Arch. Environ. Contam. Toxicol. 37, 103–109 (1999)

ARCHIVES OF
Environmental
Contamination
and Toxicology

Uptake of Mercury by Fish in an Experimental Boreal Reservoir

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Received: 28 August 1998/Accepted: 20 January 1999

Abstract. We studied the uptake of mercury (Hg) by finescale dace (*Phoxinus neogaeus*) in an experimental boreal reservoir for 2 years prior to flooding and 3 years after flooding, and in a natural wetland pond over the same 5-year period. Hg uptake was much higher after flooding as compared to uptake before flooding and in the natural pond. After flooding, Hg concentrations in late summer were usually 2–3× higher than concentrations observed prior to flooding. Net uptake of Hg by fish over the summer in the experimental reservoir was 0.25 and −0.07 µg per fish in the 2 years before flooding as compared to 0.63, 0.64, and 0.42 µg per fish in the 3 years after flooding. Thus, Hg uptake by fish responded quickly to flooding and was highest in the first 2 years following impoundment. Uptake in the reference pond ranged from 0.10 to 0.28 µg of Hg per fish over the same 5-year period. Calculated fluxes of Hg on an areal basis ranged from 0.04–0.09 µg m^{-2} $year^{-1}$ in the reference pond, were 0.08 and −0.02 µg m^{-2} $year^{-1}$ in the experimental reservoir prior to flooding, and ranged from 0.14–0.22 µg m^{-2} $year^{-1}$ in the experimental reservoir after flooding. These fluxes were much smaller than fluxes of methyl mercury (MeHg) through the zooplankton and emerging insect communities. Most (71–89%) of the mercury measured in the muscle of finescale dace was MeHg, and the proportion that was MeHg decreased over the summer period prior to flooding, but increased over the summer after flooding. Growth of fish was not significantly affected by flooding. Fish ate predominantly benthic invertebrates (64–84% of food items found in stomachs), with lesser proportions of crustacean zooplankton (16–31% of items found in stomachs) and feeding was similar after as compared to before flooding. Therefore, differences in Hg uptake did not appear to be the result of changes in diet.

Mercury is by far the most common reason for human health advisories for fish consumption in North American fresh waters (*e.g.*, Ontario Ministries of Natural Resources and Environment & Energy 1997), and mercury problems in boreal reservoirs are probably the most significant environmental problem associated with reservoir development for hydroelectric power production (Bodaly *et al.* 1984; Johnston *et al.* 1991; James Bay Mercury Committee 1995; Mucci *et al.* 1995). In temperate and tropical areas, cases of elevated mercury levels are known (Abernathy and Cumbie 1977; Yingcharoen and Bodaly 1993; Roulet and Lucotte 1995), but the problem appears to be less serious compared with boreal reservoirs. The study of mercury in reservoirs is often hampered by sampling difficulties due the large geographic scales involved, by lack of control over flooding times and amounts, and by problems of the confounding of different effects. To control these factors, the Experimental Lakes Area Reservoir Project (ELARP) took the approach of creating an experimental reservoir. The overall objective of ELARP was to investigate mercury (Hg) cycling in a flooded boreal wetland ecosystem, especially to provide quantitative information on impacts of flooding. This information will prove useful for modeling, prediction, and mitigation of mercury problems in reservoirs.

The purpose of this paper as part of the ELARP study is to describe the uptake of mercury by fish before and after flooding of a boreal wetland pond and in a reference pond, and to make comparisons to other fluxes in the reservoir, including other parts of the food chain.

Materials and Methods

Study Area and Study Plan

Our study was conducted at the Experimental Lakes Area, northwestern Ontario, Canada. We followed mercury bioaccumulation by fish in an experimental reservoir, Lake 979, for 2 years before flooding (1991 and 1992) and for 3 years after flooding (1993–1995) and in a reference pond (Lake 632) for the same 5-year period. The experimental reservoir consisted of a central pond (2.4 ha) surrounded by a 14.3-ha wetland dominated by *Sphagnum* and black spruce. A 16.7-ha reservoir was created by flooding the surrounding wetland by raising the water by 1.3 m in June 1993. Water volume increased by a factor of about 10. Water levels were drawn down in late fall and were increased each spring. The reference system, Lake 632, was a 0.8-ha pond surrounded by a *Sphagnum*/black spruce wetland. Water level remained relatively constant in this system. See Kelly *et al.* (1997) for preliminary results for the first 2 years of flooding.

