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AN AGE DEPENDENT MODEL OF PCB IN A LAKE MICHIGAN FOOD CHAIN

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It was estimated that a criterion specifying that PCB concentrations of all ages of lake trout be at or below 5 μg/g (wet weight) in the edible portion would require that dissolved PCB concentrations be reduced to somewhere between 0.5 and 2.5 μg/l. The range reflects uncertainty in the PCB assimilation efficiency of the species and the dissolved PCB concentration.

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CONTENTS

		Page
Abstract		iv
Figures		v
Tables		ix
Acknowledgements		x
Section 1.	INTRODUCTION	1
Section 2.	CONCLUSIONS	8
Section 3.	RECOMMENDATIONS	10
Section 4.	COMPILATION AND ANALYSIS OF PCB DATA	12
Section 5.	AGE DEPENDENT MODEL: THEORY	27
Section 6.	ESTIMATION OF MODEL FOOD CHAIN INTERACTIONS AND PARAMETERS	46
Section 7.	MODEL CALIBRATION	73
Section 8.	MODEL SENSITIVITY	88
Section 9.	LAKE TROUT RESPONSE DUE TO REDUCED WATER CONCENTRATION	99
References		106

ABSTRACT

An age-dependent food chain model that considers species bioenergetics and toxicant exposure through water and food is developed. It is successfully applied to PCB contamination of the Lake Michigan lake trout food chain represented by phytoplankton, Mysis, alewife, and lake trout. The model indicates that for the top predator lake trout, PCB exposure through the food chain can account for greater than 99 percent of the observed body burden. A simple steady-state computation indicates that ratios of chemical concentration in predators to that in prey in feeding experiments may be as low as 0.2 and still result in significant food chain transfer.

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FIGURES

Number		Page
1.	Lake Michigan	2
2.	Historical trend of PCB concentration in small fish of Lake Michigan (replotted from Neidermeyer and Hickey, 1976; also data from USF & WLS).	3
3.	PCB concentrations in lake trout (whole fish), 1971-1979. ( ) = No. samples, ⊡ = Arith. mean, Δ = median, Range = ± 1 Std. deviation.	5
4.	Sampling regions for 1971 fish contaminant survey by Veith (1973, 1975).	15
5.	PCB concentrations (whole fish) as a function of weight, 1971 Data from Veith (1975).	16
6.	PCB concentrations (whole fish) as a function of % lipid. 1971 Data from Veith (1975). (Numbers in parentheses are fish species, see Figure 5 for identification).	17
7.	PCB concentrations in coho salmon, 1971-1976.	19
8.	PCB concentration in alewife as a function of age, 1971.	20
9.	PCB concentration in lake trout as a function of age class for 1971 and 1975.	21
10.	Variation of PCB for lake trout from 1971-79 for age classes 2-6 years.	23
11.	Variation of PCB for lake trout from 1971-79 for age classes 7-10 years.	24
12.	Relationships between PCB and % lipid for lake trout from different sets.	25
13.	Schematic of compartment definition for a) age dependent model, b) simplified steady state model.	33
14.	Illustration of meaning of $w_i(t)$ and $v_i(t)$ showing as an example a 0-1 year old and 1-2 year old alewife.	36

FIGURES (continued)

Number		Page
15.	Illustration of equilibrium conditions for concentration of contaminant in organism. (a) Exponential increase in organism weight, Eq. (11); (b) increase in contaminant body burden, Eq. (20); (c) increase in contaminant concentration to equilibrium (steady state) level, Eq. (21).	40
16.	Effect of food chain transfer ratio, f, on ratio of chemical concentration in top predator from water plus food to that from water only. Equilibrium conditions, constant BCF and f ratio along food chain.	45
17.	Percent of total *Mysis* captured in the water column and found in alewife stomach in relation to *Mysis* length for two sampling dates. (From Janssen and Brandt, 1980).	49
18.	Feeding interactions specified in the model for alewife and *Mysis*.	50
19.	Observed incidence of fish and invertebrates in lake trout stomachs in relation to lake trout length and computed age.	53
20.	Distribution, by size, of alewives found in various size classes of lake trout.	54
21.	Feeding interactions specified in the model for lake trout.	55
22.	Wet weight of *Mysis relicta* in relation to age as calculated from a length-age and weight-length relationship (solid line) and as used in the model (dashed line).	58
23.	Observed wet weight in relation to alewife age as compared to functional relationship used in the model.	59
24.	Observed wet weight in relation to lake trout age as compared to functional relationship used in the model.	61
25.	Comparison of lake trout wet weight-age data from Carlander (1969) and Hess and Muench (1980). Lines indicate rates used in the model.	62
26.	Calculated respiration rates used in the model for *Mysis*, alewife, and lake trout in relation to wet weight as compared to a general relationship based on data for many species of freshwater fish.	67

FIGURES (continued)

Number		Page
27.	Comparison between observed and computed ingestion rates in relation to wet weight.	68
28.	Comparison between observed and computed excretion rates of PCB in relation to wet weight.	71
29.	Comparison of observed PCB concentrations in alewife and lake trout with calculated concentrations from basic model.	74
30.	Average PCB concentration in each of the 4 species included in the food chain model. Values indicated are arithmetic average over all age classes.	77
31.	Computed PCB concentration factor for lake trout due to PCB uptake from water and from water and food.	79
32.	Computed PCB concentration in lake trout due to PCB uptake from water and from water and food.	80
33.	Comparison of observed PCB concentrations in alewife and lake trout with calculated concentration from lipid-dependent model.	83
34.	Comparison of food chain model and octanol/water partitioning to lipid tissue of lake trout.	85
35.	Computed PCB concentration in alewife from octanol/water partitioning calculation.	86
36.	Computed lake trout PCB concentrations at PCB assimilation efficiencies of 0.65, 0.8, and 0.95.	89
37.	Computed lake trout PCB concentrations at the calibrated excretion rate and zero excretion rate.	91
38.	Computed alewife and lake trout PCB concentrations at dissolved PCB concentrations of 2 μg/ℓ, 5 μg/ℓ and 10 μg/ℓ.	93
39.	Computed alewife and lake trout PCB concentrations for models recalibrated to dissolved PCB concentrations of 2 μg/ℓ and 10 μg/ℓ by alteration of PCB assimilation efficiencies.	94
40.	Computed lake trout PCB concentrations for three trophic level and four trophic level food chain structures.	96

FIGURES (continued)

Number		Page
41.	Computed and observed lake trout PCB concentrations in relation to wet weight for native trout growth rates (#1) and stocked trout growth rates (#2).	98
42.	Projected response of different age lake trout to a five-fold decrease in dissolved PCB concentration - growth rate #1 for pre-stocked period.	100
43.	Relationships between equilibrium PCB concentration in 11 year old lake trout and dissolved PCB water concentrations, growth rate #1 = pre-stocked period; growth rate #2 = stocked period.	101
44.	Calculated relationship between age class of lake trout and required dissolved PCB water concentration to meet whole fish concentration of 5-10 μg/g(w). Growth rate #2 for stocked lake trout.	104

TABLES

Number		Page
1.	PCB Concentration (Whole Fish) in Lake Trout of Lake Michigan	4
2.	PCB in Surficial (0-3 cm) Lake Michigan Sediments (1975)	13
3.	Fraction of Total Alewife Consumption on Plankton and Each *Mysis* Age Class	51
4.	Fraction of Total Lake Trout Consumption on Each *Mysis* and Alewife Age Class	56
5.	Parameter Values Used in Calibration of Model	75
6.	Lake Trout Lipid Content Data Compiled in Relation to Age Class	82

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SECTION 1

INTRODUCTION

BACKGROUND

The fishes of Lake Michigan (Figure 1) have been a matter of study and concern for a great number of years (Grt. Lakes Basin Comm., 1975; Int. Joint Comm., 1978). The concern was directed toward maintaining a viable sport and commercial fishery in the face of increased fishing pressures and predation of the lake trout by the sea lamprey. Beginning about 10-15 years ago, concern was expressed over the concentration of potentially toxic substances in desirable sport and commercial fish. The elevated levels of DDT exceeded the guideline value of 5 μg/g(w) (g(w) = grams wet weight) and as a result DDT was banned in 1972. Similarly, the elevated concentrations of polychlorinated biphenyls (PCB) in the fishes of Lake Michigan, particularly the lake trout (Salvelinus namaycush) were a matter of significant concern. Concentrations of PCB in the lake trout in 1971, for example, ranged from 5-20 μg/g(w), substantially above the U.S. Food and Drug Administration (FDA) guidelines of 5 μg/g(w) in the edible portion of fish. Several fishing bans have been issued concerning the consumption of fish contaminated with PCBs. In 1978 commercial fishermen harvested more than 43 million pounds of alewife for fish meal. But in 1980, the catch dropped dramatically because of recognition that the PCB concentration of alewives exceeded the FDA guideline of 2 μg/g(w) for fish meal (Anon, 1981). This particular chemical had been in use within the Lake Michigan drainage basin for more than 20 years when it was restricted in sales (except for minor uses) in 1971. A full ban was instituted several years later.

Neidermeyer and Hickey (1976) analyzed the PCB concentration in museum specimens of small fish from Lake Michigan and observed a marked increase beginning in the 1950s. Their results are shown in Figure 2 together with additional data on PCB concentration from the U.S. Fish and Wildlife Service (USF&WLS). Concentration appears to peak in the early 1970s. The histori-

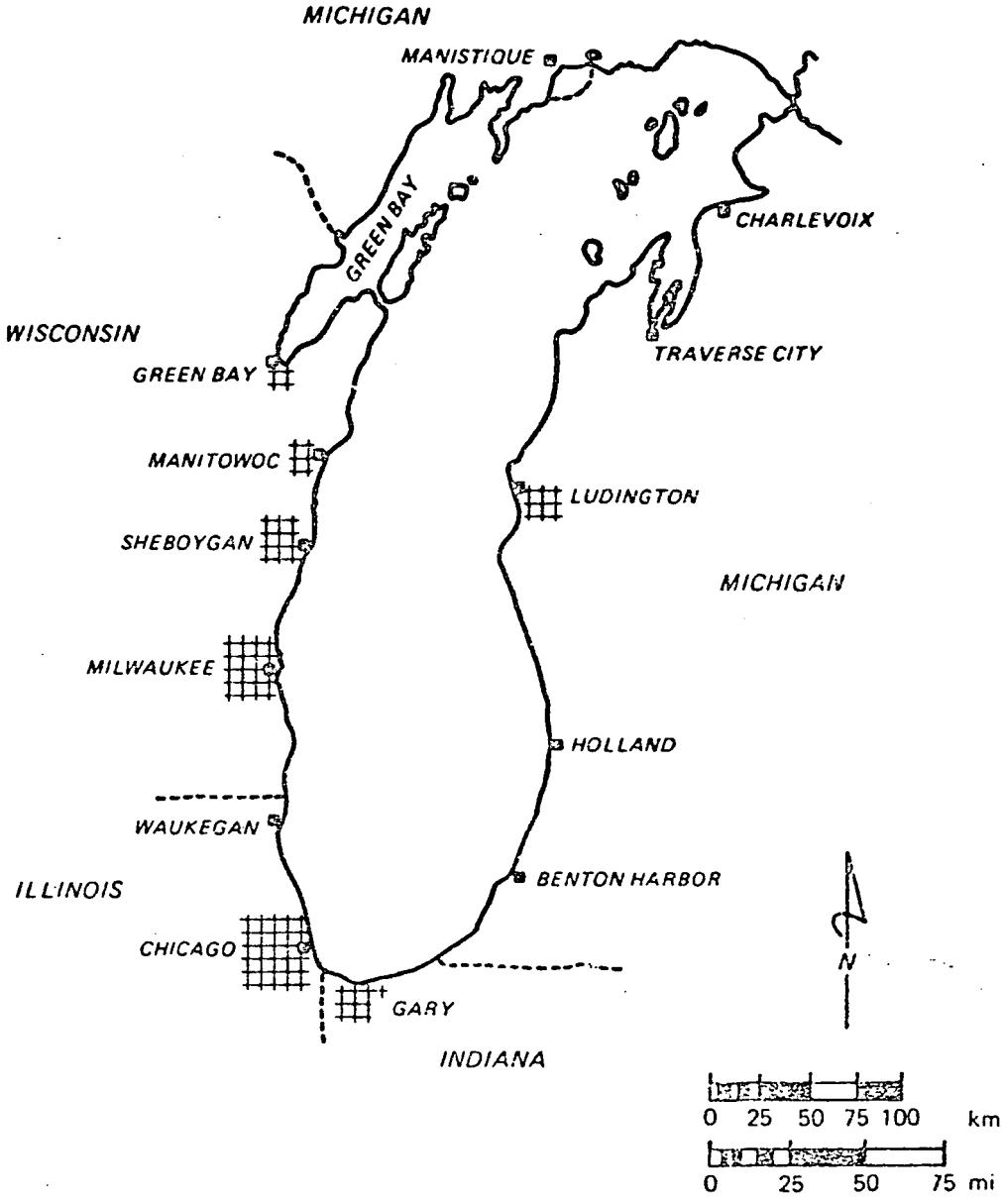

Figure 1. Lake Michigan

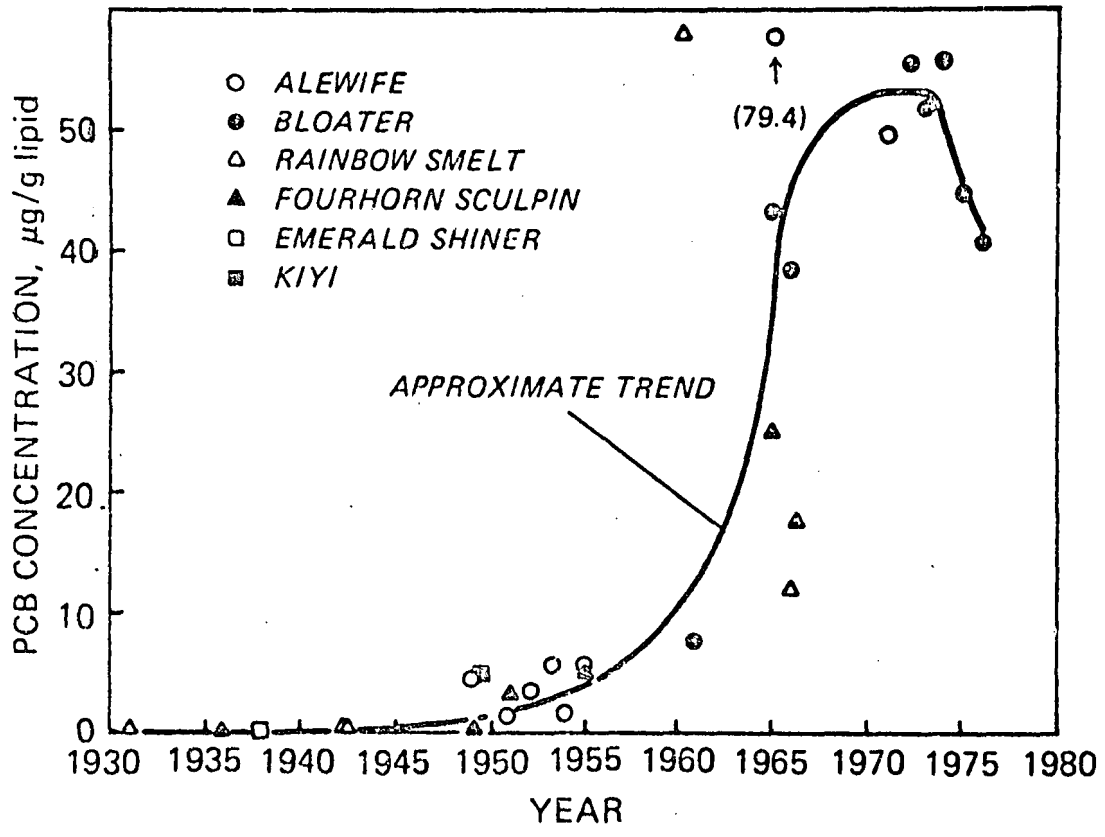

Figure 2. Historical trend of PCB concentration in small fish of Lake Michigan (replotted from Neidermeyer and Hickey, 1976; also data from USF & WLS).

cal record for PCBs in the top predators of the lake such as the lake trout is less extensive. Figure 3 and Table 1 show the general trend in PCB for the trout (whole fish) for the period 1971-1979 from several sources (Veith, 1975; EPA STORET; USF & WLS; WAPORA, 1981). As noted previously the concentration of PCB exceeds a guideline level of 5 μg/g(w) although this is for the edible portion of the fish. The data indicate that for the years 1971-77, the mean and median were greater than 13 μg/g(w). An apparent decline began in 1978 where mean and median values dropped to 5.8 and 3.3 μg/g(w) respectively. The 1979 data however indicate mean and median values of 10 and 8 μg/g(w). Across all age classes therefore there appears to have been some decline in the PCB concentration in the lake trout although the decline is not dramatic and a longer term data base is necessary to ascertain the actual trend. This is especially true when the variation in the data as indicated by the standard deviation of Figure 3 is recognized. Additional data collected through the Great Lakes Environmental Contamination Survey (GLECS) in 1975 are for filets only and averaged about 4.5 μg/g(w).

TABLE 1. PCB CONCENTRATIONS (WHOLE FISH) IN LAKE TROUT OF LAKE MICHIGAN[1]

YEAR	MEAN (μg/g(w)	STAND. DEV.	MEDIAN (μg/g(w)	NO. OF FISH
1971	13.7	4.5	13.4	139
1972	17.9	9.2	15.4	19
1973	18.0	1.0	17.8	30
1974	17.7	7.9	20.7	48
1975	11.7	6.5	7.3	48
1976	17.4	17.1	14.6	40
1977	13.3	4.4	14.8	105
1978	5.8	7.8	3.3	42
1979	10.1	7.3	8.0	182

(1) Compiled from Veith (1975), EPA STORET, USF & WLS, and Wapora (1981)

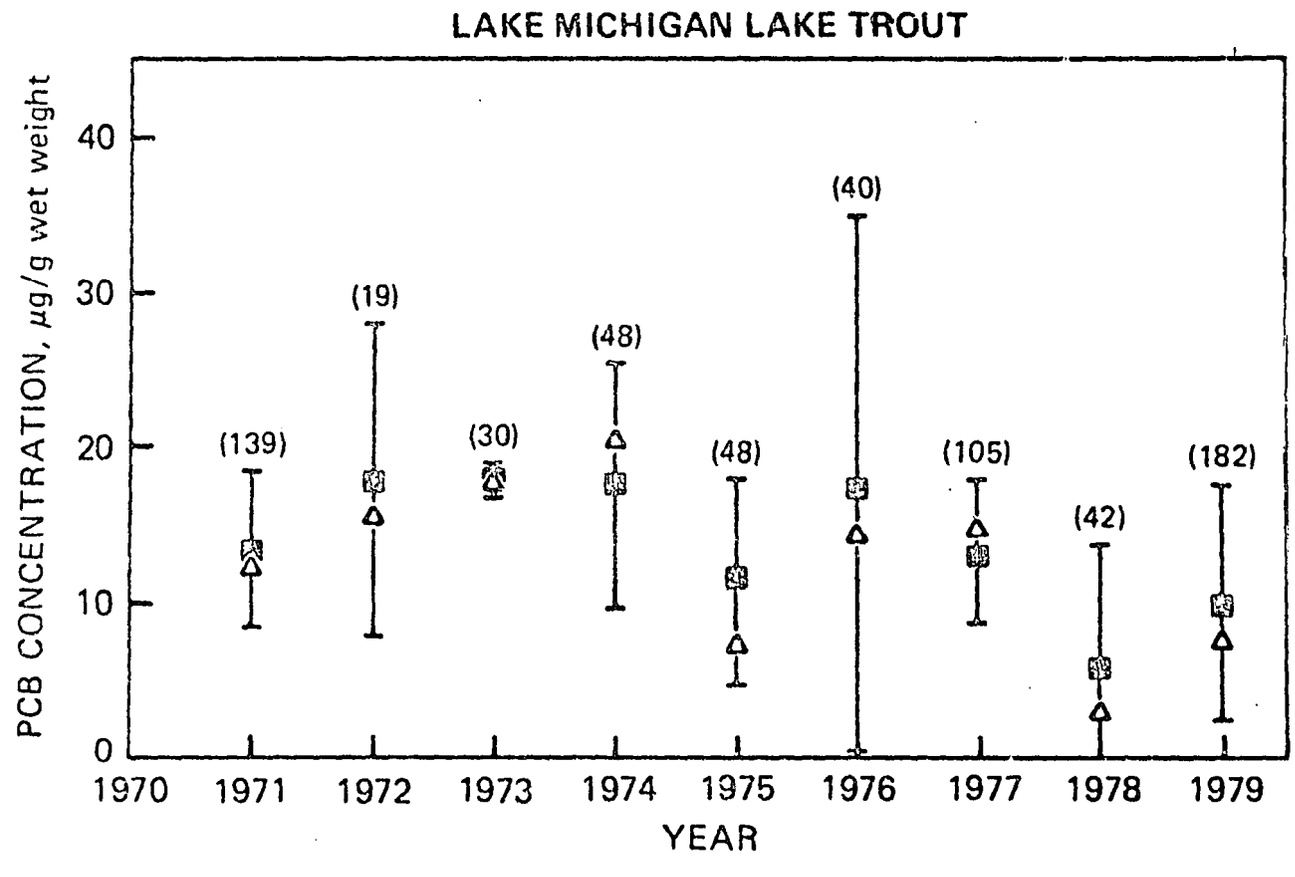

Figure 3. PCB concentrations in lake trout (whole fish), 1971-1979. ( ) = No. samples, ▣ = Arith. mean, △ = median, Range = ± 1 Std. deviation.

THE ISSUE OF WATER UPTAKE VS. FOOD CHAIN TRANSFER

These data on PCBs in the Lake Michigan fish indicate a substantial concentration of this substance in the lake trout. In order to properly understand the mechanisms that give rise to the observed data, it is necessary to analyze the data through use of a model of the principal phenomena of chemical uptake and transfer. These mechanisms include two principal routes:

1) Uptake of PCB directly from water
2) Accumulation of PCB through consumption of contaminated food

The significance of the food chain route, i.e. the degree to which a chemical is accumulated in an organism by predation has been the subject of considerable debate and confusion. For example, Scura and Theilacher (1977) in evaluating chlorinated hydrocarbon (CHC) data on a laboratory food chain concluded that "CHC accumulation is not a food chain phenomenon but rather the result of direct partitioning of the compounds between the seawater and the test organisms." On the other hand, Weininger (1978) using a model of PCB accumulation in the lake trout of Lake Michigan concluded that "Direct uptake of PCBs can only account for 2-3% of total observed PCB accumulation by adult lake trout although this route is more important for juveniles." Also Thomann (1981), in analyzing PCB data from a variety of water bodies, concluded that for top predators in the natural environment, the data suggest that the PCB concentration is about one order of magnitude greater than would result from uptake from the water alone.

It has also been suggested implicitly (Neeley et al., 1974; Kenaga, 1980) and explicitly (Neeley, 1979) that the maximum environmental concentration in fish can be estimated without recourse to a food chain route. These approaches assume that uptake from water is the principal route and for lipid soluble compounds such as the PCBs, the concentration in fish is related directly to the octanol-water partition coefficient of the compound. It is assumed in these approaches then that a first approximation to expected levels of a chemical can be obtained either from simple partitioning concepts (Neeley et al., 1974; Kenaga, 1980) or from a simple model of direct uptake from the water (Neeley, 1979).

The issue of whether a simple calculation of uptake of a chemical directly from the water is sufficient relates to the degree to which such a calculation would actually reproduce observed field data for important species such as the lake trout. If such a calculation does account for the observed data in the field, then there is no need for a model that includes a food chain component. If a simple partitioning calculation fails to reproduce the observed data, then the principal feature of the food chain must be included.

OBJECTIVES OF RESEARCH

The principal objectives of this research therefore are to

1. develop an age-dependent food chain model of uptake and transfer of potentially toxic chemicals
2. determine the relative importance of water uptake and food chain routes of PCB in a Lake Michigan food chain with specific emphasis on lake trout
3. test the utility of simple partitioning approaches for PCB that do not include the food chain route
4. provide a preliminary projection of response in PCB concentration in the lake trout following a reduction in PCB water concentration.

Although the age-dependent model for transfer of chemicals in aquatic food chains is calibrated in this work with the PCB data of the Lake Michigan food chain including the lake trout, a subsidiary and equally important objective of this research is to provide a general framework that would be applicable in a variety of other problem contexts.

SECTION 2
CONCLUSIONS

The contamination of Lake Michigan alewife and lake trout by PCB can be adequately modeled using an age-dependent computation that considers species bioenergetics and uptake of PCB from water and food. The model successfully reproduces the age-dependent trends and magnitude of PCB contamination observed in 1971.

Both the model and the PCB data compiled for this study indicate that food chain transfer is a significant route of contamination. Data from thirteen species of fish suggest an increase in PCB concentration as one proceeds up the food chain to the top predators. Transfer of PCB through the food chain is the major contributor to calculated PCB concentrations, accounting for greater than 99% of the body burden in adult lake trout. A simple steady-state computation indicates that ratios of chemical concentration in predators to that in prey in feeding experiments may be as low as 0.2 and still result in appreciable food chain transfer.

An empirical relationship between lake trout excretion rate and lipid content significantly improved the lake trout calibration, suggesting that lipid tissue is an important factor in PCB dynamics.

A simple empirical correlation between octanol/water partitioning of PCB and partitioning between water and fish lipid tissue failed to reproduce the observed concentrations in alewife and lake trout. It is concluded that although this simple partitioning approach may be useful in assessing trends, it cannot estimate actual concentrations, especially in higher trophic level species, because it does not consider food chain transfer.

Projections of the response of the lake trout food chain to reduced water concentrations indicate that a period of about 5 years is needed to reduce whole body PCB concentrations in upper age class lake trout. In order to have the PCB concentrations of all age classes of lake trout at or below 5 $\mu g/g(w)$ in the edible portion, it is estimated that the dissolved water concentrations

would have to be between 0.5-2.5 ng/ℓ. The range results from the uncertainty of the parameter values in the model. These water concentrations represent a 75-95% reduction of apparent 1961-1971 water concentrations. Young age classes can generally be exposed to higher water PCB concentrations than older age classes without exceeding the objective of 5 μg/g(w). As a result, if water quality projections indicate a lower bound in the achievable PCB water concentrations, a size dependent fish consumption guideline can be developed.

SECTION 3

RECOMMENDATIONS

This analysis of the PCB contamination of a Lake Michigan food chain has illustrated gaps of knowledge that add uncertainty to the estimation of the effect of concentration reductions. These gaps are most significant in regard to the compound-related parameters needed by the model, i.e., assimilation efficiency and excretion rate. It is therefore recommended that experimental investigations be conducted to more accurately determine these parameters. Of significant value would be relationships, both within and across species, between these parameters and characteristics of the compound and species, e.g., octanol/water partition coefficient and % lipid.

Additional significant gaps of knowledge are the PCB concentrations of the invertebrate and plankton components of the food chain and accurate estimates of the water concentration.

Differences were found between growth rates of pre-stocked and stocked lake trout. The model was shown to be sensitive to these differences. It is recommended that lake trout growth rate be investigated to provide an accurate estimate that will decrease uncertainty in the model projections.

An empirical relationship between lake trout excretion rate and % lipid significantly improved the model calibration suggesting that lipid content is an important factor in accumulation. It is recommended that the model structure be modified to include a more fundamental description of lipid, possibly separating the species into lipid and non-lipid components.

The sediment of Lake Michigan contains a substantial quantity of PCB. Because the sediment responds more slowly to reductions in PCB loading than does the water column, it will have significant PCB concentrations even when water column concentrations decline to some "acceptable" level. A significant question, then, is the extent to which benthic fauna may transfer this sediment PCB to the pelagic food chain, thus mitigating the concentration reduction in that food chain. This question should be addressed by including a benthic component in the food chain.

The food chain model was calibrated to data collected in 1971. Data are also available through 1979. These data indicate a decline of lake trout PCB concentrations after 1975. A further calibration of the model, using these data, would increase confidence in its prediction capability.

It is also recommended that the model be applied to other chemicals for which a sufficient data base exists. This would test the applicability of the model as a general framework for assessing the response of the food chain to toxic substance exposure.

SECTION 4

COMPILATION AND ANALYSIS OF PCB DATA

WATER COLUMN

The estimation of water column concentrations of PCB in the open waters of Lake Michigan is difficult because levels during the early 1970s were apparently lower than the level of detection. Veith and Lee (1971) reported from work done in 1970 that the PCB concentration in Lake Michigan water was below the detection limit of 10 ng/ℓ (.01 μg/ℓ). Torrey (1976) and Konasewich et al. (1978) have summarized the available water data including the earlier determinations of Veith and Lee, and also indicated that open water concentrations were below 10 ng/ℓ. For some nearshore (.05 km) samples reported in 1972, the concentrations ranged from 12 ng/ℓ to 56 ng/ℓ. Values of 30 ng/ℓ and 40 ng/ℓ for open lake water reported by Murphy and Rzeszutko (1977) are believed to be too high (Veith, personal communication). Data obtained in Waukegan Harbor by several laboratories and compiled by Thomann and Kontaxis (1981) indicate PCB concentrations in the water column during 1979 ranging from 40-1000 ng/ℓ. Concentrations in the nearshore region (< 0.5 km) off Waukegan Harbor ranged from 1-10 ng/ℓ. One concludes from these data, along with Veith 1973, that the PCB concentration in the open lake have been less than about 10 ng/ℓ during the 1970s and in the nearshore area (< 0.5 km) have been in the range of 10-100 ng/ℓ. Because of the absence of quantitative values of PCB water concentrations for the open lake, it is not possible to describe any trends of PCB in the water column.

SEDIMENT

The most comprehensive analysis of the PCB concentrations of the sediments of Lake Michigan has been reported by Frank et al. (1981). This 1975 survey analyzed 286 surficial sediment samples (0-3 cm) taken from throughout the lake. Table 2 summarizes some of their results. The standard deviation of 15.7 ng/g(d) (g(d) = grams dry weight) for the average of about 10 ng/g(d) indicates the non uniform spatial distribution of PCB in the sediments. The distribution of concentrations were: 20% at < 2 ng/g(d); 68% at concentra-

tions between 2 and 20 ng/g(d) and 12% greater than 20 ng/g(d). The highest value measured was 190 ng/g(d).

TABLE 2. PCB IN SURFICIAL (0-3 cm) LAKE MICHIGAN SEDIMENTS (1975) (From Frank et al., 1981)

	PCB (ng/g(d))	
	Mean	Std. Dev.
Whole Lake	9.7	15.7
Non-Depositional Zones	6.3	8.1
Depositional Zones	17.3	23.9

Harbors throughout Lake Michigan have significantly greater sediment PCB concentrations than the main lake. As summarized in Konasewich et al. (1978), Milwaukee Harbor measurements indicated a concentration of 6420 μg/g(d) while Sheboygan Harbor ranged from .06 to 0.32 μg/g(d). Thomann and Kontaxis (1981) in a summary of sediment PCB data for Waukegan Harbor collected by several laboratories, indicate a range in the harbor spanning five orders of magnitude from 0.1 to 10,000 μg/g(d).

These data indicate that the sediment of Lake Michigan contain a substantial reservoir of PCB. The distribution however is not uniform and reflects past loading of PCB and subsequent distribution throughout the lake with concentration in the principal sediment depositional zones.

AQUATIC ECOSYSTEM

Lower and Intermediate Levels

No data have been obtained on the PCB concentration in the phytoplankton or the smaller zooplankton. Veith (1973) as reported by Torrey (1976) analyzed the PCB concentration of the amphipod Pontoporeia in the spring of 1972 and obtained a range from .06 μg/g(w) at 30.5 m to 0.89 μg/g(w) at 8.2 m. An apparent inverse correlation of concentration with distance from shore (increasing depth) was observed. In the vicinity of Waukegan, the concentration decreased from 0.89 μg/g at 8.2 m to 0.45 μg/g at 30.5 m to 0.34 μg/g at 45.7 m.

