The Association Between Watershed Characteristics and Mercury Concentrations in Fish of Northern Ontario Lakes

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Abstract

Many landscape, limnological, and ecological factors synergistically affect the mercury cycle and subsequently influence total mercury (THg) concentrations in fish. In Chapter 1, the associations between watershed and lake scale characteristics with THg in piscivorous fish are examined. ArcGIS was used to delineate the waterbody catchment area and extract waterbody catchment characteristics for 243 of northern Ontario's lakes. Walleye (*Sander vitreus,* n= 121 lakes), lake trout (*Salvelinus namaycush,* n= 60 lakes), brook trout (*Salvelinus fontinalis*, n= 18 lakes), northern pike (*Esox lucius,* n =107 lakes), and smallmouth bass (*Micropterus dolomieu*, n = 37 lakes) were standardized to the mean length of the populations by using power-series regressions. Multivariate analysis (non-metric multidimensional scaling) and univariate analysis were used to determine the associations between total mercury concentrations in fish and watershed scale and lake scale variables. Watershed and lake chemistry characteristics poorly described the variability in THg concentrations. Forest harvesting and natural disturbance were not associated with fish mercury concentrations.

In Chapter 2, the relationship between walleye (*Sander vitreus*) growth rates and mercury concentrations was evaluated. The von Bertalanffy growth model was used to standardize the age of walleye to the mean total length. Walleye populations with slower growth rates had higher THg concentrations (r^2=0.333, p< 0.001), suggestive of growth efficiency. Moreover, abundance of walleyes was associated with the growth rate (r^2 =0.136, p<0.0001).

Concentrations of THg in piscivorous fish are attributed to physical, chemical, and ecological characteristics of lakes. It is likely that lake ecology exerts the strongest influence on high mercury concentrations in piscivorous species, masking the effect from from watershed disturbance.

Lay Summary

This thesis contributes to the body of research addressing mercury as a dangerous global pollutant as well as a harmful fish contaminant. In Ontario, mercury accounts for 86.2% of consumption restrictions from inland water bodies[1]. In the Boreal Shield where forest harvesting is a major landscape-scale disturbance, the connection between forest harvesting and mercury contamination of sport-fish is not well defined and requires further assessment. Forest vegetation and soil sequester and retain atmospheric and naturally occurring geologic mercury within their watersheds for long periods of time. Changes in watershed hydrology can cycle terrestrial mercury to the aquatic environment. While certain variables are known to influence the production and accumulation of MeHg in aquatic ecosystems, investigation of the relative importance of watershed and lake characteristics that simultaneously influence mercury contamination in freshwater fish is lacking. This paper specifically addresses the associations between lake chemistry, spatial characteristics of the lake and waterbody catchment environment, and biological relationships between the terrestrial and aquatic ecosystem in order to gain a better understanding of mercury contamination of sport-fish in connection with forest harvesting activities.

[1] Ontario Ministry of the Environment. 2008-2009 Guide to Eating Ontario Sport Fish. Twenty-fifth Edition, Revised. Queen's Printer for Ontario.

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0. General Introduction

Mercury as a Global Pollutant

Mercury exists in the global environment either from naturally occurring geological sources or as a result of human activities. Even as the direct toxicity of mercury has been known for thousands of years, humans have only known of the trophic transfer of mercury and other biocontaminants since the 1960's (Takizawa 1979). Our understanding of mercury's behaviour in the environment was initiated by human tragedies that were linked to point-source discharges of mercury (Munthe *et al.* 2007).

The most severe case of mercury contamination from consumption of contaminated fish occurred in Minamata Bay, Japan in the late 1950's where 2252 people were severely affected (Harada 1995). The point-source pollution came from an acetaldehyde manufacturing plant, which discharged 456 tons of mercury into Minamata Bay between 1932 and 1968 (Díez 2008). This disastrous event was the first case of mercury poisoning caused by food chain transfer of pollutants that led to very high concentrations of mercury in fish and shellfish (Harada 1995). This awakened the world to the dangers of mercury and the threat of mercury contamination in fish and other aquatic biota.

