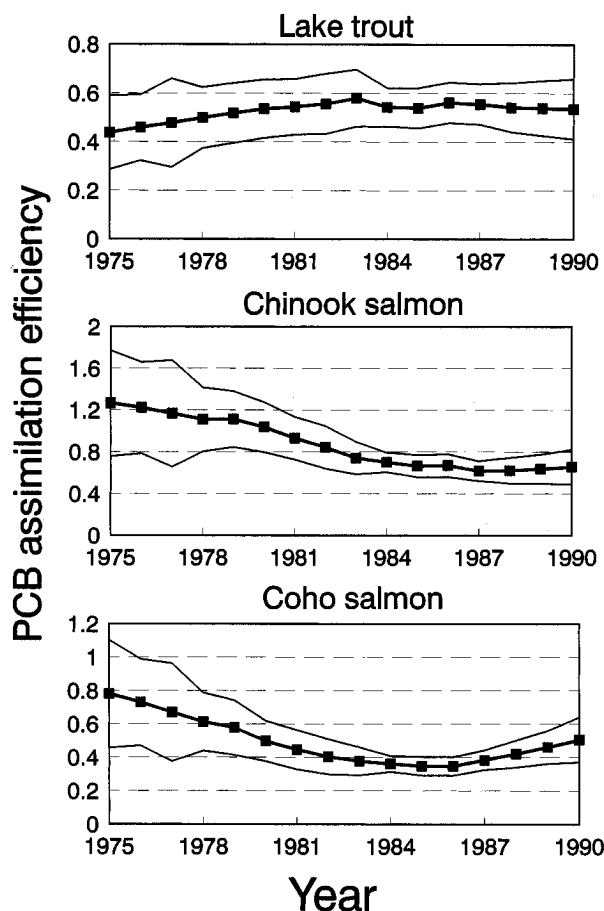


FIGURE 3. Net trophic transfer efficiencies (center line, solid squares) for three Lake Michigan salmonid species calculated from median salmonid PCB concentrations, size corrected alewife PCB concentrations, and salmonid α_j . The confidence bands about the center line represent ± 2 standard errors, calculated with a first-order error propagation.

We used α_j , diet composition and consumption estimates, and PCB concentrations of the diet items to solve for γ_j for each salmonid according to eq 4. We estimated uncertainty in γ_j with a first-order error propagation procedure (19).

Results

If $\gamma_j = \alpha_j$, a predator will incorporate prey PCBs and mass at the same rate and will have the same PCB concentration as its prey. For example, at $\gamma_j = 0.2$ and 0.4 and $\alpha_j = 0.2$ and 0.4 , predator PCB concentration is the same 1.0 mg kg^{-1} as its prey (Figure 1). If $\gamma_j > \alpha_j$, then a predator will bioaccumulate PCBs.

Estimated PCB trophic transfer efficiencies were not constant over time (Figure 3). The γ_j of lake trout increased, and those of chinook and coho salmon decreased through the 1970s to the early 1980s. From the early 1980s to 1990, γ_j has been nearly constant at ~ 0.55 for lake trout and ~ 0.65 for chinook salmon. γ_j for coho salmon increased slightly from the mid-1980s to 1990 where γ_j is estimated to be ~ 0.5 . Our first-order error propagation analysis resulted in relatively high uncertainty for γ_j values early in the time series, particularly for chinook and coho salmon. Calculated chinook salmon γ_j and the upper two standard errors for coho salmon exceed 1 in the 1970s, but the lower two standard errors for both species are well below 1 during this same period.

