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ORIGINAL ARTICLE

Mercury Contamination of Biota from Acadia National Park, Maine: A Review

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Abstract We reviewed literature reporting both total and methylmercury from biota from Acadia National Park, Maine, USA. Our review of existing data indicates that 1) mercury contamination is widespread throughout the Park's various aquatic ecosystems; 2) mercury pollution likely represents a moderate to high risk to biota inhabiting the Park; and 3) biota at all trophic levels possess elevated concentrations of both total and methylmercury. Watershed fire history and the resulting post-fire forest succession patterns are an important landscape attribute governing mercury cycling at Acadia National Park. Therefore, park service personnel should consider these factors when planning and implementing Hg biomonitoring efforts. Additional baseline funding from the National Park Service for Hg research and biomonitoring will likely be required in order to further evaluate the spatial and temporal patterns of mercury contamination in the park's biota.

Keywords Acadia National Park · biota · contaminants · literature review · mercury

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1 Introduction

Mercury (Hg) contamination of biota from freshwater ecosystems is a chronic and widespread environmental problem (Lindqvist et al., 1991; Watras & Huckabee, 1994; Richardson, Mitchell, Coad, & Raphael, 1995; Weiner, Krabbenhoft, Heinz, & Scheuhammer, 2003). Remote aquatic ecosystems, such as those in Acadia National Park (ANP), Maine (Figure 1), may be highly contaminated as a result of long-range transport and atmospheric deposition (Downs, Macleod, & Lester, 1998; Fitzgerald, Engstrom, Mason, & Nater, 1998; Morel, Kraepiel, & Amyot, 1998).

Anthropogenic Hg in the environment is commonly derived from a variety of sources. The primary sources include municipal incinerators and coal-burning industries (NESCAUM, 1998), chlor-alkali plants (Parks & Hamilton, 1987; Lodenius, 1998) and mining operations (Bonzongo, Heim, Warwick, & Lyons, 1996; Wayne, Warwick, Lechler, Gill, & Lyons, 1996; Lacerda, 1997). Natural sources of Hg inputs to the

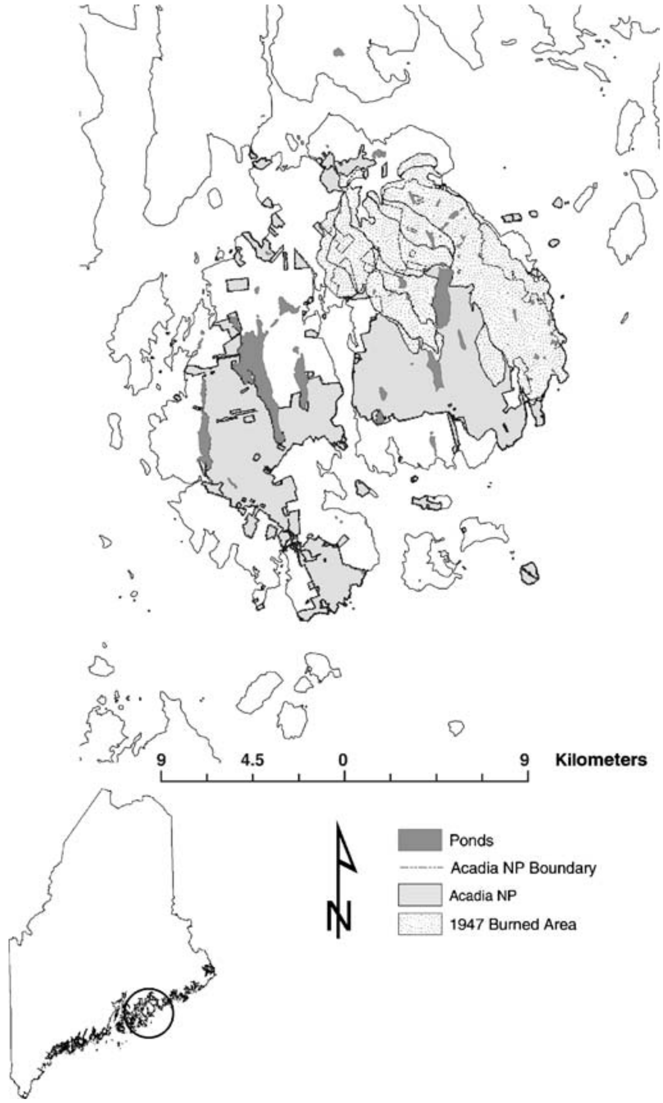

Figure 1 Map of Mount Desert Island, Maine, USA, with Acadia National Park boundary and 1947 Bar Harbor fire perimeter outlined.

atmosphere include forest fires, dust particles, volcanic eruptions, and degassing from water surfaces (Rasmussen, 1994). Lake sediment records (Swain, Engstrom, Brigham, Henning, & Brezonik, 1992) have shown that anthropogenic sources of mercury have tripled during the last 150 years indicating that approximately two-thirds of all atmospheric Hg is from human origin, and the remaining amount is from natural sources (Mason, Fitzgerald, & Morel, 1994; Morel et al., 1998).