Experimental Lakes Area Reservoir Project (ELARP) contribution no. 43.

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Sampling Methods

Because Lake 979 had a transitory fish fauna and Lake 632 was fishless, we introduced finescale dace (*Phoxinus neogaeus*) to both lakes to follow mercury uptake by fish. A common source (a separate lake) was used for both lakes and fish were transfered in the early summer in each of the 5 years of the study. Fish were held in large pens (19.6 or 38.5 m^2) that were open to the bottom of the pond and had large mesh sizes (0.6 or 1.2 cm) to allow fish to feed on naturally available food. Both mesh sizes were able to contain the fish utilized. In both lakes, pens were placed in the pond in open water near shore each year. Also, we placed pens on flooded peat in Lake 979 in 1993 and 1994 (the first 2 years after flooding). Holding fish in pens on flooded peat was not successful in 1995 due to poor survival caused by floating peat, shallow water, and low dissolved oxygen. Initial stocking densities in both lakes were 3.7–9.7 g/m^2. We sampled fish from pens every 2–3 weeks over summer and early fall, measured fresh weight, and determined mercury concentrations.

Total mercury (THg) concentrations were measured for whole bodies, less the digestive tract, by cold vapor atomic absorption spectroscopy using the hot acid extraction method of Hendzel and Jamieson (1976) modified from Armstrong and Uthe (1971). Methyl mercury (MeHg) concentrations were determined for a subsample of fish, taken from Lake 979 at the beginning and end of the sample period in 1992 (before flooding) and in 1995 (after flooding). All values are reported on a wet weight basis; fresh weight was determined before freeze-drying. MeHg analysis was carried out on freeze-dried samples of dace white muscle by extraction into a methylene chloride/hexane mixture after tissue homogenization in a solution of NaBr and $CuSO_4$, followed by flameless atomic absorption spectroscopy, modified from Uthe *et al.* (1972). Reference material used was National Research Council of Canada DORM-1 and MeHg determinations were normally within the acceptable certified range for MeHg. THg was determined from freeze-dried muscle from the same fish, and results were expressed as the proportion of THg that was MeHg. Values that exceeded 100% (6% of data points) were not changed, following Bloom (1992), who argued that the practice of rounding down values greater than 100% artificially lowers estimates of the proportion that is MeHg.

We compared uptake of Hg by fish by calculating least square regression slopes and 95% confidence intervals of mean body burden versus $\log_{10}$ day.

To determine feeding patterns of finescale dace in the two study lakes, we removed stomachs from fish immediately after capture in minnow traps, preserved them in formalin, and identified and counted the number of each food item. Items were generally identified to major groups, usually Order. The percent occurrence of each food item was calculated based on nonempty stomachs only.

Results

Growth

In the reference pond, finescale dace grew gradually over the summer periods, from initial weights of 2.1–3.9 g, to 3.1–5.0 g (Table 1). Weight gains ranged from 0.7–2.1 g and averaged 1.2 g. Growth in the experimental reservoir was very similar to that in the reference pond. From the same initial weights, fish attained final weights of 3.6–4.7 g. Weight gains ranged from 0.5–1.9 g and averaged 1.3 g. There was no noticeable effect of flooding on fish growth. The mean weight gain in Lake 979 before flooding was 1.5 g (n = 2 years) as compared to 1.2 g after flooding (n = 5 years and locations within Lake 979). Also, growth in the experimental reservoir was similar to growth in the reference pond in a given year before and after flooding (Table 1). Growth after flooding in the experimental reservoir was similar for fish held in the pond or over flooded peat in the 2 years (1993 and 1994) for which comparisons were available (Table 1).

