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Sara Tucker Wellman
The College at Brockport, wellmanst@aol.com

James M. Haynes
The College at Brockport, jhaynes@brockport.edu

James J. Pagano

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Model Estimates Bioaccumulation of Total PCBs, Dioxin-Furan TEQs, and Total Mercury in Mink Liver Based on Concentrations in Lake Ontario Water

Sara T. Wellman,[1] James M. Haynes[1,3] and James J. Pagano[2]

[1]Department of Environmental Science and Biology
The College at Brockport, State University of New York
Brockport, NY 14420

[2]Environmental Research Center and Department of Chemistry
The College at Oswego, State University of New York
Oswego, NY 13126

[3]Copies: www.brockport.eduenvsci

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Abstract

We used stable isotope analysis and a bioaccumulation model to estimate concentrations of total polychlorinated biphenyls (PCB), dioxin-furan toxic equivalents (TEQ), and total mercury (Hg) in mink and to compare predicted ranges with their chemical concentrations in mink liver (PCB, TEQ) and brain (Hg). Actual concentrations were within predicted bounds for total PCB, dioxin-furan TEQ, and Hg except in two cases (lowest PCB and highest Hg) which were very close to predicted bounds. Based on ^{15}N analysis, the trophic level of mink ranged from 3.4 to 3.9. Animals at the upper end of the range were exposed to Lake Ontario water and its food web while those at the lower end were captured at inland locations. Because of the complexity of wetland (an important habitat for mink in this study) food webs with pelagic, littoral, and terrestrial carbon sources and overlapping ^{13}C signatures, whether the origins of mink diets were aquatic or terrestrial could not be determined. We have established a non-destructive biomonitoring tool to reasonably estimate concentrations of total PCB, TEQ and total Hg in mink tissues as concentrations of these chemicals change in their water supply.

Introduction

Stable isotopes of nitrogen and carbon are used to evaluate trophic webs of ecosystems to give lifetime, integrated estimates of both trophic level and dietary sources for organisms (DeNiro and Epstein 1978, Cabana and Rasmussen 1994). Both ^{12}C and ^{14}N have stable, heavier isotopes (^{13}C and ^{15}N) which occur naturally, and the heavier and lighter isotopes are differentially absorbed and metabolized by organisms (Fry 1991). Usually the lighter isotopes are excreted preferentially, leading to a relative enrichment of the heavier isotopes in organisms relative to their environment or diet. These enrichments

are measurable through mass spectrometry, and are reported in parts per thousand (δ‰) relative to a standard:

$$\delta X = [(R_{sample} - R_{standard}) - 1] \times 10^3,$$

where X is ^{13}C or ^{15}N and R is the corresponding ratio $^{13}C/^{12}C$ or $^{15}N/^{14}N$. The standard for carbon is PeeDee Belemnite (PDB) limestone, and the standard for nitrogen is atmospheric nitrogen (Fry 1991).

Selective excretion of ^{14}N over ^{15}N by animals results in an increase of approximately 3.4‰ in the $\delta^{15}N$ at each trophic level; thus, ^{15}N analysis can determine the average trophic level at which an animal feeds (Peterson and Fry 1987, Cabana and Rasmussen 1994). Carbon is also enriched between trophic levels but at a much lower rate between 0 and 1‰. Because freshwater algae have a much less negative $\delta^{13}C$ than terrestrial plants (e.g., terrestrial leaves $\delta^{13}C$ = -27 to -31‰ versus algae > -17‰; Collier and Lyon 1991), ^{13}C analysis can differentiate between these as original sources of carbon in a diet, indicating whether the diet is primarily of aquatic or terrestrial origin.

Once trophic level and percent aquatic diet are known, the concentration of a persistent organic chemical can be calculated using a model adapted from Sample et al. (1996). The model takes into account the concentration of the chemical in the water, daily food and water ingestion rates, proportion of the diet originating from aquatic carbon sources, body mass of the animal, and bioaccumulation factor (BAF) for the chemical. The BAF is dependent upon the trophic level and the octanol-water partition coefficient of the compound (Sample et al. 1996).

