

**MERCURY CONTAMINATION IN FISHES OF MOUNT DESERT ISLAND
AND A COMPARATIVE FOOD CHAIN MERCURY STUDY**

By

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B.A. University of Southern Maine, 1994

A THESIS

Submitted in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

(in Zoology)

The Graduate School

University of Maine

December, 1997

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MERCURY CONTAMINATION IN FISHES OF MOUNT DESERT ISLAND AND A COMPARATIVE FOOD CHAIN MERCURY STUDY

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**An Abstract of the Thesis Presented
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To assess the scope of mercury contamination in lakes within and surrounding Acadia National Park, mercury concentrations were determined in fish species of ten lakes on Mount Desert Island, Maine, USA. Five fish of the most common predatory species were collected from each lake. Skinless and boneless fillets of each fish, and the remaining carcass, were homogenized separately and analyzed for total mercury. Mean size-adjusted fillet concentrations for each lake ranged from $0.17 \mu\text{g g}^{-1}$ to $1.71 \mu\text{g g}^{-1}$ (wet wt.). Smallmouth bass (*Micropterus dolomieu*) from Hodgdon Pond had a mean fillet concentration of $1.71 \mu\text{g g}^{-1}$, which is 2.5 times higher than the statewide average ($0.67 \mu\text{g g}^{-1}$) for this species. Hodgdon Pond and Seal Cove Pond, an adjacent pond with a significantly lower mean mercury concentration ($0.85 \mu\text{g g}^{-1}$, $P=0.0008$) in smallmouth bass, were chosen for a comparative food chain mercury study. Food organisms in the

chain leading to smallmouth bass and in prey fish species were determined by stomach content analysis. Water, sediment, plankton, macroinvertebrates, and prey fish species were collected seasonally from both ponds and analyzed for total and methylmercury.

In Hodgdon Pond, warmer, low-dissolved oxygen hypolimnetic water is probably contributing to the enhanced bioavailability of mercury, as seen by higher methylmercury concentrations in plankton at the base of the food chain. An extra trophic link in the food chain leading to smallmouth bass, as evidenced by the presence of larval fish and isopods in the diet of yellow perch, is also contributing to the increased biomagnification of mercury in Hodgdon Pond.

ACKNOWLEDGMENTS

I thank my advisor, Dr. Terry Haines, for his advice and guidance throughout this project. I also thank Dr. John Moring and Dr. John Dearborn for their advice as committee members.

I am grateful to Ruth Perry, Jacqueline Baltunis, and Marty Richards for their advice and assistance with laboratory analyses. I am also grateful to Dan Demers for his field and laboratory assistance. I thank Ken Oliviera for aging the eels collected in this study. I also thank Kim Seymour, Walter Golet, Mike Palmer, Laura Wilson, Patrick Elliott, Susan Anderson, and all of my friends and colleagues at the University of Maine, for their help and assistance.

Funding for this project was provided by the National Park Service, the U.S. Geological Survey – Biological Resources Division, and the University of Maine. Thanks also to the Maine Department of Inland Fisheries and Wildlife for the use of their trapnets and for providing information on the lakes involved in this study. I also thank David Manski, Bob Breen, and Bill Gawley, of the National Park Service, for their logistical help during field sampling.

Finally, I would like to thank my wife, Kelly, and our children, Jasmine and Kyle, for their love and support during this challenging project.

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INTRODUCTION

Over the past decade, there has been an increasing awareness of the widespread health threat posed to humans and wildlife by high levels of mercury in freshwater fishes. Thirty-four states, including Maine, have issued fish consumption advisories because of high levels of mercury in fishes (USEPA, 1995). High concentrations of mercury have been found in Maine sport fishes (Stafford and Haines, 1997). In particular, Hodgdon Pond, a small warmwater pond on Mount Desert Island, had the highest concentrations of mercury in fish of any lake surveyed in Maine (Stafford, 1994).

Atmospheric deposition is the primary source of mercury contamination to remote aquatic ecosystems that have no point sources of mercury pollution (Watras *et al.*, 1994). The largest sources of anthropogenic mercury pollution to the atmosphere are from coal combustion, incineration of solid wastes, and the smelting of ore for non-ferrous metal production (Nriagu and Pacyna, 1988). Natural sources, such as oceanic and soil degassing, forest fires, and volcanic eruptions, release mercury to the atmosphere as well.

Greater than 95% of the mercury in predatory fishes is in the form of monomethyl mercury (Bloom, 1992). However, most mercury is atmospherically deposited in the inorganic form (Bloom and Fitzgerald, 1988), and is subsequently methylated by sulfate-reducing bacteria to form monomethyl mercury (Gilmour and Henry, 1991).

Monomethyl mercury (CH_3Hg^+) is the most toxic form of mercury and easily passes through biological membranes (Lindqvist and Rodhe, 1985). Methylmercury uptake occurs by passive diffusion from water to phytoplankton at the base of aquatic foodchains

(Ribeyre and Boudou, 1982; Mason *et al.*, 1995). At each subsequent increase in trophic level, from zooplankton to fish, diet is the primary route of methylmercury uptake (MacCrimmon *et al.*, 1983; Cope *et al.*, 1990; Mason *et al.*, 1995). Biomagnification by aquatic organisms occurs because methylmercury is easily assimilated and resistant to depuration (McKim *et al.*, 1976). Because mercury strongly binds to sulfhydryl groups in protein, most of the methylmercury in fish is sequestered in the muscle tissue (Wiener and Spry, 1996).

Non-salmonid fish species generally have higher concentrations of mercury in their tissues than do salmonid species (Stafford and Haines, 1997). This trend may occur because non-salmonid species spend the warmer months of the year in water of warmer temperature, where enhanced uptake of methylmercury through water and diet may take place. Piscivorous fish species generally accumulate much higher concentrations of methylmercury than do fish species feeding at lower trophic levels (MacCrimmon *et al.*, 1983; Wren and MacCrimmon, 1986; Stafford and Haines, 1997).

Mercury concentrations in the same fish species vary greatly among lakes. Bioaccumulation of mercury in fish is controlled by several factors, including trophic level, growth rate, age, and the bioavailability of methylmercury. Mercury bioavailability is controlled by various environmental parameters, including temperature, pH, dissolved organic carbon (DOC), dissolved oxygen, and sulfate, which relate to the formation of methylmercury (Winfrey and Rudd, 1990).

Mercury concentrations in fish have been strongly correlated with lake epilimnetic water temperatures (Bodaly *et al.*, 1993). This relationship may result from the effect of temperature on the rate of methylmercury formation. Warmer temperatures

enhance the rate of methylation, with a temperature optimum of 35°C (Callister and Winfrey, 1986). Temperature also has an effect on bioaccumulation at the base of aquatic food chains. Ribeyre and Boudou (1982) demonstrated that increased temperature enhanced the accumulation of mercury by algae.

High mercury concentrations in fish from low-pH lakes have been found in many studies (Håkanson *et al.*, 1988; Grieb *et al.*, 1990; Lindqvist *et al.*, 1991; Watras and Bloom, 1992; Haines *et al.*, 1995). Several mechanisms have been suggested to explain this phenomenon, including: increased permeability of fish gills to mercury, increased solubility of mercury in acidic water, increased inputs of mercury from terrestrial ecosystems or precipitation, and increased in-lake production of methylmercury (Winfrey and Rudd, 1990). However, some circumneutral lakes have high levels of mercury in fish as well (Akielaszek and Haines, 1981; Wren and MacCrimmon, 1986), thus demonstrating that other factors are interrelated in regulating mercury bioavailability.

High concentrations of dissolved organic carbon in the water column can lower the bioavailability of methylmercury, and lead to decreased concentrations of methylmercury in fish (Watras *et al.*, 1994), and zooplankton (Back and Watras, 1995). However, other studies have found high positive correlations between mercury in fish and DOC in the water column (McMurtry *et al.*, 1989; Haines *et al.*, 1992). Back and Watras (1995) suggest that DOC complexes with mercury, and therefore competes with biota for available mercury. Increased concentrations of DOC in the water column result in reduced mercury methylation rates (Winfrey and Rudd, 1990). However, increased concentrations of organic carbon in sediments can enhance production of methylmercury by stimulating productivity within the microbial community (Gilmour and Henry, 1991).

Thus, the productivity of the microbial community or the availability of mercury as a substrate may limit production of methylmercury. Ultimately, the most important factor controlling methylmercury formation in one system may not be the most important factor in another type of ecosystem (Winfrey and Rudd, 1990).

The purpose of this study was to assess the scope of mercury contamination in lakes within and adjacent to Acadia National Park, Mount Desert Island, Maine. In addition, a detailed comparison of mercury concentrations in biota at various trophic levels within Hodgdon Pond and an adjacent pond was conducted in an attempt to elucidate reasons for the elevated mercury levels found in Hodgdon Pond. Because atmospheric deposition of mercury to the two ponds should be similar due to their close proximity, differences in biota mercury levels must be related to varying environmental and biological parameters within each system. Therefore, various environmental and biological parameters, including water mercury concentrations, water chemistry, water temperatures, sediment mercury concentrations and organic content, and the seasonal diets of forage and top predatory fish species were also determined for each pond. These data were collected in addition to the mercury biota data in an attempt to identify important differences that may control mercury bioavailability and bioaccumulation in these ponds.

METHODS

Survey of Mercury in Predatory Fishes of Mount Desert Island

Study Sites

Mount Desert Island is a coastal island located in Frenchman's Bay, Maine, USA. Nine lakes and ponds on the island were chosen for the initial investigation, as well as one pond located on Isle au Haut, which is an island located further offshore and a large proportion of which is part of Acadia National Park. Lakes were chosen because they are within or in close proximity to Acadia National Park, and contain predatory, sport fish species (Table 1). Hodgdon Pond was also included in the survey, but the data reported for this lake are from a previous study (Stafford and Haines, 1997).

Water samples were collected from each lake and water chemistry analyses were conducted at Sawyer Environmental Chemistry Laboratory, University of Maine, Orono, Maine. Samples were analyzed for Ca, Mg, K, Na, Cl, Al, Si, SO₄, NO₃, NH₄, DIC, DOC, TSS, color, pH, ANC, and specific conductance. Anions were determined by ion chromatography (method 300.0, U.S. EPA, 1993), cations by flame atomic absorption (method 215.1, U.S. EPA, 1983), true color by spectrophotometry (method 110.3, U.S. EPA, 1983), and acid neutralizing capacity by automated colorimetry (method 310.2, U.S. EPA, 1983).

Fishes

Fishes were collected in May and June of 1995 using trapnets, gillnets, and angling. The most abundant top predatory species in each lake was targeted for collection (Table 1). The collection objective from each lake was five fish of the same

Table 1. Lakes and ponds sampled and fish species collected in and near Acadia National Park, 1996.

Lake	Species	N	Mean Length (mm)	Mean Weight (g)
Eagle Lake	lake trout <i>Salvelinus namaycush</i>	5	462	797
Echo Lake	landlocked salmon <i>Salmo salar</i>	4	405	542
Hamilton Pond	chain pickerel <i>Esox niger</i>	5	387	332
Hodgdon Pond	smallmouth bass <i>Micropterus dolomieu</i>	6	430	1141
Jordon Pond	landlocked salmon <i>Salmo salar</i>	5	519	1311
Long Pond (MDI) ^a	smallmouth bass <i>Micropterus dolomieu</i>	5	393	834
Long Pond (IAH) ^b	brook trout <i>Salvelinus fontinalis</i>	5	258	185
Lower Hadlock Pond	white perch <i>Morone americana</i>	5	267	242
Round Pond	chain pickerel <i>Esox niger</i>	5	378	336
Seal Cove Pond	smallmouth bass <i>Micropterus dolomieu</i>	5	394	817
Somes Pond	chain pickerel <i>Esox niger</i>	5	369	276

^a Mount Desert Island

^b Isle Au Haut

species and of a typical size kept by recreational fishermen. Total length and weight of each fish was recorded upon capture. Each fish was individually wrapped in aluminum foil and placed on ice until frozen. All fish collected were frozen within 24 hours of capture.