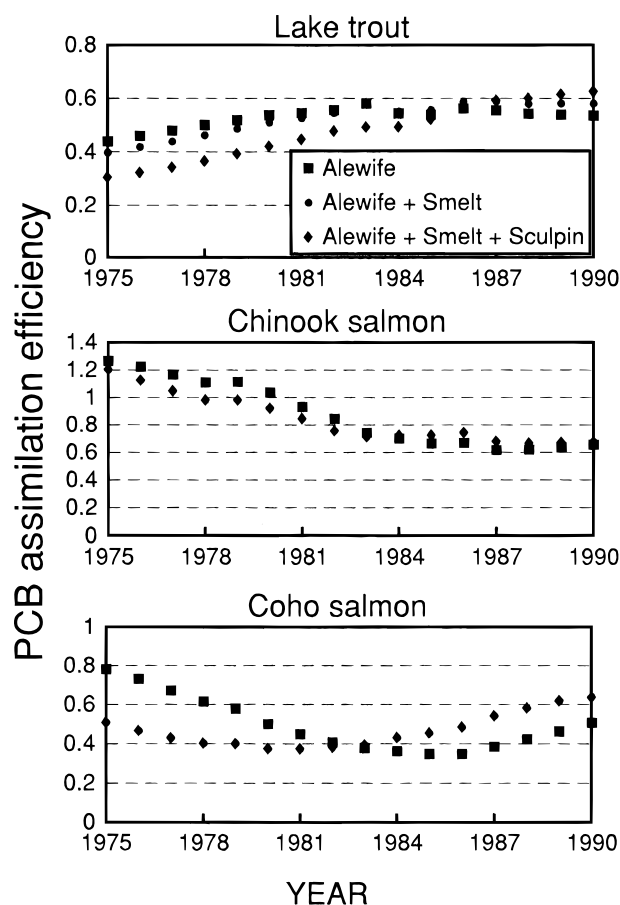


FIGURE 4. Net trophic transfer efficiencies for three Lake Michigan salmonid species for three scenarios of prey PCB concentrations. All scenarios include a PCB concentration trend in alewife adjusted for size selectivity of the salmonids. Alewife includes only the alewife concentration trend. Alewife and smelt adds a decreasing smelt concentration trend. Alewife, smelt, and sculpin adds an additional decreasing concentration trend for sculpin.

Adding decreasing smelt contaminant concentrations to the alewife PCB concentrations further decreased the calculated γ_j (Figure 4). Sculpins are important prey of young lake trout (17, 20). The addition of decreasing sculpin PCB concentrations to the decreasing alewife and smelt PCB concentrations further reduced the calculated lake trout γ_j . Chinook and coho salmon have virtually no sculpin in their diets; therefore, there was no difference when the decreasing sculpin PCB concentrations were added to the alewife and smelt concentration trends for these species.

Discussion

The rate at which PCBs bioaccumulate is a function of the ratio of the net PCB transfer efficiency (γ_j) to the gross assimilation efficiency (α_j) of a predator. The relationship between α_j and predator PCB concentration is nonlinear, as predators become proportionately more contaminated at low α_j than at high α_j for a given γ_j and food concentration (Figure 1). The observation that lake trout are more contaminated than other salmonids of the same size results from lake trout having a relatively low α_j (~ 0.17) when compared to that of the salmonids ($\alpha_j \sim 0.23$).

For each salmonid species, γ_j was relatively constant after the early 1980s despite substantial shifts in diet composition for coho and chinook salmon and a 20% drop in chinook salmon α_j . One of the important findings of this analysis is that γ_j appears to be relatively stable despite

changes in available forage, diet composition, and α_j since the mid-1980s in Lake Michigan.

The patterns of change in γ_j during the 1970s carry with them high uncertainty, and we believe do not reflect an ecologically significant trend. The error in the γ_j estimates is primarily a reflection of the relatively large variability in [PCB] of the salmonids and alewife, particularly early in the time series (15). If there was significant uptake of PCBs across gill membranes, γ_j could appear high. Most available evidence however suggests that dietary exposure is the principal route of PCB accumulation in these fish (1–4), and laboratory studies have demonstrated that gill exchange for salmonids is less than 5% of the total uptake (21, 22). Therefore, the high values for chinook salmon and the decreasing pattern of γ_j for chinook and coho salmon in the 1970s most likely only reflect the large variability in the predator and prey PCB concentrations.

The calculated γ_j of ~ 0.55 for lake trout (1983–1990) is about 25–30% lower than PCB assimilation efficiencies typically estimated from laboratory studies (3, 10, 23). It is important to recognize however that our values are *net* trophic transfer efficiencies that consider final concentrations after all depuration and transformation processes have occurred, while previous studies have reported *gross* assimilation efficiencies that do not include loss terms. Therefore, we would expect our calculated transfer efficiencies to be lower than assimilation efficiencies typically reported in the literature. Different PCB congeners are expected to have different assimilation efficiencies. We have calculated γ_j for the entire suite of PCB congeners present in Lake Michigan salmonids; this may account for some of the disparity between our field-derived estimates and laboratory results typically determined for individual Aroclors (e.g., ref 24). Congener-specific determinations may be beneficial for detailed analyses of PCB fluxes but do not exist for salmonids and their prey for the time period we have examined.

The contaminant survey data are salmonid PCB concentrations in fillets. If whole body concentrations are greater than or less than fillet concentrations, we would underestimate or overestimate, respectively, γ_j . Within-species relationships between PCB concentration and lipid content for the salmonid species included in the DNR surveys that we used are exceedingly weak to nonexistent (25). Thus, lipid-normalizing the concentrations and converting fillet to whole body concentrations with lipid ratios is not possible at present (26). Equation 4 illustrates that γ_j is a simple linear function of predator PCB concentration, α_j , and prey PCB concentrations. Therefore, when fillet to whole body conversions become available, our γ_j estimates could be adjusted to reflect this conversion.