Our study originated from reports that identified "degradation of fish and wildlife populations" and "bird or animal deformities and reproductive problems" as potential problems in the Rochester Embayment of Lake Ontario Area of Concern (AOC; RAP 1993, 1997). We conducted stable isotope analysis for ^{13}C and ^{15}N on tissues from the same mink collected for total mercury, total PCB, and dioxin-furan TEQ analyses (Haynes et al. 2009). The questions addressed here are: Can stable isotope analysis be used to evaluate mink diets, at lakeshore and inland areas, in terms of trophic levels and terrestrial and aquatic food sources? Can stable isotope results be used to construct a food web/bioaccumulation model for mink to predict body burdens of total PCB, dioxin-furan TEQ, and total mercury in mink? How do predicted concentrations of these chemicals in mink liver (PCB, TEQ) and brain (Hg) (based on concentrations in Lake Ontario water) compare with measured concentrations?

Methods

Complete methods for specimen collection, processing, and handling of tissues, as well as biological and chemical data, for the mink used in this study are in Haynes et al. (2009). All of the mink were pelt-less and 24 were tail-less which had implications for the bioaccumulation models (see the "Modeling exposure of mink in the AOC to persistent organic chemicals" section in the Results and Discussion). Thigh muscle was used for stable isotope analysis. Muscle samples, frozen and packaged with dry ice, were shipped to Cornell University's Stable Isotope Laboratory (COIL).

Tissue Analysis

At COIL, stable isotope analyses for ^{13}C and ^{15}N were done with a continuous flow Elemental Analyzer (NC2500, CE Elantech, New Jersey) interfaced with an Isotope Ratio Mass Spectrometer (Delta Plus, Thermo Electron Corp., Germany). Quality control procedures included standards to test for instrument linearity, define instrument response for the determination of elemental composition, and measure stability of precision and accuracy over the length of a run (Arthur Kasson, COIL, Ithaca, NY, personal communication). Total PCB, dioxin-furan TEQ, and total mercury analyses were performed as reported in Haynes et al. (2009).

Data Analysis

We used Microsoft ® Excel 2000 for data management and non-statistical calculations. For statistical analyses, we used Minitab™ Statistical Software Release 14.13 (2005). We used Minitab's General Linear Model routine (a 2-way ANOVA with two fixed factors, Area—AOC: in vs. out and Location—lakeshore vs. inland followed by Tukey's pair-wise comparisons) to analyze the relationships between each isotope and the areas and locations where mink were captured.

Bioaccumulation Modeling

Trophic level was calculated by dividing the $\delta^{15}N$ value of an organism by the change in $\delta^{15}N$ per trophic level, usually 3.4‰ (Minigawa and Wada 1984, Vander Zanden and Rasmussen 1999, Doucett 1999). Calculating percent aquatic diet using $\delta^{13}C$ required 1) determining the $\delta^{13}C$ value in tissue, 2) estimating the difference between the $\delta^{13}C$ values in the tissue and in the diet, and 3) calculating the relative contributions of aquatic and terrestrial sources required to yield the estimated $\delta^{13}C$ of the diet (DeNiro and Epstein 1978). COIL's analysis provided the data for step 1. Literature review provided estimated values for step 2. The equation for step 3, calculating the proportion of a diet ($\%_A$) originating from one of two dietary sources of carbon with different $\delta^{13}C$ values, is

$$\%_A = \frac{\delta^{13}C_{animal} - \delta^{13}C_B - f \cdot x}{\delta^{13}C_A - \delta^{13}C_B} \times 100,$$

where $\delta^{13}C_{animal}$ is the stable-isotope ratio in the animal, $\delta^{13}C_A$ and $\delta^{13}C_B$ are the stable-isotope ratios of the two carbon sources, f is the trophic fractionation between the animal and its diet, and x is the trophic position of the animal (adapted from Doucett 1999).