For analysis, fish were thawed, remeasured, and reweighed, and scales were collected for age determination. To determine the age of each fish, cleaned scales were mounted between glass slides and viewed on a microfiche projector to enumerate annuli. Each fish was rinsed twice, first with tap water, then with deionized water. A skinless, boneless fillet was removed from the left side of each fish. Fillets were removed by placing each fish on its side on a polyethylene cutting board and cutting posterior to the operculum down to the backbone with a stainless steel fillet knife. The cut was then continued along the backbone to the caudal peduncle. After the skin and rib cage were removed, each fillet was weighed, then homogenized using a food processor. All equipment used in the homogenization procedure was washed with detergent, then with 10% nitric acid, and rinsed with deionized water. This washing regime was used after processing each individual fish. The remaining carcass also was weighed and homogenized. All homogenized tissues were stored frozen in acid-washed plastic storage bags.

Percent moisture was determined for each homogenized sample by drying approximately 1.5 g of tissue to a constant weight at 105°C. Subsamples of homogenized tissue were digested with 5 mL of nitric acid (EM Scientific, Tracepur® Plus) and 1 mL of 30% hydrogen peroxide using a microwave-assisted digestion procedure (for a detailed description see Stafford, 1994). Samples that exceeded 10% weight loss during digestion

were not used. Digestates were diluted with deionized water to 50 mL in volumetric flasks, and stored in polyethylene bottles. All glassware, digestion vessels, and bottles were washed with detergent, soaked in 50% nitric acid overnight, and rinsed with deionized water prior to each use. The accuracy of mercury determinations was verified by analysis of standard reference materials (Dorm-1 or Dorm-2, National Resource Council Canada), spiked samples, replicate samples, and reagent blanks.

Digestate aliquots ranging from 10 mL to 50 mL were analyzed for total mercury with a cold vapor atomic absorption spectrophotometer (Coleman Mercury Analyzer System, model MAS-50B). Sample aliquots and mercury standard aliquots were diluted to 100 mL with deionized water in a graduated cylinder immediately prior to analysis. Mercury in the samples was reduced with 1 mL of 10% stannous chloride. Samples were bracketed with standards prepared from a $1.0 \mu\text{g mL}^{-1}$ mercury standard solution. Spectrophotometer output was recorded using a data shuttle (Omega Engineering, Inc.) and data acquisition software (Quicklog, Strawberry Tree, Inc.). Peak height was measured for each sample and converted into log % transmittance. Mercury concentration was then determined from a standard curve.

Approximately ten percent of the samples were reanalyzed with a cold vapor atomic fluorescence spectrophotometer (PS Analytical LTD). This instrument was used because matrix interference was encountered when the Dorm-2 certified reference material was analyzed with cold vapor atomic absorption spectroscopy.

Food Chain Study

Study sites

Hodgdon Pond and Seal Cove Pond were chosen for a comparative study of mercury in aquatic food chains. These two adjacent ponds were chosen because Hodgdon Pond has extremely high concentrations of mercury in smallmouth bass (Stafford, 1994), and Seal Cove Pond was found to have significantly lower mercury concentrations in the same species (least square means: $1.71 \mu\text{g g}^{-1}$ and $0.85 \mu\text{g g}^{-1}$ wet wt. respectively, $P=0.0008$). Hodgdon Pond has a surface area of 14.2 hectares with a maximum depth of 6.7 m, and Seal Cove Pond has a surface area of 114.6 hectares and a maximum depth of 13.4 m. A small outlet stream flows from Hodgdon Pond into Seal Cove Pond. Hodgdon Pond has one inlet stream and Seal Cove Pond has three inlets in addition to the input from Hodgdon Pond. Seal Cove Pond has a dam at its one outlet, and the outlet stream flows directly into salt water. This stream provides a migratory path for spawning alewives.

Species assemblages are very similar in both ponds, with smallmouth bass (*Micropterus dolomieu*) and chain pickerel (*Esox niger*) being the dominant predatory species. Other abundant species in both ponds include: yellow perch (*Perca flavescens*), white perch (*Morone americana*), golden shiner (*Notemigonus crysoleucas*), pumpkinseed (*Lepomis gibbosus*), redbreast sunfish (*Lepomis auritus*), American eel (*Anguilla rostrata*), brown bullhead (*Ameiurus nebulosus*), white sucker (*Catostomus commersoni*), and alewife (*Alosa pseudoharengus*). Brown trout (*Salmo trutta*) are also present in Seal Cove Pond, but not in sufficient numbers for the purposes of this study.

The food chain investigation involved two separate sampling periods in 1996. Spring samples were collected from 15 May until 30 June, and summer samples were collected from 15 August until 30 September.

Fish

The diets of smallmouth bass from both ponds were examined in an attempt to identify differences in trophic feeding levels. The diets of yellow perch, the principal prey species of the smallmouth bass, were also examined. Fish were captured using fyke nets, gillnets, and angling. Length and weight were recorded for each fish, and scales were collected for age determination. Stomach contents were removed using the gastric lavage method of Hartleb and Moring (1995). Fish were released alive after stomach contents were removed. Stomach contents were preserved in 70% ethanol and stored in polyethylene bottles. Prey items were enumerated under a dissecting microscope and identified to taxonomic family. Frequency of occurrence was determined by calculating the number of stomachs containing one or more individuals of each food category. This number was expressed as a percentage of all stomachs containing food.

Forage fish species, determined from stomach content analysis, were captured using beach seines and small-mesh gillnets. Captured forage fish were dispatched, placed in clean ziplock bags, placed on ice, and frozen within 24 hours of capture. Upon arrival in the laboratory, fish were thawed, measured, weighed, and scales were collected for age determination. Prior to homogenization, each fish was rinsed twice, first with tap water, then with deionized water. Fish were homogenized using a food blender and stored in acid-washed, plastic storage bags. Moisture content of larger forage fishes was

determined by drying approximately 1 g of homogenized tissue to a constant weight at 105°C. In order to conserve tissue for mercury analyses, smaller fishes were freeze-dried after homogenization to determine moisture content.

For total mercury determination, tissue was digested using the microwave-assisted acid digestion previously described, with the following modifications. Subsamples of approximately 1.0 g of wet, or 0.05 to 0.10 g of dry, homogenized tissue were digested with 15 mL of nitric acid (EM Scientific, Tracepur® Plus) and 2 mL of 30 % hydrogen peroxide. Digestates were diluted with deionized water to 50 mL in volumetric flasks, and stored in polyethylene bottles until analyzed. Total mercury was determined by cold vapor atomic fluorescence spectroscopy.

A method developed by Liang *et al.* (1994a), was slightly modified to digest forage fish samples for methylmercury determination. Briefly, 0.5 to 1.0 g of wet tissue, or 0.05 to 0.1 g of freeze-dried tissue, was digested in 15 mL of 25% KOH in methanol in a 25 mL glass vial with a Teflon®-lined cap. Samples were digested at 75°C for 3 hours, and analyzed within 24 hours of digestion. Methylmercury concentrations in digestates were determined by aqueous phase ethylation, gas chromatograph separation, and cold vapor atomic fluorescence detection (Brooks Rand Atomic Fluorescence Spectrometer, model 2), using a technique modified from Liang *et al.* (1994b). An aliquot of 50 µL of the digestate was diluted with 100 mL of water, neutralized to pH 4.9 with acetate buffer, and ethylated with 55 µL of sodium tetraethylborate. The sample was allowed to react for 20 minutes and was then purged with argon gas onto a CarboTrap™. The trap was then heated to 200°C and the sample was eluted through a gas chromatograph column. Eluted mercury species were decomposed into elemental mercury (Hg⁰) by heating to

900°C in a thermal decomposition tube and detected by cold vapor atomic fluorescence spectroscopy.

Macroinvertebrates

Macroinvertebrates were collected by dipnetting in surficial sediment and vegetation in the littoral zone of both ponds. Individual odonates were removed from the nets with stainless steel tweezers and placed in polyethylene vials. Amphipods and isopods were concentrated using graduated sieves before individual collection and placement in polyethylene vials. Polyethylene vials were placed on ice and invertebrates were frozen within 24 hours of collection.

Upon arrival in the laboratory, predatory macroinvertebrates were thawed, rinsed with deionized water, identified, and freeze-dried. Odonates collected in the early sampling period were identified to family, and those collected in the latter period were identified to genus, in an attempt to discern possible differences in mercury concentrations among genera. After identification and freeze-drying, macroinvertebrates were homogenized using a freezer mill (Spex model 6700), which uses a brittle fracture technique by pulverizing samples in a bath of liquid nitrogen. Homogenized dry tissue was then digested using microwave acid digestion. Approximately 0.05 g to 0.15 g of dry tissue was placed in Teflon® digestion vessels (Quesstron Corp., very high pressure 625 PSI) and 5 mL of nitric acid (EM Scientific, Tracepur® Plus) and 5 mL of deionized water was added to each vessel. Digestion vessels were then heated in a microwave digestion system (Quesstron Corp., Qwave 3000). The microwave sequence consisted of a 10 minute ramp period during which vessels were gradually heated to an internal

temperature of 170°C. The ramp phase was followed by a 15 minute dwell period, during which vessels were maintained at 170°C, and then a 10 minute cooling period. After vessels reached room temperature, digestates were diluted to 50 mL with deionized water in volumetric flasks. Aliquots of diluted digestates were analyzed for total mercury using the cold vapor atomic fluorescence spectroscopy method previously described. For methylmercury determination, subsamples of dry, homogenized tissue were digested using the alkaline digestion method and analysis procedure previously described for forage fish, with the exception that 10 mL of 25% KOH in methanol was used to digest invertebrate samples.

Plankton

Plankton was sampled with a non-metallic #20 plankton net (mesh size 76 µm). Several vertical tows were collected from the center of each pond, and combined to form composite samples. Plankton were transferred from the net to trace-clean glass jars with Teflon®-lined caps. Jars were placed on ice, in the dark, until samples could be frozen. Plankton samples were freeze-dried, and then were digested with 4 mL of 25% KOH in methanol for 3 hours at 75°C for methylmercury determination. Methylmercury digestates were analyzed using the same procedure as described for macroinvertebrates, except larger aliquots (50-100 µL) of digestate were analyzed. Plankton samples analyzed for total mercury were digested and analyzed using the same procedures described for macroinvertebrates. During the second sampling period, horizontal tows covering the range of depth in each pond were necessary to obtain an adequate sample size.

Sediment

Surficial sediment samples were collected using a light-weight, gravity corer (Forestry Suppliers Inc., 2 in. diameter). The top 3 cm of the core was collected by extruding the core with a wooden dowel. Sediment was collected directly into Whirl-Pak® bags as it was extruded. Samples were stored on ice in the dark until frozen. Samples were collected at 1.5 m and 3.0 m depths, and at the deepest point of each pond. Two samples were collected approximately 3.0 m apart at each sampling depth.

Moisture content was determined by drying sediment to a constant weight at 35°C. Total volatile matter (an estimate of organic matter) was determined by loss on ignition at 550°C. Approximately 1 g of dry sediment was placed in a 550°C muffle-furnace until a constant weight was achieved. Total mercury was determined using the same digestion and analysis procedures described for macroinvertebrates.

Methylmercury was determined by using a modified version of an extraction technique developed by Liang *et al.* (1996). Approximately 0.05 to 0.10 g of dried sediment was digested using the alkaline digestion method previously described. Methylmercury was extracted from the digestate by adding 8 mL of methylene chloride and 2 mL of hydrochloric acid to the digestate and shaking for 10 minutes. The methylene chloride layer was drawn off into a 125 mL Teflon® bottle using a separatory funnel, and the sediment slurry was washed twice with 10 mL aliquots of methylene chloride.

Methylmercury was back-extracted into 80 mL of ultrapure deionized water by boiling off the methylene chloride at 80°C. Methylmercury was determined by aqueous phase ethylation, gas chromatograph separation, and cold vapor atomic fluorescence detection, as previously described.

Water

Water samples were collected during both sampling periods at depths of 0.5 m and at 5.5 m (Hodgdon Pond) and 0.5 m and 13.0 m (Seal Cove Pond). Deep samples were collected with a Van Dorn water sampler that was rinsed with 10 % nitric acid and deionized water prior to use. Shallow water samples were collected by hand with the boat pointed into the wind. Shoulder length polyethylene gloves were worn at all times during water sample collection. A reference blank (deionized water) and field blank (deionized water collected from sampler) were used to check for contamination from the sampling device.

Total mercury was determined by oxidizing samples with 0.2N bromine monochloride, reduction of mercury with 20 % stannous chloride, purging with helium gas, and trapping of eluted mercury with gold-coated quartz sand traps. Mercury collected on traps was quantified by cold vapor atomic fluorescence spectrometry (method 1631, U.S. EPA, 1995). Methylmercury in water was determined using the distillation method of Horvat *et al.* (1993), followed by aqueous phase ethylation, gas chromatograph separation, and cold vapor atomic fluorescence detection, as previously described.