Soon after the Minamata tragedy, direct discharges of mercury were virtually eliminated from Ontario's major industrial polluters (Mohapatra *et al.* 2007; OMOE 2011). However, decades of use and inappropriate disposal of mercury have left their mark on the fish and wildlife of environmental systems. For example, the English-Wabigoon river system (located near Dryden, Ontario) has been declared contaminated since 1970. Dryden's chlor-alkali industry used the mercury-cell cathode for manufacturing caustic soda and was a major discharger of mercury directly to the aquatic environment. Between 1962 and 1970, approximately 10 metric tonnes of inorganic mercury were released in the effluent water of the chlor-alkali plant (Kinghorn *et al*. 2007). Mercury concentrations in walleye (*Sander vitreus*) from Clay Lake in 1970 exceeded 15 ppm w.w. (standardized total length of 500 mm) (Parks and Hamilton 1987). All fish species from the English-Wabigoon river system were considered unfit for human consumption due to severe mercury contamination. The contamination of fish of the English-Wabigoon system resulted in the closure of the sustenance, commercial, and recreational fisheries in May of 1970 (Kinghorn *et al.* 2007). Over the past few decades, a slow recovery from the environmental damage has been continually monitored and the levels of mercury have declined in the biota whereby certain sizes and species of fish are now able to be consumed (Kinghorn *et al.* 2007; OMOE 2011).

Around the mid 1980's, global studies found the occurrence of mercury contaminated fish in remote locations lacking substantive sources of local mercury inputs from anthropogenic or geologic sources (USGS 1995). Mercury was soon recognized as a "global pollutant" (Schroeder and Munthe 1998; Fitzgerald *et al.* 1998). The source of mercury pollution to these remote ecosystems was determined to be atmospheric fallout. Mercury's long atmospheric residence time enables the elemental form to be transported for tens of thousands of kilometres in the troposphere from its point-source (Grigal 2002). By the mid 1990's, mercury gained worldwide attention from scientists and resource managers as it became known as the most widespread contaminant of aquatic ecosystems (USGS 2010).

Mercury Emissions

Global atmospheric emissions of mercury have greatly increased since the industrial revolution. The 2005 estimate of mercury emissions from human sources was estimated to be approximately 1930 tonnes, whereby Asia is the leading polluter accountable for about two-thirds of this amount (UNEP 2008). The United States of America and India, as the second and third largest emitters respectively, equate merely to one-third of China's emissions when combined total emissions are compared (UNEP 2008).

The primary sources of atmospheric mercury from anthropogenic sources are fossil fuel combustion, mining activities, and industrial processes that smelt ores or produce cement (UNEP 2008). The largest single source of anthropogenic mercury emissions is the combustion of fossil fuels and China is by far the largest emitter, mainly due to coal combustion, compared to all other countries (Seigneur *et al.* 2003; Pacyna *et al.* 2006; UNEP 2008). Mercury pollution from coal is a result of the enormous amounts of coal being consumed rather than from the levels of mercury in coal which varies according to geologic origin from 0.01 to 1.5 ppm (Pacyna *et al.* 2006; UNEP 2008).

Secondary anthropogenic sources of mercury are released from industrial processes or from mercury-containing consumer products. Artisanal or small-scale gold mining and the chlor-alkali industry are major industrial sources of secondary anthropogenic mercury to the environment (UNEP 2008). A considerable amount of pollution is also from mercury's widespread occurrence in consumer products including common household items (batteries, paint, switches, thermometers, blood-pressure gauges, fluorescent lights), dental amalgam, pesticides, fungicides, medicines, and cosmetics (UNEP 2008). Until mercury-containing products are phased out of the global marketplace, mercury continues to be widely used and released into the environment regardless of the known potential dangers and health risks. In Canada, the use and improper disposal of products containing mercury currently represents about 27% of Canada's total emissions (Canada Gazette 2011).

Since the 1970's, Canada has taken aggressive action and has reduced its industrial mercury emissions by approximately 90% and as a result is only responsible for approximately 1.42% of the total global mercury emissions (Canada Gazette 2011). Canada accomplished this decrease by implementing regulations, pollution prevention plans, and Canada-wide standards for mercury emissions from waste incineration, base metal smelting, and coal-fired electric power generating stations (Canada Gazette 2011).

