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Seasonal patterns of mercury species in water and plankton from softwater lakes in Northeastern Minnesota

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Abstract. Twelve softwater lakes in NE Minnesota were sampled in spring, summer, and fall of 1992 and 1993 for labile (unextracted) methyl-Hg, total (extracted) methyl-Hg, and total Hg in lake water and net plankton ($\geq$300 μm). The lakes are small (5.6–56 ha), low productivity, headwater drainage or seepage lakes. They are acid-sensitive (ANC $\leq$200 μeq/L) but not low pH lakes (average pH 6.6). The lakes ranged in color from 8.5 to 70 PCU. Statistical analysis of the water chemistry variables and mercury species support the conclusion that these were a homogeneous set of lakes; therefore, seasonality of mercury forms was analyzed on combined (mean) data from the 12 lakes. Methyl-Hg in water declined throughout the growing season. Hg_T also declined sharply from spring to summer but increased again in the fall. In contrast to the methyl-Hg and Hg in water, concentrations in plankton were at the lowest levels in spring and rose to higher levels in summer. The mass of mercury in plankton increased from spring to fall, as did the methyl-Hg fraction, which increased from 20% of Hg_T in spring to 52% in autumn. Bioaccumulation factors (BAF) for methyl-Hg in net plankton increased over the growing season. Overall, *log* BAF for Hg_T in net plankton (wet wt.) was 4.45. *Log* BAF for methyl-Hg in plankton was 4.90 to 5.43 depending on the analytical form of methyl-Hg in water (labile or total). Seasonal patterns of methyl-Hg and Hg_T did not covary in water, but did covary in plankton. These results support the conclusion that measurement of Hg in water is not adequate in itself to determine the amount of bioavailable Hg (i.e., methyl-Hg) in a lake. Labile (unextracted) methyl-Hg could be a useful measurement of bioavailable Hg. Labile methyl-Hg exhibits the same seasonal patterns as total methyl-Hg, but does not require the extraction steps necessary for measuring total methyl-Hg.

Introduction

The widespread occurrence of moderate to high levels of Hg in game fish has led to the issuance of fish consumption advisories on lakes throughout North America and Europe. To understand the causes of this problem, researchers are attempting to identify the physical, chemical and biological factors that affect Hg cycling in aquatic ecosystems. Synoptic lake surveys have shown that Hg levels in fish (fish-Hg) are correlated with several water chemistry variables. Acid neutralizing capacity (ANC) and pH are often negatively correlated with

fish-Hg (e.g. Cope et al. 1990; Driscoll et al. 1994; Grieb et al. 1990; Haines et al. 1992; Lee & Iverfeldt 1991; Lathrop et al. 1991; McMurty et al. 1989; Nilsson & Håkanson 1992; Swain & Helwig 1989). Dissolved organic carbon (DOC) and/or color has been positively correlated with fish-Hg in surveys that included mostly drainage lakes (e.g. McMurty et al. 1989) and negatively correlated in seepage lakes (Cope et al. 1990; Grieb et al. 1990). Correlations do not demonstrate cause-and-effect, but these results suggest that organic carbon, pH, and ANC influence the chemistry and bioaccumulation of Hg.

In this study we looked for the influence of these chemical and physical factors on Hg and methyl-Hg in zooplankton because of their importance in the pelagic food chain. Zooplankton are food for many fish species and predaceous aquatic invertebrates. Based on measurements of mercury uptake in *Daphnia pulex* and *Gambusia affinis* (mosquito fish), Huckabee et al. (1975) concluded that mercury in fish was primarily derived from their food, but most of the mercury in zooplankton was accumulated directly from water. Therefore, the concentration of Hg in zooplankton is directly related to the bioavailable fraction of Hg in water. Bioavailable Hg is generally considered to be methyl-Hg, but it has not been shown that all methyl-Hg is bioavailable. The supply of bioavailable mercury to zooplankton is certainly a function of the total deposition of mercury entering a lake and in-lake processes. The importance of in-lake processes is exemplified by the observation of a greater fraction of methyl-Hg in zooplankton from low pH lake water than in neutral pH water (Watras & Bloom 1992). Thus, examination of water chemistry and Hg species in zooplankton should provide insights into the physicochemical influences on bioavailability of Hg.

Our objectives were to quantify statistical distributions and relationships of mercury levels in water and plankton, and among water chemistry variables and mercury species. Unlike some studies that have drawn conclusions from a single year of monitoring, we made two years of seasonal measurements on 12 lakes of similar chemistry. We measured both labile (unextracted) and total (extracted) methyl-Hg in the water column, because of their potential differences in bioavailability. In zooplankton, we measured Hg_T and methyl-Hg. The main water chemistry variables of interest were ANC, pH, and DOC, because they have been correlated with fish-Hg in other lake surveys, but other water chemistry variables (e.g., SO_4^{2-}) also were examined.