Temperature data loggers (Optic StowAway™, Onset Computer Corporation) were used to measure water temperatures at depths of 1.5 m and 3.0 m in both ponds. Water temperatures were recorded at two-hour intervals from May to December.

Stable Isotope Analysis

Subsamples of fillet tissue from smallmouth bass and chain pickerel, and subsamples of whole body tissue from forage-size yellow perch, were dried to a constant weight at 105°C. Dried samples were homogenized and analyzed for the stable isotope $\delta^{15}\text{N}$ at the Stable Isotope Laboratory, Sawyer Environmental Research Center, University of Maine, Orono, Maine.

Statistical Analyses

All statistical analyses were performed with SAS, version 6 (SAS Institute, 1990). Fish mercury concentrations for each species were normalized for differences in size of fish by calculating least square means with weight as the covariate. Least square mean mercury concentrations were calculated for each lake using the pooled data for all lakes containing the same species. The mean mercury concentration for each lake represents the mean concentration for a fish of the average weight of all fish of a given species. Mercury concentration data were log-transformed prior to statistical analyses when normality was improved by transformation. Analysis of variance was performed on water, sediment, plankton, and invertebrate mercury concentration data, to reveal differences between lakes, seasons, and/or locations (e.g., epilimnion, hypolimnion, littoral, or profundal). When significant differences were indicated by analysis of variance, the Ryan-Einot-Gabriel-Welsh Multiple F Test was used to discern differences among means. Student's t-test was used to determine differences in amphipod mercury concentrations between ponds. An alpha level of 0.05 was used in testing all hypotheses.

RESULTS

Quality assurance checks supported the accuracy of the mercury analyses. The Standard Reference Material results for the survey of Mount Desert Island fishes were as follows: all DORM-1 samples were within the certified range, and two of the DORM-2 samples were slightly outside the certified range (Table 2). The recovery of mercury from spiked samples averaged 85% (range 82-94, N=7) for fish and 97% (range 90-107, N=4) for standard reference material. The difference between replicate samples averaged 9.5% (range 0-25, N=16). All reagent blanks were below detection limits.

The Standard Reference Material results for the food chain study were as follows: nine CRM 463 samples were slightly out of the certified range (Table 2). However, some CRM 463 samples were digested using smaller than recommended weights, which may have contributed to the observed variability in mercury concentrations. All Tort-2 samples analyzed for total mercury were within the accepted range. Most of the Tort-2 samples analyzed for methylmercury were out of the specified range. However, the overall mean was only slightly out of the certified range, and all of these samples were digested using much smaller weights than recommended, because of low recoveries encountered when larger sample weights were used. Three of the Mess-2 samples slightly exceeded the certified range, and all five of the IAEA 356 samples analyzed for methylmercury were out of the certified range, but the mean of these samples was within the certified range. These results were again attributed to using smaller digestion weights than recommended, and the mean concentration seems to support the accuracy of these analyses. The recovery of inorganic mercury from spiked samples averaged 99% (range

Table 2. Standard reference material results (all results in $\mu\text{g g}^{-1}$ as Hg, dry weight basis).

Standard Material	N	Mean	SD or CI^d	Range
DORM-1 (dogfish)^a				
This study	5	0.818	0.05	0.750-0.867
Certified (total Hg)		0.798	0.074	0.724-0.872
DORM-2 (dogfish)^a				
This study	4	4.459	0.257	4.152-4.730
Certified (total Hg)		4.64	0.26	4.38-4.90
CRM 463 (tuna)^b				
This study	23	2.962	0.257	2.486-3.375
Certified (total Hg)		2.85	0.16	2.69-3.01
TORT-2 (lobster)^a				
<u>Total mercury</u>				
This study	8	0.264	0.029	0.230-0.320
Certified		0.27	0.06	0.21-0.33
<u>Methyl mercury</u>				
This study	21	0.173	0.059	0.054-0.261
Certified		0.152	0.013	0.139-0.165
MESS-2 (sediment)^a				
This study	4	0.107	0.009	0.094-0.113
Certified (total Hg)		0.092	0.009	0.083-0.101
IAEA 356 (sediment)^c				
This study	4	0.00526	0.00269	0.00247-0.00941
Certified (methyl Hg)		0.00546	0.00039	0.00507-0.00584

^a Produced by the National Research Council of Canada (NRCC).

^b Produced by the Commission of the European Communities, Community Bureau of Reference.

^c Produced by the International Atomic Energy Agency (IAEA).

^d Standard deviation for this study; 95% confidence interval for certified values.

63-125, N=4) for fish, 79% (N=1) for invertebrates, 99% (range 77-119, N=12) for CRM 463, 77% (range 48-111, N=6) for Tort-2, and 89% (range 75-103, N=2) for Mess-2. The recovery of methylmercury averaged 98% (range 23-188, N= 27) for spiked samples and 81% (range 51-116, N=13) for spiked reagent blanks. The percent difference between replicate analyses averaged 8.4% (range 0-34, N= 54) for total mercury samples, and 11.2% (range 0-35, N=42) for methylmercury samples.

The method detection limit for total mercury analysis was calculated as 0.06 ng, and the practical quantitation limit was calculated as 0.17 ng. All samples exceeded the practical quantitation limit, and all reagent blanks were less than the practical quantitation limit. Only ten reagent blanks exceeded the method detection limit. The method detection limit for methylmercury analysis was calculated as 0.01 ng. All samples exceeded the detection limit and only one reagent blank exceeded detection.

Survey of Mercury in Predatory Fishes of Mount Desert Island

Least square mean mercury concentrations indicate that fishes on Mount Desert Island generally have moderate mercury concentrations (Table 3). Six of the eleven lakes surveyed have mean fish fillet mercury concentrations that are less than the State of Maine fish consumption limit of $0.43 \mu\text{g g}^{-1}$ of wet fish tissue. Lakes containing salmonid species and two ponds containing chain pickerel (Round Pond and Some Pond) were low in fish mercury concentrations. However, chain pickerel collected from these two ponds were small, young fish and thus may not be representative of larger individuals of this species in these ponds.

Hodgdon Pond is the only lake surveyed with a mean mercury concentration in fish fillet exceeding the Food and Drug Administration consumption limit of $1.0 \mu\text{g g}^{-1}$. Four lakes exceed the Maine consumption limit, but have fish mercury concentrations less than the FDA consumption limit. Overall, most of the lakes surveyed have fish mercury concentrations less than or near the statewide mean for the species surveyed. Two ponds, Hodgdon Pond and Seal Cove Pond, exceed the statewide mean mercury concentration for species surveyed. Whole-body mercury concentrations were on average 30% less than concentrations found in fillets. Salmonid fish species had less of a difference between fillet and whole-body concentrations, than did non-salmonid fish species.

All lakes and ponds surveyed are circumneutral in pH, with values ranging from 6.69 to 7.34, but acid neutralizing capacities (ANC) are generally low (Table 4). Only one pond, Hamilton Pond, has a high concentration of dissolved organic carbon: 13.1 mg L^{-1} .

Table 3. Mercury concentrations in Mount Desert Island fishes (Mean values are least square mean concentrations).

Lake	Species	Fillet ($\mu\text{g g}^{-1}$ wet)		Whole-body ($\mu\text{g g}^{-1}$)		State avg. (fillet) ($\mu\text{g g}^{-1}$ wet) ¹
		X	Range	X	Range	
Eagle Lake	lake trout	0.25	0.21-0.32	0.20	0.17-0.25	0.45
Echo Lake	landlocked salmon	0.19	0.11-0.17	0.14	0.12-0.15	0.35
Hamilton Pond	chain pickerel	0.59	0.39-0.79	0.43	0.29-0.58	0.88
Hodgdon Pond	smallmouth bass	1.71	0.76-3.86	N/A	N/A	0.66
Jordon Pond	landlocked salmon	0.17	0.14-0.37	0.14	0.10-0.18	0.35
Long Pond (MDI)	smallmouth bass	0.56	0.41-0.54	0.27	0.24-0.31	0.66
Long Pond (IAH)	brook trout	0.21	0.09-0.49	0.17	0.09-0.40	0.31
Lower Hadlock Pond	white perch	0.69	0.36-1.03	0.31	0.29-0.39	0.65
Round Pond	chain pickerel	0.37	0.27-0.56	0.29	0.28-0.35	0.88
Seal Cove Pond	smallmouth bass	0.85	0.51-1.16	0.51	0.28-0.84	0.66
Somes Pond	chain pickerel	0.37	0.27-0.45	0.24	0.19-0.31	0.88

¹ State of Maine average mercury concentrations are from Stafford and Haines, 1997.

Table 4. Physical and chemical characteristics of lakes surveyed.

Lake	Area (ha)	Depth (m)	pH (SU)	ANC ($\mu\text{eq/L}$)	Color (Haz.)	Al ($\mu\text{g/L}$)	Ca ($\mu\text{eq/L}$)	Mg ($\mu\text{eq/L}$)	Na ($\mu\text{eq/L}$)	SO ₄ ($\mu\text{eq/L}$)	Cl ($\mu\text{eq/L}$)	DOC (mg/L)	Sp.Cond (μS)
Eagle Lake	176	34	6.93	46.6	2	21	79.8	38.7	154.0	72	140	1.82	32.5
Echo Lake	96	20	7.21	86.6	6	9	131.7	60.0	288.0	89	265	2.75	55.7
Hamilton Pond	15	4	7.34	159	130	98	135.2	66.6	274.9	50	218	13.1	53.8
Hodgdon Pond	11	7	6.96	66.2	28	59	71.9	60.9	176.6	60	146	5.85	37.0
Jordon Pond	75	46	7.08	63.2	2.5	13	93.3	42.0	140.3	80	128	1.62	33.2
Long Pond (MDI)	360	34	6.95	50.7	7.5	18	81.8	54.7	191.0	84	175	2.98	39.9
Long Pond (IAH)	30	18	6.69	32.5	3	28	74.4	107.8	443.7	125	434	2.97	79.4
Lower Hadlock Pond	14	12	6.90	47.7	6	17	83.3	38.7	175.7	76	145	3.07	34.9
Round Pond	15	6	6.72	39.1	15	27	57.4	41.1	175.3	60	144	4.51	32.3
Seal Cove Pond	114	13	6.96	61.7	9	27	82.8	59.2	188.4	75	167	3.9	39.0
Somes Pond	42	8	6.99	62.1	22	26	89.8	60.4	217.5	74	197	4.93	42.5

Food Chain Study

Predatory Fish

Total mercury concentrations were significantly higher in smallmouth bass, chain pickerel, and American eel fillets from Hodgdon Pond than from Seal Cove Pond (Table 5; $P=0.0003$, $P=0.027$, and $P=0.001$ respectively). The average weights used to compute least square mean concentrations for each species were 944 g, 380 g, and 614 g, respectively. Average weights represent the average weight of all individuals of a given species collected from both ponds.

Dietary Analysis

Yellow perch were the primary dietary item of smallmouth bass in both ponds (Fig. 1). Unidentifiable fish (partially digested), probably yellow or white perch, were the next most abundant item in smallmouth bass from Hodgdon Pond, with insects and alewives occurring infrequently. In contrast, insects and alewives were important items in the diet of Seal Cove Pond smallmouth bass.

Dietary analysis of yellow perch revealed Isopoda to be the most abundant taxon in the spring diet of Hodgdon Pond yellow perch (Fig. 2). In contrast, this taxon was absent from the spring diet of yellow perch in Seal Cove Pond. Larval fishes were not observed in the spring diet of yellow perch from Seal Cove Pond, but were relatively abundant in the spring diet of yellow perch in Hodgdon Pond. Zygoptera were relatively abundant in the diet of yellow perch in Seal Cove Pond, but were absent in the diet of yellow perch in Hodgdon Pond. Amphipoda and Anisoptera were the most abundant taxa common to the spring diets of yellow perch in both ponds.

Table 5. Total mercury least square mean concentrations in fillets of predatory fishes.

