New York State Energy Research and Development Authority

# Long-Term Monitoring and Assessment of Mercury Based On Integrated Sampling Efforts Using the Common Loon, Prey Fish, Water, and Sediment

Final Report

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# Long-Term Monitoring and Assessment of Mercury Based On Integrated Sampling Efforts Using the Common Loon, Prey Fish, Water, and Sediment

Final Report

Prepared for the
**New York State Energy Research and Development Authority**


Albany, NY
nyserda.ny.gov

Gregory Lampman
Project Manager
Environmental Monitoring, Evaluation and Protection

Mark Watson
Program Manager
Environmental Research

Prepared by:
**Wildlife Conservation Society's Adirondack Program**
Nina Schoch[1] and Michale Glennon

**Biodiversity Research Institute**
David Evers, Melissa Duron, and Allyson Jackson

**Syracuse University, Department of Civil and Environmental Engineering**
Charles Driscoll and Xue Yu

**Bureau of Habitat, NYS Dept. of Environmental Conservation**
Howard Simonin

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[1] Current affiliation: Biodiversity Research Institute, Gorham, ME

## NOTICE

This report was prepared by Nina Schoch, David Evers, Melissa Duron, Allyson Jackson, Charles Driscoll, Xue Yu, Howard Simonin, and Michale Glennon in the course of performing work contracted for and sponsored by the New York State Energy Research and Development Authority and the partners of the Wildlife Conservation Society's Adirondack Loon Conservation Program (hereafter the "Sponsors"). The opinions expressed in this report do not necessarily reflect those of the Sponsors or the State of New York, and reference to any recommendation or endorsement of it. Further, the Sponsors and the State of New York make no warranties or representations, expressed or implied, as to the fitness for particular purpose or merchantability of any product, apparatus, or service, or the usefulness, completeness, or accuracy of any processes, methods or other information contained, described, disclosed, or referred to in this report. The Sponsors, the State of New York, and the contractor make no representation that the use of any product, apparatus, process, method, or any other information will not infringe privately owned rights and will assume no liability for any loss, injury, or damage resulting from, or occurring in connection with, the use of information contained, described, disclosed, or referred to in this report.

**ABSTRACT**

We used the common loon (*Gavia immer*), a top trophic-level piscivorous predator, as an indicator species to assess mercury exposure and risk in aquatic ecosystems in the Adirondack Park of New York State. We related mercury levels in loons to long-term reproductive success to evaluate the effects of mercury contamination on the breeding loon population in the Park, and enable the development of a mercury hazard profile. We used mercury levels from abiotic (water column and sediment) and biotic (common loon blood, feathers, and eggs; prey fish; crayfish; and zooplankton) samples collected on 44 lakes over a two-year period (2003 to 2004) to develop a mercury exposure profile and to quantitatively assess the ecological risk that mercury deposition poses to Adirondack waterbodies. Loons were sampled and monitored on the study lakes from 1998-2007. Mean mercury concentrations within the food web increased by many orders of magnitude as it moved from lower trophic levels (water, zooplankton, and crayfish) to higher trophic levels (fish and loons). There was a strong correlation between large and extra-large fish mercury and loon blood mercury. Lake acidity also correlated with mercury levels, with more acidic lakes exhibiting higher mercury concentrations in fish and loons. Twenty-one percent of males and 8% of females were at a high risk of behavioral and reproductive impacts based on blood mercury exposure, and 37% of male and 7% of female birds were at high risk based on feather mercury exposure. A Wildlife Criterion Value indicated that a water mercury level of 2.00 ng Hg/L or less is protective of male loons, while a water mercury level of 1.69 ng Hg/L or less is protective of females. Female and male loons in the highest exposure category showed a 32% and 56% reduction in the number of chicks fledged per year, respectively, compared to birds in the lowest exposure category. Quantile regression found a negative correlation between loon productivity and mercury levels for both female and male loons, and indicated that the maximum Adirondack loon productivity with negligible mercury exposure would be ~1.0 chick fledged/territorial pair and loon reproductive success would be reduced by 50% when female blood mercury levels were 3.30 µg/g or male blood mercury levels were 4.50 µg/g;. Population model results indicated that the portion of the Adirondack loon population exposed to high mercury levels has a reduced growth rate ($\lambda = 1.0005$), compared to birds with low body burdens of mercury ($\lambda = 1.026$). The results of this project will assist in the continued refinement of state and national policies and regulations that effectively address the ecological injury mercury and other contaminants pose to freshwater ecosystems.

**KEYWORDS:** Common Loon, Methylmercury, Wildlife Criterion Value, Adirondack Park, Acid Deposition

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## SUMMARY

The Wildlife Conservation Society's Adirondack Program and Biodiversity Research Institute (BRI)'s Adirondack Center for Loon Conservation, in partnership with the Natural History Museum of the Adirondacks, New York State Department of Environmental Conservation, and the Audubon Society of New York, used the common loon (*Gavia immer*) as an indicator species to assess the mercury exposure and risk in aquatic ecosystems in the Adirondack Park of New York State. This report to the New York State Energy Research and Development Authority (NYSERDA), a sponsor of this project, presents the project's design, methodology, results, and recommendations for the future.

Abiotic (water column and sediment) and biotic (common loon blood, feathers, and eggs; prey fish; crayfish; and zooplankton) mercury levels were used to develop a mercury exposure profile and to quantitatively assess the ecological risk that mercury deposition poses to the Adirondack loon population. Biotic and abiotic samples were collected on 44 lakes within the Adirondack Park over a two-year period (2003 to 2004). Water and fish samples were collected from all 44 lakes, zooplankton from 43 lakes, crayfish samples from 26 lakes, and sediment samples from 32 lakes. Loons were sampled and monitored from 1998-2007. Loon blood samples were collected from 44 lakes, loon feather samples from 40 lakes, and nonviable eggs were collected from 29 lakes.

The mean adult loon blood mercury level on each lake was 1.97 µg/g (ww) (± 0.17 SE), with a wide range of variation across lakes (range 0.58 – 5.62 µg/g). Females averaged lower blood and feather mercury loads than males. Juvenile loon blood mercury level was considerably lower than adults, averaging 0.24 µg/g (ww) (±0.03 SE), with a range from 0.01 µg/g to 0.76 µg/g. Nonviable eggs were collected at 29 study lakes, and total mercury concentrations ranged from 0.35 µg/g (ww) to 2.15 µg/g (ww) (mean = 0.80 µg/g ± 0.09 SE).

Mean mercury concentrations within the food web increased by many orders of magnitude as it moved from lower trophic levels (water = 0.0000017 µg/g, zooplankton = 0.006 µg/g, and crayfish = 0.047 µg/g) to higher trophic levels (small to medium sized fish = 0.096 µg/g, large to extra-large fish = 0.167 µg/g, and loons: adult female blood = 1.72 µg/g, adult male blood = 2.16 µg/g). There was a strong correlation between large and extra-large fish mercury (represented as a yellow perch equivalent) and common loon blood mercury (represented as a male or female loon unit). Lake acidity also correlated with mercury levels, with more acidic lakes exhibiting higher mercury concentrations in both fish and loon tissues. Although no significant spatial trends in mercury availability within the Adirondacks were observed, it did appear that the southwestern Adirondacks tended to have lakes with higher loon mercury levels, corresponding to increased acid deposition for that area.

Loons were placed into four risk categories of mercury concentrations in their tissues, based on previous research for effects levels conducted by BRI and others (Thompson 1996; Evers et al. 2003, 2008; Burgess

and Meyer 2008). We determined that 21% of the male Adirondack common loons and 8% of the female common loons included in this study were at a high risk of behavioral and reproductive impacts based on blood mercury exposure (reflective of short-term mercury accumulation), and 37% of male and 7% of female study birds were at high risk of impacts based on feather mercury exposure (reflective of long-term mercury accumulation). Thirteen percent of the Adirondack loon eggs sampled were at high risk for mercury exposure, indicating that, if the chicks hatched, their behaviors would be abnormal, and they would have a reduced likelihood of surviving to fledging.

We modified the Wildlife Criterion Value (WCV) formula by Nichols et al. (1999), which estimates wildlife population viability through measurement of contaminant stressors such as surface water mercury concentrations, with variables specific to the Adirondack Park, to develop a sensitive and appropriate New York-based WCV. We determined that an unfiltered water sample equal to or less than 2.00 ng Hg/L is protective of male common loons, while a water sample of 1.69 ng Hg/L or less is protective of female common loons. These WCVs are greater than the WCV of 1.30 ng Hg/L that the Great Lakes Water Quality Initiative uses for avian species (Evers et al. 2004). Based on the water samples collected from our study lakes, we estimate that the WCV accurately protects 61% of females and 73% of males, indicating that use of a water mercury concentration to protect loons in the Adirondack Park requires further study. More rigorous sampling of the abiotic compartment over a wider temporal and spatial scale is necessary to fully understand how these water quality parameters relate to loon reproductive success.

From 1999-2007, we monitored the study lakes weekly during the summer breeding season to determine the return rate and reproductive success of more than 150 adult loons that were uniquely color-banded from 1998-2006, and had potential to return to their capture lakes. We determined annual loon productivity as the average number of chicks fledged per territorial pair per year, for territories where we obtained greater than three consecutive years of productivity data. For the 54 territories where annual loon productivity met these criteria, female loons in the highest mercury exposure category (2–4 μg/g) showed a 32% reduction in the number of chicks fledged per year compared to those in the lowest exposure category (0–1 μg/g). Males in the highest mercury exposure group (>3 μg/g) showed 56% reduction in reproductive success compared to those in the lower exposure group (1–2 μg/g).

Because common loon mercury data are from multiple tissues (i.e., adult male and female blood, juvenile blood, and loon eggs), we converted mercury concentrations to a single common unit to best evaluate and utilize existing data from various biotic compartments, thus facilitating comparisons between locations and years. The female loon unit (FLU) represents the expected or observed blood mercury of adult females, and is the more universal unit because it includes egg and juvenile data. The male loon unit (MLU) predicts adult male exposure, which is often more severe given the larger size of males in an area; thus the MLU provides an indication of the potential for population-level adverse effects of mercury exposure (Evers et al. 2011). We found a negative correlation between productivity and mercury level for both female loon unit mercury equivalent ([Productivity] = -0.128[FLU] + 0.764) and male loon unit mercury equivalent

([Productivity] = -0.0992[MLU] + 0.806). For both males and females, the slope of the regression line increased at the 80$^{th}$ and 90$^{th}$ quantiles, indicating that mercury likely exerts more pressure on the upper limits of the population in the Adirondack Park. Based on the results of our quantile regression analysis, we found that the maximum Adirondack loon productivity with negligible female or male loon mercury exposure was ~1.0 chick fledged /territorial pair, and that productivity would be reduced by 50% when female blood mercury levels were 3.3 μg/g or male blood mercury levels were 4.5 μg/g.

Data collected for this project were incorporated into a common loon population model developed by the U.S. Environmental Protection Agency (EPA) to assess the risk that mercury contamination poses to the population growth rate of Adirondack loons (Grear et al. 2009). Model results indicated that the portion of the Adirondack common loon population exposed to high mercury levels has a much reduced growth rate ($\lambda = 1.0005$), compared to that of birds that had low body burdens of mercury ($\lambda = 1.026$).

Our study provides additional support for the critical need to better regulate mercury emissions on national and local scales to protect biota living in aquatic ecosystems from the impacts of environmental mercury contamination. Our results provide valuable new information that (1) contributes to documenting the extent of mercury contamination and its impacts to New York's aquatic ecosystems; (2) provides evidence for ecological damage to public resources; (3) establishes a baseline for detecting future changes in biotic impacts from atmospheric mercury deposition; and (4) provides science-based justification for policy-makers to stringently regulate mercury and acidic emissions on local, regional, and national scales. Long-term studies of biotic mercury levels, particularly those of high-trophic level species living in acidic or high mercury habitats, contribute much information about the risks mercury and acidic deposition pose to wildlife and aquatic ecosystems. A national standardized biotic mercury monitoring program, as is proposed in the National Mercury Monitoring Program (Mason et al. 2005), would greatly inform federal and state mercury-related policies, provide data for predictive models, and characterize the biological effects in the United States from the redistribution of anthropogenic mercury on the landscape (Evers et al. 2011). The proposed mercury monitoring program would also ensure that recently implemented New York State and regional regulations, and the recently finalized U.S. EPA Mercury and Air Toxics Standards (MATS) Rule, are effective at preventing local biological mercury hotspots (Evers et al. 2007) and biotic impacts, such as the observed decreased reproductive success in a portion of the Adirondack common loon population.

# 1.0 INTRODUCTION

Environmental mercury (Hg) is a naturally occurring element in our landscape, with high concentrations prevalent in aquatic biota. Nevertheless, analyses of lake sediment cores indicate that the current rate of regional mercury deposition is 2-5 times greater than historical levels (pre-1940s; Swain et al. 1992). Scientists have traced this mercury increase to dry and wet atmospheric particulate fallout. Anthropogenically caused atmospheric mercury primarily originates from coal burning and incinerator emissions. Studies comparing fish mercury concentrations with rates of atmospheric deposition have found that these anthropogenic sources account for a major contribution to the aquatic system load (NESCAUM 1998). Pollution prevention programs exist to remove mercury from the environment and products, and stringent regulations of mercury emissions from coal fired power plants have recently been implemented in New England and New York. Federally, the U.S. EPA finalized the Mercury and Air Toxics Standards (MATS) Rule in December 2011, which requires all U.S. coal-fired power plants to adopt best available pollution control technology by 2016 (US EPA 2011, Docket Nos. EPA-HQ-OAR-2009-0234; EPA-HQOAR-2011-0044).

The current availability of methylmercury (MeHg) in aquatic ecosystems of northeastern North America is at levels posing risks to human and ecological health. This is reflected in the number of human fish consumption advisories for many waterbodies in New York State, including a blanket advisory for the Adirondack Park (New York State Dept. of Health 2011). The concentration of mercury in aquatic ecosystems varies considerably in response to methylmercury availability, which is affected by lake hydrology, biogeochemistry, habitat, topography, and proximity to airborne sources. Mercury is of especially high concern in acidic environments, such as in many Northeastern lakes, where elemental mercury is converted at a higher rate to methylmercury, the toxic form that magnifies up the food web.

The common loon (*Gavia immer*), a Species of Special Concern in New York State, breeds on waterbodies throughout New York's six-million acre Adirondack Park. Loons are piscivorous predators at the top of the food chain, and thus, have potential to be detrimentally affected by toxins, such as mercury, that bioaccumulate and biomagnify through the environment. Methylmercury, a neurotoxin, has been demonstrated to affect the reproduction, behavior, and survival of loons (Nocera and Taylor 1988, Meyer et al. 1998, Counard 2001, Evers 2001, Evers et al. 2008), and potentially other wildlife species (Thompson 1996). Two independent and geographically distinct studies have shown that, for the common loon, blood mercury concentrations above 3.0 µg/g (wet weight, ww) cause detrimental impacts to reproduction, potentially leading to population level declines (Evers et al. 2008, Burgess and Meyer 2008).

Current mercury risk assessments in the Northeast have shown that the common loon is a suitable bioindicator of aquatic mercury toxicity based on its ecology, the logistics of studying this species, and the high value the public places on these charismatic birds (Evers 2006). Large scale studies have shown that

anthropogenic inputs of mercury into the environment have resulted in an increasing gradient of mercury found in loons from west to east across North America (Evers et al. 1998). Despite the depth of knowledge about loons and mercury in other areas, this is the first project to evaluate how mercury impacts the common loon population in the Adirondack Park.

The evidence is compelling that loons with elevated mercury exposure experience numerous negative neurotoxic, physiologic, and reproductive impacts, including the production of smaller eggs (Evers et al. 2003), increased time spent in low-energy behaviors (Evers et al. 2004; 2008), reduced diving frequency (Olsen et al. 2000), decreased time spent incubating eggs (Evers et al. 2004; 2008), reduced chick feeding rates by adults (Counard 2001), and less back-riding by chicks (Nocera and Taylor 1998). Scheuhammer et al. (2008) also correlated brain mercury concentrations with changes in neurotransmitter receptor concentration and other neurochemical effects. Evers et al. (2008) found that loons with elevated blood mercury levels spent less time in high energy behavioral events, such as foraging for chicks and themselves, and incubating eggs, than birds with low mercury levels. These behavioral changes could contribute to decreased survival of eggs and chicks, providing insight into why there is reduced productivity in loons with increasing mercury body burden (Evers et al. 2008).

This study was conducted by the Wildlife Conservation Society's Adirondack Program in partnership with the Biodiversity Research Institute's (BRI) Adirondack Center for Loon Conservation, the Natural History Museum of the Adirondacks, NYS Department of Environmental Conservation (NYS DEC), and the Audubon Society of New York. It expanded on loon mercury exposure and reproductive success data collected on loons in the Adirondack Park from 1998-2000 by the U.S. Fish and Wildlife Service and BRI. Based on risk categories developed from the literature and *in situ* studies by BRI and their collaborators, results from 1998-2000 indicated that 17% of the loons sampled in the Adirondacks were estimated to be at risk for harmful effects from mercury contamination (Schoch and Evers 2002). Specifically, this project determines levels of abiotic and biotic mercury exposure to help assess the risk to human and ecological health in New York State and northeastern North America.

The project examines mercury concentrations at different levels of the aquatic food web because lakes are the recipients of mercury in the watershed, provide microhabitat for methylmercury production, and are the sites of methylmercury biomagnification (Wiener et al. 2003). To provide statistical robustness, 44 lakes inhabited by loons were examined from 2003-2004. The study lakes selected were coordinated with ongoing or previous water quality research, such as that conducted by the Adirondack Lakes Survey Corporation, Adirondack Effects Assessment Program, and the US EPA's Environmental Monitoring and Assessment Program, to complement existing datasets by other researchers in the Park.

This project was also coordinated with other research and monitoring projects in New York State and northeastern North America. Such collaboration provides a better assessment of the neurologic, behavioral, and physical impacts of mercury exposure on loons. Loon productivity and mercury results were

incorporated into a US EPA common loon population model to determine if mercury toxicity is resulting in a negative impact (i.e., recruitment does not compensate adult mortality) on the breeding loon population in the Park. Biotic and abiotic mercury data were provided to the Northeast Ecosystem Research Cooperative and the U.S. EPA for use in their region-wide databases evaluating the variation in mercury concentration across northeastern North American aquatic ecosystems as related to depositional gradients. Prey fish mercury results were provided to the NYS DEC and NYS Dept. of Health to aid in identifying Adirondack lakes that should be evaluated for fish consumption advisories to prevent human exposure to mercury.

## 2.0 OBJECTIVES

We focus on three main objectives in this report: 1) to characterize aquatic-based mercury in the Adirondack Park, 2) to use this information and published estimates of mercury risk levels to determine what percentage of the Adirondack loon population is at risk of reduced productivity, and 3) to assess the effect of mercury on the Adirondack common loon population. Each of these objectives was carried out through a variety of analyses, which are summarized below.

**1) Characterize aquatic-based mercury in the Adirondack Park.** We first wanted to quantify mercury levels in different compartments (e.g., water, fish, loons) of the aquatic ecosystem, as this is the first step needed to both set up a long-term mercury monitoring program and quantify if any injury is occurring due to current mercury loadings. We characterized mercury exposure in five ways:

a. **Individual lake mercury profiles.** We measured mercury levels in both the abiotic (water and sediment) and biotic (zooplankton, crayfish, fish, and loons) compartments of each lake. This baseline data is critical for future biomonitoring programs in the Adirondack Park.

b. **Spatial distribution of mercury.** We looked at mercury levels across the Adirondack Park to determine if certain areas are at higher risk to mercury contamination, which is important for understanding how atmospheric deposition interacts with individual watershed characteristics.

**c. Bioconcentration factor for the Adirondack Park.** We explored both percentage of methylmercury between different compartments of the aquatic ecosystem and the ratio of mercury in different trophic levels to mercury in the water column, or the bioconcentration factor. Because of intrinsic differences in geography and nutrient loading, bioconcentration factors must be calculated on a site-specific basis, and have not previously been explored in the Adirondack Park.

d. **Relationships between mercury concentrations at different levels of the food web.** We explored relationships between various aquatic compartments. Because mercury biomagnifies as it moves up the food chain, we wanted to test whether we could trace mercury levels back down the food chain from the top level predator (i.e., common loon) to the abiotic environment (i.e., water and sediment).

e. **Relationship between lake acidity and mercury.** Because other studies have shown that common loon productivity is correlated with lake acidity, we also measured lake pH to understand if there could be synergistic relationships between acidity and mercury.

**2) Develop a mercury hazard profile for the common loon and put in geographical context**

a. We used published estimates for mercury risk to common loons based on blood, feather and egg values to determine what percentage of the Adirondack loon population is at risk for reproductive impairment.

b. Using BRI's common loon mercury dataset, we compared the Adirondack Park mercury exposure to other geographical regions.

**3) Determine the effect of mercury on the Adirondack common loon population**

a. **Effect of mercury and lake acidity on loon reproductive success.** We used our long-term dataset on common loon productivity in the Adirondack Park to determine if there are differences in reproductive success related to mercury exposure or lake acidity.

b. **Model for long-term effect of mercury on the Adirondack loon population.** We utilized the US EPA common loon population model (Grear et al. 2009) to evaluate the relationship between methylmercury availability and the Park's loon population, and to assess if Adirondack loons are being negatively impacted on a population scale by mercury contamination of the aquatic ecosystem.

c. **Recommended water mercury level to protect the Adirondack common loon population.** We assessed ecological risk using a formula for a wildlife criterion value (WCV) that provides a water column mercury value that is protective of wildlife at the population level. The WCV estimates wildlife population viability through measurement of contaminant stressors such as surface water mercury concentrations (Nichols et al. 1999). Development of the WCV requires knowledge of mercury concentrations that are hazardous to the loon at the population level (i.e., test dose) as well as the bioaccumulation factor at two trophic levels (i.e. mercury increase from unfiltered water to perch; Nichols et al. 1999). A loon-based WCV provides information needed by policy makers to better regulate mercury in aquatic systems.

## 3.0 METHODS

### FIELD WORK

**Study Area and Lake Selection**

We considered a total of 831 lakes in the Adirondack Park as potential breeding habitat for loons, based on the minimum lake size loons require for breeding purposes (typically a minimum of 10 ha [~25 acres]). Then we determined the number and percent of lakes ≥ 10 ha meeting the designated criteria within each of the eight watersheds in the Adirondack Park. This provided a basis for determining the number of study lakes to be sampled in each watershed (40-50 lakes total) to statistically represent the Adirondack Park as a whole (Table 1).

Lakes that met the criteria of (1) observed presence of loons with chicks, (2) accessibility for scientists and research equipment, and (3) prior wildlife or aquatic systems research were considered for inclusion in this study. We then randomly chose lakes within each watershed to determine where sampling would occur.

**Table 1. Number of lakes sampled in 2003-2004 for each Adirondack Park (Adk Park) watershed.**

| Watershed | No. of Lakes in Watershed [a] | Target Number of Study Lakes [b] | No. of Lakes Sampled | % Study Lakes in each Watershed |
|---|---|---|---|---|
| Black | 156 | 8 - 9 | 11 | 25% |
| Champlain | 108 | 5 - 7 | 6 | 14% |
| Mohawk | 70 | 3 - 4 | 3 | 7% |
| Oswegatchie | 56 | 3 - 3 | 3 | 7% |
| St. Lawrence | 219 | 10 - 14 | 10 | 23% |
| Upper Hudson | 222 | 11 - 13 | 11 | 25% |
| **Total # Lakes:** | **831** | **40 - 50** | **44** | **100%** |

[a] Total number of lakes ≥ 10 ha within watershed

[b] If total number of study lakes = 40-50

**Loon Tissue Sample Collection**

We captured loons using nightlighting and playback techniques (Evers 2001), and followed established tissue sample collection protocols (Evers et al. 1998, 2003, 2005). We non-lethally collected loon blood samples from the tibiotarsal vein to evaluate short-term mercury accumulation in the loons. Feather samples were collected from the adults and from juvenile loons with fully emerged feathers to provide an indication of long-term mercury accumulation. Feather samples included two central tail feathers and the second secondary feather from each wing. We recorded bill and leg measurements and weight. Adult and juvenile loons (if large enough) were banded with U.S. Fish and Wildlife Service aluminum bands and a unique combination of plastic colored bands, enabling identification of individual birds to be made from a distance in subsequent observations.

A constraint of this study is that our loon mercury and productivity data (and the corresponding food web data) was limited for very acidic lakes (pH < 5.0) and for those with elevated mercury exposure levels, because our ability to capture loons is primarily restricted to birds who successfully produce chicks. Thus, loons with very elevated blood mercury concentrations may be excluded from this study, since they did not have chicks, and so were not responsive to our capture technique.

Each breeding season, we opportunistically collected abandoned non-viable loon eggs, placed them in a polyethylene bag, and froze them as soon as possible after collection. Eggs were collected from an abandoned nest only when field staff confirmed the adult loons were no longer incubating them, or they were determined nonviable (i.e., strong odor). Eggs were processed following standardized protocols (Evers et al. 2003). All sample preparation, handling, labeling, and chain of- custody procedures followed BRI's established standard operating procedures.

**Banded Loon Monitoring**

To determine the annual return rate and reproductive success of color-banded loons, trained field staff conducted observations on the territories where the birds were originally captured. Observations were conducted weekly using 10x40 binoculars from a canoe or kayak on 60-75 lakes for an 11 to 15-week period from late May until mid-August or early September during the summers of 1999-2007. If a banded loon was not found on the lake (or territory) it had occupied in previous years, then lakes (or territories) in close proximity were also checked throughout the field season to determine if the bird had returned to the area, but changed territories. We conducted regular observations of the banded loons to determine the return rate of the banded loons to the Adirondack Park and the following reproductive parameters for each territory on the study lakes:

1. Presence of a territorial pair;
2. Presence of a nesting pair (nest was observed);
3. Nesting attempts;
4. Hatching success; and
5. Fledging success, (defined as a chick that survived to 6 weeks of age or older, as chicks that survive past 6 weeks are likely to survive to the actual fledging age of 11 weeks, Evers et al. 2004).

We used the number of chicks fledged per territorial pair (CF/TP) as the reproductive endpoint of interest, since a fledged chick carries the most biological significance (Evers et al. 2004, 2008). Although studies on other species often use nesting attempts or hatching success as indicators of success, we believe that the number of chicks fledged is the most accurate assessment of overall reproductive success in common loons because nests can fail prior to detection and hatched chicks can die before confirmation.

**Water, Sediment, Zooplankton, Crayfish, and Prey Fish**

We organized sampling locations for sediment, water, zooplankton, crayfish, and prey fish by the presence of a loon territory. Food web samples were collected from 7/16/2003 to 8/28/2003 and from 7/16/2004 to

9/15/2004. We collected water mercury and chemistry samples using a standard two-person "clean hands-dirty hands" protocol (US EPA 2001). Chlorophyll samples were collected by filtration. Water chemistry samples were collected from the study lakes to evaluate the interactions between water chemistry parameters (particularly pH, dissolved organic carbon, and aluminum) and mercury levels, and enable us to interpret the water-fish-loon mercury relationships in more depth.

Sediment sample locations were determined by the locations where crayfish were collected. Sediment cores for mercury analysis were collected using a "clean hands-dirty hands" protocol (US EPA 2001). Crayfish were collected via minnow traps, visual scans, and nightlighting techniques. Crayfish were identified to species. Zooplankton samples were collected via tow-nets for taxonomy identification, biomass determination, and mercury analysis according to the protocol developed for the Regional Environmental Monitoring and Assessment Program's (REMAP) assessment of mercury in Vermont and New Hampshire lakes (Chen et al. 2000). Samples collected for taxonomic analysis were processed and preserved according to procedures presented in the protocol, and then submitted to James Sutherland with the NYS DEC and Derek Bloomquist for identification, enumeration and evaluation.

A composite of fish in each of four size-classes (small: 5–10 cm, medium: 10–15 cm, large: 15–20 cm, and extra large: 20–25 cm) was collected on each lake using hook and line, trap nets, seine nets, and gill nets. Fish species were not combined in the composite samples for mercury analysis. Staff from the NYS DEC, the Adirondack Lakes Survey Corporation, and BRI collected fish samples for this project. Fish were identified to species. Otoliths and scales were collected from fish larger than 15 cm in length to aid in aging the size classes.

We submitted the biotic and abiotic samples collected during the summers of 2003-2004 to the Center for Environmental Systems Engineering Laboratory at Syracuse University, Dartmouth College, Univ. Connecticut's Center of Environmental Sciences and Engineering, and the Trace Element Research Laboratory (TERL) at Texas A&M University for analysis of mercury concentrations and water chemistry parameters.

## LABORATORY ANALYSIS

### Water Chemistry and Water, Sediment, and Zooplankton Mercury

Water chemistry, including total mercury (THg) and methylmercury (MeHg), sediment total mercury and methylmercury, and zooplankton total mercury were analyzed at the Center for Environmental Systems and Engineering, Syracuse University, under the direction of Dr. Charles Driscoll. Crayfish and fish total mercury were analyzed at the Trace Element Research Lab under the direction of Dr. Robert Taylor. All labs met basic quality assurance/quality control guidelines.

Total mercury was analyzed via oxidation, purge and trap, and cold vapor atomic fluorescence spectroscopy (CVAFS, Tekran model 2600) based on USEPA method 1631 (2002, revision E). Methylmercury was analyzed via distillation, aqueous ethylation, purge and trap, desorption, and CVAFS based on USEPA method 1631 (2001). All samples were analyzed for total mercury. Methylmercury was analyzed in water, sediment, and zooplankton. All biotic mercury and methylmercury concentrations are expressed in µg/g on a wet weight (ww) basis. Ancillary water chemistry parameters, including pH, acid neutralizing capacity (ANC), dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), Na, K, Ca, Mg, Si, $NH_4$, $NO_3$, $SO_4$, F, Cl, total phosphorus (P), chlorophyll a, monomeric aluminum ($Al_m$) and non-labile (organic) monomeric aluminum ($Al_o$), were analyzed from the same sample that was analyzed for mercury species according to standard methods (APHA/AWWA/WEF 1998).

### Zooplankton Identification

Identification and enumeration methods for zooplankton samples collected for taxonomic analysis were as follows:

1. Bottles containing samples were concentrated to a volume of 20 to 100 mL using a filter funnel with a No. 20 (76µm) mesh to provide a concentration of organisms that was efficient for counting.

2. The sample was gently stirred to provide random mixing. Using a wide-mouth Hensen-Stempel pipette, a subsample was transferred to a 1 ml Sedgwick-Rafter counting cell. The closed counting cell keeps specimens immobile and allows identification of smaller rotifers as well as the larger crustaceans.

3. The specimens in the counting cell were identified and counted using an inverted compound microscope at 150X magnification.

4. The density (abundance) of organisms was calculated using the formula: Individuals/$m^3$ = (n x k)/$m^3$, where (n) is the number of counts, (k) is the proportion of total sample volume to subsample volume, and ($m^3$) is the amount of water filtered by the net.

Most zooplankton samples contain many more organisms than can be enumerated efficiently. The size of the sub-sample, therefore, was determined with two minimum criteria to provide a consistent level of

precision. First, the dominant taxon was enumerated until approximately 100 specimens were observed. Second, the sample was counted until no new species were observed in the pipetted sub-sample.

Preserved samples contained specimens of varied quality and ease of identification. For this reason, individual species particularly difficult to identify (e.g. certain rotifers) were lumped into larger taxonomic units. This procedure also increased the number of specimens in each unit and, thus, increases the precision for that unit (Bloomquist and Sutherland, 2005).

**Loon Tissue Laboratory Analysis**

Laboratory protocols for analyzing total mercury in common loon tissues followed Evers et al. (2003) for eggs and Evers et al. (1998) for blood and feathers (except for feathers after 2002). Analyses for methylmercury in loon tissues were not conducted as more than 95% of blood mercury is in the methyl form (Wolfe et al. 2007). Analyses of loon blood and feathers were conducted by the Animal Health Diagnostics Laboratory, Univ. of PA, New Bolton, PA, and analysis of blood and eggs were conducted at the Trace Element Research Laboratory, Texas A&M, College Station, Texas. Loon feathers from 2003 to 2005 were analyzed by the Dept. of Public Health, Harvard University, Cambridge, MA, and from 2006-2007 at University of Connecticut's Center of Environmental Sciences and Engineering Metals Laboratory, under the direction of Dr. Christopher Perkins.

We acknowledge that there is potential for inter-laboratory error when comparing tissue mercury concentrations that have been obtained from multiple laboratories. The potential for inter-laboratory error is an inherent problem associated with most long-term contaminant studies, however, the use of multiple labs for mercury analysis is common in the peer-reviewed literature (Evers et al. 2011; Meyer et al. 2011). We took all steps to alleviate this source of error, by using only labs that were nationally recognized and well-established.

## STATISTICAL ANALYSIS

***Mercury Exposure Profile***

**Bioconcentration factor:** Because we are interested in how mercury moves throughout the food chain, we first considered the bioconcentration factor for mercury in the environment, starting with the water column. Average mercury values were obtained for all lakes in each of the following trophic levels: zooplankton, crayfish, trophic level-3 fish, trophic level-4 fish, female loons, and male loons and compared to water mercury using the following formula: (Bioconcentration factor = Hg in biota / Hg in water column). All mercury concentration values were converted to parts per million (ppm; µg/g) for comparison.

We tested the assumption that methylmercury becomes more bioavailable as we move up the food chain by comparing the percentage of methylmercury in water and zooplankton and comparing these values to known values for loon blood.

**Individual lake mercury profile:** To determine the individual lake mercury profiles, we averaged all samples from each lake to obtain the mean value for water, sediment, zooplankton, crayfish, fish and loons.

**Prey fish:** We grouped fish into 4 size categories: small (5 – 10 cm), medium (10 – 15 cm), large (15 – 20 cm), and extra large (20 – 25 cm). Based on US EPA (1997) definitions, small and medium size class fish were classified as trophic level 3 (insect prey specialists) and large and extra large size class fish were classified as trophic level 4 (fish prey specialists).

Because mercury bioaccumulation can vary based on fish species, we converted all fish captured into yellow perch equivalents (YPE) for ease of comparison (Evers et al. 2004; Simonin et al. 2008a). Barr (1996) documented the loon's favored prey item was yellow perch, which is a relatively ubiquitous species in the Adirondack region of New York. In order to build the relationships permitting calculation of yellow perch equivalents, there needed to be a sufficient crossover between yellow perch and other species for like sizes. Yellow perch data were paired with other species by lake and size category. Incorporating size within the paired data internalized the effect of size within the fish species data pairing. The predictive relationship between the various species and yellow perch was examined with this in mind.

Where a linear relationship was evident between fish species, this was calculated using linear regression. Where no particular relationship was evident, we divided the mean yellow perch concentration by the mean concentration for the paired species, to derive an adjustment factor. Where sample sizes were insufficient, YPE concentrations were not calculated. Seven fish species were able to be converted into YPEs using this design (Table 2). For more information, see Appendix A.

**Table 2. Conversion factors used to calculate yellow perch equivalents (YPE) from each prey fish species.**

| Fish Species | Abbreviation | Formula |
|---|---|---|
| Pumpkinseed | PKS | YPE = 1.32 x PKS |
| Largemouth bass | LMB | YPE = 0.76 x LMB + 0.032 |
| Smallmouth bass | SMB | YPE = 0.91 x SMB |
| Brown bullhead | BRB | YPE = 2.9 x BRB |
| Creek chub | CKC | YPE = 3.54 x CKC |
| Red-ear sunfish | RSF | YPE = 0.69 x RSF |
| Rock bass | RKB | YPE = 1.14 x RKB |

**Loon unit calculations:**

To best evaluate and utilize existing data from various biotic compartments, mercury concentrations require a single common unit. Since common loon mercury data are from multiple tissues, including adult male and female blood, juvenile blood, and loon eggs, comparisons between locations and years can be difficult to conduct or assess. To address this issue, we compiled a dataset of common loon data from New York (1998-2008, n=381). Subsets of the data, in which there were multiple mercury data points from a single territory and year, were used to develop relationships between mercury in different tissues. These models were then applied to the larger dataset to present data from all tissue types, territories and years, in a

common unit, the so-called "female loon unit" (FLU; Evers et al. 2011). Egg mercury levels are correlated with female mercury exposure, as female loons depurate mercury into their eggs (Evers et al. 2003). Juvenile loon mercury levels, likewise, could assumed to be more highly correlated with female mercury, as they tend to eat prey of similar size (as opposed to males, which are larger and tend to eat larger prey). Nevertheless, there is no clear link between egg mercury or juvenile blood mercury and male blood mercury. As such, male blood mercury, juvenile blood mercury, and egg mercury were each separately regressed with female blood mercury to convert all tissues to FLUs. Female adult blood levels were also converted into "male loon units" (MLUs), as male loons on the breeding grounds tend to have higher mercury than females regardless of body weight, presumably due to the depuration of female body mercury into eggs. Thus, presentation of mercury data in FLUs presents a different picture than in MLUs: while FLUs are a more universal unit (since they include egg and juvenile data), they represent the expected or observed blood mercury of adult females. As male mercury exposure is generally higher than for females, even in the same locations and years, examination of the data in the form of MLUs is useful for predicting male exposure in the region. Equations to convert tissue samples into FLUs or MLUs, and additional information are presented in Appendix B.

**Relationships between mercury concentrations at different levels of the food web:** To test whether different aspects of the food chain were correlated, we ran linear correlations between all measured variables. We highlight biologically and statistically significant results in this report. Because sample size varied for all measured variables, we were not able to include all of these into one multivariate regression.

**Mercury Hazard Profile**

We used published estimates for risk associated with blood, feather, and egg mercury concentrations in common loons (Evers et al. 2008) to assess what proportion of the breeding Adirondack Park common loon population is at risk to reproductive impairment from mercury contamination.