When mercury enters aquatic ecosystems via atmospheric deposition, it is converted by micro-organisms, such as sulfate reducing bacteria, and chemical processes to methylmercury (MeHg), a biologically active and toxic form. Acidified freshwater ecosystems in the northeastern United States with high temperatures and dissolved organic carbon levels facilitate MeHg bioaccumulation and biomagnification (Gilmour & Henry, 1991; Weiner et al., 2003 and references therein). High concentrations of Hg in aquatic biota have the potential for sub-lethal effects (i.e., reduced foraging and inability to avoid predators; Webber & Haines, 2003) that may negatively influence population dynamics (Evers, Lane, & Savoy, 2003a). The effects of MeHg on biota can be detrimental and include reduced reproduction and development, lower productivity and growth, aberrant behavior, and death (Thompson, 1996; Weiner et al., 2003). Moreover, environmental contaminants, including Hg, may be a significant limiting factor that contributes to widespread population declines of some groups of aquatic biota, such as amphibians (Stebbins & Cohen, 1995). Aquatic biota such as fish, that occupy high trophic positions, and piscivorous wildlife commonly contain elevated amounts of Hg, even in remote, undeveloped areas with no local sources of pollution (Lucotte, Schetagne, Thérien, Langlois, & Tremblay, 1999; Evers, Taylor, Major, Taylor, Poppenga, & Scheuhammer, 2003b).

The mission of the National Park Service is "to promote and regulate the use of the...national parks... whose purpose is to conserve the scenery and the natural and historic objects and the wild life therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations" (National Park Service Organic Act, 16 U.S.C.1.). A goal of Park resource management is to protect the integrity of the Parks' natural resources. This mission and goal cannot be realized without recognition of the current condition of the Parks' abiotic and biotic resources and an understanding of factors contributing to those conditions. It is likely that the status of natural resources at many of the Nation's Parks, including Acadia, is the result of perturbations originating beyond the Parks' borders. Hg pollution management presents a daunting challenge to natural resource specialists because of its long-range transport capabilities.

Investigators often seek to document whether a persistent bioaccumulative contaminant, such as Hg, which is known to occur in elevated levels in environments across the United States due to anthropogenic sources, may present a much more insidious problem for ecosystem health than is recognized. Protection of surface water quality in the United States has enormous ramifications for economic and public health of the Nation's residents and is fundamental to maintaining the biotic integrity of the Nation's public lands. Monitoring water quality and determining Hg concentration levels in aquatic ecosystems also has important implications for protection of the Nation's aquatic biota. Recognizing the pervasiveness of Hg and other global contaminant problems and accelerating development of methods to decrease continued contaminant accumulation are critical to environmental and public health. Because of the global scale of the mercury problem, a reversal in current trends must be considered a priority in order to avoid functional collapse of aquatic ecosystems.

Recent paired and gauged watershed studies (Kahl, Fernandez, Manski, Haines, & Lent, 2002; Nelson, Johnson, Kahl, Haines & Fernandez, in press; Johnson et al., in press) in ANP have identified a number of factors influencing total Hg bioavailability (Figure 2). Hg and MeHg contamination in several species at different trophic levels have been documented across the Park. Here we present a general review of the research documenting Hg contamination of biota from ANP. Specific objectives were to summarize Hg contamination data from biota inhabiting the Mount Desert Island (MDI) region of ANP and provide recommendations for future ANP biomonitoring and research. We reviewed all biota related research examining Hg in ANP except for tree swallows (*Tachycineta bicolor*), which is reviewed by Longcore, Haines, and Halteman (in press). All quality assurance/quality control and methodology for the summarized studies may be found in the original literature sources. Data are presented as mean ± standard error unless otherwise noted. Hg data are presented as dry weight (dw), wet weight (ww), and fresh weight (fw). Notation of total mercury (Hg) and methylmercury (MeHg) are used throughout. Unless otherwise noted, Hg values represent total Hg.