Natural Geologic Sources of Mercury

Mercury is released into the global environment naturally by the weathering of mercury-containing rocks and soils, volcanic eruptions, and geothermic activity (UNEP 2008). These natural sources of mercury account for one third to a half of mercury emissions to the atmosphere (UNEP 2008).

Mercury occurs naturally in a number of geological formations but rich geological deposits of mercury are most often found in the form of cinnabar (red mineral or ore composed of mercury sulphide: HgS) or metacinnabar (Jonasson and Boyle 1971). The mercury content in cinnabar can reach concentrations as high as 86% (Jonasson and Boyle 1971). However, in Canada mercuriferous belts rich in cinnabar are generally not found east of the Rocky Mountains (National Research Council Canada 1979). None the less, natural background concentrations of mercury may be influenced by widespread geological formations that contain varying levels of mercury (Rasmussen *et al*. 1998). Mercury deposits occur in all types of rocks, but sedimentary rocks of the Palaeozoic to Recent age contain greater concentrations of mercury (Jonasson and Boyle 1972). The average mercury content is much higher in shales (0.4 ppm Hg) compared to sandstones and limestones (0.03 ppm Hg) and granites (0.08 ppm Hg) (Goldwater 1972).

Studies by the Geological Survey of Canada (GSC) found some of the highest sediment mercury values in Ontario's Lakes southwest of Thunder Bay in an area underlain by shale (Friske and Coker 1995).

Elevated mercury levels in bedrock are associated with mineralization (Jonasson and Boyle 1972). High soil mercury levels have been reported in Quebec and Ontario near areas of known gold, copper, or zinc mineralization (Environment Canada 1979). The Red Lake area, which is situated near a rich gold deposit, has mercury concentrations in the rock cores twice as high as in cores from adjacent non-mineralized regions of Northwestern Ontario (Bishop and Neary 1976). However, mercury levels in fish from the Red Lake area have not been noticeably higher than those from other off-system lakes from different geological formations (Bishop and Neary 1976).

Research across Canada has shown that physical and chemical weathering and erosion of bedrock, glacial deposits, and soil enriched in mercury ultimately increases the mercury load in lakes and streams (Jonasson and Boyle 1972; Hornbrook and Jonasson 1971; Rasmussen *et al*. 1998; Friske and Coker 1995). The weathering of local geology may be a leading contributor to the elevated mercury levels in the environment but distinguishing the relative contribution of mercury from natural sources is a major challenge. However, natural geologic sources of mercury alone are insufficient to explain the increased number of lake sediment and peat profiles with substantial increases in mercury levels during the last century (Engstrom *et al*. 2007; Fitzgerald *et al*. 1998).

The Mercury Cycle

The environmental mercury cycle has four strongly interconnected realms: atmospheric, terrestrial, aquatic, and biotic (Wiener *et al*. 2003). Mercury is reasonably reactive in the environment and cycles readily among the four compartments as it can exist naturally in the solid, liquid or gas physical state (Ullrich *et al.* 2001; Wiener *et al*. 2003). Mercury has three valence states (Hg^0, Hg^{1+}, and Hg^{2+}); in the atmosphere, mercury is mostly (>95%) gaseous elemental mercury (Hg^0). Mercury emitted from point-sources enters into the atmosphere as elemental mercury (Hg^0), gaseous ionic mercury (reactive gaseous mercury (RGM) which is generally assumed to be mercuric chloride ($HgCl_2$)), and particulate mercury (Hg^P) (Lindberg and Stratton 1998; Driscoll *et al*. 2007; Wiener *et al*. 2003). The atmospheric lifespan of each species is between 0.5 to 2 years, 0.5 to 2 days, and 0.5 to 3 days respectively (Driscoll *et al*. 2007; Wiener *et al*. 2007).