Methods

Site description. The 12 lakes selected for this study are in a forested, glaciated region, at the southern end of the Canadian Shield in northeastern Minnesota (Figure 1). The lakes were selected for long-term monitoring of acidification,

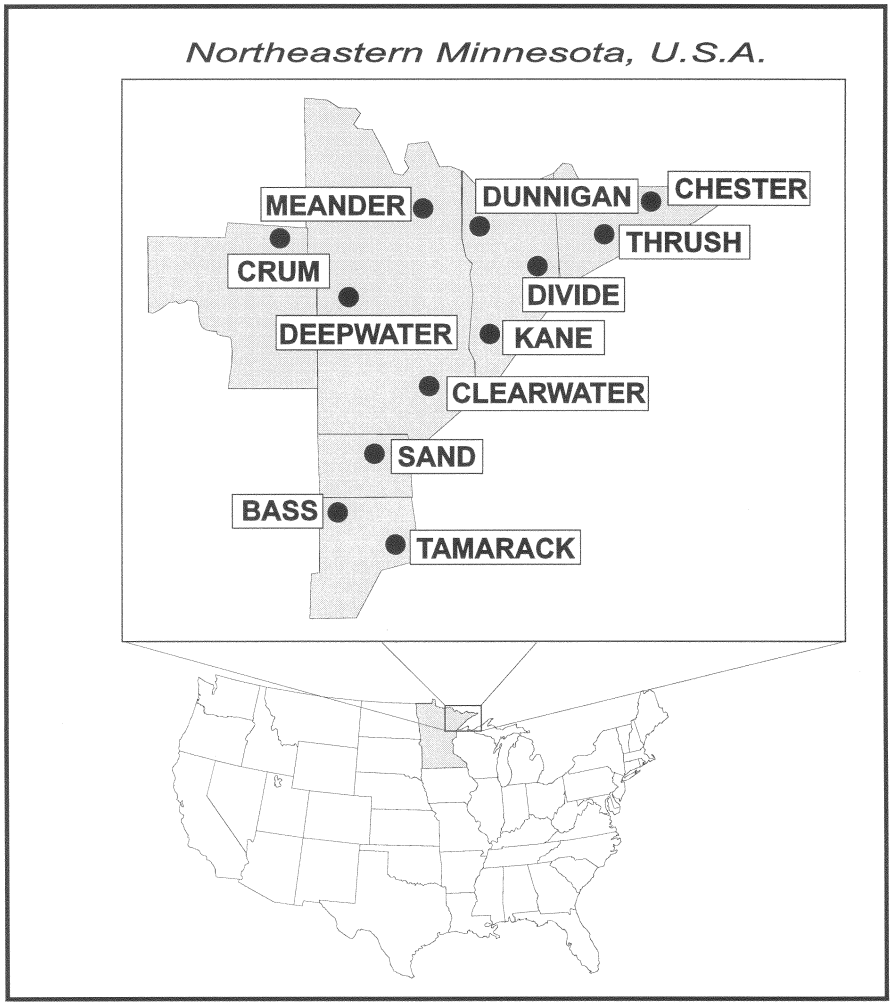

Figure 1. Location of lakes monitored.

and all have acid neutralizing capacities (ANC) <200 μeq/L (Table 1). However, they are not low-pH lakes (average pH 6.6). They are low productivity (chlorophyll *a* <15 μg/L), headwater drainage or seepage lakes, ranging in size from 5.6 to 56 ha and maximum depths from approximately 4 m to 15 m. Average color ranges from 10 to 73 PCU. The lakes were sampled in 1992 and 1993, during the months of May, August, and October.

Bottle preparation. Teflon bottles for Hg samples were rigorously cleaned before use. New bottles were heated to 95 °C in concentrated HNO_3 for 48

Table 1. Morphometric and physicochemical characteristics of the 12 study lakes in northeastern Minnesota. Chemistry values are mean ±1 standard deviation and the range is given in parenthesis.