2 Review & Discussion

2.1 Plankton

Burgess (1997) showed that Hg and MeHg concentrations in plankton (phytoplankton and zooplankton) from ANP (Table I) were generally higher than plankton from meso-oligotrophic lakes in Wisconsin,

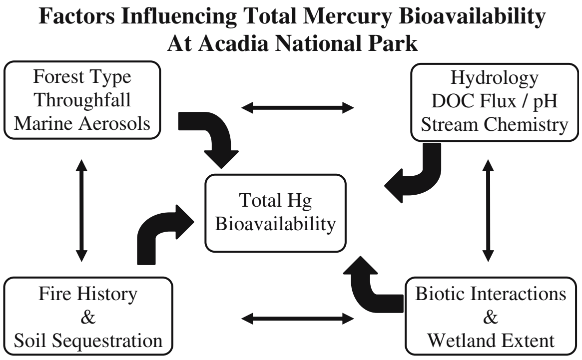

Figure 2 Hypothetical model of factors influencing total Hg bioavailability at Acadia National Park, Maine. Note all factors are not mutually exclusive.

where Watras and Bloom (1992) reported Hg and MeHg concentrations ranging from 300 to 400 ng/g dw and 40–150 ng/g dw, respectively, for phytoplankton and 280–375 and 100–270 ng/g dw, respectively, for zooplankton. Additionally, Watras et al. (1998) report mean MeHg concentrations in microseston and zooplankton for 15 Wisconsin lakes of 33 and 53 ng/g dw, respectively. In contrast, MeHg concentrations in plankton (phytoplankton and zooplankton combined) from ANP were generally higher, ranging from 46 to 206 ng/g dw. Hg concentrations in the ANP samples ranged from 234 to 761 ng/g dw. Watras et al. (1998) reported zooplankton Hg concentrations ranged from 33 to 206 ng/g dw for 15 Wisconsin lakes. Although not directly comparable (i.e., phytoplankton and zooplankton combined *versus* zooplankton alone), these data suggest that mercury concentrations are higher at the base of the food web in the ANP study lakes in contrast to the Wisconsin study lakes. The proportion of MeHg to Hg in the Acadia samples (phytoplankton and zooplankton combined) ranged 12–21%, comparable to the 16–20% for zooplankton reported by Mason and Sullivan (1997), but generally lower than the 7–34% (phytoplankton) and 11–83% (zooplankton) reported by Watras et al. (1998).

2.2 Invertebrates

Hg concentrations in dragonfly larvae from Hodgdon Pond and Seal Cove Pond (Table II) ranged 208–560 ng/g dw and are similar to concentrations found in other studies (Tremblay, Lucotte, & Rheault, 1996). Elsewhere, Hg concentrations in Odonate nymphs have ranged from 135 to 358 ng/g dw in two Swedish mesotrophic lakes (Parkman & Meili, 1993) to 136 ng/g dw in northern Québec, Canada, lakes and 283–615 ng/g dw in hydroelectric reservoirs in Northern Québec, Canada (Tremblay et al., 1996). Tremblay et al. (1996) found that Odonate nymph MeHg concentrations ranged from 283 to 615 ng/g dw and accounted for 75–100% of the total Hg. In ANP mean MeHg concentrations in Odonate nymphs ranged from 130 to 319 ng/g dw, and the proportion of MeHg to Hg ranged from 53 to 83% (Table II, Burgess, 1997).

Mean Hg concentrations in pooled Isopod samples from ANP (Table II) are comparable to those documented by Parkman and Meili (1993) in eight Swedish lakes (63–394 ng/g dw). Hodgdon Pond, ANP, isopods ranged from 306 to 448 ng/g dw, similar to the range reported for acidic dystrophic lakes by Parkman and Meili (1993). Hodgdon Pond is a circumneutral mesotrophic pond. The decrease in Hg concentrations, which occurred from spring to late summer (Table II), was also observed by Parkman and Meili (1993). This seasonal decrease in Hg concentration in isopods may be directly related to the annual appearance of a new generation in summer (Parkman & Meili, 1993).