Oxidation of Hg^0 occurs at the solid-liquid interface in cloud and fog droplets, forming the dissolved inorganic divalent mercury species (or mercuric form of mercury: Hg^{2+}) which is the most common form found in precipitation (Morel *et al*. 1998). Atmospheric mercury, once deposited onto the land surface, becomes sequestered in soils largely in the inorganic divalent form (Hg^{2+}) bound to organic matter in the humus layer or to mineral constituents in the soil (Lindqvist 1991; Wiener *et al*. 2003). Soils of the Boreal region are typically organic rich podsols that readily absorb and accumulate mercury. Aquatic ecosystems are protected from the full effects of atmospheric mercury pollution as forest soils sequester atmospheric mercury and watersheds typically act as sinks (Grigal 2002; Bishop *et al*. 2009b). Based on estimates, terrestrial soils contain the largest inventories of mercury from natural and anthropogenic emissions (Lindqvist 1991; Mason *et al*. 1994). The global inventory of mercury in surface soils far exceeds the mercury stored in the aquatic or atmospheric compartment (Wiener *et al*. 2003). The store of mercury as Hg^{2+} in soils can be reduced and re-emitted into the atmosphere as elemental mercury (Hg^0) by volatilization or be transported to the aquatic environment where it transforms into other forms of mercury, most importantly the organic form methylmercury (MeHg having the formula CH_3Hg^+) (USGS 1995; Wiener *et al*. 2003).

The conversion of inorganic forms of mercury to the organic form methylmercury by methylating organisms is a critical component of the mercury cycle (Pollution Probe 2003). Mercury methylation is mediated by microbial activity and is the conversion of both neutral mercury complexes and ionic mercury to methylmercury (CH_3Hg^+) by a methyl-donor group (Ullrich *et al.* 2001; Wiener *et al*. 2003). A variety of microorganisms are known to be involved in the mercury methylation process including sulfate-reducing bacteria (SRB) which are believed to be the most important methylating agents in anaerobic sediments (Ullrich *et al.* 2001). The SRB mediate the methylation of inorganic mercury and produce sulfide as a metabolic by-product of microbial respiration (St. Louis *et al*. 1994; Branfireun *et al*. 1996; Benoit *et al*. 1999). The transformation of inorganic mercury to methylmercury primarily occurs in semi-anoxic environments of lake sediments and wetlands (Wiener *et al*. 2003). Within the bottom sediments of lakes, sulfate reduction is greatest in the uppermost 5 cm at the oxic-anoxic interface of aquatic environments (Wiener *et al*. 2003). In wetland porewater, lower sulfate levels and the corresponding increase in methylmercury were attributed to sulfate reduction by bacterial activity (Selvendiran *et al.* 2008). Mercury methylation also occurs to a lesser extent in aerobic freshwaters, on floating periphyton mats, on the roots of some floating aquatic plants, in the intestines of fish, and on the mucosal slime layer of fish (Wiener *et al*. 2003). Iron-reducing and methanogenic bacteria also have potential to methylate mercury (Ullrich *et al.* 2001; Fleming *et al.* 2006). At the same time, sulfate reducers and methanogenic bacteria both possess the ability to demethylate mercury in freshwater sediments (Ullrich *et al.* 2001). Demethylation processes that degrade methylmercury by microbial action or photodegradation are operating simultaneously in terrestrial and aquatic ecosystems but in aquatic systems the rate of methylation typically exceeds that of demethylation (Ullrich *et al*. 2001; Wiener *et al*. 2003). Thus, methylation and demethylation are complicated processes affecting the methylmercury concentrations in aquatic food webs (Wiener *et al*. 2003).

Aquatic organisms take up both mercury and methylmercury but methylmercury is assimilated more efficiently than ionic mercury (Mason *et al*. 1994). Mercury assimilation efficiency varies by species and is 5-10 fold higher for methylmercury than inorganic mercury (Stokes and Wren 1987; Trudel and Rasmussen 1997). A process known as "bioaccumulation" occurs when organisms take up contaminants, such as the inorganic and organic forms of mercury, more rapidly than their bodies can eliminate them (Pollution Probe 2003). Mercury bioaccumulation can also result from the direct uptake of mercury across fish gills (Ponce and Bloom 1991).