Once the trophic level and aquatic portion of an animal's diet are known, the animal's exposure to a persistent organic chemical can be modeled knowing the concentration of the compound in ambient water. We started with Equation 28 from Sample et al. (1996), adding the units for clarity:

$$C_w\left(\frac{mg}{L}\right) = \frac{NOAEL\left(\frac{mg}{kg \cdot d}\right) \times bw(kg)}{W\left(\frac{L}{d}\right) + \left[F\left(\frac{kg}{d}\right) \times BAF\left(\frac{L}{kg}\right)\right]}, \qquad (1)$$

where C_w is the concentration of the POP in the water, *NOAEL* is the No Observed Adverse Effects Level; *W* and *F* are the daily water and food consumption rates in L/day and kg/day, respectively; *BAF* is the bioaccumulation factor for the chemical of concern (based on the trophic level of the animal and the octanol-water partition coefficient, k_{ow}, a measure of hydrophobicity or lipophilicity of the compound); and *bw* is the body mass of the animal in kilograms (Sample et al. 1996).

We solved for *NOAEL* and, taking into account the aquatic portion of the animal's diet (P_{aq}), got an equation to predict the exposure concentration of an animal to a chemical in water:

$$NOAEL\left(\frac{mg}{kg\,(bw)\cdot d}\right)=\frac{C_w\left(\frac{mg}{L}\right)\left[W(L)+\left(F(kg)\times P_{aq}\times BAF\left(\frac{L}{kg}\right)\right)\right]}{bw(kg)}. \quad (2)$$

The inclusion of the P_{aq} factor in this equation implies that there is no contribution of the chemical from the terrestrial portion of the diet. We made this assumption because our literature review (Gerell 1967, Melquist et al. 1981, Dunstone and Birks 1987, USEPA 1993, Sullivan 1996) indicated that the terrestrial portion of minks' diet consists mainly of lagomorphs and small rodents, which are herbivores and would have negligible bioaccumulations of persistent organic chemicals.

According to Sample et al. (1996), the dietary concentration C_f (mg/kg) equivalent to the NOAEL is:

$$C_f\left(\frac{mg}{kg}\right)=\frac{NOAEL\left(\frac{mg}{kg\cdot d}\right)\times bw(kg)}{F\left(\frac{kg}{d}\right)}. \quad (3)$$

Substituting equation (2) for *NOAEL* into equation (3) for C_f, the *bw* terms in the numerator and denominator cancel out and give a dietary concentration equivalent to the exposure concentration based on a chemical's concentration in water, the food and water consumption rates of mink, the percent aquatic diet, and the bioaccumulation factor (Sample et al. 1996):

$$C_f\left(\frac{mg}{kg}\right)=\frac{C_w\left(\frac{mg}{L}\right)\left[W\left(\frac{L}{d}\right)+F\left(\frac{kg}{d}\right)\times P_{aq}\times BAF\left(\frac{L}{kg}\right)\right]}{F\left(\frac{kg}{d}\right)}. \quad (4)$$

This dietary concentration equivalent can be directly compared to dietary concentrations of persistent organic chemicals known to cause adverse effects in mink.

Using the highest and lowest values for diet-to-tissue biomagnification factors (BMF_t) calculated from the literature (see the "Predicting tissue concentrations" section in Results and Discussion), we predicted concentrations of selected chemicals in mink tissue with the equation

$$C_t\left(\frac{\mu g}{g}\right)=C_f\left(\frac{\mu g}{g}\right)\times BMF_t. \quad (5)$$

Results and Discussion

Stable Isotope Analysis

Areas (AOC: in and out) and locations (lakeshore and inland) in which mink were captured had significant effects on $\delta^{15}N$ values (Table 1). Mink captured in the AOC had higher $\delta^{15}N$ values than mink out of the AOC ($P = 0.024$) and mink captured near the lakeshore had higher $\delta^{15}N$ values than inland mink ($P = 0.001$). The highest mean $\delta^{15}N$ ($13.2 \pm 0.5‰$) was in the AOC-lakeshore region, as was the highest individual $\delta^{15}N$ value (16.9‰). The lowest individual $\delta^{15}N$ value (9.2 ‰) was found in the AOC-inland region. Areas and locations where mink were captured had no significant effect on the $\delta^{13}C$ values (Table 2), indicating similar dietary compositions.