2.3 Amphibians

Bank, Loftin, and Jung (2005) measured an average total Hg concentration of 66.1 ± 3.4 ng/g ww (n = 116) for 1–3 year old larval northern two-lined salamanders (*Eurycea bislineata bislineata*) from 14, first and second order streams in ANP. Green frog (*Rana clamitans*) tadpoles (~1 year old) had an average Hg concentration of 25.3 ± 1.5 ng/g ww (n = 61), and bullfrog (*Rana catesbeiana*) tadpoles (1–2 years old) had an average Hg concentration of 19.1 ± 0.74 ng/g

Table I Seasonal plankton Hg concentrations (cell values are ng/g dw); % MeHg = ratio of MeHg to Hg. Spring samples were collected during May–June 1996; summer samples were collected during August–September 1996

			Total Hg			MeHg			Percent MeHg	
Site	Season	*N*	Mean	SD	*N*	Mean	SD	*N*	Mean	SD
Hodgdon Pond	Spring	3	531.5	141.8	5	115.8	38.8	3	21.1	12.3
	Summer	–	–	–	1	205.6	–	–	–	–
	Combined	3	531.5	141.8	6	130.7	50.5	–	–	–
Seal Cove Pond	Spring	3	431.1	171.1	4	75.6	23.6	2	16.8	4.4
	Summer	1	760.5	–	2	69.71	33.8	1	12.3	–
	Combined	4	513.5	216.0	6	73.6	23.9	3	15.3	4.0

Data are from Burgess (1997).

Table II Hg concentrations in homogenized aggregates of invertebrate samples categorized among seasons

Site	Season	Order/suborder	Number (aggregate N)	Total Hg Mean	Total Hg SD	MeHg Mean	MeHg SD	Percent MeHg Mean	Percent MeHg SD
Hodgdon Pond	Spring	Anisoptera	49 (5)	560[a]	200	319[a]	193	53	31
		Zygoptera	30 (2)	491	72	290	–	54	–
		Isopoda	44 (1)	448	–	278	–	62	–
		Amphipoda	>100 (1)	212	–	102	–	48	–
	Summer	Anisoptera	200 (13)	377[c]	119	264[a]	75	83	38
		Isopoda	>50 (1)	306	–	214	–	70	–
		Amphipoda	>100 (1)	206	–	118	–	57	–
	Combined	Anisoptera	249 (18)	448	174	278	110	71	37
		Isopoda	>100 (2)	377	100	246	45	66	6
		Amphipoda	>200 (2)	209[a]	4	110[a]	11	53	6
Seal Cove Pond	Spring	Anisoptera	31 (3)	208[b]	40	130[a]	46	65	7
		Zygoptera	51 (2)	505	226	259	–	39	–
		Amphipoda	>100 (1)	323	–	148	–	46	–
	Summer	Anisoptera	121 (14)	366[c]	127	220[a]	134	55	26
		Amphipoda	>100 (1)	457	–	103	–	22	–
	Combined	Anisoptera	152 (17)	318	130	209	129	57	23
		Amphipoda	>200 (2)	390[a]	95	126[a]	32	34	17

[a,b,c] Means with the same letter within the same column are not significantly different ($P > 0.05$).

Spring samples were collected during May–June 1996; summer samples were collected during August–September 1996. Data are from Burgess (1997).

ww (n = 31, Bank, Loftin, & Amirbahman, 2004). Larval two-lined salamanders likely had higher mercury concentrations due to their invertebrate diet (Petranka, 1984) in contrast to both green frog and bullfrog tadpoles which are grazers. No reptile species have been sampled for Hg exposure on MDI, and currently only three species of amphibians from the park have been analyzed for Hg contamination.

2.4 Fish

Fish tissue results (Tables III, IV, and V) from MDI suggest that salmonid fish species (landlocked salmon, *Salmo salar*; brook trout, *Salvelinus fontinalis*; lake trout, *Salvelinus namaycush*) generally have lower Hg concentrations than non-salmonid species (chain pickerel, *Esox niger*, alewife, *Alosa pseudoharengus*; yellow perch, *Perca flavescens*; white perch, *Morone americana*; and smallmouth bass, *Micropterus dolomieu*). The fish tissue results also suggest that Hg concentrations in most fish species, with the exception of smallmouth bass, are generally lower on MDI than in other regions of Maine (Stafford & Haines, 1997; Burgess, 1997). Hg concentrations in lake trout from MDI ranged 0.21–0.32 μg/g ww, which is considerably lower than the statewide mean of 0.45 μg/g ww (Stafford & Haines, 1997). Similarly, Hg concentrations in landlocked salmon from MDI ranged 0.11–0.37 μg/g ww, whereas the statewide mean for this species is 0.35 μg/g ww (Stafford & Haines, 1997).