Fish PCB Data Base-Overview and Summary

The PCB data for the fishes of Lake Michigan were compiled from five principal sources: a) Veith (1973, 1975), b) USF & WLS, and a Great Lakes Environmental Contamination Survey (GLECS) c) the Storage and Retrieval (STORET) system of EPA and d) data for 1977-79 obtained from state agencies and tabulated by Wapora (1981). In addition, literature data were evaluated from sporadic sampling conducted by individual investigators.

The data reported by Veith (1973, 1975) were obtained in 1971, generally during the fall at the areas shown in Figure 4. Thirteen species of fish were taken from the fourteen regions and analyzed for PCBs and analogs of DDT. Additional parameters included the weight, length and % fat of the fish. Figure 5 and Figure 6 display the summary PCB data of Veith for the thirteen species as a function of mean species weight and as a function of mean % lipid. There is an apparent relationship between weight and concentration (with the exception of the principal detrital feeders such as carp and the suckers). The apparent relationship with weight may reflect differing positions in the food chain. The highest concentration reported was for the lake trout (#8). The redhorse sucker (#5), white sucker (#6), whitefish (#7) and carp (#9) all have relatively low concentrations of PCB. This undoubtedly reflects the simple food chain structure of these primarily herbivorous and bottom feeding species. Figure 6 shows no apparent relationship between the PCB concentration and the mean species % lipid. This probably reflects a confounding of lipid concentration and trophic level position. Thus, although the bloater (#4) had the highest mean species % lipid (20 $\pm$ 5.9%), the mean PCB concentration is only 6 μg/g(w).

The available PCB data from the surveys of the USF & WLS extended from 1972-1976. Bloater chubs and lake trout were obtained off Saugatuck, Michigan and coho salmon from east-central Lake Michigan in the area of the Platte River. Bloater PCB concentration varied from about 5.7 μg/g(w) in 1972 to about 3.1 μg/g(w) in 1978 with an apparent downward trend (see also Figure 2).

The available GLECS data set (Michigan, 1975) is for surveys and analyses of PCB conducted in 1975 by the Michigan Department of Agriculture, Michigan Department of Natural Resources, Michigan Department of Public Health and U.S.

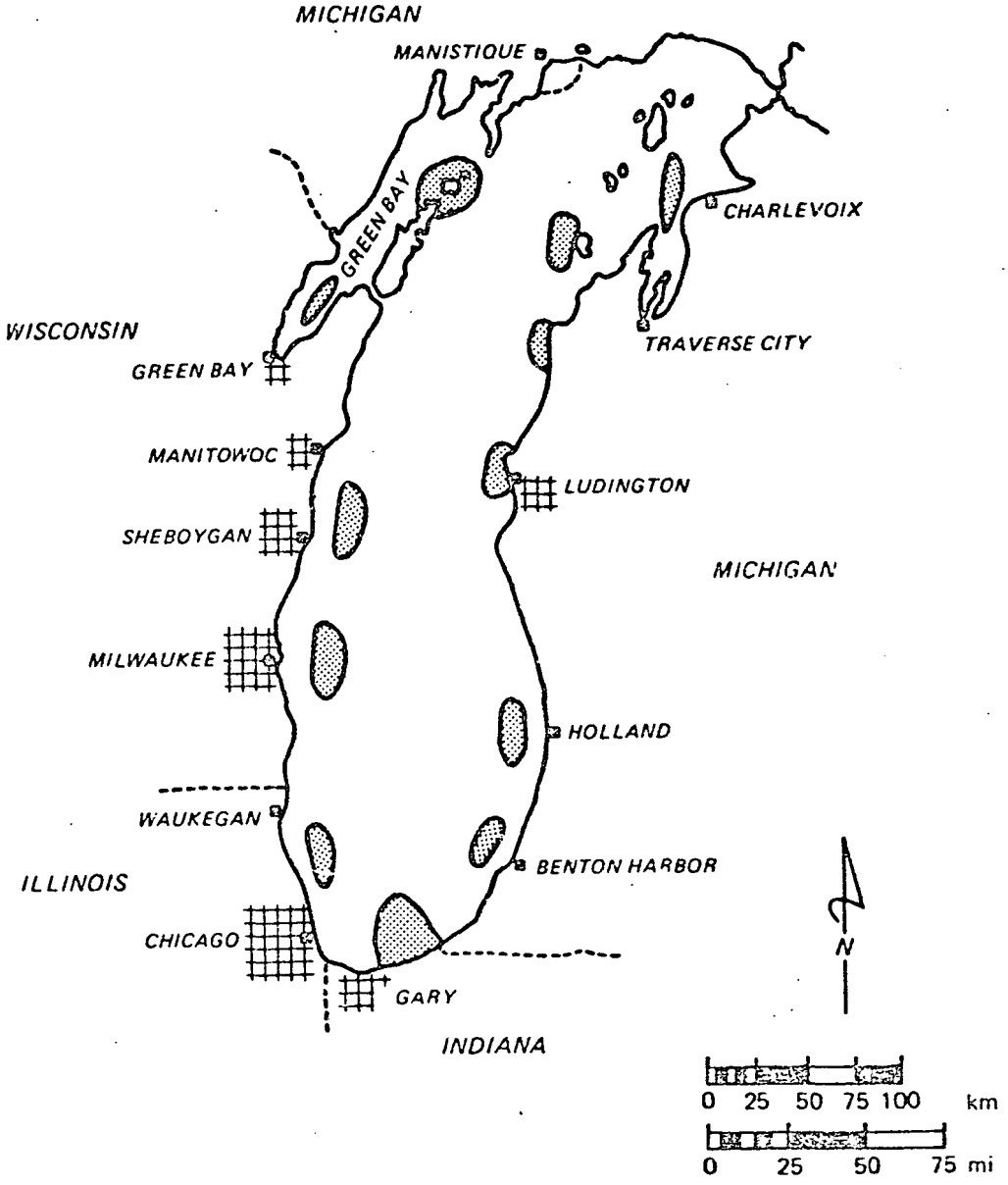

Figure 4. Sampling regions for 1971 fish contaminant survey by Veith (1973, 1975).

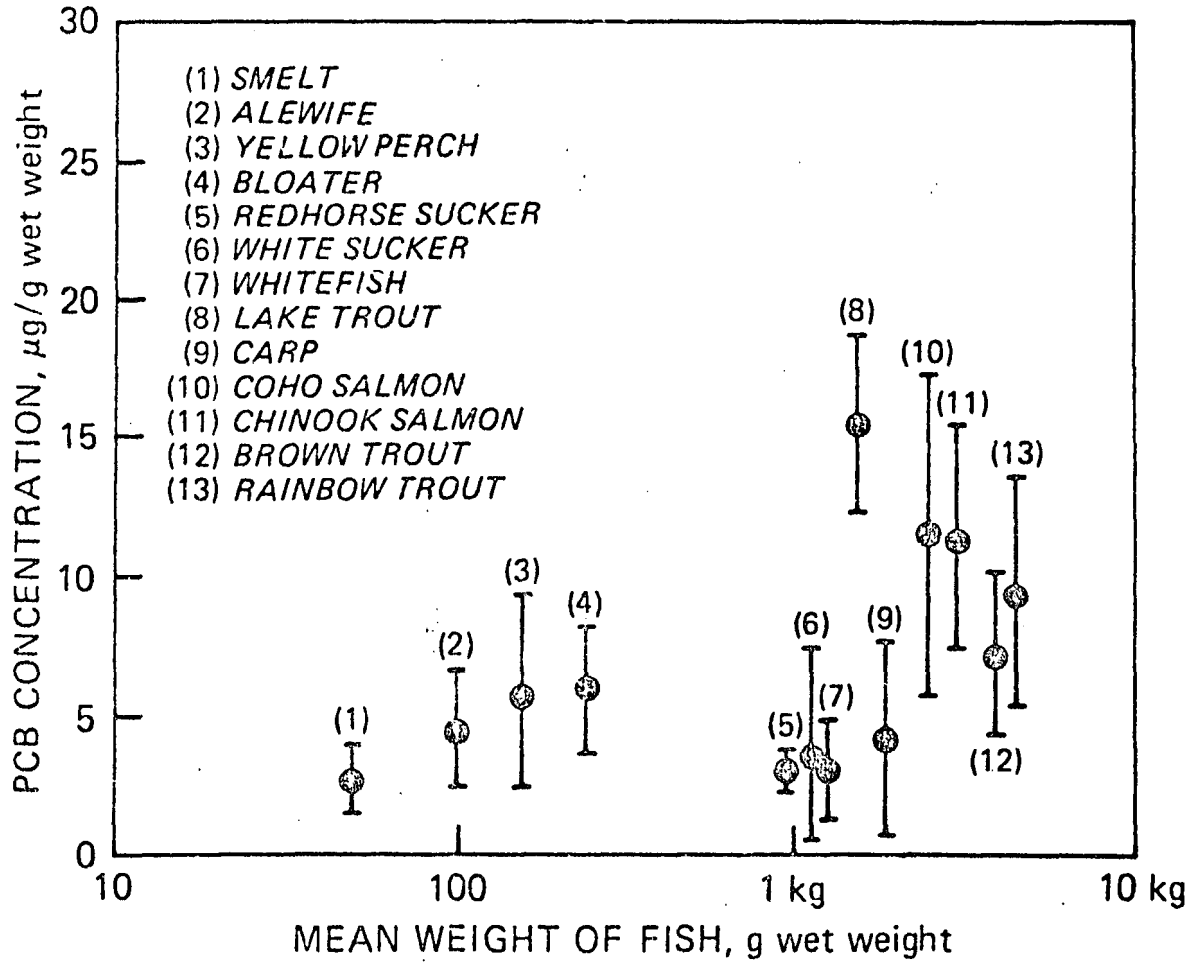

Figure 5. PCB concentrations (whole fish) as a function of weight, 1971 data from Veith (1975).

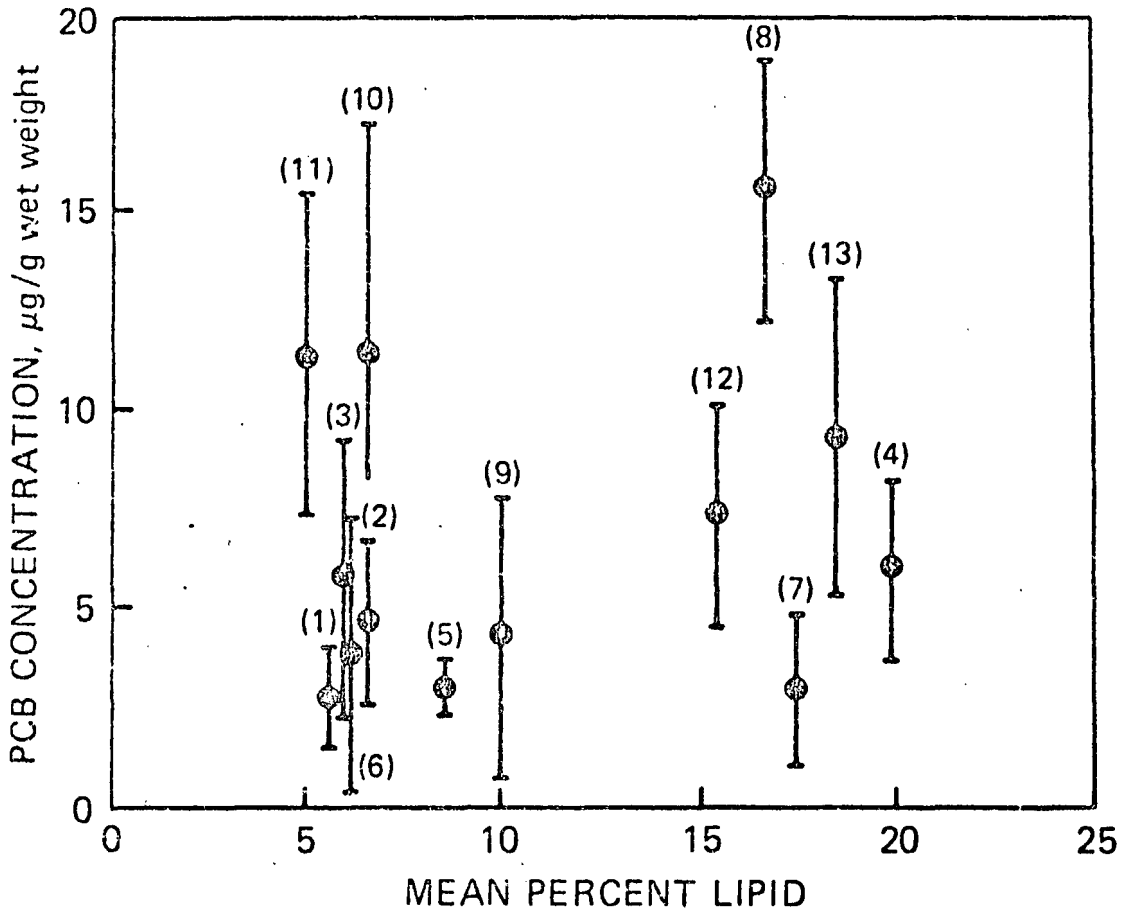

Figure 6. PCB concentrations (whole fish) as a function of % lipid. 1971 data from Veith (1975). (Numbers in parentheses are fish species, see Figure 5 for identification).

Food and Drug Administration. The collections were made throughout the Michigan waters of Lake Michigan. Twenty seven PCB samples were obtained for the lake trout and six for burbot in these waters. The data represent the concentration in the filet of the fish and are discussed in the analysis below.

The STORET data base includes data from a variety of sources including some samples from Wisconsin and Illinois and other agencies. The data compiled by Wapora (1981) include samples collected up to 1979 and are incorporated here to provide a more up-to-date data set. Zubik et al. (1978) reported PCB values for whole freshwater white and longnose mullet from 0.06-0.71 μg/g(w) for 1975-76.

The data sets discussed above have been compiled and analyzed with specific reference to the lake trout and the alewife as a principal food for the trout. For the combined data sets, Figure 3 shows the variation of PCB from 1971-1979 for the lake trout and Figure 7 shows a similar plot for the coho salmon. The lake trout range from 10-20 μg/g(w) while the coho salmon range from about 8-15 μg/g(w). No apparent trend exists for the coho while an apparent decrease occurs in the lake trout beginning after 1976.

The data were also summarized by age class using weight-age relationships discussed in Section 6, Estimation of Model Food Chain Interactions and Parameters. Figure 8 shows the PCB concentration for the alewife in 1971 and indicates that the average concentration was about 5 μg/g(w). The PCB concentration is essentially independent of the age of the alewife.

The lake trout data as a function of age class are shown in Figure 9. These data were assigned to age classes using growth rates generally representative of pre-stocked trout. Different growth rates and the associated distribution of PCB with age class are discussed in Section 9. For 1971, there is a decided trend towards increasing concentration with age. The initial high concentrations of 10-12 μg/g(w) for the 2-3 year old age class followed by the decline in the 4 year olds may be the result of exposure to higher water concentrations in in-shore areas. There is no apparent difference between the 1971 and 1975 data.

The distributions of PCB for the lake trout for the years 1971-79 and by

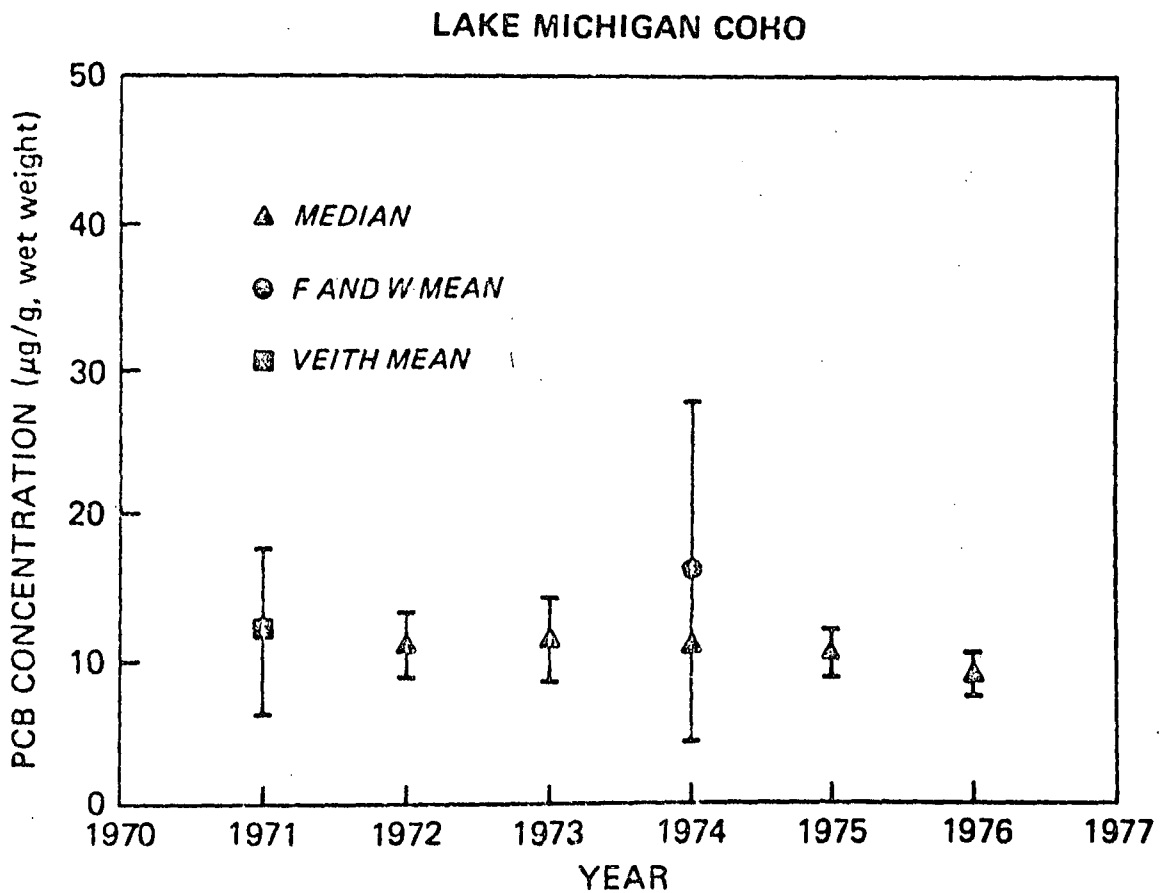

Figure 7. PCB concentrations in coho salmon, 1971-1976.

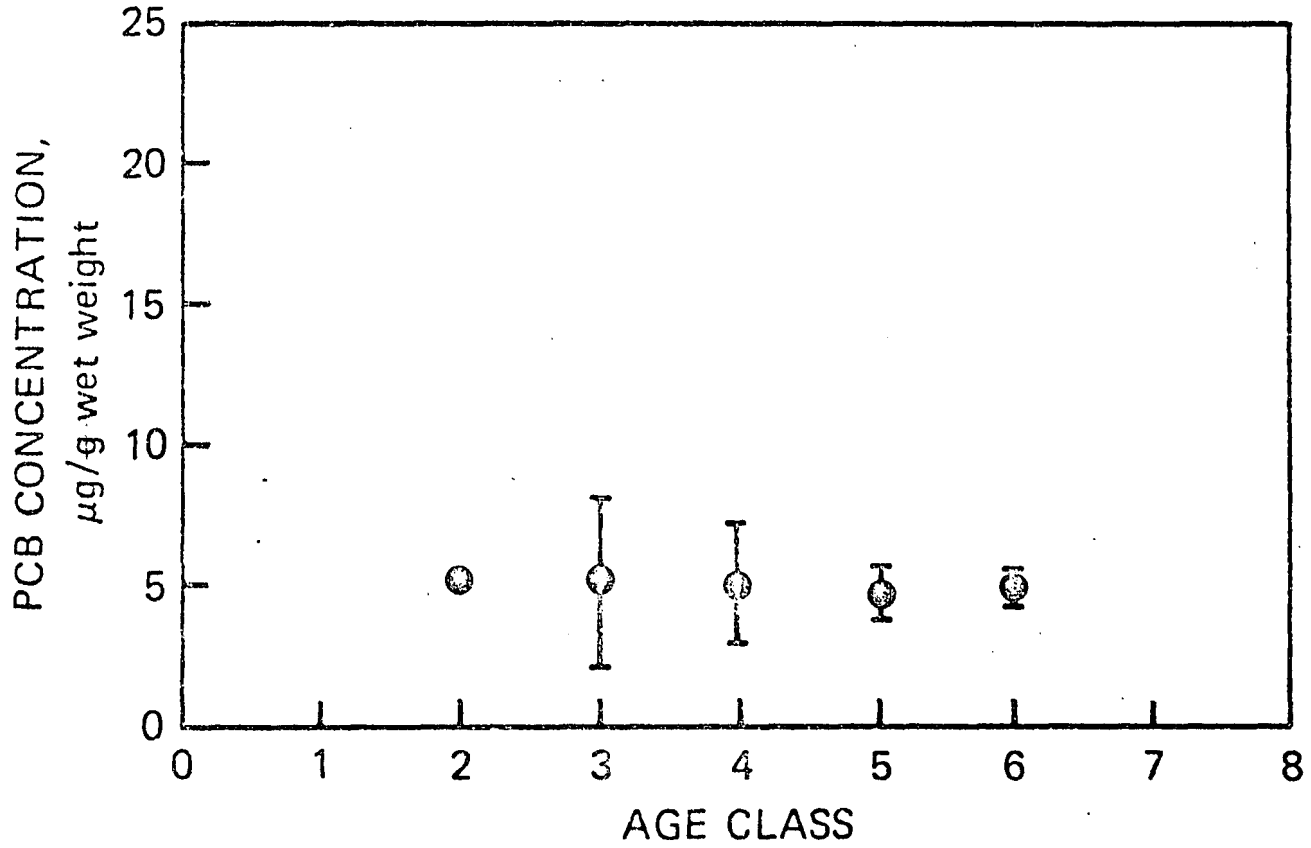

Figure 8. PCB concentration in alewife as a function of age, 1971.

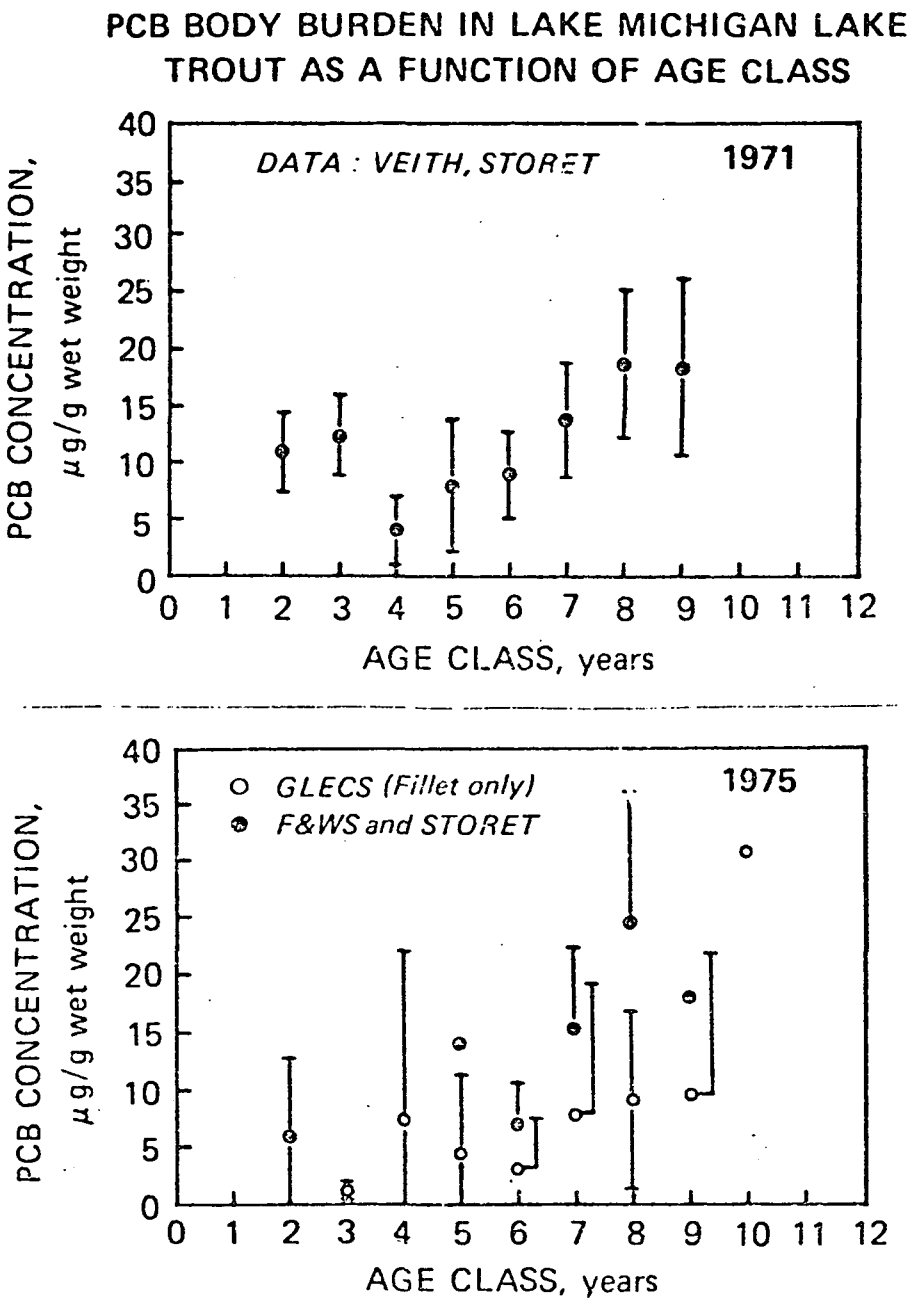

Figure 9. PCB concentration in lake trout as a function of age class for 1971 and 1975.

age class are shown in Figures 10 and 11. There is a clear decline in several of the upper age classes after 1976 with an apparent leveling off or increase in 1978 and 1979. The nine and ten year old age classes still had concentrations of 10 μg/g(w) or greater for these latter years. The data are therefore suggestive that there has been some decline in the fish PCB concentration although the decline appears to have leveled off in the upper age classes at greater than 5 μg/g(w).

Due to the high lipid solubility of PCBs, one would expect that the PCB concentration may be correlated with the lipid content of the lake trout. Figure 12 shows the available data. The relationship is not strong for the Veith data but is marked for the STORET data. For this latter data set, the PCB concentration is about directly proportional to the % lipid. The GLECS data are for filet only and also indicate some tendency toward increasing concentrations with increasing lipid content.

This compilation and analysis of the PCB concentration in the Lake Michigan ecosystem indicates the following:

1) Data on PCB are virtually absent for trophic levels below the fish, including zooplankton and phytoplankton
2) The data for the fish suggest an increase in PCB concentration as one proceeds up the food chain to the top predators. Lake trout PCB concentrations are the highest of the top predators
3) Across thirteen species there is no apparent relationship between PCB concentration and % lipid. For lake trout however, there is some evidence for some data bases that indicate increasing PCB concentration with increasing lipid content
4) Evaluation of the 1971 and 1975 data by age class for the lake trout indicates an approximate four fold increase in concentration of PCB from the 4 to 9 year old age classes.
5) The lake trout PCB data for 1971-79 appears to indicate a declining trend after 1975-76 with some leveling off in 1978 and 1979.

These summary conclusions regarding the PCB concentration in the Lake Michigan food chain also indicate the need to develop a modeling structure

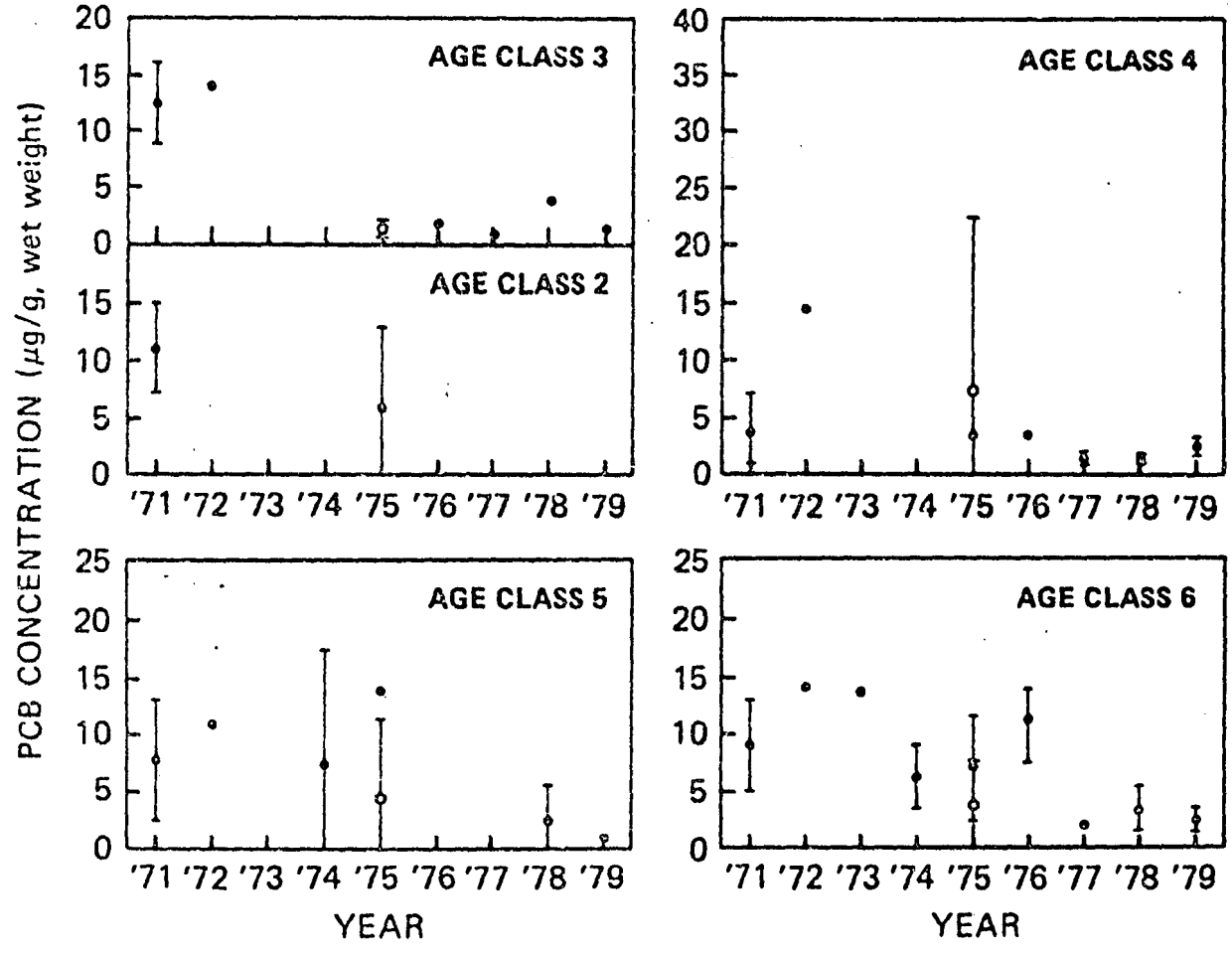

Figure 10. Variation of PCB for lake trout from 1971-79 for age classes 2-6 years