Construction of the Bioaccumulation Model

Calculation of trophic level. Using the $\delta^{15}N$ value of 11.9 (grand mean of 40 mink in our study, Table 1) and the commonly accepted value of 3.4‰ $\delta^{15}N$ per trophic level, the average trophic level of mink in our study was 3.50. If we use 3.5‰ $\delta^{15}N$ per trophic level, as reported by Cabana and Rasmussen (1994) for the Lake Ontario food web, the trophic level of our mink averaged 3.40. The higher mean $\delta^{15}N$ of 13.2 for mink in the lakeshore/AOC region resulted in higher values for the trophic level of those mink (3.87 or 3.76 using 3.4‰ or 3.5‰ $\delta^{15}N$ per trophic level, respectively). All of these values agree well with estimates found in the literature; USEPA (1995a) reported estimates for mink prey ranging from 2.5 to 2.9 which would imply a mink trophic level of 3.5 to 3.9.

For modeling purposes, we chose 3.8 as the trophic level of mink for several reasons. As the ultimate purpose is to protect mink populations, we wanted to represent the mink in the AOC at greatest risk, those living near the lakeshore. We chose an intermediate value between the two estimates of trophic level for mink in the lakeshore/AOC region (3.76 and 3.87) because, although Cabana and Rasmussen (1994) studied Lake Ontario, they analyzed only the pelagic food web. Therefore, their estimate is not fully appropriate for the diet of mink that feed in the littoral zone of the lake, associated wetlands, or streams.

The mean $\delta^{15}N$ in mink from the lakeshore was 1.7‰ higher than the mean from inland areas (Table 1). This represents a one-half trophic level difference between lakeshore and inland mink. The mean $\delta^{15}N$ for mink in the AOC was 1.1‰ higher than the mean out of the AOC (Table 1), about one-third of a trophic level. If the lakeshore minks' diet includes a higher proportion of aquatic-based prey, then inferring a higher trophic level for lakeshore than inland mink may be confounded by the fact that aquatic primary producers typically have $\delta^{15}N$ values up to 8‰ higher than terrestrial plants. (Peterson et al. 1985, Fry 1991). However, the hypothesis that lakeshore mink feed at a somewhat higher trophic level than inland mink is supported by analyses of persistent organic chemicals in mink tissues (Haynes et al. 2009).

Aquatic portion of the diet. Potential wetland sources of ^{13}C include phytoplankton, C_3 vascular plants (terrestrial, emergent, floating-leaved, submersed), and epiphytic and filamentous algae. Keough et al. (1996) used $\delta^{13}C$ analysis to determine that the Lake Superior trophic web and that of an associated wetland were based on phytoplankton, but there was a 5-6‰ difference between the carbon signatures of the two, corresponding to the $\delta^{13}C$ of the dissolved inorganic carbon in the respective waters. This difference was greater than the differences between the various classes of primary producers in the wetland, including terrestrial producers (Keough et al. 1996). Also, Keough et al. (1996) considered only invertebrates and fish as consumers in the wetland, while we needed to include other mink prey such as the muskrat (*Ondatra zibethicus*) which consumes emergent vegetation as well as occasional terrestrial vegetation and small animals such as shellfish, fish, turtles, and frogs (Kurta 1995). Thus, having at least three probable carbon sources in our trophic web, insufficient information about their $\delta^{13}C$ values, and probable overlap between them, we concluded that we could not use Doucett's (1999) equation to calculate the proportion of aquatic foods in the diet of the mink in our study.