Site	Species	<u>Length (mm)</u>			<u>Weight (g)</u>		<u>Age (yr.)</u>		<u>THg ($\mu\text{g g}^{-1}$ wet wt.)</u>	
		N	X	Range	X	Range	X	Range	X	Range
Hodgdon Pond	American eel	4	607	476-680	505	220-662			1.63	0.94-1.70
	Chain pickerel	8	347	278-440	270	120-552	3.8	2-5	0.48	0.16-0.70
	Smallmouth bass	7	433	329-513	1138	419-1686	8.7	5-12	1.72	0.77-3.86
Seal Cove Pond	American eel	4	678	632-756	723	571-1036			0.77	0.65-1.41
	Chain pickerel	6	418	333-465	527	203-699	4.3	3-6	0.36	0.25-0.86
	Smallmouth bass	8	380	273-457	774	252-1326	7.4	5-9	0.91	0.32-1.30

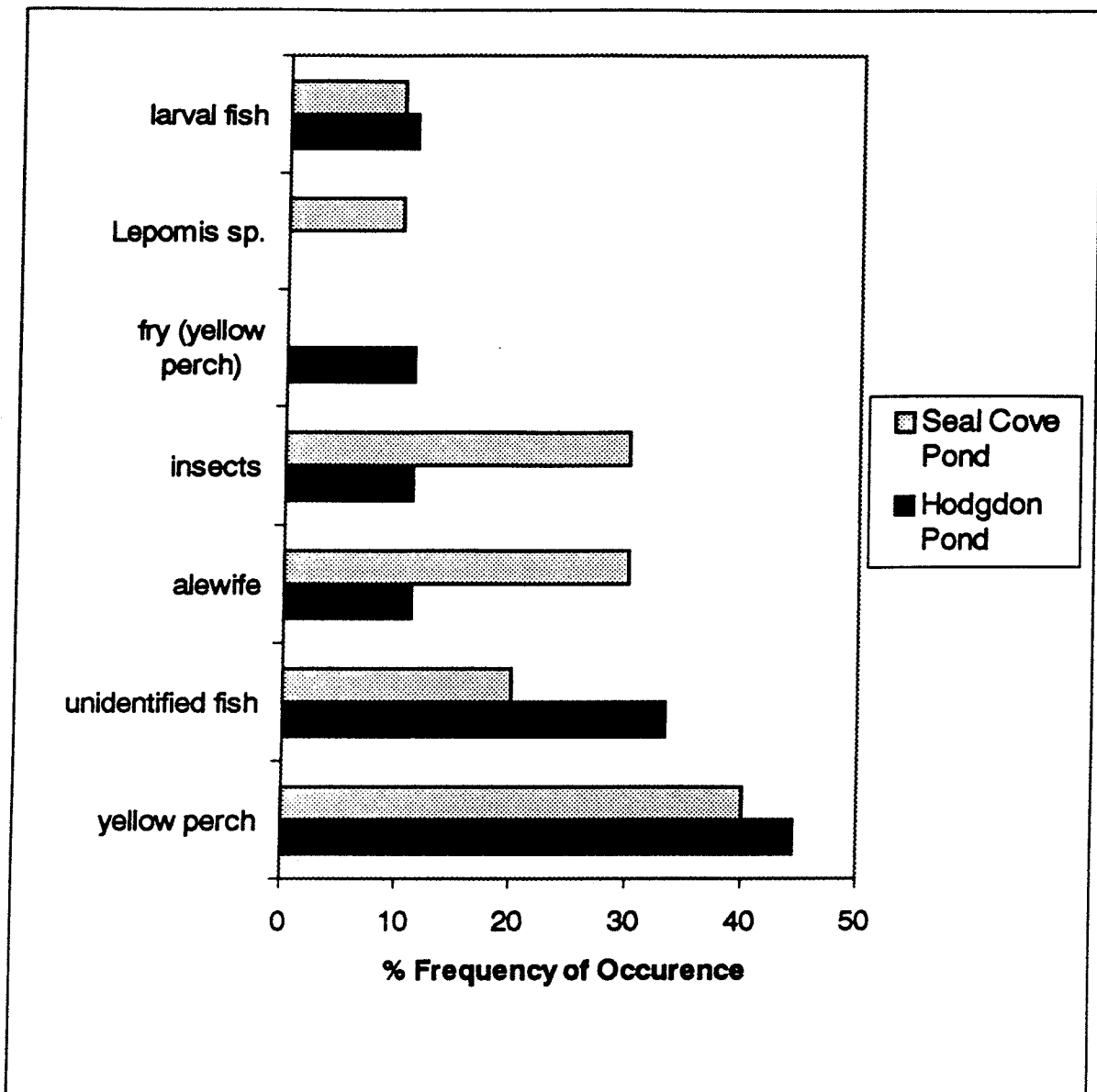


Figure 1. Percent frequency of occurrence of food items in the diet of smallmouth bass. Frequency expressed as a percentage of stomachs containing food. (Hodgdon Pond: N=16, 7 empty; Seal Cove Pond: N=18, 8 empty).

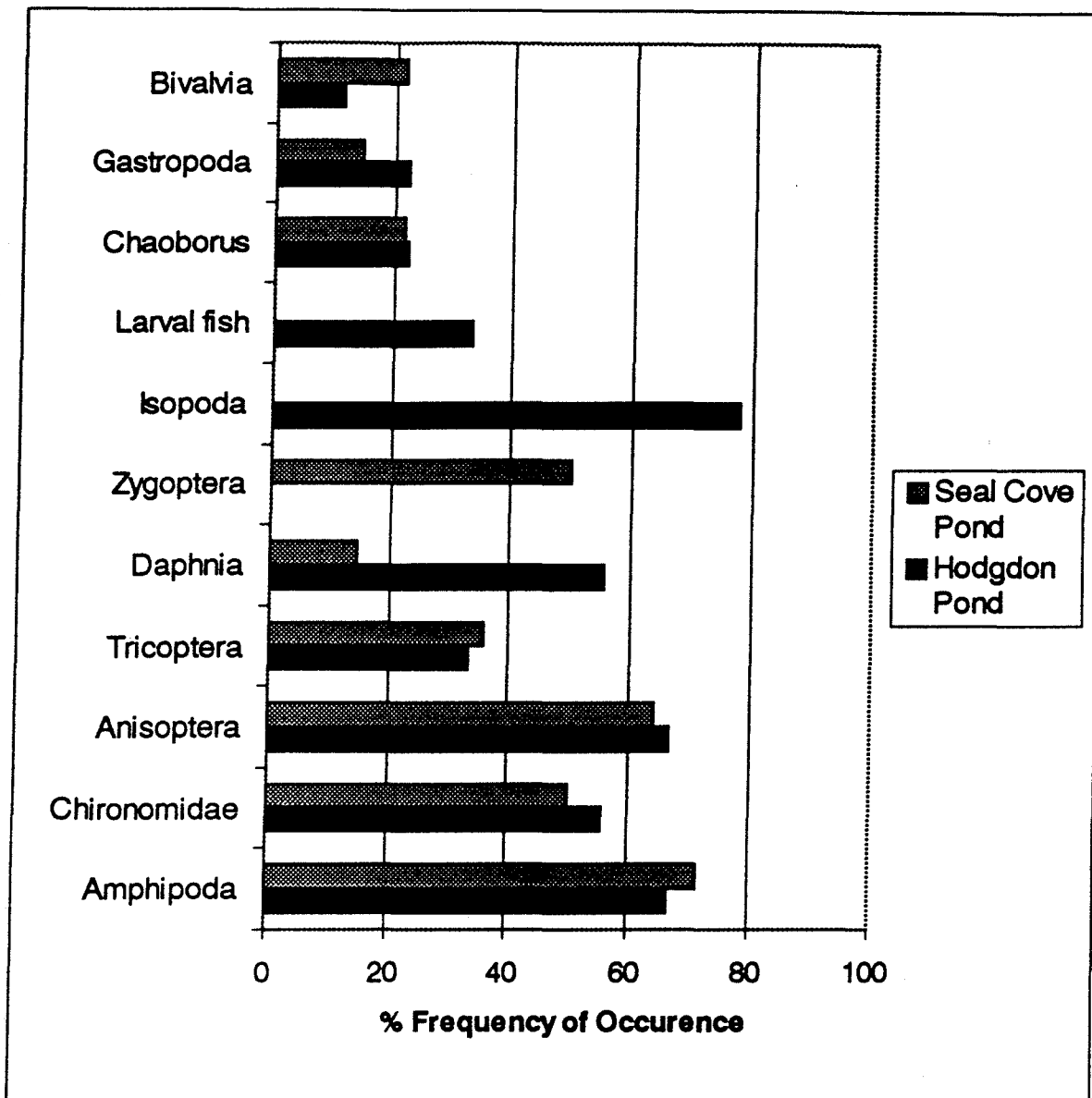


Figure 2. Percent frequency of occurrence of food items in the spring diet of yellow perch. Frequency expressed as a percentage of stomachs containing food. (Hodgdon Pond: N=9, Seal Cove Pond: N=14).

Therefore, these two taxa and isopods were selected for seasonal collection and mercury analysis.

Analysis of summer diets of yellow perch revealed an increased importance of Amphipoda and Chironomidae in both ponds (Fig. 3). A striking feeding shift away from Isopoda occurred in yellow perch from Hodgdon Pond, with this taxon being absent from the summer diet of this species. However, larval fishes remained relatively important in the diet of yellow perch in Hodgdon Pond, and unimportant in the diet of this species in Seal Cove Pond. Anisoptera occurred much less frequently in the summer diets of yellow perch from both ponds.

Analysis of the overall diets (combined seasonal data) of yellow perch revealed Amphipoda to be the most abundant dietary item, followed closely by Chironomidae (Fig. 4). Anisoptera were the third most important taxon in the overall diets of yellow perch in both ponds. Isopoda and larval fishes were moderately important in the overall diet of yellow perch in Hodgdon Pond, and almost non-existent in the overall diet of this species in Seal Cove Pond.

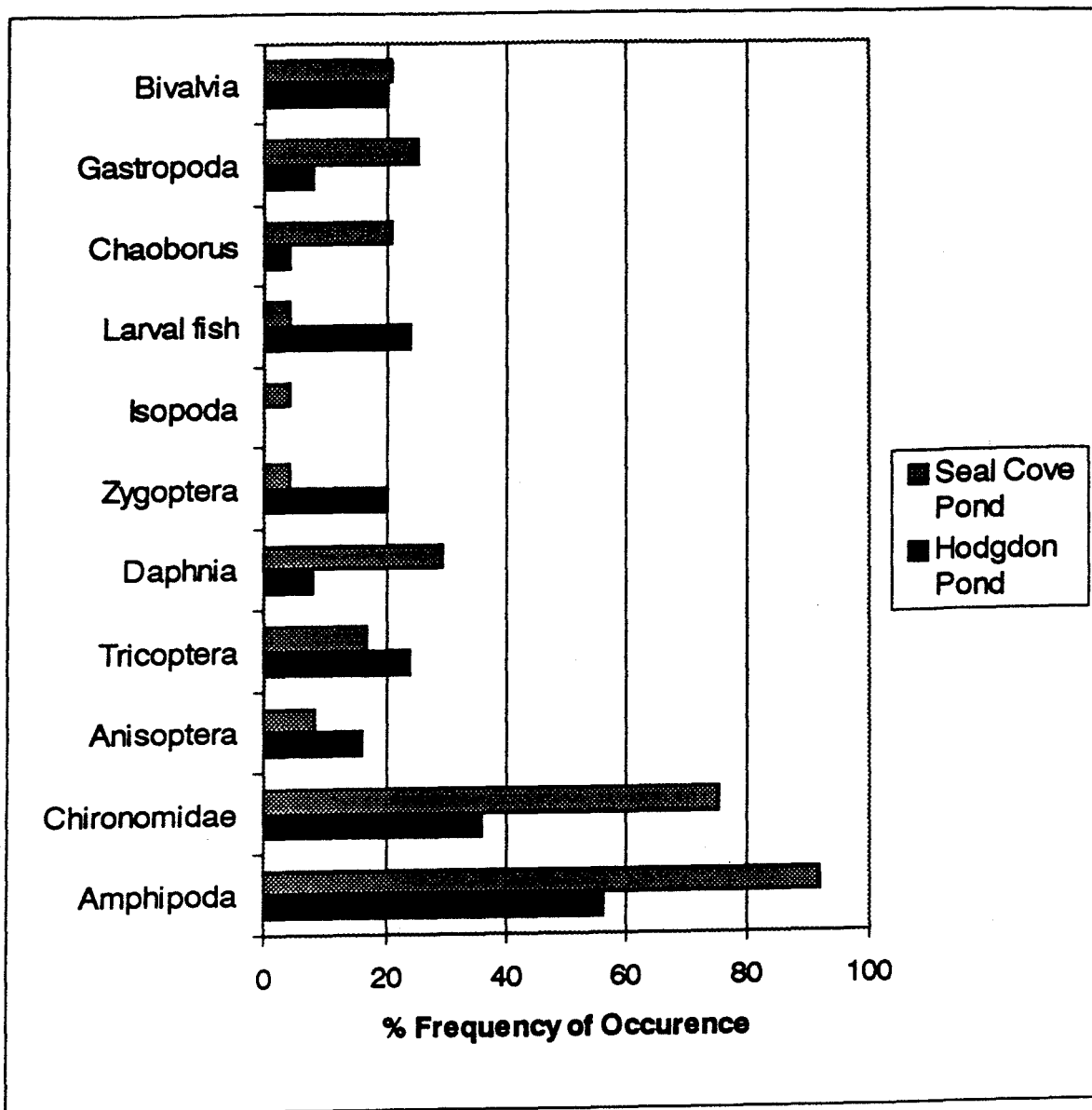


Figure 3. Percent frequency of occurrence of food items in the summer diet of yellow perch. Frequency expressed as a percentage of stomachs containing food. (Hodgdon Pond: N=25, Seal Cove Pond: N=24).