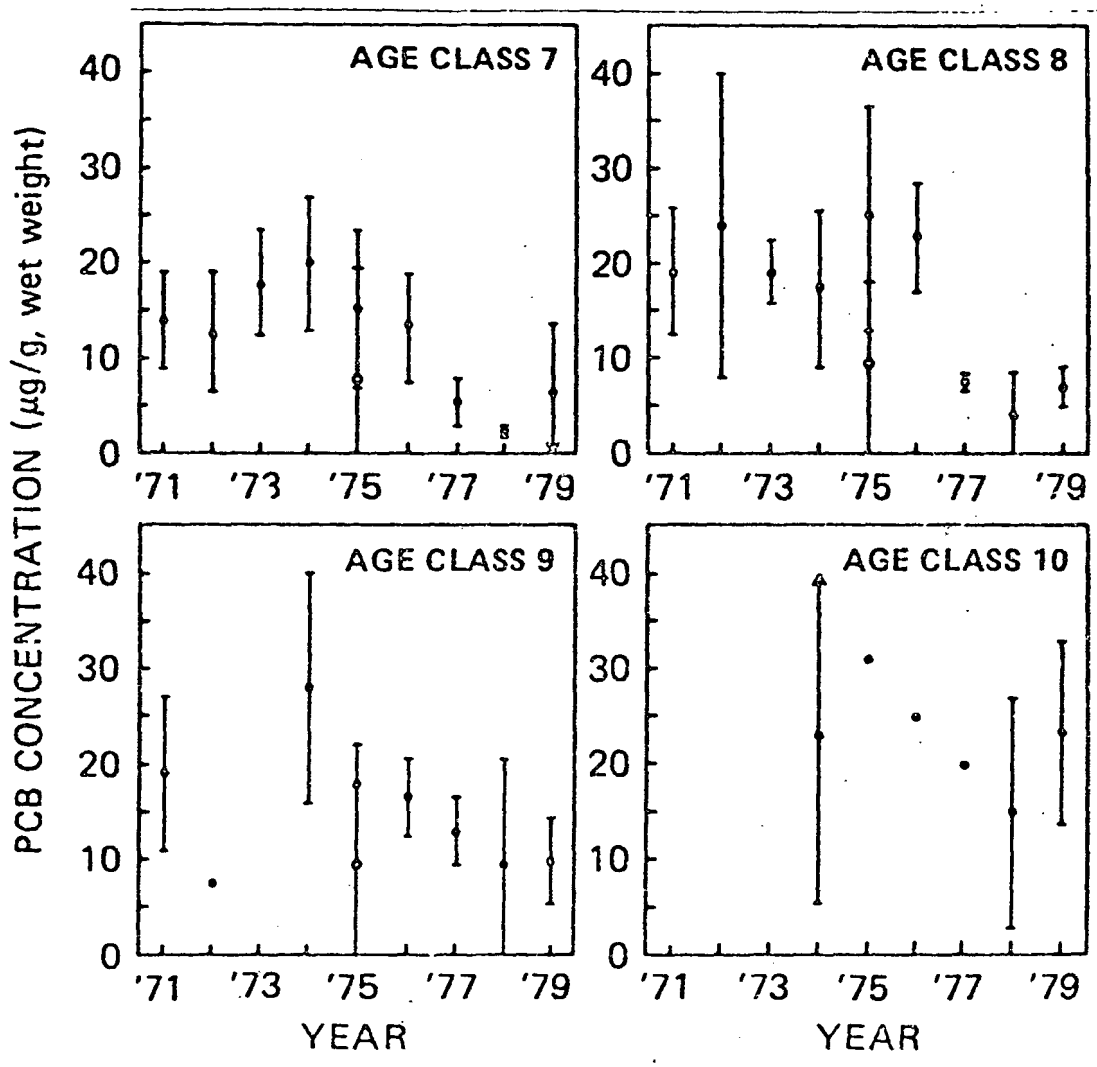

Figure 11. Variation of PCB for lake trout from 1971-79 for age classes 7-10 years

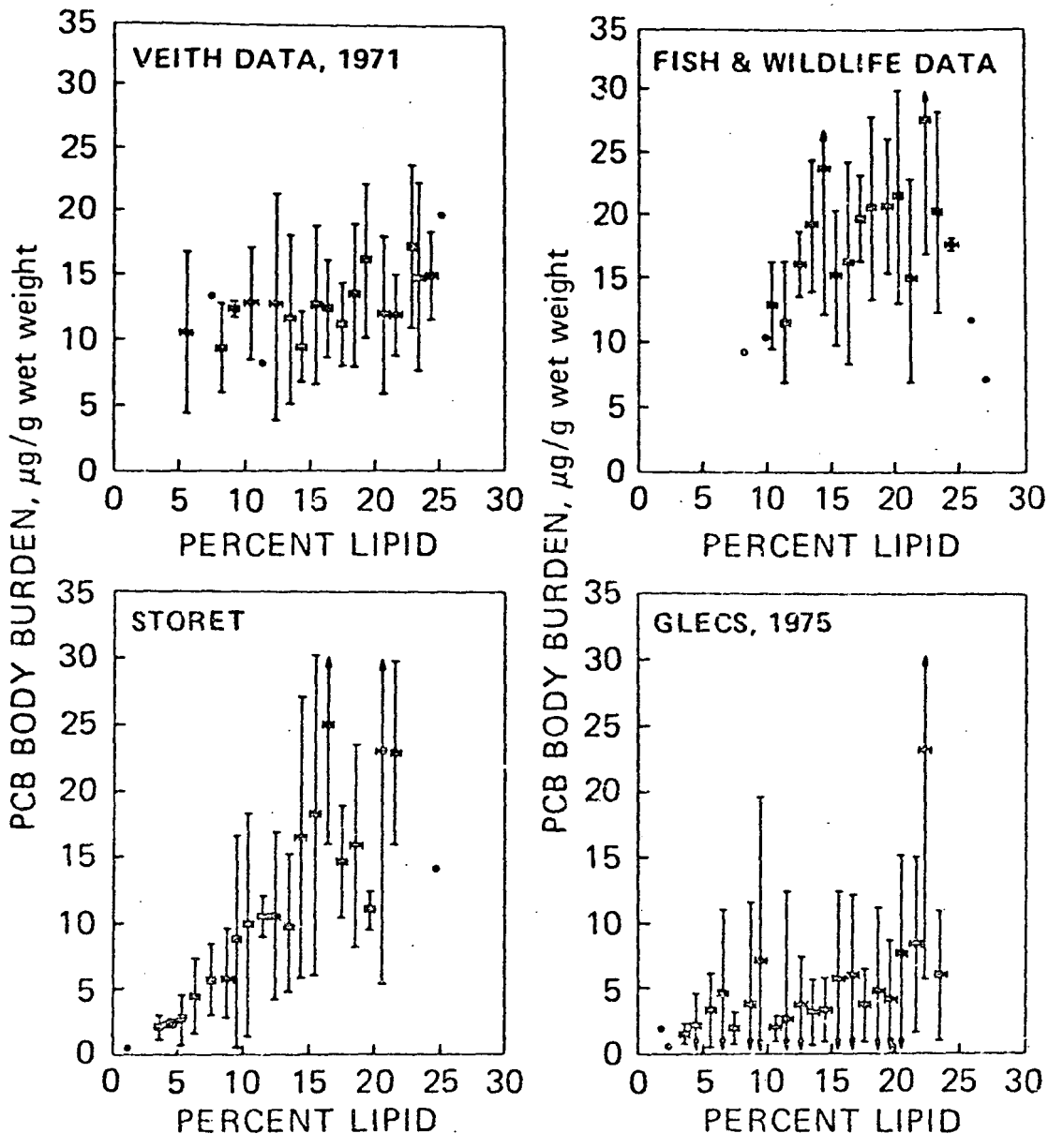

Figure 12. Relationships between PCB and % lipid for lake t·out from different sets.

that will address the aforementioned issues in light of the available data. A modeling framework can increase the understanding of the basic observed data to help resolve the questions of relative routes in the accumulation of PCB in the top predators.

SECTION 5
AGE DEPENDENT MODEL: THEORY

PREVIOUS WORK

A substantial literature exists on compartment models for transfer of material through food chains and food webs. The ecological concepts of compartment analyses have been reviewed by Dale (1970) and Patten (1971) and examples of compartment models include Gillet (1974), Hill et al. (1976), Lassiter et al. (1976), Haefner and Gillet (1976), and Aoyama et al. (1978). These approaches incorporate a variety of mechanisms in a timevariable mode.

Weininger (1978) formulated an age dependent model for PCBs in the lake trout of Lake Michigan. In that work, mass balance equations were written across the age of the trout and included the principal mechanisms of uptake directly from water and from food. Uptake from water is functionally dependent on fish respiration and related to the dissolved oxygen transfer across the gill surfaces. Food chain transfer is included through feeding on contaminated prey where the concentrations of PCB in the food is specified. Excretion of PCB is included as a first order loss mechanism and in the actual calibration of the model to lake trout PCB body burden excretion is assumed to be zero. Weininger concluded from his work that direct uptake from water is small and that the principal route for PCB accumulation in the lake trout is the food chain. The model described below builds on this earlier work, but rather than specifying the food PCB concentration, the model framework is extended to include calculation of PCB at each level of the food chain.

BIOCONCENTRATION AND DEPURATION

The uptake of a chemical directly from water through transfer across the gills as in fish or through surface sorption and subsequent cellular incorporation as in phytoplankton is an important route for transfer of toxicants. This uptake is often measured by laboratory experiments where test organisms are placed in aquaria with known (and fixed) water concentrations

of the chemical. The accumulation of the chemical over time is then measured and the resulting equilibrium concentration in the organism divided by the water concentration is termed the bioconcentration factor (BCF). A simple representation of this mechanism is given by a mass balance equation around a given organism. Thus,

$$\frac{d\nu'}{dt} = k_u wc - K\nu' \tag{1}$$

where ν' is the whole body burden of the chemical (μg), k_u is the uptake sorption and/or transfer rate (ℓ/d-g(w)), w is the weight of the organism (g(w)), c is the available water concentration (μg/ℓ), K is the desorption and excretion rate (d^{-1}) and t is time. This equation indicates that the mass input (μg/d) of toxicant given by $k_u wc$ is offset by the depuration mass loss rate (μg/d) given by $K\nu'$. It is assumed in the mass balance that the rate of change of the whole body burden is directly proportional to the concentration of the chemical in the water. For PCB, this is a good assumption as shown by Vreeland (1974) for oysters and Hansen et al. (1974) for pinfish. For both these studies, the resultant PCB concentration in the test animal was linear to the exposure water concentration. Eq. (1) is the proper starting point for analysis of the mechanisms controlling uptake directly from water. The whole body burden ν' is given by

$$\nu' = \nu w \tag{2}$$

where ν is the concentration of the chemical (μg/g(w)). Substitution of (2) into (1) gives

$$\frac{d\nu w}{dt} = k_u wc - K\nu w \tag{3}$$

Expanding the derivative and grouping terms yields

$$\frac{d\nu}{dt} = k_u c - \left(\frac{dw}{dt}/w + K\right)\nu \tag{4}$$

Letting $\frac{dw}{dt}/w = G$ (5)

where $G(d^{-1})$ is the net growth rate of the organism, and

$$K' = K + G \tag{6}$$

then

$$\frac{d\nu}{dt} = k_u c - K'\nu \tag{7}$$

It is seen that the loss term on the concentration includes the loss (or gain) due to the changing weight of the organism during the test and may therefore be termed an apparent depuration. The solution to Eq. (7) is

$$\nu = \frac{k_u c}{K'}(1 - \exp(- K't)) + \nu_o \exp(-K')t \tag{8}$$

where ν_o is the initial concentration of the chemical in the test organism. Note that this expression indicates that the rate of accumulation is a function of K', the sum of the depuration rate and the net growth rate. At equilibrium or steady state,

$$\nu = \frac{k_u c}{K'} \tag{9}$$

and the BCF is given by

$$N = \frac{\nu}{c} = \frac{k_u}{K+G} \tag{10}$$

The ratio N, the bioconcentration factor, is shown here in units μg/g ÷ μg/ℓ. In a form representative of a pseudo dimensionless ratio, the BCF is in typical units of μg/kg(w) ÷ μg/ℓ, i.e. ppb/ppb and then

$$N^1 = 1000\ N \tag{11}$$

The BCF is defined from Eq. (10) as the ratio of the uptake rate, k_u to the sum of the depuration and growth rate and represents the ratio of the equilibrium steady state chemical concentration to the dissolved water concentration. It is evident that BCF tests extending over time periods where the weight of the organism changes are affected by such weight changes. This effect should

be incorporated in analysis of the test data through Eq. (8) or (10). This is especially true for analyses of chemical concentration time history data from independent information on uptake or depuration rates.

For fish, the primary mechanism for uptake directly from water is through transfer of the chemical across the gill surface. The rate of transfer can be calculated from the rate of transfer of oxygen from water to the blood of the fish.

The rate of mass transport of a substance is given by

$$M = \frac{DA}{\delta} c \tag{12}$$

where M is the mass transport [μg/d], D is the diffusivity of the substance [cm^2/d], A is the effective surface area of gill for transfer [cm^2], δ is the effective thickness of the gill [cm], and c is the concentration of the substance in the water [μg/ℓ]. If it is assumed that the mechanism for uptake of the chemical, e.g. PCB is identical to oxygen uptake then

$$\frac{M_{PCB}}{M_{O_2}} = \frac{D_{PCB}\ c_{PCB}}{D_{O_2}\ c_{O_2}} \tag{13}$$

where the subscripts PCB and O_2 are for the chemical PCB and dissolved oxygen respectively. From (13),

$$M_{PCB} = (\beta \frac{M_{O_2}}{c_{O_2}})\ c_{PCB} \tag{14}$$

where $\beta = D_{PCB}/D_{O_2}$, the ratio of the diffusivity of PCB to that of oxygen. From (14),

$$M_{PCB} = k'_u\ c_{PCB} \tag{15}$$

where

$$k'_u = \beta\ M_{O_2}/c_{O_2} \tag{16}$$

The quantity k_u' represents the mass uptake for the whole fish and has units, ℓ/d. Dividing k_u' by the fish weight gives the uptake rate per unit weight, i.e.

$$k_u = \frac{k_u'}{w} = \beta \frac{M_{O_2}/w}{c_{O_2}} \tag{17}$$

The quantity M_{O_2}/w is the respiration rate, r, of the fish, i.e.

$$r = M_{O_2}/w$$

where r has units $[gO_2/g(w) - d]$. The uptake rate for the chemical is therefore related to the respiration rate of the organism by

$$k_u = \beta \frac{r}{c_{O_2}} \tag{18}$$

This representation is used in the model calibration discussed in the next section.

In depuration experiments, the organism is transferred to a tank with zero toxicant concentration and the time history of the loss of the chemical is measured. Eq. (8) then becomes for c=o,

$$\nu = \nu_o \exp(-K')t \tag{19}$$

or in terms of whole body burden

$$\nu' = \nu_o' \exp(-K)t \tag{20}$$

Again, the weight change of the organism must be properly taken into account. Eq.(19) can be used to estimate K' and from the net growth rate, the depuration rate K can be obtained. Or, Eq. (20) can be used directly where the whole body burden is plotted and the depuration rate obtained from such data.

MODEL FRAMEWORK

The model utilizes a mass balance of PCB around a defined compartment of the aquatic ecosystem. In the most general case, a compartment is defined as a specified age class of a specified organism or in the steady state simplified version discussed below, a compartment is considered as an "average" age class or range of ages for a given organism. Figure 13 schematically shows the compartments. As indicated in Figure 13(a), each age class of a given trophic level is considered as a compartment and a mass balance equation can be written around each such age class. The zero trophic level is considered to be the phytoplankton-detritus component representing one of the principal sorption mechanisms for incorporating toxicants into the food chain.

Consider then the phytoplankton, detrital organic material, and other organisms, all of size approximately <100 μm as the base of the food chain. An equation for this compartment is given by a simple reversible sorption-desorption linear equation as:

$$\frac{d\nu_o}{dt} = k_{uo}c - K_o\nu_o \tag{21}$$

where all terms have been defined, the subscript zero refers to the base of the food chain, and t is real time.

This simple expression indicates that the concentration at the base of the food chain is represented as a balance between the uptake directly from the water ($k_{uo}c$) and the loss due to desorption and excretion ($K_o\nu_o$). The chemical concentration is the concentration of the average phytoplankton or detrital particle on a wet weight basis. Eq. (21) therefore is not a balance equation for the mass of toxicant in the entire biomass.

For a compartment above the phytoplankton/detritus level, the mass input of the toxicant due to ingestion of contaminated food must be included. This mass input will depend on a) toxicant concentration in the food, b) rate of consumption of food and c) the degree to which the ingested toxicant in the food is actually assimilated into the tissues of the organisms.

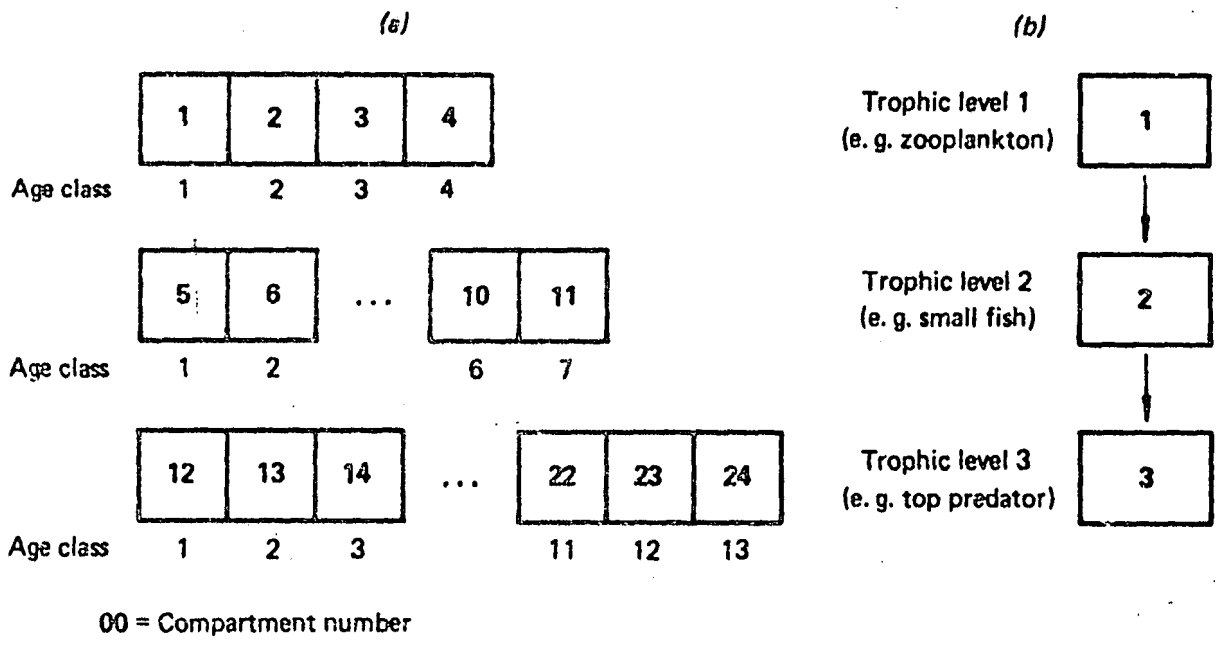

Figure 13. Schematic of compartment definition for a) age dependent model, (b) simplified steady state model.

The general mass balance equation for the whole body burden for a given compartment, i, is then similar to Eq. (1) for water uptake but with the additional mass input due to feeding. Therefore,

$$\frac{d\nu_i'}{dt} = \frac{d(\nu w)_i}{dt} = \frac{w_i d\nu_i}{dt} + \frac{\nu_i dw_i}{dt} = k_{ui} w_i c - K_i \nu_i'$$

$$+ \sum_j p_{ij} \alpha_{ij} C_{ij} \nu_j w_i \qquad i = 1 \ldots m \qquad (22)$$

where as before ν'_i and ν_i are the chemical whole body burden and concentrations respectively in the ith compartment, w_i is the weight of the average individual in the ith compartment, k_{ui} and K_i are, as before, the uptake or sorption rate and desorption and excretion rate, respectively, for the ith organism, α_{ij} is the chemical assimilation efficiency (μg chemical absorbed/μg chemical ingested), C_{ij} is the weight-specific consumption of organism i on j (g(w) prey/g(w) pred.-d), p_{ij} is the food preference of i on j, and t is real time (days). The quantity $(1 - \alpha)$, represents that portion of ingested chemical which is immediately or rapidly egested by the organism and never accumulated in body tissue. The excretion rate therefore in Eq. (22) is interpreted as excretion and desorption of the chemical that has been sorbed onto or into the organism. Consider now a sequential food chain where predation is only on the next lowest trophic level. This is done only for convenience. The general model as discussed in the next section includes a generalized feeding structure.

An equation for the individual organism weight is

$$\frac{dw_i}{dt} = (a_{i,i-1} C_{i,i-1} - r_i) w_i, \quad i = 1 \ldots m \qquad (23)$$

where $a_{i,i-1}$ is the biomass assimilation efficiency (g(w) pred./g(w) prey) and r_i is the respiratory weight loss (d^{-1}) due to routine metabolism, swimming, and other activities. The weight change is therefore

$$G_i = \frac{dw_i}{dt}/w_i = a_{i,i-1} C_{i,i-1} - r_i \qquad (24)$$

Equation (22) can then be written as

$$\frac{d\nu_i}{dt} = k_{ui} c + \alpha_{i,i-1} C_{i,i-1} \nu_{i-1} - K'_i \nu_i, \quad i = \ldots m \qquad (25)$$

where the preference term is unity.

The first term of Eq. (25) represents the direct uptake of the chemical by the organism from the water. The second term represents the flux of the chemical into the organism through feeding. Note that the units of this term are gram of chemical assimilated per gram predator per day. The third term is the loss of chemical due to desorption and excretion from body tissue at a rate K_i plus the change in concentration due to growth of the individual. The equation pair, (24) and (25), can be solved analytically for the case of known food concentration and constant coefficients or numerically for time-variable coefficients. Norstrom et al. (1976) and Weininger (1978) are examples; in each case, however, the food toxicant concentration is assumed known.

The interpretation of w_i and n_i in Equations (23) and (25) is further explained in Figure 14. The variation of the weight (and toxicant concentration) of a given compartment (i.e. a given age class of a given organism) is shown with real time. If as an example, w_5 is a 0-1 year old alewife then it is seen that the weight of this age class may vary from year to year. Similarly, the distribution of the toxicant may change from year to year for a given compartment depending on, for example, the variation in the water column toxicant concentration. The specification of the boundaries of a given compartment depends on the life cycle of the organism.

It should be recognized therefore that several of the biological and toxic parameters of the weight change equation (24) and the food chain equation (25) are functions of organism weight within the time interval defined by the compartment. One of the principal weight dependent parameters is respiration. Since the uptake of the chemical is dependent on the respiration through eq. (18) and the specific consumption also depends on the respiration through eq. (24), the weight dependence of the respiration assumes a significant role in the model structure.

Schmidt-Nielsen (1970) has reviewed the relationship between energy utilization and body size for an entire size spectrum of organisms. In addition, Norstrom et al. (1976), among others, have summarized the literature on respiration of fish as a function of weight. From an energetics viewpoint, the generalized relationship between respiration and body weight is given by

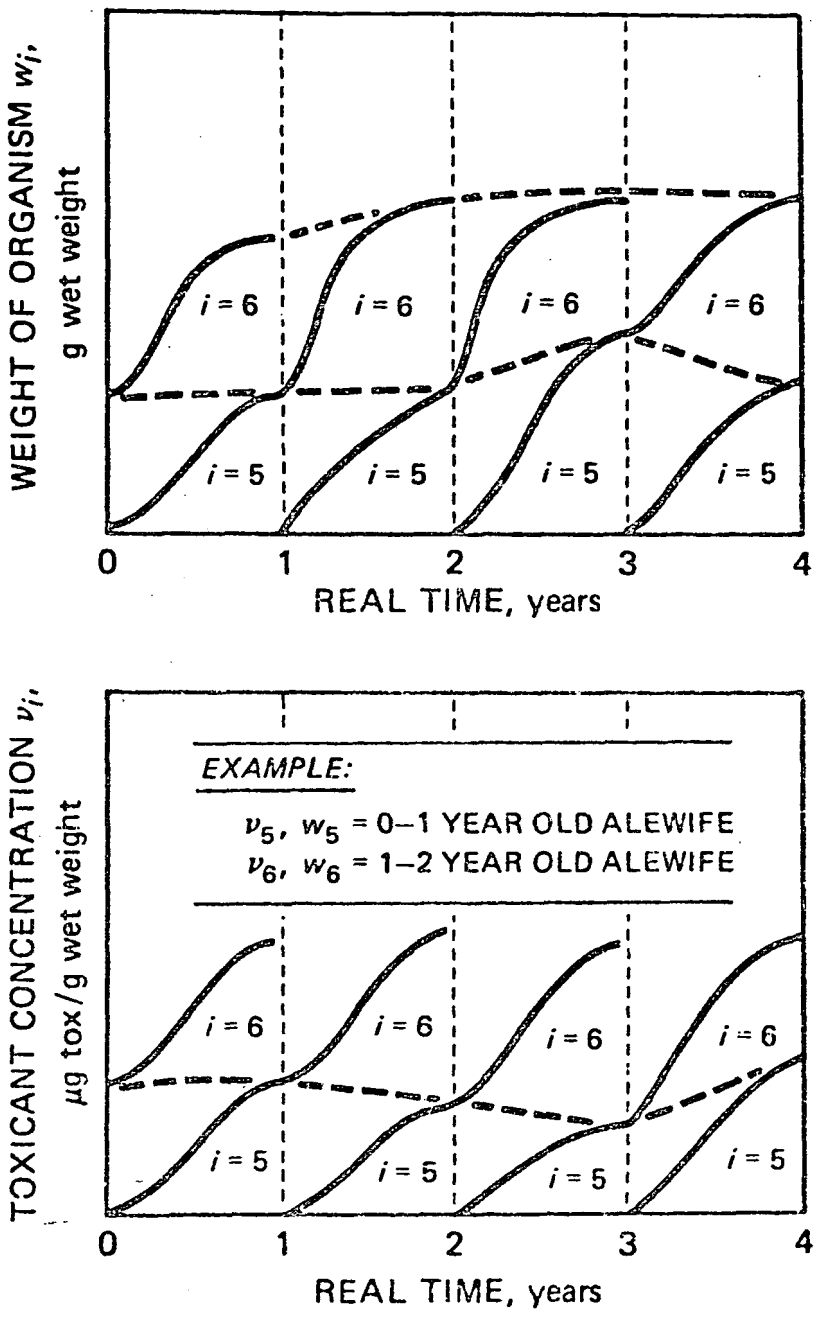

Figure 14. Illustration of meaning of $w_i(t)$ and $v_i(t)$ showing as an example a 0-1 year old and 1-2 year old alewife.

$$R = \alpha w^{\gamma} \tag{26}$$

where R is the kcal respired/day and for 1 kcal $\approx$ 1 g(w) also is approximately equivalent to g(w) respired/day and α and γ are empirical constants. For low routine metabolism,

$$R = \alpha_{\ell r} w^{\gamma} \tag{27}$$

Norstrom et al. (1976) suggest the following values for fish at 20°C:

$$\alpha_{\ell r} = 0.2 \frac{\text{kcal}}{\text{week} - g(w)^{0.8}}$$

and

$$\gamma \approx 0.8$$

The latter estimate for the slope of the relationship between metabolic rates and body weight is also approximately shown in Schmidt-Nielsen (1970) over a range from 10^{-12} to 10^{4}g. The range in γ is about 0.75 to 0.9 with a mean close to 0.75. See also Section 6, Eq. (53). Thus, in general

$$R \approx \alpha_{\ell r} w^{0.8} \tag{28}$$

for R in g(w)/day. Normalizing this expression by the weight of the organism gives

$$r' \approx a_r w^{-.2} \tag{29}$$

where r' is the fish respiration rate in g(w) respired/g(w)-day and a_r is a constant which may be organism dependent. In terms of oxygen uptake, the respiration rate is given approximately by

$$r = 2.7\ \eta a_c r' \tag{30}$$

where r is in gO_2/g(w)-day, η is the dry weight to wet weight ratio (g(d)/g(w)) and depends on the species, a_c is the carbon to dry weight ratio [g Carbon/g(d)]

set at 0.4 in this work and 2.7 is the gO_2 utilized/g Carbon.

Following eq. (18), the chemical uptake rate directly from the water is then given by

$$k_u = \frac{2.7\ \eta a_c \beta}{c_{O_2}}\ w^{-0.2} \tag{31}$$

or

$$k_u(w) = a_k w^{-.2} \tag{32}$$

where $a_k = \frac{2.7\eta a_c \beta}{c_{O_2}}$

From Eq. 24, for a growth rate, G, determined from weight-age data, respiration from eq. (29) and a constant food assimilation ratio, a, the specific consumption can be computed as

$$C(w) = \frac{G(w) + r_i'}{a} = \frac{G(w) + a_r w^{-.2}}{a} \tag{33}$$

Similarly, the depuration rate is considered as a function of weight, i.e., $K = K(w)$. Thus the general model calculation is given from eqs. (25), (32) and (33) as

$$\frac{d\nu_i}{dt} = a_k w_i^{-.2}\ c + \frac{\alpha_{i,i-1}}{a_{i,i-1}} (G_i(w) + a_r w_i^{-.2}) - (K(w) + G(w))\nu_i \tag{34}$$

In this form, it is difficult to understand the behavior of the equation and the relative importance of each mechanism. Some additional insight into the behavior of the equations derived above therefore can be obtained by examining a simple steady state case.

EQUILIBRIUM (STEADY STATE) CONDITION

The ith compartment around which the mass balance is taken to derive Eq. (25) is defined as a given age-class of an organism in the food chain. The properties of that age-class include the weight change as a result of net growth, as shown in Eq. (23). The net growth rate of the age-class can be considered to be a constant in time, i.e. that the average growth rate of a

2 to 5-yr-old top predator is the same from 1 year to the next. This assumption implies no temporal change in the feeding rate, food availability or respiration rate. Such an assumption may be entirely reasonable for the lower levels of the food chain but may be questionable for the top predators. Nevertheless, considerable insight can be gained into the behavior of the contaminant equations by assuming a constant growth rate. In a similar fashion, the assumption is made that the uptake, assimilation, and excretion rates are constant within a defined age-class (i.e. a defined compartment).

The solution to Eq. (24) for constant coefficients is

$$w_i = w_{oi}\exp(G_i t) \tag{35}$$

where w_{oi} is the initial weight at the beginning of the age-class i.

The solution to Eq. (22), the body burden equation under an assumption of constant coefficients, is

$$\nu'_i = \frac{(k_{ui}c + \alpha_{i,i-1}C_{i,i-1}\nu_{i-1})w_{oi}}{K'_i}(\exp(G_i t) - \exp(-K_i t)) + \nu'_{oi}\exp(-K_i t) \tag{36}$$

where ν'_{oi} is the initial contaminant body burden at the beginning of the age-class i.

The solution to Eq. (25) is

$$\nu_i = \frac{k_{ui}c + \alpha_{i,i-1}C_{i,i-1}\nu_{i-1}}{K'_i}(1 - \exp(-K'_i t)) + \nu_{oi}\exp(-K't). \tag{37}$$

Figure 15 shows the form of Eqs. (35)-(37). The increase in the weight of the given age-class is exponential as shown in Fig. (15a), with a constant net growth rate of G_i. Figure (15b) shows the increase in body burden and indicates that no equilibrium is reached, i.e. the total mass of contaminant in

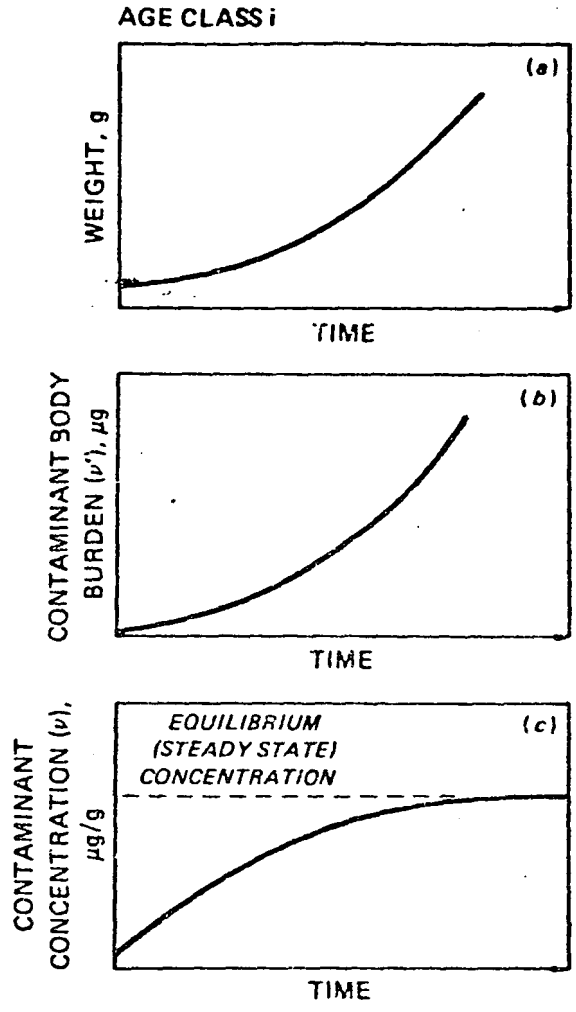

Figure 15. Illustration of equilibrium conditions for concentration of contaminant in organism. (a) Exponential increase in organism weight, Eq. (11); (b) increase in contaminant body burden, Eq. (20); (c) increase in contaminant concentration to equilibrium (steady state) level, Eq. (21).

the organism continues to grow as a result of uptake, assimilation, and increase in weight. Indeed, Eq. (36) shows that only if the net growth rate is zero (i.e. weight of the organism is a constant) is there an equilibrium condition on the body burden. A constant weight assumption, however, is a particularly poor one. Figure (15c) shows that even as weight is increasing exponentially and the body burden increasing according to Eq. (36), the concentration of the contaminant approaches a constant value, an equilibrium or steady state condition. This equilibrium concentration represents a balance between the inputs of the contaminant to the organism and the mechanics of excretion and increasing weight of the organism. This can be seen from Eq. (25) where at equilibrium, $d\nu_i/dt = 0$, and

$$K'_i \nu_i = k_{ui} c + \alpha_{i,i-1} C_{i,i-1} \nu_{i-1}$$

or

$$\nu_i = \frac{k_{ui} c + \alpha_{i,i-1} C_{i,i-1} \nu_{i-1}}{K'_i} \tag{38}$$

which can also be obtained from Eq. (37) at large t.