Non-salmonid species on MDI generally have similar or lower Hg concentrations than the state average for these species. Hg concentrations in chain pickerel from MDI had much lower Hg concentrations than the statewide mean (Stafford & Haines, 1997). Differences in sizes or ages of collected fish may explain the difference in Hg concentrations. MDI chain pickerel were generally smaller individuals than those collected in the statewide effort (Stafford & Haines, 1997). Smallmouth bass collected on MDI had higher concentrations of Hg than the state average ranging from 0.41 to 3.86 μg/g ww in contrast to the range of 0.31–1.12 μg/g ww reported by Stafford and Haines (1997). Total Hg concentrations in yellow perch collected on MDI ranged 0.07–0.25 μg/g ww (Table V). Watras et al. (1998) reported that total Hg concentrations in Wisconsin yellow perch ranged from 0.03 to 0.24 μg/g ww, which are similar to Hg levels reported for MDI yellow perch.

Table III Hg concentrations in fish collected on Mount Desert Island, Maine

Lake	Species	N	Mean length (mm)	Mean weight (g)	Fillet (μg/g, ww)		Whole-body (μg/g, ww)		State average (fillet - μg/g, ww)
					Mean	Range	Mean	Range	
Eagle Lake	lake trout	5	462 (417–475)	797 (704–918)	0.25	0.21–0.32	0.20	0.17–0.25	0.45
Echo Lake	landlocked salmon	4	405 (350–412)	542 (322–730)	0.15	0.11–0.17	0.14	0.12–0.15	0.35
Hamilton Pond	chain pickerel	6	387 (304–432)	332 (152–440)	0.59	0.39–0.79	0.43	0.29–0.58	0.88
Hodgdon Pond	smallmouth bass	7	433 (329–513)	1138 (419–1686)	1.72	0.77–3.86	N/A	N/A	0.66
Jordon Pond	landlocked salmon	5	519 (460–569)	1311 (838–2042)	0.17	0.14–0.37	0.14	0.10–0.18	0.35
Long Pond (MDI)	smallmouth bass	5	393 (340–417)	834 (510–1152)	0.48	0.41–0.54	0.27	0.24–0.31	0.66
Long Pond (Isle au Haut)	brook trout	6	258 (233–336)	185 (120–408)	0.21	0.09–0.49	0.17	0.09–0.40	0.31
Lower Hadlock Pond	white perch	5	267 (229–299)	242 (154–338)	0.69	0.36–1.03	0.31	0.29–0.39	0.65
Round Pond	chain pickerel	5	378 (323–458)	336 (190–672)	0.37	0.27–0.56	0.29	0.28–0.35	0.88
Seal Cove Pond	smallmouth bass	5	394 (350–417)	817 (640–1042)	0.85	0.51–1.16	0.51	0.28–0.84	0.66
Somes Pond	chain pickerel	5	369 (297–429)	276 (138–444)	0.37	0.27–0.45	0.24	0.19–0.31	0.88

Fish were collected May–June 1995. Mean Hg values are least square mean concentrations. Data are from Burgess (1997). State of Maine average mercury concentrations are from Stafford and Haines (1997).

2.5 Birds

Aquatic MeHg bioavailability in ANP is indicated by concentration data from three species of birds: Bald eagle (*Haliaeetus leucocephalus*, Welch, 1994, Table VI), common loon (*Gavia immer*, Evers et al., 2003a, b, Tables VII and VIII), and tree swallow (*Tachycineta bicolor*, Longcore et al., in press). The U.S. Environmental Protection Agency has identified bald eagles and common loons as species with high risk for Hg contamination (USEPA, 1997) and has recently selected those species as primary indicators for monitoring environmental Hg in North America (M. Wolfe, personal communication).

Juvenile eagle blood and feather samples (n = 7) collected during 1991–1992 on MDI (Welch, 1994) contained a mean blood Hg level of 0.10 ± 0.40 μg/g, ww (range = 0.03 to 0.15) and a mean feather Hg level of 5.2 ± 1.6 μg/g, fw (range = 2.8 to 6.5). Juvenile eagle blood and feather Hg levels represent upper trophic level Hg biomagnification from coastal marine habitats surrounding ANP and MDI. Compared to Hg levels in juvenile eagles elsewhere, MDI marine habitats do not pose a risk to eagles (Welch, 1994; Jagoe, Bryan, Brant, Murphy, & Brisbin, 2002). Eagle populations with similar low risk to Hg exposure are known in South Carolina (mean = 3.1 μg/g fw in feathers and 0.10 μg/g ww in blood; Jagoe et al.,

Table IV Hg (μg/g ww) least square mean concentrations in fillets of predatory fishes collected on MDI Maine during May–June 1996