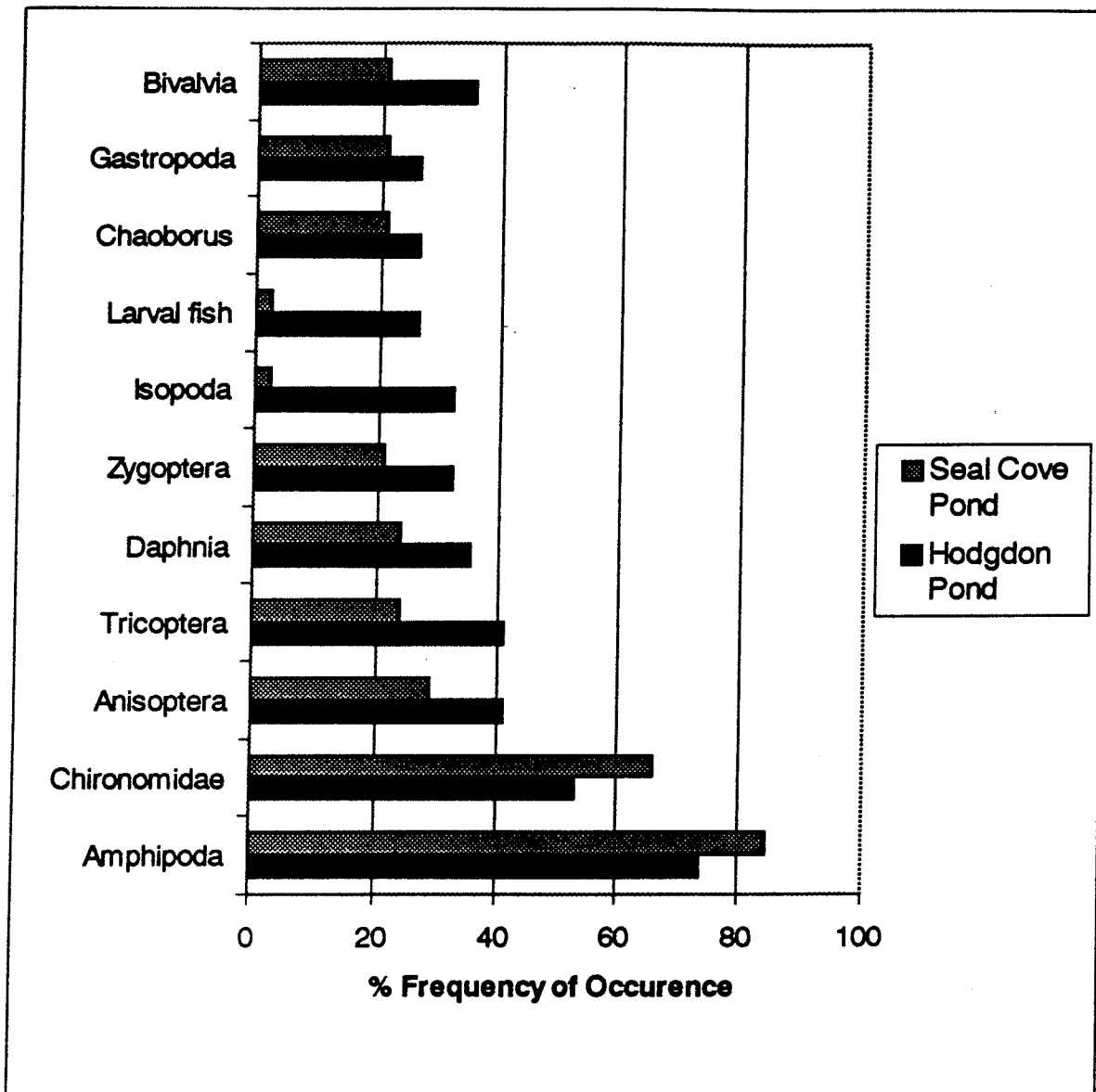


Figure 4. Percent frequency of occurrence of food items in the overall diet of yellow perch. Frequency expressed as a percentage of stomachs containing food. (Hodgdon Pond: N=34, Seal Cove Pond: N=38).

Forage Fish

In the spring, total mercury concentrations in yellow perch were significantly higher in Hodgdon Pond than in Seal Cove Pond (Table 6, ANOVA, $P=0.0009$). In the summer, total mercury concentrations in yellow perch from both ponds were not different. There was a significant seasonal difference in total mercury concentrations for yellow perch in Hodgdon Pond (ANOVA, $P=0.0025$), but not in Seal Cove Pond. Overall, total mercury concentrations in yellow perch were significantly higher in Hodgdon Pond than in Seal Cove Pond (ANOVA, $P=0.0211$).

Methylmercury concentrations in yellow perch from Hodgdon Pond were significantly higher than from Seal Cove Pond in the spring (Table 6; ANOVA, $P=0.0178$); however, in the summer methylmercury concentrations were not different (Table 6, ANOVA, $P>0.05$). There was a significant seasonal difference in methylmercury concentrations in yellow perch from Seal Cove Pond (ANOVA, $P=0.0167$, summer > spring), but not in Hodgdon Pond. There was no difference in overall methylmercury concentrations in yellow perch between ponds (ANOVA, $P>0.05$).

Mean total mercury and methylmercury concentrations in age 0+ yellow perch from Hodgdon Pond were not significantly different from concentrations in age 0+ alewives from Seal Cove Pond (Table 6; t-test, $P>0.05$).

Stable Isotope Analyses

Yellow perch collected from Hodgdon Pond in the spring had a significantly higher ratio of the stable isotope $\delta^{15}\text{N}$ than did yellow perch collected in the spring from

Table 6. Mercury concentrations in forage fishes (THg = total mercury; MeHg = methylmercury; mean concentrations are least square means; concentrations are in wet wt.; data are whole body concentrations).

Site	Species	N	<u>Length (mm)</u>		<u>Weight (g)</u>		<u>Age (yr.)</u>		<u>THg ($\mu\text{g g}^{-1}$)</u>		<u>MeHg ($\mu\text{g g}^{-1}$)</u>	
			X	Range	X	Range	X	Range	X	Range	X	Range
Hodgdon Pond	Yellow perch (young of year)	10	53.1	46-56	1.7	1.2-2.4	0+	0+	0.058	0.031-0.085	0.057	0.040-0.083
	yellow perch (spring)	10	138	113-174	29	13-57	3.6	3-4	0.18 ^a	0.14-0.23	0.16 ^a	0.11-0.21
	yellow perch (summer)	10	138	105-180	28	12-54	3.1	2-5	0.13 ^b	0.07-0.19	0.13 ^{ab}	0.10-0.19
Seal Cove Pond	Alewife (young of year)	10	75.3	68-80	3.4	2.4-4.2	0+	0+	0.055	0.033-0.073	0.059	0.052-0.067
	yellow perch (spring)	10	139	102-160	29	10-45	3.7	2-5	0.12 ^b	0.078-0.21	0.11 ^b	0.061-0.18
	yellow perch (summer)	10	128	111-159	22	13-38	3.2	2-4	0.13 ^b	0.080-0.25	0.16 ^a	0.10-0.24

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

Seal Cove Pond (t-test, $P=0.0002$), with mean values of 7.47 ($N=9$) and 6.69 ($N=10$), respectively.

Macroinvertebrates

Mercury concentrations were not significantly different among dragonfly larvae genera (suborder Anisoptera) or families; therefore, data were pooled for reporting purposes. Total mercury concentrations in dragonfly larvae were significantly higher in Hodgdon Pond than in Seal Cove Pond in the spring (ANOVA, $P=0.0028$), but were not different during the summer (Table 7). In Hodgdon Pond, total mercury in dragonflies was greater in spring than in summer (ANOVA, $P=0.0411$), whereas in Seal Cove Pond it was lower in spring than in summer (Table 7; ANOVA, $P=0.0459$).

Methylmercury concentrations in dragonfly larvae were not significantly different between seasons within each pond or between ponds in each season (Table 7; ANOVA, $P>0.05$ for all comparisons). However, the overall methylmercury and total mercury concentrations (combined seasonal data) in dragonfly larvae from Hodgdon Pond were significantly greater than the overall methylmercury and total mercury concentrations in dragonfly larvae from Seal Cove Pond (ANOVA, $P=0.0473$ and 0.0341 , respectively).

The overall total mercury and methylmercury concentrations in amphipods from Hodgdon Pond did not differ between spring and summer, and were not different from Seal Cove Pond (t-test, $P>0.05$, Table 7). Amphipods in Seal Cove Pond had higher total mercury and lower methylmercury concentrations in the summer than in the spring. Total and methylmercury concentrations in isopods from Hodgdon Pond were slightly lower in summer, than in spring (Table 7).

Table 7. Seasonal mercury concentrations in homogenized aggregates of invertebrates. (Num. = total number of individuals in aggregates, (number of aggregates); THg = total mercury, ng g⁻¹ dw; MeHg = methylmercury, ng g⁻¹ dw; % MeHg = ratio of methylmercury to total mercury; statistical comparisons are within taxa).

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

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

Plankton

In Hodgdon Pond, methylmercury concentrations were higher in summer plankton than in spring plankton (Table 8); however, only one summer sample was obtained for analysis. Methylmercury concentrations in plankton from Seal Cove Pond did not differ between spring and summer. Total mercury concentration in plankton from Seal Cove was higher in summer (Table 8), but only one sample could be analyzed. The differences in mercury concentrations between ponds were not significant (ANOVA, $P>0.05$).

Sediment

There were no significant differences in total or methylmercury concentrations in sediment between ponds, seasons, or locations (i.e., littoral and profundal zones), within and between ponds (Table 9; ANOVA, $P>0.05$ for all comparisons). The only difference in the ratio of methylmercury to total mercury was in Hodgdon Pond, where spring profundal sediment had a higher ratio than summer profundal sediment (ANOVA, $P=0.0018$).

Percent volatile organic content in profundal sediment was greater in Hodgdon Pond than in Seal Cove Pond, when seasonal data were pooled (Kruskal-Wallis one-way ANOVA, $P=0.0209$). No other differences were found in percent volatile organic matter in sediment ($P>0.05$ for all comparisons). Percent volatile organic matter in sediment was strongly correlated with both total and methylmercury in Hodgdon Pond (Spearman rank correlation coefficient, $r_s=0.797$, $P=0.0019$ and $r_s=0.699$, $P=0.0114$, respectively), but not in Seal Cove Pond ($r_s=-0.081$, $P=0.8110$ and $r_s=0.545$, $P=0.0666$, respectively).

Table 8. Seasonal plankton mercury concentrations. (THg = total mercury, ng g⁻¹ dw; MeHg = methylmercury, ng g⁻¹ dw; % MeHg = ratio of methylmercury to total mercury).

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

Table 9. Seasonal sediment mercury concentrations. (%VO = percent volatile organic; THg = total mercury, ng g⁻¹ dw; MeHg = methylmercury, ng g⁻¹ dw; % MeHg = ratio of methylmercury to total mercury).