If then it is assumed that the growth and contaminant related coefficients are constant over the length of time defined by the age-class of the compartment, then an equilibrium or steady state condition in the concentration of the contaminant is reached. This is the basis of considering an equilibrium or steady state model where the time derivative in Eq. (25) is set equal to zero.

One can also obtain from this line of reasoning the appropriate age-class interval for each compartment that will guarantee an approximate equilibrium condition. If a criteria of 90% of equilibrium is chosen, then the age interval t_a for a compartment is

$$t_a \geq 2.3/K' \tag{39}$$

For example, for a low excretion rate (e.g. $K = 0.001/d$) and a high net growth rate that are representative of the upper levels of the food chain (e.g. $G = 0.01/d$), $t_a \geq 3.1$ yr. Therefore if one considers the lower level to be repre-

sentative of an age-class spanning about 0.5 yr and the upper food chain level to be representative of an age-class spanning about 3 yr, then an equilibrium, steady state concentration will be reached. Such a concentration will be approximately independent of the initial concentration at the beginning of the age-class as can be determined from Eq. (37). On the other hand, if excretion is very low, say approximately zero, and if the net weight change G is also very low, say approximately zero, then Eq. (39) indicates that the age span approaches infinity. This simply implies that an equilibrium condition will never be reached under those compartment characteristics. Such cases can occur for low excretion rates and older top predators. To summarize, equilibrium or steady state implies constant positive net weight change and constant uptake, assimilation, and excretion rates over an age-class interval defined to guarantee 90% of equilibrium.

Continuing with this simplification, Eq. (38) can be written as

$$\nu_i = N_i c + f_{i,i-1}\nu_{i-1} \tag{40}$$

$$= \nu_{iw} + \nu_{if}$$

where N_i is the BCF given by

$$N_i = \frac{k_{ui}}{K_i + G_i} = \frac{\nu_{iw}}{c} \tag{41}$$

$f_{i,i-1}$ is a food chain transfer number (a biomagnification ratio) given by

$$f_{i,i-1} = \frac{\alpha_{i,i-1}C_{i,i-1}}{K_i + G_i} \tag{42}$$

and ν_{iw} and ν_{if} are the components of the chemical concentration in organism i due to uptake from water and food chain transfer respectively.

The f factor therefore represents the ratio of the chemical in the predator due to food chain transfer relative to the chemical concentration of the prey, i.e.

$$f_{i,i-1} = \frac{\nu_{if}}{\nu_{i-1}} \tag{43}$$

this ratio can be obtained experimentally through feeding experiments where the water concentration is zero or where enough experimental information is obtained to calculate f from Eq. (43). Lieb et al. (1974), in feeding experiments using rainbow trout and Aroclor 1254, obtained a ratio of about 2 indicating that the approximate equilibrium concentration of the trout was about twice that of the PCB in the diet. Note that the meaning of f is contrasted to the BCF, the former as a ratio of chemical concentration in predator to that in food, the latter a ratio of chemical concentration in predator to that in the water. Comparisons then of the numerical magnitude of the BCF (for PCB, order 100 μg/g ÷ μg/ℓ) and the f factor (for PCB, order 1 μg/g(w) ÷ μg/g(w)) can be misleading.

For example, consider a four step food chain representative of phytoplankton, zooplankton, small fish and large fish. The chemical concentration at each level are then given from Eq. (40) as

$$\nu_1 = N_1 c \tag{44a}$$
$$\nu_2 = N_2 c + f_{21}\nu_1 \tag{44b}$$
$$\nu_3 = N_3 c + f_{32}\nu_2 \tag{44c}$$
$$\nu_4 = N_4 c + f_{43}\nu_4 \tag{44d}$$

Successive substitutions beginning with the lowest level and proceeding to the fourth level give the chemical concentration for the top predator as

$$\begin{aligned}\nu_4 &= N_4 c + [f_{43}N_3 + f_{43}f_{32}N_2 + f_{43}f_{32}f_{21}N_1]c \\ &= \nu_{4w} + \nu_{4f}\end{aligned} \tag{45}$$

The first term, ν_{4w}, represents as before the chemical concentration in the top predator due to uptake from water only and the second term, ν_{4f}, represents the chemical concentration due to the cumulative transfer of the chemical through the food chain. Now for PCB, a reasonable assumption is that the BCF is approximately constant across the food chain. Therefore,

$$N_4 = N_i \qquad i = 1...3$$

and Eq. (45) becomes

$$\nu_4 = [1 + f_{43} + f_{43}f_{32} + f_{43}f_{32}f_{21}]\nu_{4w} \tag{46}$$

The effect of food chain transfer is now clearly additive to the concentration from water uptake only. In this model then, there will always be an increase in chemical concentration in a predator due to food chain transfer. The degree of relative significance of that transfer is the question, not whether there is any food chain transfer at all. Consider the case where f is a constant across the food chain at a value of 2 (the value obtained by Lieb et al. (1974) for rainbow trout). Then

$$\nu_4 = [1 + 2 + 4 + 8]\nu_{4w}$$

$$= 15\ \nu_{4w}$$

showing that the concentration of the top predator is fifteen times the level due just to uptake from water. The cumulative effect from the lowest level (phytoplankton) to the top predator can also be seen.

Figure 16 summarizes the effect of the food chain transfer factor in the ratio of chemical from water and food to that from water only. Note that for f = 0.5, and a four level food chain, the resultant concentration in the top predator would be almost twice that from water uptake only. A water quality standard for chemicals that were based solely on water uptake to the food chain would therefore exceed an attainable chemical concentration by about two times if the food chain were not taken into account.

If food chain transfer is considered "significant" at say 20% of the concentration due to water only, then the approximate constant f level can be calculated. Thus if

$$\frac{\nu_4}{\nu_{4w}} = 1.2$$

then f is about 0.2. If the ratio of chemical concentration in the predator to that in prey in feeding experiments is therefore only 0.2, there will be an approximate 20% increase in the top predator over the concentration from water only. A lower bound of f of about 0.2 is suggested then as the boundary between significant and non-significant food chain transfer effects.

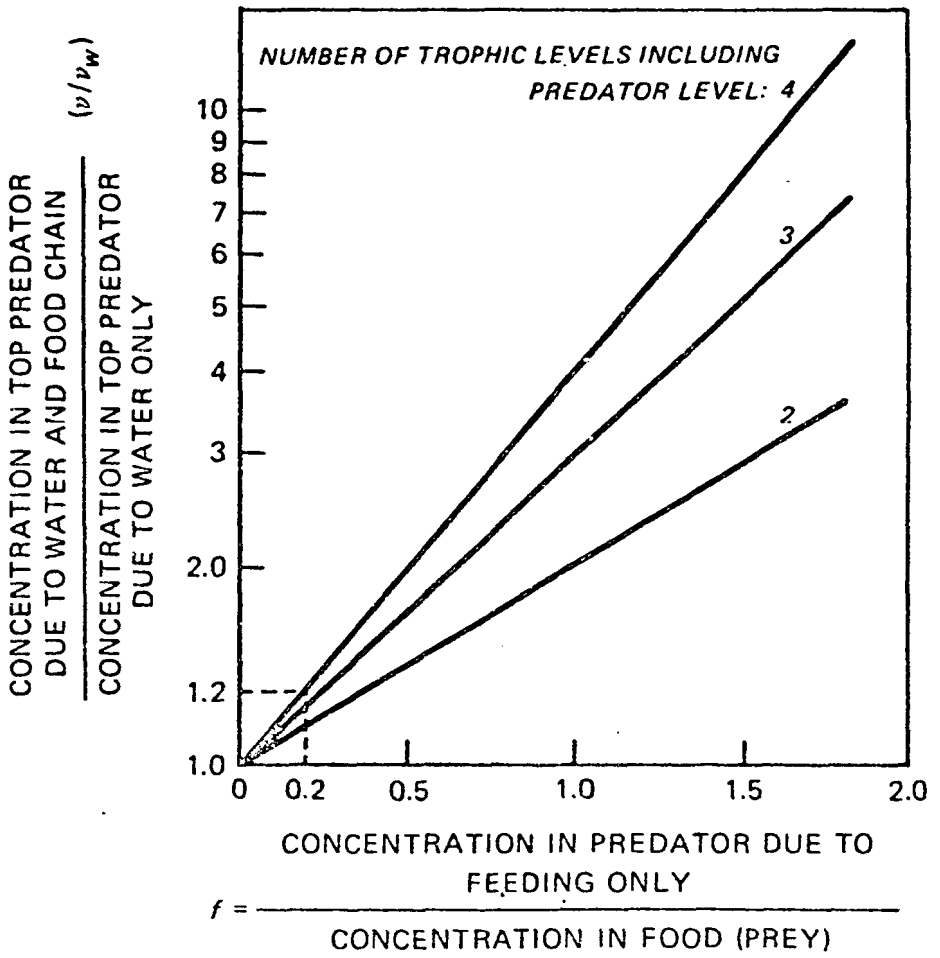

Figure 11. Effect of food chain transfer ratio, f, on ratio of chemical concentration in top predator from water plus food to that from water only. Equilibrium conditions, constant BCF and f ratio along food chain.

SECTION 6

ESTIMATION OF MODEL FOOD CHAIN INTERACTIONS AND PARAMETERS

The accumulation of PCBs in the Lake Michigan food chain is modelled assuming a four species food chain consisting of phytoplankton, *Mysis relicta*, alewife (*Alosa pseudoharengus*), and lake trout (*Salvelinus namaycush*). As discussed below, this species linkage constitutes the major energy transport route to the lake trout. Both *Mysis* and alewife are viewed as representative species of the middle levels of the food chain acknowledging that other invertebrates and small fish also contribute to the observed PCB levels in Lake Trout. The phytoplankton component of the model is assumed to represent non-living particulate organic material as well as living plankton.

Phytoplankton are represented by a single compartment that is assumed to be in dynamic equilibrium with water column dissolved PCB. The other species are separated into discrete age classes. For any age class growth rate and predator-prey relationships are constant. Dynamics of feeding and growth within an age class are not considered.

FEEDING STRUCTURE

Mysis

Mysis relicta is a crustacean of the order Mysidacea. It is found in North America in the deep waters of the Great Lakes, some of the Canadian lakes, Green Lake of Wisconsin and the Finger Lakes of upper New York State (Klots, 1966). It is a filter feeder, feeding primarily on plankton and detritus although it also may seize zooplankton (Mozley and Howmiller, 1977).

Mysis exhibits a diurnal migration cycle, moving vertically from the bottom at dusk, feeding in the water column and then returning to the bottom waters. Beeton (1960) has described this cycle in Lake Michigan showing that light and vertical thermal structure control the migration. As surface light intensity increases the height of migration decreases. If a sharp thermocline is present, the mysids will generally concentrate just below it until their descent prior to sunrise.

Mysis are present throughout Lake Michigan and may be found all year at depths greater than 30 m. (Mozley and Howmiller, 1977). In winter, populations may be found at shallower depths. They are an important link in the food chain, especially as a forage for fishes (Grossnickle, 1978).

Morgan and Beeton (1978) found the life cycle of *Mysis* in Lake Michigan to be about 16 months for females and slightly longer than 1 year for males. Females can produce two broods of young, the first at 12 months and the second at 16 months. Juveniles are released at distinct intervals throughout the year with a periodicity of about 4 months.

The feeding habits of Lake Michigan *Mysis* have been determined by examination of gut contents. In a qualitative analysis McWilliam (1970) found that plankton were the dominant food source, although some cladocerans, copepods and juvenile mysids were present in *Mysis* guts. Grossnickle (1978) found that vertically migrating mysids in Lake Michigan have nearly empty guts as they begin their ascent but feed rapidly as they move up in the water column, suggesting that *Mysis* may be regarded as a pelagic consumer. Grossnickle also indicated that the analysis of fecal pellets suggested that feeding was primarily on phytoplankton. In laboratory experiments he did show, however, that adult mysids were omnivorous when offered naturally occurring phytoplankton assemblages and specific zooplankton species.

The food chain model is structured with four 4 month age classes of *Mysis* reflecting life span and birth frequency. All classes consume phytoplankton exclusively.

Alewife

The alewife is an anadromous fish found in the Atlantic Coast waters of North America. It also has become established in certain freshwater bodies including the Finger Lakes of upper New York and all the Great Lakes. In Lake Michigan, alewives have increased from their first reported appearance in 1949 to become the most abundant species (Janssen and Brøndt, 1980). They feed almost exclusively on crustacea ranging in size from small copepods and cladocera to *Pontoporeia hoyi* and *Mysis relicta* (Morsell and Norden, 1968).

Algae have been found in alewife stomachs but this is believed to be incidental to feeding on zooplankton (Morsell and Norden, 1968). Alewives live approximately seven years spawning in late June in Lake Michigan (Carlander, 1969).

An intensive study of the feeding habits of adult alewives in Lake Michigan (Janssen and Brandt, 1980) showed that they concentrated near the bottom during the day and migrated to the base of the thermocline at night in a pattern that coincided with that of Mysis. The authors suggested that the vertical migration of alewife was mechanistically linked to their feeding behavior, i.e., they were actually following and consuming Mysis. Based on stomach content data, the authors concluded that Mysis was the major constituent in the diet of adult alewives. Pontoporeia and microcrustacean zooplankton were also present in alewife stomachs but were much less common than Mysis. In addition, average body length of Mysis in alewife stomachs were longer than average body length of Mysis in the water column (Fig. 17), a possible consequence of larger Mysis being more visible and thus more vulnerable to predation and/or predation on only the layer of large Mysis in the observed vertical segregation of Mysis by size.

In the model the alewife component is divided into 7 single year classes. The feeding structure (Fig. 18) reflects the field observations described above, young-of the-year alewife consuming phytoplankton and all other age classes consuming Mysis with a bias toward the larger Mysis (Table 3).

Lake Trout

The lake trout is a predatory fish found throughout Canada, the Great Lakes drainage basin and parts of New England, New York, Wisconsin, Minnesota, and Montana (Carlander, 1969). Up to the mid-1940's the lake trout was abundant in Lake Michigan and a species of primary importance to the commercial fishery. Beginning in 1946 production dropped catastrophically due to overfishing and predation by the sea lamprey. By the mid-1940's lake trout were virtually extinct in Lake Michigan.

In an effort to restore the fishery, sea lamprey production was controlled

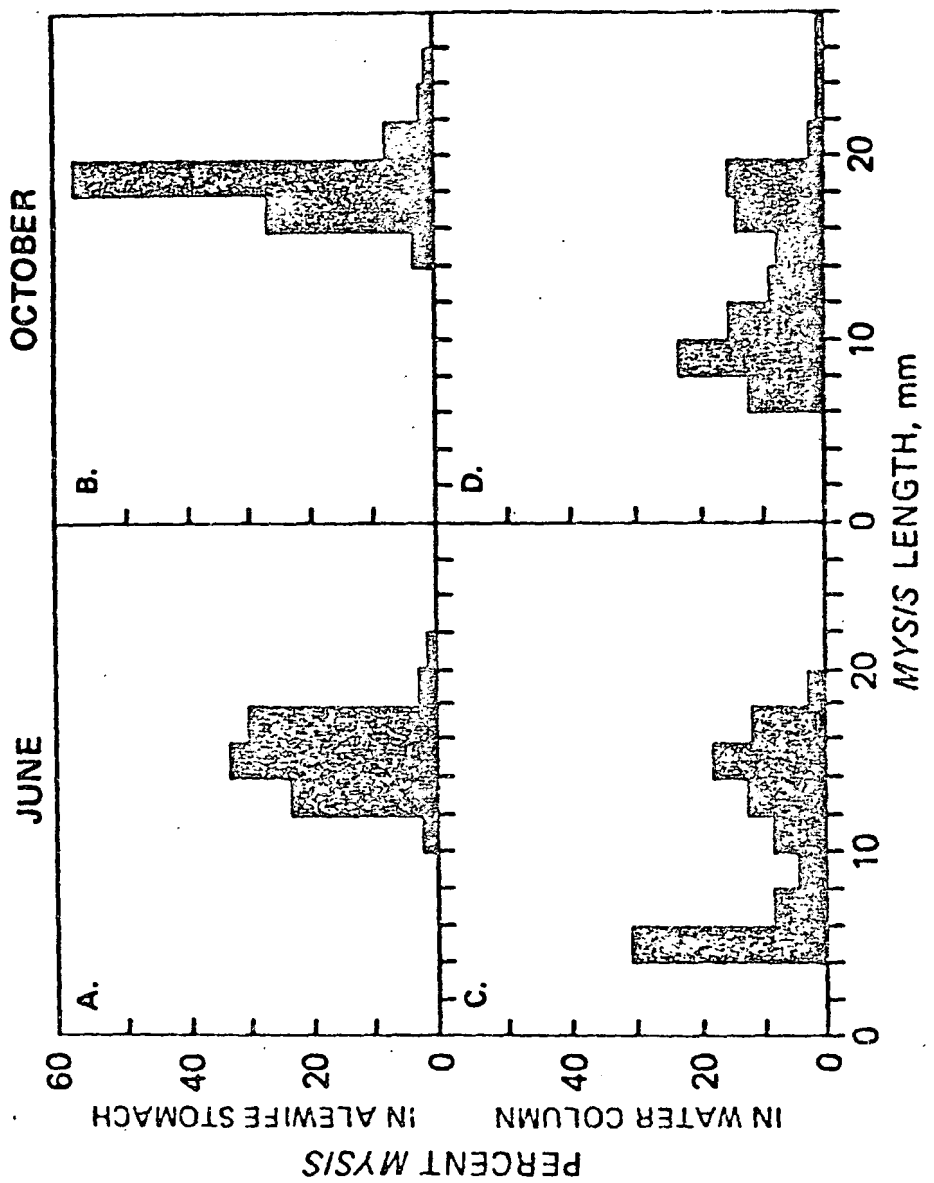

Figure 17. Percent of total Mysis captured in the water column and found in alewife stomach in relation to Mysis length for two sampling dates. (From Janssen and Brandt, 1980).

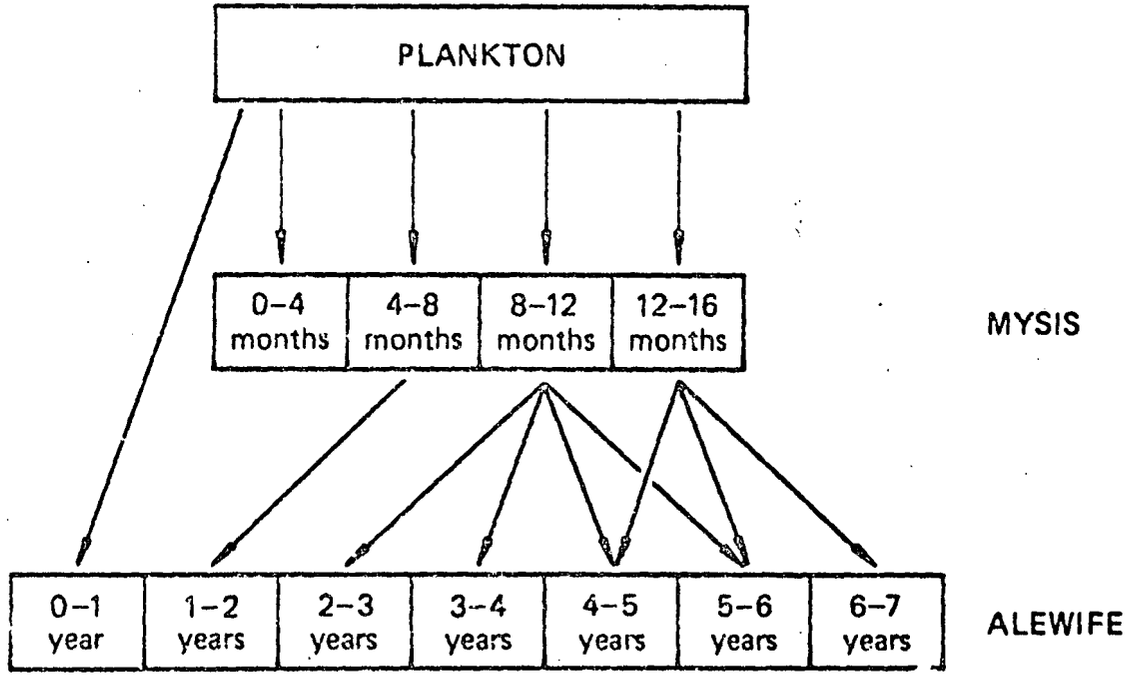

Figure 18. Feeding Interactions specified in the model for alewife and *Mysis*.

TABLE 3. FRACTION OF TOTAL ALEWIFE CONSUMPTION ON PLANKTON AND EACH *MYSIS* AGE CLASS

ALEWIFE AGE CLASS	PLANKTON	0-4 MO. MYSIS	4-8 MO. MYSIS	8-12 MO. MYSIS	12-16 MO. MYSIS
0-1 yr	1.0				
1-2 yr			1.0		
2-3 yr				1.0	
3-4 yr				1.0	
4-5 yr				0.5	0.5
5-6 yr				0.5	0.5
6-7 yr					1.0

by poisoning larvae and blocking spawning routes and in 1965 lake trout stocking was begun. Each year several million yearling lake trout are planted at various sites throughout the lake.

A summary of lake trout food items in relation to season and lake trout size (Weininger, 1978) shows that invertebrates (largely Mysis relicta and Pontoporeia hoyi) are the major food items for trout up to 200 mm total length (approximately 2 years old) and that alewife and to a lesser extent sculpin and smelt are the major food items for all larger trout.

Data compiled from stomach content examinations reported by Wright (1968) (Fig. 19) also show invertebrates to be the major food source of young trout, fish the main food source of older trout and alewife the major component of that fish.

Also compiled from data reported by Wright (1968) is the distribution of alewife age classes in the lake trout diet (Fig. 20). These data indicate that trout eat over a wide range of alewife age classes and that the range of age classes consumed increases with the age of the trout.

The lake trout component of the model is divided into 13 single-year age classes (Fig. 21). Reflecting the above data, the first two age classes consume Mysis exclusively, the next class consumes Mysis and first and second year alewife. Older trout consume alewife exclusively, with an age class distribution commensurate with the stomach content data presented (Table 4).

BIOLOGICAL PARAMETERS

Weight-Age Relationship

The biological parameters required by the model and described in Section 5 include growth rate, respiration rate and assimilation efficiency of food. These parameters must be specified for each age class of all species other than phytoplankton used in the model.

Growth rate, which is a function of both temperature and diet, is measured either as the change in length or the change in weight with age. Length change with age may usually be described by an equation of the form (Schnute and Fournier, 1980):

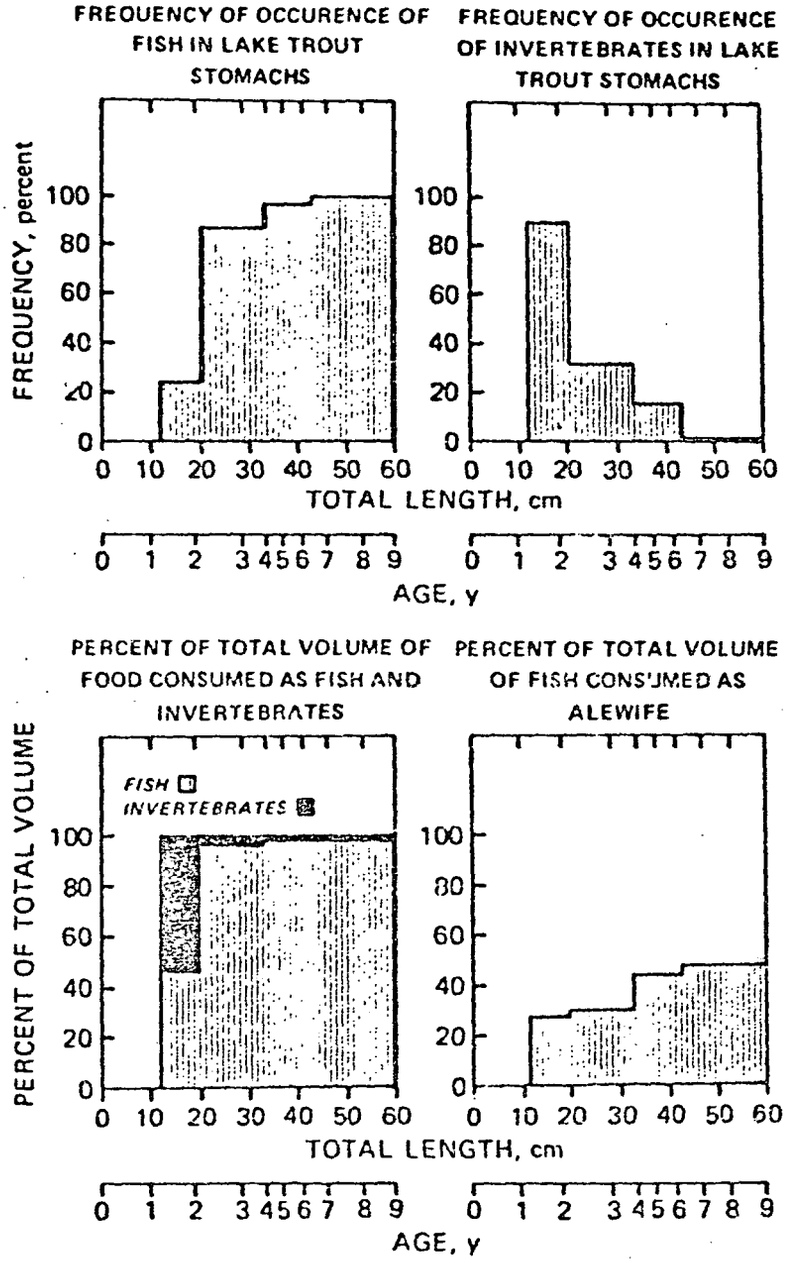

Figure 19. Observed incidence of fish ad invertebrates in lake trout stomachs in relation to lake trout length and computed age.

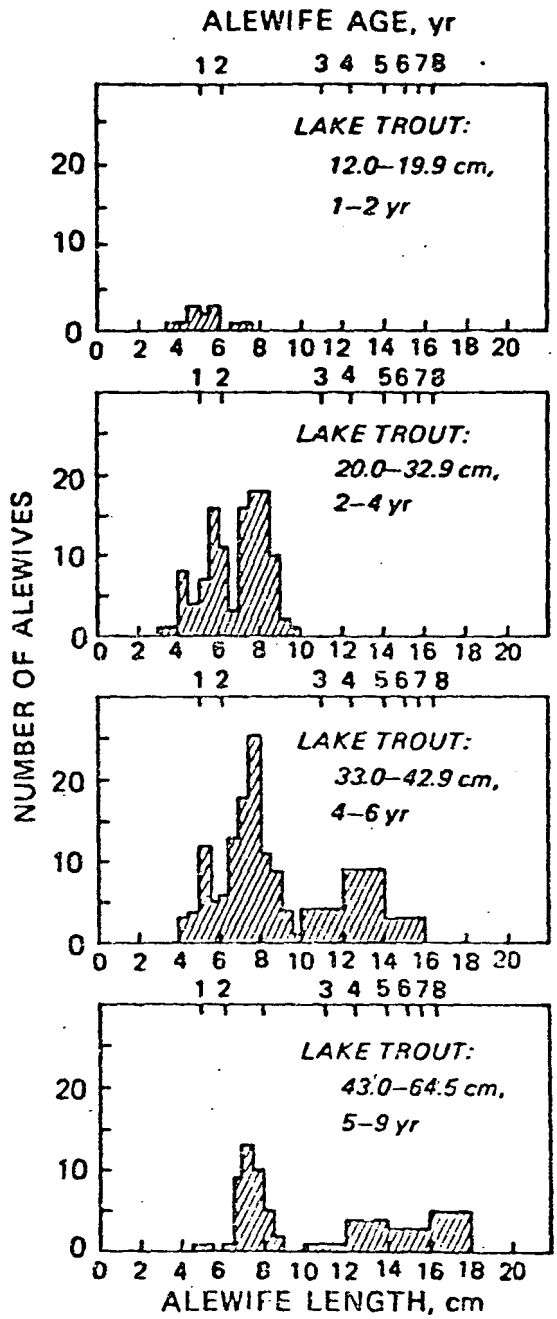

Figure 20. Distribution, by size, of alewives found in various size classes of lake trout.

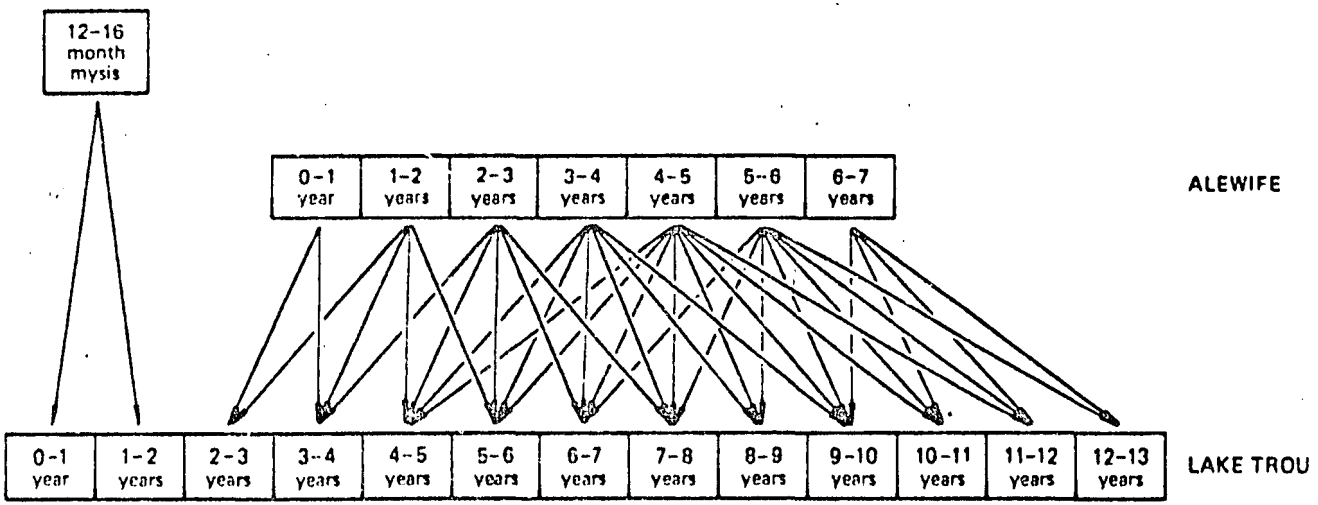

Figure 21. Feeding interactions specified in the model for lake trout.