Lake	Morphometry		Water chemistry				
	Max. depth (m)	Surface (ha)	Chl *a* (μg/g)	pH (SU)	ANC (μeq/L)	DOC (mg/L)	Color (PCU)
Bass	3.7	12.9	4.2 ± 2.31 (1.7–6.9)	6.2 ± 0.12 (6.2–6.5)	32.8 ± 6.13 (28.3–42.2)	6.5 ± 0.26 (6.1–6.8)	32.0 ± 10.76 (21.5–51.8)
Chester	10.7	19.7	2.4 ± 1.47 (0.8–4.8)	6.8 ± 0.15 (6.2–7.2)	104.5 ± 8.79 (94.3–115.2)	5.9 ± 0.17 (5.7–6.1)	23.7 ± 3.24 (20.4–28.1)
Clearwater	7.6	5.6	6.2 ± 4.54 (1.4–14.8)	6.7 ± 0.36 (6.3–7.4)	88.3 ± 11.11 (74.0–103.7)	6.9 ± 0.71 (6.0–8.1)	36.8 ± 20.10 (24.7–77.1)
Crum	4.0	8.4	3.1 ± 1.59 (1.3–5.3)	6.2 ± 0.17 (6.0–6.4)	23.3 ± 7.76 (14.2–35.4)	6.1 ± 0.57 (5.6–7.2)	12.2 ± 1.93 (9.2–14)
Deepwater	9.1	8.5	2.6 ± 1.63 (0.4–5.4)	6.4 ± 0.24 (6.2–6.8)	52.0 ± 7.80 (39.2–59.1)	5.9 ± 0.73 (5.2–7.3)	15.0 ± 7.20 (7.2–28.0)
Divide	4.6	27.7	2.7 ± 1.92 (0.6–5.3)	6.3 ± 0.15 (6.0–6.4)	29.7 ± 7.49 (20.3–40.1)	4.5 ± 0.17 (4.2–4.6)	8.2 ± 1.91 (4.2–12.8)
Dunnigan	5.2	33.7	2.9 ± 1.81 (1.1–6.3)	6.5 ± 0.07 (6.4–6.7)	58.6 ± 9.24 (44.8–70.2)	6.7 ± 0.36 (6.1–7.1)	18.0 ± 2.90 (13.8–21.7)
Kane	5.2	43.4	2.7 ± 1.18 (1.0–4.2)	6.8 ± 0.10 (6.7–7.0)	88.1 ± 7.16 (79.0–97.7)	7.5 ± 0.59 (6.9–8.3)	32.6 ± 6.27 (25.4–41.9)
Meander	7.6	56.3	2.1 ± 0.79 (0.6–3.0)	6.6 ± 0.18 (6.5–6.9)	69.8 ± 5.70 (64.4–80.3)	6.3 ± 0.12 (6.2–6.5)	21.1 ± 3.69 (16.8–25.8)
Sand	8.2	49.8	2.6 ± 1.41 (0.9–5.0)	6.5 ± 0.11 (6.4–6.6)	61.3 ± 9.78 (47.6–74.1)	10.2 ± 0.40 (9.8–10.8)	73.3 ± 14.70 (58.4–94.0)
Tamarack	14.3	32.1	2.9 ± 1.91 (1.1–6.2)	6.8 ± 0.34 (6.5–7.3)	96.7 ± 9.89 (81.3–110.0)	5.9 ± 0.46 (5.2–6.6)	16.4 ± 14.52 (7.1–45.5)
Thrush	14.6	6.4	1.0 ± 0.41 (0.5–1.7)	6.8 ± 0.24 (6.5–7.2)	112.5 ± 5.16 (105.4–120.2)	4.7 ± 0.44 (4.2–5.3)	10.0 ± 3.80 (6.4–16.6)
All lakes (n = 72)			3.0 ± 2.20 (0.4–14.8)	6.6 ± 0.30 (6.0–7.4)	68.1 ± 30.11 (14.2–120.2)	6.4 ± 1.48 (4.2–10.8)	24.9 ± 19.30 (4.2–94.0)

h, cooled, rinsed with distilled-deionized water (DDW), filled with 1% HCl, and heated again for 24 h. They were rinsed again and dried in a laminar-flow hood fitted with a gold-coated air filter. Clean bottles were double bagged in new polyethylene zip-lock bags and stored in a clean room until used. For subsequent cleaning, bottles were soaked in 50% HNO_3 overnight, rinsed three times with DDW, dried in a hood, and double-bagged.

Field method. Water samples were collected just below the water surface into 1-L opaque, TEFLON PTFE bottles using clean techniques. Water samples were collected in polyethylene bottles for other water chemistry analyses. Samples were kept on ice in the field and at 1–4 °C in the laboratory. Zooplankton samples were collected with a 300 μm mesh nitex net fitted with a Minnesota Bucket and thus consisted of larger zooplankton without apparent phytoplankton or detritus. Zooplankton samples for Hg were transferred to acid-cleaned 125-mL TEFLON PFA bottles, immediately put on ice, and frozen upon return to the laboratory.

Analytical methods

Mercury. Methyl-Hg and Hg_T were analyzed by Cold Vapor Atomic Fluorescence Spectrometry (CVAFS) (Bloom 1989). Hg_T in water was oxidized by BrCl, followed by reduction to Hg^0 by $SnCl_2$ (Bloom & Fitzgerald 1988). Hg^0 was purged from aqueous samples with N_2 and concentrated onto gold-coated sand. The amalgamated Hg was thermally desorbed to an analytical gold trap with He gas, thermally desorbed and carried into an AFS detector. The Hg peak was quantified by peak height, with reference to a standard curve.

Methyl-Hg analysis was adopted from Bloom (1989). Total methyl-Hg was extracted from water samples by methylene chloride and back extracted to mercury-free water. Ethylated methyl-Hg and mercuric ion (Hg^{2+}) were collected on Carbotrap®, thermally desorbed, separated by chromatography, thermally decomposed, and carried into an AFS detector by He. Hg species were identified by retention time and quantified in the same manner as total Hg. Total methyl-Hg is assumed to include free methyl-Hg dissolved in water and methyl-Hg bound to particles. Labile methyl-Hg was determined by analyzing water samples by the preceding method minus solvent extraction. Labile methyl-Hg is assumed to included unbound and weakly bound methyl-Hg.