Site	Season	Location	N	%VO		THg		MeHg		%MeHg	
				X	SD	X	SD	X	SD	X	SD
Hodgdon Pond	Spring	Littoral	4	25.7	3.0	138.5	21.1	6.96	1.81	4.97	0.58
		Profundal	2	28.8	0.3	132.6	30.7	8.06	0.87	6.07	0.23
	Summer	Littoral	4	29.8	10.3	121.0	78.5	4.76	2.04	4.61	1.95
		Profundal	2	28.9	1.0	210.4	41.4	6.32	1.13	3.01	0.05
	Combined	Littoral	8	27.7	7.4	129.8	54.0	5.86	2.14	4.79	1.35
		Profundal	4	28.9	0.6	171.5	51.2	7.19	1.30	4.54	1.77
Seal Cove Pond	Spring	Littoral	4	23.2	10.0	91.3	25.3	6.76	3.14	8.31	5.49
		Profundal	2	25.8	0.9	151.9	53.2	10.39	1.23	7.13	1.69
	Summer	Littoral	4	17.7	9.6	124.1	39.2	4.80	0.90	4.52	2.83
		Profundal	2	26.1	0.2	104.1	26.7	6.87	1.11	6.96	2.85
	Combined	Littoral	8	20.4	8.9	110.0	35.9	5.78	2.38	6.14	4.26
		Profundal	4	25.9	0.5	128.0	44.1	8.63	2.24	7.05	1.92

Water

Total mercury in water averaged 3.08 ng L^{-1} in Hodgdon Pond and 2.31 ng L^{-1} in Seal Cove Pond (Table 10). Methylmercury in water averaged 0.212 ng L^{-1} in Hodgdon Pond and 0.182 ng L^{-1} in Seal Cove Pond. There were no significant differences in mercury concentrations between ponds, seasons, or locations (i.e., epilimnion, hypolimnion), (ANOVA, $P > 0.05$ for all comparisons).

Temperatures in Hodgdon Pond were warmer than in Seal Cove Pond in late May, with approximately 12°C near the bottom and 16°C near the surface (Fig. 5) compared with approximately 10°C and 12°C , respectively, in Seal Cove Pond (Fig. 6). Temperatures in Hodgdon Pond increased to approximately 16°C near the bottom and 23°C near the surface by late August. Hypolimnetic water in Seal Cove Pond remained cooler than in Hodgdon Pond, with a temperature of 13°C by late August (Fig. 6). Average, monthly littoral zone water temperatures did not differ between the two ponds in May or from September to early December (Fig. 7, ANOVA, $P > 0.05$). However, Seal Cove Pond had warmer littoral zone water temperatures during June, July, and August (ANOVA, $P = 0.0001$).

In May, oxygen concentrations near the bottom were higher in Hodgdon Pond than in Seal Cove Pond, with concentrations of 7 mg L^{-1} and 4.5 mg L^{-1} , respectively. Oxygen concentrations near the surface were the same in both ponds in May, with concentrations of approximately 10 mg L^{-1} . In Hodgdon Pond, anoxic conditions developed near the bottom by early July and rose to a depth of approximately 4.5 m by late August (Fig. 5). In contrast, anoxic conditions in Seal Cove Pond did not develop until late August (Fig. 6).

Table 10. Seasonal water mercury concentrations. (Mercury concentrations are in ng L⁻¹)

Pond	Season	Depth (m)	N	THg		N	MeHg	
				X	SD		X	SD
Hodgdon								
	Spring	0.5	1	2.94	---	1	0.180	---
		5.5	1	2.84	---	1	0.160	---
		Average	2	2.89	0.07	2	0.170	0.014
	Summer	0.5	1	2.18	---	1	0.324	---
		5.5	1	4.35	---	1	0.183	---
		Average	2	3.27	1.53	2	0.254	0.010
	Combined		4	3.08	0.91	4	0.212	0.076
Seal Cove								
	Spring	0.5	1	2.21	---	1	0.165	---
		13.0	1	4.76	---	1	0.275	---
		Average	2	3.49	1.80	2	0.220	0.078
	Summer	0.5	1	0.91	---	1	0.146	---
		13.0	1	1.34	---	1	0.140	---
		Average	2	1.13	0.30	2	0.143	0.004
	Combined		4	2.31	1.72	4	0.182	0.063

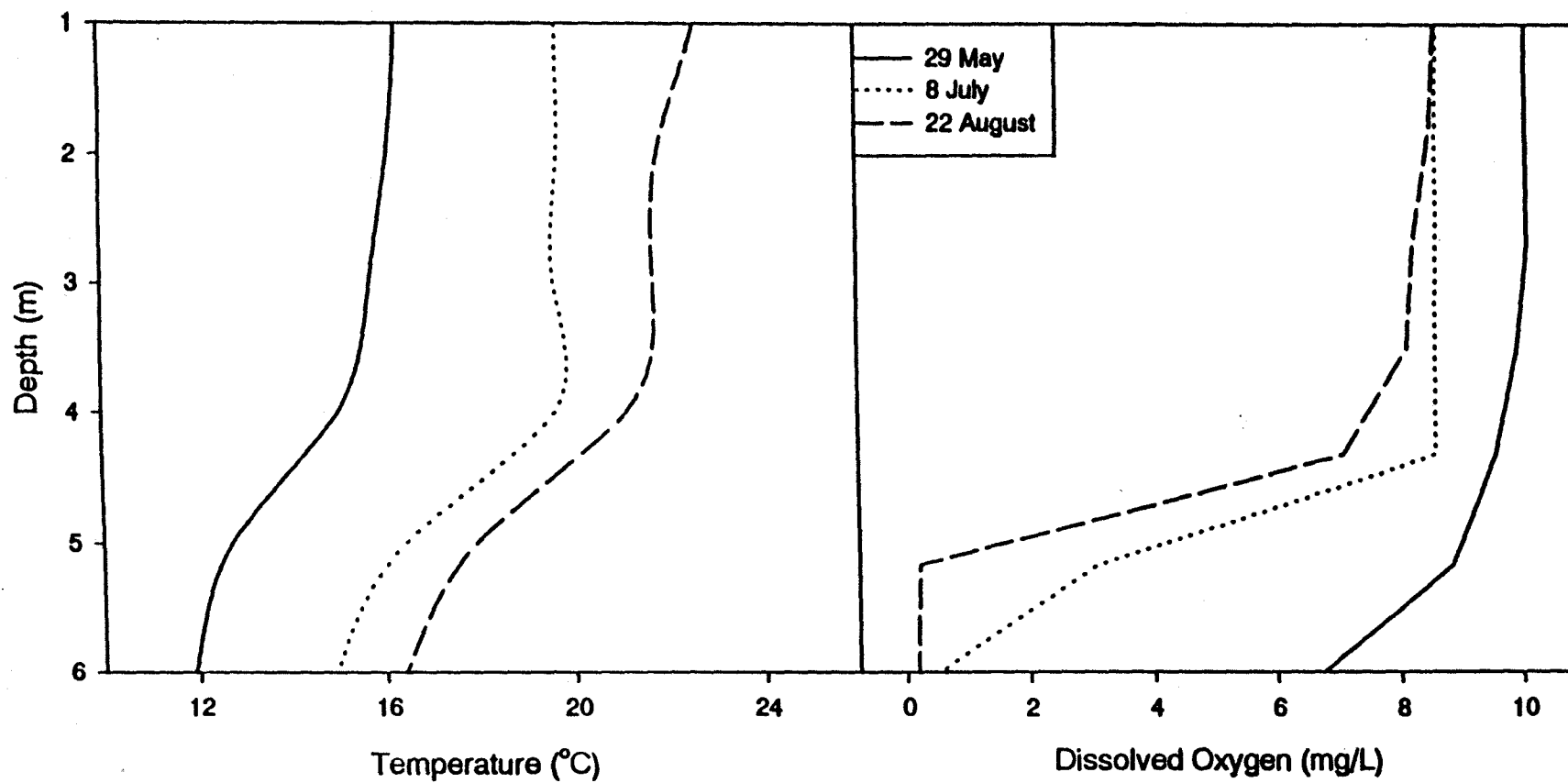


Figure 5. Temperature and oxygen profiles for Hodgdon Pond from May to August 1996.

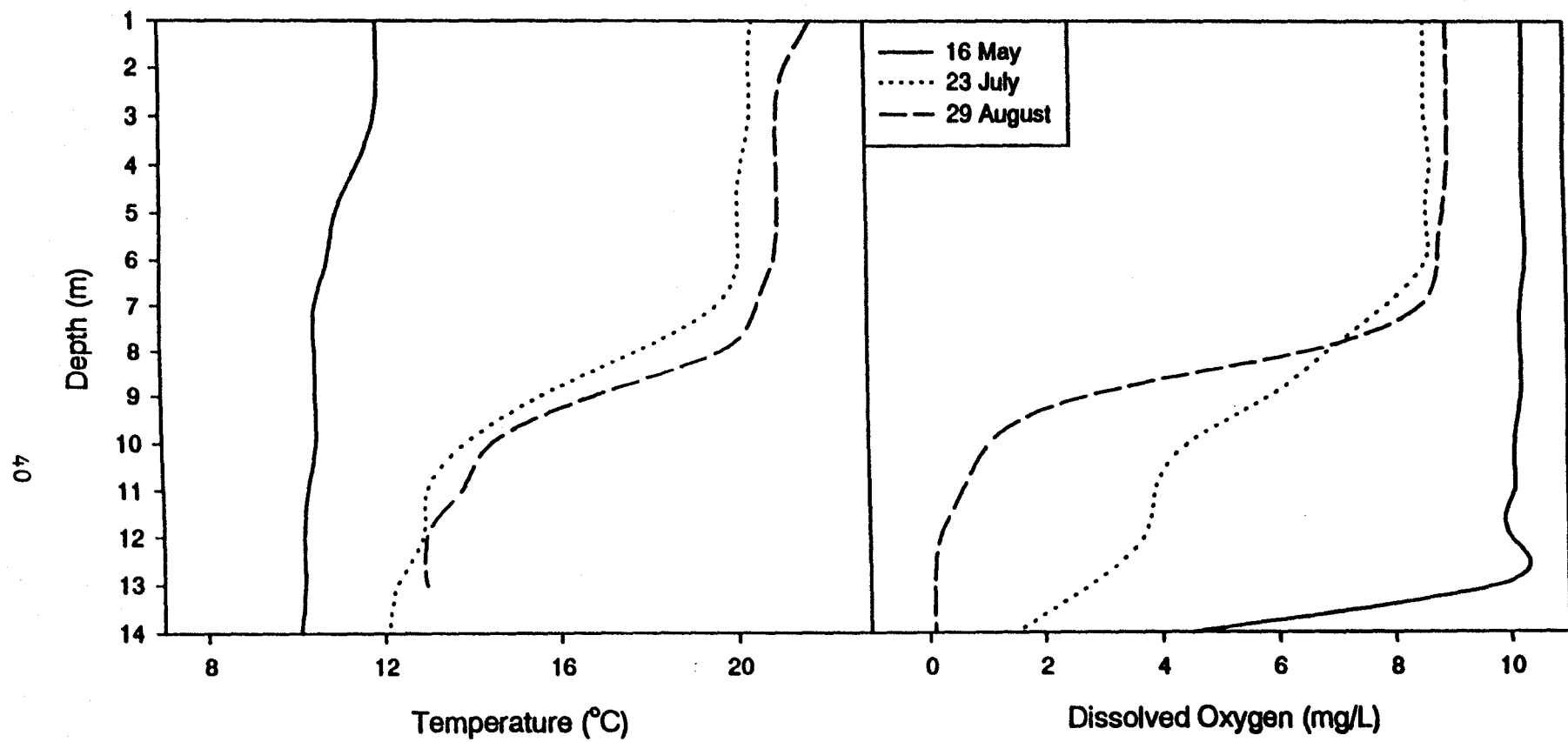


Figure 6. Temperature and oxygen profiles for Seal Cove Pond from May to August 1996.