Table 1. Initial and final mean weights of finescale dace placed into and sampled from pens in Lake 979 (before and after flooding) and Lake 632 (reference pond)[a]

Year	Lake	Initial Weight (g)	Final Weight (g)	Change in Weight (g)
1991	979	2.82 (69)	4.27 (30)	1.45
	632		4.96 (34)	2.14
1992	979	2.14 (130)	3.62 (9)	1.48
	632		3.08 (28)	0.94
1993	979 (pond)	3.87 (50)	4.37 (47)	0.50
	979 (peat)		4.72 (62)	0.85
	632		4.53 (53)	0.66
1994	979 (pond)	2.29 (98)	4.09 (19)	1.80
	979 (peat)		4.16 (37)	1.87
	632		4.00 (66)	1.71
1995	979 (pond)	2.65 (60)	3.61 (75)	0.96
	632		3.32 (70)	0.67

[a] Initial weights are the same for both lakes in each year because pens were stocked from a common source at the same time. Fish held in pens in the pond or over the flooded peat after flooding in Lake 979 are shown separately. Sample sizes are shown in brackets after means

Mercury Concentrations in Fish

In the reference pond, THg concentrations were generally stable over the summer during the 5 years of observation (Figure 1). In 4 of the 5 years, THg concentrations started and remained near 0.1 µg/g (weight weight); however, in 1993, initial concentrations were higher and remained higher over the summer.

In the experimental reservoir, concentrations were similar to those in the reference pond in 1991 and 1992, prior to flooding; the final mean concentration was slightly higher in Lake 979 as compared to Lake 632 in 1991, but slightly lower in 1992 (Figure 1). As in the reference pond, concentrations before flooding did not tend to increase over the summer. In contrast, after flooding, concentrations increased noticeably over the summer, usually reaching 0.2–0.25 µg/g wet weight by the end of the summer. There was a clear separation of all pre- and postflooding samples by late summer (Figure 1). In the years that comparisons between fish held in different zones of the reservoir (pond and flooded peat) were possible (1993 and 1994), concentrations in fish were similar to both zones.

Mercury Body Burdens in Fish

In Lake 632, the reference pond, total body burdens of mercury generally changed little or increased slightly over the course of the summer in the 5 years studied (Figure 2). The greatest increase was in 1991 when mean body burden increased by 0.28 µg per fish over the summer and the smallest increase was in 1992 when burdens were little changed (0.03 µg per fish) over the summer. Other years were intermediate between 1992 and