**Geographical Context**

We queried the BRI common loon database for all common loons captured on their breeding grounds (between 1 May and 30 September) to put the Adirondack loon mercury concentrations in context. We show both mean and maximum mercury concentrations.

***Effect of Mercury on the Adirondack loon population***

**Common Loon Productivity**

We used data collected from 1999-2007 on the reproductive success of banded common loons on our full set of 80 loon study lakes throughout the Adirondack Park to calculate the number of fledglings produced per territorial pair per year. Because we are comparing reproductive data over multiple years, we compare it to a "loon unit" value (which creates a type of average) instead of blood concentrations from any

individual year. A territory is defined as a certain lake or subsection of a lake, so that actual members of the territorial pair are allowed to vary between years (as would be expected if one member of the pair died or switched territories), and individual lakes can contain more than one defined territory. Territories on the same lake were not averaged, but instead kept as individual data points. To examine the interactions between mercury contamination, lake acidity, and loon productivity, we included only territories that were monitored every year for three or more years. We refer to the average number of fledglings produced per territorial pair per year as the "overall productivity" of each territory. Since we are interested in how productivity is related to mercury body burden and lake acidity, we regressed female loon unit, male loon unit, and lake pH with productivity. We conducted standard linear least squares regression, along with quantile regression to discern the slope of the regression line at the $90^{th}$ quantile. Quantile regression has been used in other studies to show how the upper limit of loon productivity can be constrained by mercury load (Burgess and Meyer 2008). Because we are working with relatively small sample sizes, it is possible for the $90^{th}$ quantile regression to be non-significant due to lack of data at the high end of the dependent variable. In cases where the $90^{th}$ quantile regression line was non-significant, we also present the $80^{th}$ quantile regression line.

We assigned the study birds to a mercury category based on their blood mercury levels when they were captured for banding and sampling. Male loon units were grouped into 4 categories: low (0–1 µg/g), low-moderate (1–2 µg/g), moderate-high (2–3 µg/g), and high (> 3 µg/g). Because we had few female loon units in the high mercury category, FLUs were grouped into 3 categories: low (0–1 µg/g), low-moderate (1–2 µg/g), and moderate-high (2+ µg/g). We grouped lake pH into two categories: high acidity (pH < 6.3) and low acidity (pH > 6.3) based on previous studies (Alvo 1996; Meyer et al. 1995).We compared groups for both mercury level and for lake pH using a non-parametric Kruskal-Wallis test.

**Common Loon Population Model**

We utilized Grear et al.'s (2009) density-independent stage-based matrix population model to determine a population growth rate ($\lambda$) estimate for the Adirondack study loon population.

$$\lambda_{Adks} = \frac{\{P_j \; F_a\}}{\{G_j \; P_a\}}$$

Where $P_j$ is the probability of juvenile survival without transitioning to adulthood (estimated at 0.5702), $G_j$ is the probability of a juvenile growing into the adult class (estimated at 0.1842), $F_a$ is the number of female offspring produced per adult female per year ($= P_a^{(10/12)}\, b*m*r$, where b = pairing propensity, m = number of chicks fledged per territorial pair (CF/TP), and r = the sex ratio of chicks, which was set at 50:50), and $P_a$ is the annual adult survival (estimated at 0.9200, Mitro et al. 2008). The overall Adirondack loon study population included all loons sampled for this study, but we also looked at the population growth within the low-moderate and high-extra high mercury risk categories separately.

We used the population growth rate calculated to project common loon population growth over 50 years across four different scenarios of mercury risk. Starting with an estimated population of 1000 birds, we first calculated how many loons would fall into each risk category, then we modeled the population growth (based on calculations of lambda) for each subset of the population. We calculated overall population growth by adding together the projections for low-moderate and high-extra high groups, to allow us to directly compare between different mercury risk scenarios.

**Wildlife Criterion Value**

As was accomplished for the common loon population in Maine (Evers et al. 2005), we used the Wildlife Criterion Value (WCV) formula developed by Nichols et al. (1999), modified with newly acquired information to develop a sensitive and appropriate New York-based WCV. The generic WCV has several major limitations that we improved in this study. The WCV estimates wildlife population viability through measurement of contaminant stressors such as surface water mercury concentrations (Nichols et al. 1999). We utilized three standard matrices, including loon blood, feather, and egg mercury levels, to develop the wildlife criterion values (Evers et al. 2004). The WCV was calculated using the following formula (Nichols et al. 1999). We calculated a separate WCV for male and female loons in the Adirondack Park. Each specific variable is discussed in detail in the Results section.

$$\text{Wildlife Criterion Value (WCV)} = \frac{(TD \times [1/\{UF_L \times UF_A \times UF_S\}] \times WT_A}{W_A + ([FD_3 \times F_A \times BCF_3] + [FD_4 \times F_A \times BCF_4])}$$

Whereas:

- TD = Tested dose from toxicity studies with wildlife species (ug Hg/kg body weight/day).
- $UF_L$ = The uncertainty factor between the lowest observed adverse effect level (LOAEL) and the no observed adverse effect level (NOAEL).
- $UF_A$ = The uncertainty factor between species.
- $UF_S$ = The uncertainty factor between subchronic and chronic levels of impacts.
- $WT_A$ = Average species weight (kg).
- $W_A$ = Average daily volume of water consumed (L/day).
- $FD_{3,4}$ = Fraction of diet from trophic level 3 and 4.
- $F_A$ = Average daily mass of food consumed (kg/day).
- $BCF_{3,4}$ = Aquatic life bioconcentration factor for trophic level 3 and 4 (L/kg of Hg in fish / Hg in water).

We used the correlation between water total mercury and MLU/FLU created from this study to assess the accuracy of the WCV.

# 4.0 RESULTS

## SAMPLING EFFORT

### Lake Selection

Within the six Adirondack watersheds, we selected a total of 44 study lakes to sample for abiotic and biotic mercury levels during the 2003 - 2004 field seasons (Table 1, Figure 1). Three other lakes where loons were captured in 2003 or 2004 (Duck Pond, Little Charley Pond, and Putnam Pond) were omitted from the lakeset due to seasonal time limitations in collecting the biotic and abiotic samples. In addition, although loons were sampled on both Spitfire and Upper St. Regis Lakes, non-loon samples were collected only on Spitfire because the two water bodies are connected by a channel, and so, only Spitfire was included as a full study lake. Lake characteristics are described more fully in Appendix C.

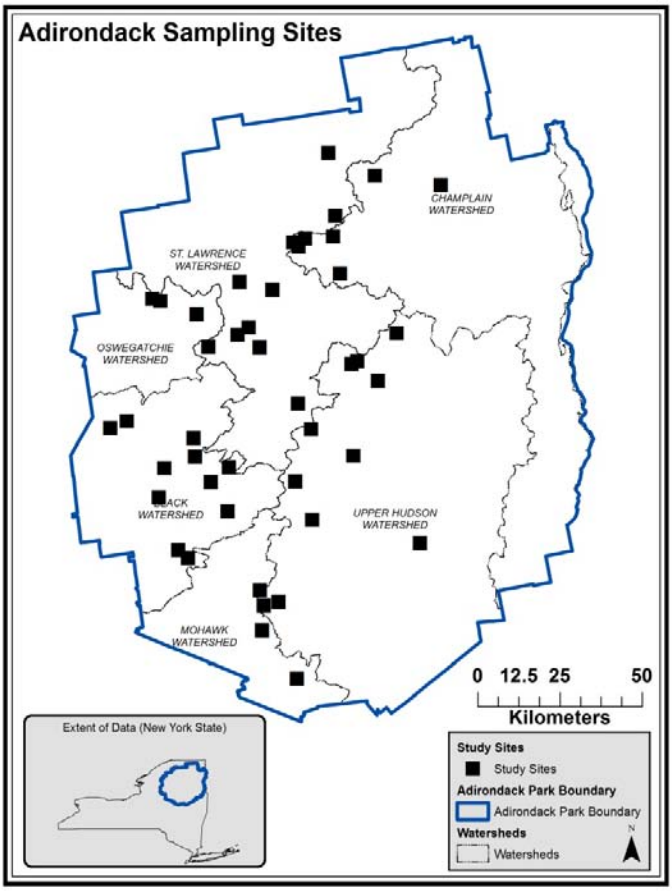


**Figure 1. Study lakes (2003–2004) located within the Adirondack Park.**

### Sampling Effort

Over the two-season sampling period, water mercury and chemistry samples were collected from all 44 study lakes, and zooplankton samples were collected from 43 of the study lakes (one lake was too shallow for a vertical tow). Sediment samples were collected from 32 lakes and crayfish from 26 of the study lakes.

Fish samples were collected from all lakes (Appendix D). Between 1998 and 2007, we sampled loon blood at all 44 of the lakes, loon feathers at 40 of the lakes and loon eggs at 29 of the lakes (Appendix D).

## ZOOPLANKTON IDENTIFICATION

### Rotifers

Thirteen species of rotifers were observed in the samples collected from the 43 lakes and ponds during 2003-2004. Three rotifers were identified to the genera level, including *Asplanchna*, *Euchlanis* and *Pleosoma*. The most common species were *Conochilus unicornis*, *Asplanchna spp*. and *Kellicottia longispina*, which occurred in 25, 22 and 19 waters, respectively; the remaining 10 rotifer species occurred only infrequently (Table 3).

**Table 3. Rotifer species identified in 43 Adirondack Park study lakes.**

| Rotifer Species | Abbreviation | Species Occurrence[a] |
|---|---|---|
| *Conochilus unicornis* | Cu | 25 |
| *Asplancha spp.* | Asp | 22 |
| *Kellicottia longispina* | Kl | 19 |
| *Trichocerca cylindrica* | Tc | 6 |
| *Kellicottia bostoniensis* | Kb | 5 |
| *Keratella cochlearis* | Kco | 5 |
| *Keratella taurocephala* | Kt | 3 |
| *Polyarthra remata* | Pr | 2 |
| *Conochiloides dossuarius* | Cd | 1 |
| *Euchlanis spp* | Esp | 1 |
| *Keratella crassa* | Kcr | 1 |
| *Keratella quadrata* | Kq | 1 |
| *Pleosoma spp.* | Psp | 1 |

[a] *total number of waters*

**Rotifer Species Numbers and Density**. The greatest number of rotifer species in the lakes and ponds sampled was 5, which occurred in four different waters including Garnet, Horseshoe, South Pond, and Woodruff (Figure 2). There were six waters, including Big Moose, Henderson, North, Seventh, South Lake, and Squaw, in which no rotifer species were observed in the aliquots examined from the collected samples. An average of 2.1 (±1.5 SD) rotifer species occurred in the 43 lakes and ponds that were sampled. The 43 study waters exhibited a wide range in total rotifer density (#/$m^3$), with an average density of 6837 per $m^3$ (±13,327 SD). Total rotifer density was spread across several orders of magnitude in the waters that contained any rotifer species. Study sites such as Beaver, Canada, Limekiln and Wolf exhibited very low (<100 organisms/$m^3$) densities, while Durant, Garnet, Kushaqua, Middle Saranac, and Spitfire displayed very high (>10,000 organisms/$m^3$) total rotifer densities (Figure 3).

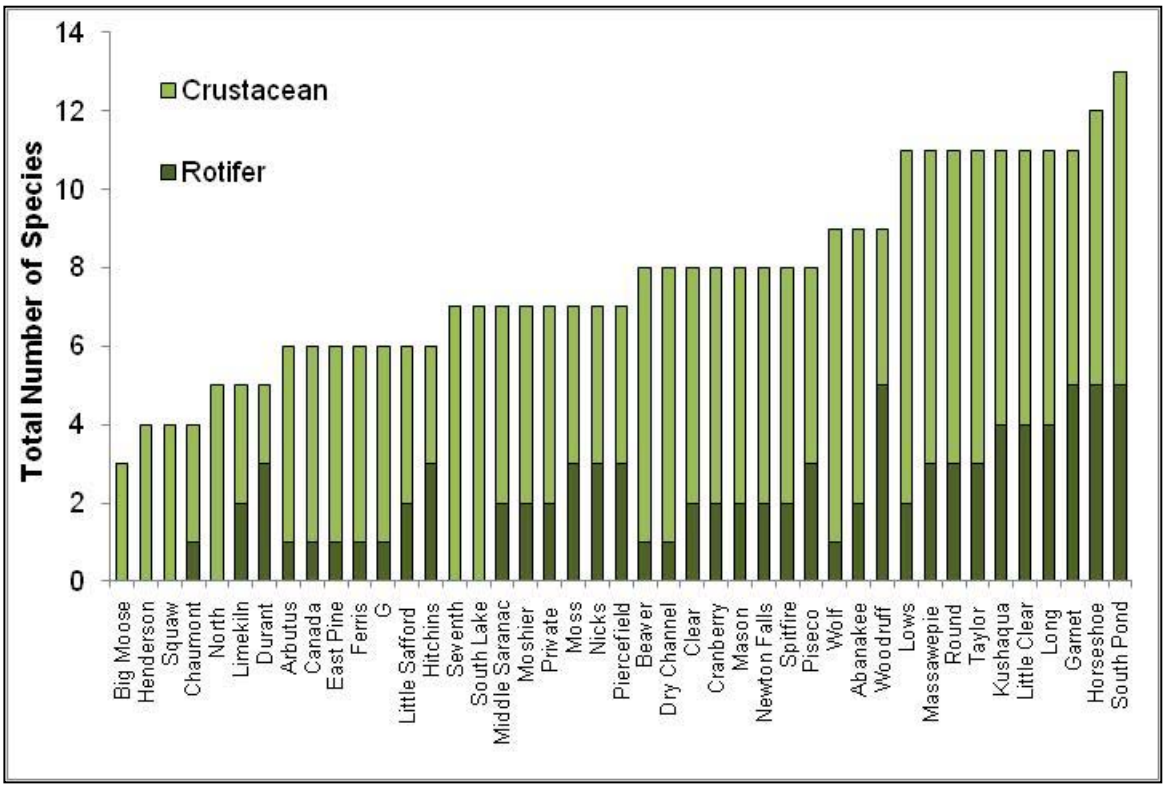


**Figure 2. Total zooplankton diversity (number of species in each lake), separated by crustacean and rotifer species.**

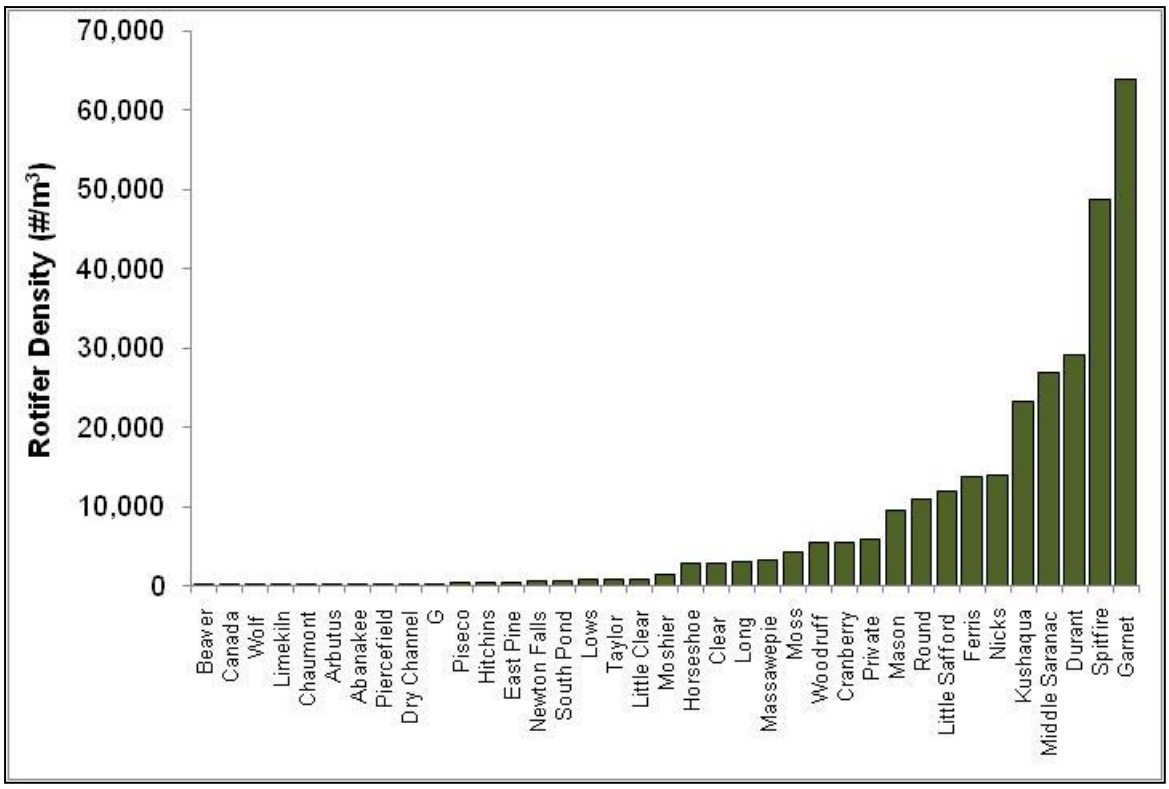


**Figure 3. Rotifer density at each sampled study lake.**

**Crustaceans**

A total of 20 crustacean species were identified in samples collected from the 43 Adirondack lakes and ponds during 2003-2004, including four calanoid copepods, four cyclopoid copepods and 12 cladocerans (Table 4). All of the crustacean zooplankton species were limnetic forms except two, *Polyphemus pediculus* and *Chydorid* spp. Evaluations to the genera level occurred with the chydorids and with *Diaphanosoma* spp., which often was distorted in the 2003 collections, making the distinction of either *birgei* or *brachyurum* impossible. As a result, all of the graphics for the 2003-2004 data show the collective *Diaphanosoma* spp. to maintain consistent reporting. The most common crustacean species, and the number of waters in which they occurred, were *Bosmina longirostris* (38 lakes), *Mesocyclops edax* (36 lakes), *Leptodiaptomus minutus* (32 lakes), *Holopedium gibberum* (27 lakes), and *Diaphanosoma* spp. (25 lakes).

**Table 4. Crustacean species identified in 43 Adirondack Park lakes.**

| | Crustacean Species | Abbreviation | Species Occurrence [a] |
|---|---|---|---|
| Calanoid copepods | *Agliodiaptomus leptopus* | Al | 1 |
| | *Epischura lacustris* | El | 3 |
| | *Leptodiaptomus minutus* | Lm | 33 |
| | *Skistodiaptomus oregonensis* | So | 3 |
| | *Calanoid nauplii* | | 1 |
| Cyclopoid copepods | *Cyclops scutifer* | Cs | 3 |
| | *Diacyclops bicuspidatus thomasi* | Dbt | 1 |
| | *Mesocyclops edax* | Me | 36 |
| | *Tropocyclops prasinus* | Tp | 15 |
| | *Unknown cyclopoid* | | 11 |
| | *Cyclopoid nauplii* | | 14 |
| Cladocerans | *Bosmina longirostris* | Bl | 38 |
| | *Chydoridae* | Csp | 3 |
| | *Ceriodaphnia reticulata* | Cr | 9 |
| | *Daphnia ambigua* | Da | 5 |
| | *Daphnia catawba* | Dc | 9 |
| | *Daphnia parvula* | Dpa | 3 |
| | *Daphnia pulex* | Dpu | 18 |
| | *Daphnia retrocurva* | Dr | 5 |
| | *Diaphanosoma spp.* | Dsp | 25 |
| | *Holopedium gibberum* | Hg | 27 |
| | *Polyphemus pediculus* | Pp | 3 |

[a] Total number of waters

**Crustacean Species Numbers and Density.** The greatest number of crustacean species in the study lakes was 9 in Lows Lake, followed by 8 species in Massawepie, Round, South Pond, Taylor and Wolf (Figure 2). Lake Durant contained only 2 species, while Big Moose, Chaumont, Hitchins and Limekiln each had 3 crustacean species. The average number of crustacean species in the 43 lakes and ponds was 5.6 (±1.7 SD).

The average total density of crustacean species in the 43 Adirondack study waters was 18,806 per $m^3$ (± 15783 SD) (Figure 4). For example, Chaumont, Hitchins, Limekiln and Piercefield Flow exhibited low (<1000 organisms/$m^3$) densities, while Arbutus, Horseshoe, Massawepie and the Private water all contained very high (>30,000 organisms/$m^3$) total crustacean densities.

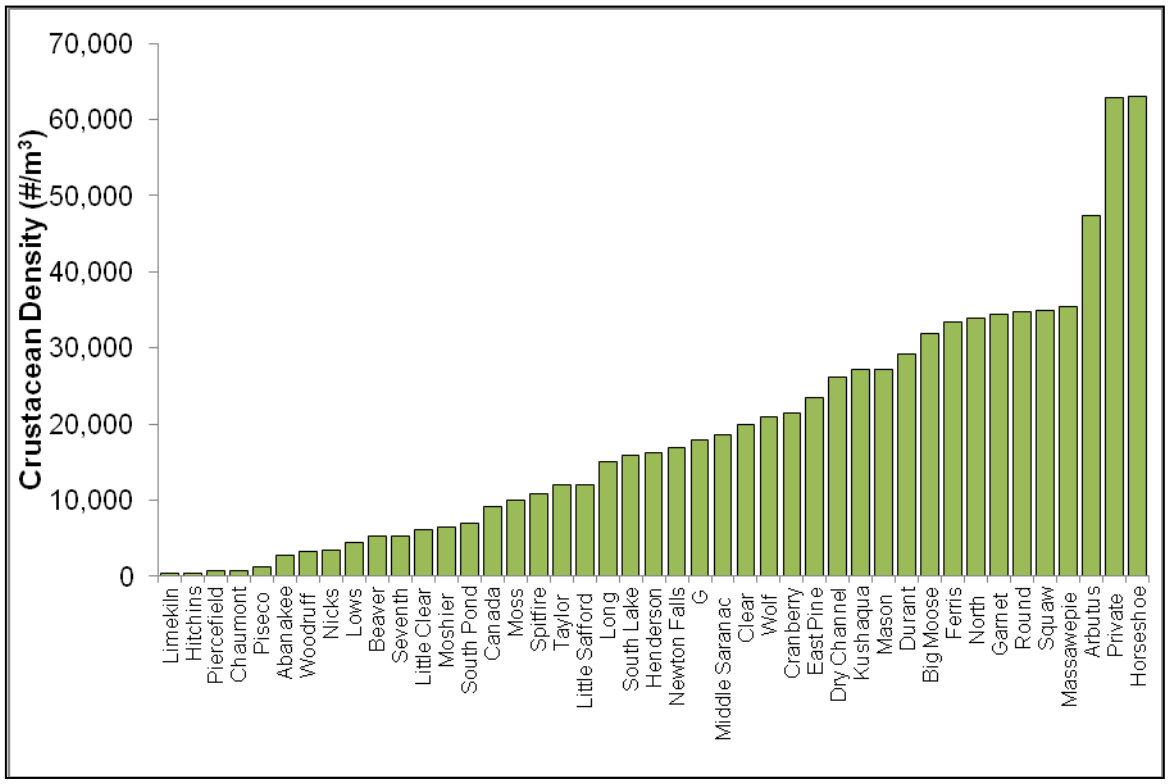


**Figure 4. Total crustacean density in sampled study lakes.**

## AQUATIC-BASED MERCURY IN THE ADIRONDACK PARK

### Individual Lake Mercury Exposure Profiles

**Abiotic Mercury Profile**. We collected 49 water samples from 44 lakes and analyzed each for various measures of water chemistry (Figure 5, Table 5, Appendix E). More than half of the study lakes were characterized by relatively low-DOC (25 out of 44 lakes <5.00 mg umol/L, Appendix E). The lakes had a mean pH of 6.54 (range: 5.26-7.82) and a mean ANC of 101 µeq/L (range: 8.28-331.49 µeq/L). Most of the study lakes were oligotrophic, with low concentrations of total P (mean: 5.00 µg/L, range: 0.00-74.88 µg/L) and chlorophyll a (mean 2.73 µg/L, 0.30–15.70 µg/L (Table 5, Appendix E).

Across all study lakes, the average water total mercury concentration equaled 1.73 ppt (SD = 0.92) and ranged from 0.096 ppt to 4.64 ppt (Figure 6, Appendix F). Methylmercury was detected in the water column at low levels (range 0.002 – 0.482 ppt). Water methylmercury was positively correlated with total mercury, but only weakly ($R^2 = 0.20$, $F = 8.91$, $p = 0.005$).

Sediment mercury samples were obtained from 26 loon territories on 26 different lakes where crayfish were captured in 2003 and 2004 (presented as ppt (parts per trillion), ww (wet weight)). The mean sediment total mercury concentration was 15,022 ppt, which is over 8,600 times higher than the water total mercury concentration. Sediment total mercury ranged from 1407 ppt (ww) to 83,799 ppt (ww) (Figure 7, Appendix

F). Methylmercury in the sediment was much lower than total mercury (range 1.19 – 4499 ppb, ww) and also showed a positive relationship between methylmercury and total mercury ($R^2 = 0.46$, $F = 19.78$, $p <$ **0.001) .**

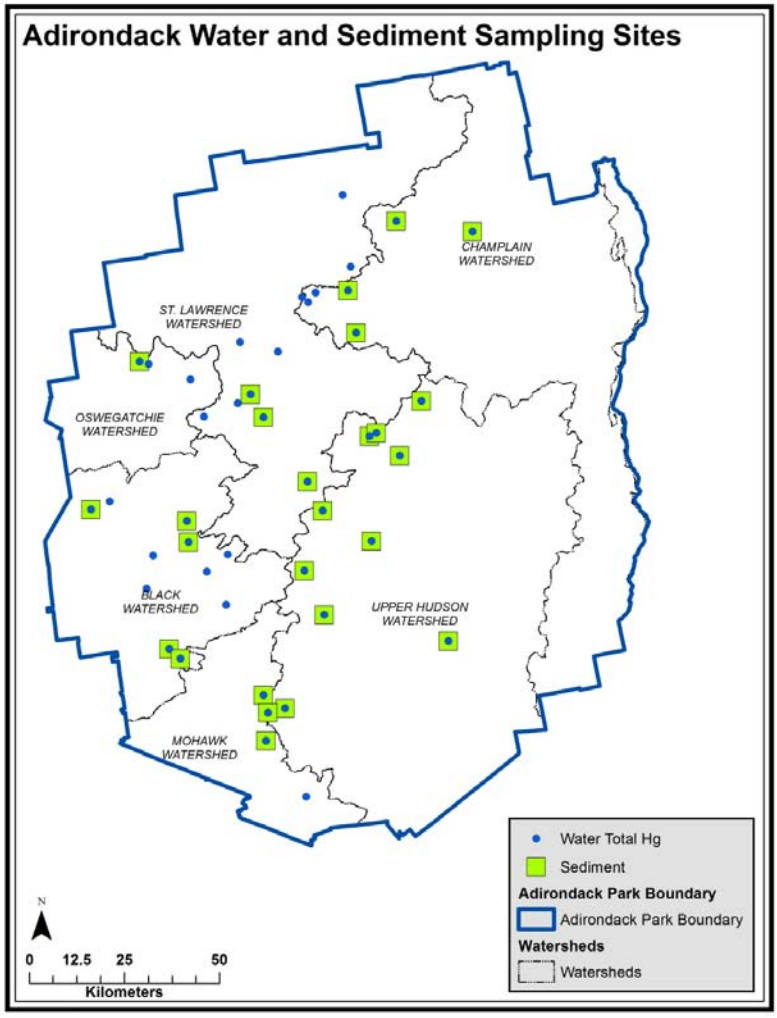


**Figure 5. Abiotic sampling locations within the Adirondack Park.**

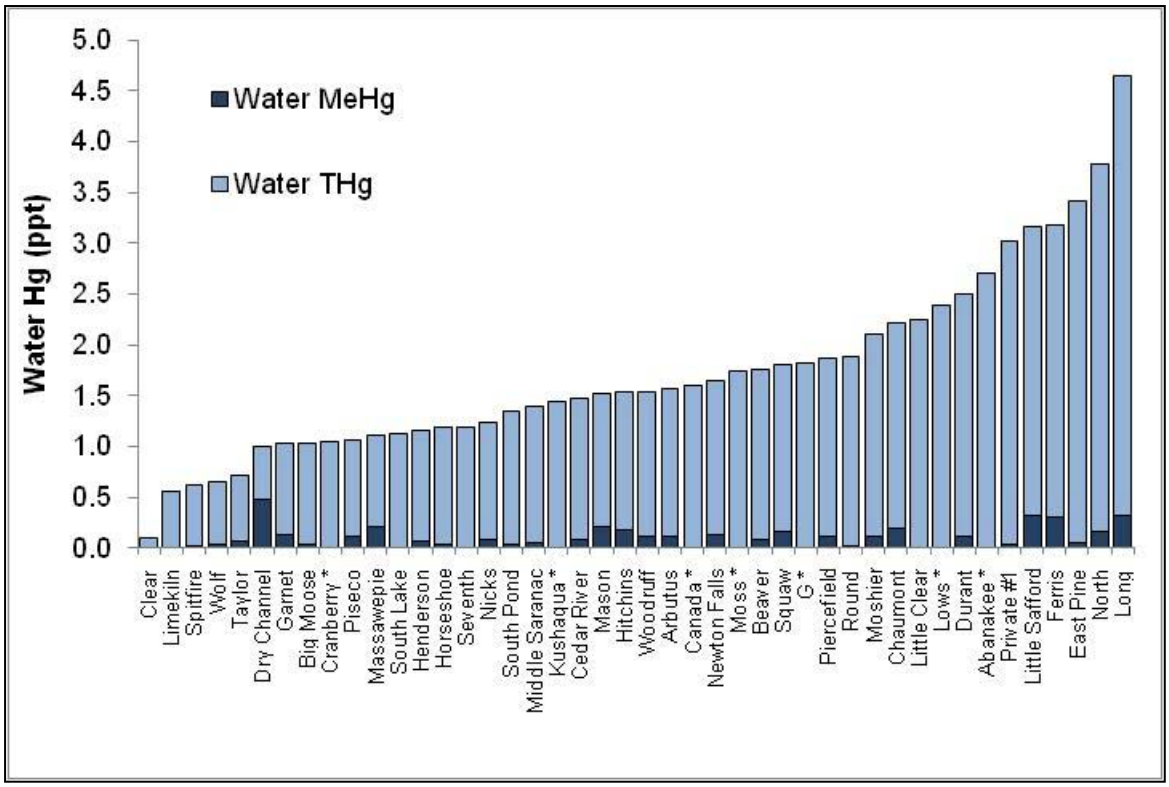


**Figure 6. Total mercury found in unfiltered water samples and proportion of total mercury that is made up of methylmercury. Asterisk (*) indicates methylmercury was not reported.**

**Table 5. Descriptive statistics for water chemistry on all New York study lakes.**

| Water Chemistry Variable | Abbreviation | Mean | Median | N (# Lakes) | SE | Min | Max |
|---|---|---|---|---|---|---|---|
| Total Mercury (ppt) | THg | 1.73 | 1.54 | 44 | 0.14 | 0.10 | 4.64 |
| Methylmercury (ppt) [a] | MeHg | 0.12 | 0.08 | 37 | 0.02 | 0.00 | 0.48 |
| pH | pH | 6.54 | 6.66 | 44 | 0.09 | 5.26 | 7.82 |
| Dissolved inorganic carbon (mg/L) | DIC | 147.35 | 119.57 | 44 | 12.63 | 32.07 | 403.53 |
| Dissolved organic carbon (umol/L) | DOC | 82.67 | 4.38 | 44 | 21.72 | 1.55 | 502.02 |
| Acid neutralizing capacity (ueq/L) | ANC | 101.33 | 85.94 | 44 | 11.93 | 8.28 | 331.49 |
| Ammonium (umol/L) | NH4 | 1.79 | 1.57 | 44 | 0.14 | 0.16 | 4.26 |
| Silicon (umol/L) | Si | 61.49 | 58.18 | 44 | 4.62 | 1.59 | 129.17 |
| Total Phosphorus [b] | Total P | 5.00 | 2.34 | 35 | 2.14 | 0.00 | 74.88 |
| Chlorophyll a (ug/L) | Chloro | 2.73 | 1.95 | 44 | 0.43 | 0.30 | 15.70 |
| Potassium (mg/L) | K | 0.42 | 0.43 | 44 | 0.02 | 0.14 | 0.72 |
| Sodium (mg/L) | Na | 1.85 | 0.91 | 43 | 0.36 | 0.30 | 13.73 |
| Calcium (mg/L) | Ca | 2.52 | 2.20 | 44 | 0.17 | 1.01 | 6.29 |
| Magnesium (mg/L) | Mg | 0.62 | 0.52 | 44 | 0.05 | 0.20 | 1.46 |
| Flourine (umol/L) | Fl | 3.44 | 3.05 | 44 | 0.26 | 0.95 | 7.17 |
| Chlorine (umol/L) | Cl | 63.74 | 11.85 | 43 | 16.93 | 5.35 | 589.53 |
| Nitrate (umol/L) | NO3 | 3.01 | 1.08 | 44 | 0.56 | 0.00 | 13.83 |
| Sulfate (umol/L) | SO4 | 41.30 | 39.41 | 44 | 1.24 | 25.11 | 73.28 |
| Monomeric aluminum (umol/L) | Al m | 0.78 | 0.67 | 44 | 0.09 | 0.33 | 3.42 |
| Non-labile (organic) monomeric aluminum (umol/L) | Al o | 0.84 | 0.77 | 44 | 0.05 | 0.43 | 2.24 |

[a] Results for 6 lakes, Abanakee, Canada, Cranberry, G, Kushaqua, and Lows, were removed from the analysis because the methylmercury was below the detection limit. Moss Lake was removed because methylmercury exceeded total mercury.

[b] Results for 5 lakes were removed from the analysis because their phosphorus value was negative (Seventh, Little Clear, Limekiln, Henderson, Dry Channel).

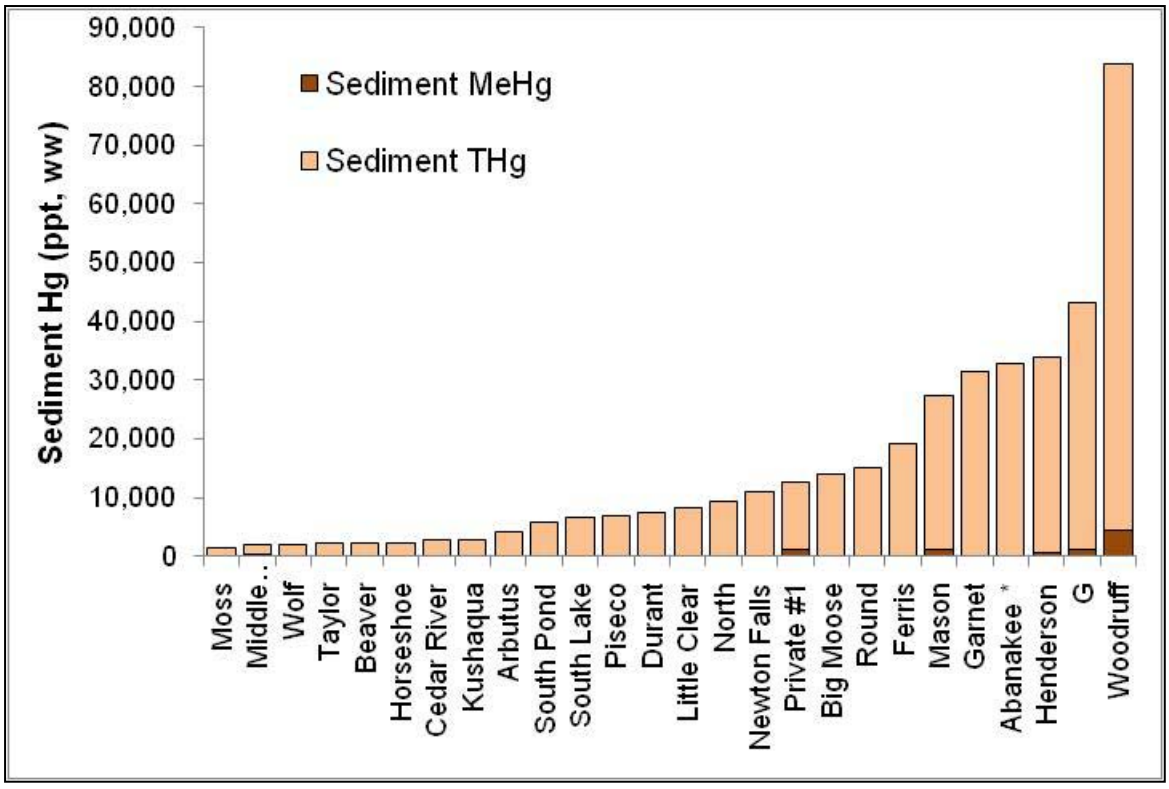


**Figure 7. Total mercury found in the sediment and proportion of total mercury made up of methylmercury. Asterisk (*) indicates methylmercury is not reported.**

**Invertebrate Prey Mercury Profile**. We obtained zooplankton mercury results for 40 study lakes, which are presented as parts per billion (ppb, ww) (Figure 8). Zooplankton total mercury ranged from 6.6 ppb to 820.3 ppb; the average for all lakes was 304 ppb (SD = 172) (Figure 9). The average methylmercury concentration in zooplankton was 70.2 ppb (SD = 3.6), ranging from 3.6 ppb to 216.7 ppb. Log-transformed methylmercury was positively correlated with log-transformed total mercury ($R^2 = 0.29$, $F = 13.75$, $p < 0.001$).

Thirty-nine crayfish mercury samples were obtained from 26 loon territories in 2003 and 2004 (Figure 8). Despite sampling effort at all 44 sites, crayfish were not found at all sites. Although crayfish samples were not analyzed for methylmercury, we do compare total mercury concentrations for whole body versus the tail only (Figure 10). On average, crayfish tail mercury (mean = 58.6 ppb, ww) was higher than crayfish whole body mercury (mean = 46.6 ppb, ww), but the two metrics are closely correlated ($R^2 = 0.98$, $F = 1361.9$, $p < 0.001$).