For total Hg in zooplankton, a subsample from lyophilized samples was digested in concentrated H_2SO_4 and HCl at 90 °C for 2 h and diluted to 100 mL with distilled-deionized water (DDW). A 1 mL aliquot of digestate was added to pre-sparged 100 mL DDW and analyzed for total Hg in the same manner as a water sample. Methyl-Hg and Hg^{2+} were simultaneously determined in zooplankton samples following the method of Liang et al. (1994). Samples were extracted in 25% methanolic KOH at 60 °C for 2 h, and 0.2–1.0 mL of extract added to pre-sparged 100 mL DDW was analyzed in the same way as a water sample.

Data quality assurance was based on adherence to clean techniques, reference standards (e.g., DORM II), and frequent analysis of standards and

replicates in each analytical run. Data accuracy was monitored by analyzing known standards after at least every 20 samples. Precision was monitored by analyzing replicates for at least every tenth sample. In nearly all methyl-Hg runs, blanks were indistinguishable from zero; methyl-Hg is not a common contaminant in air or reagents. Blanks for Hg_T typically were in the range of 10 to 40 pg/100 mL. The method detection limit (MDL) for Hg_T was 0.05 ng L^{-1} and for methyl-Hg was 0.013 ng L^{-1}. Coefficients of variation for Hg_T and methyl-Hg replicates were $\leq$7%.

Dissolved organic carbon (DOC) and color. Dissolved organic carbon (DOC) was determined with a Dohrman Model DC-80 carbon analyzer with a Horiba PIR-2000 detector. Organic color was measured on centrifuged samples spectrophotometrically at 420 nm using a chloroplatinate standard for calibration.

Other water chemistry. Because these lakes were monitored for acidification as well as Hg content, an extensive list of water chemistry variables was analyzed, including anions, cations, nitrogen, phosphorus, dissolved oxygen, and temperature. Water chemistry analyses followed standard methods (APHA 1989; USEPA 1987; see Owen & Axler 1991; Ameel et al. 1993). Sulfate was analyzed by ion chromatography, and acid-neutralizing capacity by Gran titrations. Chlorophyll was analyzed spectrophotometrically and fluorometrically on 90% acetone extracts (Axler & Owen 1994). QA/QC procedures were based on a detailed field and laboratory manual (Owen & Axler 1991).

Results

Correlations among mercury species and water chemistry parameters. Pairwise Pearson correlation coefficients among mercury species and water chemistry parameters (Table 2) were low ($\leq$0.3) and none was significant at $p < 0.05$. The strongest correlation among the variables was between Hg_T in zooplankton and water color (r = 0.30), but it still was not significant (p = 0.193). Water color exhibited the strongest correlation with Hg_T probably because color exhibited a wide range (8.5 PCU to 70 PCU); other water chemistry variables had narrow ranges. This is not surprising given the lakes were selected to have a similarly low ANC.

ANC & DOC grouping. Although the lakes in this study were selected for low ANC, they separated into three distinct groups within that variable. Based on the upper and lower quartiles of the three groups, the study lakes were

Table 2. Pairwise Pearson correlation coefficients for mercury species and water chemistry variables from the twelve lakes monitored in this study.

Hg species	Water chemistry parameter						
	Chl	pH	ANC	DOC	Color	TP	SO4
Water							
THg	0.110	−0.092	−0.012	0.225	0.197	0.228	0.092
TmeHg	−0.109	−0.152	−0.147	0.087	0.129	0.023	0.034
LmeHg	−0.095	0.061	0.139	0.096	0.177	0.159	−0.057
Plankton							
Thg	0.200	−0.223	−0.029	0.184	**0.304**	−0.084	−0.123
MeHg	0.189	0.101	0.124	−0.084	−0.098	−0.091	0.060

categorized into ANC classes of low (0–40 μeq L^{-1}; n = 3), medium (40–80 μeq L^{-1}; n = 4), high (80–120 μeq L^{-1}; n = 5) (Figure 2a). Median values of DOC for the three ANC groups (Figure 2b) were not significantly different. However, comparison of the upper and lower quartiles of DOC shows that ANC and DOC appeared to covary; the lowest DOC concentrations were in the lowest ANC group and the highest DOC in the highest ANC group.

No statistically significant trends were found in Hg species concentrations across the three ANC groups (Figure 3). Total methyl-Hg concentrations in water were highest in the low ANC group and lowest in the high ANC group, but the difference was not significant. Mean values of labile methyl-Hg were within a narrow range (0.03–0.04 ng/L) for the three groups. Mean concentrations of Hg_T and methyl-Hg in plankton were also not significantly different among the three ANC groups. These findings support the conclusion that the lakes in this study are relatively homogeneous, having similar water chemistry characteristics and Hg concentrations. This is different from typical synoptic surveys of lakes where the variability among the lakes is used to develop empirical relationships among the variables of interest. However, it provides a basis for combining data for the 12 lakes to make conclusions about low ANC lakes.