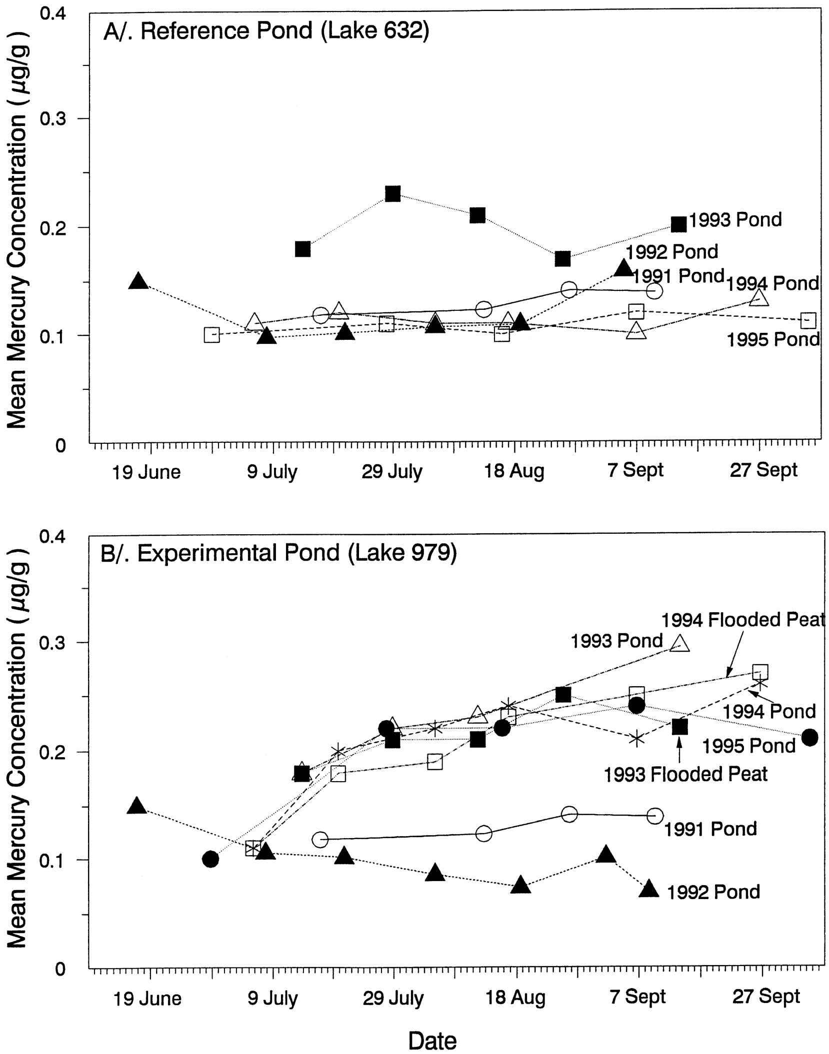

Fig. 1. Total mercury concentrations in finescale dace in Lake 632 (reference pond) and Lake 979 (before and after flooding). Initial sample sizes were 69 (1991), 130 (1992), 50 (1993), 98 (1994), and 60 (1995). Final sample sizes for both lakes ranged from 9–75 and averaged 44.2 (see Table 1). Intermediate sample sizes for Lake 632 ranged from 15–52 and averaged 33, whereas intermediate sample sizes for Lake 979 ranged from 8–51 and averaged 24

1991. Net uptake (change in mean body burden over the summer) on an areal basis ranged from 0.04–0.09 μg Hg m^{-2} $year^{-1}$ in the reference pond (Figure 3). The regression slope of burden versus time was highest in 1991 and similar in other years and all confidence intervals of slopes were overlapping for Lake 632 (Table 2).

In Lake 979, the experimental reservoir, prior to flooding, mercury body burdens were very similar to those in the reference pond; uptake in 1991 was much higher than uptake in 1992 (Figure 2). In fact, net uptake in Lake 979 in 1992 was negative (Figure 3). In the experimental reservoir after flooding, mercury burdens increased noticeably over the summer period in all 3 years. Final burdens were always greater than in the highest preflooding year and in the reference lake (Figure 2). Regression slopes of burden versus time were lower in 1991 and 1992 before flooding in Lake 979 as compared to after flooding (Table 2). Net uptake in Lake 979 generally followed the same year-to-year pattern as in Lake 632, except that net uptake was greatly elevated in Lake 979 after flooding (Figure 3). In Lake 979, net uptake in fish held over flooded peat was less than in the pond in 1993, but was similar in 1994 (Figure 2). Although the length of time that fish were held in pens in the two lakes was not the same in each year, body burdens were either largely unchanged over the summer or, if increasing over the summer, had reached a plateau by early September (Figure 2).

Net mercury uptake by fish in Lake 979 was highest in the first 2 years after impoundment and declined in the third year of impoundment (Figure 3). Confidence intervals of slopes were overlapping for all postflooding years in Lake 979, but were nonoverlapping compared with before flooding except for the last year of observations, 1995 (Table 2). Net uptake in Lake 979 after flooding and Lake 632 followed a similar year-to-year pattern (Figure 3). The average difference between net uptake by finescale dace after flooding (mean of 0.18 μg/m^2/year) compared with before flooding (mean of 0.03 μg/m^2/year) or as compared to net uptake in the reference system (0.05 μg/m^2/year) was a factor of 4–6.

Methyl Mercury in Fish

Concentrations of MeHg in finescale dace muscle from Lake 979 were similar in the early summer of 1992 and 1995, decreased over the summer in 1992 (before flooding), but increased over the summer in 1995 (after flooding) (Table 3).