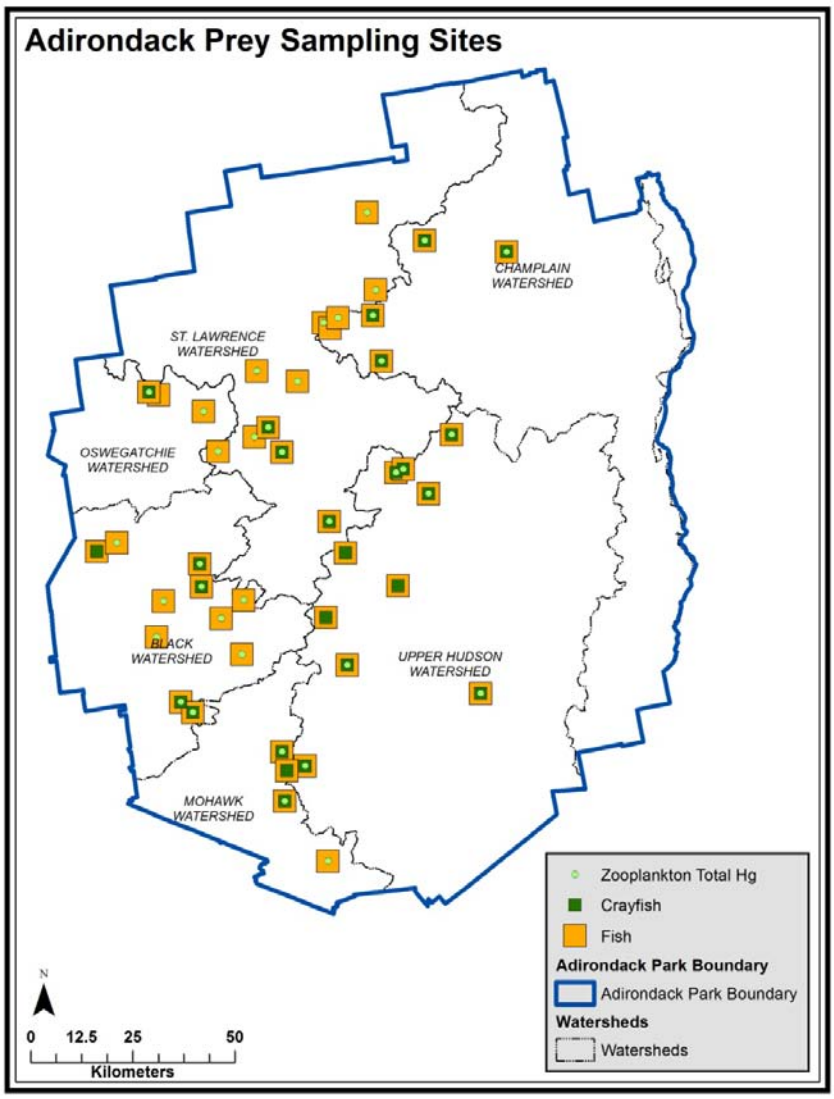


**Figure 8. Sampling locations for each of the prey sampling categories (zooplankton, crayfish, and fish).**

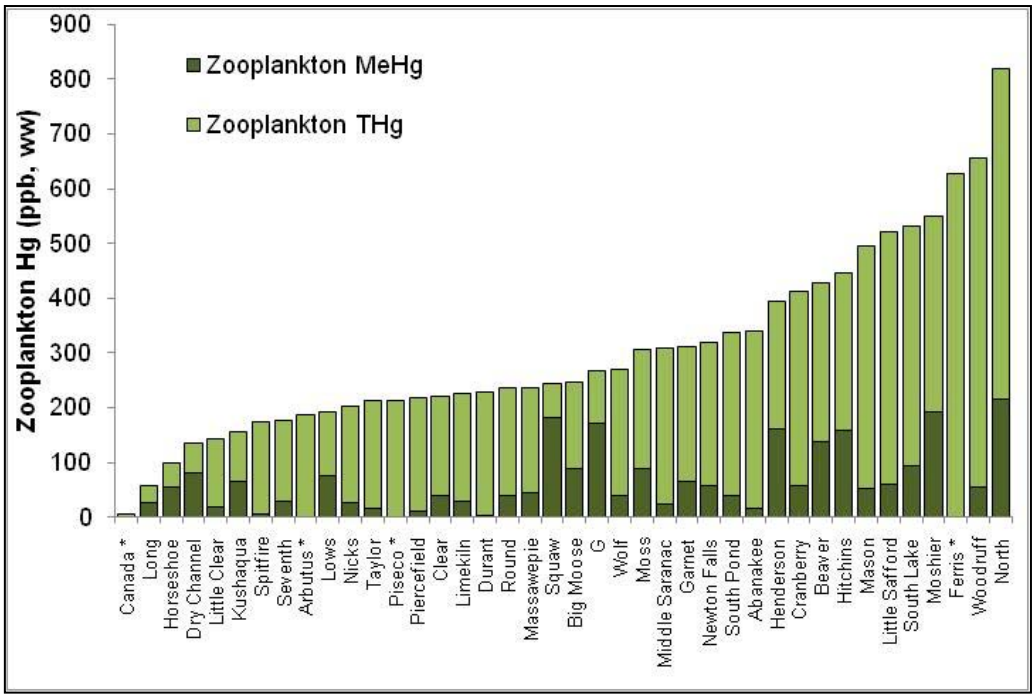


**Figure 9**. **Total mercury found in zooplankton and proportion of total mercury that is made up of methylmercury. Asterisk (*) indicates that methylmercury was not reported.**

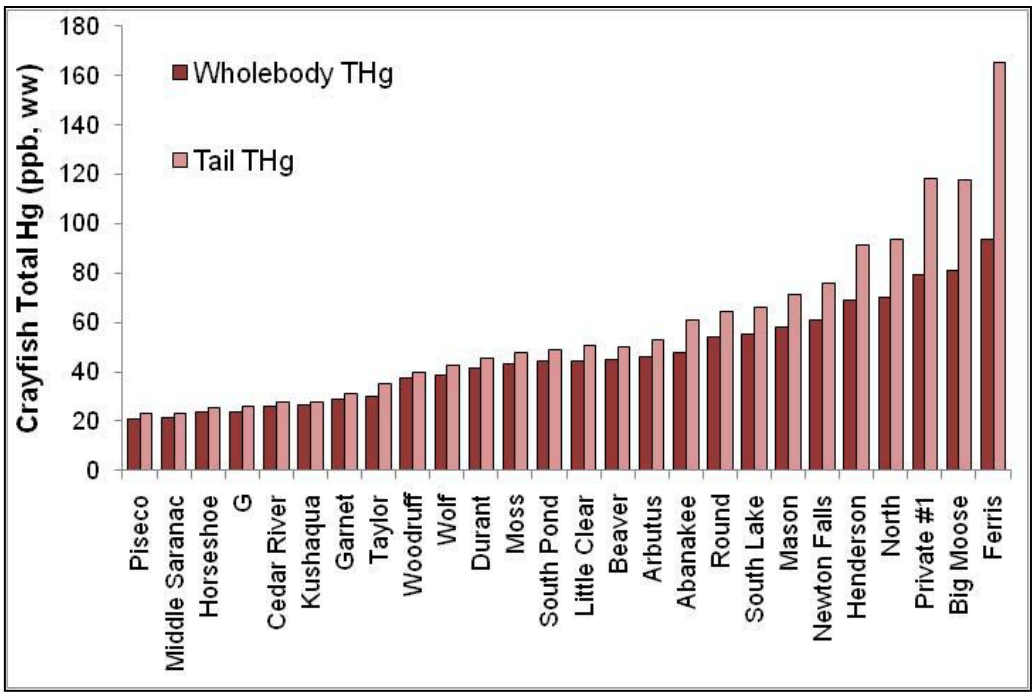


**Figure 10. Total mercury found in crayfish whole body and tail samples. Crayfish were not tested for methylmercury.**

**Fish Mercury**. We sampled yellow perch from 28 (64%) of the 44 study lakes. On average, trophic level 4 fish (mean = 0.17 µg/g, ww) had higher mercury levels than trophic level 3 fish (0.10 µg/g, ww). Because larger fish bioaccumulate more mercury than smaller fish, we report both the mercury concentrations for each fish size class (Figure 11) and for a conversion to change all species to a yellow-perch equivalent (reported for medium-sized fish only to aid in comparisons, Figure 12).

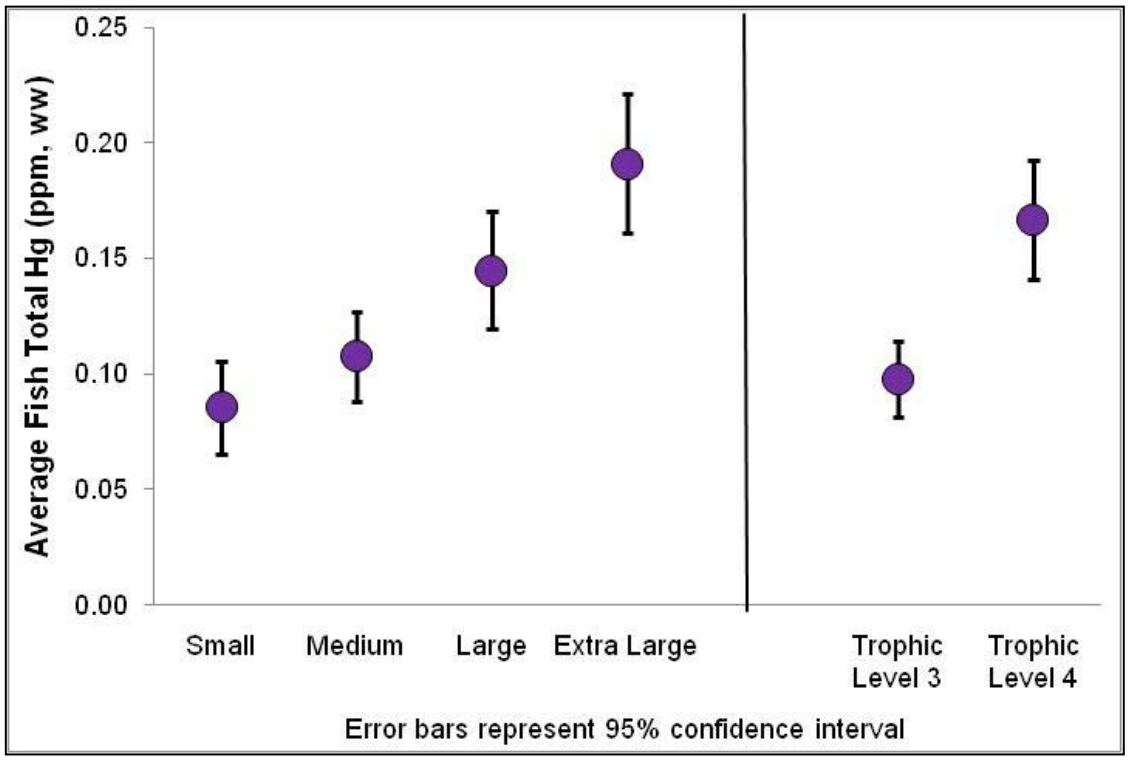


**Figure 11. Mean fish total mercury values for fish in each size category (small, medium, large, and extra large) and mean for trophic level 3 (TL3) and trophic level 4 (TL4).**

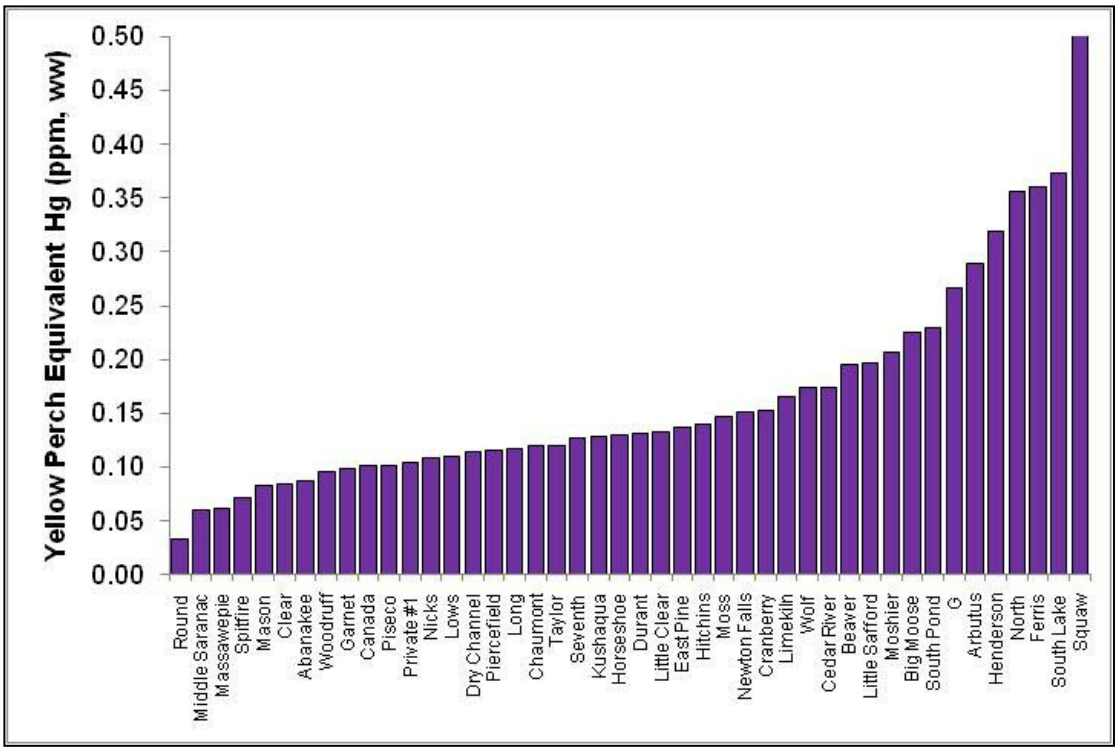


**Figure 12. Average yellow perch equivalent (medium-sized fish) total mercury for each study lake.**

**Loon Mercury**. Between 1998 and 2007, we sampled at least one type of loon tissue at each of the study lakes (blood, feather, or egg). The mean adult blood mercury level on each lake was 1.97 µg/g (ww) (± 0.173 SE), with a wide amount of variation across lakes (range 0.58 – 5.62 µg/g). As expected, females averaged lower blood and feather mercury loads than males (Table 6). Juvenile loon blood mercury level was considerably lower than adults, averaging 0.239 µg/g (ww) (±0.029 SE), with a range from 0.01 µg/g to 0.76 µg/g. Adult feathers similarly show a large amount of variation in mercury levels, ranging from 3.940 µg/g (fw) to 73.21 µg/g (fw) (Table 6). Eggs were collected at 29 study lakes, and total mercury concentrations ranged from 0.35 µg/g (ww) to 2.15 µg/g (ww) (mean = 0.8 µg/g ± 0.085 SE). We averaged multiple tissue samples collected at the same lakes to determine an average mercury value for each lake. There was a 5–fold difference in blood mercury concentrations between the highest lake (Ferris) and the lowest lake (Cedar River Flow) (Figure 13). Wolf Lake had the highest average feather mercury concentration, which was roughly 5 times higher than the lowest lake (Cedar River Flow) (Figure 14). Although juvenile blood mercury levels were much lower than adults, we also see variation between lakes (Figure 15). Egg mercury levels showed roughly 4-fold variation across lakes (Figure 16).

Because male and female common loons are different sizes, and therefore feed on slightly different prey items, it is important to consider the sexes separately. Although there is a strong correlation between average male and female blood mercury levels on each lake, males generally have higher blood mercury levels than females (Figure 17). We converted all blood mercury concentrations of sampled loons (female, male and juvenile) into female loon units and male loon units since we did not have blood samples from

both male and female loons at all lakes. We used both FLU and MLU for many of the analyses in this report.

**Table 6. Summary statistics for mercury levels in Adirondack loons, 1998-2007.**

| Media | Variable [a] | N | Mean | SD | SE | Min | Max | 1st quartile | Median | 3rd quartile |
|---|---|---|---|---|---|---|---|---|---|---|
| Loon Blood [b] | Female Blood | 36 | 1.716 | 1.040 | 0.173 | 0.430 | 5.870 | 1.080 | 1.460 | 2.265 |
| | Male Blood | 37 | 2.164 | 1.028 | 0.169 | 0.520 | 5.360 | 1.340 | 1.900 | 2.915 |
| | All Adult Blood | 42 | 1.970 | 0.989 | 0.153 | 0.580 | 5.620 | 1.243 | 1.850 | 2.533 |
| | All Juvenile Blood | 34 | 0.239 | 0.166 | 0.029 | 0.010 | 0.760 | 0.128 | 0.205 | 0.295 |
| Loon Feather [c] | Female Feather | 34 | 11.631 | 5.496 | 0.943 | 3.940 | 35.260 | 8.693 | 10.525 | 13.563 |
| | Male Feather | 36 | 19.792 | 12.758 | 2.126 | 5.190 | 73.210 | 13.570 | 15.550 | 21.138 |
| | All Adult Feathers | 40 | 16.385 | 8.522 | 1.348 | 4.570 | 48.210 | 12.168 | 13.735 | 19.193 |
| Loon Egg [b] | Eggs | 29 | 0.802 | 0.459 | 0.085 | 0.350 | 2.150 | 0.490 | 0.610 | 0.960 |
| Loon Units [b] | Average Female Loon Unit | 44 | 1.474 | 0.835 | 0.126 | 0.310 | 4.130 | 0.813 | 1.345 | 1.995 |
| | Average Male Loon Unit | 42 | 2.260 | 0.974 | 0.150 | 0.740 | 5.660 | 1.510 | 2.210 | 2.853 |

[a] All Hg values in parts per million (µg/g)
[b] Values reported at wet weight (ww)
[c] Feather Hg in fresh weight (fw)

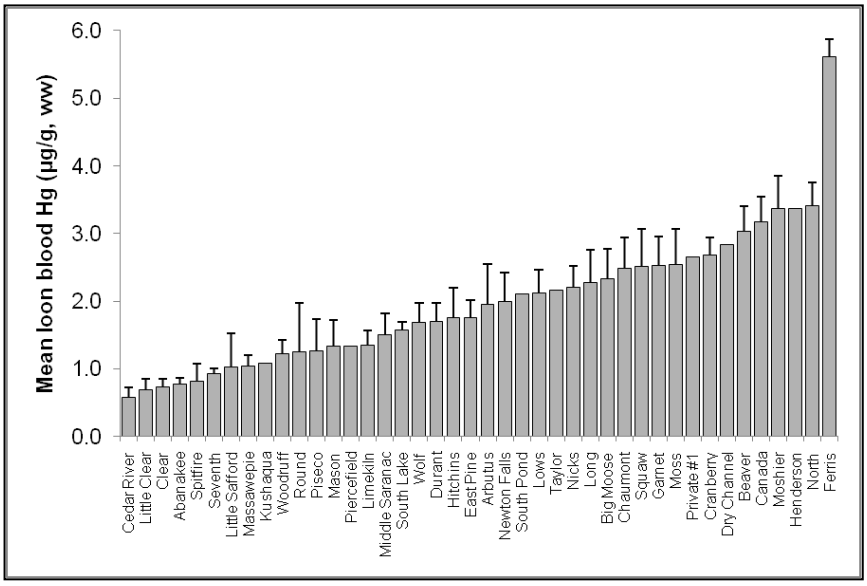


**Figure 13. Average adult (male and female combined) common loon blood mercury levels (µg/g, ww) for each study lake. Error bars indicate standard error.**

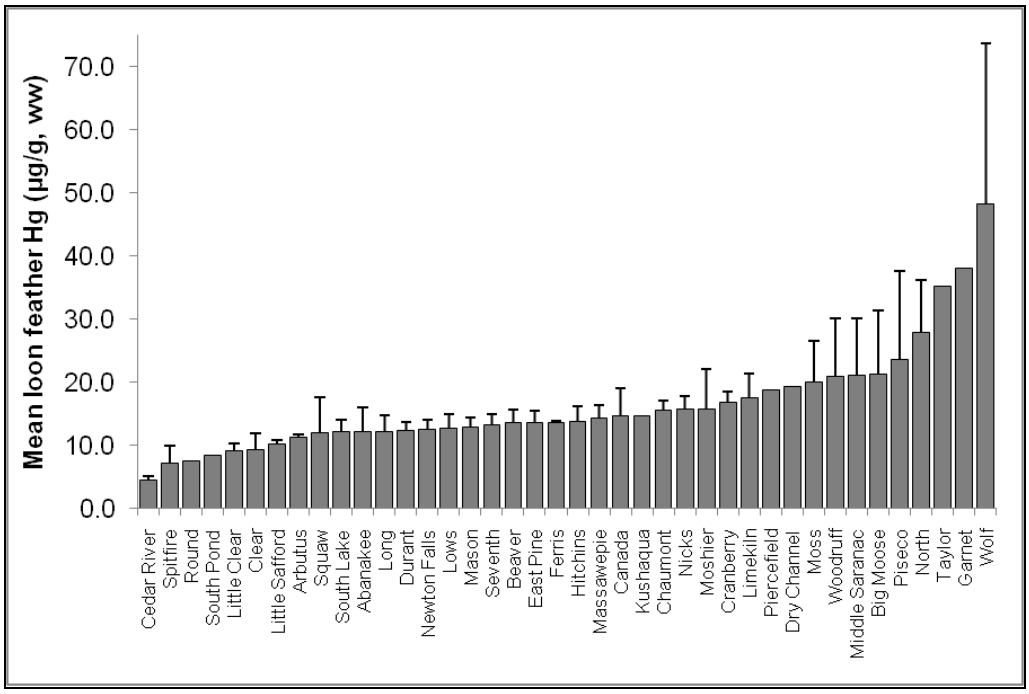


**Figure 14. Average common loon adult (male and female combined) feather mercury levels (µg/g, fw) for each study lake. Error bars indicate standard error.**

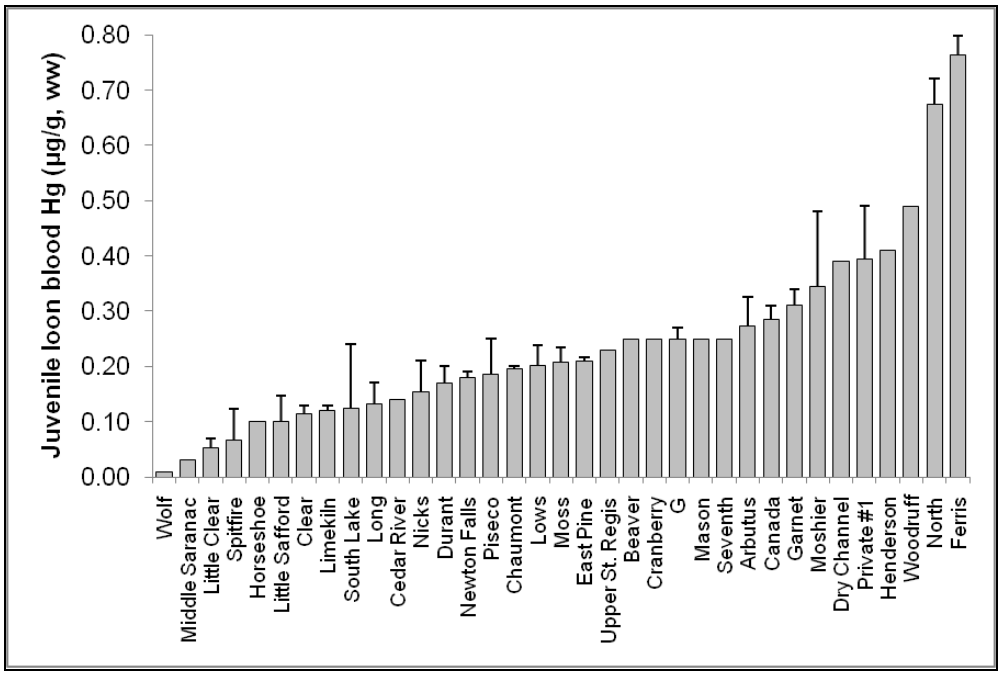


**Figure 15. Average juvenile common loon blood mercury levels (µg/g, ww) at each study lake. Error bars indicate standard error.**

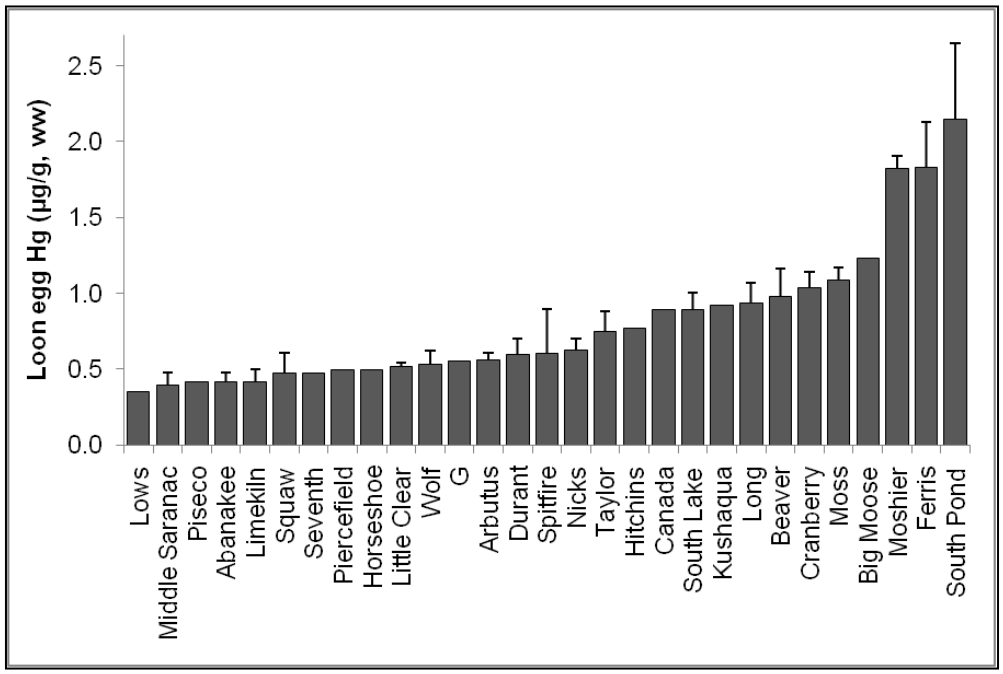


**Figure 16. Average common loon egg mercury levels (µg/g, ww) for each study lake. Error bars indicate standard error.**

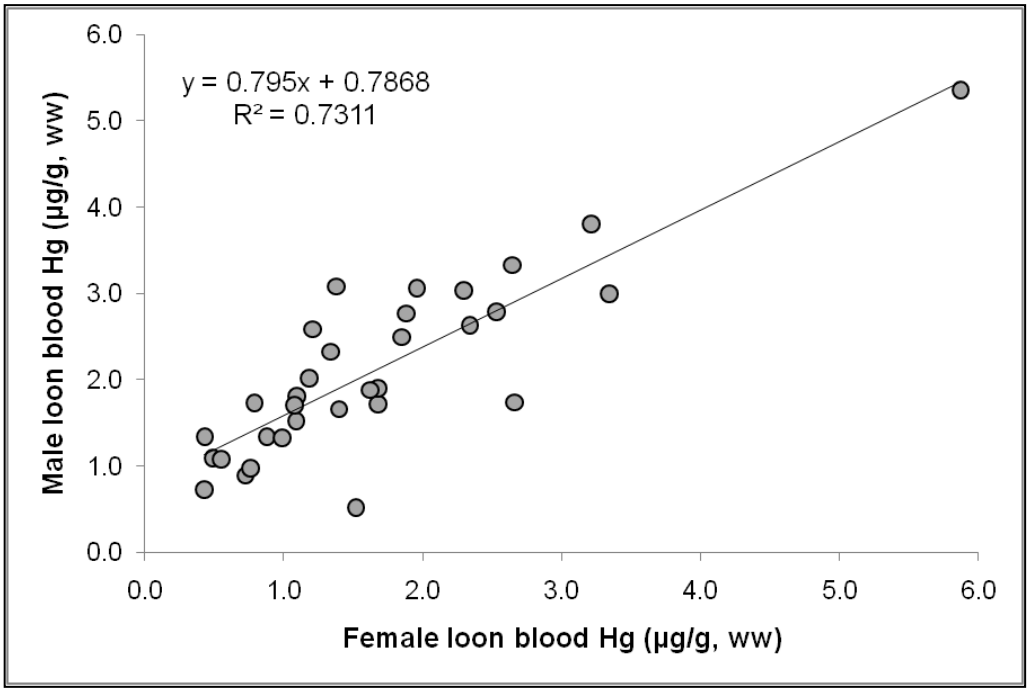


**Figure 17. Correlation between average male and female blood mercury levels at each study lake.**

**Bioconcentration factor for the Adirondack Park**

Mercury concentrations within the food web varied by many orders of magnitude between water and loon samples (Table 7). We found that average mercury concentrations followed the predicted pattern of biomagnification through the food web, with an increase in mercury as it moved from water to prey (zooplankton, crayfish, and insectivorous fish) to upper level predators (piscivorous fish and loons) (Figure 18). As expected, trophic level 4 fish have higher average mercury values than trophic level 3 fish. Average sediment mercury concentration falls between the values for zooplankton and crayfish. We also analyzed water, sediment and zooplankton for methylmercury, and found evidence of methylmercury increasing from abiotic to biotic variables (Figure 19).

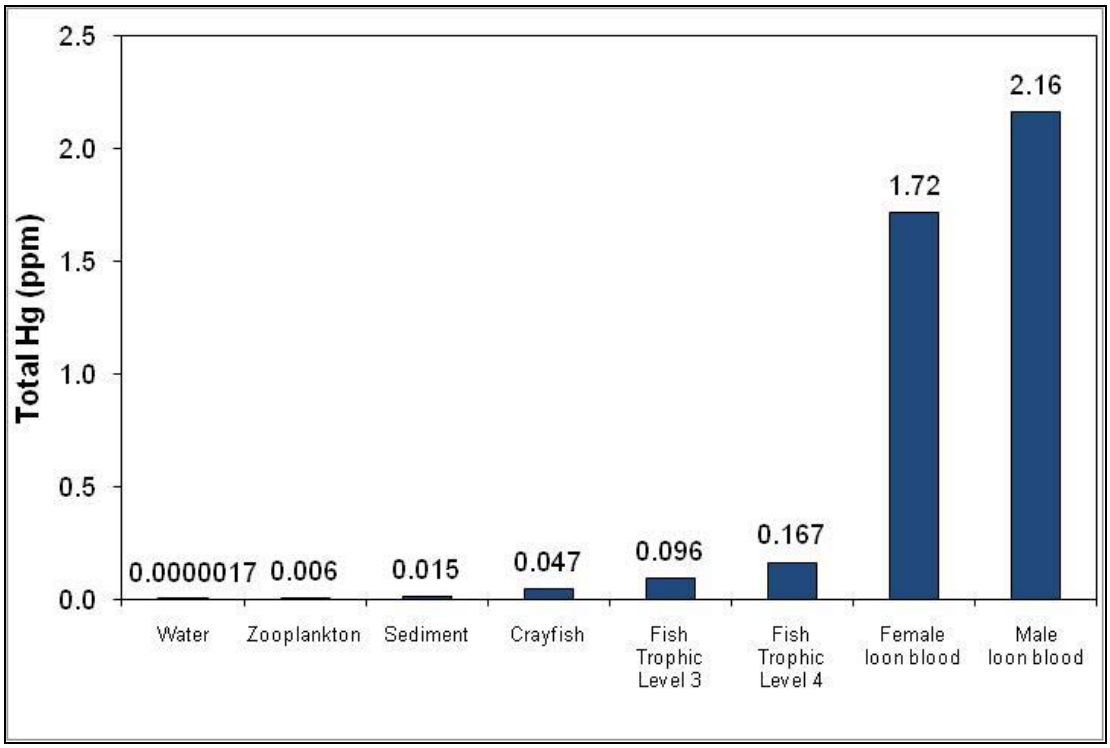


**Figure 18. Average total mercury concentration for each sampling group, used to calculate bioconcentration factor.**

**Table 7. Descriptive statistics for all measured mercury values.**

| Media | Variable | N (# Lakes) | Mean [a] | SD | SE | Min | Max | 1st quartile | Median | 3rd quartile |
|---|---|---|---|---|---|---|---|---|---|---|
| Water | Water Total Hg | 44 | 0.00000172 | 0.00000092 | 0.00000014 | 0.00000010 | 0.00000460 | 0.00000110 | 0.00000150 | 0.00000218 |
| | Water MeHg | 37 | 0.00000012 | 0.00000011 | 0.00000002 | 0.000000002 | 0.00000048 | 0.00000003 | 0.00000008 | 0.00000016 |
| Sediment [b] | Total Hg | 26 | 0.015 | 0.018 | 0.004 | 0.001 | 0.084 | 0.003 | 0.008 | 0.021 |
| | MeHg | 25 | 0.00043 | 0.00093 | 0.00019 | 0.00000 | 0.00450 | 0.00004 | 0.00010 | 0.00025 |
| Zooplankton [b] | Total Hg | 40 | 0.006 | 0.004 | 0.001 | 0.000 | 0.017 | 0.004 | 0.005 | 0.009 |
| | MeHg | 38 | 0.00255 | 0.00493 | 0.00080 | 0.00008 | 0.02725 | 0.00059 | 0.00118 | 0.00221 |
| Crayfish [b] | Wholebody | 26 | 0.047 | 0.020 | 0.004 | 0.021 | 0.094 | 0.028 | 0.045 | 0.059 |
| | Tail | 26 | 0.059 | 0.036 | 0.007 | 0.020 | 0.170 | 0.030 | 0.050 | 0.073 |
| Fish [b] | Small Fish | 42 | 0.086 | 0.066 | 0.010 | 0.030 | 0.430 | 0.058 | 0.070 | 0.090 |
| | Medium Fish | 43 | 0.108 | 0.066 | 0.010 | 0.030 | 0.410 | 0.070 | 0.100 | 0.120 |
| | Large Fish | 41 | 0.145 | 0.083 | 0.013 | 0.040 | 0.460 | 0.095 | 0.120 | 0.160 |
| | Extra Large Fish | 36 | 0.191 | 0.093 | 0.015 | 0.060 | 0.450 | 0.123 | 0.170 | 0.230 |
| | All Fish | 44 | 0.129 | 0.066 | 0.010 | 0.040 | 0.350 | 0.090 | 0.110 | 0.140 |
| Trophic Level Fish [b] | Trophic Level 3 | 44 | 0.096 | 0.055 | 0.008 | 0.040 | 0.290 | 0.063 | 0.080 | 0.110 |
| | Trophic Level 4 | 42 | 0.167 | 0.085 | 0.013 | 0.040 | 0.450 | 0.110 | 0.145 | 0.203 |
| Yellow Perch Equivalent [b] | All | 44 | 0.164 | 0.099 | 0.015 | 0.030 | 0.520 | 0.100 | 0.130 | 0.198 |
| | Small YPE | 33 | 0.104 | 0.067 | 0.012 | 0.030 | 0.330 | 0.060 | 0.080 | 0.125 |
| | Medium YPE | 40 | 0.145 | 0.112 | 0.018 | 0.030 | 0.520 | 0.080 | 0.100 | 0.178 |
| | Large YPE | 38 | 0.174 | 0.103 | 0.017 | 0.050 | 0.460 | 0.110 | 0.140 | 0.225 |
| | Extra Large YPE | 33 | 0.202 | 0.085 | 0.015 | 0.080 | 0.450 | 0.135 | 0.190 | 0.260 |

[a] All Hg values in parts per million (µg/g)

[b] Values reported as wet weight (ww)

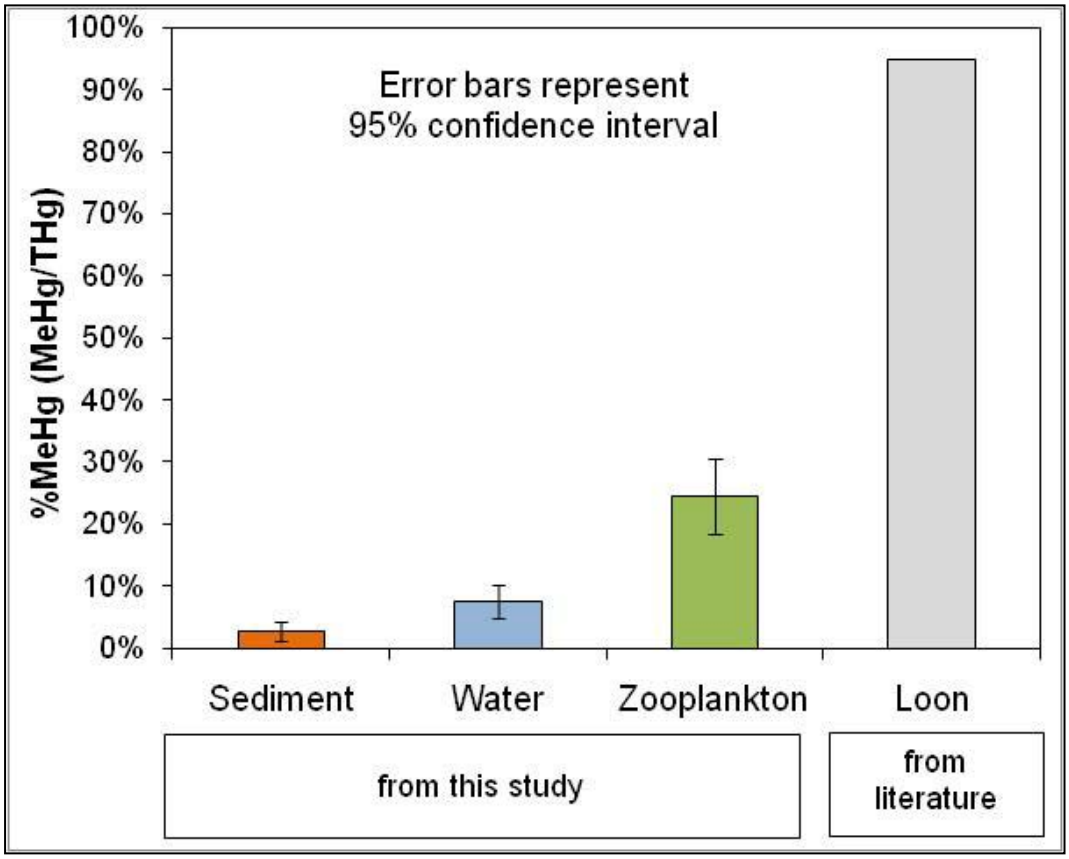


**Figure 19. Percent of total mercury that is made up of methylmercury in sediment, water, zooplankton, and loons. Loon sample comes from literature review (e.g., Wolfe et al. 2007).**

**Relationships between mercury concentrations at different levels of the food web**

We looked at the correlations between mercury levels in different components of the food chain. We highlight the biologically and statistically significant correlations here. Zooplankton appeared to be correlated to both small and medium yellow perch equivalents, but not as strongly correlated to large or extra large yellow perch equivalents (Figure 20). Crayfish, on the other hand, appeared to be correlated to large and extra large fish, but not to small or medium sized fish (Figure 21).