TABLE 4. FRACTION OF TOTAL LAKE TROUT CONSUMPTION ON EACH MYSIS AND ALEWIFE AGE CLASS

TROUT AGE CLASS	0-4 MO. MYSIS	4-8 MO. MYSIS	8-12 MO. MYSIS	12-16 MO. MYSIS	0-1 YR. ALEWIFE	1-2 YR. ALEWIFE	2-3 YR. ALEWIFE	3-4 YR. ALEWIFE	4-5 YR. ALEWIFE	5-6 YR. ALEWIFE	6-7 YR. ALEWIFE
0-1 yr				1.0							
1-2 yr				1.0							
2-3 yr				0.5	0.25	0.25					
3-4 yr					0.2	0.4	0.4				
4-5 yr						0.2	0.5	0.2	0.1		
5-6 yr						0.2	0.5	0.2	0.1		
6-7 yr							0.25	0.25	0.25	0.25	
7-8 yr							0.25	0.25	0.25	0.25	
8-9 yr								0.25	0.25	0.25	0.25
9-10 yr								0.25	0.25	0.25	0.25
10-11 yr									0.33	0.33	0.34
11-12 yr									0.33	0.33	0.34
12-13 yr										0.5	0.5

$$L_i = L_M(1 - e^{-K_i A}) \tag{47}$$

where

L_i = length at age i
L_M = maximum length
K = growth rate constant
A = age

An age-weight relationship may then be obtained from the relationship between length and weight, w, which is usually of the form (Ricker, 1978):

$$w = a\,L^b \tag{48}$$

The age-weight relationship is usually sigmoidal (logistic) in shape. It is most conveniently modeled by computing instantaneous first-order rates of growth for successive time intervals (Ricker, 1978).

A linear length (L,mm) - age (A,mo) relationship has been observed for Mysis relicta in Lake Michigan (Morgan and Beeton, 1978):

$$L = 3.0 + 0.91A \tag{49}$$

Combining this relationship with an observed wet weight (mg) - length (mm) relationship (Reynolds and DeGraeve, 1972):

$$w = 0.007L^{2.92} \tag{50}$$

yields the weight-age relationship shown by the solid line in Fig. 22. This relationship was approximated in the food chain model by first-order growth rates of 0.0193, 0.0107, 0.0073, and 0.0056/d for the successive 4 month periods indicated by the dashed lines in Fig. 22.

Growth rates for alewives were determined by fitting first-order rate expressions to weight-age data for Lake Ontario alewives reported by Carlander (1969). A rate of 0.00245/d was calculated for the first 4 age classes and a rate of 0.00047/d was calculated for the remaining age classes (Fig. 23).

Figure 22. Wet weight of *Myxis reifiota* in relation to age as calculated from a length-age and weight-length relationship (solid line) and as used in the model (dashed line).

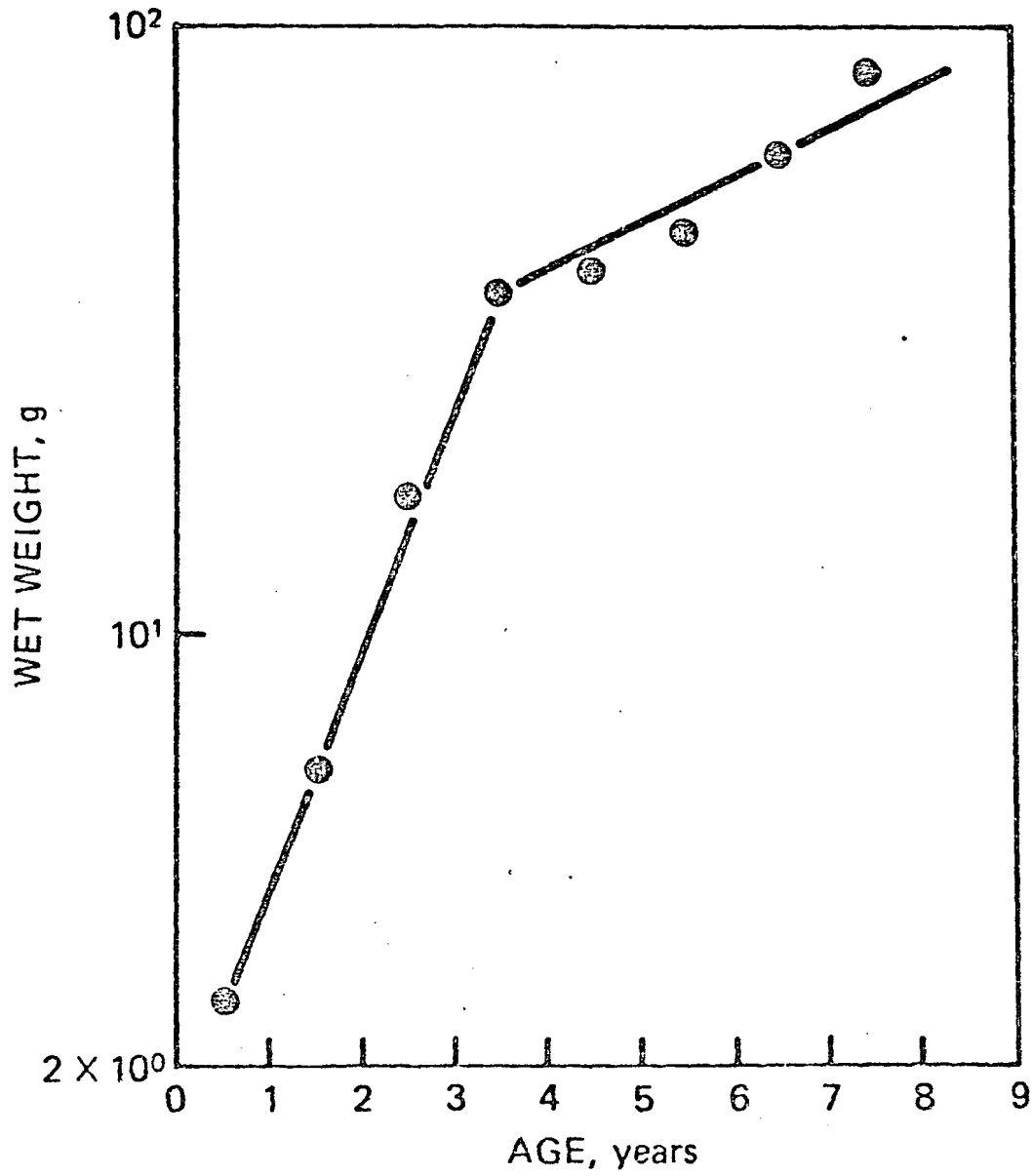

Figure 23. Observed wet weight in relation to alewife age as compared to functional relationship used in the model.

These rates are low in comparison to both Mysis and lake trout and indicate that the alewife is a slow growing fish.

Lake Trout growth rates were calculated using weight-age data for trout from Lake Michigan and Lake Superior that was reported by Carlander (1969). As with alewives, two first-order rate expressions were used (Fig. 24); 0.0058/d for the first two age classes (zero and 1 year olds) and 0.0012/d thereafter. These growth rates are representative of native lake trout since the data used were gathered prior to 1965, the first year of lake trout stocking in Lake Michigan. Hatchery-reared lake trout are faster growing and shorter lived than the native trout (R. Hess, Illinois Dept. of Conservation personal communication).

A recent study of the age-length relationship of fin-clipped (stocked) lake trout (Hess and Muench, 1980) indicates growth rates significantly different than those calculated for the native trout. The weights of juvenile and young adult trout are substantially higher and weights of the last two age classes are lower (Fig. 25). The weights (g(w)) were calculated from total lengths (cm) using a relationship determined from data compiled in STORET:

$$w = 0.00636\ L^{3.1125}$$

Calculated growth rates are 0.0026/d for one to four year olds, 0.00072/d for four to seven year olds, and 0.00024/d for seven to thirteen year olds.

The 1971 PCB data, to which the model is calibrated, includes both native and stocked trout. Because the oldest stocked trout is of year class 1964, all fish that were greater than 7 years old in 1971 must be native trout. Fish 7 years old and younger in 1971 are likely stocked trout since their population is far greater than that of native trout. The calculated growth rates therefore suggest that two non-homogeneous populations of trout are included in the PCB data set. Using the calculated growth rates of native trout will simulate the population represented by the older trout. Because these trout exhibit the maximum PCB concentrations they are the trout calibrated to in this work. It is acknowledged that the stocked trout calculated growth rates are, likely, more representative of the younger trout and a comparison of computed PCB using both estimates of growth rates is shown in Section 8.

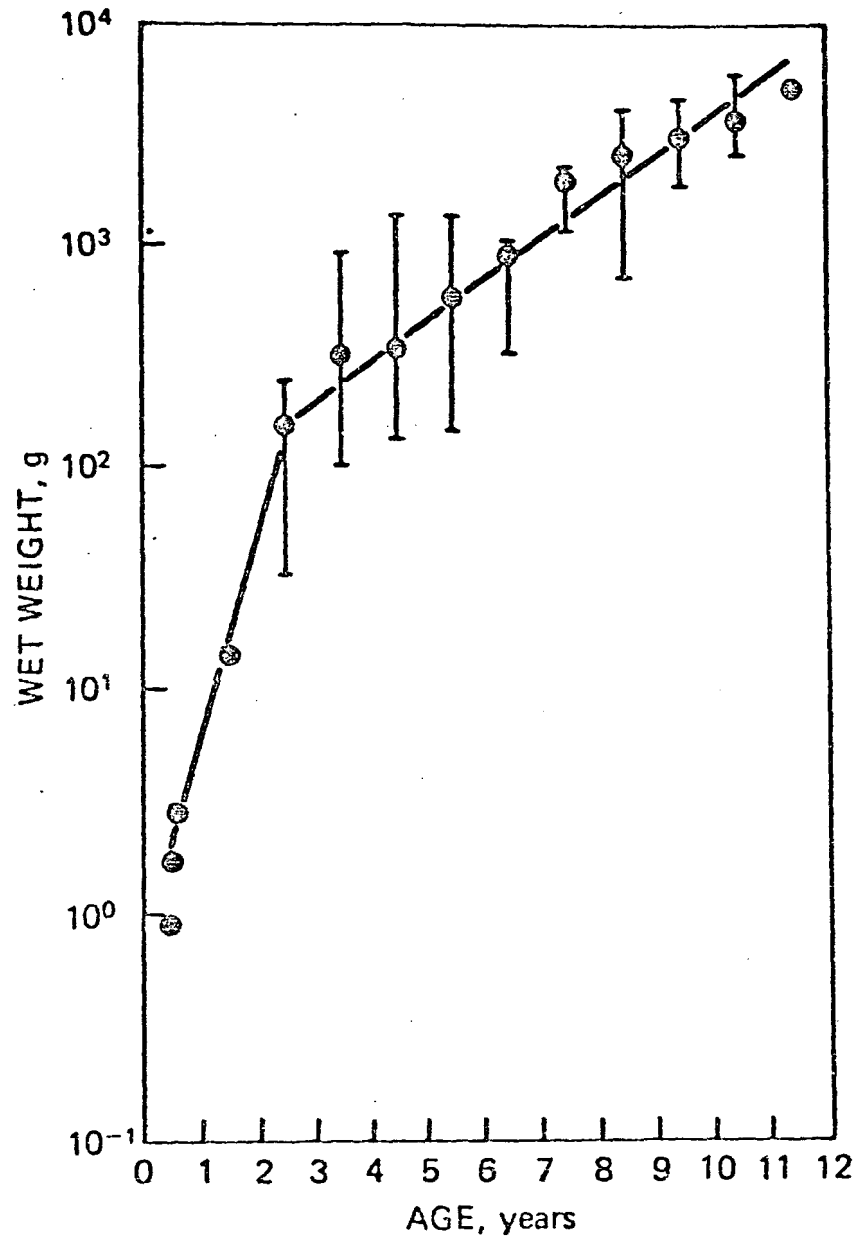

Figure 24. Observed wet weight in relation to lake trout age as compared to functional relationship used in the model.

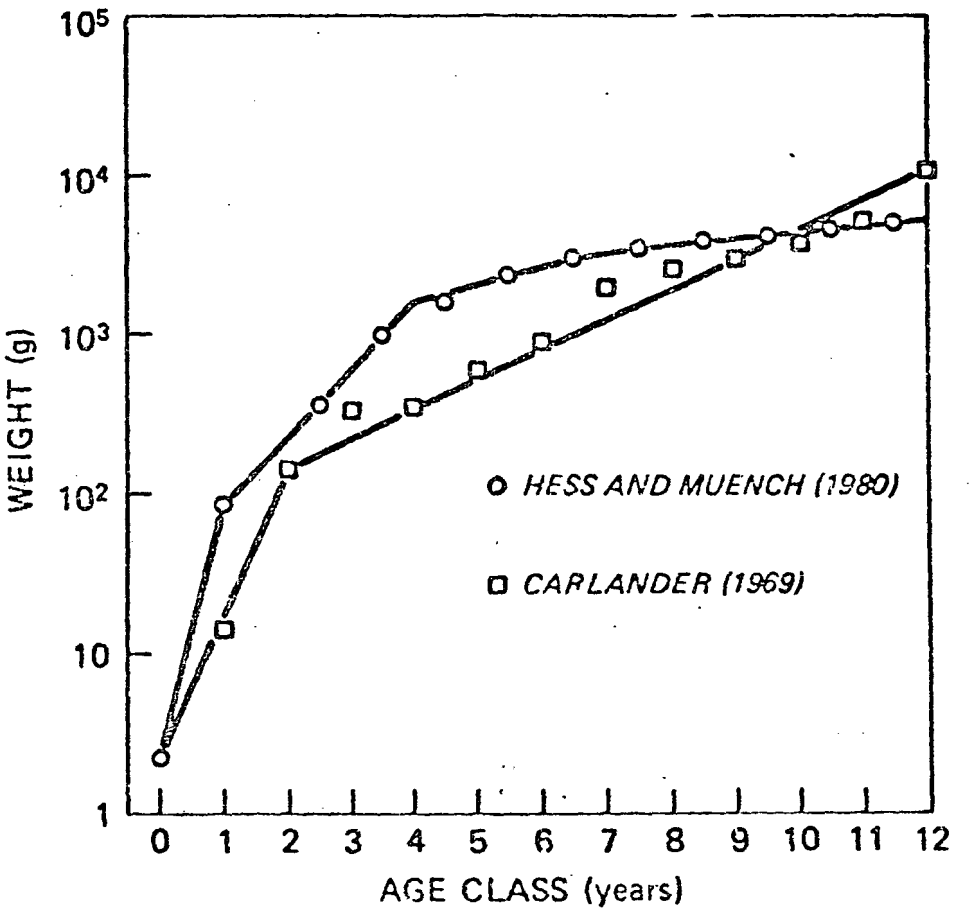

Figure 25. Comparison of lake trout wet weight-age data from Carlander (1969) and Hess and Muench (1980). Lines indicate rates used in the model.

Respiration

Respiration or metabolic rate is composed of the basal metabolic rate (the rate of energy expenditure necessary to sustain life) and the rate of energy usage for activity. The sum of these components, under normal conditions, is termed routine metabolism. Metabolic rate is a function of temperature and size or weight of the individual.

The relationship between metabolic rate and temperature is a complex function of species and environment. For a particular species and environment the logarithm of basal or standard metabolism (the minimum rate observed) generally increases in a parabolic fashion with temperature. Although no single functional relationship is capable of describing the relationship across species, within 10°C of the midpoint of a species normal, environmental temperature range an experimental relationship of the form:

$$R_{s_t} = R_{s_m} e^{0.0833(T-T_m)} \tag{51}$$

where

R_{s_t} = metabolic rate at temperature, T(g(w) respired/day)

R_{s_m} = metabolic rate at midpoint temperature T_m

approximates the temperature effect (Brett and Graves, 1979).

As discussed in Section 5, Eq. (26), the relationship between metabolic rate, R, and body weight, w, is described by the relationship:

$$R = \alpha w^{\gamma} \tag{52}$$

The values of the coefficients α and γ have been the focus of much research with reported values of γ ranging from 0.75 (Kleiber, 1961) to 0.86 (Glass, 1969). A review of the field by Hemmingsen (as cited by Schmidt-Nielsen, 1970) showed that for poikilotherms γ was approximately 0.77.

Routine metabolism of free-swimming Mysis relicta from an arctic and a temperate lake has been determined for various body weights at several temperatures (Lasenby and Langford, 1972). At all temperatures the relationship between respiration and body weight conformed to Eq. 52. None of the γ values calculated by regression differed significantly from the mean value for all tests of 0.75. Respiration increased with temperature, doubling for mysids from the arctic lake as temperature increased from 0°C to 8°C and doubling for the mysids from the temperate lake as temperature increased from 2°C to 13°C. Expressing respiration as mgO_2/hr, the value of α for the temperate lake mysids at 10°C, the temperature used in the food chain model, was 0.000265. Converting to g(wet weight)/d by assuming 40% dry weight as carbon and 0.2 g(dry weight)/g(wet weight) and then dividing by weight the respiration-weight relationship at 10°C may be written as:

$$r' = 0.0157w^{-0.25} \tag{53}$$

where r' is expressed as g/g/d and w is g(w) (see also Eq. (29)).

The respiration-weight relationship applied to alewife in the food chain model was calculated from respiration data reported by Weaver (1975) for alewife in the James River, Virginia. Weaver calculated respiration (cal/d) as the difference between field measurements of food assimilation and growth. Using a caloric density of 1061 cal/g(wet weight) and fixing γ at 0.8 the relationship between respiration (g/g/d) and weight (g(w) was computed to be

$$r' = 0.047w^{-0.2} \tag{54}$$

The value of γ was fixed because only 4 respiration measurements were available and thus using a generally accepted γ value increased the significance of the α determination.

A regression of laboratory measured alewife respiration rates by Stewart (1980) yielded a weight exponent of -0.215 and a temperature and swim speed dependency (see Eq. 55 below) that, at 10°C, results in equivlent α values

for respiration (g/g/d) ranging from 0.0212 to 0.0278 as fish size and swim speed increases. Respiration rates calculated using these values are approximately a factor of 2 lower than those calculated using Eq. 54. Use of these lower rates in the model would require a slight increase in toxicant assimilation efficiency to maintain consistency with observed PCB concentrations in alewife and lake trout.

The respiration of lake trout in relation to body weight has been analyzed by Weininger (1978). Combining the respiration-weight relationship for basal metabolism (Eq. 52) with a functional representation of temperature effects and of activity in terms of swimming speed, Weininger presented the following general relation:

$$R = \alpha w^{\gamma} e^{\beta T} e^{\phi V} \tag{55}$$

where

T = temperature (°C)
V = swim speed (cm/sec)
β, ϕ = empirical constants

Regression using an extensive data set of 183 respiration experiments yielded an equation that when swim speed is converted to trout length, L(cm), using the empirical relation (Weininger, 1978):

$$V = 0.46L, \tag{56}$$

respiration is expressed as g/g/d, and temperature is set at 10°C, may be written as

$$r' = 0.03w^{-0.295} e^{0.01L} \tag{57}$$

In the model length, L, is determined from the weight-length relationship (Weininger, 1978):

$$\ln(L) = 0.296\ln(w) + 1.803 \tag{58}$$

Calculated respiration in relation to body weight for Mysis, alewife, and trout is shown in Fig. 26. Also shown is a general relationship based on data for many species of freshwater fish (Winberg, 1956). Lake trout respiration levels off at high body weight because of the increasing contribution of swimming activity to respiration.

Assimilation Efficiency

The assimilation efficiency of food, as specified in the model, is the fraction of food ingested that does not appear in the feces. It is a function of the type of food eaten and the rate of consumption. Elliott (1976) found that the assimilation efficiency of brown trout eating Gaunmarus pulex varied from 0.86 at 10% of maximum consumption rate to 0.77 at maximum consumption rate. He also found that efficiency increased with temperature and was independent of fish body weight. In general, assimilation efficiency in carnivorous fish ranges from about 0.75 to 0.9 (Brett and Graves, 1979). The digestibility of plant life (algae, grasses) is comparatively low and herbivorous fish consequently have lower assimilation efficiencies. A summary by Brett and Graves (1979) indicates a range of 0.16 to about 0.65. Based on this information the assimilation efficiencies in the model were set at 0.8 for lake trout and alewife and 0.3 for Mysis.

The food ingestion rate (g/g/d) used in the model is computed as the sum of the growth and respiration rates normalized by the food assimilation efficiency. Fig. 27 shows the ingestion rate calculated for each species in relation to body weight. The discontinuities that are seen are a consequence of the change in inputted growth rate with age. Also shown are laboratory observed ingestion rates for Mysis relicta (Cooper and Goldman, 1980) and a field observation of alewife ingestion rate (Weaver, 1975). The fit of the computed rates to the data further supports the values used in the model for growth, respiration and assimilation.

PCB PARAMETERS

The concentration of PCB in the bodies of Mysis, alewife, and lake trout is controlled by the rates at which PCB is adsorbed, metabolized, and excreted. Each of these rates is a function of the species bioenergetics.

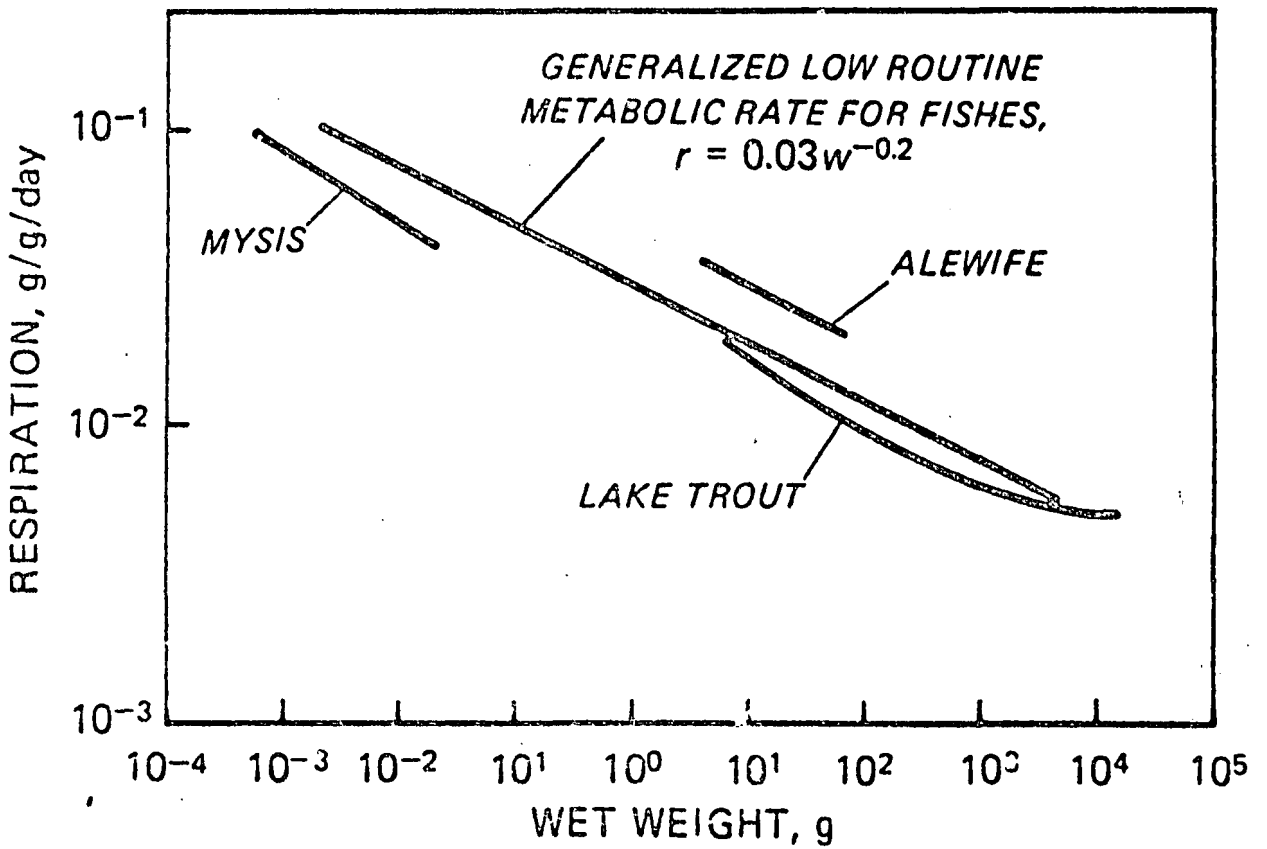

Figure 26. Calculated respiration rates used in the model for *Mysis*, alewife, and lake trout in relation to wet weight as compared to a general relationship based on data for many species of freshwater fish.

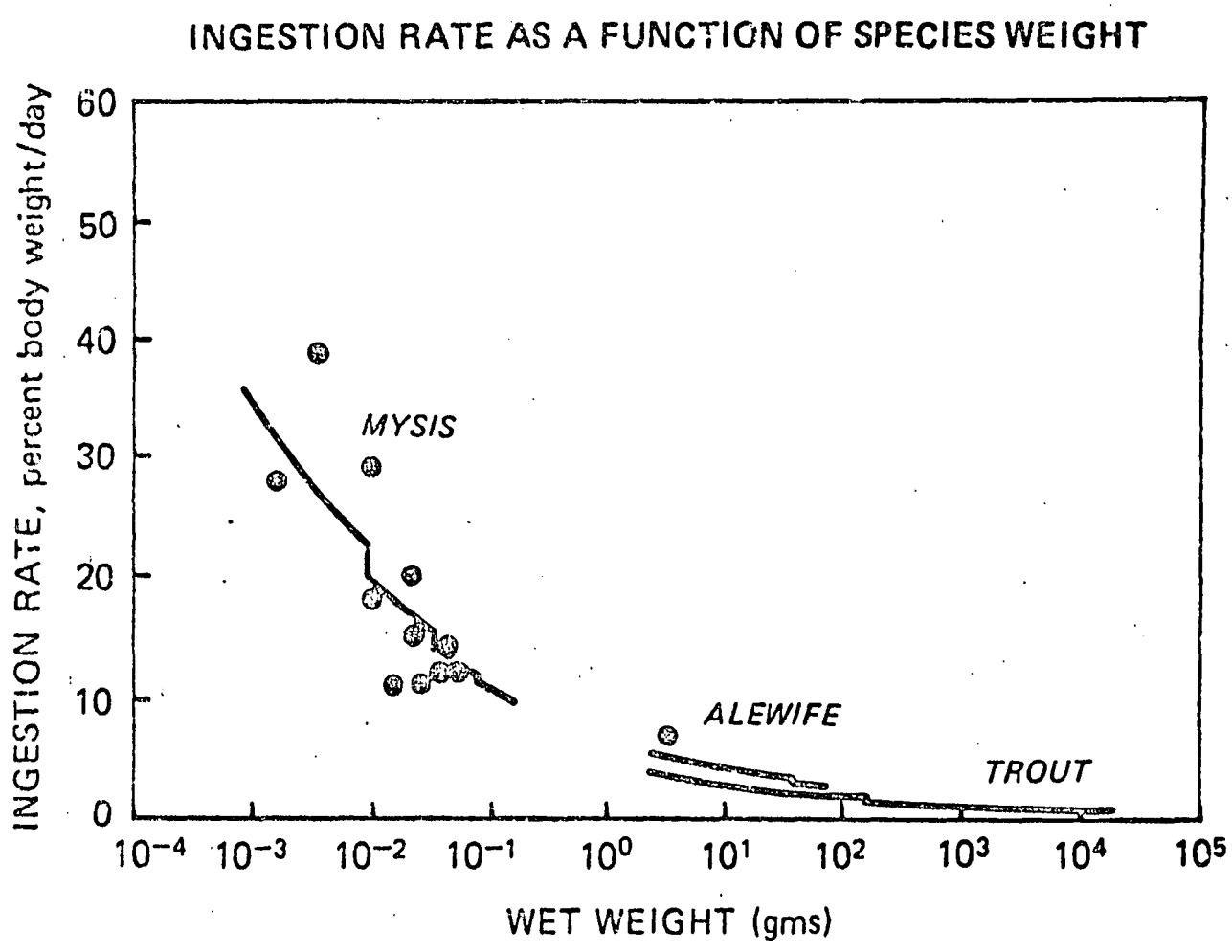

Figure 27. Comparison between observed and computed ingestion rates in relation to wet weight.

Absorption occurs at the gill membrane, skin, and digestive tract. The skin is normally not significant in this regard and is not considered here. Adsorption through the gill membrane is assumed to be proportional to respiration as described in Section 5. Absorption through the lining of the digestive tract is computed as the product of the ingestion rate of PCB (food ingestion rate x μg PCB/gfood) and an assimilation efficiency for PCB.

The assimilation efficiency is the fraction of ingested contaminant that is passed across the gut lining into the organism. It is normally computed as the ratio of the sum of the measured mass of contaminant in a test organism and the mass excreted to the mass ingested. However, it is difficult to accurately determine the mass excreted because significant contaminant must be excreted before measurable dissolved contaminant concentrations are reached. As an alternative, excretion rates determined from depuration studies may be used to estimate the mass excreted if contaminant concentrations in the organism are measured over the duration of feeding.

Another method to determine assimilation efficiency is suggested from the equilibrium condition described by Eq. (40). If there is no uptake from water, as in most experiments designed to determine assimilation efficiency, this equation may be written as:

$$\nu_i = \frac{\alpha_{i,j} C_{i,j} \nu_j}{K_i + G_i} \tag{59}$$

Solving for the assimilation efficiency yields:

$$\alpha_{i,j} = \frac{\nu_i (K_i + G_i)}{C_{i,j} \nu_j} \tag{60}$$

Thus if the feeding rate, $C_{i,j}$, the food concentration, ν_j, the excretion rate, K_i, and the growth rate, G_i, are known then the assimilation efficiency may be calculated from the equilibrium concentration of the organism under study.

No assimilation efficiencies of PCB by lake trout, alewife, or *Mysis* have been reported. The assimilation efficiency for PCB Aroclor 1254 in synthetic

food by rainbow trout has been reported to be 0.68 based on the retention of ingested PCB over a 32-week test period, assuming no excretion (Lieb et al., 1974). No significant excretion was detected during a 16-week depuration study. From a study of short-term PCB Aroclor 1254 uptake from PCB-contaminated phytoplankton (Wyman and O'Connors, 1980), the PCB assimilation efficiency of the estuarine copepod Acartia tonsa was calculated to be approximatly 0.2. This low efficiency relative to the rainbow trout is consistent with the food assimilation efficiencies discussed previously in this section. The PCB assimilation efficiencies used in the model are 0.55 for Mysis, 0.7 for alewife, and 0.8 for lake trout. These values were determined by calibrating the model to observed PCB in lake trout and alewife. Assimilation efficiency was varied during the calibration because of the uncertainty associated with it.

Metabolism of PCB is not significant. The composition of PCB residues in rainbow trout (Lieb, et al., 1974) and fathead minnows (DeFoe et al., 1978) was found to be essentially identical to the composition of PCB used for the experiment.