In freshwater food webs, trophic scale interactions with mercury begin with the bioaccumulation of ionic mercury and methylmercury by primary producers (Driscoll *et al*. 2007). The inorganic forms of mercury are excreted rapidly compared to methylmercury which is more efficiently absorbed and accumulated (Trudel and Rasmussen 1997). Fish are able to assimilate 65 to 80% of the methylmercury present in the food they eat (Wiener *et al.* 2003). An increase in the ratio of methylmercury to inorganic mercury contributing to the total mercury concentration occurs with each additional step in the trophic level (Tan *et al.* 2009). Thus, the predominance of methylmercury in fish is a consequence of the greater trophic transfer efficiency of methylmercury from food and slower rates of methylmercury excretion relative to inorganic mercury (Trudel and Rasmussen 1997). The trophic transfer of methylmercury is more efficient with each step in the food chain and average contribution that MeHg makes to the total

Hg level increases from 10% in the water column to 15% in phytoplankton, 30% in zooplankton, and 95% in fish (Watras and Bloom 1992; Driscoll *et al*. 2007). In piscivorous fish it is inferred that 99% or more of total mercury is methylmercury (Grieb *et al.* 1990; Bloom 1992).

The trophic transfer of bioaccumulated methylmercury is more efficient with each step in the food chain leading to the process known as "biomagnification" which is the increase in concentration of a contaminant with each additional trophic level (Pollution Probe 2003). Due to bioaccumulation and biomagnification, concentrations of methylmercury in sport fish commonly exceed those in ambient surface water by a factor of 10^6 or 10^7 (Wiener *et al*. 2003). High mercury levels are most often found in piscivorous fish such as pike, walleye, bass and trout (Wiener *et al*. 2003).

Contamination of Aquatic Ecosystems

The long range transport and deposition of anthropogenically-derived mercury is largely responsible for the elevated mercury concentrations in fish of remote areas (Fitzgerald *et al*. 1998). Atmospheric modelling of mercury deposition has shown that the Great Lakes region can be influenced from sources up to 2000 km away (Cohen *et al*. 2004). Lake sediment cores from mid-continental United States of America show that the atmospheric deposition of mercury has tripled in the past 140 years (Swain *et al.* 1992). Mercury concentration in lake sediment cores from central and northern Canada and Hudson Bay have increased on average by 2 fold over the past half century (Lockhart *et al.* 1998). At the Experimental Lakes Area in northern Ontario, the anthropogenic component of current mercury inputs was calculated to be approximately 9 $\mu g\ m^{-2}\ y^{-1}$ (Lockhart *et al.* 1998). An estimated 96% of the mercury that is being deposited to Canada's land and water every year comes from foreign emission sources (Fitzgerald *et al*. 1998; Canada Gazette 2011).

In recognition of the global nature of mercury pollution, international action was taken through the United Nations Environment Programme (UNEP) to establish a committee with a mandate to prepare a legally binding document with global standards for mercury emissions for 2013 (Canada Gazette 2011). If no action is taken to reduce global mercury emissions, the estimated cost of global mercury pollution is projected to be $10 billion a year by 2020 (Pacyna *et al.* 2008).

Health Impacts and Human Fish Consumption Guidelines

Despite Canada's decreased emissions, thousands of recreational fish consumption advisories are issued each year in Canada due to the persistent nature of mercury and increased industrialization in other countries (Canada Gazette 2011). For the general population, the main route of exposure to methylmercury is consumption of contaminated fish and other seafood (Health Canada 2007). Coastal Arctic communities are also exposed to mercury through consumption of marine mammals (Van Oostdam *et al.* 2005; Donaldson *et al*. 2010). Canada's aboriginal people are faced with health concerns when relying on a traditional diet high in fish and marine mammals. At the same time, fish are an excellent source of high quality protein with many nutritional benefits and are an essential component of a traditional diet for many Canadians (Health Canada 2007). It is difficult to show a definite correlation between consumption of contaminated fish and direct clinical effects caused by methylmercury exposure (Wheatley and Paradis 1995). Therefore, considering the risk of methylmercury exposure along with the many health benefits of fish consumption, consumers are advised to modify their behaviour by choosing lower risk species and sizes of fish for consumption rather than decreasing their overall fish consumption (Health Canada 2007; Dórea 2008).

Once humans ingest methylmercury from contaminated food, roughly 95% is absorbed in the gastrointestinal tract and distributed to all tissues in the body as it is bound to the hemoglobin in red blood cells (Díez 2008). Methylmercury is slowly metabolized; the half-life of methylmercury in the body is about 50 days, with a range of 20 to 70 days (Díez 2008). The toxic effect of mercury negatively impairs the reproductive, nervous, cardiac, immune and endocrine organ systems (Wolfe *et al.* 1998; Sams 2004; Clarkson and Magos 2006; Mergler *et al.* 2007; Scheuhammer *et al*. 2007; Tan *et al.* 2009; Sandheinrich and Wiener 2011).