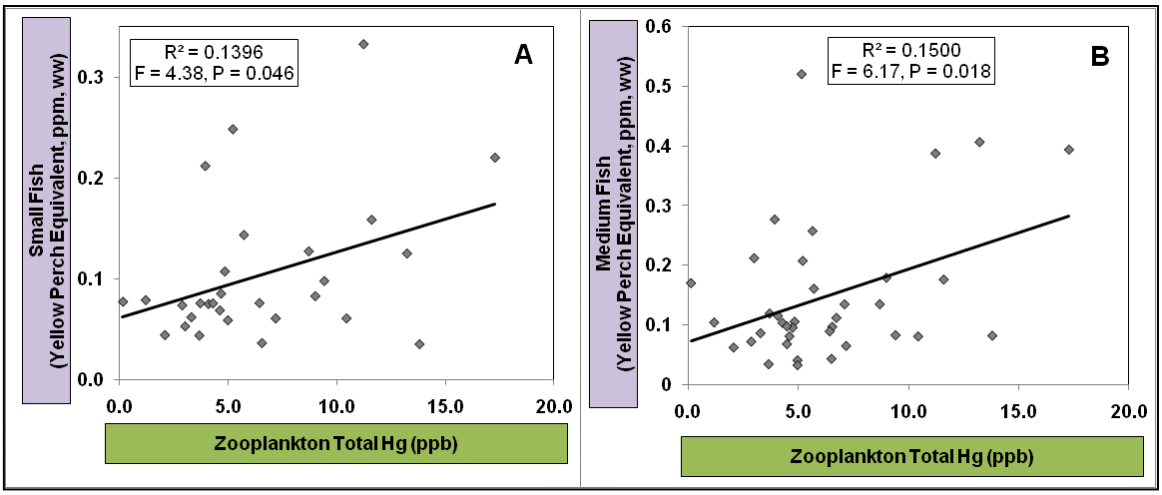


**Figure 20. Correlation between zooplankton total mercury and A) small fish and B) medium fish. All fish are converted to yellow perch equivalent for comparison purposes.**

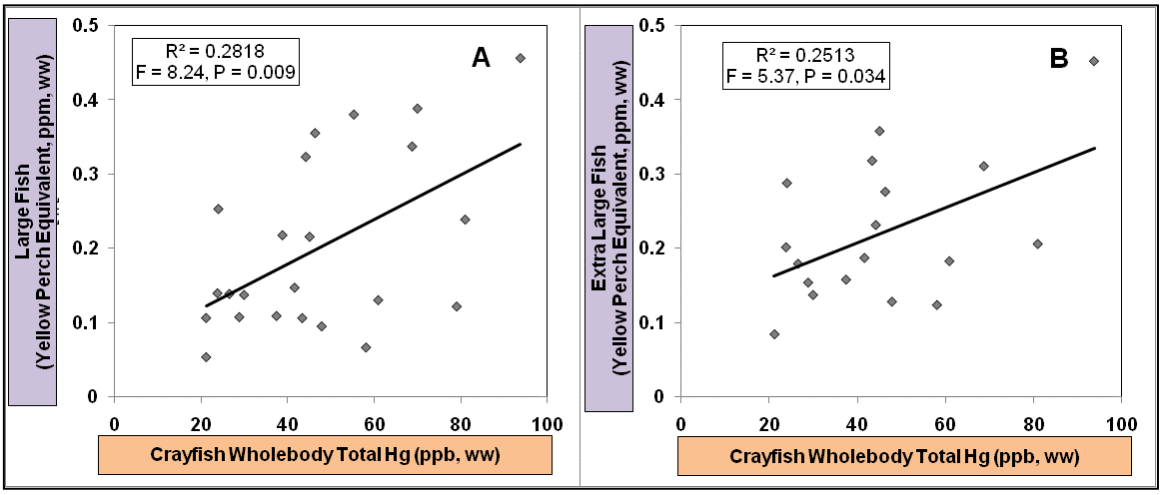


**Figure 21. Correlation between crayfish wholebody total mercury and A) large fish total mercury and B) extra large fish total mercury*.**

* All fish are converted to yellow perch equivalent for comparison purposes.

Interestingly, we did not find correlations between water total or methylmercury and any of the prey items, but we did see a correlation between both total mercury and methylmercury and the two different loon unit measurements (FLU and MLU, Figure 22). Although not their preferred food item, both male and female loon units appeared to be correlated with crayfish mercury levels (Figure 23), probably reflecting the fact that crayfish compose a substantial portion of the diet of juvenile Adirondack loons. Also, both loon units are correlated with large and extra large yellow perch equivalents (Figure 24).

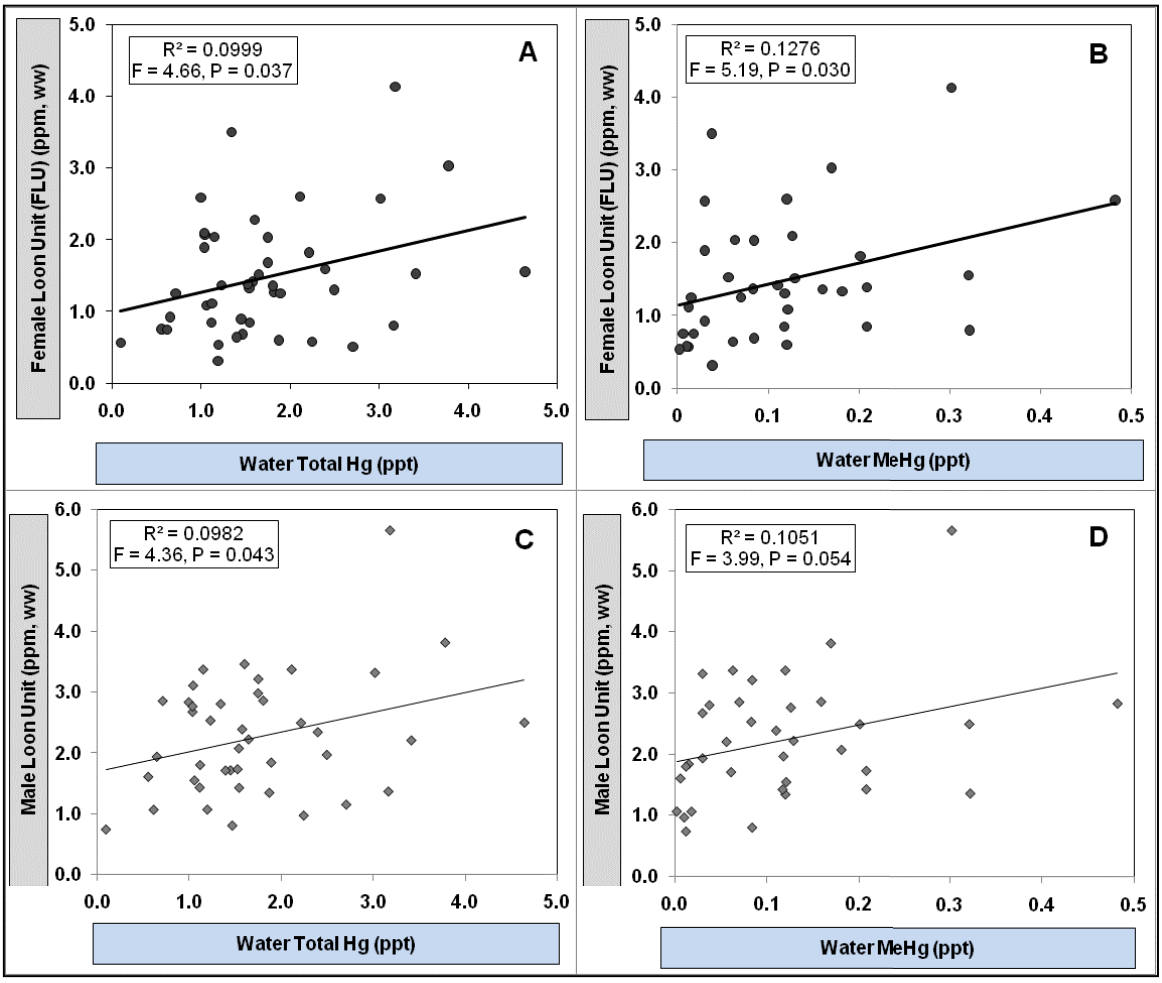


**Figure 22. Correlations between mercury in water samples and mercury in loon units. A) correlation between water total mercury and FLU, B) correlation between water methylmercury and FLU, C) correlation between water total mercury and MLU and D) correlation between water methylmercury and MLU.**

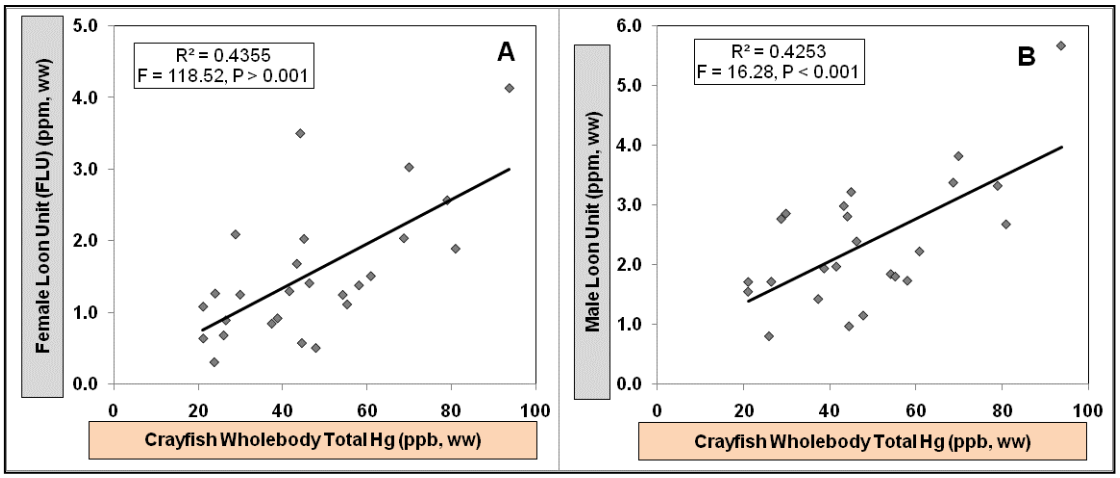


**Figure 23. Correlation between crayfish whole body total mercury and A) female loon unit and B) male loon unit.**

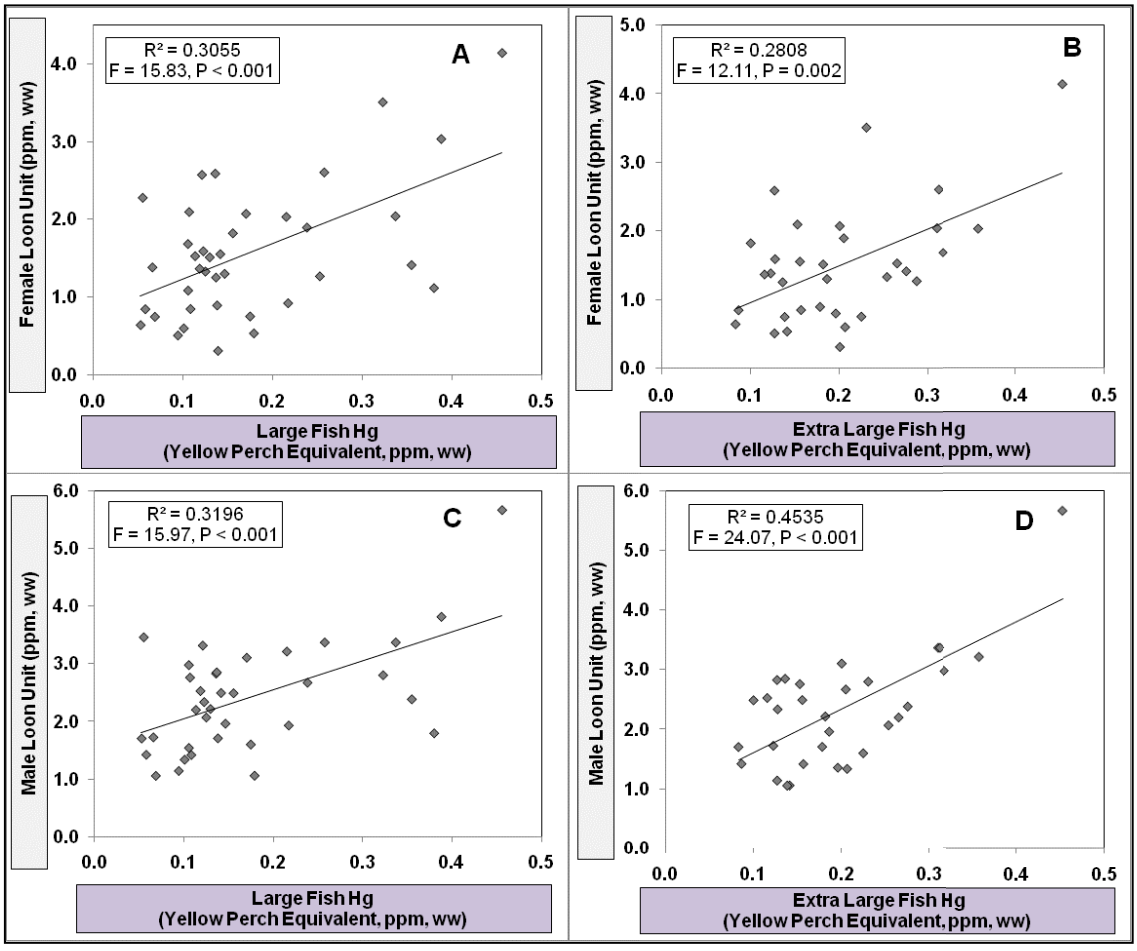


**Figure 24. Correlation between fish total mercury and loon units. A) correlation between large fish mercury and FLU, B) correlation between extra large fish mercury and FLU, C) correlation between large fish mercury and MLU, and D) correlation between extra large fish mercury and MLU.**

**Relationship between lake acidity and mercury**

We found a negative correlation between lake pH and mercury concentrations in both trophic level 3 and 4 fish (Figure 25a,b), and both female and male loon units (Figure 25c,d).

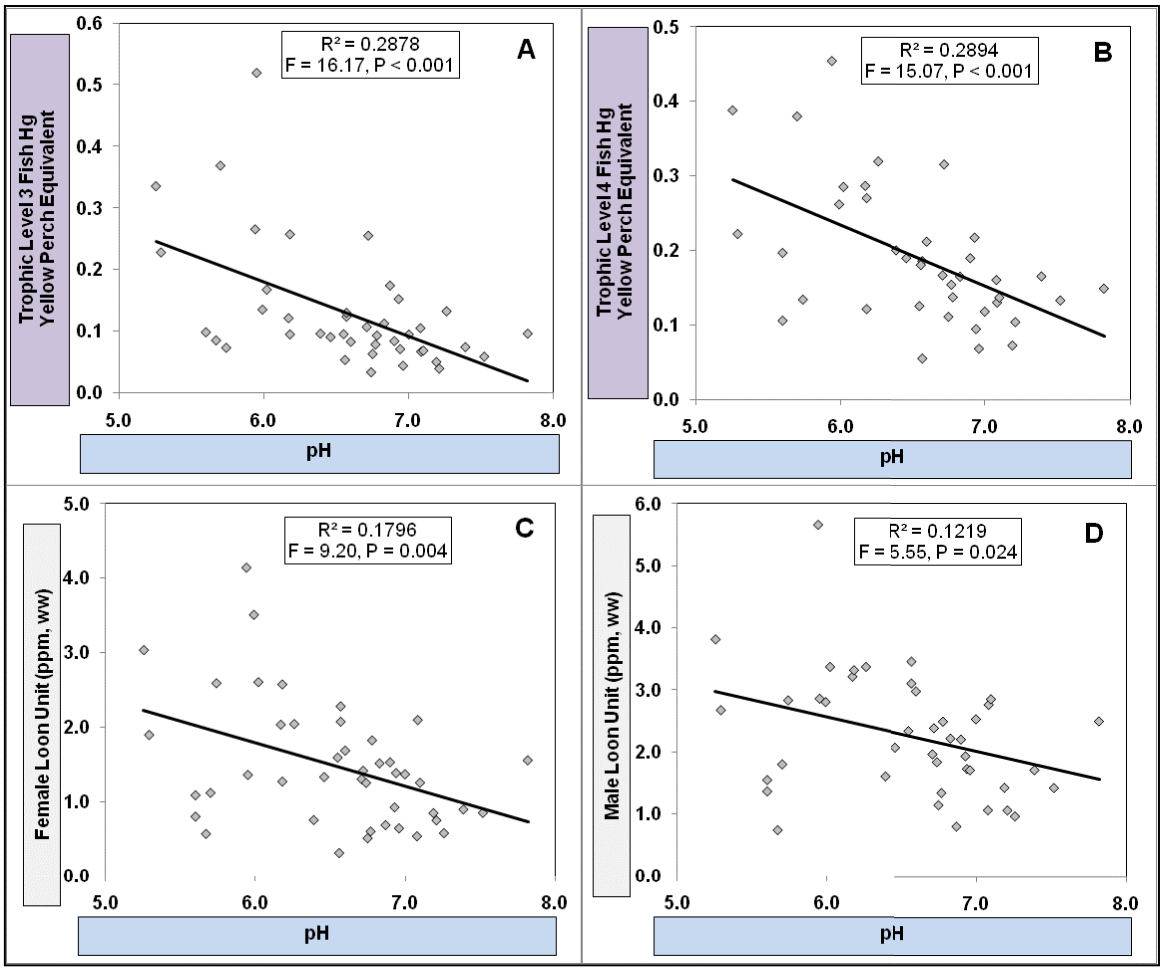


**Figure 25. Correlation between pH and mercury for A) trophic level 3 fish, B) trophic level 4 fish, C) female loon units and D) male loon units.**

Based on previous research conducted by BRI and others, we are able to assemble reasonable guidelines for effects based on mercury risk categories (Table 8). Using these guidelines for blood effects levels, we determined that 21% of the male Adirondack common loons and 8% of the female common loons examined in this study are at high risk from detrimental impacts (i.e., behavioral impairment or decreased reproductive success) due to mercury exposure (Figure 26). Looking at feather mercury loads, 37% of male common loons are at risk of adverse effects, while only 7% of females are at risk (Figure 27). In terms of egg mercury levels, approximately 13% of nonviable Adirondack common loon eggs evaluated in this study exceeded the guidelines for high mercury dose (Figure 28).

**Table 8. Categories for mercury risk assessment of common loon tissue and prey samples (µg/g).**

| Matrix | Type | Low | Moderate | | High | Extra High | Reference Base |
|---|---|---|---|---|---|---|---|
| **Adult Blood** | ww | 0 - 1.0 | **Low-Moderate** | **Moderate-High** | 3.0 - 4.0 | > 4.0 | Evers et al. 2008, Burgess and Meyer 2008 |
| | | | 1.0 - 2.0 | 2.0 - 3.0 | | | |
| **Egg** | ww | 0 - 0.5 | 0.5 - 1.3 | | 1.3 - 2.0 | > 2.0 | Evers et al. 2003 |
| **Feather** | fw | 0 - 9.0 | 9.0 - 20.0 | | 20.0 - 35.0 | > 35.0 | Thompson 1996 |

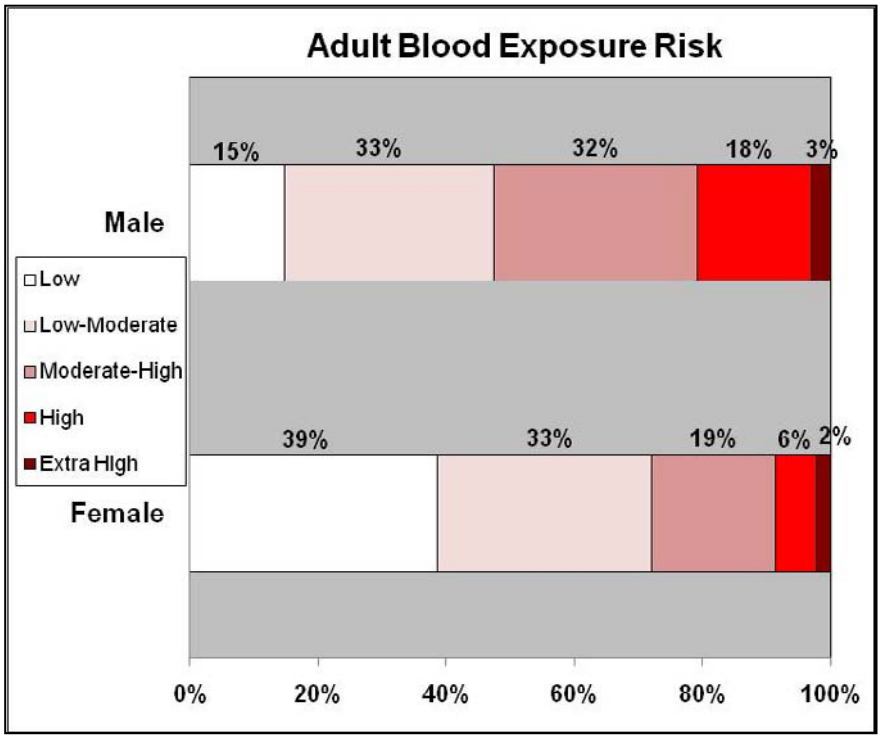


**Figure 26. Risk ratios for mercury exposure based on adult blood mercury exposure groups: low (0–1 µg/g), low-moderate (1–2 µg/g), moderate-high (2–3 µg/g), high (3–4 µg/g) and extra high (>4 µg/g).**

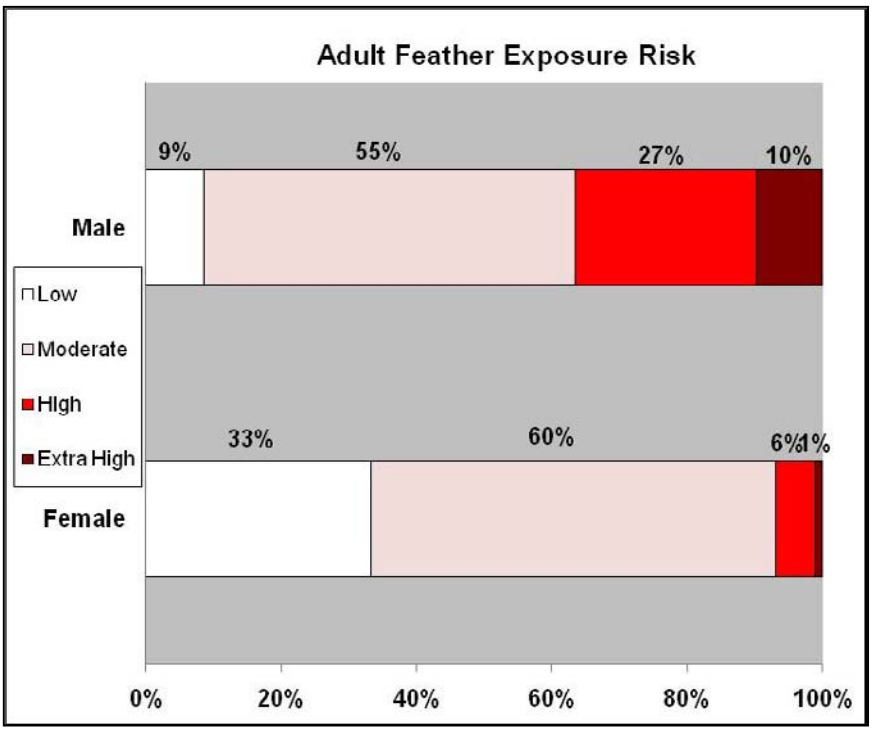


**Figure 27. Risk ratios for mercury exposure based on adult feather mercury exposure groups: low (0–9 µg/g), moderate (9–20 µg/g), high (20–35 µg/g), and extra high (>35 µg/g).**

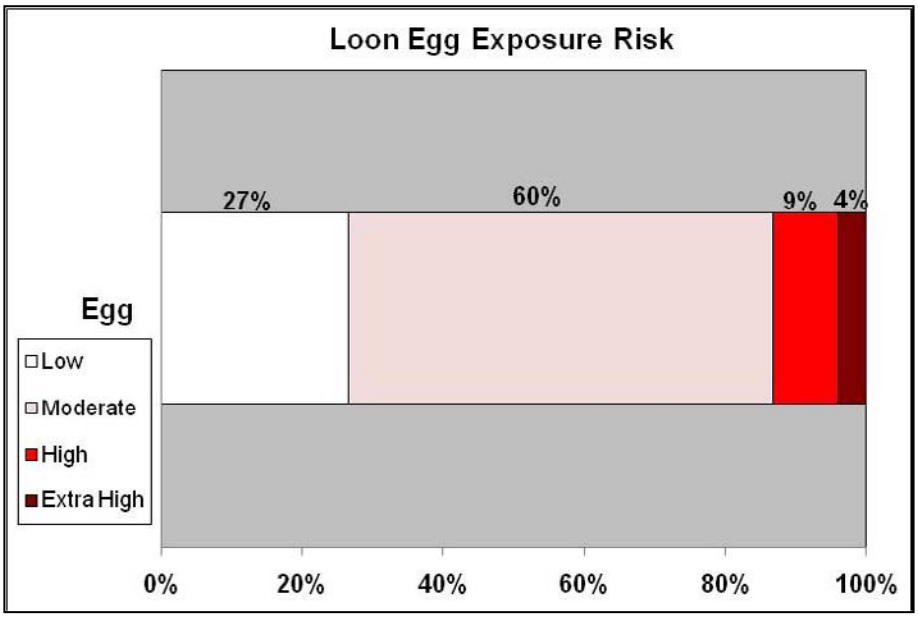


**Figure 28. Risk ratios for mercury exposure in common loon eggs, based on risk groups: low (0–0.5 µg/g), moderate (0.5–1.3 µg/g), high (1.3–2.0 µg/g) and extra high (>2.0 µg/g).**

**Spatial distribution of mercury**

In order to better understand if mercury contamination in loons is concentrated in one part of the Adirondack Park, we grouped the female loon units for each lake into four risk categories, based on previous findings (Evers et al. 2005). Although we did not see any strong trends of loon mercury levels in one part of the Park, it does appear that the southern end of the study area tends to have lakes with higher FLU values (Figure 29). The trend is stronger when we consider MLU values, as there are more male birds within the high risk categories (Figure 30). This corresponds with the increased acid deposition that the lakes in the southwestern part of the Adirondacks receive due to increased lake effect snow load, and the rainshadow effect from the mountains in the northeastern part of the Park.

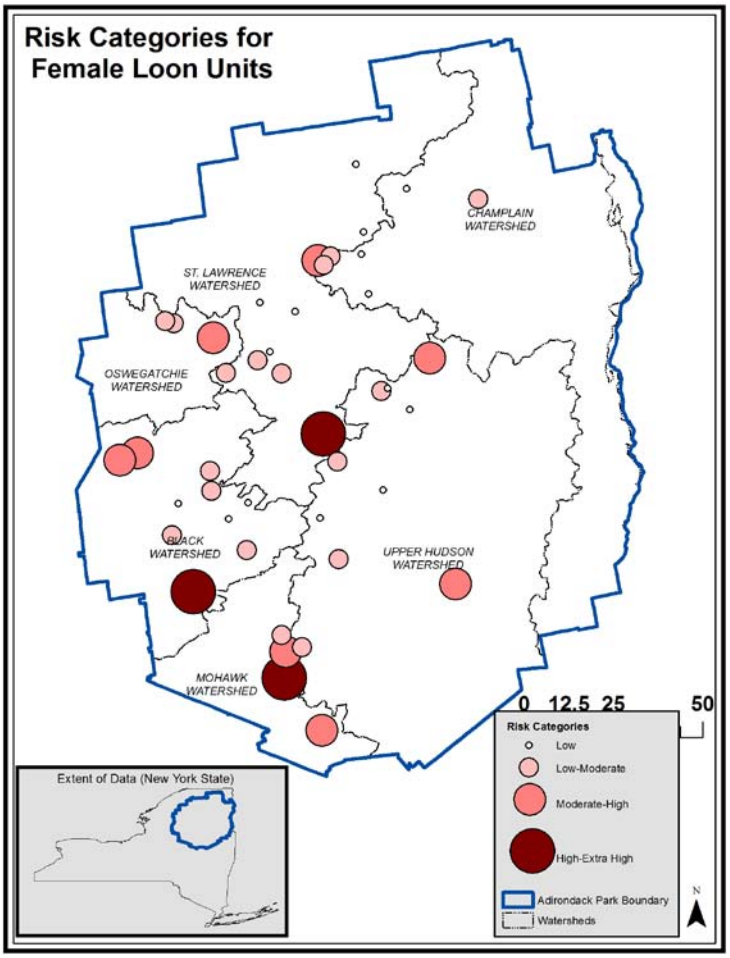


**Figure 29. Spatial distribution of lakes with low, moderate, high and extra high female loon units. Low (0–1 µg/g), low-moderate (1–2 µg/g), moderate-high (2–3 µg/g), and high-extra high (3+ µg/g).**

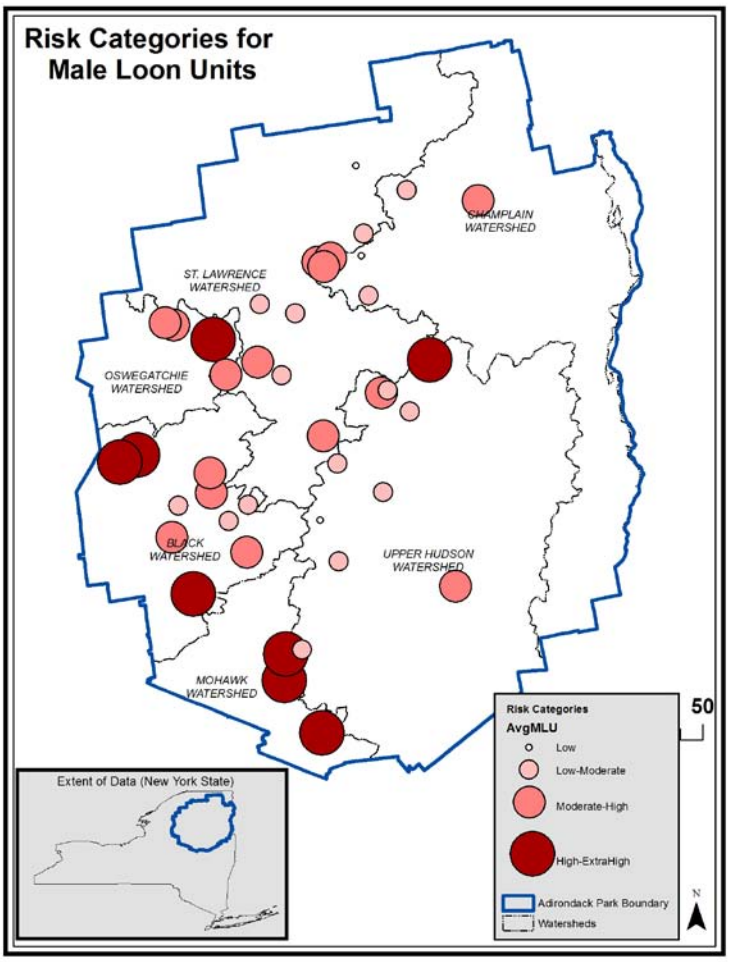


**Figure 30. Spatial distribution of lakes with low, moderate, high and extra high male loon units. Low (0–1 µg/g), low-moderate (1–2 µg/g), moderate-high (2–3 µg/g), and high-extra high (3+ µg/g).**

### Geographical Context

After querying the BRI common loon database, we found 2040 records of adult common blood mercury concentrations, between 1988 and 2011. The average blood mercury concentration for all loons sampled across North America was 1.876 ppm (ww, SD = 1.369). When means are compared between U.S. states and Canadian provinces, New York common loon mean mercury concentrations (Mean = 1.954 ppm, Figure 31) falls between the lowest mercury concentrations, found in Alaska (Mean = 0.642 ppm) and the highest mercury concentrations, found in Nova Scotia (Mean = 4.827 ppm). Mercury concentrations in loons generally increase from west to east across North America, with maximum values representing the highest detected risk to loons in each region (Figure 32; Evers et al. 1998). As in New York, female common loons generally have lower mercury body burdens than males (Table 9).