We did, however, find several estimates in the literature for the aquatic portion of mink diets. Although most diet studies only report frequencies of occurrence of diet items in scats, digestive tracts, or dens (USEPA 1993 summarizes the results of 19 such studies), USEPA (1995a) points out that this is not a good representation of biomass assimilated by mink. However, USEPA (1995b) cites a study by Alexander (1977) reporting that the aquatic portion of minks' diets was 75% to 90%, based on wet mass of stomach contents year-round. Sample and Suter (1999) averaged the results of five studies to conclude that the aquatic portion of minks' diets is 54.6%. (The standard deviation for that average was reported as ± 0.21%, which seems very low, as it included Alexander's 1977 study; it is much more likely that the standard deviation was actually 21%). USEPA (1995b) used both 90% and 50% to calculate Wildlife Values for DDT, Hg, 2,3,7,8-TCDD, and PCB; therefore, we chose the same bounds on the aquatic portion of the diet of our mink.

Modeling exposure of mink in the AOC to persistent organic chemicals. Other values needed for the model are the body mass of the animal (g), daily consumption rates of food (g/day) and water (L/day), the bioaccumulation factor (BAF) of the chemical of concern (which also requires knowing the k_{ow} of the compound), and the concentration of the chemical in the water. The mean body mass (pelt-less, tail-less) of females in our study was 457 ± 42 g while males averaged 782 ± 27 g. Because we had six females and 35 males, we averaged the male and female means for a representative (pelt-less, tail-less) average body mass of 620 g. We then had to correct for the absence of tails and pelts on the mink received from trappers since we presumed that the body mass in the model would have included these. The tails that we removed from mink (n=16) averaged 1% of the body mass of those mink, and Aulerich et al. (1999) give the mass of a mink pelt (excluding the tail) as 17% of whole body mass. Subtracting the contributions of the pelt and tail (18%) from the whole body mass (100%) and taking the inverse of 82% gave a multiplying factor of 1.22 to convert from our average tail-less, pelt-less carcass mass to an estimated average whole body mass of about 760g.

Several sources give daily food and water consumption rates along with body masses of mink. Sample and Suter (1999) cited Bleavin's and Aulerich's (1981) value of 137 g of food per day and estimated daily intakes of 0.099 L of water, using a model by Calder and Brown (1983), for mink averaging 970 g body mass. USEPA (1995b) estimated intakes of 177 g of food (using an allometric model by Nagy 1987) and 0.081 L water per day (using Calder's and Brown's 1983 model) for mink with a body mass of 800 g. For captive adult males averaging 2200 g, Aulerich et al. (1999) reported that they drank 0.127 L/day and daily food consumption ranged from 147 g to 275 g depending upon the caloric content of the food and the season. The extrapolated whole body mass for our largest mink was only 1350 g, based on the estimated pelt and tail mass for a carcass of 1110 g. Since we wanted to make our model conservative (protective of mink at the highest trophic level) but not unrealistic, we discounted Aulerich's laboratory consumption rates as too high, and chose the larger of the remaining two values for daily food and water intakes. Thus, for our model, the daily food and water consumption rates were 177 g and 0.1 L (= 100 g), respectively.

The k_{ow} and BAF values used in the model were taken from Sample et al. (1996), who assumed that all fish consumed by mink are trophic level 3 (small fish). However, Melquist et al. (1981) reported that mink feed on kokanee (land-locked *Oncorhyncus nerka*) after spawning; therefore, it is probable that mink near the Lake Ontario shore feed on the abundant dying and dead piscivorous salmonines available every fall. Still, the average trophic level of 3.8 for mink in Braddock Bay Wildlife Management Area (Figure 1) indicates that if salmonines (trophic level 4) do contribute a significant portion of the minks' diet, they are balanced by a comparable portion of level 2 aquatic prey such as aquatic invertebrates. The BAF factor we used for Hg was for methyl mercury chloride (Sample et al. 1996), whereas we used measured concentrations of total Hg for both lake water and mink brain. Because >90% of mercury in mink brain is methyl-Hg (Evans et al. 2000) and because of the success of the model (see below), it appears that the BAF provided by Sample et al. is appropriate. Therefore, we used the BAF factors provided by Sample et al. (1996) for prey of trophic level 3, which is slightly higher and thus more protective than the prey from a trophic level of 2.8 implied by our results.