Seasonality. We examined seasonal changes in Hg species by combining the two years of seasonal data from the 12 lakes and compared means ($\pm$1 S.E.) for spring, summer, and fall (Figure 4). A seasonal pattern in Hg species' concentrations was clearly evident in lake water and in plankton. In water, methyl-Hg declined throughout the growing season; the average decrease from spring to summer was greater for total methyl-Hg than for labile methyl-Hg. Hg_T also declined sharply from spring to summer, but increased again in the fall. The high Hg_T levels in spring and fall likely reflect spring and

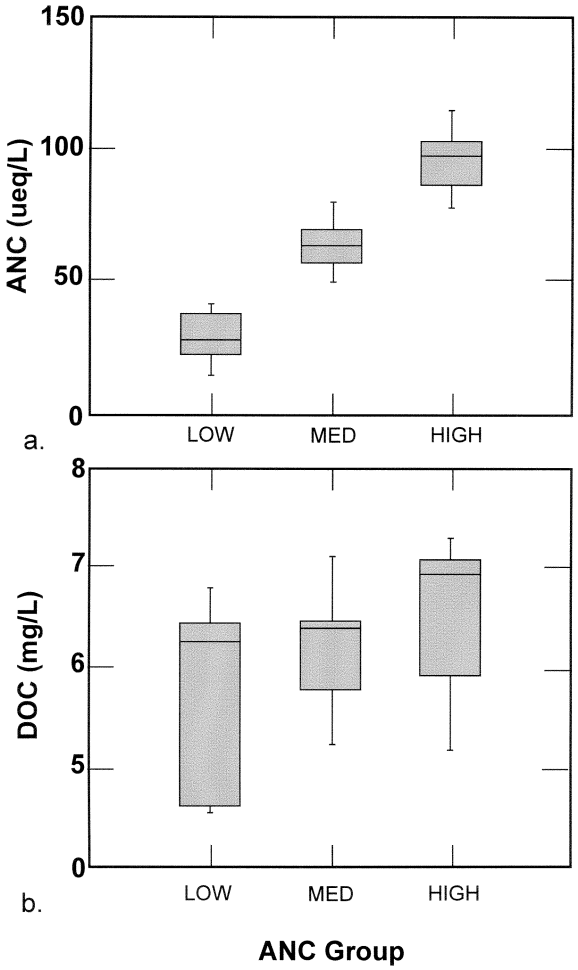

Figure 2. Box-whisker plots of (a) ANC grouping of the 12 lakes and (b) DOC concentrations within the ANC groups.

fall mixing and/or seasonal patterns in rainfall and runoff. Because these are seepage/headwater lakes rather than drainage lakes, the observed patterns are more likely caused by in-lake processes than by seasonal changes in hydrology.

In contrast to Hg concentrations in water, concentrations in plankton were at their lowest levels in spring, and highest (by a factor of four) in the fall (Figure 4). These patterns were repeated in 11 of the 12 lakes. In Tamarack Lake, which was sampled in more detail than the other lakes, the calculated total mass of methyl-Hg in zooplankton >300 μm was 3% of the mass in