Site	Species	N	Length (mm)		Weight (g)		Age (year)		Total Hg	
			Mean	Range	Mean	Range	Mean	Range	Mean	Range
Hodgdon Pond	American eel	4	607	476–680	505	220–662	unk	unk	1.63	0.94–1.70
	Chain pickerel	8	347	278–440	270	120–552	3.8	2–5	0.48	0.16–0.70
	Smallmouth bass	7	433	329–513	1138	419–1686	8.7	5–12	1.72	0.77–3.86
Seal Cove Pond	American eel	4	678	632–756	723	571–1036	unk	unk	0.77	0.65–1.41
	Chain pickerel	6	418	333–465	527	203–699	4.3	3–6	0.36	0.25–0.86
	Smallmouth bass	8	380	273–457	774	252–1326	7.4	5–9	0.91	0.32–1.30

Data are from Burgess (1997).

Table V Hg concentrations in forage fishes (concentrations are in ww)

Site	Species	N	Length (mm)		Weight (g)		Age (year)		Total Hg (μg/g)		MeHg (μg/g)	
			Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Hodgdon Pond	Yellow perch (young of year)	10	53.1	46–56	1.7	1.2–2.4	<1	<1	0.058	0.031–0.085	0.057	0.040–0.083
	yellow perch (spring)	10	138	113–174	29	13–57	3.6	3–4	0.18[a]	0.14–0.23	0.16[a]	0.11–0.21
	yellow perch (summer)	10	138	105–180	28	12–54	3.1	2–5	0.13[b]	0.07–0.19	0.13[a,b]	0.10–0.19
Seal Cove Pond	Alewife (young of year)	10	75.3	68–80	3.4	2.4–4.2	<1	<1	0.055	0.033–0.073	0.059	0.052–0.067
	yellow perch (spring)	10	139	102–160	29	10–45	3.7	2–5	0.12[b]	0.078–0.21	0.11[b]	0.061–0.18
	yellow perch (summer)	10	128	111–159	22	13–38	3.2	2–4	0.13[b]	0.080–0.25	0.16[a]	0.10–0.24

[a,b] Means with the same letter within the same column are not significantly different ($P > 0.05$).

Methylmercury concentrations are least square means, and data are whole body concentrations. Spring samples were collected during May–June 1996; summer samples were collected during August–September 1996. Data are from Burgess (1997).

2002), central Florida (mean = 3.2 μg/g fw in feathers and 0.16 μg/g ww in blood; Wood et al., 1996), and Lake Erie (mean = 3.7 μg/g fw in feathers; Bowerman, Evans, Geisy, & Postupalsky, 1994). Eagle populations at higher risk have been documented from the upper Great Lakes (8.0–20.0 μg/g fw in feathers) and from interior Maine (0.07–1.46 μg/g ww in blood and 8.0–36.7 μg/g dw in feathers; Welch, 1994).

Common loon samples collected from MDI have documented near-term (blood and egg matrices) as well as long-term (adult feathers) MeHg uptake. Hg exposure is confounded by age and sex (Evers et al., 1998). Mean feather Hg level was 7.5 ± 1.6 μg/g, fw, and mean adult blood Hg concentration was 1.03 ± 0.45 μg/g, ww (n = 5 lakes). Mean egg Hg concentration was 0.57 ± 0.09 μg/g, (n = 1 lake). Egg and adult female blood Hg levels were highly correlated ($r^2 = 0.79$, $P < 0.001$, Evers et al., 2003b). Because loon blood and prey fish Hg levels are highly correlated on breeding lakes (Evers et al., 2003a), egg Hg levels primarily represent dietary Hg uptake on the breeding lake. Compared to Hg exposure levels elsewhere in Maine (Evers et al., 2003a) and across North America (Burgess, Evers, Kaplan, Duggan, & Kerekes, 1998; Evers et al., 1998, 2003b; Meyer, Evers, Hartigan, & Rasmussen, 1998), near-term and long-term Hg exposure on the larger lakes of MDI likely represents a low risk to breeding loons. Although loon reproductive success is below sustainable levels on MDI (Evers et al., 2003a), Hg levels are not considered a contributing factor.