Excretion of PCB may be estimated from the ratio of PCB concentration in the body of a particular species to the concentration of PCB the species is exposed to in a non-feeding test, i.e., the bioconcentration factor (See Equation (10)). If growth is not significant during the test because of a low or zero feeding rate then Eq. (10) simplifies to

$$N = \frac{K_u}{K} \tag{61}$$

The bioconcentration factor for PCB has been shown to be nearly constant over a wide range of aquatic species at a value of approximately 10^5 ($\mu g/g_{dry}$: $\mu g/g_{water}$) (Thomann, 1981). Using this value and the uptake rate computed from the respiration rate (Eq. (18)) a value for excretion rate may be calculated directly. Fig. 28 compares the resulting excretion rates with rates measured in the laboratory for various species. Note that no excretion rate data is available for large fish such as the adult lake trout. The values used in the model are within the reported ranges. The data in Fig. 28 indicate significant variability of excretion rate. Much of this variability is

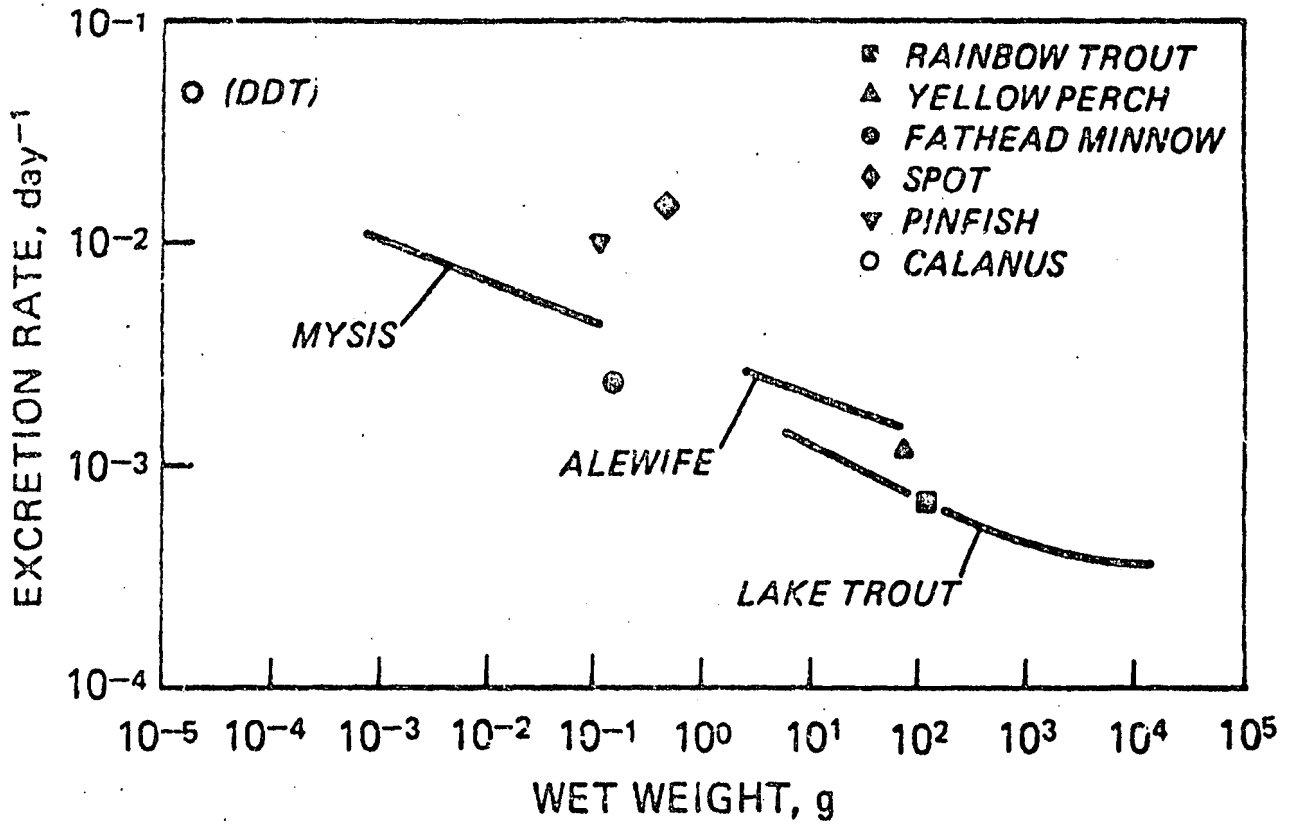

Figure 28. Comparison between observed and computed excretion rates of PCB in relation to wet weight.

likely analytical error. Because PCB excretion rates are low the decrease in body burden during most excretion experiments is less than the uncertainty of the PCB measurement.

SECTION 7

MODEL CALIBRATION

The final calibration is the result of a series of model runs that determined a consistent set of parameter values that were in agreement with observed values and reproduced the observed PCB concentrations in Lake Michigan lake trout and alewife. Data for 1971 were used in the calibration. A constant dissolved PCB concentration of 5 ng/ℓ was assumed. A constant value implies that the alewife and lake trout sampled in 1971 were exposed to a constant PCB concentration for their entire lives, which for the oldest trout represented is ten years. A time variable dissolved PCB concentration was not used because no accurate data history exists. The values assigned to the PCB assimilation efficiency were adjusted to reproduce the observed PCB distribution. This parameter was chosen as the calibration variable because of the uncertainty of its value relative to the other parameters in the model. Sensitivity between adjusting the PCB assimilation efficiency and the dissolved PCB concentration was tested and is discussed in Section 8.

The basic model presented in Section 5 and a model including an empirical function relating lake trout excretion rate to lipid content were calibrated to the data. The lipid-dependent excretion was included because of the observed relationship between lake trout PCB concentration and percent of body weight as lipid. (See Figure 6).

BASIC MODEL

The comparison between observed data and calculated PCB concentrations in alewife and lake trout is shown in Fig. 29. The parameter values used in the model and presented in Section 6 are summarized in Table 5. The model reproduces the observed data with the exception of the early age classes of lake trout. No combination of parameters was successful at reproducing the high PCB values in age class 2 and 3 lake trout while maintaining consistency with reported parameter values and reproducing the observed concentrations in the

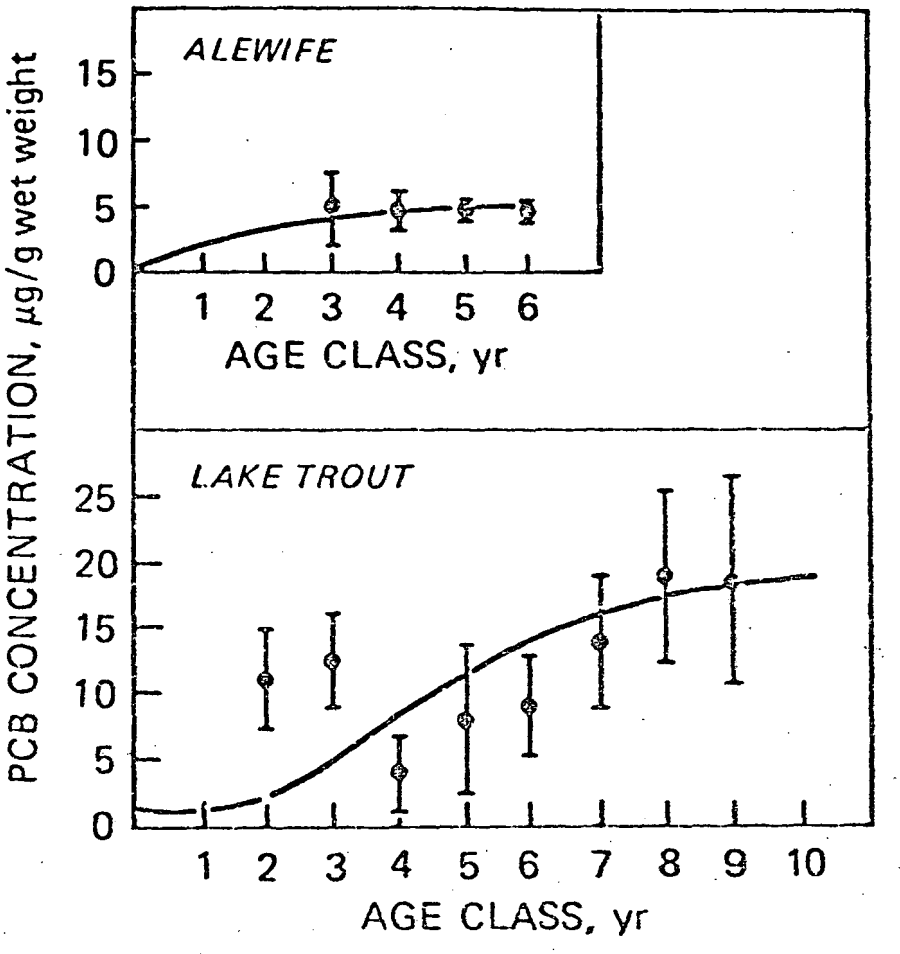

Figure 29. Comparison of observed PCB concentrations in alewife and lake trout with calculated concentrations from basic model.

TABLE 5. PARAMETER VALUES USED IN CALIBRATION OF MODEL

Parameter	Phytoplankton	Mysis	Alewife	Lake Trout
Growth Rate (d^{-1})	-	0.0193 - Age Class 0 0.0107 - Age Class 1 0.0073 - Age Class 2 0.0056 - Age Class 3	0.00245 Age Class 0-3 0.00047 Age Class 4-6	0.0058 Age Class 0-1 0.0012 Age Class 2-12
Respiration Rate α	-	0.0157	0.047	0.03
(g/g/d) γ	-	-0.25	-0.20	-0.295
($R = \alpha W^{\gamma} e^{\phi V}$) ϕ	-	0	0	0.022
Food Assimilation Efficiency	-	0.3	0.8	0.8
PCB Assimilation Efficiency	-	0.35	0.7	0.8
Bioconcentration Factor from Water ($\mu g/g_d \div \mu g/g_{water}$)	2×10^5	2.5×10^5	4×10^5	4×10^5

upper age classes. A possible explanation of these high values is that young trout may be exposed to higher dissolved PCB concentrations than older trout because of a tendency to stay in shallow near-shore waters. This effect is not simulated by the model because of the assumed constant dissolved PCB concentration.

The data and the model both indicate that PCB concentrations in lake trout are 3 to 4 times those in alewife. The computed increase results from the higher PCB concentration in lake trout prey (alewife) relative to alewife prey (Mysis). All species in the model accumulate similar PCB concentrations from dissolved PCB because of the constant relationship maintained between uptake rate from water and excretion rate (i.e., constant BCF). Some variation does occur depending on the influence of growth rate as a concentration loss mechanism. The increase in computed PCB concentration with increasing trophic level (Fig. 30) is a direct consequence of the transfer of accumulated PCB from one level to the next. Section 5 discusses this effect in a simple steady state situation.

As a further illustrative example, using values consistent with the preceding discussion, consider a 4 level food chain (see Eq. 45) in which each level has a bioconcentration factor from water of 10^5 μg/g ÷ $\mu g/g_{water}$ and a bioconcentration factor from food of 0.3×10^1 μg/g ÷ μg/g. As in the Lake Michigan PCB calibration assume that the water concentration is 5 ng/ℓ and that the first level of the food chain is exposed only through water thus achieving a concentration of 0.5 μg/g. The second level receives the same 0.5 μg/g from water but also receives 1.5 μg/g from food since it bioconcentrates by a factor of 3 the 0.5 μg/g accumulated by the first level. It then has a concentration of 2 μg/g. In a similar manner the third level accumulates 0.5 μg/g from water and 6 μg/g from food and the fourth level accumulates 0.5 μg/g from water and 19.5 μg/g from food.

From this example it is evident that even for small accumulation factors from food there is a significant geometric increase in the concentration achieved by ascending levels of the food chain. Also evident is the increase in importance of uptake from food at ascending levels of the food chain. At the first level 100% of the accumulated concentration is from water while by

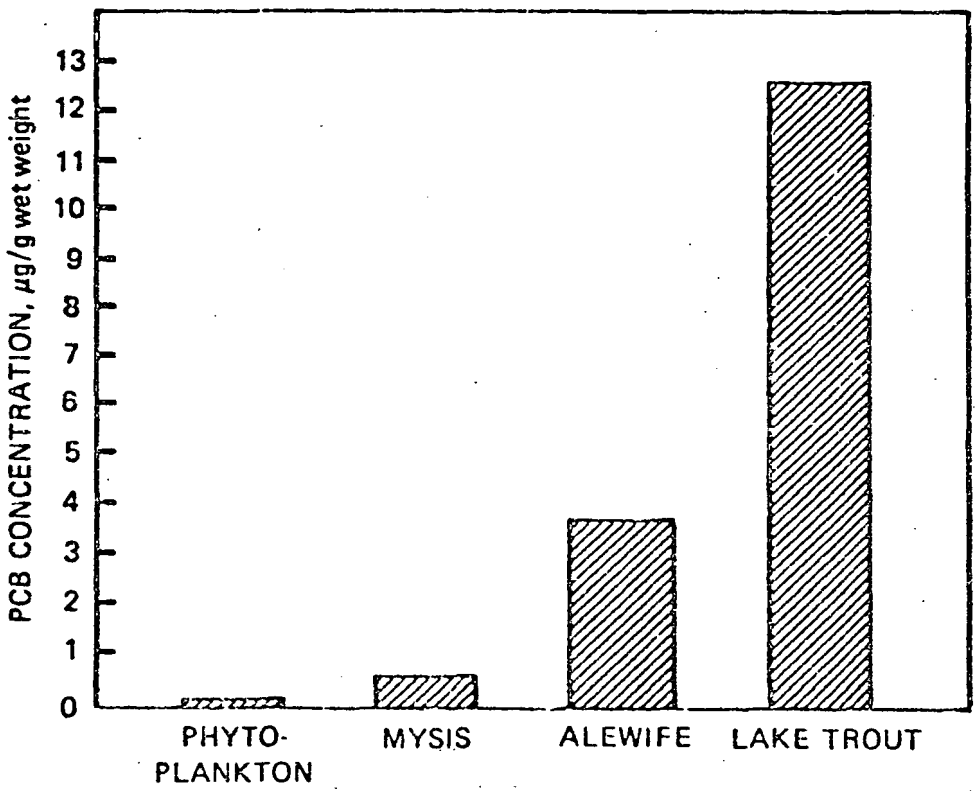

Figure 30. Average PCB concentration in each of the 4 species included in the food chain model. Values indicated are arithmetic average over all age classes.

the fourth level only 2.5% of the accumulated concentration is from water. The geometric increase shown by this simple example is also computed by the calibrated food chain model as illustrated in Fig. 30.

The relative importance of food and water to the computed PCB accumulation by lake trout is illustrated in Fig. 31. The PCB accumulated from water is relatively constant of all age classes at a wet-weight BCF of approximately 2×10^4. Including food results in over an order of magnitude higher BCF for juvenile trout increasing with the concentration in the prey to an approximately 2 order of magnitude higher BCF for the oldest trout. Expressing PCB accumulation in terms of concentration, Fig. 31 shows that greater than 99% of the PCB in adult trout is taken up in food. The contributions of food and water to the lake trout calibration are shown directly in Fig. 32. Again, the overwhelming influence of the food component is evident. Considering the other species or levels in the model the average fraction of total PCB concentration that is contributed from food is 0 for phytoplankton, 0.5 for *Mysis*, and 0.85 for alewife.

LIPID-DEPENDENT MODEL

Empirical evidence indicates that the extent of accumulation of organic chemicals by aquatic species in laboratory studies is related to the lipophilic nature of the chemicals. The lipophilic nature is normally expressed as the equilibrium concentration ratio of the chemical partitioned between noctanol and water, i.e., the octanol-water partition coefficient. A linear relationship between the log of the BCF and the log of the octanol-water partition coefficient for a wide range of chemicals has been demonstrated for rainbow trout (Chiou, et al., 1977) and fathead minnows (Veith, et al., 1979).

The lipophilic nature of a chemical controls its rate of transport across the cell membranes of the gut and gills and its partitioning between the blood and lipid tissue of an exposed species. The former affects the assimilation efficiency of the chemical and the latter affects the apparent excretion rate of the compound because the sequestering of chemical in the lipid tissue slows the rate at which the chemical reaches excreting organs. The BCF then increases with the lipophilic nature of the chemical because of an increased uptake rate and a decreased excretion rate.

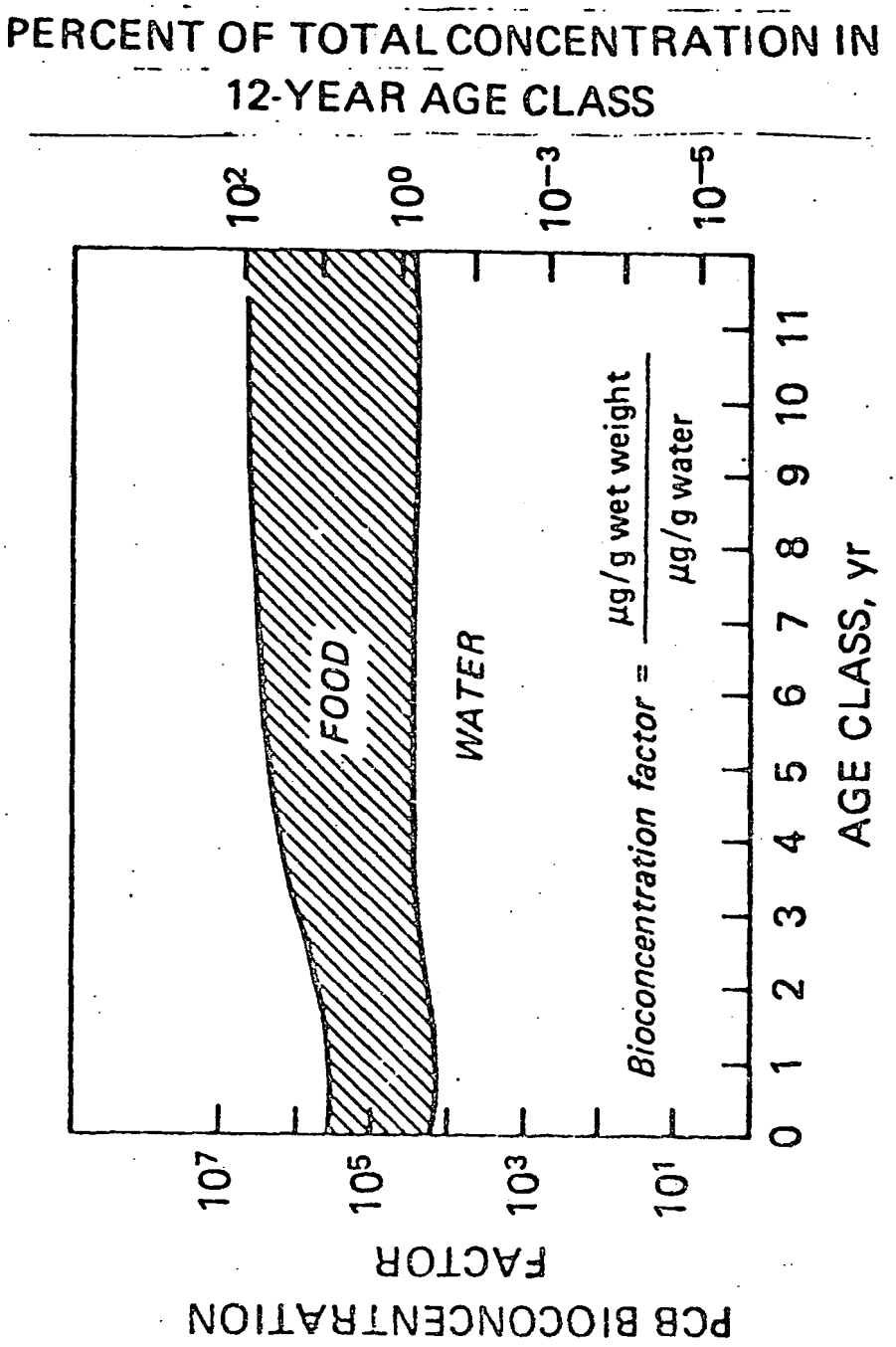

Figure 31. Computed PCB concentration factor for lake trout due to PCB uptake from water and from water and food.

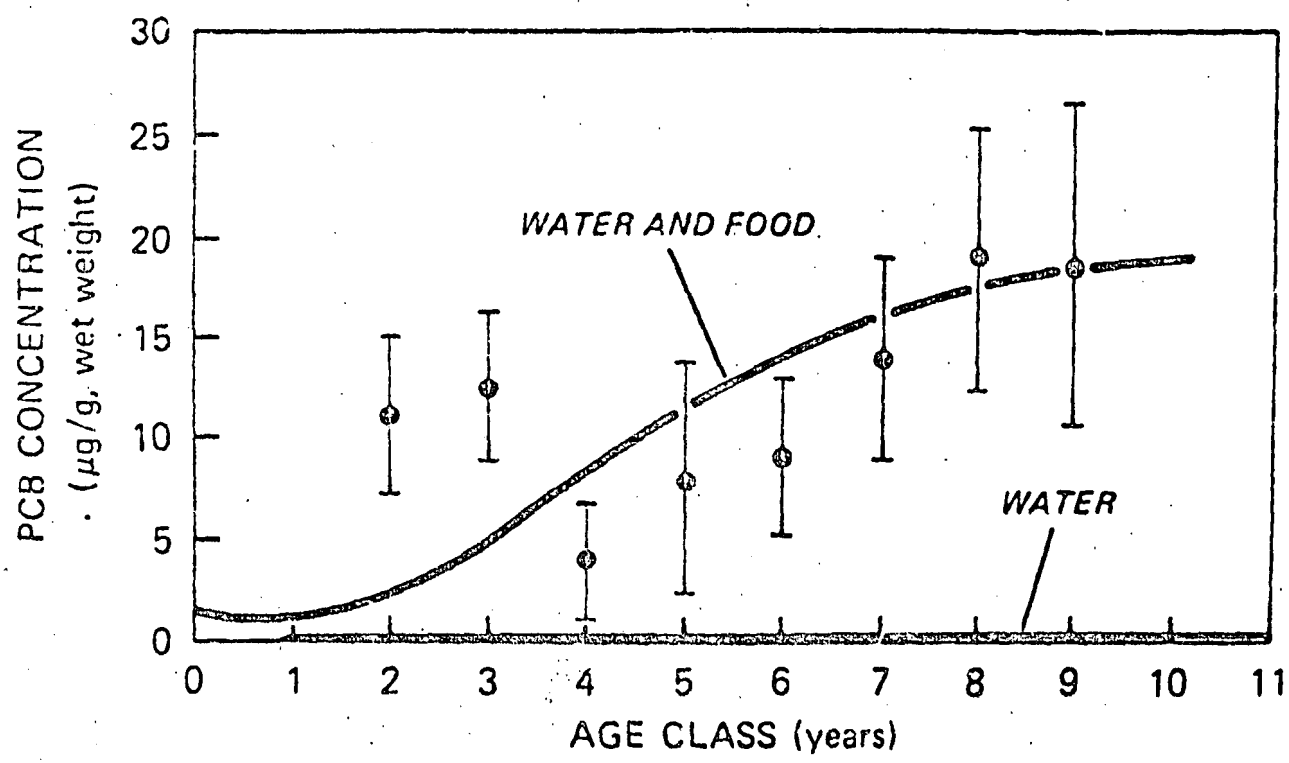

Figure 32. Computed PCB concentration in lake trout due to PCB uptake from water and from water and food.

The mechanisms that relate the BCF and lipophilic nature of the chemical also suggest that the BCF should be related to the lipid content of the test species. De Foe et al. (1978) found that the BCF of PCB Aroclor 1248 and PCB Aroclor 1260 in fathead minnows was directly related to the lipid content of the fish and that a nearly constant BCF was obtained when PCB concentrations were expressed on a lipid basis ($\mu g/g_{lipid}$). Roberts et al. (1977) reported that tissue retention of the insecticide chlorodane by northern redhorse suckers was directly proportional to the lipid content of the fish. In addition, their data indicated that the assimilation efficiency of chlordane was directly related to the percent of body weight as lipid while excretion rate was inversely related.

The high lipid-solubility of PCBs suggests that accumulation of PCBs should be correlated with the lipid content of the exposed species. The analysis of data presented in Section 4 indicates that across thirteen species there is no apparent relationship between PCB concentration and % lipid. However, for the lake trout the data does indicate increasing PCB concentration with increasing lipid content.

To test if an improvement of the lake trout calibration would result, excretion rate was formulated as an empirical function of the lake trout lipid content. Lipid content in relation to lake trout age (Table 6) was compiled from the PCB data sources for fishes of Lake Michigan listed in Section 4. An inverse power-law relationship between excretion rate and % lipid was chosen and the coefficients of the relationship determined by calibration of the model to the lake trout PCB data. The resulting relationship is:

$$K = \frac{K'}{6 \times 10^{-6}(\text{lipid})^5} \tag{62}$$

where

K' = excretion rate used in the basic model.

TABLE 6. LAKE TROUT LIPID CONTENT DATA COMPILED IN RELATION TO AGE CLASS

Lake Trout Age Class	Mean % Lipid	Std. Deviation	Max.	Min.	No. of Points
2	8.6	3.3	14.8	3.2	8
3	7.0	3.4	13.2	1.1	10
4	6.4	2.9	12.6	3.5	16
5	6.8	3.5	16.6	2.1	38
6	10.8	4.9	21.3	1.8	154
7	16.3	5.8	50.0	4.0	242
8	18.1	7.0	36.2	6.1	73
9	16.3	9.3	49.2	1.5	70
10	13.6	5.5	45.0	5.7	10

With this formulation an improvement of the lake trout calibration was achieved (Fig. 33). The model still does not reproduce the early age class data but does very well at fitting the shape as well as the magnitude of the later data. This result indicates that lipid content is important to the dynamics of PCB accumulation in lake trout. The empirical relationship between excretion and lipid used here has no fundamental basis but is important because it points out the significance of lipid content. A more mechanistic formulation would separate each lake trout age class into lipid and non-lipid components.

The demonstrated importance of lipid content and the observed increase in lipid content with lake trout age suggests the possibility that the increasing PCB concentration with lake trout age may be explained by partitioning from water to lipid rather than uptake from food. Some evidence, in fact, indicates that field observed contaminant concentrations may be directly estimated from water concentrations using lipid content and the octanol-water partition coefficient of the contaminant (Schnoor, 1980). To test this possibility, the highest reported octanol-water partition coefficient available in the literature ($10^{6.72}$ for HCB) was used with the lake trout lipid content to predict lake trout PCB concentration. Following Schnoor (1980) the octanol-water partition coefficient was assumed to be equivalent to the lipid-based BCF (μg/

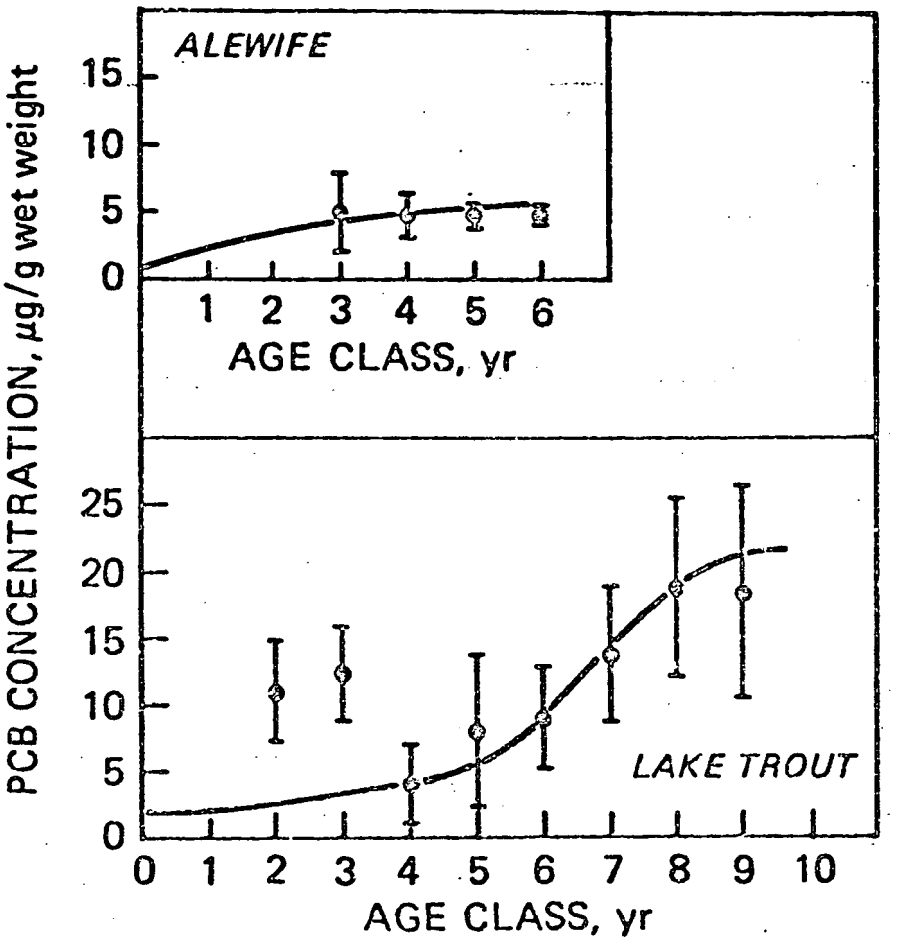

Figure 33. Comparison of observed PCB concentrations in alewife and lake trout with calculated concentration from lipid-dependent model.

$g_{lipid} \div \mu g/g_{water}$). The resulting PCB concentrations (Fig. 34) calculated from the equation:

$$\nu = BCF_L * c * f_L$$

where

f_L = fraction of body weight as lipid (g_{lipid}/g_w)

were 4 to 5 times lower than the data and food chain model calculation for adult trout and clearly an unsatisfactory estimation of lake trout contamination. The poor fit results from the failure to consider exposure through food which, as shown earlier, is the dominant contributor of PCB to the top predator lake trout. Similarly for the alewife, the simple lipid partitioning calculation yields concentrations 2 to 4 times lower than the observed data (Fig. 35). This calculation is closer to the data than the lake trout calculation possibly because the importance of contaminated food to body burden is less in the lower trophic level alewife. It should be noted that the uncertainty in the dissolved PCB concentration is such that the actual concentration may be as high as 10 ng/ℓ, i.e., twice the concentration used here, thus doubling the fish PCB concentrations calculated by simple lipid partitioning. These high calculated values would come close to observed alewife concentrations but would still be significantly below observed lake trout concentrations.

The fact that satisfactory results have previously been achieved with the simple octanol-water BCF approach may reflect the different focus of this method relative to the work presented here. The octanol-water BCF approach is based on correlation of data over a range of chemicals, fish species and water bodies. Interpretations are made on general trends that include a significant amount of statistical variability. Estimation to within an order of magnitude of observation is considered satisfactory. In contrast, this work is focused on a finer scale where accurate prediction is necessary to determine the impact of a specific chemical, on a specific fishery, in a specific water body. While the former method is valuable in assessing trends, the latter is necessary to answer specific questions about the management and fate of a given system.

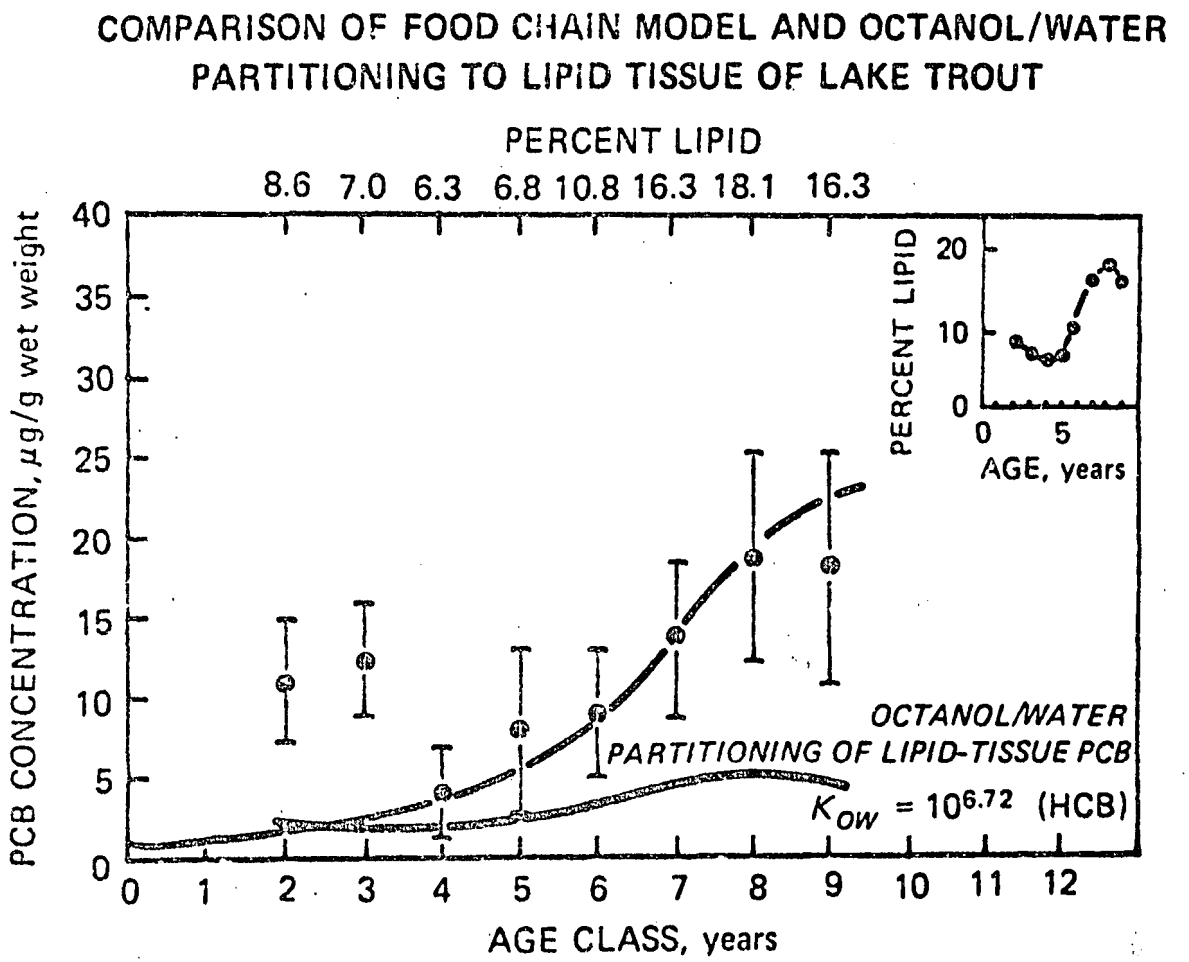

Figure 34. Comparison of food chain model and octanol/water partitioning calculation.