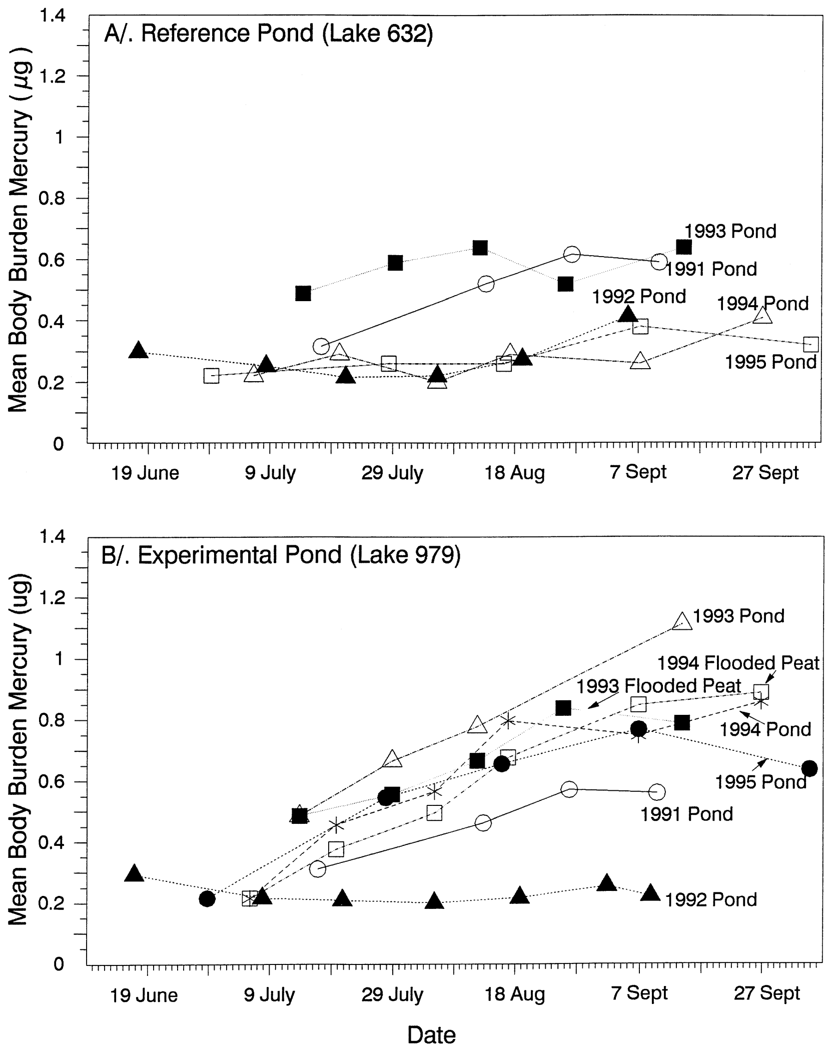

Fig. 2. Mean mercury body burdens in finescale dace in Lake 632 (reference pond) and Lake 979 (before and after flooding). See Figure 1 for sample sizes

Mean percent MeHg was always >70%, and ranged from 71–89% (Table 3). The proportion of total mercury that was MeHg increased over the summer in 1995, after flooding, but decreased over the summer of 1992, prior to flooding.

Feeding

Benthic invertebrates comprised most of the diet of finescale dace in both years and lakes (Table 4). Chironomids were the most common benthic animal consumed, but many other benthic groups were also eaten. Crustacean zooplankton, especially cladocerans, were an important, but secondary, food source. Differences in the proportion of empty stomachs may have been due to differences in field sampling techniques because fish were immediately placed on ice in 1995, but not in 1992, to slow digestion of stomach contents.

The proportion of benthic invertebrates, crustacean zooplankton, and algal material food items in the stomachs of finescale dace from Lake 979 did not change appreciably after flooding (Table 4). Benthic invertebrates comprised 78% of food items before flooding and 83% of food items after flooding. Fish from Lake 632 consumed more zooplankton and less benthic invertebrates in 1995 as compared to 1992, but feeding patterns were similar to those observed in Lake 979 (Table 4).