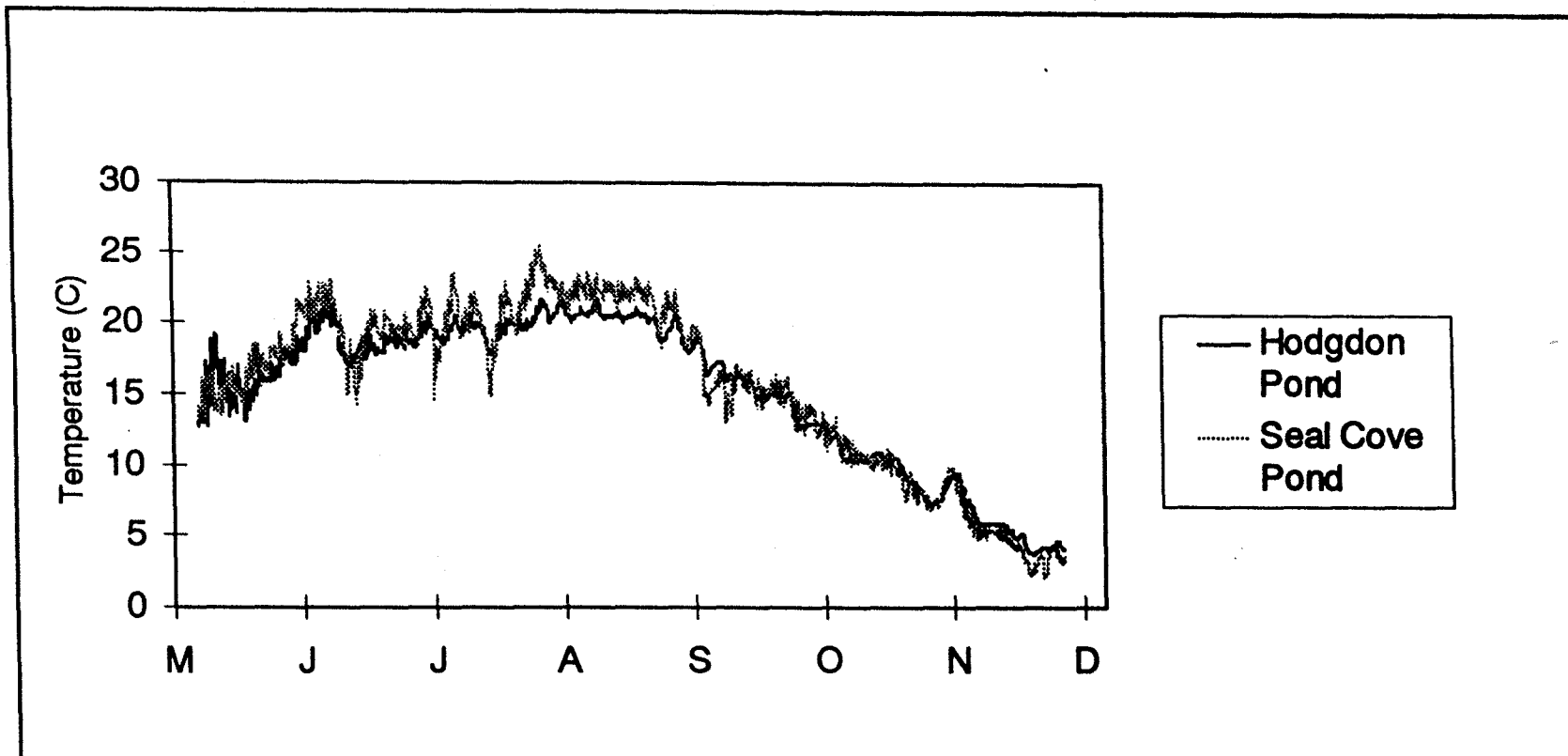


Figure 7. Littoral zone water temperatures (1.5 m depth) from mid-May to early December. Water temperatures at 3.0 m depth not shown, but there were no differences in temperature between depths.

Bioconcentration Factors

Bioconcentration factors [$BCF = \log C_b/C_w$, where C_b = concentration in biota and C_w = concentration in water) were calculated for each trophic levels on a wet weight basis. Bioconcentration factor calculations indicate that methylmercury is bioconcentrated to a greater extent than is total mercury (Fig. 8). Bioconcentration factors for methylmercury are generally higher for each trophic level within Hodgdon Pond. In contrast, bioconcentration factors for total mercury are higher at lower trophic levels within Seal Cove Pond.

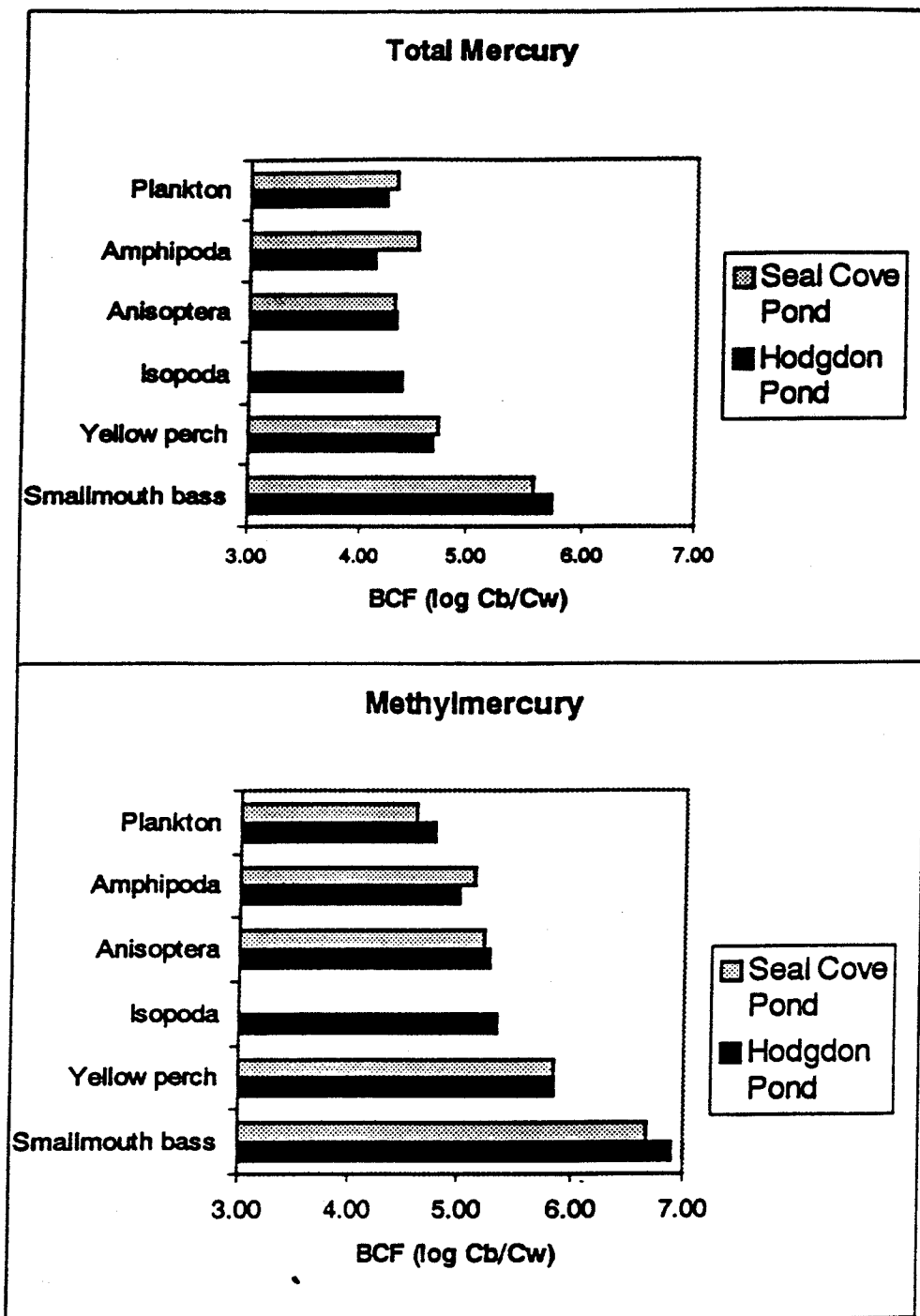


Figure 8. Bioconcentration factors [$BCF = \log C_b/C_w$, where C_b = conc. in biota and C_w = conc. in water) for total and methylmercury in various biota from both ponds. Plankton calculations are based on 90% moisture, Amphipoda and Isopoda 80%, and Anisoptera 85%. Smallmouth bass methylmercury concentrations based on assumption of 95% methylmercury (Bloom, 1992).

DISCUSSION

Results of this study suggest that salmonid fish species on Mount Desert Island generally have lower mercury concentrations than those species in other regions of Maine. For example, mercury concentrations in lake trout from northern Maine lakes range from 0.45-0.94 $\mu\text{g g}^{-1}$ (Normandeau Associates, 1978) and 0.34-0.84 $\mu\text{g g}^{-1}$ (Akielaszek and Haines, 1981). A statewide mean of 0.45 $\mu\text{g g}^{-1}$ found by Stafford and Haines (1997). In contrast, mercury concentrations in lake trout analyzed in this study ranged from 0.21-0.32 $\mu\text{g g}^{-1}$. Mercury concentrations in landlocked salmon were also lower in this study, with values ranging from 0.11-0.37 $\mu\text{g g}^{-1}$, whereas the statewide mean for this species is 0.35 $\mu\text{g g}^{-1}$ (Stafford and Haines, 1997).

Non-salmonid species on Mount Desert Island generally had similar or lower mercury concentrations than the state average for each species. White perch averaged 0.69 $\mu\text{g g}^{-1}$ in this study, which is very similar to the state average of 0.65 $\mu\text{g g}^{-1}$. Chain pickerel in this study have much lower mercury concentrations than the statewide mean for chain pickerel reported by Stafford and Haines (1997). However, the size differences of chain pickerel collected in both studies may explain the difference in mercury concentrations. Because it is well established that larger, older individuals of a predatory fish species generally have higher mercury concentrations (MacCrimmon *et al.*, 1983; Lange *et al.*, 1993; Stafford and Haines, 1997), and the chain pickerel collected in this study were generally smaller individuals than those collected in the Stafford and Haines study, then the average concentration of mercury in chain pickerel on Mount Desert Island is probably not substantially different from the statewide average for this species. The one species on Mount Desert Island that seems to have higher concentrations of

mercury then average for the state of Maine is smallmouth bass. Concentrations in smallmouth bass ranged from 0.41-3.86 $\mu\text{g g}^{-1}$ in this study, in contrast to the range of 0.31- 1.12 $\mu\text{g g}^{-1}$ reported by Stafford and Haines (1997). However, the trend of high concentrations in this species was not consistent for the entire island. Only two of the three ponds (Hodgdon Pond and Seal Cove Pond) containing smallmouth bass had elevated levels of mercury in this species. The third pond, Long Pond had lower mercury concentrations in smallmouth bass than the state average, with an average concentration of 0.56 $\mu\text{g g}^{-1}$ (Table 3).

In the food chain study, mercury concentrations in water, sediment, and biota generally agree with concentrations found in other studies. Watras *et al.* (1994) reported epilimnetic total and methylmercury concentrations in seven Wisconsin lakes ranging from 0.7-2.1 ng L^{-1} and 0.05-0.3 ng L^{-1} , respectively. Epilimnetic mercury concentrations in the present study agree well with these ranges. The sediment mercury concentrations are similar to concentrations found in previous surveys of mercury in freshwater sediments (Andersson and Håkanson, 1992; Tremblay *et al.*, 1995; Tremblay and Lucotte, 1997). For example, Tremblay *et al.* (1995) reported a mean total mercury concentration of 80 ng g^{-1} dw in surficial sediment from 73 Canadian lakes, and Andersson and Håkanson (1992) reported a mean concentration of 238 ng g^{-1} dw in surficial sediment from 25 Swedish lakes.

The mercury concentrations in yellow perch from Hodgdon Pond were not abnormally high, as one might expect from the high concentrations in smallmouth bass and American eel. In fact, the concentrations were lower in yellow perch than were of similar size and age to those analyzed in other studies. For example, Grieb *et al.* (1990)

found age 3 and age 4 yellow perch from 35 lakes in northern Michigan to have mean mercury concentrations of $0.20 \mu\text{g g}^{-1}$ and $0.25 \mu\text{g g}^{-1}$ (wet wt.), respectively.

Additionally, Wren and MacCrimmon, (1986) found mercury concentrations in yellow perch from two adjacent, uncontaminated lakes in Ontario, Canada, to average $0.24 \mu\text{g g}^{-1}$ and $0.34 \mu\text{g g}^{-1}$, with respective mean ages of 2.7 and 4.2 years. In contrast, in this study mean mercury concentrations of similar age yellow perch ranged from 0.12 to $0.18 \mu\text{g g}^{-1}$ (Table 6).