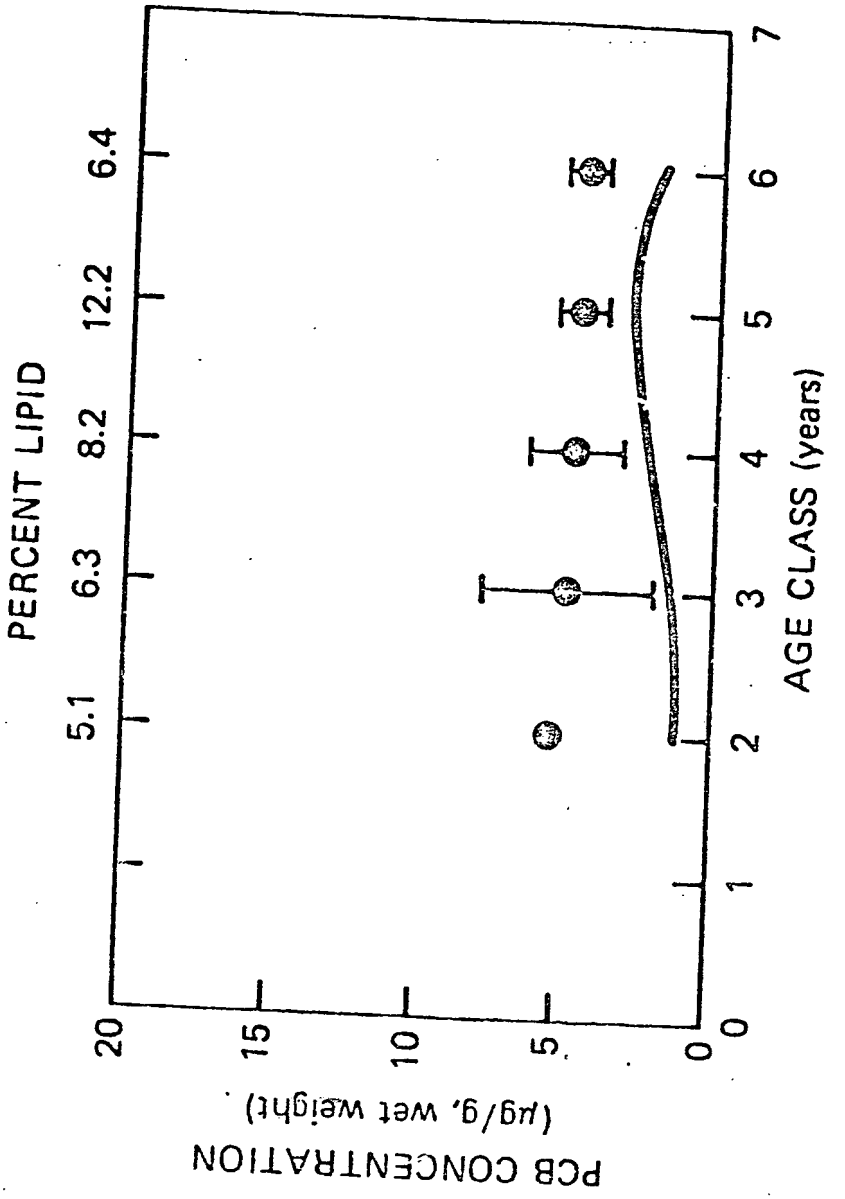

Figure 35. Computed PCB concentration in alewife from octanol/water partitioning calculation.

CONCLUSIONS

The calibration of the age-dependent food chain model to data from Lake Michigan illustrates several points:

1. the model is capable of reproducing the age-dependent trends and magnitude of PCB contamination observed in Lake Michigan alewife and lake trout in 1971.

2. transfer of PCB through the food chain is the major contributor to observed PCB concentrations in lake trout, accounting for greater than 99% of the body burden in adult trout.

3. an empirical relationship between lake trout excretion rate and lipid content significantly improves the lake trout calibration, suggesting that lipid tissue is an important factor in PCB dynamics.

4. a simple empirical correlation between octanol-water partitioning of PCB and bioconcentration of PCB fails to reproduce the observed concentrations in alewife and lake trout.

SECTION 8
MODEL SENSITIVITY

Of the parameter values chosen for calibration of the model to the alewife and lake trout PCB data, the values for PCB assimilation efficiency, PCB excretion rate, and dissolved PCB concentration are the least well known. It is thus pertinent to examine the effect of variation of these parameter values on the PCB concentrations computed by the model.

In addition, because the food chain structure leading to a top predator such as the lake trout is more complex than that of other species of sport or commercial interest, it is useful to examine the effect of a simpler structure on the computed concentrations.

This significance of the differences in lake trout growth rate that are discussed in Section 6 is also examined.

SENSITIVITY OF MODEL TO PCB ASSIMILATION EFFICIENCY

Very little data are available that defines the assimilation efficiency of toxicants by fish. As cited earlier, Lieb, et al. (1974) reported that the PCB Aroclor 1254 assimilation efficiency of rainbow trout was approximately 0.68. An analysis of this data by Weininger (1978) showed that for fish greater than 5.5 g, the assimilation efficiency ranged from 0.64 to 0.79. In a previous modeling study (Thomann, 1981) the PCB assimilation efficiency of lake trout was assumed to 0.9.

A comparison of computed lake trout PCB concentrations at lake trout PCB assimilation efficiencies of 0.95, 0.8 and 0.65 and observed PCB concentrations is shown in Fig. 36. The high and low assimilation efficiencies bracket the estimates noted above and 0.8 is the value used in the model calibration. Computed concentration is approximately 4 μg/g lower at the upper age classes when the assimilation efficiency is 0.65. The results at the 0.95 assimilation efficiency reproduce the observed data for the middle age classes fairly well.

It is significant that the computed values at assimilation efficiencies

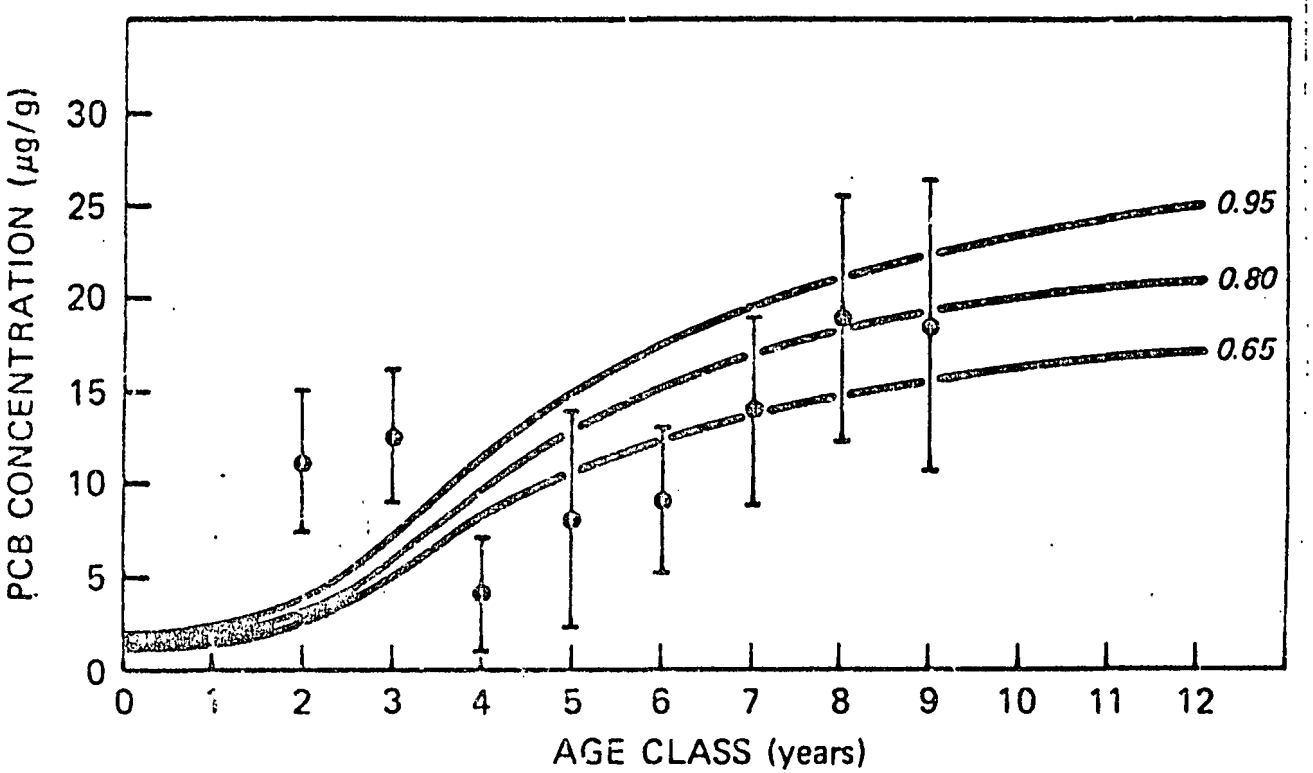

Figure 36. Sensitivity of computed lake trout PCB concentration to assimilation efficiency of PCB in food.

0.65 and 0.8 both fall within the observed data since the available experimental data indicate that 0.65 to 0.8 is the assimilation efficiency range. The μg/g difference at upper age classes between the two computations, however, is substantial because it will effect estimates of the efficiency of PCB load reductions in lowering lake trout PCB concentrations to below the FDA action limit of 5 μg/g. This variability therefore must be considered in management decisions based on the model's results.

SENSITIVITY OF MODEL TO LAKE TROUT PCB EXCRETION RATE

The PCB excretion rates used in the calibration are calculated from observed BCFs assuming that the BCF represents the ratio of uptake to excretion rate (see Eq. 105)). As shown in Section 6 the resultant values compare well with rates reported in the literature. However, there is substantial variability in the reported rates and some literature data indicate that the PCB excretion rate is effectively zero (DeFoe et al., 1978; McLeese et al., 1980). To examine the influence of this variability on the computed PCB concentrations the excretion rate for lake trout was altered from the values used for calibration to zero.

A comparison of computed PCB concentrations in lake trout that have excretion rate specified as zero with computed PCB concentrations from the calibration is shown in Fig. 37. Reducing lake trout excretion to zero increases computed concentrations in adult trout by approximately 3 μg/g or 13 percent. Because this increase results in a less satisfactory fit of the model to observed data, it suggests that either zero excretion rate is unreasonable or that other parameter values in the model are incorrectly specified.

As discussed earlier, PCB assimilation efficiency is not well defined and for lake trout could vary at least from 0.65 to 0.8. Also presented were the results of lowering the value of lake trout assimilation efficiency from 0.8 to 0.65, i.e., a reduction in adult trout PCB concentrations of approximately 4 μg/g or 17 percent. This reduction would more than compensate for the increase resultant from assignment of zero excretion rate. Therefore, it is concluded that the uncertainty in assimilation efficiency precludes a judgement as to the most appropriate of the reported excretion rates.

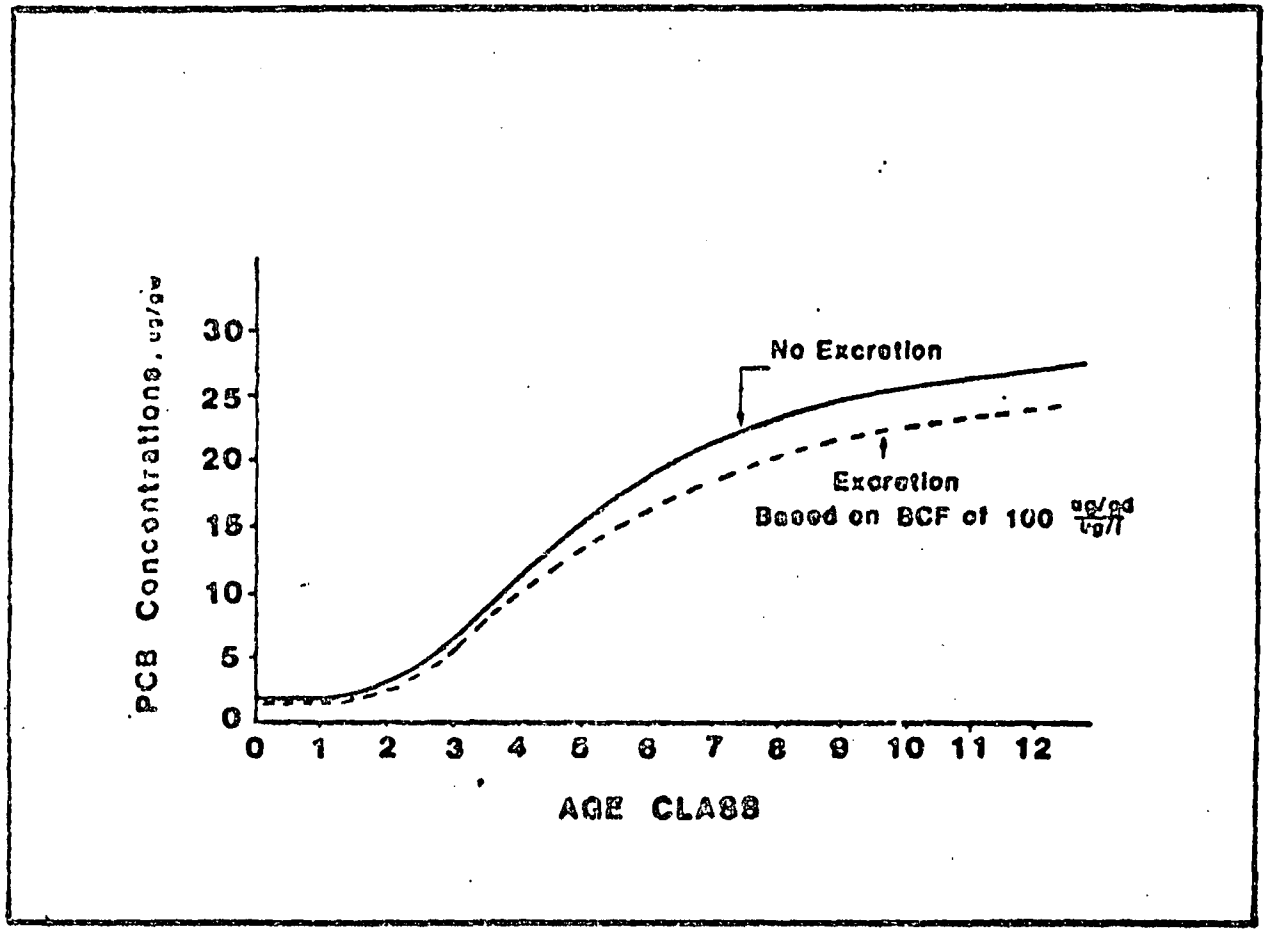

Figure 37. Computed lake trout PCB concentrations at the calibrated excretion rate and zero excretion rate.

SENSITIVITY OF MODEL TO DISSOLVED PCB CONCENTRATION

As discussed in Section 4, the main-lake dissolved PCB concentration during the early seventies is believed to have been less than 10 ng/ℓ. A more precise estimate is unavailable. In the calibration a value of 5 ng/ℓ is assumed.

An examination of the effect of variation of the dissolved PCB concentration on computed alewife and lake trout PCB concentrations can be used to assess the likelihood that the dissolved PCB concentration was significantly different than 5 ng/ℓ. A criteria for calibration of the model is that the bioenergetic and PCB associated parameter values be in agreement with observed values. This criteria is satisfied at a dissolved PCB concentration of 5 ng/ℓ. If this criteria cannot be met at higher and lower concentrations it suggests that the concentration must have been close to 5 ng/ℓ.

Assuming dissolved PCB concentrations of 10 and 2 ng/ℓ alters the computed alewife and lake trout PCB concentrations as shown in Fig. 38. As expected, at 10 ng/ℓ computed concentrations are much greater than observed values and at 2 ng/ℓ they are much lower than observed values. To adjust these computations to better simulate the data the PCB assimilation efficiency is adjusted since this parameter exhibits the widest range of observed values, all other parameters having been shown in Section 6 to be in good agreement with experimental data.

At the 2 ng/ℓ dissolved PCB concentration, the model failed to reproduce the observed data even with assimilation efficiencies for alewife and trout increased to 0.99 (Fig. 39). Calculated concentrations were consistently low for alewife and low for upper age class trout.

At the 10 ng/ℓ dissolved PCB concentration, decreasing *Mysis* assimilation efficiency to 0.2 (the value reported by Wyman and O'Connors (1980) for the copepod *Acartia tonsa*) and alewife assimilation efficiency to 0.45 produced calculated concentrations essentially identical to the calibration at a dissolved PCB concentration of 5 ng/ℓ (Fig. 39). Although the assimilation efficiency of 0.2 for *Mysis* is reasonable, the value of 0.45 for alewife is low relative to available fish data. However, since no data on middle level con-

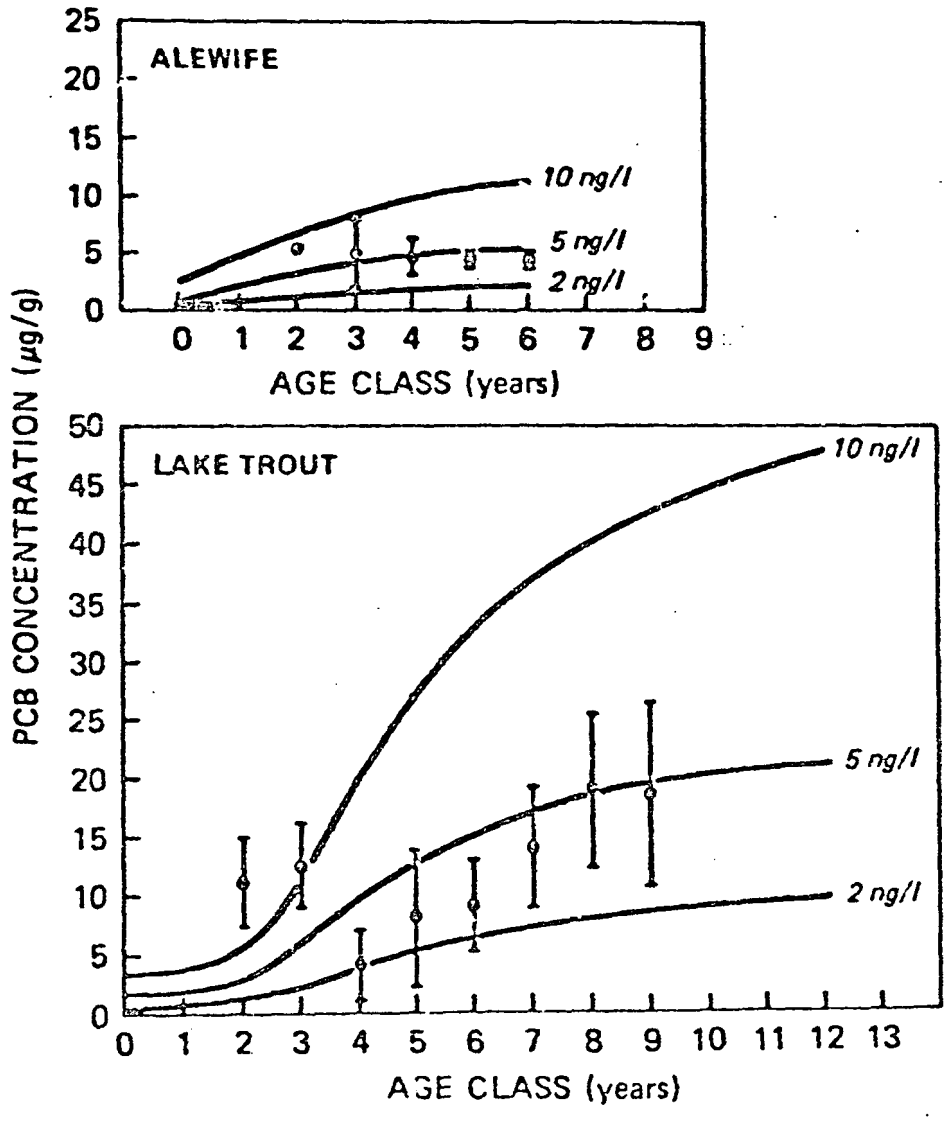

Figure 38. Computed alewife and lake trout PCB concentrations at dissolved PCB concentrations of 2 µg/ℓ, 5 µg/ℓ and 10 µg/ℓ.

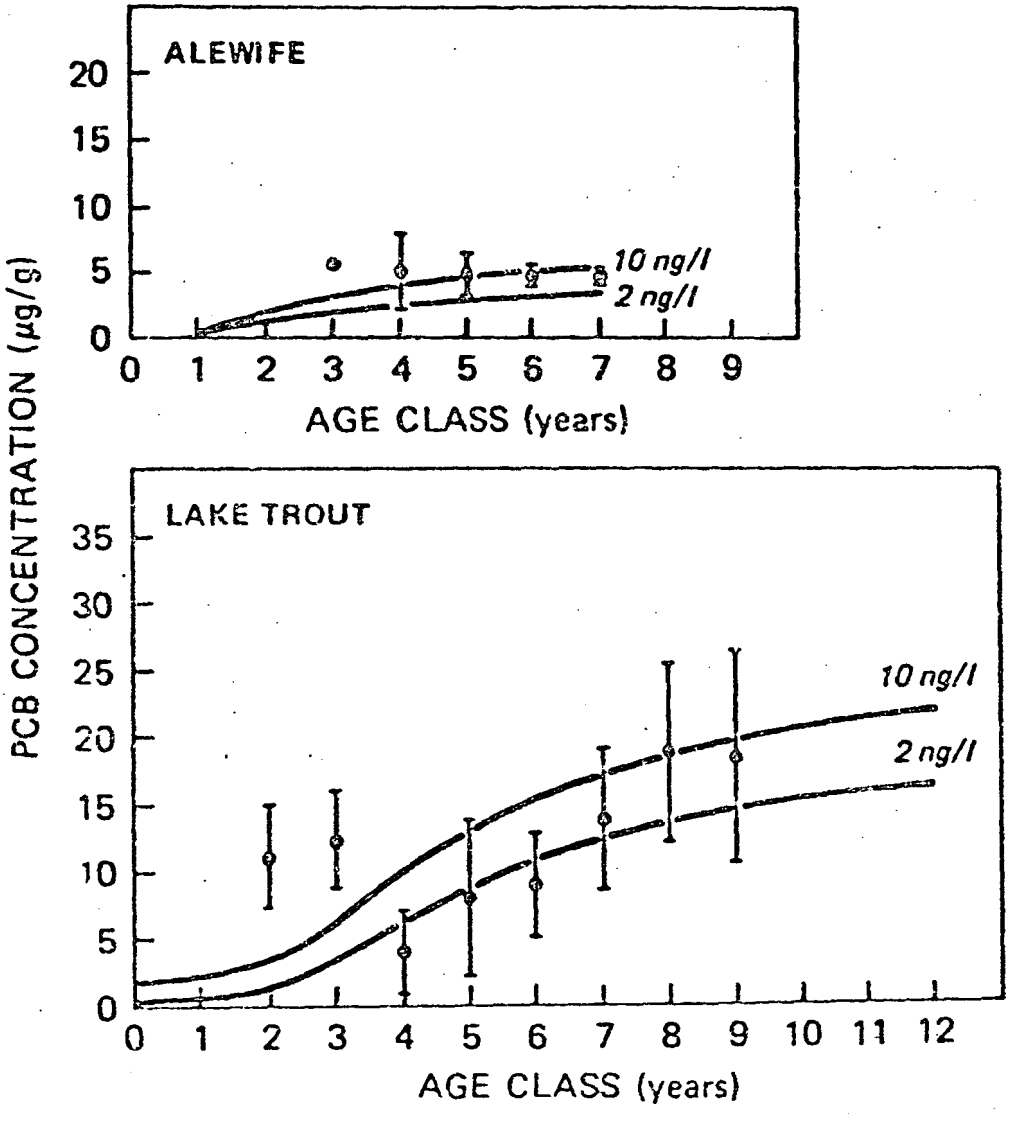

Figure 39. Computed alewife and lake trout PCB concentrations for models recalibrated to dissolved PCB concentrations of 2 ug/ℓ and 10 µg/ℓ by alteration of PCB assimilation efficiencies.

sumers like alewife are available, the value of 0.45 cannot be considered unreasonable and 10 ng/ℓ must be assumed to be a possible value for the dissolved PCB.

It may be concluded from the above analysis that the average dissolved PCB concentration in the main-lake of Lake Michigan was likely in the range of 5 to 10 ng/ℓ. The implication of using 5 ng/ℓ in the calibration is that a conservative estimate of the loading reduction necessary to achieve a certain concentration in the food chain will result from any projection. The model is linear with respect to dissolved PCB concentration and the undertainty of the model results will be directly proportional to the uncertainty of the dissolved PCB concentration. Since an uncertainty in dissolved concentration of approximately a factor of 2 is suggested by the above analysis a similar factor must be placed on any prediction mode using the model.

SENSITIVITY OF MODEL TO FOOD CHAIN STRUCTURE

The lake trout food chain considered here includes four trophic levels. The geometric increase in PCB contamination with trophic level demonstrated in Section 7 suggests that if the lake trout food chain were, say, the three trophic level chain that is exhibited by many other species then PCB concentrations in lake trout would be much less.

To illustrate this effect the lake trout PCB concentrations were computed with Mysis removed from the food chain structure and then with alewife removed from the food chain structure (Fig. 40). For both cases there is a dramatic reduction in lake trout PCB. At the upper age classes the food chain without Mysis has higher concentrations than the food chain without alewife. This results from the higher assimilation efficiency of the alewife relative to Mysis. At the lower age classes the food chain without alewife has higher concentrations because the lake trout are consuming Mysis whereas in the mode without Mysis they are consuming the lower concentration plankton.

For adult trout the loss of a trophic level reduces concentration by a factor of 5 to 10. This illustrates the importance of food chain transfer and the significance of a species trophic position on its accumulation of PCB from the environment.

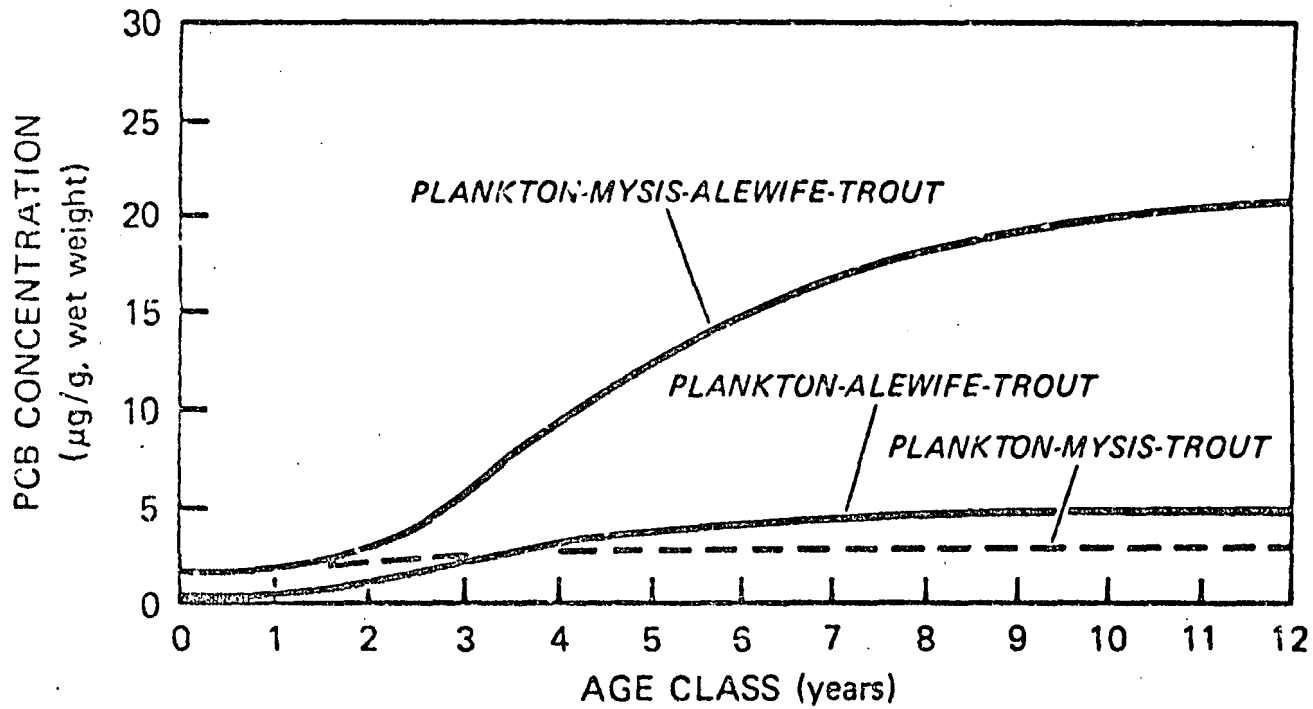

Figure 40. Computed lake trout PCB concentrations for three trophic level and four trophic level food chain structures.

SENSITIVITY OF MODEL TO LAKE TROUT GROWTH RATES

As presented in Section 6, native and stocked lake trout in Lake Michigan may have different rates of growth based on observed length-age and weight-age relationships. The growth rate difference is significant because the 1971 data used for calibration of the model include fish from both populations. Fish greater than 7 years old in 1971 were native to Lake Michigan because they were born prior to the beginning of the stocking program. Fish less than 7 years old, that were caught and analyzed for PCB, were likely stocked since the stocked population was far greater than the native population.

Computed PCB concentrations using both sets of growth rates are compared to each other and observed PCB concentrations in Fig. 41. Weight, rather than age class, is used in the figure because the age class divisions are different for each set of growth rates. As in the previous calibration figures, computed concentrations using the growth rates for native trout are below the data for young trout, high for intermediate age trout and within the data for the older trout. The differences between the model and data for the smaller trout are accentuated by the compressed scale for the early age classes. The computed concentrations using the growth rates for stocked trout fit the data for trout up to 3000 g(w) much better than the calibration. The continued increase for heavier trout (greater than age 7), however, is not supported by the data.

These results indicate that the optimal model calibration would consider stocked trout up to about age 7 and native trout thereafter. This supports the contention that there are two trout populations with different growth rates. An important consequence of this population difference is indicated by the computation of greater PCB accumulation in stocked trout. With stocked trout comprising the bulk of the population in Lake Michigan, the calculated PCB concentration reduction necessary to reduce lake trout PCB to a specified level is greater than predicted using native trout growth rates. This is illustrated in Section 9.