Discussion

The results of this study demonstrate significant increases in concentrations and net uptake by fish in a boreal reservoir after flooding. Concentrations increased 2–3× to 0.2–0.25 μg/g (wet weight), similar to increases that have been observed in other nonpiscivorous fish in larger reservoirs (Bodaly *et al.* 1984, 1987, 1997; Scruton *et al.* 1994; James Bay Mercury Committee 1995). Piscivorous fish, had they been present, would have been expected to reach much higher concentrations. Concentrations of MeHg in zooplankton increased 10× in Lake 979 after flooding (Paterson *et al.* 1997, 1998), and concentrations of MeHg in emerging and nonemerging invertebrates in the lake increased up to 3× after flooding (Malley *et al.* 1996; Hall *et al.* 1998; A. P. Wiens and D. M. Rosenberg, Freshwater Inst., unpublished data).

Fluxes (net uptake) of mercury to fish increased about 4–6× after flooding. Net fluxes before and after flooding and in the reference pond ranged from less than zero to 0.24 μg/m^2/year.

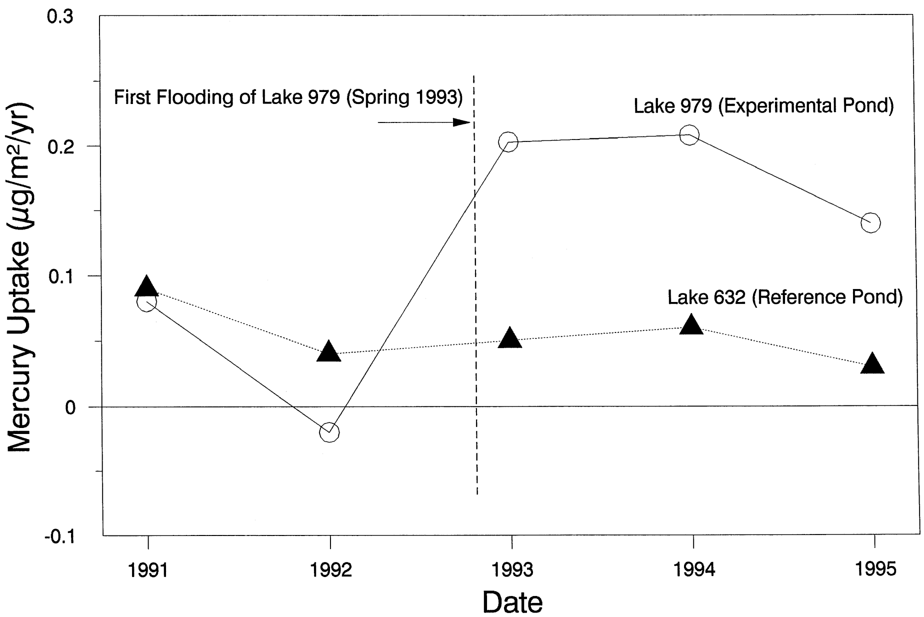

Fig. 3. Net uptake of Hg by finescale dace in Lake 632 (reference pond) and Lake 979 (before and after flooding). Net uptake per fish calculated as the difference between Hg body burdens at the end of the summer and at time of fish introduction (Figure 2). Net uptake on an areal basis calculated assuming an average density of 1.3 g m^{-2} over the whole pond or reservoir area and mean fish weight of 4 g

Table 2. Slopes (and 95% confidence intervals) of least squares linear regressions of mean Hg burden versus $\log_{10}$ day for Lakes 632 and 979, 1991–1995

Lake	Year	Slope	95% Confidence Interval
632	1991	2.84	0.66–5.01
	1992	0.48	−0.97–1.93
	1993	0.82	−1.14–2.79
	1994	0.86	−0.32–2.03
	1995	0.69	−0.14–1.53
979	1991	0.37	0.15–0.59
	1992	−0.36	−0.62–−0.10
	1993 (pond)	5.16	4.50–5.81
	1993 (peat)	2.95	1.03–4.87
	1994 (pond)	3.83	1.93–5.72
	1994 (peat)	4.44	3.50–5.39
	1995 (pond)	2.42	−0.13–4.97