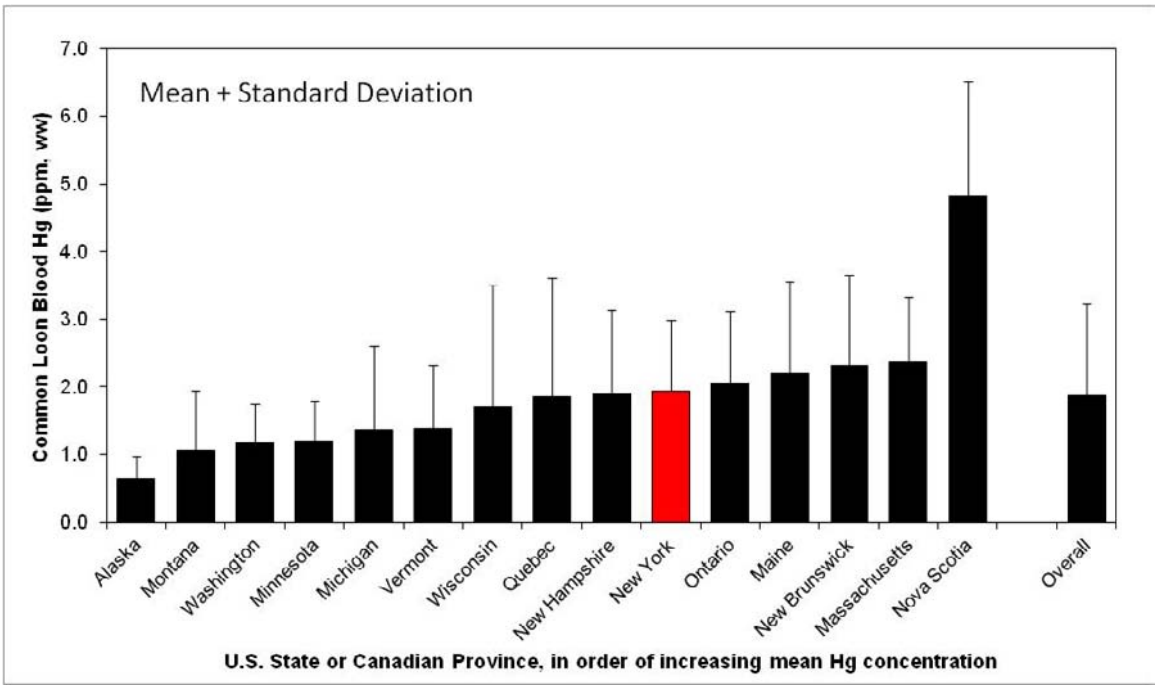


**Figure 31. Mean blood mercury concentration for breeding common loons in each U.S. state or Canadian province, in order of increasing mercury.**

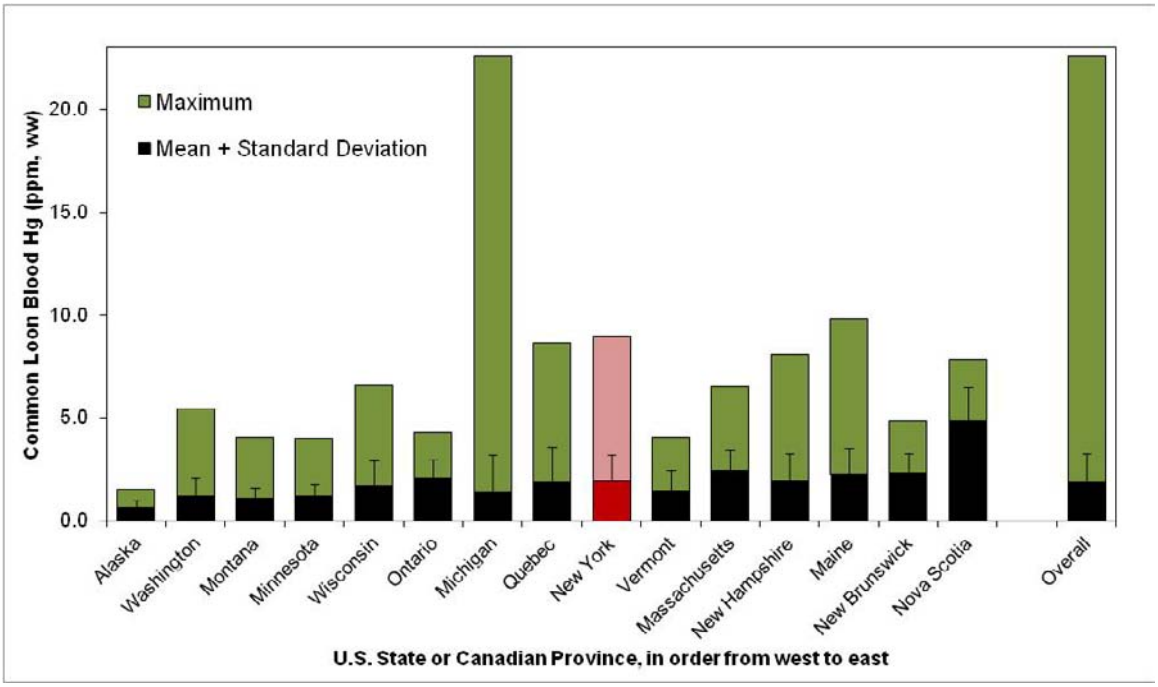


**Figure 32. Mean and maximum common loon blood mercury concentrations in each geographic region, in order from west to east.**

**Table 9. Common loon blood mercury concentrations for each state/province by sex.**

| State/Province | Female N | Female Mean Blood Hg | Female SD | Female Max Blood Hg | Male N | Male Mean Blood Hg | Male SD | Male Max Blood Hg |
|---|---|---|---|---|---|---|---|---|
| **Average Canada** | **52** | **2.207** | **1.812** | **7.440** | **67** | **2.618** | **1.815** | **8.630** |
| New Brunswick | 12 | 2.226 | 0.583 | 3.240 | 11 | 2.413 | 1.247 | 4.840 |
| Nova Scotia | 9 | 4.697 | 1.678 | 6.720 | 8 | 5.283 | 1.821 | 7.800 |
| Ontario | 8 | 1.548 | 0.529 | 2.570 | 14 | 2.351 | 1.006 | 4.260 |
| Quebec | 23 | 1.452 | 1.744 | 7.440 | 34 | 2.167 | 1.743 | 8.630 |
| **Average U.S.** | **905** | **1.512** | **1.040** | **8.980** | **975** | **2.146** | **1.479** | **22.600** |
| Alaska | 25 | 0.520 | 0.276 | 1.300 | 24 | 0.815 | 0.339 | 1.540 |
| Massachusetts | 31 | 2.084 | 0.885 | 4.380 | 34 | 2.560 | 0.964 | 5.770 |
| Maine | 355 | 1.819 | 0.987 | 6.510 | 367 | 2.589 | 1.513 | 9.790 |
| Michigan | 74 | 1.070 | 0.709 | 4.000 | 88 | 1.632 | 2.333 | 22.600 |
| Minnesota | 97 | 0.932 | 0.374 | 2.500 | 113 | 1.436 | 0.620 | 4.000 |
| Montana | 28 | 1.009 | 0.674 | 4.080 | 22 | 1.161 | 0.384 | 2.300 |
| New Hampshire | 130 | 1.521 | 1.229 | 6.890 | 154 | 2.253 | 1.346 | 8.100 |
| New York | 112 | 1.666 | 1.256 | 8.980 | 119 | 2.234 | 1.150 | 6.556 |
| Vermont | 13 | 0.928 | 0.810 | 2.900 | 13 | 1.499 | 0.803 | 2.940 |
| Washington | 17 | 0.902 | 0.232 | 1.500 | 13 | 1.565 | 1.243 | 5.430 |
| Wisconsin | 23 | 1.545 | 1.191 | 4.600 | 28 | 1.861 | 1.278 | 6.600 |
| **Total** | **957** | **1.550** | **1.106** | **8.980** | **1042** | **2.177** | **1.506** | **22.600** |

## EFFECT OF MERCURY ON THE ADIRONDACK COMMON LOON POPULATION

**Effect of mercury and lake acidity on loon reproductive success.**

Between 1999 and 2007, we made 564 re-observations of banded loons on 82 Adirondack lakes for a total of 541 loon "territory-years" (defined as a pair of loons that were observed on a territory for a breeding season) to assess the reproductive success of our study birds. The number of territorial and nesting pairs, chicks hatched, chicks fledged, and the number of chicks fledged per territorial pair (CF/TP) observed annually are summarized in Table 10. The annual productivity across all study lakes was 0.594 (SE = 0.046) CF/TP per year. We also grouped the study birds by their mercury exposure level, and found that the productivity of the low-moderate mercury loons was 0.673 CF/TP, while for high/extra-high mercury loons it was 0.483 CF/TP. The low-moderate mercury loons hatched more chicks (1.16 CH/NP) than did the high/extra-high mercury birds (0.866 CH/NP; Table 11, Figure 33).

To examine the interactions of mercury and lake acidity on reproductive success, we included only territories that were monitored every year for three or more years, resulting in 304 territory-years of data on common loon productivity across 53 territories in the Adirondack Park from 1999 to 2007 (Appendix G). We found a significant relationship between female loon units and annual productivity for territories with three or more years of productivity observations, using a simple least squares regression model ($\beta = -0.128$, $R^2 = 0.075$, $F = 4.15$, $p =$

0.047). We also found a significant relationship between annual productivity and FLU at the 80$^{th}$ quantile ($\beta$ = -0.141, t = -2.64, p = 0.01). At the 90$^{th}$ quantile, the productivity decreased more rapidly ($\beta$ is larger), but the trend is not significant ($\beta$ = -0.185, t = -0.99, p = 0.32, Figure 34). We also used FLU equivalent values to compare annual productivity between three different mercury groups (Figure 35): low risk (0-1 µg/g; mean Hg = 0.680 µg/g: avg CF/TP/year = 0.697, SE = 0.076, n = 19); low-moderate risk (1-2 µg/g; mean Hg = 1.351 µg/g: avg CF/TP/year = 0.563, SE = 0.064, n = 24); and moderate-high risk (2+ µg/g; mean Hg =2.505 µg/g: avg CF/TP/year = 0.474, SE = 0.120, n = 10). Using a non-parametric Kruskal-Wallis test, we found a marginally significant difference between all the mercury groups ($\chi^2$ = 5.136, df = 2, p = 0.077), and a significant difference when comparing low to moderate-high groups (p = 0.036). The trends indicate a 19% reduction in productivity between low and moderate groups and a 32% reduction in productivity between low and moderate-high groups (Figure 35b).

Although FLU and MLU are highly correlated, we also wanted to explore the relationship between MLU and productivity. We found a significant relationship using a simple least squares regression ($\beta$ = -0.099, $R^2$ = 0.079, F = 4.30, p = 0.04) and a marginally significant relationship at the 80$^{th}$ quantile ($\beta$ = -0.113, t = -1.91, p = 0.06). Due to low sample size at the upper quantiles, we did not find a significant relationship at the 90$^{th}$ quantile, but the slope increases dramatically above the standard least squares regression, indicating that productivity is more steeply regulated by mercury at the upper end of the distribution ($\beta$ = -0.136, t = -1.04, p = 0.30, Figure 36). We also broke the productivity data into 4 groups based on MLU values: low risk (0–1 µg/g, mean Hg = 0.771 µg/g: avg CF/TP/year = 0.467, SE = 0.167, n = 5); low-moderate risk (1–2 µg/g, mean Hg = 1.551 µg/g: avg CF/TP/year = 0.736, SE = 0.056, n = 22); moderate-high risk (2–3 µg/g, mean Hg = 2.493 µg/g: avg CF/TP/year = 0.572, SE = 0.091, n = 18); and high-very high risk (> 3 µg/g, mean Hg = 3.805 µg/g: avg CF/TP/year = 0.320, SE = 0.078, n = 7). We detected a significant difference between groups using a Kruskal-Wallis test ($\chi^2$ = 10.897, df = 3, p = 0.0012), and found that there was a significant difference between moderate and very high mercury groups (p < 0.05). There was a 56% difference in annual productivity between these groups (Figure 37).

Because we are concerned about the effects of acid deposition to Adirondack ecosystems, we also looked at the relationship between lake pH and common loon productivity. We found a marginally significant positive relationship between lake pH and annual productivity using a simple least squares regression ($\beta$ = 0.185, $R^2$ = 0.082, F = 2.959, p = 0.095), which showed a stronger and steeper trend at the 90$^{th}$ quantile ($\beta$ = 0.185, t = 2.348, p = 0.025, Figure 38). We grouped lakes into two groups based on pH: high acidity (pH < 6.3; mean pH = 5.86: avg CF/TP/year = 0.570, SE = 0.07, n = 6) and low acidity (pH > 6.3; mean pH = 6.90: avg CF/TP/year = 0.660, SE = 0.07, n = 29), and did not find a significant difference in productivity between the two groups ($\chi^2$ = 0.734, df = 1, p= 0.392; Figure 39).

**Table 10. Number of territorial pairs and reproductive success of common loons in the Adirondack Park, NY, study area, 1999-2007.**

| Year | # Territories Monitored | #TP | #NP | #Chick Hatch | #Chick Fledge | CF/TP |
|---|---|---|---|---|---|---|
| **1999** | 15 | 6 | 3 | 4 | 2 | 0.33 |
| **2000** | 31 | 19 | 14 | 19 | 17 | 0.89 |
| **2001** | 42 | 32 | 26 | 25 | 16 | 0.50 |
| **2002** | 51 | 38 | 28 | 31 | 23 | 0.61 |
| **2003** | 71 | 53 | 44 | 48 | 35 | 0.66 |
| **2004** | 72 | 55 | 47 | 49 | 38 | 0.69 |
| **2005** | 79 | 53 | 43 | 39 | 25 | 0.47 |
| **2006** | 85 | 53 | 40 | 31 | 20 | 0.38 |
| **2007** | 95 | 47 | 40 | 49 | 35 | 0.74 |

**Table 11. Loon productivity parameters by mercury exposure for all territorial pairs observed from 1999-2007.**

| Productivity by Mercury Exposure | #TP | #NP | Nesting (NP/TP) | #Chick Hatch | #Chick Fledge | Chick Survival (CF/CH) | CH/TP | Overall Productivity (CF/TP) | Hatch (CH/NP) | CF/NP |
|---|---|---|---|---|---|---|---|---|---|---|
| **Low-Mod Hg Loons** | 211 | 168 | 0.80 | 195 | 142 | **0.728** | 0.924 | **0.673** | 1.161 | **0.845** |
| **High-Very High Hg Loons** | 149 | 119 | 0.80 | 103 | 72 | **0.699** | 0.691 | **0.483** | 0.866 | **0.605** |
| **All Loons Combined** | 360 | 287 | 0.80 | 298 | 214 | 0.718 | 0.828 | 0.594 | 1.038 | 0.746 |

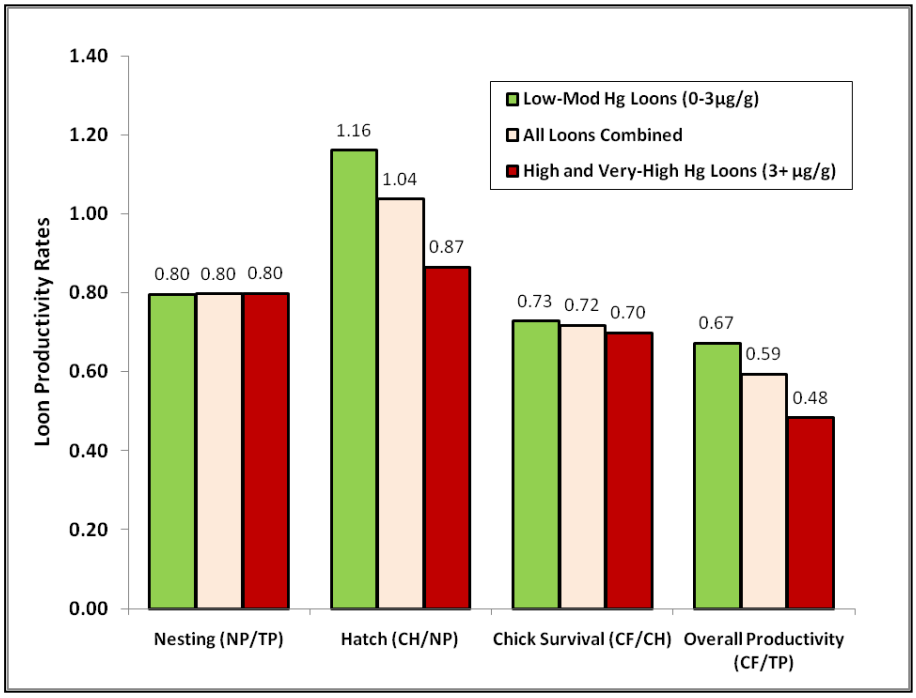


**Figure 33. Summary of change in reproductive measures between low-moderate (<3.0 µg/g blood Hg) and high/very-high (3.0+ µg/g blood Hg) risk categories for Adirondack study loons, and all loons combined, 1999-2007.**

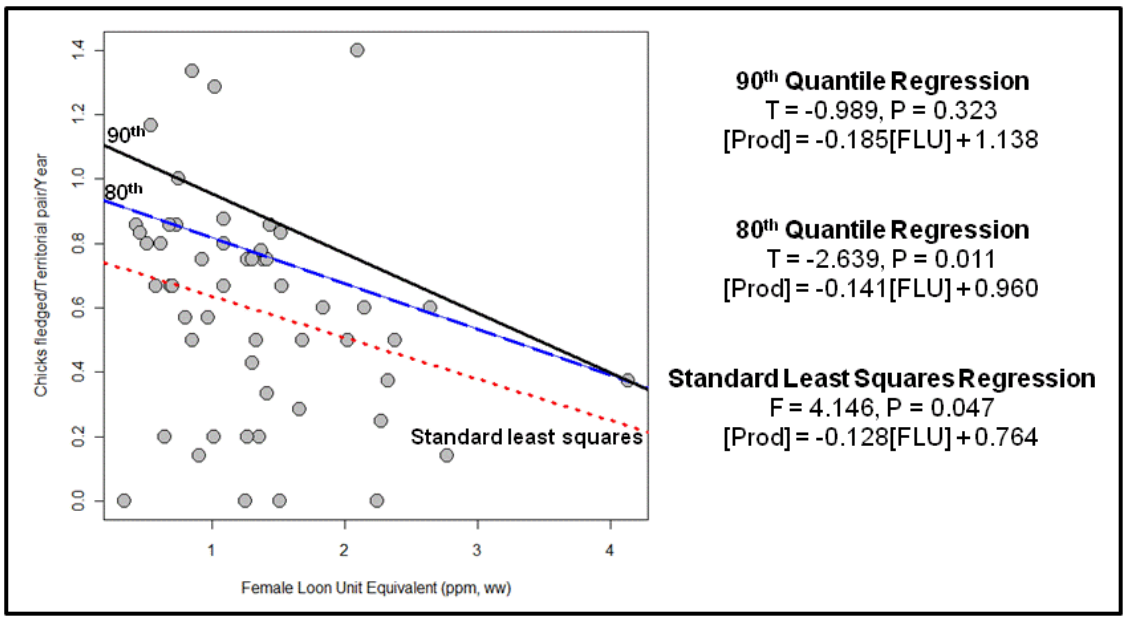


**Figure 34. Relationship between female loon unit and productivity (chicks fledged per territorial pair per year).***

* Lines indicate the standard least squares regression, 80th quantile regression, and 90th quantile regression.

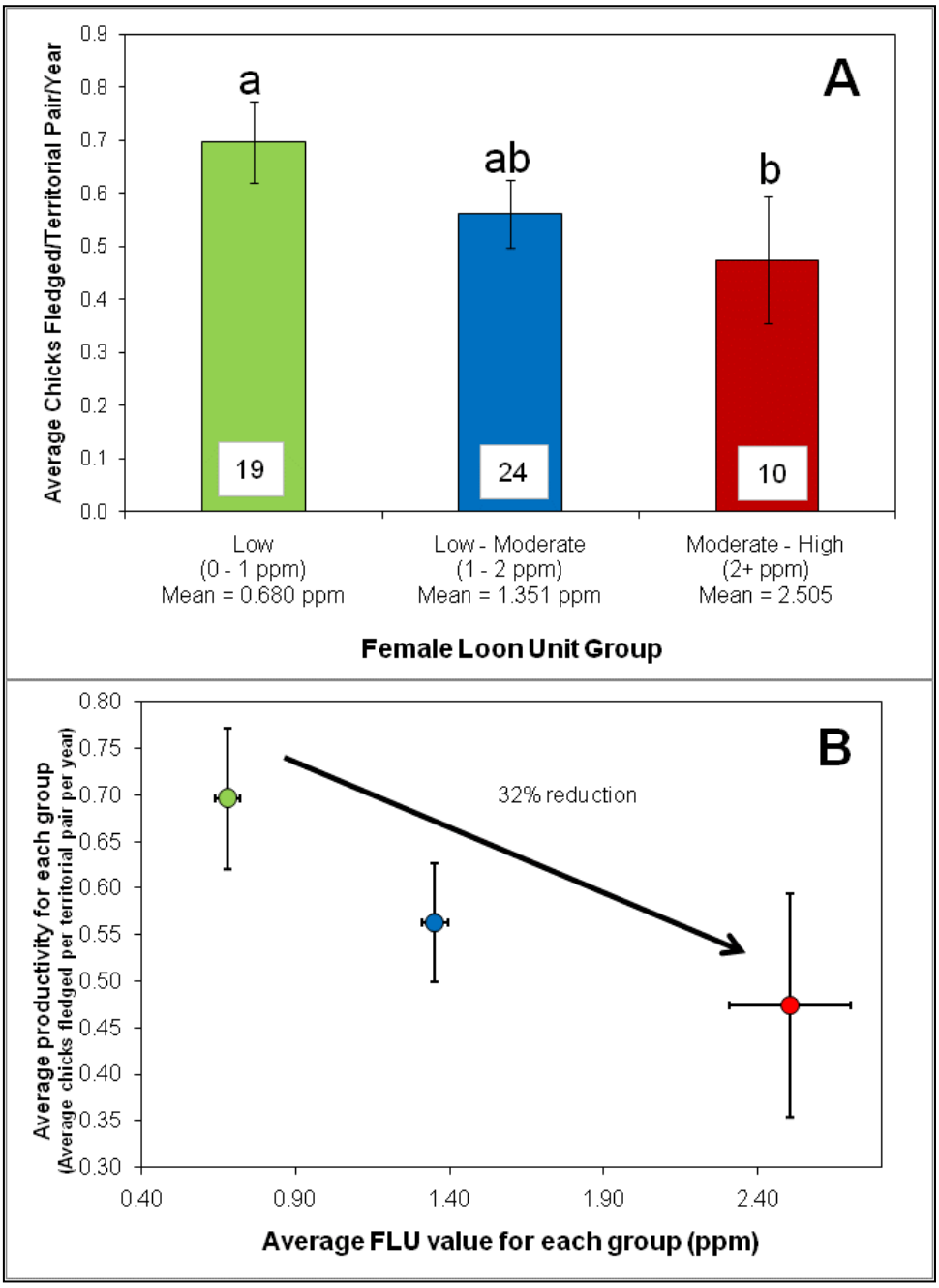


**Figure 35. Comparison of annual productivity by female loon unit groups for A) three mercury risk groups and B) based on average mercury value within each group*.**

* Numbers within bars indicate number of territories where productivity and female loon unit were both measured, letters indicate marginally significant differences between groups (Kruskal-Wallis $\chi^2 = 5.136$, df = 2, p = 0.077) and error bars indicate standard error.

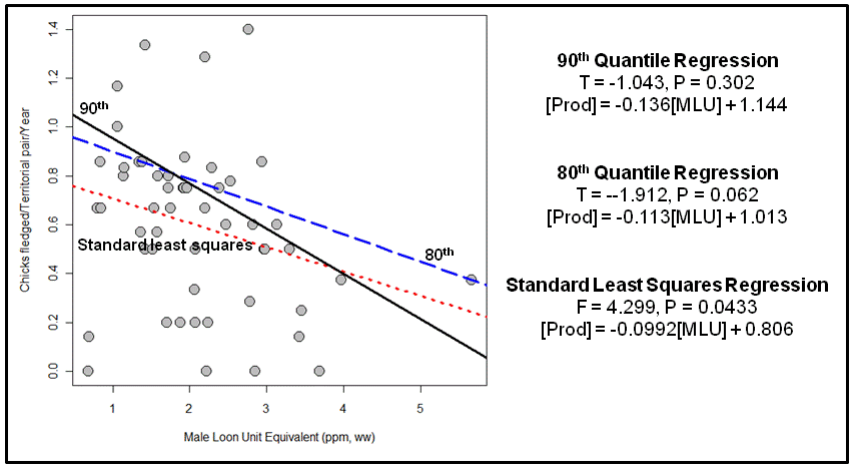


**Figure 36. Relationship between male loon unit and productivity (chicks fledged per territorial pair per year)*.**

* Lines indicate the standard least squares regression, 80th quantile regression, and 90th quantile regression.

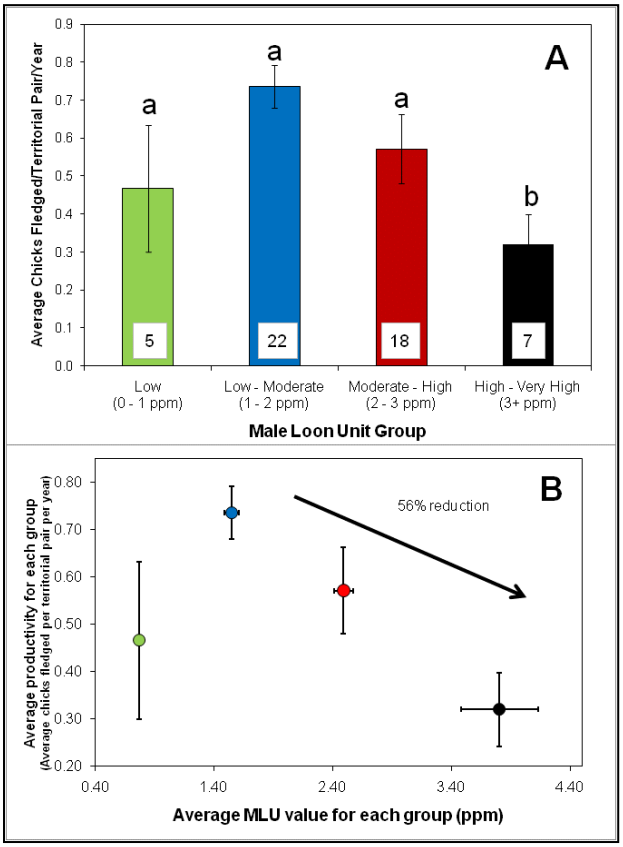


**Figure 37. Comparison of annual productivity by male loon unit groups for A) four mercury risk groups and B) based on average mercury value within each group*.**

* Numbers within bars indicate number of territories where productivity and male loon unit were measured, letters indicate significant differences between groups (Kruskal-Wallis $\chi 2 = 10.897$, df = 3, p = 0.0012) and error bars indicate standard error.

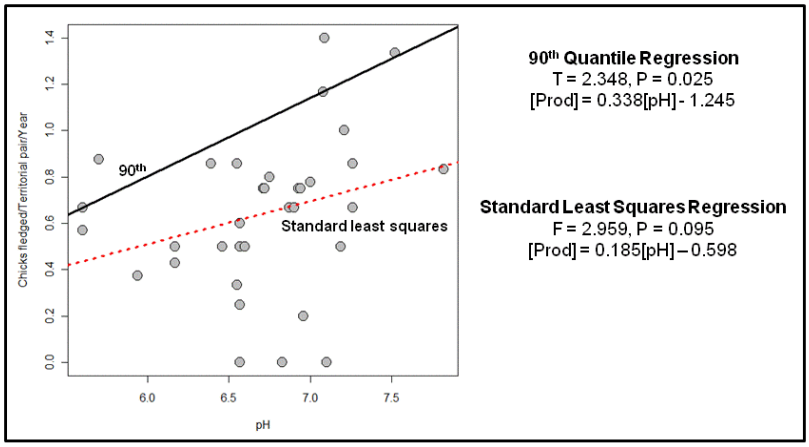


**Figure 38. Relationship between pH and productivity (chicks fledged per territorial pair per year)*.**

* Lines indicate the standard least squares regression and 90th quantile regression.

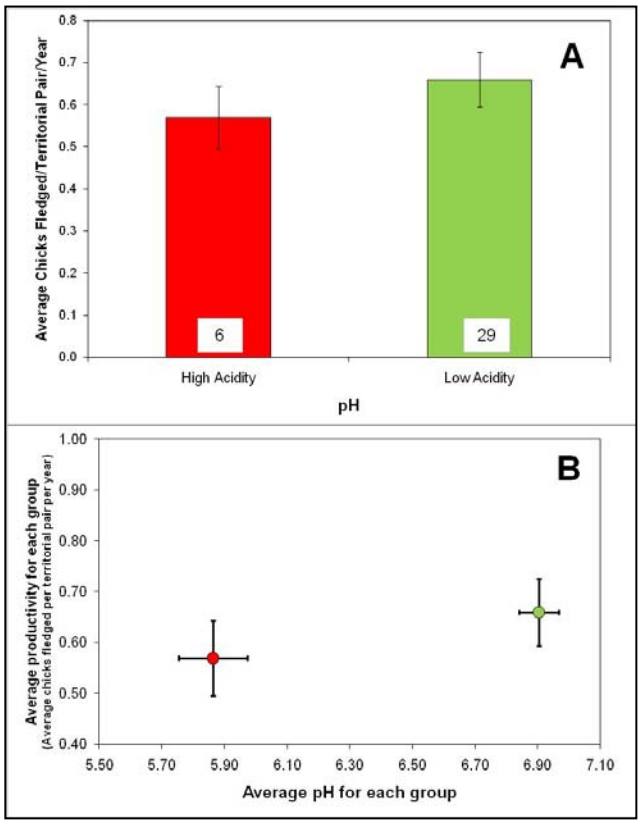


**Figure 39. Comparison of annual productivity by pH groups for A) two lake acidity risk groups and B) based on average pH within each group*.**

* Numbers within bars indicate number of territories where pH and productivity data were collected, there was no significant difference between groups ($\chi 2 = 0.734$, $p = 0.392$), and error bars indicate standard error.

**Model for long-term effect of mercury on the Adirondack loon population**

To assess if mercury body burden was affecting the population growth of Adirondack loons, we looked at three categories of productivity (average chicks fledged per territorial pair) for the study birds from 1999-2007 to incorporate into Grear et al.'s (2009) loon population model (Figure 40). A value of lambda greater than 1 generally predicts that current vital rates (i.e., birth and survival rates) are sufficient to support a stable or growing population, but it is important to note the inherent error associated with models of this type. These projections are meant as estimates of overall population growth across many years; high variability within the population could cause yearly population growth to range below 1.

1. **The overall Adirondack loon study population.** The CF/TP of this group was 0.594 (n=360 TP, with 332 nest attempts; n=558 productivity years), resulting in $\lambda_{Adk\text{-}overall} = 1.0157$.

2. **Low-moderate mercury body burden (blood Hg level <3.0 µg/g).** The CF/TP of this group was 0.673 (n=211 TP, with 192 nest attempts; n=338 productivity years), resulting in $\lambda_{Adk\text{-}Low\text{-}ModHg} = 1.0260$.

3. **High or extra-high mercury body burden (blood Hg level >/= 3.0 µg/g).** The CF/TP of this group was 0.483 (n=149 TP, with 140 nest attempts; n=220 productivity years), resulting in $\lambda_{Adk\text{-}High/X\text{-}HighHg} = 1.0005$.

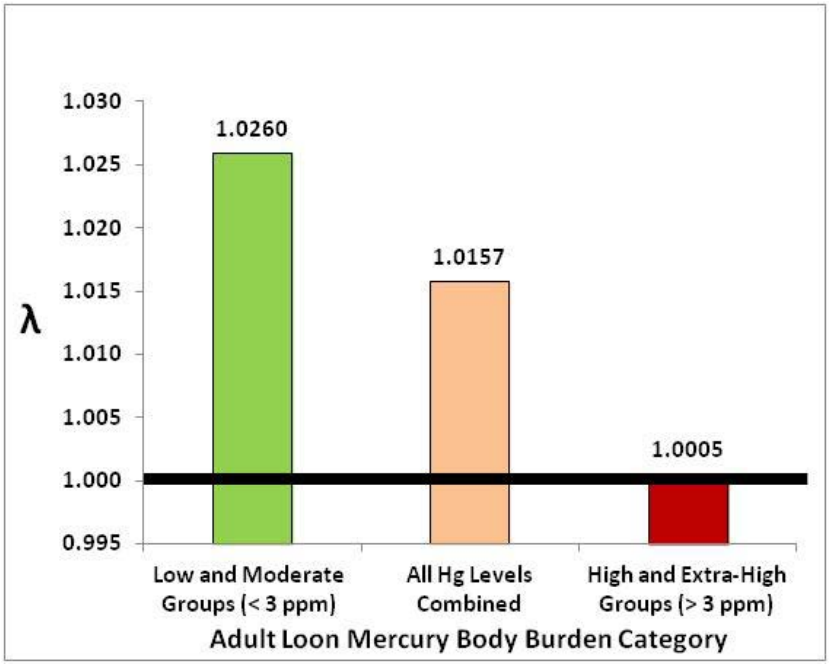


**Figure 40. Adirondack adult loon population growth rate by mercury body burden category, based on Grear et al. (2009) loon population model. Black line shows lambda = 1.0, or no change in population size.**

**Hypothetical Population Model Projections**

To assess how the resulting population growth rates would affect the Adirondack loon population over time, a projection of the adult loon population over 50 years was conducted. It is important to note that these projections do not include effects of competition (density dependence) for limited breeding habitat or other potential limits to loon populations. For example, some of our models predict a loon population of over 3,000 birds after 50 years, but it is unlikely that enough habitat exists in the Adirondack Park to support a population of this size. We have not set out to measure the carrying capacity of the area, however, so we decided to include these density-independent projections as representations of what differences in lambda could mean for the population. A population living in a natural environment can experience yearly disruptions, as well as catastrophic events that limit population growth independent of mercury contamination, and these projections give us an estimate of how well the population would be able to recover from these setbacks. Similarly, we can use these projections to estimate how mercury exposure is likely to limit the growth of loon populations if other limitations or stressors (e.g., disease, human disturbance, and predation) could be removed through restoration or conservation.

Parker et al. (1986) estimated the adult Adirondack loon population at 800-1,000 birds from surveys conducted in the Adirondacks in the mid-1980s. Based on an approximate starting point of 1,000 adult loons, we can project what these growth rates calculated from the loon population model hypothetically mean for the Adirondack population. Our mercury risk models in the previous section indicate that not all loons in the Adirondack Park are exposed to mercury at levels high enough to cause reproductive impairment. We report mercury risk ratios based on the comparison of loon blood, feather and egg samples to current accepted risk categories, and determined that between 8% (based on female blood) and 37% (based on adult feathers) of the population is likely to fall within the high or extra high risk categories. This range of estimates for the overall impact of mercury within the loon population necessitates that we explore different scenarios for population growth, based on varying proportions of the population being under risk (Table 12, Figure 41).

The first scenario (i.e., S.1, Hypothetical No Hg Risk) projects the population growth for a hypothetical population with no mercury risk (all loons are within low and moderate groups). The second scenario (i.e., S.2, Current Low Hg Risk) uses the mercury risk ratios developed for female loon blood in the Adirondack Park, which shows that approximately 8% of the population is within the high and extra high risk groups (refer to Figure 26 from the previous section). The third scenario (i.e., S.3, Current High Hg Risk) uses the mercury risk ratios developed for adult loon feather concentrations, which shows that approximately 37% of the population is at high or extra high risk. The fourth scenario (i.e., S.4, Hypothetical Complete Hg Risk) estimates population growth under a worst-case-scenario, where all the loons are in the high and extra high risk groups. For each scenario, we start with a hypothetical population of 1,000 birds, and model the effect of mercury when different percentages of the population are at risk (Figure 42).

**Table 12. Different scenarios for Adirondack Park loon population growth, based on varying proportions of the population being under risk.**

| Scenario | % Population in Low-Moderate Group | % Population in High-Extra High Group |
|---|---|---|
| S.1 Hypothetical No Hg Risk | 100 | 0 |
| S.2 Current Low Hg Risk | 92 | 8 |
| S.3 Current High Hg Risk | 63 | 37 |
| S.4 Hypothetical Complete Hg Risk | 0 | 100 |

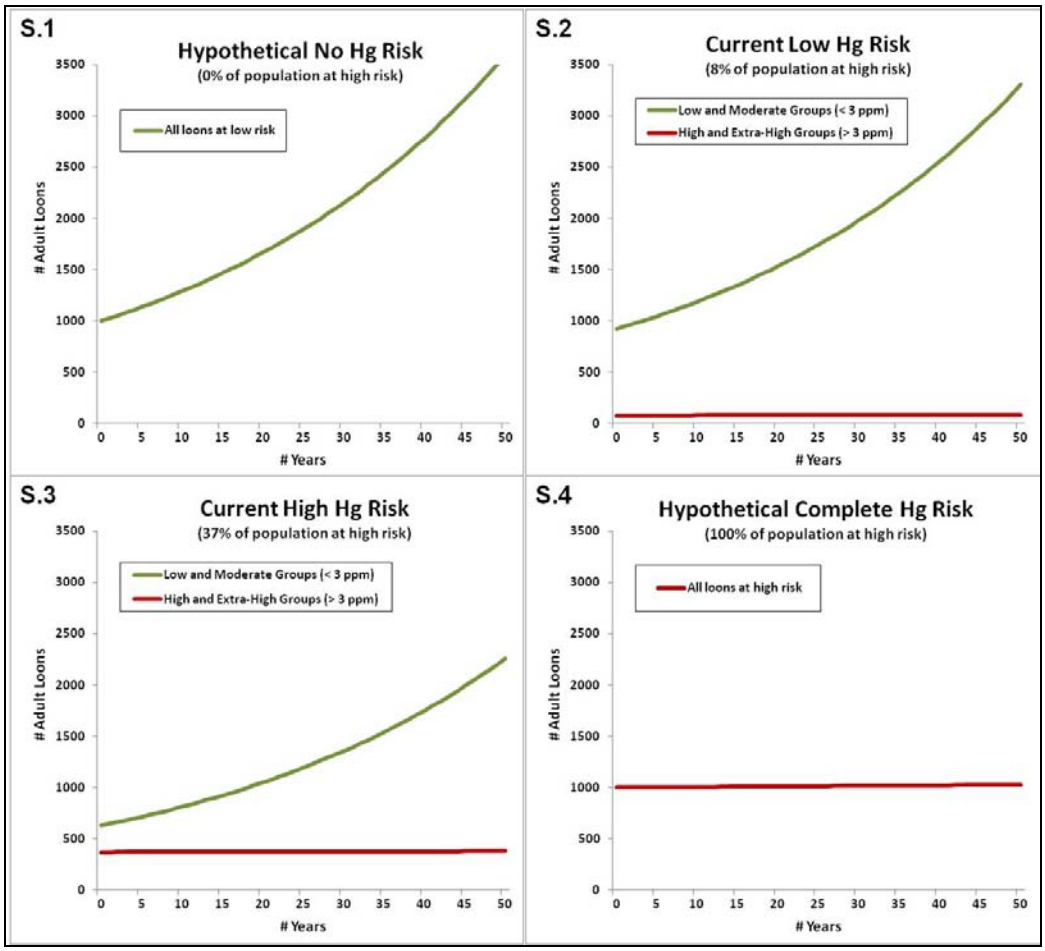


**Figure 41. Adirondack Park loon population growth under four different scenarios. S.2 and S.3 assume a hetergenous population, where the high and extra high groups are growing at a slower rate than the low and moderate groups.**

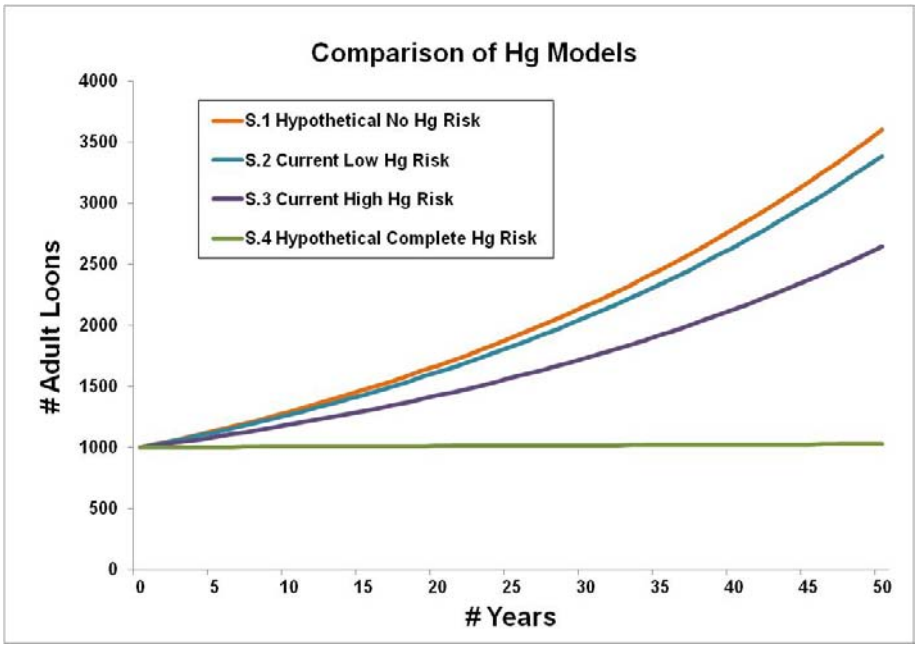


**Figure 42. Comparison of overall population growth in the four different scenarios.**

**Recommended water mercury level to protect the Adirondack common loon population**

The equation used to calculate the WCV and components of the WCV are explained in the Methods section. We determined the exposure parameters for the WCV based on the following:

**Uncertainty factors (UFL, UFA, UFS)**

The uncertainty factors identified by Nichols et al. (1999) were not quantified for the Adirondack Park, and we therefore defaulted to the Great Lakes Water Quality Initiative (GLI) values. The uncertainty factor between species ($UF_A$) is 3, the uncertainty factor between subchronic and chronic levels of impacts ($UF_S$) is 1, and the uncertainty factor between the lowest observed adverse effect level (LOAEL) and the no observed adverse effects level (NOAEL) ($UF_L$) is 2.