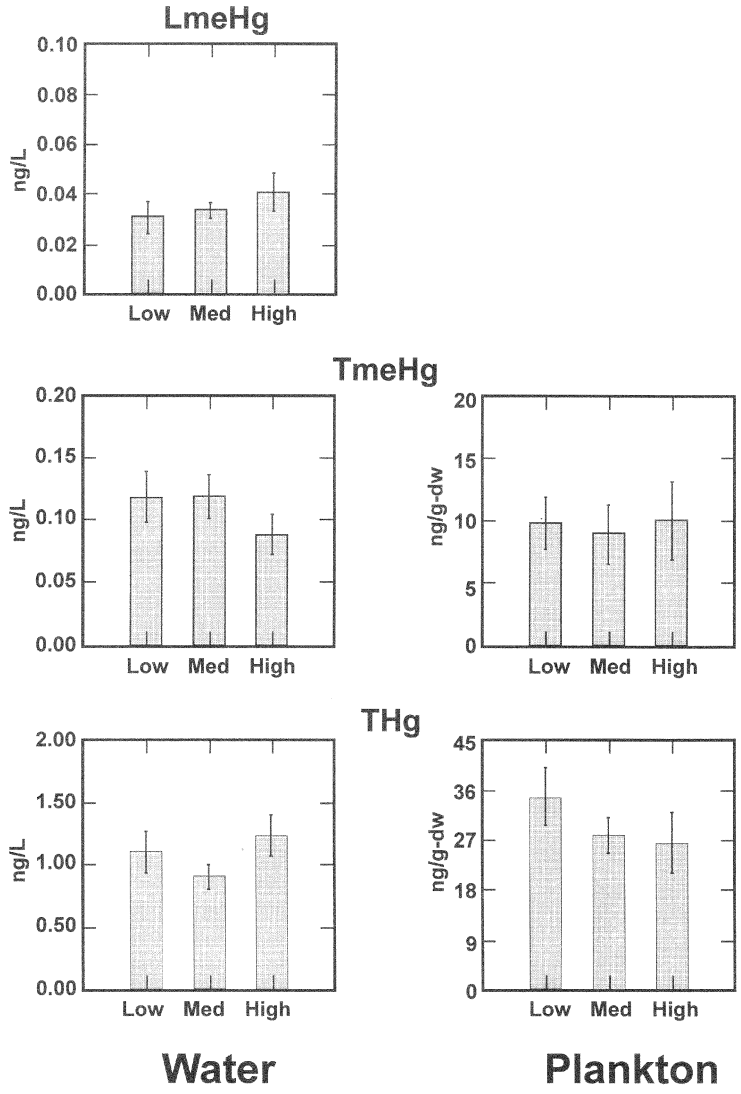

Figure 3. Comparison of Hg species by ANC group (mean ± SE).

water during spring, 6% in summer, and 7% in fall. Hg_T in zooplankton from Tamarack Lake was 0.2% of the mass in water during spring and 1.2% in summer and fall. Most of the zooplankton community is less than 300 μm; thus these are minimum estimates. These results support the conclusion that at least some of the methyl-Hg in water is repartitioned to zooplankton as the growing season progresses.

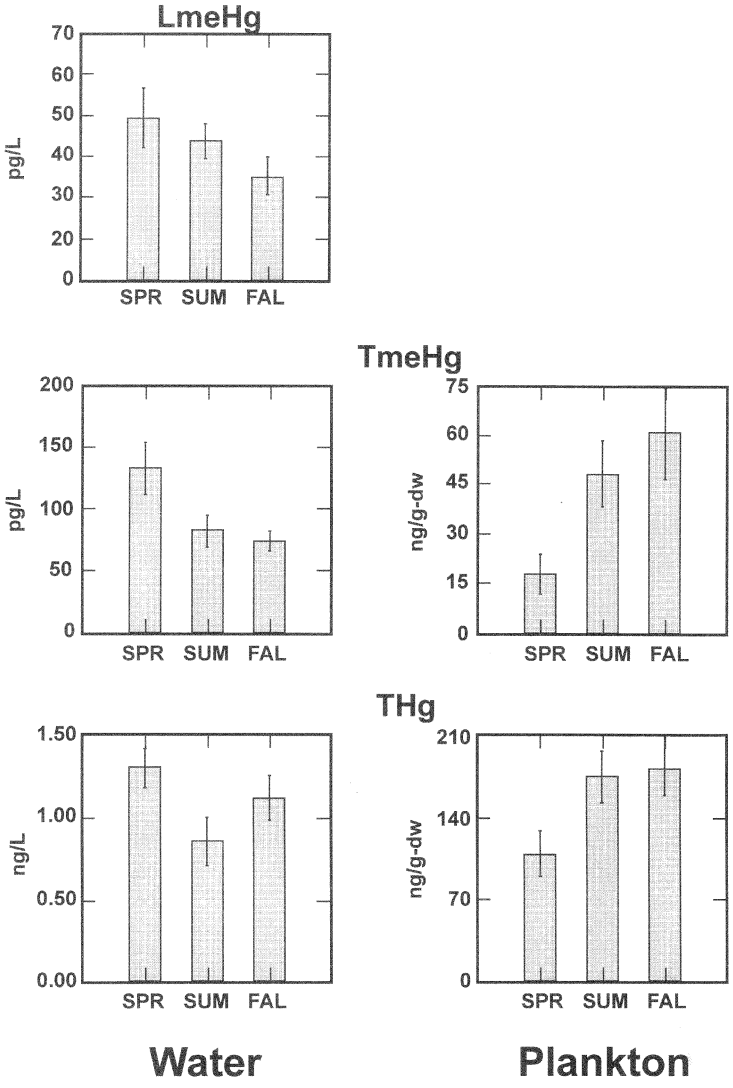

Figure 4. Comparison of Hg species by season (mean ± SE) for the 12 lakes.

The mean fraction of methyl-Hg in plankton (methyl-Hg/Hg_T) increased from 20% in spring to 52% in fall (Figure 5). The reason for the increasing fraction of methyl-Hg has yet to be evaluated. Over a growing season, zooplankton species and genotypes within species change, precluding the likelihood that the increases in methyl-Hg are occurring within the same individual organisms.