Table 3. Percent methyl mercury in finescale dace sampled at the beginning and end of the caging period in 1992 and 1995, Lake 979[a]

Date Sampled	n	Live Weight (g)	Total Hg (µg/g w.w.)	Methyl Hg (µg/g w.w.)	%MeHg
17 June 92	20	3.4	0.29	0.26	89
		(2.8–4.4)	(0.09–0.39)	(0.07–0.35)	(60–124)
1 September 92	22	3.7	0.21	0.15	71
		(3.0–7.2)	(0.10–0.37)	(0.05–0.30)	(48–93)
17 June 95	19	2.8	0.38	0.27	75
		(1.9–3.8)	(0.12–1.45)	(0.11–1.24)	(33–96)
4 October 95	20	3.6	0.43	0.36	83
		(2.0–6.1)	(0.25–0.57)	(0.23–0.59)	(57–103)

[a] Means and ranges for all parameters shown

Watras *et al.* (1994) estimated fluxes of Hg to fish in Little Rock Lake, Wisconsin, as 0.34 µg/m²/year (0.6 g/year for the whole lake) which is similar to our findings, whereas Henry *et al.* (1995) estimated fluxes to fish in Onondaga Lake, New York, to be much larger at 17 µg/m²/year (0.20 kg/year for the whole lake). This latter system has received discharges from Hg-cell chloralkali facilities.

Fish accumulate most of their MeHg burdens from their food (Harris and Snodgrass 1993; Rodgers 1994; Hall *et al.* 1997). Finescale dace fed on benthic invertebrates and crustacean zooplankton before and after flooding, and feeding changed little as a result of flooding. Thus, increased concentrations and fluxes to fish cannot be accounted for by changes in the proportion of various food types eaten by the fish and are more likely due to increased concentrations in and fluxes through the invertebrate communities of the reservoir. Estimated fluxes through the lower food chain in Lake 979 increased significantly as a result of flooding and were large enough to support estimated fluxes into fish in the reservoir. Paterson *et al.* (1998) estimated fluxes of MeHg through the zooplankton community of the Lake 979 pond to be 0.02 µg/m²/year in 1992 before flooding and 1.38, 1.13, and 0.37 µg/m²/year in 1993–95 after flooding. AP Wiens and DM Rosenberg (unpublished data) estimated fluxes through the emerging insect community as 0.05 µg/m²/year in 1992 before flooding and 0.05, 0.07, and 0.21 µg/m²/year in 1993–95 after flooding. Total flux of MeHg out of the reservoir was approximately 0.2 µg/m²/year before flooding and was 7.0, 5.4, and 2.0 µg/m²/year in the 3 years after flooding (Kelly *et al.* 1997; Rudd *et al.* unpublished data). MeHg in water in Lake 979 increased on average 10× after flooding, as compared to before flooding.

Increases in MeHg fluxes to fish in Lake 979 took place without significant changes in fish growth rates, despite increases in food availability (Paterson *et al.* 1997; Wiens and Rosenberg in preparation). Rodgers (1996) found by modeling simulations that increases in weight-specific Hg concentrations in reservoir fish were due to increases in ambient MeHg, not to changes in growth rates. Our results agree with this finding.