**Water consumed (WA)**

Average daily volume of water ($W_A$ = 0.12 L/d) consumed is based on the generic value for common loons given in Nichols et al. (1999).

**Body weight (WTA)**

The average female body weight for common loons in the Adirondack Park was 4.31 kg (N = 94, SD = 0.358) and the average male body weight was 5.59 kg (N = 101, SD = 0.392).

**Ingestion rate (FA)**

Ingestion rate is the average daily mass of food consumed by common loons. It varies between male and female, due to their variation in body size. We calculate ingestion rate as 20% of the average body weight, which equals 1.118 kg/d for males and 0.862 kg/d for females (based on adult body weight estimated above).

**Tested dose (TD)**

The tested dose from toxicity studies with wildlife species is calculated using the prey mercury level known to cause effects and the ingestion rate of the species in question. We use a prey fish mercury level of 0.16 µg/kg that has been shown in two different studies to be a relevant level for common loon reproduction (Evers et al. 2008, Burgess and Meyer 2008). To calculate the tested dose for Adirondack Park loons, we multiply the ingestion rate by the fish mercury level of 0.16 µg/kg.

**Diet fraction (FD3, FD4)**

The fraction of diet coming from either trophic level 3 or 4 is based on Barr (1996), which found that juvenile loons in Ontario consumed 83% trophic level 3 fish and 17% trophic level 4 fish. Because the juveniles were not yet at their full body weight, we must adjust the amount of fish consumed in each size class to represent the size classes used in this study. We calculated that Adirondack Park females (mean = 4310 g) are 15% larger than the juveniles (mean = 3680 g) and so consume 15% less trophic level 3 fish (71% trophic level 3 and 29% trophic level 4). Adirondack Park males (mean = 5590) are 34% larger than the juveniles (mean = 3680) and so likely consume 34% less trophic level 3 fish (55% trophic level 3 and 45% trophic level 4).

**Bioconcentration factor (BCF3, BCF4)**

Data collected between 2003 and 2004 for the different trophic levels indicate mean BCFs of 56,000 for trophic level-3 fish and 97,000 for trophic level-4 fish, based on the relationship of total mercury in unfiltered water with total mercury in fish (Figure 43).

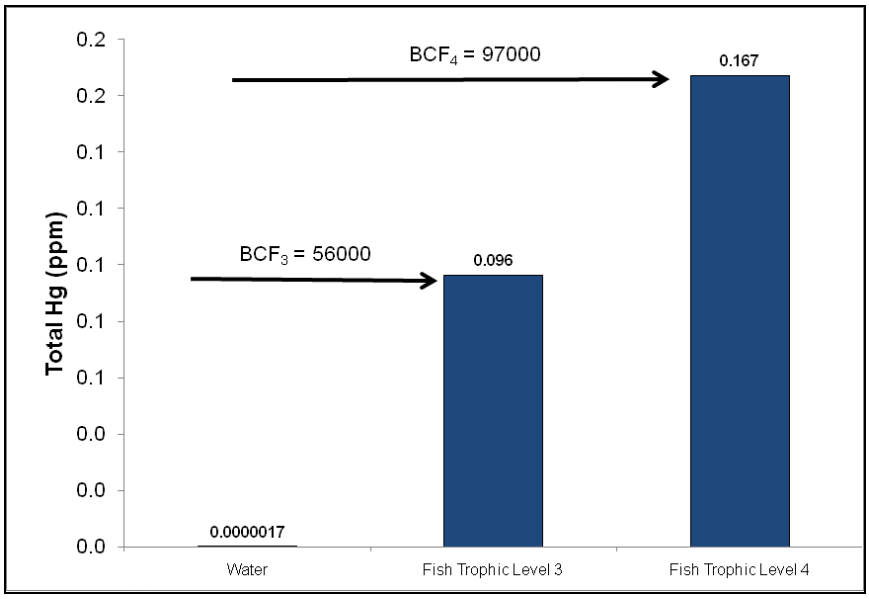


**Figure 43. Bioconcentration factors for fish in trophic level-3 and fish in trophic level-4, compared to unfiltered water.**

Using the algorithm based on Nichols et al. (1999) and our exposure parameters (Table 13), we calculated the WCV for male loons as 2.002 ng Hg/L (Figure 44) and for female loons as 1.693 ng Hg/L (Figure 45).

**Table 13. Sex-specific variable used to calculate the Wildlife Criterion Value.**

| Variable | Male | Female |
|---|---|---|
| TD | 0.179 | 0.138 |
| $UF_L$ | 2 | 2 |
| $UF_A$ | 3 | 3 |
| $UF_S$ | 1 | 1 |
| $WT_A$ | 5.59 | 4.31 |
| $W_A$ | 0.12 | 0.12 |
| $FD_3$ | 0.55 | 0.71 |
| $BCF_3$ | 56000 | 56000 |
| $FD_4$ | 0.45 | 0.29 |
| $F_A$ | 1.118 | 0.862 |
| $BCF_4$ | 97000 | 97000 |

$$WCV_{male} = \frac{\{0.179 \times [1/(2 \times 3 \times 1)]\} \times 5.59}{0.12 + [0.55 \times 1.118 \times 56000) + (0.45 \times 0.45 \times 97000)]}$$

$$WCV_{male} = 2.002 \text{ ng Hg/L}$$

**Figure 44. Calculation of the Wildlife Criterion Value for male common loons in the Adirondack Park.**

$$WCV_{female} = \frac{\{0.138 \times [1/(2 \times 3 \times 1)]\} \times 4.31}{0.12 + [0.71 \times 0.862 \times 56000) + (0.29 \times 0.862 \times 97000)]}$$

$$WCV_{female} = 1.693 \text{ ng Hg/L}$$

**Figure 45. Calculation of the Wildlife Criterion Value for female common loons in the Adirondack Park.**

**Accuracy of the Wildlife Criterion Value within the Adirondack Park**

To evaluate how accurate the Wildlife Criterion Value is at predicting protection of common loons in the Adirondack Park, we must compare the values to the water mercury concentrations obtained from our sampling effort. The WCV can be considered accurate for all instances where the water mercury level was below the WCV value and the loon mercury concentration was below the threshold for effect (< 3 ppm, ww) and all instances where the water mercury level was above the WCV value and the loon mercury concentration was above the threshold for effects. We found that the WCV values accurately predicted loon risk for 61% of females (Figure 46) and 73% of males (Figure 47).

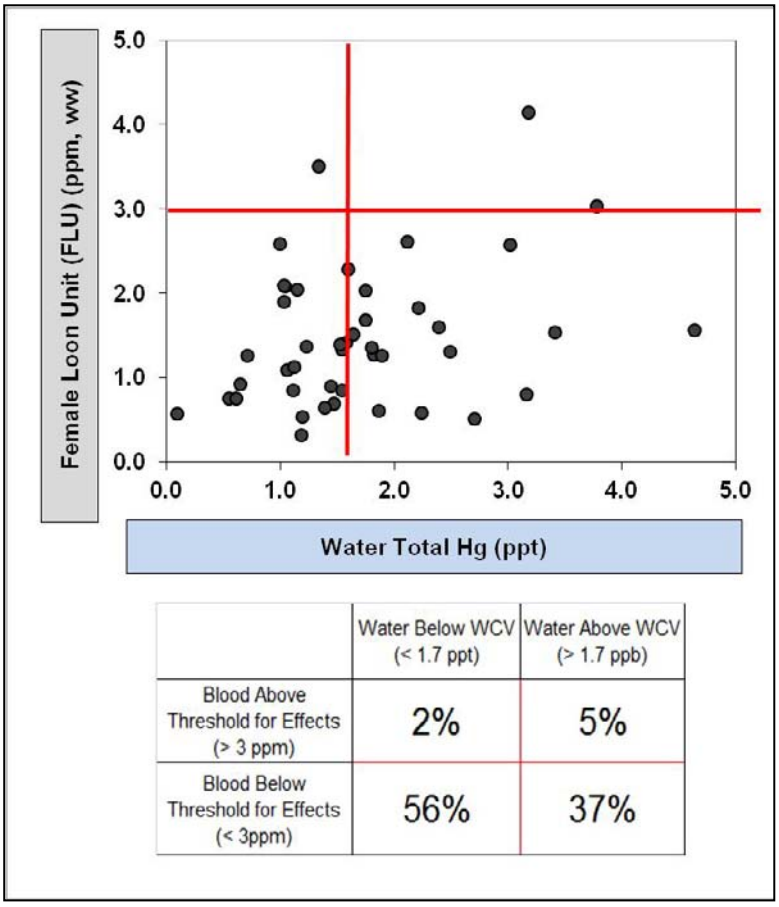


**Figure 46. Relationship between Wildlife Criterion Value (WCV) for female common loons and water mercury concentration, showing that the WCV accurately protects 61% of female loons.**

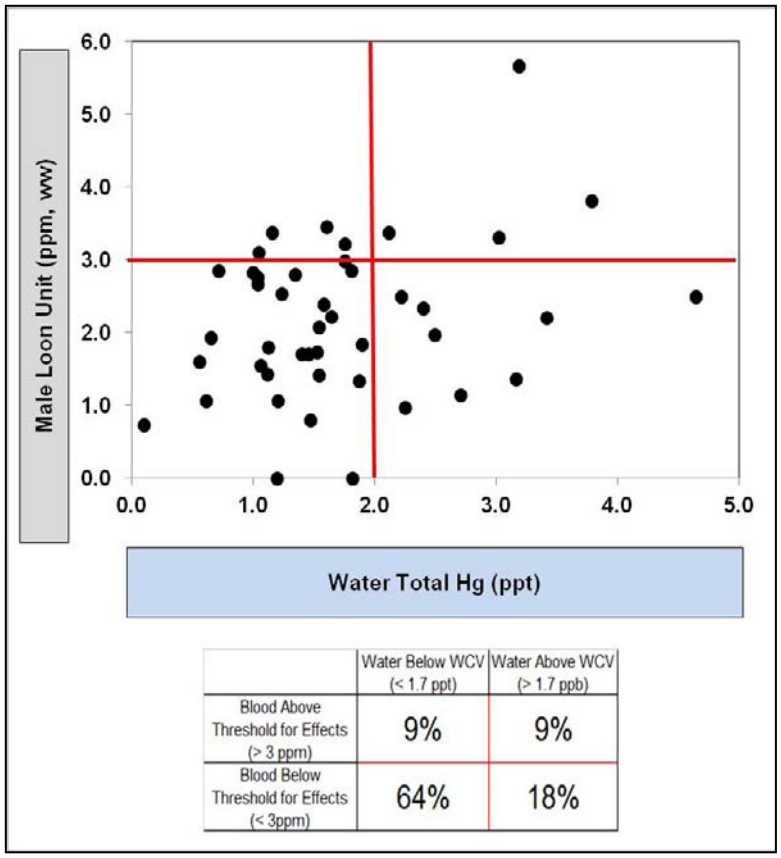


**Figure 47. Relationship between Wildlife Criterion Value (WCV) for male common loons and water mercury concentration, showing that the WCV accurately protects 73% of male loons.**

## 5.0 DISCUSSION

In this project, we employed the common loon as an indicator species to assess the mercury exposure and risk in aquatic ecosystems in New York's Adirondack Park. We utilized abiotic and biotic mercury levels to characterize aquatic-based mercury and to quantitatively assess the ecological risk that mercury deposition poses to Adirondack freshwater habitats. Using common loon mercury levels, we developed a mercury hazard profile, and determined that, in the worst-case scenario, 37% of the Adirondack loon population is at risk. We showed that loon reproductive success is negatively affected by both increased mercury load and increased lake acidity; the upper level of loon productivity is likely limited by both. At a population-level, our results indicate that the growth of the Adirondack Park loon population is limited by mercury exposure. We modified the Wildlife Criterion Value developed by Nichols et al. (1999) to develop a sensitive and appropriate New York-based WCV, and determined that a water column mercury value of 2.002 ng/L and of 1.693 ng/L is protective of male and female common loons, respectively. This information is invaluable for policy makers looking to make better decisions about regulating environmental mercury contamination.

### AQUATIC-BASED MERCURY IN THE ADIRONDACK PARK:

Our first objective was to characterize aquatic-based mercury within our study site, by examining individual lake profiles, the spatial distribution of mercury, and the relationships between mercury in different compartments of the food web. The important findings from this large-scale profile fall into three main categories, which will be discussed below: 1) fish and loon mercury concentrations in many lakes exceed human and wildlife health criteria, 2) the southwestern portion of the Park tends to have higher mercury levels, and 3) relationships between mercury in food web compartments are complex.

**Fish and loon mercury concentrations.** It is of concern that many of our study lakes have fish and loon blood mercury levels exceeding criteria established for the protection of human and wildlife health. Mercury concentrations in 7% of all fish and 12% of the YPE samples exceeded the EPA tissue criterion for methylmercury in fish (0.3μg/g). Twenty-three percent (10/44) of our Adirondack study lakes had at least one fish sample with mercury concentration above 0.3 μg /g, and half of those lakes were not currently listed on the New York State fish consumption advisory (Yu et al. 2011). Sixty-four percent (28) of the 44 Adirondack lakes had at least one fish (YPE) in excess of the 0.16 μg/g mercury level, which has been shown to significantly decrease loon reproduction (Evers et al. 2008); 48% (21) of the study lakes had at least one YPE mercury level greater than the 0.21 μg/g threshold that Burgess and Meyer (2008) found to reduce loon productivity by 50%; and 9% (4) of the study lakes had one or more YPE mercury level in excess of the 0.41 μg/g mercury threshold value at which Barr (1986) and Burgess and Meyer (2008) predicted that loon reproduction would fail completely (Yu et al. 2011).

Elevated mercury levels may also affect fish behavior (Webber and Haines 2003), resulting in piscivorous species such as loons disproportionately feeding on fish with reduced predator-avoidance abilities (Evers et al. 2008). Mercury levels in fish are reflective of their diet. Fish mercury levels increased with size class and fish length, with trophic level-4 large and extra large fish (fish prey specialists) having higher mercury concentrations than trophic

level-3 small and medium fish (insect prey specialists; USEPA 1997). Fish mercury concentrations were similar to those found in other studies in the Adirondacks (Dittman and Driscoll, 2009; Simonin et al. 2008b) and the Northeast (Kamman et al. 2005).

**Geographic distribution of mercury.** The southwestern Adirondacks had a tendency toward lakes with higher loon blood mercury levels, corresponding to increased acid deposition in that area of the Park (Evers et al. 2011). The acidic lakes in the southwestern part of the Park had higher fish and zooplankton mercury levels than non-acidic lakes. The highest loon blood mercury lake (Ferris, FLU = 4.135) had five-fold higher mercury levels than the lowest lake (Cedar River Flow, FLU = 0.685), and was also considerably more acidic (pH = 5.94 vs. pH = 6.87). Lake pH correlated with loon mercury levels, indicating that mercury uptake in loons was driven, in part, by lake acidity increasing the bioavailability of mercury via an increased methylation rate. Loons that breed in mercury "hotspots," such as the southwestern Adirondacks (Driscoll et al. 2007b, Evers et al. 2007), are likely to increase their mercury body burden annually due to the inability to sufficiently depurate and demethylate an elevated dietary uptake of mercury. Mercury hotspots have potential to cause age-related increases in mercury concentrations leading to a reduction in an individual's lifetime reproductive success, and eventually altering the age-structure of the population towards younger individuals (Evers et al. 2008).

Evers et al. (2011) found that geographic patterns of loon mercury exposure and risk indicated that the greatest risk to aquatic systems in the Adirondacks occurred in the leading edge of the western part of the Park. Our results indicate that, based on the variability of biotic mercury concentrations for nearby lakes, landscape factors likely affect mercury bioaccumulation in the Adirondacks, in addition to lake chemistry and biology. Yu et al. (2011) found slight positive correlations between elevation and mercury bioconcentration factor in zooplankton, crayfish, and fish for our study lakes. The greater atmospheric mercury and acidic depositions to high elevation lakes (Miller et al. 2005) probably both contribute to increased mercury bioaccumulation through increased mercury bioavailability, as well as increased sensitivity to surface water acidification because of thin soils and surficial deposits (Driscoll et al. 1991, Ito et al. 2002). The spatial patterns of biotic mercury concentrations in our study were probably driven more by the influence of lake pH on methylmercury bioavailability and/or trophic transfer than by elevation because a relationship between total mercury or methylmercury in lake water and elevation was not found (Yu et al. 2011).

**Relationships between mercury concentrations at different levels of the food web**. Neither total mercury nor methylmercury in water correlated with lake acidity in our study lakes, which is similar to findings by other researchers (e.g., Driscoll et al. 1994; Dittman and Driscoll 2009). However, Yu et al. (2011) found that biotic mercury levels on our study lakes, particularly in trophic level 3 and 4 fish and in common loons, were negatively correlated with increasing lake pH and acid neutralizing capacity, indicating that the acid-base status of a lake influences the methylmercury accumulation in biota living in the lake, probably due to the increased methylation of mercury within acidic aquatic systems. The increased availability of sulfate with acidic deposition promotes mercury methylation through an increase in sulfur reducing bacteria that drive the methylation process (Jeremiason et al. 2006). The role of acidity in assimilation of methylmercury at the bottom of the food web and/or trophic transfer up

the food web potentially explains the relationship between biotic mercury levels and lake acidity (Wyn et al. 2009). Likewise, in the Canadian Maritimes and Wisconsin, Burgess and Meyer (2008) found a strong negative relationship between lake acidity and mercury concentrations in small fish and blood mercury levels in common loons over a wide range of pH (i.e. 4.3–9.5).

Loon and fish mercury levels were directly related to methylmercury concentrations at lower levels of the food web (Yu et al. 2011). A variety of factors contribute to the availability of mercury in Adirondack waterbodies, including numerous wetlands facilitating the transport of mercury to downstream lakes and the production of methylmercury (Selvendiran et al. 2008), and the poor productivity of Adirondack lakes, which enhances bioconcentration of mercury (Chen and Folt 2005). It is likely that landscape characteristics (e.g., numerous wetlands, thin soils with poor buffering capacity) contribute to the sensitivity of Adirondack ecosystems to mercury inputs as well as ongoing effects of acidic deposition, resulting in the Adirondacks being classified as a biological mercury hotspot.

Zooplankton mercury levels were correlated with small and medium fish mercury levels, but not as strongly with large or extra-large fish mercury. Although zooplankton are a low aquatic trophic level, their methylmercury bioconcentration factor values are relatively high (Driscoll et al. 2007b), reflecting the importance of the lower food web in establishing the degree of mercury concentration for taxa at higher levels of the food web, and thus, affecting mercury exposure for wildlife and humans (Driscoll et al. 1994; Kamman et al. 2005). Crayfish mercury levels, unlike zooplankton, correlated to mercury concentrations in large and extra-large fish, but not those in smaller fish, reflecting that crayfish are a prey item for larger fish, but smaller fish are more likely to eat lower trophic level items. Loon mercury levels also correlated with large and extra-large fish as well as crayfish mercury concentrations, all of which are common prey for these birds. The relationships between compartments of the food web are not as strong as expected, indicating that many factors come into play when determining how mercury bioaccumulates in the food chain. For this reason, loon monitoring programs designed to only sample loon prey (i.e., fish and crayfish) would likely overlook potential loon mercury hotspots. Thus, despite the increased logistics involved, we recommend continuing to sample common loons as a part of any long-term mercury biomonitoring program.

All of our food web samples were collected within the loon breeding season in 2003 and 2004, in order to sample the most biologically-relevant time period. Temporal variation in mercury in abiotic and biotic compartment does exist, and this study was not designed to assess this issue. More intensive sampling throughout the spring, summer, and fall is the only way to fully assess overall mercury concentrations. Because of this, although common loons are more logistically difficult to sample, they represent an important biological endpoint; thus, by sampling loons, we can accurately show how much mercury is actually bioaccumulated in wildlife tissue, instead of assessing solely how much mercury is in the water.

## MERCURY HAZARD PROFILE FOR THE COMMON LOON

Our second objective was to develop a mercury hazard profile using the common loon as an indicator species for biota living in Adirondack freshwater ecosystems. Loon blood mercury levels reflect recent dietary exposure; strong

evidence indicates that adult blood mercury levels reflect prey mercury levels in their breeding territory (Evers et al. 2004; Burgess and Hobson 2006). In the Canadian Maritimes and Wisconsin, Burgess and Meyer (2008) found that loon blood mercury concentrations were increased in lakes with high fish mercury levels. As in other studies, male loon blood and feather mercury levels in the Adirondacks were greater than female blood and feather levels, due to the larger males consuming larger (and likely older) prey items that have higher mercury concentrations, and the ability of females to sequester mercury in the eggs they lay (Evers et al. 2004). Adult loon blood mercury levels were significantly higher than chick blood mercury levels, reflecting their increasing mercury body burden over time, and the increased exposure of adults feeding on larger, higher trophic level prey items. Feather mercury provides insight into the lifetime mercury body burden of an individual loon, as muscle protein reservoirs are remobilized during feather molt. Evers et al. (2008) found that loon feather mercury increased by an average of 8.4% per year.

Loons were placed into low, moderate, high, and extra-high risk categories for interpretive purposes based on mercury concentrations in female loon units. Low risk indicates background mercury levels that are minimally impacted by anthropogenic inputs. Birds in the moderate risk category have elevated mercury levels but the impact levels on the percent of individuals have not yet been determined. Loons in the high-risk category are exposed to toxic levels of environmental mercury that potentially have molecular, organism, and/or population effects. The extra high mercury category is based on known impacts on loons and other birds. The upper limit of the low risk category is considered the no observed adverse effect level (NOAEL), and the lower limit of the high risk category is the lowest observed adverse effect level (LOAEL; Evers et al. 2004).

Adult loons with blood mercury concentrations of more than 3.0 ug Hg/g or feather mercury > 20 ug Hg/g, or loon eggs with mercury >2.0 ug Hg/g are at high risk for significant adverse physiological, behavioral and reproductive effects (Evers et al. 2008). Our results indicated that 21% adult male and 8% of female Adirondack loons in our study were at high risk of behavioral and reproductive impacts due to blood mercury exposure, and 37% of male and 7% of female study birds were at high risk due to feather mercury exposure. When such a high proportion of the breeding population is above the LOAEL, mercury exposure is likely to cause population-level impacts (Evers et al. 2011). Female blood mercury levels are highly correlated with egg mercury levels (Evers et al. 2004, Evers et al. 2008, Burgess and Meyer 2008, Evers et al. 2003), thus eggs are also relevant tissues for predicting mercury risk within a breeding territory. Thirteen percent of the Adirondack loon eggs sampled were at high risk for mercury exposure, indicating that if the chicks hatched, their behaviors would be abnormal, and they would have a reduced likelihood of surviving to fledging. Several controlled studies have found that mercury exposure impairs egg development and hatchability at levels (i.e., 0.5-4.4 μg/g) which were found in this study (Borg et al. 1969, Fimreite 1971, Heinz 1979, Spann et al. 1972, Gilbertson 1974).

Putting our data into a geographic context, the Adirondack Park common loon mercury concentrations accurately reflect a west to east increase in mercury levels within North America, as has been noted in other studies (Evers et al. 1998).

**EFFECT OF MERCURY ON THE ADIRONDACK COMMON LOON POPULATION:**

We used three separate analyses to explore the effect of mercury on the common loons in the Adirondack Park: 1) by analyzing the effect of mercury and lake acidity on loon fecundity, 2) by applying a population model to assess the long-term impact of mercury on the Adirondack breeding loon population, and 3) by developing a Wildlife Criterion Value to establish a water column mercury value that is protective of wildlife.

**Effect of mercury and lake acidity on loon reproductive success**

Using quantile regression, Burgess and Meyer (2008) found that mercury exposure was associated with a linear upper limit on loon productivity, supporting the hypothesis that mercury exposure in common loons was a limiting factor on reproductive success. Similarly, in the Adirondacks, we found that loon productivity decreased significantly with increasing mercury body burdens. High risk territorial pairs (>3.0 µg/g) fledged approximately 20% fewer chicks per pair than loons with lower mercury levels (<3 µg/g). Similar patterns of lower productivity were found for other reproductive parameters. Other loon populations with blood mercury levels greater than 3.0 ug/g in the Northeast are also experiencing significant reproductive impacts – for example, breeding loons in Maine with high mercury concentrations fledged 40% fewer young than pairs with mercury levels below 1.0 µg/g (Evers et al. 2004, Burgess and Meyer 2008).

Quantile regression indicated that the maximum Adirondack loon productivity would be ~1.0 chick/territorial pair if female or male loon mercury exposure was zero, and that productivity would be reduced by 50% when female blood mercury levels were 3.3 µg/g or male blood mercury levels were 4.5µg/g. Because of the small sample size at the upper quantiles, we did not find a significant relationship at the $90^{th}$ quantile. However, the slope of the $90^{th}$ quantile is much steeper than the $80^{th}$, indicating that mercury regulates loon productivity more dramatically at the upper end of the distribution. Thus, mercury appears to be a primary anthropogenic stressor for the Adirondack common loon population, resulting in decreased productivity.

Like Burgess and Meyer (2008), we also found that some loons with low mercury exposure also had low productivity, indicating intrinsic (e.g.: species longevity, intraspecific interactions due to density), extrinsic (e.g.: predation, weather), or anthropogenic (e.g.: human disturbance, other contaminants) stressors other than mercury are impacting their reproductive success. However, several studies have identified mercury as a cause of reduced loon productivity (Barr 1986, Burgess and Meyer 2008, Evers et al. 2008). And, as in Wisconsin, New Brunswick, and Nova Scotia (Burgess and Meyer 2008), we found that Adirondack loon productivity was never high when mercury exposure was high.

It is interesting that the annual productivity we observed for the overall Adirondack study population, 0.594 chicks fledged per territorial pair, was considerably lower than that observed by both Trivelpiece et al. (1979) in the 1970s (0.83 CF/TP) and Parker et al. (1986) in the 1980s (0.96 CF/TP) New York loon population surveys. Differences in study methodology may potentially account for the difference in productivity results, as both previous surveys evaluated two years of loon productivity for a larger number of lakes with only two to four visits per lake annually,

while our study evaluated nine years of intensive (weekly) observations on a smaller number of loon territories and lakes.

We examined relationships between lake acidity, loon productivity, and mercury exposure to assess potential impacts of acid deposition to aquatic ecosystems. The results of our study support the conclusion that increased mercury exposure of loons breeding on acidic lakes detrimentally affects their productivity because we found a positive trend between loon reproductive success and increasing lake pH. Based on the results of our quantile regression, lake acidity was potentially a limiting factor on loon productivity, with maximum productivity attained at pH = 6.64 and a 50% reduction in productivity with lakes that had a pH of 5.16.

In Wisconsin, loons occupying low pH (< 6.3) lakes had significantly higher blood mercury levels in comparison to loons nesting on neutral pH lakes (Meyer et al. 1995). Alvo (1996; 2009) found no loon productivity on lakes with pH < 4.4, significantly lower productivity on lakes with pH < 5.8 and no impact on lakes with pH > 6.6. Alvo (2009) attributed chick mortalities on lakes with very low pH to reduced growth after hatching due to inadequate food resources, and concluded that the critical lake pH for loon breeding success was 4.3, and that a lake pH of approximately 6.0 was an important threshold for loon productivity. Although we evaluated loon productivity on territories with three or more years of observations based on lake acidity (pH<6.3 vs. pH > 6.3), we did not observe a significant difference between the two groups. Parker (et al. 1986; 1988), in a two-year study of loons breeding on acidic and non-acidic Adirondack lakes with and without fish, concluded that the presence of loons, the incidence of breeding, hatching, and fledging success were not affected by the acidity of a lake.

Alvo (2009) attributed the low fledging success of loons breeding on acidic lakes in Ontario to the decreased availability of food resources in those lakes. Parker (1988) felt that the impact of lake acidification on loons would manifest as quality food not being sufficient for larger chicks who have increasing energetic demands, possibly weakening and predisposing them to other factors resulting in mortality. However, Alvo (2009) and Parker (1988) did not examine the potentially confounding factor of increased mercury exposure affecting loon productivity on acidic breeding lakes. Burgess and Meyer (2008) concluded that the increased mercury exposure of loons living on acidic lakes is more likely to be the cause of reduced fledging success. They found that, although fish species diversity decreased in acidic Maritime lakes, the biomass of small fish (of suitable prey sizes for loons) actually increased, confirming that decreased loon productivity on acidic lakes was not due to lower prey abundance. Burgess and Meyer (2008) also found that data from Parker (1988) indicated that, for 24 Adirondack lakes, including several of our study lakes, there was no relationship between lake acidity and prey biomass.

**Model for the long-term effect of mercury on the Adirondack loon population.**

Parker (1987) developed a deterministic population model for the Adirondack loon population, using a fledging rate of 1.0 which was observed in the 1980s NY loon survey, to assess what life history parameter levels were required to have a stationary population, and to have a seven percent annual growth rate. Because of the lack of specific knowledge about many common loon life history parameters, Parker estimated several parameters for his model, many of which are now known or better determined due to extensive observations of banded loons throughout North

America (Mitro et al. 2008). Grear et al. (2009) were able to utilize these known parameters (annual adult survivorship of 92%, subadult survival of 41%, and average first year breeding age of 6 years; Mitro et al. 2008, Evers 2007) in the development of their density-independent stage-based matrix loon population model to better assess how mercury impacts common loon populations, and enabling them to further refine the model to provide a foundation for conducting risk assessments (Nacci et al. 2005) and population viability analyses.

The estimated population growth rates ($\lambda$) for the three estimates of mercury body burden in Adirondack loons were all above 1.0, indicating that the current birth and survival rates of the Adirondack loon population are likely able to support a stable or growing population. The overall population growth rate is estimated at 1.6%, a much lower rate than the 7% annual growth rate calculated by Parker et al. (1986) in the 1980s. It is notable that the high/extra-high mercury group had a considerably lower population growth rate (0.05 percent, just high enough for maintenance of a population) when compared to the low mercury group (2.6 percent), suggesting that environmental mercury contamination has indeed affected the growth of a portion of the Adirondack loon population. Grear et al.'s (2009) population matrix model indicates that loon breeding populations producing fewer than 0.48 chicks fledged per territorial pair are population sinks (Evers et al. 2008). The Adirondack high/extra-high mercury loons are producing 0.483 CF/TP, and thus, are probably acting as a population sink. The remaining Adirondack loon population is likely acting as a buffer population by filling unoccupied territories and producing enough chicks to maintain, and possibly even expand, the population as a whole.

Our projected population simulations over 50 years for the different mercury body burden scenarios provide a graphic extrapolation of how the Adirondack loon population could grow based on the effect of mercury contamination. Nevertheless, because of natural environmental variability, it is unknown if these numbers are indeed sufficient to ensure long term population growth. It is important to remember that these scenarios are representing a hypothetical situation in which mercury is the only factor affecting population growth, and that in reality, numerous other intrinsic (e.g., intraspecific competition), extrinsic (e.g., predation, carrying capacity of the habitat/availability of high-quality territories) and anthropogenic (e.g., recreational disturbance, lakeshore development) stressors affect the Adirondack loon population, and could potentially negatively impact the population growth rate. The productivity (CF/TP) patterns correlated with patterns of the mercury body burden in the loons, but determining the causative relationship can be difficult, as such factors as lake acidification and size could affect food availability, and may also be contributing factors influencing loon population growth.

An increasing Adirondack loon population is also supported by numerous anecdotal observations from many Adirondack residents (Schoch, pers. comm.), and preliminary analysis of an annual New York loon count conducted since 2001 (Schoch and Sauer, unpubl. data), indicating a current adult population of 1500-2000 birds. The life history characteristics of longevity, slow to mature, and low fecundity of this species mean that a population enduring annual and continual impacts from a stressor such as mercury contamination would result in erosion of the affected population over time (Evers et al. 2004). We assume that the overall Adirondack loon population is equally exposed to other extrinsic and anthropogenic stressors present on the breeding grounds (Evers et al. 2008), thus, the results of our population modeling indicate that the risk imposed by mercury bioavailability in Adirondack aquatic

ecosystems to high trophic level obligate piscivores such as common loons causes a long-term impact on the population growth and size of the segment of the Adirondack loon population breeding on acidic lakes in the Park.

**Recommended water mercury level to protect the Adirondack common loon population.**

The Wildlife Criterion Value utilizes measurement of contaminant stressors, such as surface water mercury concentrations, to estimate the viability of wildlife population exposed to the stressors (Nichols et al. 1999). The results of our calculations indicate that water levels equal to or less than 2.002ng Hg/L are protective of male loons, and equal to or less than 1.693 ng Hg/L are protective of female loons at the population level. Both of these WCVs are greater than the WCV of 1.30 ng Hg/L that the Great Lakes Water Quality Initiative uses for avian species (Evers et al. 2004).

We assessed how the WCV for water mercury compared to the water mercury concentrations that we sampled within the Adirondack Park and found roughly 2/3 of birds were accurately classified based on the water mercury concentration of their respective study lake. Due to logistical constraints in this study, we collected water quality data within a relatively short time frame, which standardizes the mercury comparison between lakes, but likely misses a large amount of yearly, monthly, and daily variation in water mercury that occurs in lakes. Although our water mercury concentrations are likely a small "snapshot" of mercury within the lakes, it is important to note that the WCV is calculated independent of actual water mercury values within the study area. Because of this, we believe that the WCV is likely accurate, and that more rigorous testing of water mercury within the study lakes would show better correlations between the WCV, loon blood and water mercury.

## CONCLUSIONS

In summary, the results of our study indicate that:

1. Mercury appears to be a primary anthropogenic stressor for the Adirondack common loon population, resulting in decreased productivity, as Adirondack loon productivity was never high when mercury exposure was high.

2. Increased mercury exposure of loons breeding on acidic lakes detrimentally affects their productivity.

3. The Adirondack loon population is apparently increasing, although at a much lower rate than the seven percent annual growth rate calculated by Parker et al. (1986) in the 1980s.

4. The risk imposed by mercury bioavailability in Adirondack aquatic ecosystems to high trophic level obligate piscivores causes a long-term impact on the population growth and size of the segment of the Adirondack loon population breeding on acidic lakes in the Park.

In conclusion, our results provide valuable new information that (1) contributes to documenting the extent of mercury contamination and its impacts to New York's aquatic ecosystems; (2) provides evidence for ecological damage to public resources; (3) establishes a baseline for detecting future changes in biotic impacts from atmospheric mercury deposition; and (4) provides science-based justification for policy-makers to stringently

regulate mercury and acidic emissions on local, regional, and national scales. Thus, our study provides support for the critical need to better regulate mercury emissions on national and local scales to protect biota living in aquatic ecosystems from the impacts of environmental mercury contamination.

## POLICY IMPLICATIONS

Because elevated mercury concentrations in aquatic biota are linked to acidic deposition, it is likely that increasingly stringent regulations for atmospheric emissions of sulfur dioxide and nitrogen oxides (Driscoll et al. 2007a) will have the co-benefit of reducing biotic mercury levels (Yu et al. 2011). Evaluation of mercury levels and productivity of loons breeding on Adirondack lakes with pH levels below 5.0 would provide further insight into the potentially synergistic interactions between mercury and acidic environmental contamination and the risk that mercury and acidic deposition pose to aquatic ecosystems in the Park. In this study, like Burgess and Meyer (2008), our loon mercury and productivity data (and the corresponding food web data) was limited for very acidic lakes ($pH < 5.0$) and for those with elevated mercury exposure levels because our ability to capture loons is primarily restricted to loons who successfully produce chicks. Thus, loons with very elevated blood mercury concentrations may be excluded from this study, since they did not have chicks, and so were not responsive to our capture technique. Therefore, on very acidic lakes where loon capture is not logistically feasible, evaluation of mercury concentrations throughout the food web would provide especially critical information regarding the impact of environmental mercury pollution to aquatic ecosystems.

It is critical to develop standardized state, regional, and national monitoring networks for both abiotic and biotic mercury in the aquatic foodweb, as is proposed in the National Mercury Monitoring Program (Mason et al. 2005), to inform federal and state mercury-related policies, provide data for predictive models, and characterize the biological effects in the United States from the redistribution of anthropogenic mercury on the landscape (Evers et al. 2011). Long-term studies of biotic mercury levels, particularly high-trophic level species living in acidic or high mercury habitats, would contribute much information about the risks mercury and acidic deposition pose to wildlife and aquatic ecosystems. A standardized biotic mercury monitoring program would ensure that recently implemented New York State and regional regulations are effective at preventing local mercury hotspots (Evers et al. 2007) and biotic impacts such as decreased reproductive success in common loons.

There are indications that the acidity of Adirondack lakes, and potentially elsewhere in North America, has been improving over time as sulfur emissions decrease with the implementation of the 1990 Clean Air Act Amendments, leading to biological recovery in some previously extremely acidic lakes (Driscoll et al. 2007b). In Ontario, Alvo (2009) found that some very acidic lakes previously incapable of supporting loon reproduction could do so as the pH increased from the mid-1980s through the 1990s.