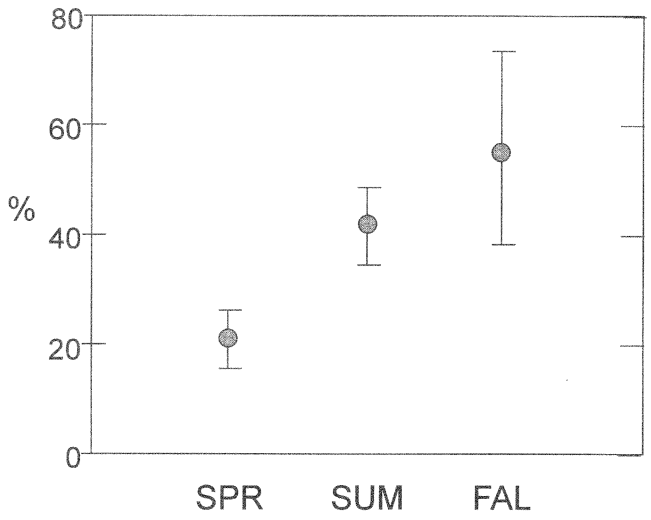

Figure 5. Mean (±1 S.E.) percent methyl-Hg (meHg/total Hg) in Zooplankton from the 12 lakes.

Bioaccumulation factors. The relative importance of uptake at the base of the food chain is evident from a plot of Hg in water, plankton, and fish from one of our lakes (Figure 6). There was about an order of magnitude difference between plankton and fish concentrations for both Hg_T and methyl-Hg, but the difference between water concentrations and plankton was 4.5 orders of magnitude for Hg_T and 5.5 orders of magnitude for methyl-Hg. Physical and chemical in-lake processes that influence the bioaccumulation of Hg in zooplankton also will greatly influence the levels of Hg throughout the rest of the pelagic food chain.

The bioaccumulation factor (BAF) relates ambient concentrations of chemicals to biotic tissue concentrations. The concept of BAF assumes an equilibrium between the organism and the environment: as environmental concentrations increase, equilibrium concentrations in organisms also are assumed to increase. Our results suggest that plankton are not in equilibrium with the water concentrations of methyl-Hg. As water concentrations of methyl-Hg declined, plankton concentrations increased, and BAFs for methyl-Hg in plankton thus increase with season (Figure 7). Labile methyl-Hg concentrations were a small fraction of the total methyl-Hg in lake water; therefore, $BAF_{Lmethyl-Hg}$ values were greater than $BAF_{Tmethyl-Hg}$. There was a significant increase in $BAF_{Tmethyl-Hg}$ from spring to summer, but spring and summer $BAF_{Lmethyl-Hg}$ were fairly constant. However, there was a large increase in $BAF_{Lmethyl-Hg}$ in fall.

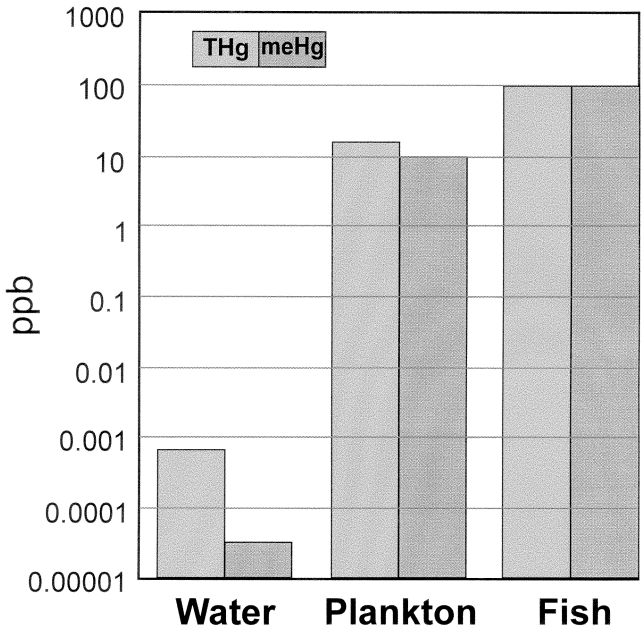

Figure 6. Total Hg (THg) and methyl-Hg (Me-Hg) concentrations in Tamarack Lake (water in μg/L, plankton and fish in ng/g-ww). Fish concentrations were from fillets of young-of-the-year blue-gill sunfish and plankton were collected in tows of a 300 μm mesh plankton net. Plankton were assumed to be 80% water.

Discussion

Total Hg and methyl-Hg in water were similar to other undisturbed seepage lakes and drainage lakes in temperate regions (Table 3a), despite differences in hydrologic balances among the lakes. It is more difficult to compare Hg concentrations in plankton from various studies because of differences in plankton net mesh sizes (Table 3b). The lowest concentrations tend to be found when smaller mesh size nets are used (e.g. Sorensen et al. 1990; mesh size 80 μm); considerably higher concentrations were measured in some Swedish lakes (Meili & Parkman 1988; mesh size 250 μm). Nevertheless, the ranges of Hg in plankton showed considerable overlap.