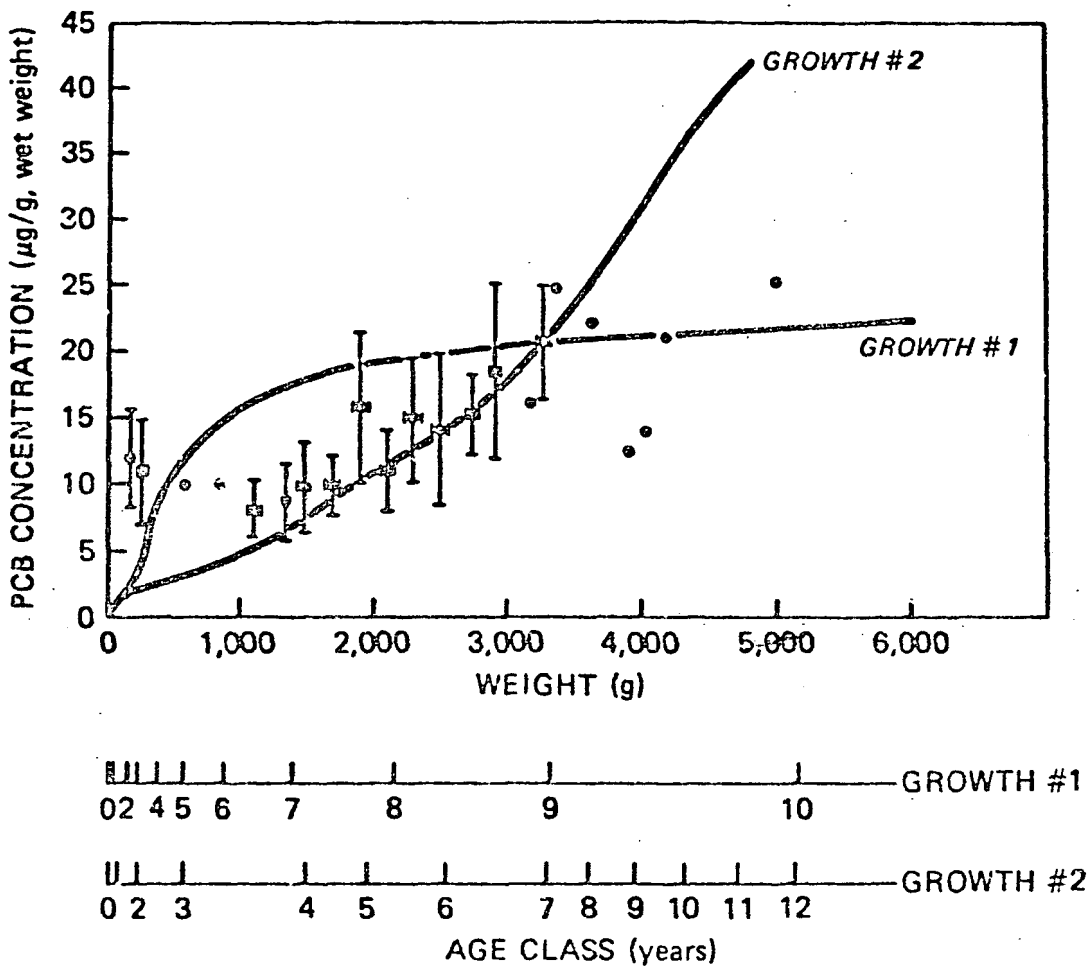

Figure 41. Computed and observed lake trout PCB concentrations in relation to wet weight for native trout growth rates (#1) and stocked trout growth rates (#2).

SECTION 9

LAKE TROUT RESPONE DUE TO REDUCED WATER CONCENTRATION

The objectives of this research as noted in the Introduction include a preliminary projection of the response in PCB concentration in the lake trout following a reduction in water column concentration. The purpose of such a projection is to determine the approximate dissolved water column concentration that is necessary to have PCB concentrations in the edible portion of the lake trout below 5 μg/g(w).

Figure 42 shows the result of a simulation of the response of the lake trout to a five fold decrease in the dissolved PCB concentration. The time history of 3 different year classes is shown. The dashed lines show the simulated PCB concentration after the water concentration is instantaneously dropped from 5 ng/ℓ to 1 ng/ℓ. For the 7-year old lake trout, initially at a concentration of about 19 μg/g(w), there is a decline for a period of 5 years to about 6 μg/g(w) at which point the assumed maximum age of the fish of 12 years is reached. The four year old fish is calculated to approach 4 μg/g(w) after about a four-five year period of decrease. The one year old traces out the PCB distribution that is calculated over its life cycle and as shown reaches a maximum value of 4 μg/g(w). One concludes from this projection that when water levels drop from the assumed concentration of 5 ng/ℓ dissolved to 1 ng/ℓ, that the lake trout will reach maximum concentrations of about 4 μg/g(w) on a whole fish basis. A period of about 5 years would be required to "clear out" the higher concentrations from the upper age class fish. The calculations shown in Figure 42 however are for the growth rate of the lake trout for the pre-stocked period. The time variable response under the higher growth rates representing the period of extensive stocking would be somewhat different although not markedly so.

The entire modeling framework for the PCB in the lake trout is linear to the dissolved water concentration. A simple relationship can therefore be obtained between lake trout concentration and the PCB concentration in the water. This is shown in Figure 43. As noted previously in Section 8 - Model

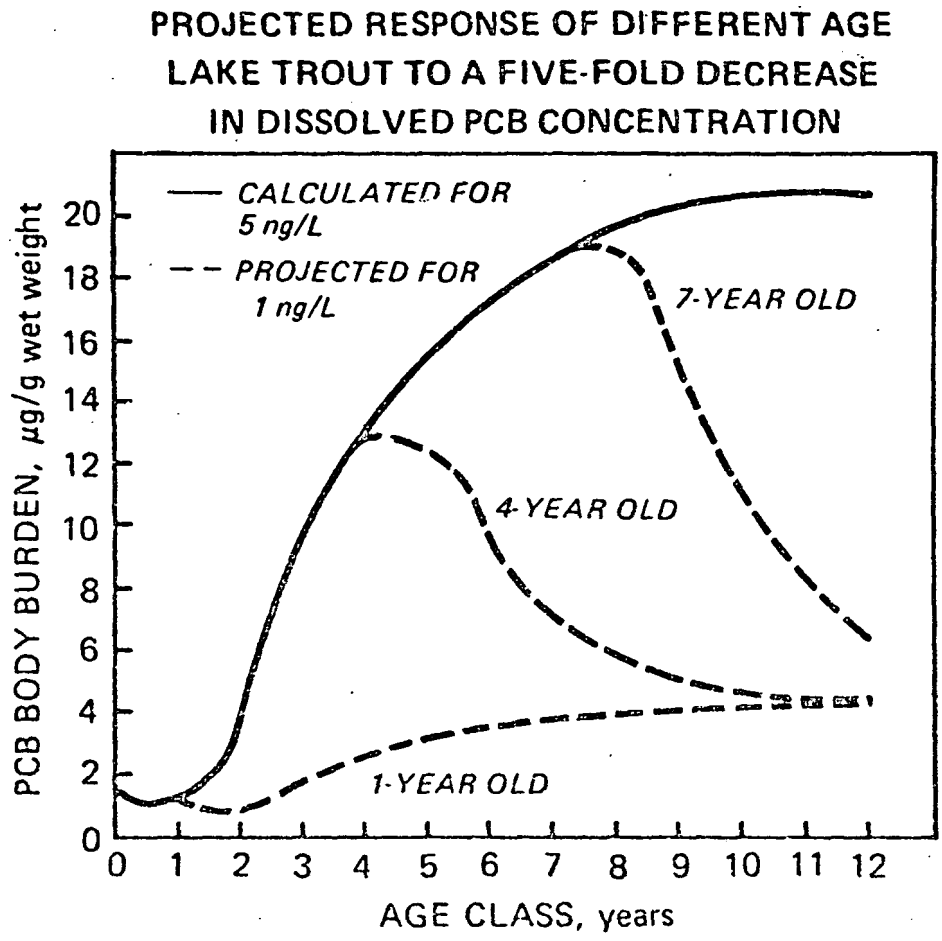

Figure 42. Projected response of different age lake trout to a five-fold decrease in dissolved PCB concentration - growth rate #1 for pre-stocked period.

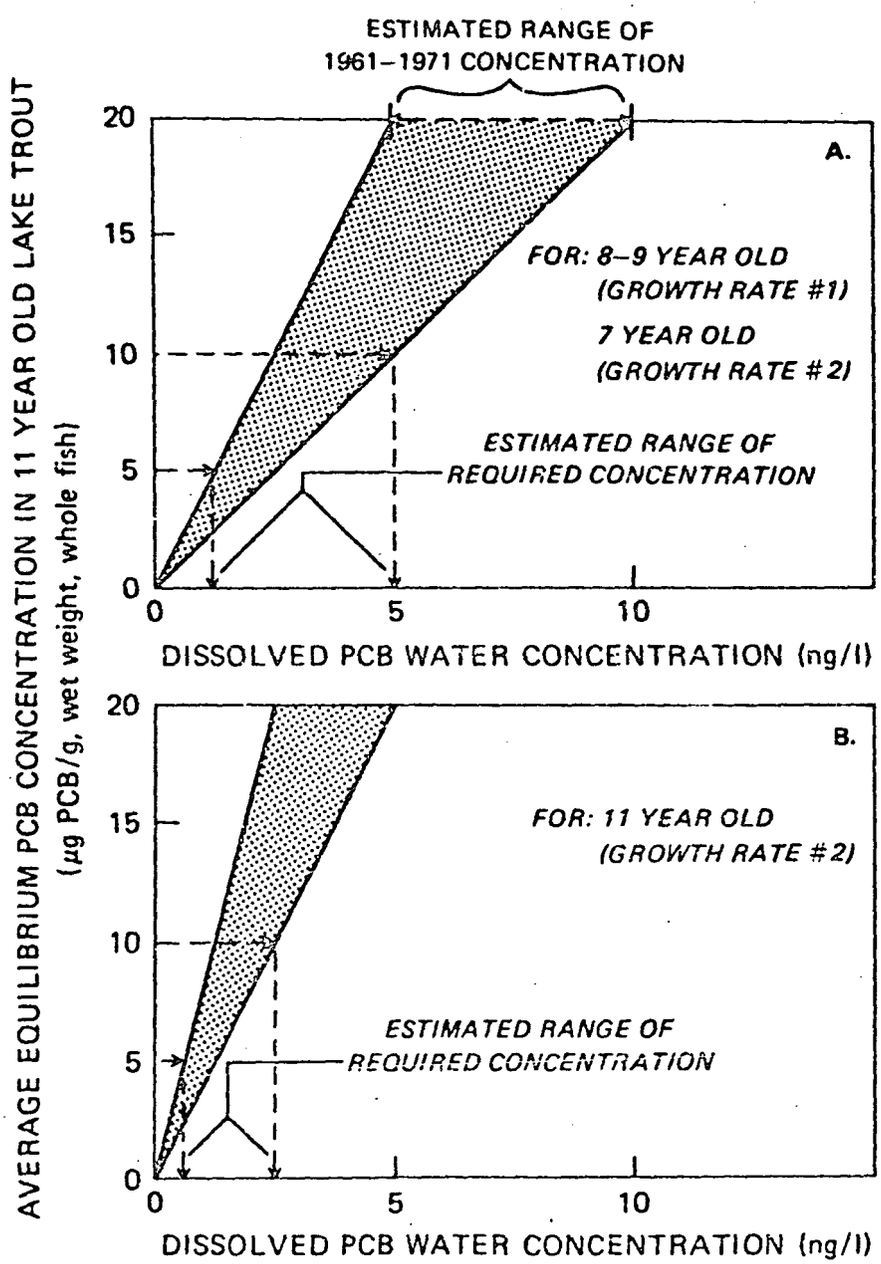

Figure 43. Relationships between equilibrium PCB concentration in 11 year old lake trout and dissolved PCB water concentrations, growth rate #1 - pre-stocked period; growth rate #2 - stocked period.

Sensitivity, the 1971 lake trout data using growth rate #1 (pre-stocked growth rate) could be calibrated within an acceptable range of parameters for water concentrations between 5 and 10 ng/ℓ. Figure 43(a) shows this calibration range for the 20 μg/g(w) as observed in the 8-9 year age class and the pre-stocked growth rate (#1). As shown in Figure 41, the 20 μg/g(w) also applies to the 7 year olds under the stocked period growth rate (#2). The calibration and sensitivity analyses indicate that the dissolved PCB water concentration for the 1961-1971 period was then probably between 5 and 10 ng/ℓ.

The objective for PCB concentration in the edible portion of the lake trout is 5 μg/g(w). The model calculation is for the PCB concentration in the whole fish. Recognizing the variability in the concentration of the edible portion as a function of the whole fish (see for example, Figure 9), a range of 5-10 μg/g(w) on a whole fish basis was chosen. This assumes that the edible portion concentration is about 50% of the whole fish concentration, a fraction indicated approximately by the data of Figure 9. This range is then used to estimate the required water concentration such that the edible portion would be expected to be below 5 μg/g(w). Figure 43(a) shows the linear relationship between whole fish concentration and dissolved water concentration. The shaded area indicates the region of uncertainty as a result of the calibration uncertainty in the water concentration. If whole fish concentrations of 5-10 μg/g(w) are acceptable (and are considered to result in edible concentrations less than 5 μg/g(w)), then a range of about 1-5 ng/ℓ dissolved PCB water concentration is necessary. A 50-90% reduction in the 1971 water concentration would therefore result in 7-9 year old whole fish concentrations of 5-10 μg/g(w) and edible portion concentrations generally less than 5 μg/g(w). The degree to which any reduction in water concentration have been accomplished since 1971 is subject to further study.

The situation is slightly different for the older fish (e.g., 11 years old) under the higher growth rate of the stocked fish. This is shown in Figure 43(b) where it is noted that 20 μg/g(w) in an 11 year old results from only 2.5-5 ng/ℓ (as contrasted to 5-10 ng/ℓ under the slower growth rate). The difference is attributed to the apparent higher growth rate of the stocked lake trout (see discussion in Section 8). For a similar objective of 5-10 μg/g(w) in the 11

year old fish, Figure 43(b) indicates that the required dissolved PCB water concentration would have to be in the range of 0.5-2.5 ng/ℓ. This range represents a 75-95% reduction of apparent 1961-1971 concentrations.

It is apparent then that for a fixed water concentration, the objective of 5 μg/g(w) in the edible portion may be met by some age classes but not others. Indeed, the critical age classes are the fish that are generally older than about 6 years. The overall relationship between age class and the required dissolved water concentration to maintain 5-10 μg/g(w) on a whole fish basis is shown in Figure 44. The range shown reflects uncertainty in the PCB assimilation efficiency and dissolved PCB concentration. As indicated, the older age classes require the lowest dissolved PCB water concentration to meet the 5-10 μg/g(w) level. This is just another representation of Figures 42 and 43. Thus, if a level of 2 ng/ℓ were obtained, then whole fish 6 years and older would have concentrations between 5 and 10 μg/g(w). Conversely, whole fish less than 6 years old would have PCB concentrations less than 5 μg/g(w). It is possible to use a plot such as Figure 44 to develop an age dependent (or more practically a weight or size dependent) basis for opening up the fishery to consumption. For example, if projections for the next 10 years indicate that the lowest attainable PCB water concentration is 1 ng/ℓ because of diffuse and atmospheric sources, then lake trout 8 years or older (i.e., 3.5 kg or 70 cm) would be prohibited for consumption. The whole body concentrations in those fish would be expected to be equal to or greater than 5 μg/g(w).

CONCLUSIONS

The projections of the behavior of the lake trout food chain to reduced water concentrations indicate the following:

1. Following reductions in water column PCB concentrations, a period of about 5 years is needed to reduce whole body PCB concentrations in upper age class lake trout.
2. In order to have the PCB concentrations of all age classes of lake trout at or below 5 μg/g(w) in the edible portion, it is estimated that the dissolved water concentrations would have to be between 0.5-2.5 ng/ℓ using growth rates representative of stocked fish.

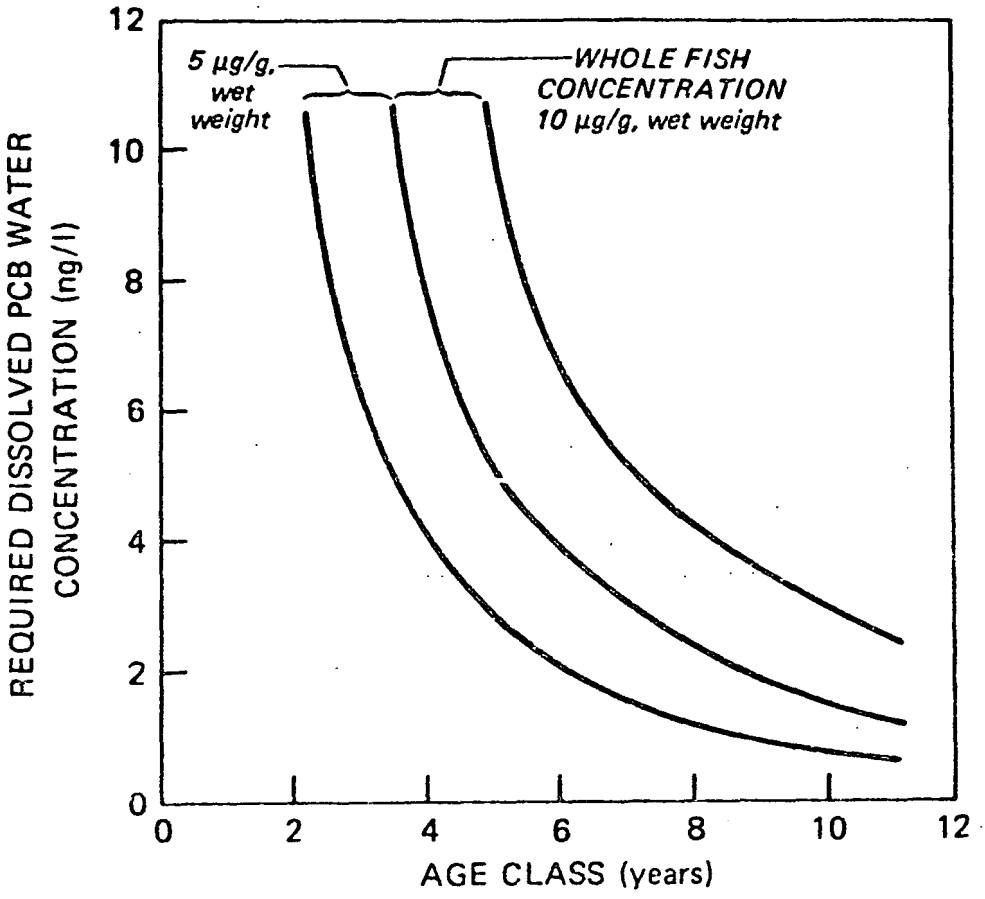

Figure 44. Calculated relationships between age class of lake trout and required dissolved PCB water concentration to meet whole fish concentration of 5-10 µg/g(w). Growth rate #2 for stocked lake trout.

This range represents a 75-95% reduction of apparent 1961-1971 water concentrations.

3. Younger age classes can generally be exposed to higher water PCB concentrations than older age classes without exceeding the objective of 5 μg/g(w). As a result, if water quality projections indicate a lower bound in the achievable PCB water concentrations, a size dependent fish consumption guideline can be developed.

REFERENCES

Anon. 1981. Sea Grant Today. Utilizing Alewives, 11(2):10.

Aoyama, I., Y. Inoue and Y. Inoue. 1978. Simulation analysis of the concentration process of trace heavy metals by aquatic organisms from the viewpoint of nutrition ecology. Water Res. 12:827-842.

Beeton, A.M. 1960. The Vertical Migration of Mysis relicta in Lakes Huron and Michigan. J. Fish. Res. Bd. Can. 17:517-539.

Brett, J.R. and T.D.D. Groves. 1979. Physiological Energetics. In Fish Physiology (W.S. Hoar, D.J. Randall, and J.R. Brett, eds.), Vol. 8, pp. 280-352. Academic Press, New York.

Carlander, K.D. 1969. Handbook of Freshwater Fishery Biology. Vol. 1. The Iowa State University Press, Ames, Iowa.

Chiou, C.T., V.H. Freed, D.W. Schmedding, and R.L. Kohnert. 1977. Partition Coefficient and Bioaccumulation of Selected Organic Chemicals. Environ. Sci. Technol. 11(5):475-478.

Cooper, S.D. and C.R. Goldman. 1980. Opossum Shrimp (Mysis relicta) Predation on Zooplankton. Can. J. Fish. Aquat. Sci. 37:909-919.

Dale, M.G. 1970. Systems analysis and ecology. Ecology 51:1-16.

DeFoe, D.L., G.D. Veith, and R.W. Carlson. 1978. Effects of Aroclor 1248 and 1260 on the Fathead Minnow (Pimephales promelas). J. Fish. Res. Bd. Can. 35:997-1002.

Elliot, J.M. 1976. Energy Losses in the Waste Products of Brown Trout (Salmo tratta L.). J. Anim. Ecol. 45:561-580.

Frank, R., R.L. Thomas, H.E. Braun, D.L. Gross, and T.T. Davies. 1981. Organochlorine insecticides and PCB in surficial sediments of Lake Michigan (1975). J. Great Lakes Res. 7(1):42-50.

Gillett, J.W., J. Hill, A.W. Jarvinen, and W.D. Schnoor. 1974. A conceptual model for the movement of pesticides through the environment. Nat. Environ. Res. Cntr., Off. Res. Dev., U.S. Environ. Prot. Agency, Corvallis, Oregon, EPA 660/3-74-024; 79 p.

Glass, N.R. 1969. Discussion of Calculation of Power Function with Special Reference to Respiratory Metabolism in Fish. J. Fish. Res. Bd. Can. 26: 2643-2650.

Great Lakes Basin Commission. 1975. Great Lakes Basin Framework Study, Appendix 8, Fish. Prepared by the Fish Work Group, Sponsored by Mich. DNR, GLBC, Ann Arbor, Mich., 290 pp.

Grossnickle, N.E. 1978. The Herbivorous and Predaceous Habits of Mysis relicta in Lake Michigan. Ph.D. Thesis, The University of Wisconsin-Madison. 107 p.

Haefner, J.W., and J.W. Gillett. 1976. Aspects of mathematical models and microcosm research. Proc. Conference in Environmental Modeling and Simulations, U.S. Environ. Prot. Agency, Off. Res. Devel., Off. Plann. Man., Washington, D.C. EPA600/9-76-016:624-628.

Hansen, D.J., P.R. Parish and J. Forester. 1974. Aroclor 1016: Toxicity to and uptake by estuarine animals. Envir. Res. 7:363-373.

Hess, R., and B. Muench. 1980. 1980 Report of the Lake Trout in Illinois Waters of Lake Michigan. Illinois Department of Conservation, Division of Fisheries and Wildlife, Dec. 1980.

Hill, J., H.P. Kallig, D.F. Paris, N.L. Wolfe, and R.G. Zepp. 1976. Dynamic behavior of vinyl chloride in aquatic ecosystems, U.S. Environ. Prog. Agency, Off. Res. Dev., Environ. Res. Lab., Athens, Georgia. EPA-600/3-76-001: 64 p.

Int. Joint Comm. 1978. Environmental management strategy for the Great Lakes system. Final Report to the IJC from PLUARG, Int. Joint Comm., Windsor, Ont., 115 pp.

Janssen, J. and S.B. Brandt. 1980. Feeding Ecology and Vertical Migration of Adult Alewives (Alosa pseudoharengus) in Lake Michigan. Can. J. Fish. Aquat. Sci. 37:177-184.

Kenaga, E.E. 1980. Correlation of bioconcentration factors of chemicals in aquatic and terrestial organisms with their physical and chemical properties. Env. Sci. Technol. 14:553-556.

Kleiber, M. 1961. The Fire of Life. An Introduction to Animal Energetics. Wiley, New York.

Klots, E.B. 1966. The New Field Book of Freshwater Life. G.P. Putnam's Sons, New York. 398 p.

Konasewich, D., W. Traversy and H. Zar. 1978. Status report on organic and heavy metal contaminants in the Lakes Erie, Michigan, Huron and Superior basins. Int. Joint Comm., Windsor, Ont., 373 pp.

Lasenby, D.C. and R.R. Langford. 1972. Growth, Life History, and Respiration of Mysis relicta in an Arctic and Temperate Lake. J. Fish. Res. Bd. Can. 29:1701-1708.

Lassiter, R.R., J.L. Malanchuk, and G.L. Baughman. 1976. Comparison of processes determining the fate of mercury in aquatic systems. Proc. Conf. on Environmental Modeling and Simulation. U.S. Environ. Prot. Agency. Off. Res. Dev., Environ. Res. Lab., Washington, D.C. EPA 600/9-76-016:619-623.

Lieb, A.J., D.D. Bills, and R.O. Sinnhuber. 1974. Accumulation of Dietary Polychlorinated Biphenyls (Aroclor 1254) by Rainbow Trout (Salmo gairdneri). J. Agr. Food Chem. 22(4):638-642.

McLeese, D.W., C.D. Metcalfe, and D.S. Pezzack. 1980. Uptake of PCBs From Sediment by Nereis virens and Crangon septemspinosa. Arch. Environ. Contam. Toxicol. 9:507-518.

McWilliam, P.S. 1970. Seasonal Changes in Abundance and Reproduction in the Opossum Shrimp, Mysis relicta Loven, in Lake Michigan. M.S. Thesis, University of Sydney. 94 p.

Morgan, M.D. and A.M. Beeton. 1978. Life History and Abundance of Mysis relicta in Lake Michigan. J. Fish. Res. Bd. Can. 35:1165-1170.

Morsell, J.W. and C.R. Norden. 1968. Food Habits of the Alewife, Alosa pseudoharengus (Wilson), in Lake Michigan. Proc. 11th Conf. Great Lakes Res.:
96-102.

Mozley, S.C. and R.P. Howmiller. 1977. Environmental Status of the Lake Michigan Region. Vol. 6. Zoobenthos of Lake Michigan. Argonne National Laboratory, Argonne, Illinois. ANL/ES-40.

Murphy, T.J. and C.P. Rzeszutko. 1977. Precipitation inputs of PCBs to Lake Michigan. J. Great Lakes Res. 3(3-4):305-312.

Neely, W.B., D.R. Branson, and G.E. Blau. 1974. Partition coefficient to measure bioconcentration potential or organic chemicals in fish. Environ. Sci. Technol. 8:1113-1115.

Neely, W.B. 1979. A preliminary assessment of the environmental exposure to be expected from the addition of a chemical to a simulated aquatic ecosystem. Intern. J. Env. Studies, 13:101-108.

Neidermeyer, W.J. and J.S. Hickey. 1976. Chronology of organochlorine compounds in Lake Michigan fish, 1929-66. Pesticides Monitoring Jour. 10(3): 92-95.

Norstrom, R.J., A.E. McKinnon, and A.S.W. DeFreitas. 1976. A bioenergetics-based model for pollutant accumulation by fish. Simulation of PCB and methylmercury residue levels in Ottawa River yellow perch (Perca flavescens). J. Fish. Res. Board Can. 33:248-267.

Patten, B.C. [Ed.] 1971. Systems analysis and simulation in ecology. Academic Press, New York. 607 p.

Reynolds, J.B. and G.M. DeGraeve. 1972. Seasonal Population Characteristics of the Opossum Shrimp, Mysis relicta, in Southeastern Lake Michigan, 1970-71. Proc. 15th Conf. Great Lakes Res.:117-131.

Ricker, W.E. 1979. Growth Rates and Models. In Fish Physiology (W.S. Hoar, D.J. Randall, and J.R. Brett, eds.), Vol. 8, pp. 678-744. Academic Press, N.Y.

Roberts, J.R., A.S.W. deFrietas, and M.A.J. Gidney. 1977. Influence of Lipid Pool Size on Bioaccumulation of the Insecticide Chlordane by Northern Redhorse Suckers (Moxostoma macrolepidotum). J. Fish. Res. Bd. Can. 34:89-97.

Schmidt-Nielsen, K. 1970. Energy Metabolism, Body Size, and Problems of Scaling. Federation Proceedings, 29(4):1524-1532.

Schmoor, J.L. 1980. Field Validation of Water Quality Criteria for Hydrophobic Pollutants. Presented at the Fifth ASTM Symposium on Aquatic Toxicology, 7-8 October, 1980. Philadelphia, Pa.

Schnute, J. and D. Fournier. 1980. A New Approach to Length-Frequency Analysis: Growth Structure. Can. J. Fish. Aquat. Sci. 37:1337-1351.

Scura, E.C. and G.H. Theilacker. 1977. Transfer of the chlorinated hydrocarbon PCB in a laboratory food chain. Mar. Biol. 40:317-325.

Stewart, D.J. 1980. Salmonid Predators and their Forage Base in Lake Michigan: A Bioenergetics-Modeling Synthesis. Ph.D. Thesis, The University of Wisconsin-Madison, 225 pp.

Thomann, R.V. 1981. Equilibrium model of fate of microcontaminants in diverse aquatic food chains. Can. J. Fish. Aquat. Sci., 38(3):280-296.

Thomann, R.V. and M.T. Kontaxis. 1981. Mathematical modeling estimate of environmental exposure due to PCB-contaminated harbor sediments of Waukegan Harbor and North Ditch. Prep. for USEPA, Ind. Env. Res. Lab., Cinn, Ohio by HydroQual, Inc., Mahwah, N.J., 109 pp.

Torrey, M.S. 1976. Environmental status of the Lake Michigan region. Vol. 3, Chemistry of Lake Michigan. ANL/ES-40, Vol. 3. Prepared for US Energy Res. & Devel. Admin., Argonne Nat. Lab., Argonne, Ill. 418 pp.

Veith, G.D. and G. Fred Lee. 1971. PCBs in fish from the Milwaukee region. Proc. 14th Conf. Grt. Lks. Res. Int. Assoc., Great Lakes Res:157-169.

Veith, G.D. 1973. Chlorinated hydrocarbon in fish from Lake Michigan. USEPA, Water Quality Off., Project No. 16020 PBE, 129 pp.

Veith, G.D. 1975. Baseline concentrations of polychlorinated biphenyls and DDT in Lake Michigan Fish, 1971. Pest. Monitor. Jour. 9(1):21-29.

Veith, G.D., D.L. DeFoe, and B.V. Bergstedt. 1979. Measuring and Estimating the Bioconcentration Factor of Chemicals in Fish. J. Fish. Res. Bd. Can. 36:1040-1048.

Vreeland, V. 1974. Uptake of chlorobiphenyls by oysters. Eviron. Pollut. 6:135-140.

Wapora. 1981. Environmental assessment of Waukegan Harbor. In preparation.

Weaver, J.E. 1975. Food Selectivity, Feeding Chronology, and Energy Transformations of Juvenile Alewife (Alosa pseudoharengus) in the James River near Hopewell, Virginia. Ph.D. Thesis, University of Virginia. 87 p.

Weininger, D. 1978. Accumulation of PCBs by Lake Trout in Lake Michigan. Ph.D. Thesis, University of Wisconsin-Madison. 232 p.

Winberg, G.G. 1956. Rate of Metabolism and Food Requirements of Fishes. Nauchn. Tr. Beloruss. Gos. Univ. V.I. Lenina, Minsk, 253 p. (Transl. from Russian by Fish. Res. Bd. Can. Transl. Ser. No. 194, 1960.)

Wright, K.J. 1968. Feeding Habits of Immature Lake Trout (Salvelinus namaycush) in the Michigan Waters of Lake Mighigan. M.S. Thesis, Michigan State University, Lansing, MI. 42 p.

Wszolek, P.C., D.J. Lisk, T. Wachs, and W.D. Youngs. 1979. Persistence of polychlorinated biphenyls and 1,1-Dichloro-2,2 bis (p-chlorophenyl)ethylene (p,p'-DDE) with age in lake trout after 8 years. Environ. Sci. & Thecnol. 13(10):1269-1271.

Wyman, K.D. and H.B. O'Connors, Jr. 1980. Implications of Short-term PCB Uptake by Small Estuarine Copepods (Genus Acartia) from PCB-contaminated Water, Inorganic Sediments and Phytoplankton. Estuarine and Coastal Marine Science. 11:121-131.

Zabik, M.E., B. Olson and T.M. Johnson. 1978. Dieldrin, DDT, PCBs, and mercury levels in freshwater mullet from the Upper Great Lakes, 1975-76. Pest. Monitor. J. 12(1):36-39.