Table VI Hg concentrations for blood (μg/g, ww) and feathers (μg/g, fw) from 6–8 week old bald eagles from MDI and vicinity, 1991–1992 (Welch, 1994)

Location	Year	Feather Hg	Blood Hg
Placentia Island, Frenchboro, ME	1991	2.8	0.106
Placentia Island, Frenchboro, ME	1991	3.2	0.101
Placentia Island, Frenchboro, ME	1992	6.51	0.141
Somes Sound, Mt Desert, ME*	1991	6.2	0.062
South Twinnie Island, ME	1992	6.4	0.027
Sheep Porcupine Island, Bar Harbor, ME	1992	5.92	0.086
Schoodic Island, Winter Harbor, ME	1991	5.4	0.148

*Egg sample collected in 1991, MeHg = 0.410 μg/g (adjusted ww).

2.6 Mammals

Mink (*Mustela vison*) and river otter (*Lutra canadensis*) feed on aquatic prey and thus are possible indicators of aquatic MeHg bioavailability in ANP. Sampling these species, however, has been limited to a few carcasses provided by trappers. Although sample sizes of both species from ANP are small (Table VIII), data indicate a potential low risk of Hg to these piscivorous mammals compared to levels documented in other studies (Mierle, Addison, MacDonald, & Joachim, 2000; Evans, Addison, Villeneuve, Macdonald, & Joachim, 1998, 2000). Blood Hg levels reflect short-term exposure, are independent of age, and are primarily in the MeHg form, in contrast to levels reported from liver and kidney tissues. Live-trapping of these species to collect blood Hg may provide a more accurate indication of Hg exposure and potential effects originating from MDI.

Table VII MeHg concentrations (μg/g) in common loon blood (ww), feathers (fw), and eggs (ww) from Acadia National Park, Maine, USA and vicinity, 1997–2002 (Evers et al., 1998, 2003b; BioDiversity Research Institute unpublished data)

Lake/territory	Age	Sex	Year	Blood Hg	Feather Hg	Egg Hg
Eagle Lake	Adult	M	2002	0.89	–	–
Eagle Lake	Adult	F	2002	0.80	–	–
Echo Lake	Juvenile	U	1997	0.01	–	–
Echo Lake	Adult	M	1997	1.20	5.51	–
Echo Lake	Adult	F	1997	0.42	6.97	–
Long Pond	Adult	F	2001	0.76	8.45	–
Seal Cove	Adult	F	1997	1.71	9.07	–
Somes Pond	Adult	M	2002	1.46	–	
Echo Lake	–	–	2000	–	–	0.66
Echo Lake	–	–	2000	–	–	0.58
Echo Lake	–	–	2002	–	–	0.48

3 Considerations for Future Research and Monitoring

Although there have been many recent advances in evaluating Hg exposure and its toxicological effects on aquatic biota (Pickhardt, Folt, Chen, Klaue, & Blum, 2002; Webber & Haines, 2003), many questions remain unanswered. Weiner et al. (2003) concluded that there has been little progress in addressing toxicological effects of Hg contamination on biota, compared to scientific advances in identifying environmental and biogeochemical factors influencing Hg bioavailability and biotic exposure to MeHg. There are three primary areas of future Hg related research that warrant attention (Weiner et al., 2003), including 1) examination of reproductive effects of MeHg on aquatic biota including fish, birds, and mammals; 2) evaluation of variation in the susceptibility of certain taxa to adverse effects of MeHg exposure; and, 3) evaluation of the combined effects of MeHg with other anthropogenic stressors (e.g., acidification, other heavy metals, climate change, habitat degradation, and UV-B radiation).

Current efforts to monitor Hg and MeHg bioavailability in ANP should be maintained using palustrine amphibian species such as bullfrogs or green frogs for wetlands, two-lined salamanders for small order streams, common loons for lake systems, and Bald Eagles for marine habitats. Tree Swallows are useful indicators of MeHg availability to insectivores (Longcore et al., in press), and mink are good representatives for Hg exposure to mammals on MDI because of their high trophic position and since sublethal sampling would be possible with this species. Raccoons (*Procyon lotor*) may also be useful (see Gaines et al., 2002) for similar reasons. Hg exposure should be determined in other potentially affected species, including long-lived reptiles such as snapping turtles (*Chelydra serpentine*, Golet & Haines, 2001), other aquatic birds such as the belted kingfisher (*Ceryle alcyon*) and common merganser (*Mergus merganser*), and most biota found in wetlands. We also recommend that future Hg biomonitoring efforts include estuarine biota since little is known about mercury bioavailability in MDI coastal ecosystems. Mummichogs (*Fundulus heteroclitus*) are abundant and widely distributed in MDI estuaries and thus may be a good indicator species for monitoring Hg in these environments. Moreover, since fire history plays an important role in influencing total Hg flux in ANP

Table VIII Hg concentrations (μg/g) in river otter and mink fur (fw), brain (ww) and liver tissue (ww)

Date	Species	Sex	Location	Fur Hg	Brain Hg	Liver Hg
11/15/01	Otter	M	Long Pond	9.90	0.33	0.48
11/15/01	Otter	F	Round Pond	1.14	0.38	0.58
11/5/01	Mink	M	Round pond	4.30	0.26	0.86

All samples from MDI, November 2001 (BioDiversity Research Institute, unpublished data).