There is also encouraging evidence that biotic mercury levels decrease in response to declines in atmospheric deposition of acids and mercury. In northern Wisconsin, Hrabik and Watras (2002) found that fish mercury levels declined in conjunction with decreases in acidic and mercury deposition, and their results suggested that, over short-time scales, small changes in acid rain or mercury deposition could affect the bioaccumulation of mercury. Munthe

et al (2007) synthesized information on the connections between changes in mercury ecosystem loading and fish methylmercury levels. They concluded that fish mercury levels responded to increases or decreases in mercury loading, but that the timing and magnitude of the response depended on ecosystem-specific variables and the form of mercury deposited into the environment (Munthe et al. 2007). In an experimental study (the Mercury Experiment to Assess Atmospheric Loading in Canada and the United States) that manipulated deposition rates of different mercury isotopes in an entire ecosystem, Harris et al (2007) found that biotic mercury levels rapidly increased linearly with mercury deposition on the lake surface, but that new inputs into the surrounding watershed filtered slowly over a long time period into the lake. They predicted that initially fish mercury concentrations will rapidly (within years) decrease in response to reduced atmospheric deposition of mercury and in direct relation to the decreased atmospheric input, followed by a more gradual (decades) decline over time with decreasing mercury inputs from the watershed. Additionally, they concluded that lakes with small watersheds relative to their surface areas will respond the most effectively to decreasing mercury deposition (Harris et al. 2007).

Our study provides additional evidence, based on the ecological injury mercury poses to biota living in freshwater ecosystems, for the need to stringently regulate mercury emissions on national and global scales. Since a primary source of environmental mercury contamination is airborne deposition, which does not recognize local or national boundaries, it is essential to regulate mercury emissions from all sources throughout North America as well as globally. Strict mercury emission regulations for coal-fired power plants have recently been implemented in the Northeast and New York, which will minimize impacts due to local point sources. However, national mercury emission regulations for coal-fired power plants (Mercury and Air Toxics Standards), have only recently been finalized, and have yet to be implemented (US EPA 2011). And, although the United Nations Environment Programme (UNEP) Global Mercury Partnership is working to protect human and environmental health globally from mercury by minimizing and eventually eliminating mercury releases to the environment due to anthropogenic sources (UNEP 2011), a comprehensive global mercury pollution policy has not yet been implemented.

Thus, despite new state and regional regulations, New York and the Northeast continue to receive mercury deposition, and common loons summering in the Adirondack Park will continue to be affected by mercury pollution until all sources of mercury emissions are greatly reduced or eliminated entirely. We look forward to the day when the haunting call of the Adirondack loon will echo across the lakes and mountains of the Adirondacks unhindered by impacts from environmental pollutants such as mercury.

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# 7.0 APPENDIX A. NYSERDA FISH TISSUE STANDARDIZATION

## APPROACH

Individual relationships between yellow perch (YLP) and other fish species were evaluated with an eye toward predicting YLP from the various other species. In order to build the relationships permitting calculation of "yellow perch equivalents," there needed to be a sufficient crossover between yellow perch and other species for like sizes. The design of the dataset, where length was expressed as an ordinal variable, was well suited to this problem. Yellow perch data were paired with other species by lake and size category. Incorporating size within the paired data internalized the effect of size within the fish species data pairing. The predictive relationship between the various species and yellow perch was examined with this in mind.

Where a linear relationship was evident between fish species, this was calculated using linear regression. Where no particular relationship was evident, the mean yellow perch concentration was divided by the mean concentration for the paired species, to derive an adjustment factor. Where sample sizes were insufficient, YLP equivalent concentrations were not calculated. The following adjustment factors were used to estimate YLP equivalent concentrations:

| Yellow perch predicted from: | Adjustment factor |
|---|---|
| PKS (Pumpkinseed) | 1.32 |
| LMB (Largemouth Bass) | 0.76*(LMB)+.032 |
| SMB (Smallmouth Bass) | 0.91 |
| BRB (Brown Bullhead) | 2.9 |
| CKC (Creek Chub) | 3.54 |
| RSF (Red-ear Sunfish) | 0.69 |
| RKB (Rock Bass) | 1.14 |

**Table A-1. Adjustment factors to determine yellow perch equivalent from a given fish species.**

## RESULTS

Mean yellow perch equivalent values were then calculated for each lake using analysis of covariance, and adjusting the mean concentration to a size medium fish (Table A-1). This was done on root-transformed data to achieve statistical normality and homoscedascicity. The ANCOVA was highly significant for the effect of lake and size ($R^2 = 0.84$, $F_{44,136} = 15.8$, $p<0.001$). Leverage and prediction plots showing the strength of the model are as follows.

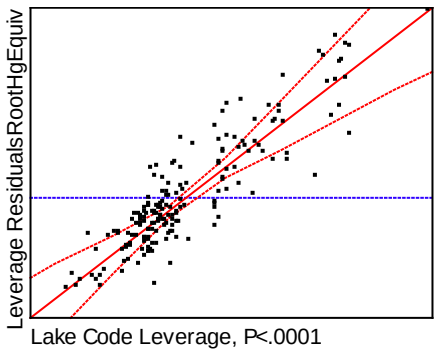


**Figure A-1.**

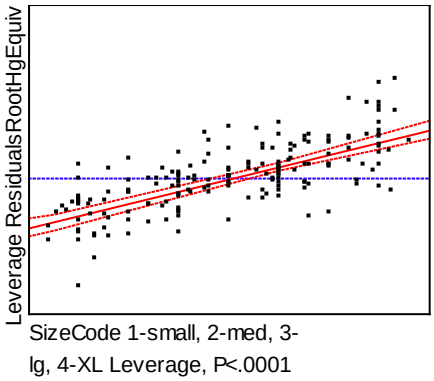


**Figure A-2.**

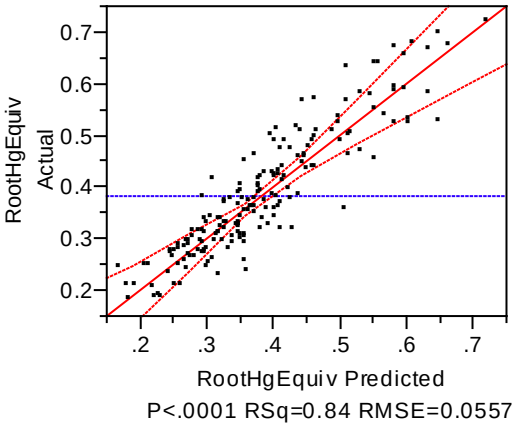


**Figure A-3.**

**Table A-2. Least square mean yellow perch equivalent concentrations calculated from various fishes collected on lakes sampled in this project.**

| Lake name | Adk Lakes Survey Corporation ID # | Length-std Hg (w.w.) | Std Error | Lake arithmetic mean |
|---|---|---|---|---|
| Round Lake | 060118 | 0.043 | 0.058 | 0.056 |
| Middle Saranac | 020110 | 0.047 | 0.033 | 0.060 |
| Massawepie Lake | 030369 | 0.060 | 0.029 | 0.094 |
| Spitfire | 030264 | 0.066 | 0.029 | 0.089 |
| Mason Lake | 050613 | 0.081 | 0.029 | 0.094 |
| Lake Abanakee | 050587B | 0.085 | 0.029 | 0.087 |
| Woodruff Lake | 050681 | 0.090 | 0.029 | 0.096 |
| Taylor Lake | 020227 | 0.091 | 0.029 | 0.120 |
| Garnet Lake | 050520 | 0.093 | 0.029 | 0.105 |
| Piseco Lake-Big Bay | 050234 | 0.102 | 0.041 | 0.102 |
| Nicks | 040804 | 0.105 | 0.022 | 0.108 |
| Dry Channel | 030128 | 0.105 | 0.024 | 0.114 |
| Piercefield Flow | 060085 | 0.106 | 0.020 | 0.116 |
| Horseshoe | 060143 | 0.108 | 0.026 | 0.130 |
| Lows Lake | 060156 | 0.109 | 0.029 | 0.110 |
| Beaver Lake | 070717 | 0.112 | 0.033 | 0.101 |
| Kushaqua | 020055 | 0.113 | 0.026 | 0.129 |
| Deer Pond | 050689 | 0.116 | 0.029 | 0.126 |
| Long Pond | 020149 | 0.119 | 0.026 | 0.117 |
| Private #1 | 050235 | 0.120 | 0.033 | 0.104 |
| Newton Falls | 040301A | 0.121 | 0.029 | 0.152 |
| Chaumont | 040303 | 0.124 | 0.018 | 0.150 |
| East Pine | 020147 | 0.128 | 0.029 | 0.137 |
| Hitchins | 060144 | 0.133 | 0.029 | 0.140 |
| Moss Lake | 040746 | 0.134 | 0.029 | 0.147 |
| Little Safford Lake | 040735 | 0.134 | 0.058 | 0.197 |
| Clear Pond | 030085A | 0.136 | 0.058 | 0.085 |
| Lake Durant | 050645A | 0.139 | 0.026 | 0.131 |
| Seventh Lake | 040787B | 0.139 | 0.026 | 0.147 |
| Limekiln | 040826 | 0.141 | 0.033 | 0.165 |
| Little Clear Pond | 020191 | 0.157 | 0.041 | 0.132 |
| Beaver Lake | 040449 | 0.188 | 0.019 | 0.195 |
| South Pond | 060245 | 0.190 | 0.029 | 0.230 |
| Wolf Pond | 050688 | 0.190 | 0.024 | 0.174 |
| Moshier Reservoir | 040478 | 0.215 | 0.024 | 0.260 |
| Big Moose | 040752 | 0.225 | 0.029 | 0.225 |
| G Lake | 070859 | 0.240 | 0.033 | 0.266 |
| Cedar River Flow | 050667 | 0.245 | 0.058 | 0.174 |
| Henderson | 050715 | 0.254 | 0.034 | 0.320 |
| Arbutus Lake | 050684 | 0.276 | 0.022 | 0.290 |
| Ferris Lake | 070777 | 0.342 | 0.029 | 0.360 |

| Lake name | Adk Lakes Survey Corporation ID # | Length-std Hg (w.w.) | Std Error | Lake arithmetic mean |
|---|---|---|---|---|
| North Lake | 041007 | 0.370 | 0.026 | 0.357 |
| South Lake | 041004 | 0.389 | 0.026 | 0.373 |
| Squaw Lake | 040850 | 0.558 | 0.058 | 0.520 |

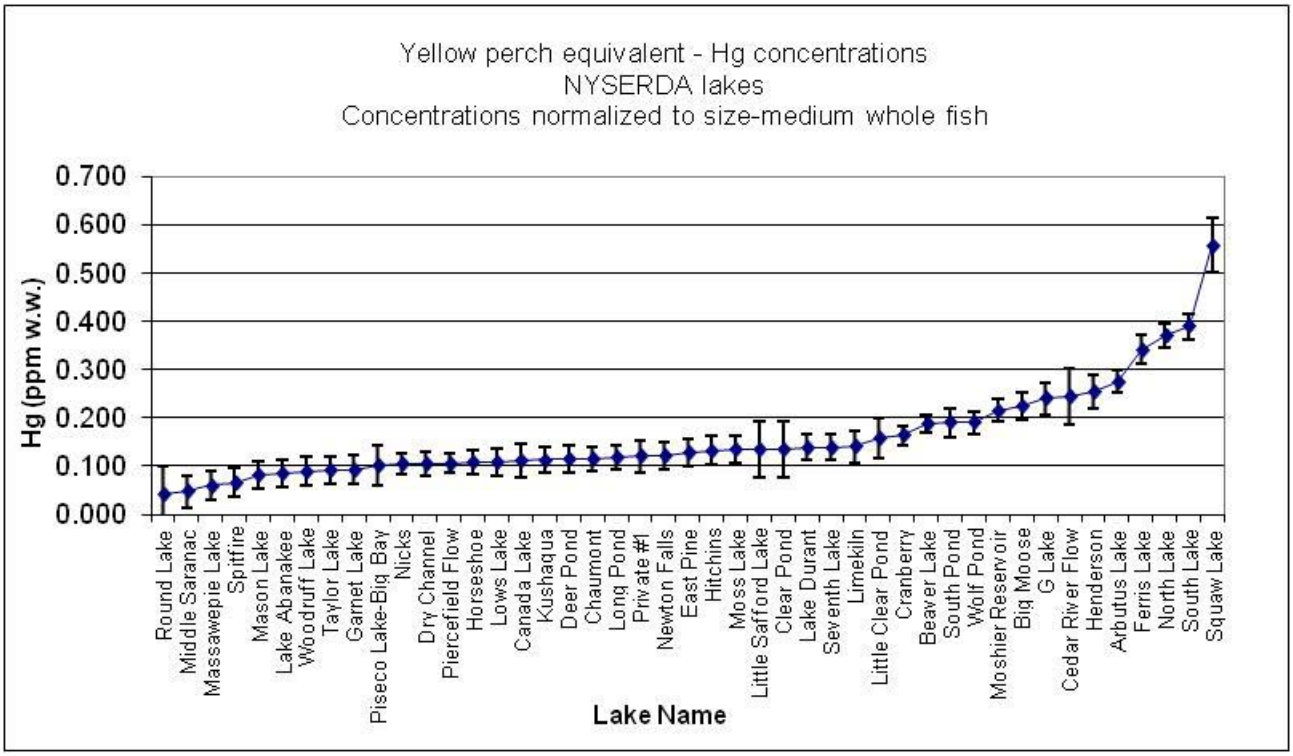


**Figure A-4. Yellow perch equivalent mercury concentrations for Adirondack study lakes.**

# 8.0 APPENDIX B. COMMON LOON DATA COMPILATION AND LOON UNIT MODELING (ADAPTED FROM EVERS ET AL. 2011)

## THE NEED TO DEVELOP A COMMON UNIT FOR LOON MERCURY DATA

To best evaluate and utilize existing data from various biotic compartments, mercury concentrations require a single common unit. Since common loon mercury data are from multiple tissues, including adult male and female blood, juvenile blood, and loon eggs, comparisons between locations and years can be difficult to conduct or assess. To address this issue, we compiled a dataset of common loon data from New York (1998-2008, n=381). Subsets of the data, in which there were multiple mercury data points from a single territory and year, were used to develop relationships between mercury in different tissues. These models were then applied to the larger dataset to present data from all tissue types, territories and years, in a common unit, the so-called "female loon unit" (FLU). Egg mercury levels are clearly correlated with female mercury exposure, as female loons depurate mercury into their eggs. Juvenile mercury, likewise, could be assumed to be more highly correlated with female mercury, as they tend to eat prey of similar size (as opposed to males, for instance, which are larger and tend to eat larger prey). Nevertheless, there is no clear link between egg mercury or juvenile blood mercury and male blood mercury. Male blood mercury, juvenile blood mercury, and egg mercury were each separately regressed with female blood mercury to convert all tissues to FLUs. Female adult blood levels were also converted into "male loon units" (MLUs), as male loons on the breeding grounds tend to have higher mercury than females regardless of body weight, presumably due to the depuration of female body mercury into eggs. Presentation of mercury data in FLUs presents a different picture than in MLUs: while FLUs are a more universal unit (since they include egg and juvenile data), they represent the expected or observed blood mercury of adult females. As male mercury exposure is generally higher than for females, even in the same locations and years, examination of the data in the form of MLUs is useful for predicting male exposure in the region.

## CALCULATION OF FLUS AND MLUS FROM ADULT BLOOD MERCURY DATA

We used JMP Version 4.0 (2000) to regress female blood mercury with male blood mercury from the same territories and years (n= 49 complete cases). When we had blood samples from more than one individual from the same sex from a single territory and year, we referenced the original notes from loon surveys to ensure that we used only the two birds that constituted the territorial pair at that site (each bird is individually banded with a unique ID number, so they can be readily distinguished). FLUs are the best estimate of female mercury exposure for specific territories for which we have data. If we have a female blood mercury result for a given territory and year, this blood mercury value is used as the "FLU" for that location/year. If we only have a male blood mercury value for that area, then we use the models developed below to convert it to an estimated female blood level. Body weight was originally intended for inclusion in the model, but had no significant relationship with mercury for either sex. We suspect that mercury may vary more with methylmercury availability between lakes than with body weight of individuals. As such, mass was not included in the models. Male and female blood mercury levels, like most

contaminant data, are highly right-skewed and are not normally distributed. We used a natural log transformation on blood mercury levels to normalize the data.

In the case of mercury in the blood of male and female adult loons, there is no clear dependent relationship between the two variables. For this reason, we determined that orthogonal regression with an assumption of equal variances was the best regression method. Using an assumption of equal variance allows for the minimization of residuals in both directions, rather than strictly along the x-axis. It also allows for prediction of both x and y values, unlike ordinary least squares regression (Freund et al. 2003). When examining all 49 cases together, the measure of linear correlation between the two variables (the Pearson correlation coefficient, *r*) for an orthogonal regression with equal variances was 0.78 (Equation 1). The regression between male and female (natural log-transformed) blood mercury values is shown in Figure B-1 and presented in Equations 1 and 2, along with the 95% confidence limits (CLs) for the models. ♂Hg represents male blood mercury values, and ♀Hg is female blood mercury:

**FLUs (Equation 1)**

$FLU = ♀Hg = e^{-0.64939+1.354771 \cdot \ln(♂Hg)}$ 95% *CLs*: 1.073-1.739 $r = 0.78$

**MLUs (Equation 2)**

$MLU = ♂Hg = e^{0.479338+0.738132 \cdot \ln(♀Hg)}$ 95% *CLs*: 0.574-0.932 $r = 0.78$

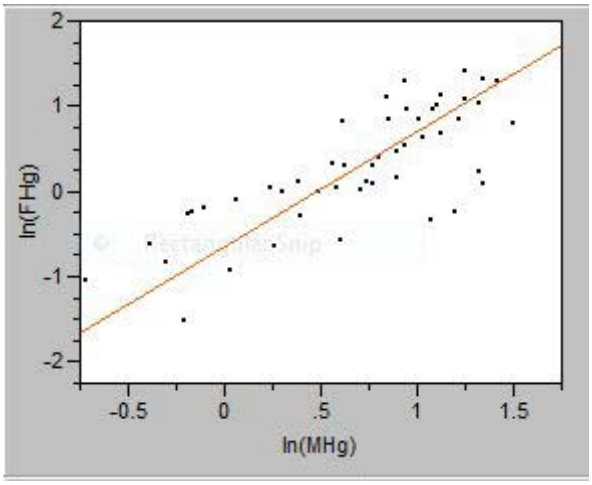


**Figure B-1. Regression of log-transformed male and female loon blood Hg values from the same territories and years (n= 49; r=0.78).**

When the above models were applied to the full adult blood mercury dataset of 636 records, the models described the observed data well, with *r*=0.88 for a regression of the natural log-transformed FLU and MLU values. The full dataset includes, as mentioned above, the observed blood mercury values for males and females when available, and estimated values based on the above models where samples from one or the other sex were not taken.

**CALCULATION OF FLUS FROM JUVENILE BLOOD MERCURY DATA**

Using a large dataset of known-age juvenile common loons from states and provinces throughout northern North America, BRI has established a strong relationship ($R^2$=0.95) between chick age and body weight (Figure 2). We used this relationship to assign an estimated age, in days, to all juveniles of known weight in the dataset. Chicks were split into three age categories, based on biological differences in mercury exposure and depuration of mercury (Evers et al. 2010):

<4 weeks old: Mostly down- covered, and on average eating smaller prey of a lower trophic level than older chicks or adult females.

4-6 weeks old: Chicks are molting in feathers and depurating the majority of mercury body burden into feathers; are also more likely to be eating fish of similar species and size classes to adult females on the same lakes.

>6 weeks old: Juveniles are finished with at least the majority of molt and begin foraging more independently (although still fed by parents 50% of time by week 8).

Thus, for FLU modeling purposes, we separately regressed female blood mercury with mercury in the blood of young chicks (<4 weeks old), older chicks (4-6 weeks) and oldest chicks (>6 weeks), as we expect these relationships to vary.

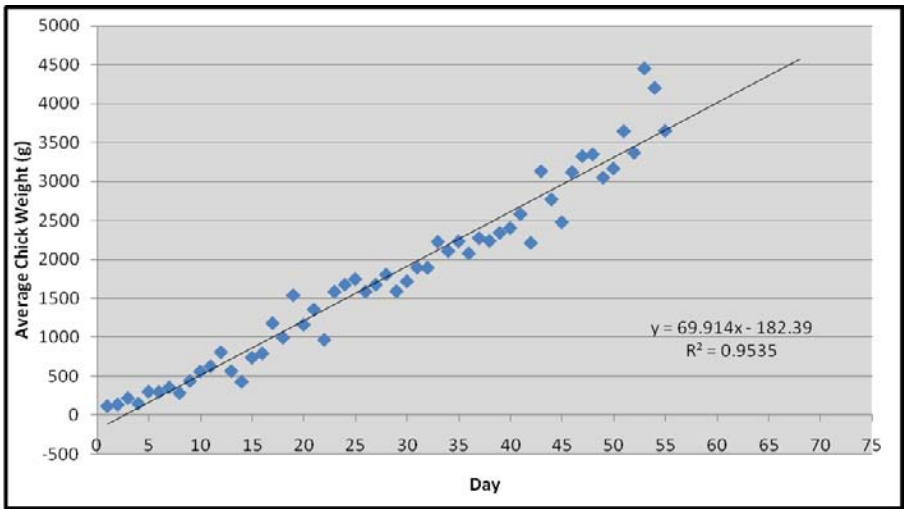


**Figure B-2. Relationship between chick age and body weight. Based on n=376 known-age chicks from across North America.**

Within about three weeks of hatching, loon chick blood mercury levels become independent of maternal transfer of methylmercury (Kenow et al. 2003). Although we could potentially use an ordinary least squares regression model for the youngest chick age group for this reason, using an assumption of equal variance allows for the minimization of residuals in both directions, rather than strictly along the x-axis (which is valuable in this case, where both female and chick mercury levels are highly variable). It also allows for prediction of both x and y values, unlike ordinary

least squares regression (Freund et al. 2003). Thus, we again used orthogonal regression with an assumption of equal variances to model chick blood mercury levels with adult female blood levels from the same territories and years (Equations 3-4). For territories where we had mercury values for the female and both her chicks, and the chicks fell into the same age group, we averaged the chick mercury values prior to regression. For loon families with chicks whose body weights placed them in different age classes, we used only the older (larger) chicks in regression models in these cases. Both female and chick mercury values were natural log transformed and regressed (n=18 complete cases for young chicks; n=10 for 4-6 week chicks; and n=5 for older chicks). There were too few older chicks to develop a reliable regression model with female blood mercury for New York lakes. As discussed in Evers et al. (2011), however, there is no clear relationship between female loon mercury and mercury in chicks of this age group even when assessed using data from five Great Lakes states and provinces. Due to these facts, we excluded chicks >6 weeks from further modeling efforts. One record each was excluded as an outlier from the <4 week and 4-6 week regression models.

**FLUs from <4 week old juvenile blood values (Equation 3)**

$FLU = e^{1.117769+0.441887*ln(\text{JuvHg})}$ 95% *CLs*: 0.324-0.571 $r = 0.76$

**FLUs from 4-6 week old juvenile blood values (Equation 4)**

$FLU = e^{1.818148+0.752218*ln(\text{JuvHg})}$ 95% *CLs*: 0.568-0.976 $r = 0.72$

## CALCULATION OF FLUS FROM EGG MERCURY DATA

Egg mercury levels were related to female levels based on the conversion established in Evers et al. (2003), which used 108 records with both female blood mercury and egg mercury (1988-2001) to establish a ratio between egg mercury (wet weight) and female blood mercury. Samples were from eight states in North America, including Maine, New Hampshire, Vermont, and states in the Midwest and western U.S. Eggs were either analyzed for wet weight mercury, or for dry weight mercury with an associated estimate of moisture loss. Dry weight measurements were converted to wet weight prior to modeling using the percentage of moisture in the samples. The relationship between female blood mercury and eggs from the same territories was highly correlated (Equation 5).

**FLUs from egg mercury values (Equation 5)**

$FLU = 0.2238 + 1.5544 * EggHg$ $r^2 = 0.79$

As stated in Evers et al. (2003), variability in this relationship is likely due to: (1) differences in egg mercury concentrations within a clutch, as there was found to be a mean 25% difference in mercury between eggs collected from the same nests; and (2) differences in the mercury exposure of female loons laying the eggs (and depurating part of their body burden of mercury into the eggs).

## COMMON LOONS IN THE ADIRONDACK REGION OF NEW YORK STATE

### RESULTS

We applied the above regression models to 381 records of common loon blood and eggs from 100 lakes in New York state (1998-2008; Biodiversity Research Institute and its collaborators). Average FLUs and MLUs were calculated for each territory using all available data from that territory, across all years.

The average FLU value for the Adirondack region is 1.34 µg/g wet weight, over all lakes and years (±1.00). Seven percent of FLUs are above the LOAEL (lowest observed adverse effect level) of 3.0 µg/g wet weight in blood, at which adult loons demonstrate physiological, behavioral, and reproductive impacts, including 41% fewer fledged young than birds with blood mercury values below 1.0 µg/g (Evers et al. 2008). Over half (56%) of FLU values were above 1.0 µg/g wet weight. The average MLU over all territories and years is 2.13 µg/g wet weight (±1.03), and 20% of MLU values are above this 3.0 µg/g LOAEL threshold.

**Table B-1. FLU summary statistics for the Adirondack region of New York, 1998-2008 (260 territories and years).**

| Year Range | # of Samples | # of Lakes Represented | Mean | Standard Deviation | Minimum | Maximum |
|---|---|---|---|---|---|---|
| 1998-2000 | 94 | 45 | 1.32 | 1.1 | 0.11 | 5.08 |
| 2001-2004 | 169 | 72 | 1.43 | 0.96 | 0.1 | 5.87 |
| 2005-2008 | 102 | 60 | 1.21 | 0.96 | 0.11 | 5.69 |
| **All Years** | **365** | **101** | **1.34** | **1** | **0.1** | **5.87** |

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## 9.0 APPENDIX C. LAKE CHARACTERISTICS OF EACH ADIRONDACK PARK STUDY LAKE

| Lake | ALSC Pond# | Latitude | Longitude | County | Town | Other Studies |
|---|---|---|---|---|---|---|
| Abanakee | 050587B | 43.766 | -74.255 | Hamilton | Indian Lake | |
| Arbutus | 50684 | 44.014 | -74.243 | Essex | Newcomb | EMAP, LTM |
| Beaver | 40449 | 43.875 | -75.163 | Lewis | Watson | ALSC |
| Big Moose | 40752 | 43.836 | -74.852 | Herkimer | Webb | LTM, AEAP, Hg |
| Canada | 70717 | 43.170 | -74.512 | Fulton | Caroga | EMAP, Hg |
| Cedar River | 50667 | 43.704 | -74.479 | Hamilton | Lake Pleasant | |
| Chaumont | 40303 | 44.212 | -74.952 | St Lawrence | Clifton | ALSC |
| Clear | 030085A | 44.588 | -74.286 | Franklin | Duane | |
| Cranberry | 40309 | 44.171 | -74.818 | St Lawrence | Clifton and Colton | Hg |
| Dry Channel | 30128 | 44.351 | -74.438 | Franklin | Santa Clara | ALSC, EMAP |
| Durant | 050645A | 43.844 | -74.409 | Hamilton | Indian Lake | |
| East Pine | 20147 | 44.339 | -74.419 | Franklin | Altamont | ALSC |
| Ferris | 70777 | 43.306 | -74.632 | Hamilton | Arietta | ALSC, Hg |
| G | 70859 | 43.415 | -74.633 | Hamilton | Arietta | LTM, AEAP |
| Garnet | 50520 | 43.518 | -74.025 | Warren | Johnsburg & Thurman | EMAP |
| Henderson | 50715 | 44.090 | -74.067 | Essex | Newcomb | EMAP |
| Hitchins | 60144 | 44.109 | -74.666 | St Lawrence | Colton | ALSC, EMAP |
| Horseshoe | 60143 | 44.129 | -74.623 | St Lawrence | Piercefield | |
| Kushaqua | 20055 | 44.520 | -74.114 | Franklin | Franklin | ALSC, EMAP |
| Limekiln | 40826 | 43.714 | -74.795 | Herkimer | Ohio | ALSC, LTM, AEAP, Hg |
| Little Clear | 20191 | 44.362 | -74.285 | Franklin | Santa Clara | ALSC |
| Long | 20149 | 43.759 | -74.967 | Franklin | Santa Clara | ALSC |
| Lows | 60156 | 44.360 | -74.392 | St Lawrence | Colton | |
| Mason | 50613 | 44.081 | -74.780 | Hamilton | Lake Pleasant | ALSC |
| Massawepie | 30369 | 43.598 | -74.423 | St Lawrence | Piercefield | ALSC, Hg |
| Middle Saranac | 20110 | 44.254 | -74.648 | Franklin | Harrietstown & Santa Clara | |
| Moshier | 40478 | 44.261 | -74.267 | Herkimer | Webb | ALSC, Hg |
| Moss | 40746 | 43.892 | -75.100 | Herkimer | Webb | ALSC, LTM, AEAP |
| Newton Falls | 040301A | 43.786 | -74.851 | St Lawrence | Clifton | |
| Nicks | 40804 | 44.219 | -74.981 | Herkimer | Webb | ALSC |
| North | 41007 | 43.681 | -74.994 | Herkimer | Ohio | ALSC, EMAP, LTM, AEAP, Hg |
| Piercefield | 60085 | 43.536 | -74.931 | St Lawrence and Franklin | Piercefield and Altamont | |
| Piseco-Big Bay | 50234 | 44.226 | -74.527 | Hamilton | Arietta | EMAP |
| Private #1 | 50235 | 43.381 | -74.566 | Hamilton | Arietta | |
| Round | 60118 | 43.373 | -74.622 | Hamilton | Long Lake | |
| Safford | 40735 | 44.073 | -74.586 | Herkimer | Webb | ALSC |
| Seventh | 040787B | 43.752 | -74.725 | Hamilton | Inlet | AEAP |
| South Lake | 41004 | 43.512 | 74.896 | Herkimer | Ohio | EMAP, LTM, AEAP |
| South Pond | 60245 | 43.915 | -74.453 | Hamilton | Long Lake | |
| Spitfire/Upper St. Regis | 30264 | 44.417 | -74.273 | Franklin | Brighton | ALSC, EMAP |
| | | | | | | |

| Lake | ALSC Pond# | Latitude | Longitude | County | Town | Other Studies |
|---|---|---|---|---|---|---|
| Squaw | 40850 | 43.634 | -74.739 | Hamilton | Morehouse | EMAP, LTM, ALSC, AEAP |
| Taylor | 20227 | 44.483 | -73.865 | Clinton | Black Brook | DEC Hg |
| Wolf | 50688 | 44.021 | -74.220 | Essex | Newcomb | |
| Woodruff | 50681 | 43.964 | -74.147 | Essex | Newcomb | |

## 10.0 APPENDIX D. SAMPLES COLLECTED FROM EACH STUDY LAKE IN THE ADIRONDACK PARK.

| Lake | 2003 -2004 | | | | | 1998 - 2007 | | |
|---|---|---|---|---|---|---|---|---|
| | Water | Sediment | Zooplankton | Crayfish | Fish | Loon Egg | Loon Blood | Loon Feather |
| Abanakee | x | x | x | x | x | x | x | x |
| Arbutus | x | x | x | x | x | x | x | x |
| Beaver | x | x | x | x | x | x | x | x |
| Big Moose | x | x | x | x | x | x | x | x |
| Canada | x | | x | | x | x | x | x |
| Cedar River | x | x | | x | x | | x | x |
| Chaumont | x | | x | | x | | x | x |
| Clear | x | x | x | | x | | x | x |
| Cranberry | x | | x | | x | x | x | x |
| Dry Channel | x | | x | | x | | x | x |
| Durant | x | x | x | x | x | x | x | x |
| East Pine | x | x | x | | x | | x | x |
| Ferris | x | x | x | x | x | x | x | x |
| G | x | x | x | x | x | x | x | |
| Garnet | x | x | x | x | x | | x | x |
| Henderson | x | x | x | x | x | | x | |
| Hitchins | x | x | x | | x | x | x | x |
| Horseshoe | x | x | x | x | x | x | x | |
| Kushaqua | x | x | x | x | x | x | x | x |
| Limekiln | x | | x | | x | x | x | x |
| Little Clear | x | x | x | x | x | x | x | x |
| Long | x | x | x | | x | | x | x |
| Lows | x | | x | | x | x | x | x |
| Mason | x | x | x | x | x | x | x | x |
| Massawepie | x | | x | | x | | x | x |
| Middle Saranac | x | x | x | x | x | | x | x |
| Moshier | x | | x | | x | x | x | x |
| Moss | x | x | x | x | x | x | x | x |
| Newton Falls | x | x | x | x | x | x | x | x |
| Nicks | x | | x | | x | | x | x |
| North | x | x | x | x | x | x | x | x |
| Piercefield | x | x | x | | x | | x | x |
| Piseco-Big Bay | x | x | x | x | x | x | x | x |
| Private #1 | x | x | x | x | x | x | x | x |
| Round | x | x | x | x | x | | x | |
| Safford | x | | x | | x | | x | x |
| Seventh | x | | x | | x | x | x | x |

| Lake | 2003 -2004 | | | | | 1998 - 2007 | | |
|---|---|---|---|---|---|---|---|---|
| | **Water** | **Sediment** | **Zooplankton** | **Crayfish** | **Fish** | **Loon Egg** | **Loon Blood** | **Loon Feather** |
| South Lake | x | x | x | x | x | x | x | x |
| South Pond | x | x | x | x | x | x | x | x |
| Spitfire/Upper St. Regis | x | x | x | | x | x | x | x |
| Squaw | x | | x | | x | x | x | x |
| Taylor | x | x | x | x | x | x | x | x |
| Wolf | x | x | x | x | x | x | x | x |
| Woodruff | x | x | x | x | x | | x | x |
| ***Total Number of Lakes:*** | **44** | **32** | **43** | **26** | **42** | **29** | **44** | **40** |