Significant differences in net uptake of Hg on a year-to-year basis were observed. In Lake 632, net uptake by finescale dace varied 10× among years and in Lake 979, net uptake was very different in the 2 preflooding years. However, net uptake of Hg varied similarly between the reference pond and experimental

Table 4. Percent occurrence of food items in the stomachs of finescale dace from Lake 979 in 1992 and 1995, before and after flooding, and from Lake 632 (reference) in the same years

	Lake 979		Lake 632	
	1992	1995	1992	1995
Number of stomachs examined	21	17	18	18
Insecta				
Unidentified parts	7	10	—	13
Chironomidae	21	27	23	30
Chaoborus	—	8	—	—
Ephemeroptera	5	—	4	5
Odonata	6	16	4	5
Coleoptera	5	—	8	3
Trichoptera	12	2	4	7
Hemiptera	17	—	16	—
Crustacea				
Unidentified parts/eggs	2	—	—	—
Copepoda	5	2	—	11
Cladocera	12	22	16	18
Ostracoda	—	—	—	2
Arachnida—Hydracarina	5	2	19	—
Pelecypoda—Sphaeriidae	—	—	4	1
Algae/organic detritus	2	11	—	5
% Empty stomachs	5	—	27	—

reservoir before flooding and also followed a similar year-to-year pattern after flooding, despite the fact that uptake was much elevated in the experimental reservoir. This would suggest that year-to-year differences in factors such as temperature and water chemistry are important, however attempts to correlate net uptake of Hg with weather averages were not successful.

Hg fluxes to fish increased significantly in the first year after flooding and thus showed a very rapid response to flooding, in agreement with observations on MeHg in water, suspended particles, and zooplankton in Lake 979 and other boreal reservoirs (Bodaly *et al.* 1997; Kelly *et al.* 1997; Plourde *et al.* 1997; Paterson *et al.* 1998). In other boreal reservoirs, response times for increased Hg in fish have not been as rapid, even for lower trophic levels such as lake whitefish (Bodaly *et al.* 1984, 1997; James Bay Mercury Committee 1995). However, finescale dace studied in Lake 979 were smaller and younger than most fish sampled in reservoirs and would be expected to respond faster. Other reservoirs probably also respond very quickly with increased MeHg production, but this production takes some years to be measurable in larger fish.

Is there evidence for a slowing in Hg bioaccumulation by fish over the 3-year period after flooding in Lake 979? Net uptake of Hg was lower in the third year of flooding as compared to the first 2 years, and the difference in net uptake of Hg between fish in the flooded reservoir and fish in the reference pond decreased (Figure 3). This trend is consistent with at least some other observations concerning MeHg production and bioaccumulation in the reservoir. Fluxes through the zooplankton community decreased progressively over the first 3 years of flooding (Paterson *et al.* 1998). However, fluxes through the emerging insect community increased over the first 3 years after flooding, although fluxes from the flooded peat part of the reservoir decreased between 1994 and 1995 (Wiens and Rosenberg unpublished data). Total net flux of MeHg from the reservoir decreased over the first 5 years after flooding (Rudd *et al.* unpublished data). Further monitoring will be required to conclusively answer the question of trends in Hg fluxes to fish. Bioenergetics modeling of Hg in fish in new reservoirs has shown that increases in the supply of MeHg to the food chain for as little as 3 years are sufficient to produce decadal-long elevations of Hg concentrations in predatory fish due to long depuration times and long life spans (R. C. Harris, personal communication).

Flooding of a boreal wetland may represent a worst case for reservoir construction with regard to MeHg bioaccumulation (Kelly *et al.* 1997). Natural wetlands are important sources of MeHg and uplands are sinks for MeHg (St. Louis *et al.* 1994, 1996). Results of this study can be used directly in models predicting the cycling of mercury in reservoirs and Hg concentrations in fish (*e.g.*, Johnston *et al.* 1991; Harris and Snodgrass 1993; Harris and Bodaly 1998). Flooding of the Lake 979 wetland increased MeHg production substantially. In contrast, uplands have lesser amounts of stored carbon and flooding may not increase methylation to the same degree. We have started a new study at the Experimental Lakes Area that will flood forested uplands, the Upland Flooding Project, to answer this question.

Acknowledgments. We thank Neil Strange for laboratory mercury determinations, and Shannon Harris, Sheena Majewski, John Embury, John Rudd, Vincent St. Louis, Allen Wiens, Britt Hall, Karen Scott, and Bill Mavros for field assistance. Drs. Carol Kelly, Tom Johnston, Michael Paterson, and John Rudd provided many valuable ideas.

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