When our results and assumptions (summarized in Table 3) are incorporated into the equation for exposure, the equation becomes

$$Exp = \frac{C_w[100\,g + (177\,g \times P_{aq} \times BAF)]}{760\,g}.$$

Given C_w (the concentration of a chemical in the water), P_{aq} (the aquatic proportion of the diet), and BAF (the bioaccumulation factor of the chemical of concern at trophic level 3), the results of this equation are the estimated concentrations of the chemical to which a mink at the highest trophic level among the four regions of study would be exposed daily.

Predicting tissue concentrations. Multiplying the exposure concentrations by diet-to-tissue biomagnifications factors (BMF) from the literature yielded predicted tissue concentrations of a chemical. Table 4 shows biomagnification factors (BMF_t) calculated from the literature (see below) and resulting estimated concentrations of total PCB, dioxin-furan TEQ, and total Hg in mink tissue. The low predicted values were calculated using the lowest C_w found in either Luckey and Litten (2005) or in Environment Canada's 2004 survey of Lake Ontario (J. Vincent, personal communication), assuming

50% aquatic diet, and using the lowest diet-to-tissue BMF calculated from the literature (Hg: Wobeser et al. 1976; TEQ: Heaton et al. 1995, Tillitt et al. 1996; PCB: Bursian et al. 2006a, b). The high predicted values were calculated using the highest C_w, 90% aquatic diet, and highest BMF calculated from the literature (Hg: Wobeser et al. 1976; TEQ: Heaton et al. 1995, Tillitt et al. 1996; PCB: Halbrook et al. 1999). The measured concentrations in these tissues were reported by Haynes et al. (2009).

Comparison of Estimated and Actual Chemical Concentrations

Table 4 compares estimated low and high values of total PCB, dioxin-furan TEQ, and total Hg to the lowest and highest tissue concentrations found in liver (PCB, TEQ) and brain (Hg) of lakeshore mink (Haynes et al. 2009). The model worked well; the estimated low and high values bounded measured values with two minor exceptions. The predicted low bound for total PCB (19.2 ng/g) was 5.6 ng/g or 29% higher than the lowest measured concentration for a lakeshore mink (mink 53, 13.6 ng/g; see Haynes et al. 2009). Mink 17 (1.55 mg/kg Hg; see Haynes et al. 2009) exceeded the predicted high bound for total Hg (1.34 mg/kg) by 0.21 mg/kg (14%) but it also had the highest δN of all mink in the study (16.9‰), an entire trophic level higher than the other mink in the lakeshore/AOC region which had the highest mean δN (13.2‰) among the four regions in the study area (Table 1). Mink 17 also had exceptionally high concentrations of total PCB and dioxin-furan TEQ in liver and adipose (Haynes et al. 2009).

Conclusion

We used stable isotope analysis to evaluate mink diets in terms of trophic levels and terrestrial and aquatic food sources. Analysis of $\delta^{15}N$ showed that mink in the study area feed on prey at an average trophic level of 2.5 (with the highest level, 2.8, along the lakeshore in the AOC). The percent aquatic diet could not be determined for lack of $\delta^{13}C$ values for carbon sources in the wetland areas inhabited by mink. Using the $\delta^{15}N$ stable isotope results we constructed a food web/bioaccumulation model for mink in the Rochester Embayment AOC which gave good estimates of actual tissue concentrations of dioxin-furan TEQ and total PCB in mink liver and of total Hg in mink brain.

There are uncertainties associated with each of the assumptions we made in constructing the model. For example, we used averages for the trophic level and body mass of the mink, and chose from among literature values for food and water consumption, percent aquatic diet, and diet-to-tissue biomagnification factors. In each case, our aim was to use realistically conservative values that would be protective of mink in the study area. Nevertheless, using an average trophic level means that some mink feed at a higher trophic level than that calculated by the model (average prey level = 2.8) and are more at risk than the model implies. (This was certainly the case for 2-3 mink in our study; see Haynes et al. 2009.) However, this possibility is mitigated by the fact that Sample et al. (1996) provided BAF factors only for integer trophic levels, requiring us to use prey BAF factors for trophic level 3, thus making the model more protective.