The developing human fetus is extremely sensitive to methylmercury exposure as it easily crosses the blood-brain barrier and placental membrane and can lead to neurological damage and impaired development (Health Canada 2007; Díez 2008). Methylmercury has been shown to impact cognitive development, measured as intelligence quotient (IQ), in children whose diet contained a large portion of seafood (Pacyna *et al.* 2008). In addition to the neurological development abnormalities, infants exposed to methylmercury in utero have exhibited delays in walking and talking, cerebral palsy, altered musclar tone, and deep tendon reflexes (Díez 2008). In adults, mercury exposure can increase the risk of cardiovascular diseases, especially myocardial infarction (Pacyna *et al.* 2008). Physical lesions in the brain can damage the central nervous system resulting in tingling and numbness in fingers and toes, malaise, impaired balance, loss of coordination, difficulty walking, generalized weakness, impairment of hearing or vision, impaired speech, tremors, and loss of consciousness leading to death (Health Canada 2007; Díez 2008).

Canada's federal, provincial and territorial governments play a role in protecting the public's health from the hazards of mercury by issuing mercury standards for fish inspections and offering consumption advice. In 2007, Health Canada changed its risk management strategy to reduce the threat of unacceptable mercury exposure from the retail sale of fish by strengthening the standards based on a new synthesis of knowledge on the health hazards of methylmercury. The Health Canada standard for total mercury allowable in commercially sold fish, which is enforceable by the Canadian Food Inspection Agency, is 0.5 ppm (Health Canada 2007). The 0.5 ppm standard applies for all species of fish except for certain long-living piscivorous fish that are consumed less frequently: shark, swordfish, escolar, marlin, orange roughy, and fresh/frozen tuna, which are subject to a 1.0 ppm total mercury standard (Health Canada 2007). At the moment, Canada's higher-risk standard is equivalent to the American "action level" of 1.0 ppm for total mercury in all commercial fish. However, this standard set by the United States Food and Drug Administration (FDA) is currently under reassessment (FDA 2011).

Health Canada offers consumption advice based on the tolerable daily intake (TDI) standard. For women of childbearing age, pregnant women, and young children the TDI for methylmercury is 0.2 micrograms per kilogram body weight per day (0.2 µg/kg bw/day). For the general population, Health Canada employs the methylmercury TDI that was developed by the joint FAO/WHO committee on Food Additives, which is 0.47 micrograms per kilogram body weight per day (0.47 µg/kg bw/day) (Health Canada 2007).

The provincial and territorial governments are responsible for implementing contaminant monitoring programs and issuing consumption advisories for sport fish in Canada. For the Province of Ontario, the Ministry of Natural Resources and the Ministry of the Environment work together to produce the biennial "Guide to Eating Ontario Sport Fish". The Ministry of the Environment determines what fish, based on size and species, are suitable for human consumption. The consumption restriction calculations for sport fish include consideration for all contaminants and are such that "No one shall exceed their tolerable daily intake (over a one month period) for any contaminant in sport fish if they follow the advice in the Guide" (Environment Canada 2001). Consumption restrictions for women of child-bearing age and children under 15 begin at 0.26 parts per million (ppm) and total restriction is advised for levels over 0.52 ppm of total mercury in fish. For the general public, fish consumption restrictions start at 0.61 ppm and total restriction is advised for levels over 1.84 ppm total mercury (OMOE 2009). Similarly, The Great Lakes Fish Advisory Workgroup (2007) recommends consumption bans for fish mercury levels >0.95 ppm (wet weight).

Northern Ontario Lakes and Fish Communities

The distribution of Ontario's fish fauna is mostly a consequence of geological and ecological forces (Radforth 1944; Hartviksen and Momot 1987; Holm *et al*. 2009). The topography of northern Ontario and the formation of drainage basins is a relic of the last episode of glaciation which occurred 100,000 to 18,000 years ago (Holm *et al*. 2009). After the continental ice sheets disappeared approximately 6,000 years ago, the land rebounded upwards forming the watersheds that are present today (Holm *et al*. 2009). Following glacial retreat, fish movement occurred between connected waterways. The coldwater species of fish (communities dominated by lake trout [*Salvelinus namaycush*]) were the first to colonize Ontario's new formed lakes followed by the coolwater species (communites dominated by walleye and northern pike [*Esox lucius*]) (Holm *et al*. 2009). The newly formed watersheds formed a physical barrier to fish movement. However, human interference on fish movement has allowed species to move past their historical range (Holm *et al*. 2009).