Our results are in general agreement with Meili & Parkman's (1988) observations on the seasonality of Hg in water and plankton. They examined Hg in plankton collected with a 250 μm mesh net from eight "small oligo-mesotrophic headwater" lakes in central and northern Sweden – four lakes with high levels of Hg in predatory fish and four with lower levels of Hg. They found the highest Hg levels in water in early spring, but peak Hg levels in plankton occurred in June. For the Minnesota lakes, seasonality of methyl-

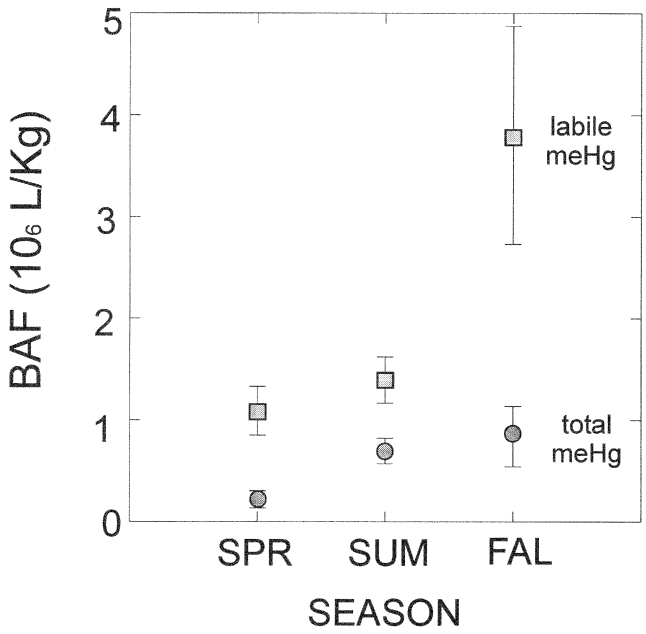

Figure 7. Mean (±1 S.E.) methyl-Hg BAF for zooplankton based on labile and total methyl-Hg from the 12 lakes.

Hg was not the same as that for Hg_T, and plankton-Hg concentrations were inversely related to seasonal methyl-Hg concentrations in water.

The lack of significant correlations of plankton-Hg with ANC, pH, DOC, sulfate, chlorophyll, and phosphorus was contrary to other reports from regional studies of Hg in lakes (although many of those studies have found only weak correlations with these water chemistry variables). The approach of this study was different from the others, however, in that the lakes were selected to be low ANC lakes rather than a broad spectrum to support regression analysis. The statistical analysis of Hg species and water chemistry parameters has demonstrated that this is a fairly homogenous set of lakes. They are arguably an optimum set of lakes for long term monitoring because they provide a statistical robustness that would not be available in a more heterogenous set of lakes.

No relationship was found between methyl-Hg and Hg_T in water from the 12 lakes in this study. Therefore, until there is more detailed understanding of the in-lake processes that affect methylation and methyl-Hg uptake in biota, methyl-Hg should be monitored in lakes, along with Hg_T. Measuring labile (unextracted) methyl-Hg is an alternative to the more laborius measurement of total methyl-Hg. Labile methyl-Hg exhibits the same seasonal patterns as total methyl-Hg, but the analysis of labile methyl-Hg avoids time consuming

Table 3. Summary of total Hg and Methyl-Hg in undisturbed lakes in temperate regions.

a. Lake water		Surface water			
			(ng/L Hg, unfiltered)		
System*	N	Location	Methyl-Hg	Total Hg	Reference
S, H	12	Minnesota	0.04–0.34 (total) 0.01–0.16 (labile)	0.2–3.2	This tudy
S	5	Wisconsin	0.04–0.15	0.5–2.2	Bloom & Watras 1989
S	2	Adirondacks	0.07–0.12	1.8	Driscoll et al. 1994
D	14	Adirondacks	0.03–0.70	1.4–6.5	Driscoll et al. 1994
D, S	10,2	Glacier National Park, Montana	0.01–0.10	0.35–2.85	Watras et al. 1995
D, S	12	Wisconsin	0.04–2.20	0.43–4.79	Back and Watras 1995
D	3	Ontario	0.02–0.08	0.2–1.1	Bloom & Effler 1990
D	4	Sweden	0.08–0.24	1.2–2.5	Lee 1987

b. Net plankton		Plankton			
			Net mesh size	Total Hg	
System*	N	Location	(μm)	(ng/g Hg, dw)	Reference
S, H	12	Minnesota	300	53–300	This study
D, S	53	Minnesota	80	10–209	Sorensen et al. 1990
D, S	12	Wisconsin	153[1]	20–153[2]	Back & Watras 1995
D, S	24	Ontario/Québec	225	26–377	Tremblay et al. 1995
S	2	Wisconsin	[1]	280[3,4], 375[3,5]	Watras & Bloom 1992
H	8	Sweden	250	100–700	Meili & Parkman 1988

∗ Systems: D – drainage
H – headwater
S – seepage

[1] Hand-picked zooplankton for analysis
[2] Predator taxa (e.g. Chaoborus)
[3] Herbivore taxa (e.g. Daphnia)
[4] Little Rock Lake, Reference Basin
[5] Little Rock Lake, Acidified Basin

extraction steps in the analysis of total methyl-Hg and thus provides a cost-saving advantage.

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