The bioaccumulation model developed in this study can be used to estimate bounds for concentrations of dioxin-furan TEQ, total PCB, and total Hg in mink tissues as new data on the concentrations of these chemicals in Lake Ontario water become available. Comparing results of the model to NOAELs or LOAELs for these compounds in mink, as the initial step in a biomonitoring program, is preferable to sacrificing mink that would not otherwise be caught by trappers.

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Table 1. $\delta^{15}N$ values for mink in four areas, inland and near the shore of Lake Ontario and in and out of the Rochester Embayment Area of Concern (AOC)**,** and the results of the General Linear Model for AOC: in vs. out and lakeshore vs. inland. See Haynes et al. (2009) for locations of study areas.

Area	N	Mean $\delta^{15}N$ (SE) (‰)	Min δN (‰)	Max δN (‰)	P
Entire Study	40	11.9 (0.26)	9.2	16.9	
AOC: In	20	12.4 (0.40)	9.2	16.9	
Inland	10	11.6 (0.49)	9.2	14.6	
Lakeshore	10	13.2 (0.54)	11.1	16.9	
AOC: Out	20	11.3 (0.31)	9.4	14.3	0.024
Inland	10	10.5 (0.25)	9.4	11.6	
Lakeshore	10	12.2 (0.42)	10.5	14.3	
Inland	20	11.0 (0.30)	9.2	14.6	
Lakeshore	20	12.7 (0.35)	10.4	16.9	0.001

Table 2. $\delta^{13}C$ values for mink in four areas, inland and near the shore of Lake Ontario and in and out of the Rochester Embayment Area of Concern, and the results of the General Linear Model for AOC: in vs. out and lakeshore vs. inland. See Haynes et al. (2009) for locations of study areas.

Area	N	Mean $\delta^{13}C$ (SE) (‰)	Min δC (‰)	Max δC (‰)	P
Entire Study	40	-25.4 (0.24)	-28.3	-19.9	
AOC: In	20	-25.1 (0.36)	-28.1	-19.9	
Inland	10	–25.3 (0.24)	–26.6	–24.4	
Lakeshore	10	-25.0 (0.70)	-28.1	–19.9	
AOC: Out	20	-25.6 (0.31)	-28.3	-23.1	0.315
Inland	10	-26.0 (0.49)	–27.0	–23.1	
Lakeshore	10	-25.3 (0.39)	-27.0	-23.2	
Inland	20	–25.6 (0.28)	-28.3	-23.1	
Lakeshore	20	–25.6 (0. 39)	-28.1	-19.9	0.319

Table 3. Critical data used in construction of the bioaccumulation model. K_{ow} is the octanol-water constant, TEQ is dioxin-furan toxic equivalents, PCB is polychlorinated biphenyls, Me-Hg is methyl-mercury, and BAF is the bioaccumulation factor for prey at trophic level 3 (Sample et al. 1996; K_{ow} for Me-Hg was not given).

Trophic level of mink prey	3
Aquatic portion of mink diet	50-90%
Body weight of mink	760 g
Daily food intake	177 g
Daily water intake	0.1 L
K_{ow} TEQ	6.53
K_{ow} PCB	6.5
K_{ow} Me-Hg	N/A
BAF TEQ	172,100
BAF PCB	1,850,000
BAF Me-Hg	27,900

Table 4. Lowest and highest values for predicted and measured concentrations of dioxin-furan toxic equivalents (TEQs), total polychlorinated biphenyls (PCBs), and total mercury (Hg) in Lake Ontario shoreline mink. Predicted values are based on Lake Ontario water concentrations (C_w: Luckey and Litten 2005; J. Vincent, Environment Canada, personal communication) and diet-to-tissue biomagnification factors (BMF; Wobeser et al. 1976, Heaton et al. 1995, Tillitt et al. 1996, Halbrook et al. 1999, Bursian et al. 2006a, b). Measured values were reported by Haynes et al. (2009).