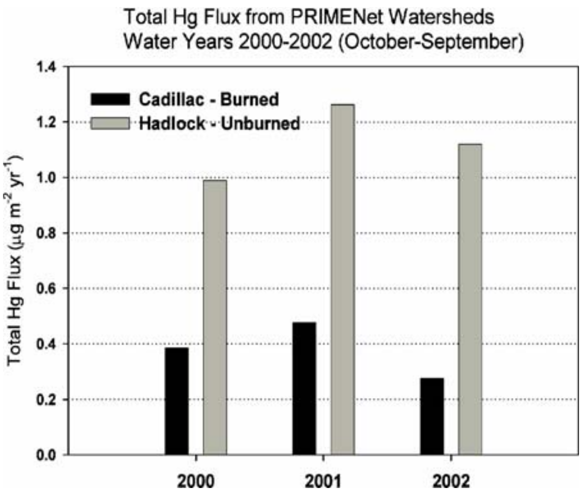

Figure 3 Total mercury (Hg) streamwater flux from paired-gauged watersheds at Acadia National Park, Maine, for water years 2000–2002. A water year is October 1 of the first year to September 30 of the following year. Flux is determined from event samples, regular (at least biweekly) samples, and hourly discharge values provided by the US Geological Survey. Data are from Kahl et al., in press.

watersheds (Nelson et al., in press; Figure 3), Hg biomonitoring programs should account for the gradient of fire disturbance in the park.

Although Hg concentrations for pond invertebrates have been reported (Table I and II), invertebrates collected from streams and bogs in ANP have not been analyzed for Hg. These taxa, which are food sources for many of the park's aquatic and terrestrial fauna, should be included in future ANP food web studies of Hg contamination and biomagnification. The relative importance and variability of trophic positions and energy sources and their fluctuations with watershed Hg concentrations, fire history, and water chemistry should also be examined.

Studies of Hg in ANP biota indicate that Hg contamination in the Park is widespread, current Hg pollution likely represents a moderate to high risk to aquatic biota in the Park, and all trophic levels show elevated levels of Hg and MeHg. Fish and amphibians are likely at a higher risk of negative effects from MeHg contamination than other species such as birds, mammals, and reptiles who store MeHg in body parts away from vital organs such as feathers, hair, or carapaces, respectively. Detoxification mechanisms of MeHg should be studied further, especially for amphibian species whose depuration potential is largely unknown (Sparling, Linder, & Bishop, 2000).

Wetlands occur in 20% of the ANP land area (Calhoun et al., 1994); future research should examine whether methylation rates differ with wetland type (St. Louis et al., 1996) as well as how the adjacent marine environment influences Hg bioavailability to ANP's aquatic environments. Evaluating how ANP beaver (*Castor canadensis*) influence dissolved organic carbon budgets and local and watershed-scale wetland vegetation dynamics may reveal additional influences on Hg cycling in the park. Moreover, Hg research and biomonitoring efforts at ANP should also evaluate Hg bioavailability dynamics among the different surface water types in the park including vernal pools, semi-permanent and permanent wetlands, ponds, lakes and stream ecosystems. It is likely that there is significant variation in methylation capacity and MeHg biomagnification rates among the different surface water types. Therefore, future research should develop and test habitat specific, predictive models across a trophic and taxonomic gradient based on the abiotic and biotic characteristics of the different aquatic ecosystems.

Hg cycling is potentially complex. Investigations that consider multiple spatial and temporal scales and recognize complexities inherent in watersheds will likely be more informative. Geographic information systems may help resource managers identify spatial trends in Hg bioavailability and relationships with landscape and watershed attributes to develop watershed-scale risk assessments of aquatic habitats and their inhabitants may be developed. Additionally, studies that use isotopic Hg tracers (Hintelmann et al., 2002) to examine the fate and transport of recently and historically deposited Hg will provide information suggesting the effectiveness of current Hg pollution abatement programs. Additional baseline funding from the National Park Service for Hg research and biomonitoring will likely be required to meet these objectives.

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