Total mercury concentrations in dragonfly larvae in the present study ranged from 208-560 ng g^{-1} dw (dry weight), and are similar to concentrations found in other studies. Previous findings of odonate total mercury concentrations have ranged from 135-358 ng g^{-1} dw in two Swedish mesotrophic lakes (Parkman and Meili, 1993), to 136 ng g^{-1} in northern Québec (Canada) lakes, and from 283-615 ng g^{-1} in northern Québec hydroelectric reservoirs (Tremblay *et al.*, 1996). In the Tremblay *et al.* (1996) study, odonate methylmercury concentrations ranged from 283 to 615 ng g^{-1} , and accounted for 75-100% of the total mercury. Likewise, in a study comparing natural lakes and reservoirs by Tremblay and Lucotte (1997), the mean proportion of methylmercury to total mercury in odonates was 85%, and concentrations ranged from approximately 100 to 400 ng g^{-1} . In the present study, mean methylmercury concentrations in odonates ranged from 130 to 319 ng g^{-1} and the proportion of methylmercury to total mercury ranged from 53 to 83%.

Mean total mercury concentrations in pooled isopod samples from the present study are comparable to values from a previous study by Parkman and Meili (1993), in which they found concentrations ranging from 63 to 394 ng g^{-1} in eight Swedish lakes.

The concentrations in isopods from Hodgdon Pond ranged from 306 to 448 ng g⁻¹ and agree more with the data from the acidic dystrophic lakes in the Parkman and Meili study, even though Hodgdon Pond is a circumneutral mesotrophic pond. The seasonal change in mercury concentrations in Isopods, from higher values in the spring to lower values in late summer, was also observed by Parkman and Meili. They suggest that this seasonal drop in concentration is directly related to the annual appearance of a new generation in summer. This observation seems to apply to the present study as well, for personal observations indicated that the summer sample was composed of smaller sized individuals.

Total and methylmercury concentrations in plankton in the present study are similar to the ranges found by Watras and Bloom (1992) of 300-400 ng g⁻¹ dw and 40-150 ng g⁻¹ dw, respectively for phytoplankton, and 280-375 ng g⁻¹ dw and 100-270 ng g⁻¹ dw, respectively for zooplankton. The proportion of methylmercury to total mercury in the present study ranged from 12-21%, which agrees well with the results of the above study, as well as with the 1-3% for phytoplankton and 16-20% for zooplankton presented by Mason and Sullivan (1997).

Bioconcentration factor (BCF) calculations agree with the laboratory work of Boudou and Ribeyre (1981), and the field work of Watras and Bloom (1992), who both showed that methylmercury is more efficiently transferred up aquatic food chains, than is inorganic mercury. Bioconcentration of methylmercury is also greater at the base and at the top of the food chain in Hodgdon Pond. However, the degree of concentration is the same in yellow perch from both ponds. It is interesting to note the degree of

concentration in isopods. This taxon has a slightly greater BCF than does Anisoptera, which should have a greater BCF because of its predaceous nature.

Water chemistry results showed that pH is not a contributing factor leading to the elevated mercury concentrations in fish from Hodgdon Pond. Although many studies have found elevated mercury concentrations in fish from low pH lakes (Håkanson *et al.*, 1988; Grieb *et al.*, 1990; Lindqvist *et al.*, 1991; Watras and Bloom, 1992; Haines *et al.*, 1995), both Hodgdon Pond and Seal Cove Pond are circumneutral, with a pH of 6.96. Thus, high mercury concentrations in fishes from Hodgdon Pond are not related to acidic water, and the significant differences in mercury concentration in predatory fishes from these two ponds is not explained by differences in pH.

Warmer water temperatures are generally believed to promote increased mercury concentrations in fishes because of increased methylmercury production in sediments (Callister and Winfrey, 1986; Winfrey and Rudd, 1988) and ultimately from an increased assimilation of mercury into organisms at the base of aquatic food chains (Boudou *et al.*, 1979). Additionally, Bodaly *et al.* (1993) found that methylation rate was positively dependent on epilimnetic water temperature in six Canadian lakes. Therefore, I hypothesized that Hodgdon Pond may have warmer water temperatures in the littoral zone than does Seal Cove Pond. However, contrary to my expectations, Seal Cove Pond had significantly warmer littoral zone water temperatures during the months of June, July, and August, and there was no difference in littoral zone water temperatures between the ponds during May and after August. However, the profundal zone and surface water in Hodgdon Pond did warm sooner than in Seal Cove Pond, and the profundal zone remained 3-4°C warmer throughout the summer. Therefore, the rate of methylmercury

production in profundal sediment in Hodgdon Pond should be greater than in Seal Cove Pond. However, comparative analyses of both total and methylmercury in sediments from both ponds failed to show any differences between the ponds.

Alternatively, Watras *et al.* (1995a) found that hypolimnetic water just below the thermocline was an important site for methylmercury production in some lakes. They found that a layer of bacterioplankton near the top of the anoxic hypolimnion was responsible for more methylmercury production than from profundal sediments. A layer of anoxic water began to form in Hodgdon Pond in early July, and water below a depth of approximately 5m was completely anoxic by August. In contrast, the anoxic zone in Seal Cove Pond developed much later and at a depth of approximately 10m. These temporal and depth differences in the anoxic zone formation may lead to enhanced production of methylmercury within this zone in Hodgdon Pond. Although water analyses of total and methylmercury failed to show any differences between the ponds, only a few samples were collected and those were only from depths of 0.5m from the surface and approximately 0.5m from the bottom of each pond. Therefore, a layer of enhanced methylmercury formation in the water column of Hodgdon Pond may exist, but remains to be demonstrated.

The methylmercury concentration in hypolimnetic water from Hodgdon Pond in the summer was twice as high as in the spring, whereas hypolimnetic water from Seal Cove Pond contained only one-half as much methylmercury as in the spring. Because Watras *et al.* (1995a) showed that the zone of methylation in their study lake only spanned a layer that was 0.5m to 1.0m thick, the water sampling conducted in Hodgdon Pond may not have been truly representative of the amount of methylmercury production

occurring. Thus, the warmer profundal zone water in Hodgdon Pond may be an important part of the explanation for why fish from this pond have such elevated mercury concentrations.

Humic matter in the water column (as indicated by color and dissolved organic carbon) is an additional factor that may contribute to enhanced bioavailability of methylmercury in Hodgdon Pond. The concentration of dissolved organic carbon in Hodgdon Pond is 1.5 times greater than the concentration in Seal Cove Pond, and color is 3 times greater. A positive relationship exists between DOC or color and mercury in fish. McMurtry *et al.* (1989) demonstrated a significant positive relationship between DOC in Ontario lakes, and mercury in smallmouth bass and lake trout. Haines *et al.* (1992) reported a high positive correlation between color in six Russian lakes and mercury in perch (*Perca fluviatilis*). Watras *et al.* (1995b) found that DOC explained the most variation in both total and methylmercury in the surface waters of 19 northern Wisconsin lakes. However, there is evidence that DOC has the opposite effect on methylmercury at lower trophic levels. Back and Watras (1995) found an inverse relationship between DOC and the bioconcentration of methylmercury in zooplankton in twelve northern Wisconsin lakes. In addition, Watras *et al.* (1994) observed a significant decrease in bioconcentration of methylmercury in one-year old yellow perch with increasing DOC. Thus, the effect of DOC on mercury bioaccumulation seems to vary with trophic position, and the full explanation remains unclear.

The differences in mercury concentrations in American eel, chain pickerel, and smallmouth bass between the ponds shows that a significant difference in mercury bioaccumulation exists between these ponds. Because mercury concentrations in yellow

perch and the ratio of the stable isotope $\delta^{15}\text{N}$ (an indicator of trophic level) in yellow perch (the primary diet item of smallmouth bass in both ponds) were significantly higher in Hodgdon Pond in the spring, there seems to be a difference in the trophic position of this species between these ponds. A significant positive relationship between mercury in fish and the stable isotope $\delta^{15}\text{N}$ has been shown in previous work by Kidd *et al.* (1995). However, the stable isotope results should be viewed with caution because a lake-specific baseline $\delta^{15}\text{N}$ signature was not obtained to correct for variability in the $\delta^{15}\text{N}$ of primary producers (Vander Zanden *et al.*, 1997). Nonetheless, the dietary analysis seems to support the finding of varying trophic positions. The spring diet of yellow perch from Hodgdon Pond contained isopods and larval fish, which were absent from the diet of yellow perch from Seal Cove Pond. The presence of larval fish and possibly isopods as well, adds an additional link in the food chain, which leads to the enhanced biomagnification of mercury in this pond.

The seasonal change in diet and mercury concentrations in yellow perch from Hodgdon Pond suggest a link to the presence of isopods in the spring diet and their subsequent absence from the summer diet. Although growth dilution could cause a seasonal drop in mercury concentration, the statistical comparisons were made using size-adjusted means, which should have removed this bias. Therefore, the seasonal difference in mercury concentration is probably not related to growth, but rather to dietary differences and/or the increase in productivity related to spring run-off and overturn.

Isopods have an omnivorous diet and are generally not active predators on other benthic invertebrates (Wetzel, 1975). Therefore, the high mercury concentrations in isopods are surprising, because they are comparable with concentrations in predaceous

odonates. The fact that the methylmercury concentrations are comparable between these two taxons is strong evidence that isopods in Hodgdon Pond are occupying a trophic level equal to that of the odonates. If these organisms are not active predators, then they must be accumulating methylmercury from detrital animal remains.

Amphipods are omnivorous substrate feeders that feed on bacteria, algae, and animal and plant remains (Wetzel, 1975). Therefore, the amphipod concentrations found in this study, which are much lower than odonate concentrations, are as expected for an organism feeding at a lower trophic level. The proportion of methylmercury to total mercury was also much lower than in the odonates, which also is as expected for a non-predaceous omnivore.

Although the limited sample sizes prevented a robust statistical comparison of plankton mercury concentrations, and no significant differences were found between the ponds, Hodgdon Pond appears to have a higher concentration and proportion of methylmercury in plankton. Therefore, this small, statistically insignificant difference in methylmercury at the base of the aquatic food chain may have a great effect on biomagnification, as manifested in the high concentrations found in piscivorous fish species in this pond.

In conclusion, the elevated mercury levels found in piscivorous fishes from Hodgdon Pond probably result from a combination of interrelated factors. Warmer, low-dissolved oxygen hypolimnetic water is probably contributing to the enhanced bioavailability of mercury in this pond, as seen by higher methylmercury concentrations in plankton at the base of the food chain. An extra trophic link in the food chain leading to smallmouth bass, as evidenced by the presence of larval fish and isopods in the diet of

yellow perch, is also contributing to the increased biomagnification of mercury in Hodgdon Pond. Finally, an additional factor that was not examined in this study is the effect of the size of the watershed drainage area to lake area. Hodgdon Pond has a much larger watershed ratio than that in Seal Cove Pond, with ratios of 28:1 and 8:1, respectively (Kahl *et al.*, 1985). Because Lee and Iverfeldt (1991) have found high concentrations of methylmercury in run-off, the differences in watershed ratios may offer an explanation for the elevated mercury concentrations found in spring biota within Hodgdon Pond.

CONCLUSIONS

Mercury concentrations in predatory fishes of Mount Desert Island are generally similar to or lower than concentrations in predatory fishes from other regions of Maine. Salmonid fish species, in particular, seem to have lower mercury concentrations than the statewide averages for these species. However, non-salmonid fish species on the island are similar, and in the case of smallmouth bass, higher than the respective statewide averages for each species.

The reasons for the elevated mercury levels in fishes from Hodgdon Pond seem to be related to trophic feeding levels and the production of methylmercury within the water column. An additional trophic link in the food chain leading to smallmouth bass in Hodgdon Pond, compared with the food chain in Seal Cove Pond, seems to lead to enhanced biomagnification of mercury in this pond. The additional link in the food chain was revealed by the presence of larval fishes and isopods in the diet of yellow perch from Hodgdon Pond, and the absence of these two items in the diet of yellow perch from Seal Cove Pond. Additionally, the presence of warmer, low-dissolved oxygen hypolimnetic water in Hodgdon Pond is probably contributing to the enhanced bioavailability of mercury in this pond, as seen by higher methylmercury concentrations in plankton at the base of the food chain.

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BIOGRAPHY

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