The strength of our estimates is that they are derived from field data. Caution has been suggested when extrapolating laboratory measurement to field populations (27). Our field-derived calculations are for animals growing under natural conditions whereas laboratory studies typically report assimilation efficiencies for the flux of contaminants into clean animals. Bias in γ_j could result from bias in prey and/or predator PCB concentrations. Low trawl efficiency for small alewives makes their effective sampling difficult and would cause them to be underrepresented in contaminant surveys. To explore the possible effect of bias in prey and predator PCB concentrations, we performed a sensitivity analysis on data for 1975 and 1990. We independently varied prey and predator PCB concentrations

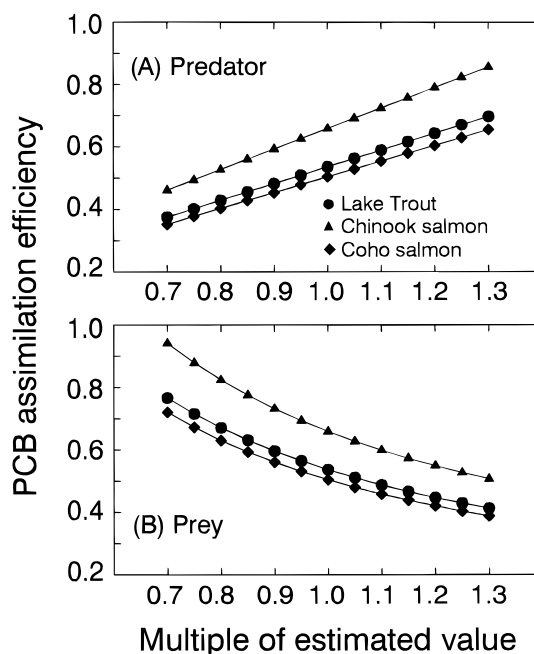


FIGURE 5. Sensitivity analysis of the effect of bias in predator (A) and prey (B) PCB concentrations on the calculated predator γ_j . The estimated value of 1.0 is the median predator PCB concentration from contaminant surveys in 1990. Prey concentrations include decreasing alewife concentrations (Figure 2C) and constant values for smelt and sculpin.

by factors that ranged from 0.7 to 1.3 times the nominal values used in our analyses. For 1990, data bias in predator PCB concentrations led to a change of equal magnitude in predator γ_j (Figure 5A). Underestimating prey PCB concentrations by 30% would result in a 43% increase in γ_j (Figure 5B). Similarly, a 30% overestimate in prey PCB concentration would result in a 23% decrease in γ_j . The same degree of bias applied to 1975 data produced the same relative change on predator γ_j . This result emphasizes the importance of knowing diet selectivity by the piscivorous fishes, coupled with reliable estimates of prey PCB concentrations. Salmonids selectively seek the largest alewives available (13), but the degree to which trawl and salmonid selectivity match is unknown. Nevertheless, γ_j since 1983 are virtually constant and are among the best estimates available for empirically derived values.

Models of contaminant uptake from food typically assume that a fish assimilates a constant fraction of the contaminant it ingests (3, 7, 28–30). It seems logical that γ_j would be more constant than α_j . Variation in feeding rate, respiration, and growth rate cause variation in α_j (31). Assimilated carbon can be lost, but PCBs rarely leave a fish once assimilated and incorporated into tissue. Our results show that relative to PCB concentrations, γ_j is fairly constant. Therefore, changes in salmonid PCB concentration are mainly driven by prey PCB concentrations, and it is appropriate to use a constant γ_j for studies of the bioaccumulation of total PCBs in fish.

Acknowledgments

We thank S. R. Carpenter, C. A. Stow, and two anonymous reviewers for their constructive comments on earlier drafts of the manuscript. L.J.J. was funded by a Natural Engineering Research Council of Canada (NSERC) postdoctoral fellowship and by the University of Wisconsin Sea Grant Institute under grants from the National Sea Grant College

Program and from the State of Wisconsin, Federal Grant NA90AA-D-SG469, project R/MW-41. D.E.S. was funded in part by an NSERC graduate fellowship.

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Received for review June 29, 1995. Revised manuscript received March 4, 1996. Accepted March 7, 1996.[®]

ES950464A

[®] Abstract published in *Advance ACS Abstracts*, May 1, 1996.