## 11.0 APPENDIX E. WATER CHEMISTRY FOR EACH ADIRONDACK PARK STUDY LAKE

| Watershed | Lake | pH | DIC | DOC | ANC | NH4 | Si | Total P | Chloro | K | Na | Ca | Mg | Fl | Cl | NO3 | S04 | Al | Al o |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Black | Beaver | 6.17 | 88.57 | 4.27 | 40.08 | 1.31 | 66.17 | 21.06 | 0.7 | 0.63 | 0.69 | 1.80 | 0.35 | 5.35 | 7.11 | 10.15 | 37.51 | 0.72 | 1.11 |
| | Big Moose | 5.29 | 56.97 | 2.52 | 12.55 | 1.48 | 20.18 | 1.93 | 1.4 | 0.51 | 0.53 | 1.41 | 0.26 | 3.66 | 6.26 | 1.10 | 40.50 | 0.70 | 0.92 |
| | Limekiln | 6.39 | 70.20 | 2.32 | 38.71 | 0.87 | 45.78 | | 0.8 | 0.53 | 0.74 | 2.88 | 0.35 | 3.25 | 14.50 | 9.42 | 41.61 | 0.41 | 0.72 |
| | Little Safford | 5.60 | 121.92 | 8.43 | 18.32 | 0.99 | 82.94 | 2.34 | 2.5 | 0.24 | 0.63 | 1.75 | 0.33 | 3.93 | 5.35 | 0.51 | 32.58 | 2.70 | 2.10 |
| | Moshier | 6.02 | 71.67 | 4.22 | 21.76 | 1.10 | 67.85 | 3.99 | 0.9 | 0.61 | 0.64 | 1.89 | 0.33 | 6.10 | 8.37 | 10.09 | 37.98 | 0.69 | 0.95 |
| | Moss | 6.60 | 129.54 | 4.44 | 85.97 | 1.53 | 100.86 | 0.87 | 1.9 | 0.64 | 0.92 | 2.58 | 0.50 | 7.02 | 9.62 | 4.67 | 45.37 | 0.69 | 0.70 |
| | Nicks | 7.00 | 200.51 | 4.31 | 141.55 | 1.33 | 52.19 | 2.48 | 3.3 | 0.38 | 0.73 | 2.96 | 0.66 | 6.51 | 14.86 | 0.00 | 40.18 | 0.34 | 0.51 |
| | North | 5.26 | 72.79 | 6.04 | 9.74 | 1.69 | 74.21 | 3.07 | 0.3 | 0.31 | 0.56 | 1.84 | 0.28 | 2.34 | 6.15 | 4.35 | 35.57 | 3.42 | 2.24 |
| | Seventh | 7.08 | 255.72 | 4.19 | 188.60 | 0.97 | 71.56 | | 0.9 | 0.43 | 5.60 | 4.38 | 0.81 | 3.15 | 237.48 | 8.43 | 50.10 | 0.42 | 0.60 |
| | South Lake | 5.70 | 70.77 | 2.60 | 12.65 | 1.60 | 49.65 | 0.71 | 1.1 | 0.46 | 0.54 | 1.42 | 0.42 | 2.42 | 7.33 | 12.79 | 37.86 | 0.92 | 0.67 |
| | Squaw | 5.95 | 64.18 | 3.60 | 21.03 | 0.86 | 26.13 | 0.09 | 1.4 | 0.57 | 0.42 | 1.46 | 0.35 | 2.40 | 8.47 | 4.94 | 37.36 | 0.57 | 0.97 |
| Champlain | East Pine | 6.90 | 251.99 | 3.38 | 163.57 | 1.28 | 122.32 | 1.77 | 5.6 | 0.60 | 0.94 | 2.97 | 0.76 | 1.81 | 11.85 | 0.00 | 34.24 | 0.36 | 0.71 |
| | Kushaqua | 7.39 | 403.53 | 5.81 | 331.49 | 1.48 | 106.26 | | 7.2 | 0.58 | 1.21 | 5.09 | 1.46 | 7.17 | 17.17 | 0.02 | 39.05 | 0.38 | 0.74 |
| | Little Clear | 7.26 | 32.07 | 1.55 | 250.13 | 2.04 | 126.56 | | 2.5 | 0.35 | 1.35 | 1.67 | 1.27 | 2.02 | 10.90 | 0.11 | 49.99 | 0.38 | 0.80 |
| | Long | 7.82 | 114.33 | 3.68 | 105.73 | 1.18 | 69.65 | 4.87 | 4.7 | 0.25 | 0.86 | 2.05 | 0.52 | 1.40 | 7.67 | 0.44 | 39.99 | 0.56 | 0.43 |
| | Middle Saranac | 6.96 | 192.36 | 3.94 | 158.97 | 1.87 | 85.29 | 1.04 | 1.6 | 0.40 | 4.27 | 3.71 | 0.92 | 1.86 | 181.51 | 0.16 | 46.99 | 0.41 | 0.49 |
| | Taylor | 7.10 | 237.17 | 3.28 | 162.74 | 1.30 | 55.62 | 74.88 | 1.7 | 0.67 | 1.12 | 3.40 | 0.97 | 3.15 | 19.76 | 0.19 | 50.55 | 0.33 | 0.54 |
| Mohawk | Canada | 6.57 | 118.16 | 202.06 | 58.70 | 2.25 | 50.05 | 2.19 | 0.9 | 0.25 | 4.12 | 2.21 | 0.48 | 1.67 | 189.78 | 5.00 | 43.08 | 1.17 | 1.02 |
| | Ferris | 5.94 | 78.19 | 362.01 | 31.98 | 0.93 | 21.93 | 4.51 | 1.6 | 0.17 | 0.45 | 1.10 | 0.32 | 1.34 | 8.40 | 0.73 | 35.03 | 1.72 | 1.36 |
| | G | 6.18 | 69.08 | 176.39 | 23.79 | 2.93 | 43.74 | 1.54 | 1 | 0.17 | 0.52 | 1.16 | 0.31 | 1.99 | 8.62 | 2.51 | 37.51 | 1.06 | 0.94 |
| Oswegatchie | Chaumont | 6.78 | 186.48 | 375.24 | 118.97 | 3.04 | 66.29 | 1.89 | 2.2 | 0.47 | 1.30 | 3.40 | 0.65 | 5.15 | 30.68 | 3.28 | 64.14 | 0.94 | 0.89 |
| | Cranberry | 6.57 | 109.12 | 345.95 | 64.40 | 1.93 | 69.16 | 0.04 | 2.3 | 0.45 | 0.91 | 2.05 | 0.45 | 3.96 | 12.72 | 5.37 | 44.87 | 0.99 | 0.90 |
| | Newton Falls | 6.83 | 173.36 | 5.33 | 116.16 | 1.48 | 81.13 | | 15.7 | 0.39 | 1.73 | 4.04 | 0.76 | 6.60 | 64.09 | 3.87 | 73.28 | 0.42 | 0.77 |

| Watershed | Lake | pH | DIC | DOC | ANC | NH4 | Si | Total P | Chloro | K | Na | Ca | Mg | Fl | Cl | NO3 | S04 | Al | Al o |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| St. Lawrence | Clear | 5.67 | 32.07 | 2.07 | 8.28 | 0.47 | 1.59 | | 1.2 | 0.35 | | 1.67 | 1.27 | 0.95 | 589.53 | 0.03 | 25.11 | 0.33 | 0.45 |
| | Dry Channel | 5.74 | 81.30 | 3.70 | 20.36 | 3.23 | 10.10 | | 2 | 0.14 | 0.51 | 1.01 | 0.20 | 1.40 | 6.27 | 0.11 | 33.82 | 0.79 | 0.65 |
| | Hitchins | 6.46 | 167.02 | 4.52 | 79.09 | 4.26 | 60.73 | | 2.3 | 0.56 | 0.42 | 2.11 | 0.48 | 4.09 | 6.58 | 1.91 | 34.55 | 0.42 | 0.77 |
| | Horseshoe | 6.56 | 205.89 | 4.56 | 261.87 | 0.16 | 92.36 | 2.66 | 3.1 | 0.32 | 1.42 | 2.48 | 0.66 | 4.73 | 37.96 | 0.00 | 36.42 | 0.39 | 0.84 |
| | Lows | 6.55 | 116.63 | 371.26 | 67.17 | 2.24 | 22.49 | 4.34 | 6.6 | 0.36 | 0.64 | 1.98 | 0.42 | 3.87 | 8.71 | 1.33 | 38.68 | 1.02 | 0.93 |
| | Massawepie | 7.19 | 317.83 | 311.97 | 235.69 | 1.67 | 129.17 | 0.19 | 2.2 | 0.46 | 2.59 | 3.98 | 1.14 | 2.85 | 109.34 | 1.05 | 46.00 | 0.88 | 0.84 |
| | Piercefield | 6.77 | 88.42 | 6.23 | 87.07 | 1.35 | 50.07 | 6.00 | 0.9 | 0.46 | 2.06 | 2.27 | 0.52 | 2.66 | 65.74 | 3.50 | 39.90 | 0.70 | 0.57 |
| | Round | 6.74 | 108.15 | 6.68 | 85.59 | 2.25 | 69.86 | 1.85 | 3.1 | 0.56 | 0.30 | 2.65 | 0.56 | 4.49 | 30.05 | 0.25 | 39.99 | 0.50 | 0.71 |
| | South Pond | 5.99 | 66.86 | 3.68 | 16.12 | 0.77 | 88.40 | 0.18 | 0.75 | 0.35 | 3.96 | 1.90 | 0.43 | 5.01 | 160.95 | 4.64 | 44.75 | 0.71 | 0.76 |
| | Spitfire | 7.21 | 244.49 | 2.84 | 176.08 | 1.03 | 87.79 | 1.60 | 2.3 | 0.72 | 4.36 | 3.36 | 0.91 | 1.83 | 166.53 | 0.05 | 38.34 | 0.34 | 0.58 |
| Upper Hudson | Abanakee | 6.75 | 177.45 | 261.34 | 106.56 | 2.76 | 87.34 | 0.00 | 2 | 0.32 | 2.32 | 2.77 | 0.64 | 2.11 | 108.36 | 0.74 | 39.02 | 0.92 | 0.85 |
| | Arbutus | 6.72 | 118.79 | 4.14 | 72.52 | 0.51 | 43.40 | 2.56 | 1 | 0.52 | 0.77 | 2.78 | 0.50 | 5.98 | 9.43 | 0.37 | 52.71 | 0.64 | 0.63 |
| | Cedar River | 6.87 | 177.09 | 3.91 | 110.15 | 2.32 | 46.08 | 4.29 | 3.4 | 0.62 | 0.89 | 2.38 | 0.70 | 5.84 | 6.63 | 0.06 | 39.76 | 0.41 | 0.81 |
| | Durant | 6.71 | 168.02 | 6.84 | 102.88 | 3.77 | 19.67 | 4.69 | 1.8 | 0.58 | 6.03 | 1.29 | 0.67 | 3.94 | 320.43 | 0.82 | 36.98 | 0.49 | 0.72 |
| | Garnet | 7.09 | 286.83 | 313.58 | 203.85 | 2.71 | 41.98 | 4.07 | 7.8 | 0.17 | 0.76 | 3.31 | 1.03 | 1.90 | 11.50 | 0.75 | 36.91 | 0.91 | 0.84 |
| | Henderson | 6.26 | 65.24 | 2.68 | 30.76 | 1.88 | 90.78 | | 1 | 0.36 | 0.87 | 1.78 | 0.23 | 2.72 | 5.35 | 13.83 | 39.77 | 0.78 | 1.04 |
| | Mason | 6.94 | 269.07 | 273.98 | 172.15 | 3.01 | 28.44 | 2.62 | 2.4 | 0.54 | 13.73 | 6.29 | 1.23 | 1.71 | | 0.74 | 39.02 | 0.89 | 0.92 |
| | Piseco | 5.60 | 153.97 | 3.65 | 101.09 | 2.23 | 50.95 | 3.81 | 1.5 | 0.25 | 3.43 | 2.10 | 0.71 | 2.16 | 153.06 | 3.86 | 37.52 | 0.60 | 0.49 |
| | Private #1 | 6.18 | 120.34 | 502.02 | 55.62 | 3.12 | 31.00 | 0.00 | 3.7 | 0.17 | 0.60 | 1.43 | 0.37 | 1.81 | 7.11 | 5.37 | 44.87 | 1.52 | 1.36 |
| | Wolf | 6.93 | 106.12 | 2.98 | 85.90 | 1.48 | 46.53 | 0.47 | 0.3 | 0.20 | 1.41 | 2.19 | 0.41 | 2.94 | 7.86 | 4.59 | 47.12 | 0.55 | 0.48 |
| | Woodruff | 7.52 | 243.04 | 9.11 | 202.02 | 1.91 | 51.31 | 6.29 | 8.72 | 0.42 | 1.81 | 3.92 | 0.86 | 3.98 | 40.98 | 0.14 | 30.48 | 0.39 | 0.69 |

## 12.0 APPENDIX F. LAKE SPECIFIC MERCURY VALUES

**Table F-1. Abiotic (water and sediment) average total mercury (THg) and methylmercury (MeHg) for each study lake.**

| Watershed | Lake | Water THg (ppt) | Water MeHg (ppt) [a] | Sediment THg (ppt) | Sediment MeHg (ppt) [b] |
|---|---|---|---|---|---|
| Black | Beaver | 1.75 | 0.08 | 2261.16 | 9.13 |
| | Big Moose | 1.04 | 0.03 | 13992.08 | 117.18 |
| | Limekiln | 0.56 | 0.01 | | |
| | Little Safford | 3.16 | 0.32 | | |
| | Moshier | 2.11 | 0.12 | | |
| | Moss | 1.75 | | 1407.28 | 5.46 |
| | Nicks | 1.23 | 0.08 | | |
| | North | 3.78 | 0.17 | 9300.57 | 192.55 |
| | Seventh | 1.20 | 0.00 | | |
| | South Lake | 1.12 | 0.01 | 6734.23 | 90.24 |
| | Squaw | 1.80 | 0.16 | | |
| Champlain | East Pine | 3.41 | 0.06 | | |
| | Kushaqua | 1.45 | | 2884.47 | 29.47 |
| | Little Clear | 2.24 | 0.01 | 8203.28 | 1.19 |
| | Long | 4.64 | 0.32 | | |
| | Middle Saranac | 1.39 | 0.06 | 1888.07 | 307.42 |
| | Taylor | 0.71 | 0.07 | 2226.01 | 12.81 |
| Mohawk | Canada | 1.60 | | | |
| | Ferris | 3.18 | 0.30 | 19073.59 | 181.52 |
| | G | 1.82 | | 43267.81 | 1207.75 |
| Oswegatchie | Chaumont | 2.21 | 0.20 | | |
| | Cranberry | 1.04 | | | |
| | Newton Falls | 1.64 | 0.13 | 10860.15 | 174.32 |
| St. Lawrence | Clear | 0.10 | 0.01 | | |
| | Dry Channel | 1.00 | 0.48 | | |
| | Hitchins | 1.54 | 0.18 | | |
| | Horseshoe | 1.19 | 0.04 | 2353.69 | 200.54 |
| | Lows | 2.40 | | | |
| | Massawepie | 1.11 | 0.21 | | |
| | Piercefield | 1.87 | 0.12 | | |
| | Round | 1.89 | 0.02 | 15033.78 | 189.63 |
| | South Pond | 1.34 | 0.04 | 5791.82 | 93.07 |
| | Spitfire | 0.61 | 0.02 | | |
| Upper Hudson | Abanakee | 2.71 | | 32907.26 | |
| | Arbutus | 1.58 | 0.11 | 4063.43 | 80.27 |
| Upper Hudson | Cedar River | 1.47 | 0.08 | 2772.16 | 3.53 |
| | Durant | 2.49 | 0.12 | 7365.43 | 61.47 |
| | Garnet | 1.04 | 0.13 | 31539.45 | 104.48 |
| | Henderson | 1.15 | 0.06 | 33815.40 | 681.49 |
| | Mason | 1.52 | 0.21 | 27388.56 | 1127.79 |
| | Piseco-Big Bay | 1.06 | 0.12 | 7014.08 | 98.03 |
| | Private #1 | 3.02 | 0.03 | 12667.09 | 1235.14 |
| | Wolf | 0.65 | 0.03 | 1972.23 | 41.35 |
| | Woodruff | 1.54 | 0.12 | 83799.30 | 4498.74 |

[a] Results for six lakes, Abanakee, Canada, Cranberry, G, Kushaqua, and Lows, were removed from the analysis because their water MeHg levels were below the detection limit. Moss Lake was removed because MeHg exceeded total Hg

[b] Abanakee was removed from analysis because its sediment MeHg value was below the detection limit.

Table F-2. Invertebrate prey (zooplankton and crayfish) total mercury (THg) and methylmercury (MeHg) in study lakes.

| Watershed | Lake [a] | Zooplankton THg (ppb) [b] | Zooplankton MeHg (ppb) [c] | Crayfish Wholebody THg (ppb) | Crayfish Tail THg (ppb) |
|---|---|---|---|---|---|
| Black | Beaver | 427.12 | 138.01 | 45.04 | 50.22 |
| | Big Moose | 246.91 | 88.52 | 81.00 | 117.90 |
| | Limekiln | 225.52 | 28.43 | | |
| | Little Safford | 520.44 | 59.38 | | |
| | Moshier | 550.51 | 193.49 | | |
| | Moss | 305.09 | 89.47 | 43.33 | 47.81 |
| | Nicks | 203.39 | 25.78 | | |
| | North | 820.25 | 216.69 | 69.97 | 93.70 |
| | Seventh | 175.66 | 29.18 | | |
| | South Lake | 532.75 | 94.03 | 55.27 | 66.18 |
| | Squaw | 244.56 | 182.26 | | |
| Champlain | East Pine | | | | |
| | Kushaqua | 156.13 | 66.37 | 26.49 | 27.96 |
| | Little Clear | 142.37 | 17.73 | 44.53 | 50.39 |
| | Long | 56.38 | 26.13 | | |
| | Middle Saranac | 308.96 | 23.03 | 21.13 | 23.13 |
| | Taylor | 213.00 | 17.39 | 29.89 | 34.91 |
| Mohawk | Canada | 6.58 | | | |
| | Ferris | 627.89 | | 93.78 | 165.25 |
| | G | 268.11 | 172.83 | 23.95 | 25.80 |
| Oswegatchie | Chaumont | | | | |
| | Cranberry | 412.88 | 56.74 | | |
| | Newton Falls | 319.49 | 57.58 | 60.90 | 76.08 |
| St. Lawrence | Clear | 221.19 | 40.74 | | |
| | Dry Channel | 136.12 | 80.60 | | |
| | Hitchins | 446.84 | 157.98 | | |
| | Horseshoe | 98.31 | 55.52 | 23.74 | 25.40 |
| | Lows | 193.59 | 76.97 | | |
| | Massawepie | 235.93 | 43.46 | | |
| | Piercefield | 218.46 | 9.95 | | |
| | Round | 235.77 | 38.93 | 54.23 | 64.44 |
| | South Pond | 337.05 | 40.01 | 44.16 | 48.98 |
| | Spitfire | 173.48 | 6.47 | | |
| Upper Hudson | Abanakee | 340.50 | 15.70 | 47.84 | 61.16 |
| | Arbutus | 186.70 | | 46.31 | 52.76 |
| | Cedar River | | | 25.96 | 27.46 |
| Upper Hudson | Durant | 229.38 | 3.55 | 41.57 | 45.43 |
| | Garnet | 310.44 | 65.91 | 28.78 | 31.06 |
| | Henderson | 393.22 | 161.07 | 68.74 | 91.18 |
| | Mason | 495.51 | 53.72 | 58.08 | 71.03 |
| | Piseco-Big Bay | 213.10 | | 21.10 | 23.11 |
| | Private #1 | | | 79.04 | 118.47 |
| | Wolf | 271.08 | 39.51 | 38.76 | 42.63 |
| | Woodruff | 655.24 | 55.43 | 37.39 | 40.00 |

[a] Due to limited depth at Cedar River Flow, zooplankton sampling was not possible.

[b] Hg levels for Chaumont Pond, Private Lake #1, and East Pine were excluded because values were outside normal ranges and therefore suspected of being in error.

[c] Canada Lake MeHg was below the detection limit. Ferris, Arbutus, and Piseco-Big Bay MeHg were not reported

**Table F-3. Fish species captured at each study lake.**

| Lake | Yellow Perch | Banded Killifish | Blacknose Dace | Brown Bullhead | Brown Trout | Common Shiner | Creek Chub | Fallfish | Golden Shiner | Hornyhead Chub | Lake Chub | Largemouth Bass | N. Redbelly Dace | Pumpkinseed | Rainbow Smelt | Red-ear Sunfish | Rock Bass | Smallmouth Bass | Walleye | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Abanakee | 3 | | | | | | | | | | | | | 1 | | | | | | 4 |
| Arbutus | | | | 7 | | | | | | | | | | | | | | | | 7 |
| Beaver | 9 | | | | | | | | | | | | | | | | | | | 9 |
| Big Moose | 1 | | | | | | | | | | | 1 | | 2 | | | | | | 4 |
| Canada | 3 | | | | | | | 1 | | | | | | | | | | | | 4 |
| Cedar River | | | | | | | 1 | | | 2 | | | | | | | | | | 3 |
| Chaumont | 3 | | | | | | | | | | | | | 1 | | | 1 | | | 5 |
| Clear Pond | | | | | | | | | | | | | | 1 | | | | | | 1 |
| Cranberry | | | | | | | | | | | | 1 | | 2 | | | | 2 | | 5 |
| Dry Channel | 2 | | | | | | | | | | | 1 | | 3 | | | | | | 6 |
| Durant | 3 | 1 | | | | | | | | | | 1 | | 1 | | | | | | 6 |
| East Pine | 4 | | | | | | | | | | | | | | | | | | | 4 |
| Ferris | 3 | | | | | | | | | | | | | 1 | | | | | | 4 |
| G | | | | 3 | | | | | 1 | | | | | | | | | | | 4 |
| Garnet | 3 | | | | | | | | | | | | | | | | | 1 | | 4 |
| Henderson | 3 | | | | | | | | | | 2 | | | | | | | | | 5 |
| Hitchins | 3 | | | | | | | | | | | 1 | | | | | | | | 4 |
| Horseshoe | 5 | | | | | | | | | | | | | | | | | | | 5 |
| Kushaqua | 5 | | | | | | | | | | | | | | | | | | | 5 |
| Limekiln | 3 | 1 | | | | | | | | | | | | | | | | | | 4 |
| Little Clear | | | | | 1 | | | | 2 | | | | | 2 | 2 | | | | | 7 |
| Little Safford | | | | | | | | | 2 | | | 1 | | | | | | | | 3 |
| Long | 3 | | | | | | | 1 | | | | | | 2 | | | | | | 6 |
| Lows | | | | | | | | | | | | 4 | | | | | | | | 4 |
| Mason | 1 | | | | | | | | | | | 2 | | | | | | 1 | | 4 |
| Massawepie | | | | | | | | | | | | | | | | | 1 | 3 | | 4 |
| Middle Saranac | 3 | 1 | | | | | | | | | | | | | | | | | | 4 |
| Moshier | | | | | | | | | | | | | | 2 | | | 2 | 2 | | 6 |
| Moss | 3 | | | | | 1 | | | | | | 1 | | | | | | | | 5 |

| Lake | Yellow Perch | Banded Killifish | Blacknose Dace | Brown Bullhead | Brown Trout | Common Shiner | Creek Chub | Fallfish | Golden Shiner | Hornyhead Chub | Lake Chub | Largemouth Bass | N. Redbelly Dace | Pumpkinseed | Rainbow Smelt | Red-ear Sunfish | Rock Bass | Smallmouth Bass | Walleye | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Newton Falls | 4 | 1 | | | | | | | | | | | | | | | | | | 5 |
| Nicks | 1 | | | | | | | | | | | 4 | | 2 | | | | | | 7 |
| North | 3 | | | | | | 2 | | 1 | | | | | | | | | | | 6 |
| Piercefield | 2 | | | | | | | | | | | | | 2 | | 3 | | | 1 | 8 |
| Piseco-Big Bay | 1 | | | | | | | | | | | | | 1 | | | | | | 2 |
| Private #1 | | | | | | | | | | | | | | 3 | | | | | | 3 |
| Round | | 1 | | | | | | | | | | | | | | | | 1 | | 2 |
| Seventh | | | | | | | | | | | | | | 2 | | | 2 | 1 | | 5 |
| South Lake | | | | 2 | 1 | | 3 | | | | | | | | | | | | | 6 |
| South Pond | 4 | | 1 | | | | | | | | | | | | | | | | | 5 |
| Spitfire | 2 | | | | | | | | | | | | | | | | | 2 | | 4 |
| Squaw | | | | | 1 | | 1 | | | | | | 1 | | | | | | | 3 |
| Taylor | 4 | 1 | | | | | | | | | | | | | | | | | | 5 |
| Wolf | | | | | | | | | | | | | | 3 | | 3 | | | | 6 |
| Woodruff | 3 | | | | | | | | | | | | | 1 | | | | | | 4 |
| **Total** | **90** | **6** | **1** | **12** | **3** | **1** | **7** | **2** | **6** | **2** | **2** | **17** | **1** | **33** | **2** | **6** | **6** | **13** | **1** | **211** |
| **% of lakes this species was found in** | **64** | **14** | **2** | **7** | **7** | **2** | **9** | **5** | **9** | **2** | **2** | **23** | **2** | **41** | **2** | **5** | **9** | **18** | **2** | **100** |

**Table F-4. Fish total mercury in each size class.**

| Watershed | Lake | Small | Medium | Large | Extra Large | YPE |
|---|---|---|---|---|---|---|
| Black | Beaver | 0.08 | 0.18 | 0.22 | 0.36 | 0.19 |
| | Big Moose | 0.19 | 0.16 | 0.24 | 0.23 | 0.22 |
| | Limekiln | 0.08 | 0.10 | 0.17 | 0.23 | 0.17 |
| | Little Safford | | 0.17 | 0.14 | 0.22 | 0.20 |
| | Moshier | 0.13 | 0.16 | 0.22 | 0.33 | 0.21 |
| | Moss | 0.06 | 0.08 | 0.11 | 0.32 | 0.15 |
| | Nicks | 0.06 | 0.08 | 0.10 | 0.11 | 0.11 |
| | North | 0.22 | 0.27 | 0.22 | | 0.36 |
| | Seventh | 0.06 | 0.10 | 0.15 | 0.15 | 0.13 |
| | South Lake | 0.09 | 0.12 | 0.09 | | 0.37 |
| | Squaw | 0.43 | 0.15 | | 0.37 | 0.52 |
| Champlain | East Pine | 0.09 | 0.08 | 0.11 | 0.27 | 0.14 |
| | Kushaqua | 0.06 | 0.09 | 0.14 | 0.18 | 0.13 |
| | Little Clear | 0.04 | 0.11 | 0.14 | 0.11 | 0.13 |
| | Long | 0.06 | 0.09 | 0.10 | 0.16 | 0.12 |
| | Middle Saranac | 0.04 | 0.04 | 0.05 | 0.08 | 0.06 |
| | Taylor | 0.08 | 0.07 | 0.14 | 0.14 | 0.12 |
| Mohawk | Canada | 0.08 | 0.17 | 0.05 | 0.06 | 0.10 |
| | Ferris | 0.09 | 0.41 | 0.46 | 0.45 | 0.36 |
| | G | 0.12 | 0.09 | 0.09 | 0.10 | 0.27 |
| Oswegatchie | Chaumont | 0.08 | 0.10 | 0.14 | 0.10 | 0.12 |
| | Cranberry | 0.08 | 0.10 | 0.18 | 0.21 | 0.15 |
| | Newton Falls | 0.11 | 0.11 | 0.13 | 0.18 | 0.15 |
| St. Lawrence | Clear | 0.06 | | | | 0.09 |
| | Dry Channel | 0.06 | 0.05 | 0.13 | 0.13 | 0.11 |
| | Hitchins | 0.09 | 0.08 | 0.12 | 0.25 | 0.14 |
| | Horseshoe | 0.04 | 0.06 | 0.14 | 0.20 | 0.13 |
| | Lows | 0.06 | 0.11 | 0.12 | 0.13 | 0.11 |
| | Massawepie | 0.05 | 0.04 | 0.06 | 0.09 | 0.06 |
| | Piercefield | 0.09 | 0.08 | 0.12 | 0.19 | 0.12 |
| | Round | 0.06 | 0.04 | | | 0.03 |
| | South Pond | 0.07 | 0.13 | 0.32 | 0.23 | 0.23 |
| | Spitfire | 0.04 | 0.04 | 0.07 | 0.14 | 0.07 |
| Upper Hudson | Abanakee | 0.05 | 0.07 | 0.09 | 0.13 | 0.09 |
| | Arbutus | 0.07 | 0.10 | 0.12 | 0.09 | 0.29 |
| | Cedar River | 0.05 | 0.03 | 0.04 | | 0.17 |
| | Durant | 0.08 | 0.11 | 0.15 | 0.20 | 0.13 |
| | Garnet | 0.04 | 0.10 | 0.11 | 0.15 | 0.10 |
| Upper Hudson | Henderson | 0.10 | 0.12 | 0.34 | 0.31 | 0.32 |
| | Mason | 0.04 | 0.06 | 0.07 | 0.12 | 0.08 |
| | Piseco-Big Bay | | 0.07 | 0.11 | | 0.10 |
| | Private #1 | 0.07 | 0.08 | 0.09 | | 0.10 |
| | Wolf | 0.13 | 0.15 | 0.24 | | 0.17 |
| | Woodruff | 0.03 | 0.08 | 0.11 | 0.16 | 0.10 |

**Table F-5. Average loon blood, feather, and egg mercury for all samples between 1998 and 2007.**

| | | Loon Blood Mercury | | | Loon Feather Mercury | | | Loon Egg Mercury | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Watershed | Lake | Hg (µg/g) | n | SD | Hg (µg/g) | n | SD | Hg (µg/g) | n | SD |
| Black | Beaver | 3.03 | 7 | 0.97 | 13.57 | 6 | 5.28 | 0.98 | 3 | 0.31 |
| | Big Moose | 2.33 | 2 | 0.63 | 21.27 | 2 | 14.25 | 1.23 | 1 | |
| | Limekiln | 1.35 | 5 | 0.49 | 17.55 | 5 | 8.49 | 0.42 | 3 | 0.14 |
| | Little Safford | 1.02 | 2 | 0.71 | 10.30 | 2 | 0.76 | | | |
| | Moshier | 3.37 | 2 | 0.68 | 15.81 | 2 | 8.86 | 1.82 | 3 | 0.14 |
| | Moss | 2.54 | 3 | 0.90 | 20.01 | 3 | 11.47 | 1.09 | 5 | 0.19 |
| | Nicks | 2.20 | 4 | 0.63 | 15.73 | 4 | 4.17 | 0.63 | 4 | 0.15 |
| | North | 3.41 | 3 | 0.60 | 27.97 | 3 | 14.26 | | | |
| | Seventh | 0.92 | 4 | 0.16 | 13.32 | 4 | 3.38 | 0.48 | 1 | |
| | South Lake | 1.57 | 3 | 0.21 | 12.16 | 3 | 3.22 | 0.89 | 6 | 0.27 |
| | Squaw | 2.51 | 2 | 0.78 | 11.98 | 2 | 8.03 | 0.47 | 3 | 0.24 |
| Champlain | East Pine | 1.75 | 3 | 0.45 | 13.66 | 3 | 3.35 | | | |
| | Kushaqua | 1.08 | 1 | | 14.81 | 1 | | 0.92 | 1 | |
| | Little Clear | 0.69 | 9 | 0.47 | 9.18 | 9 | 3.63 | 0.52 | 3 | 0.05 |
| | Long | 2.28 | 6 | 1.18 | 12.20 | 6 | 6.33 | 0.94 | 2 | 0.19 |
| | Middle Saranac | 1.50 | 3 | 0.57 | 21.20 | 3 | 15.58 | 0.40 | 3 | 0.14 |
| | Taylor | 2.16 | 1 | | 35.26 | 1 | | 0.75 | 3 | 0.24 |
| Mohawk | Canada | 3.17 | 4 | 0.74 | 14.71 | 2 | 6.23 | 0.89 | 1 | |
| | Ferris | 5.62 | 2 | 0.36 | 13.68 | 2 | 0.26 | 1.83 | 3 | 0.52 |
| | G | | | | | | | 0.55 | 1 | |
| Oswegatchie | Chaumont | 2.49 | 2 | 0.64 | 15.65 | 2 | 2.19 | | | |
| | Cranberry | 2.69 | 10 | 0.82 | 16.81 | 10 | 5.67 | 1.04 | 6 | 0.25 |
| | Newton Falls | 2.00 | 3 | 0.74 | 12.52 | 3 | 2.88 | | | |
| St. Lawrence | Clear | 0.74 | 2 | 0.16 | 9.43 | 2 | 3.58 | | | |
| | Dry Channel | 2.83 | 1 | | 19.33 | 1 | | | | |
| | Hitchins Pond | 1.75 | 4 | 0.90 | 13.79 | 4 | 4.96 | 0.77 | 1 | |
| | Horseshoe | | | | | | | 0.50 | 1 | |
| | Lows | 2.13 | 9 | 1.01 | 12.76 | 4 | 4.34 | 0.35 | 1 | |
| | Massawepie | 1.03 | 3 | 0.29 | 14.31 | 3 | 3.63 | | | |
| | Piercefield | 1.34 | 1 | | 18.78 | 1 | | 0.50 | 1 | |
| | Round | 1.25 | 2 | 1.02 | 7.54 | 1 | | | | |
| | South Pond | 2.11 | 1 | | 8.40 | 1 | | 2.15 | 2 | 0.71 |
| | Spitfire | 0.82 | 2 | 0.37 | 7.28 | 2 | 3.89 | 0.61 | 2 | 0.41 |
| Upper Hudson | Abanakee | 0.78 | 3 | 0.14 | 12.19 | 3 | 6.80 | 0.42 | 2 | 0.09 |
| | Arbutus | 1.95 | 3 | 1.03 | 11.32 | 2 | 0.54 | 0.56 | 4 | 0.10 |
| | Cedar River | 0.58 | 2 | 0.21 | 4.57 | 2 | 0.88 | | | |
| | Durant | 1.70 | 5 | 0.59 | 12.43 | 5 | 2.98 | 0.60 | 2 | 0.14 |

| | | Loon Blood Mercury | | | Loon Feather Mercury | | | Loon Egg Mercury | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Watershed** | **Lake** | **Hg (µg/g)** | **n** | **SD** | **Hg (µg/g)** | **n** | **SD** | **Hg (µg/g)** | **n** | **SD** |
| Upper Hudson | Garnet | 2.53 | 3 | 0.73 | 38.15 | 1 | | | | |
| | Henderson | 3.37 | 1 | | | | | | | |
| | Mason | 1.33 | 3 | 0.67 | 12.93 | 3 | 2.72 | | | |
| | Piseco | 1.26 | 2 | 0.66 | 23.72 | 2 | 19.62 | 0.41 | 1 | |
| | Private #1 | 2.65 | 1 | | | | | | | |
| | Wolf | 1.68 | 5 | 0.64 | 48.21 | 5 | 57.07 | 0.53 | 3 | 0.16 |
| | Woodruff | 1.22 | 3 | 0.35 | 20.91 | 4 | 18.66 | | | |

## 13.0 APPENDIX G. AVERAGE COMMON LOON PRODUCTIVITY FOR EACH ADIRONDACK TERRITORY

**Table G-1. Average productivity (number of fledglings per territorial pair per year) for each territory with three or more years of observed productivity data. FLU is the average female loon unit for that territory and MLU is the average male loon unit for that territory. pH is the measurement taken for the lake where the territory was located.**

| Lake | Territory | N | Productivity | FLU | MLU | pH |
|---|---|---|---|---|---|---|
| Abanakee | NYT079 | 5 | 0.800 | 0.509 | 1.144 | 6.75 |
| Arbutus | NYT051 | 8 | 0.750 | 1.412 | 2.385 | 6.72 |
| Beaver Falls | NYT064 | 5 | 0.200 | 1.355 | 1.884 | |
| Beaver-Lewis | NYT053 | 4 | 0.500 | 2.380 | 3.300 | 6.17 |
| Beaver-Lewis | NYT229 | 7 | 0.429 | 1.301 | | 6.17 |
| Canada | NYT055 | 4 | 0.250 | 2.276 | 3.459 | 6.57 |
| Cedar River | NYT056 | 3 | 0.667 | 0.685 | 0.798 | 6.87 |
| Cooks | NYT059 | 7 | 0.571 | 0.970 | 1.579 | |
| Cranberry | NYT037 | 5 | 0.600 | 2.148 | 2.819 | 6.57 |
| Cranberry | NYT036 | 6 | 0.500 | 2.023 | 2.971 | 6.57 |
| Cranberry | NYT041 | 6 | 0.000 | 2.241 | 3.685 | 6.57 |
| Dart | NYT060 | 5 | 0.200 | 1.266 | 2.242 | |
| Deer | NYT061 | 7 | 0.143 | 2.774 | 3.430 | |
| Durant | NYT019 | 8 | 0.750 | 1.300 | 1.964 | 6.71 |
| East Pine | NYT065 | 6 | 0.667 | 1.527 | 2.202 | 6.90 |
| Ferris | NYT066 | 8 | 0.375 | 4.135 | 5.662 | 5.94 |
| Francis | NYT068 | 5 | 0.600 | 1.837 | 2.477 | |
| Garnet | NYT070 | 5 | 1.400 | 2.093 | 2.762 | 7.09 |
| Hitchins | NYT075 | 8 | 0.500 | 1.329 | 2.070 | 6.46 |
| Limekiln | NYT031 | 7 | 0.857 | 0.682 | 1.385 | 6.39 |
| Little Clear | NYT002 | 7 | 0.857 | 0.427 | 0.834 | 7.26 |
| Little Clear | NYT001 | 3 | 0.667 | 0.575 | 0.847 | 7.26 |
| Little Clear | NYT004 | 7 | 0.857 | 0.727 | 1.341 | 7.26 |
| Little Safford | NYT084 | 7 | 0.571 | 0.799 | 1.360 | 5.60 |
| Long Pond | NYT005 | 6 | 0.833 | 1.520 | 2.289 | 7.82 |
| Lower Mitchell | NYT090 | 5 | 0.000 | 0.335 | 0.680 | |
| Lows | NYT043 | 3 | 0.333 | 1.412 | 2.068 | 6.55 |
| Lows | NYT046 | 7 | 0.857 | 1.439 | 2.940 | 6.55 |
| Mason | NYT086 | 4 | 0.750 | 1.382 | 1.727 | 6.94 |
| Massawepie | NYT087 | 6 | 0.500 | 0.846 | 1.425 | 7.19 |
| Middle | NYT089 | 5 | 0.600 | 2.645 | 3.132 | |
| Middle Saranac | NYT007 | 5 | 0.200 | 0.641 | 1.706 | 6.96 |
| Moss | NYT092 | 8 | 0.500 | 1.683 | 2.978 | 6.60 |
| Newton Falls | NYT047 | 5 | 0.000 | 1.512 | 2.218 | 6.83 |
| Nicks | NYT093 | 9 | 0.778 | 1.366 | 2.529 | 7.00 |
| Oliver | NYT094 | 5 | 0.800 | 0.611 | 1.591 | |
| Piseco-Big Bay | NYT097 | 6 | 0.667 | 1.085 | 1.544 | 5.60 |
| Sand | NYT021 | 4 | 0.750 | 1.264 | 1.918 | |
| Seventh | NYT102 | 6 | 1.167 | 0.535 | 1.062 | 7.08 |
| Silver | NYT104 | 5 | 0.200 | 1.009 | 2.079 | |

| Lake | Territory | N | Productivity | FLU | MLU | pH |
|---|---|---|---|---|---|---|
| Sixth | NYT105 | 6 | 0.833 | 0.454 | 1.150 | |
| South Lake | NYT024 | 8 | 0.875 | 1.088 | 1.935 | 5.70 |
| Spitfire | NYT011 | 6 | 1.000 | 0.748 | 1.059 | 7.21 |
| Taylor | NYT108 | 4 | 0.000 | 1.252 | 2.851 | 7.10 |
| Thirteenth | NYT109 | 5 | 0.800 | 1.090 | 1.721 | |
| Turtle | NYT110 | 3 | 0.667 | 0.700 | 1.750 | |
| Twin Pond | NYT112 | 4 | 0.500 | 1.330 | 1.518 | |
| Twitchell | NYT113 | 8 | 0.375 | 2.329 | 3.970 | |
| Upper St. Regis | NYT015 | 7 | 1.286 | 1.017 | 2.200 | |
| Wolf | NYT115 | 4 | 0.750 | 0.923 | 1.932 | 6.93 |
| Woodhull | NYT029 | 7 | 0.143 | 0.899 | 0.696 | |
| Woodhull | NYT025 | 7 | 0.286 | 1.660 | 2.785 | |
| Woodruff | NYT116 | 3 | 1.333 | 0.848 | 1.421 | 7.52 |

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**New York State Energy Research and Development Authority**

17 Columbia Circle
Albany, New York 12203-6399

**toll free:** 1 (866) NYSERDA
**local:** (518) 862-1090
**fax:** (518) 862-1091

info@nyserda.ny.gov
nyserda.ny.gov


# Long-Term Monitoring and Assessment of Mercury Based On Integrated Sampling Efforts Using the Common Loon, Prey Fish, Water, and Sediment

Final Report
September 2011

**State of New York**
Andrew M. Cuomo, Governor

**New York State Energy Research and Development Authority**
Francis J. Murray, Jr., President and CEO