Chemical (tissue)	Bound	C_w (pg/kg)	BMF	Tissue Level Predicted	Tissue Level Measured
				ng/kg	ng/kg
TEQ (liver)					
	Low	0.00006	10.7	0.06	0.2
	High	0.024	16.7	62.1	47.6
				ng/g	ng/g
PCB (liver)					
	Low	26.0	0.8	19.2	14.7
	High	915	15.5	23602	5871
				mg/kg	mg/kg
Total Hg (brain)					
	Low	440	1.0	0.006	0.090
	High	5130	10.4	1.340	1.550

Figure 1. Box plots of $\delta^{15}N$ values for mink in four areas, inland and near the shore of Lake Ontario and in and out of the Rochester Embayment Area of Concern (AOC).

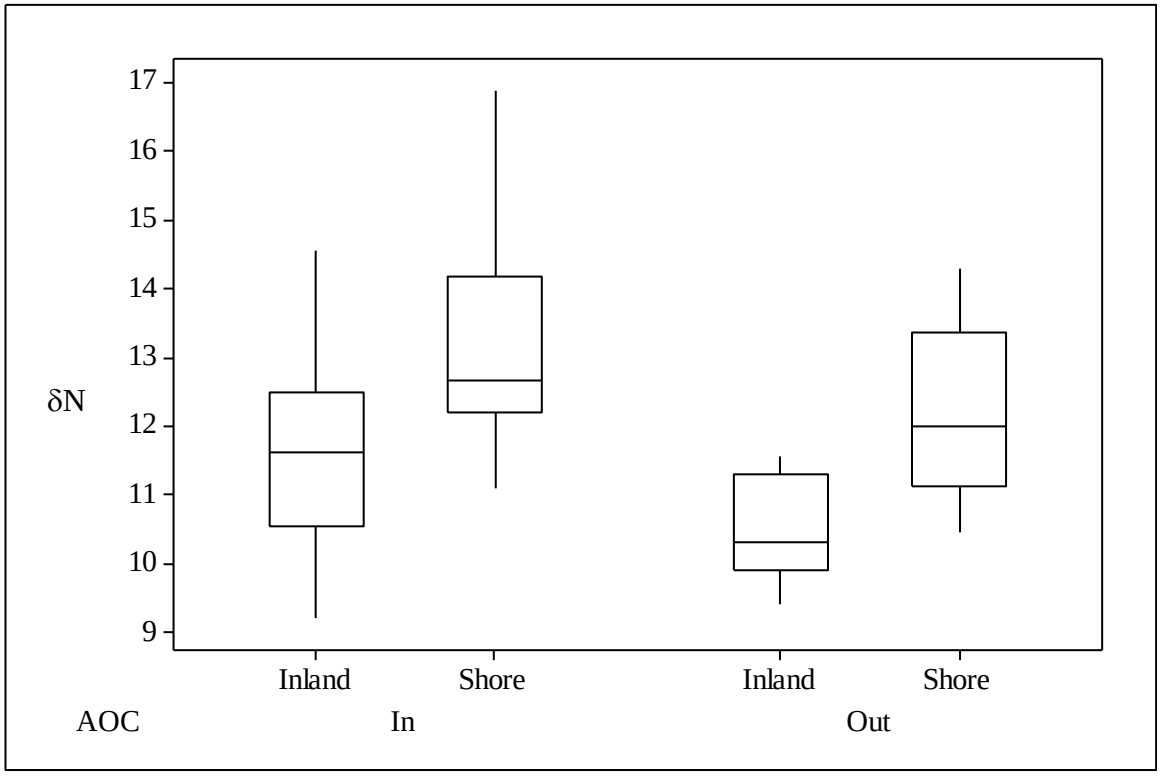

Figure 2. Box plots of $\delta^{13}C$ values for mink in four areas, inland and near the shore of Lake Ontario and in and out of the Rochester Embayment Area of Concern (AOC).