There are three distinct fish communities present in the lake, river, and wetland habitats of Ontario's Boreal region (Browne 2007). In the lakes and rivers of the Boreal Shield zone, walleye and northern pike are the most common top predators and are the most widely distributed community type (Browne 2007). Although walleye and pike often feed on similar prey, their diets diverge considerably due to their feeding habits. Northern pike are sit-and-wait predators that inhabit shallow inshore areas of lakes and slow moving waters of rivers with abundant structure such as aquatic vegetation or fallen trees (Browne 2007). Walleye are roaming predators that inhabit both slow and fast current areas of rivers as well as both near-shore and offshore lake environments (Browne 2007). The diet of northern pike is dominated by minnows, white suckers, and yellow perch, whereas the diet of walleye is dominated by yellow perch, cisco, and minnows. Walleye and northern pike also prey on one another (Scott and Crossman 1973).

Lake trout/whitefish/cisco communities are a less common community type with the majority of occurrences in the western half of northern Ontario (Browne 2007). Lake trout, lake whitefish, and cisco inhabit deep (> 8 m), low productivity lakes. In autumn, lake trout move into the shallows in preparation for spawning. After spawning, lake trout disperse freely within the entire lake at various depths and remain dispersed throughout the winter months. In spring, lake trout occur in shallow surface waters immediately after the break-up of ice. Lake trout retreat to the cooler deep water of the hypolimnion during the warm summer months. Lake trout are predaceous and feed on a broad range of organisms (zooplankton, freshwater sponges, crustaceans, aquatic and terrestrial insects, many species of littoral and pelagic fishes, other lake trout, and small mammals) (Scott and Crossman 1973; Browne 2007). Food varies with season, particularly in small lakes where a thermal barrier constricts their foraging environment (Scott and Crossman 1973).

The third community type, the brook trout (*Salvelinus fontinalis*) community, is commonly associated with the major rivers, tributary streams, and creeks as well as lakes and beaver ponds (Browne 2007). Brook trout are most abundant in waters with depauperate fish communities. Lake-dwelling populations are rare in the north due to the presence of other top predators such as walleye, perch, and northern pike which are competitors and predators of brook trout (Browne 2007).

Mercury in Fish of Northern Ontario

Methylmercury contamination of fish is a global problem that has diminished the recreational, economic, and nutritional benefits derived from fisheries resources in many fresh waters (Sandheinrich and Wiener 2011). Mercury accounts for 80% of the fish-consumption advisories in the United States and 97% of the fish consumption advisories in Canada (U.S. EPA 2009; U.S. EPA 2001). In Ontario, mercury is the main contaminant which accounts for 86.3% of all fish contaminant consumption restrictions from inland water bodies (OMOE 2009; OMOE 2011). Consumption restrictions due to mercury alone are particularly common in remote inland locations (OMOE 2011). Many of Ontario's inland lakes and rivers contain fish with high mercury concentrations which may vary widely between and within systems.

In Canada and Scandinavia, forestry harvesting activities have been shown to increase mercury concentrations aquatic ecosystems and biota (Garcia and Carignan 1999, 2000, 2005; Lamontagne *et al.* 2000; Porvari *et al.* 2003; Desrosiers *et al.* 2006). The physical and chemical characteristics of the waterbody as well as the degree of impact from human activities will influence the formation and bioavailability of methylmercury (Kidd *et al.* 2012). Mercury levels in fish may become elevated if methylmercury at the base of the food web is enhanced by significant influxes from external sources, high *in situ* rates of production in bed sediments and anoxic hypolimnia, or a combination of biogeochemical, biological, trophic and human factors (Kidd *et al.* 2012).

Watershed characteristics and human influences on the processes that affect mercury bioaccumulation and biomagnification are particularly important when evaluating mercury contamination in fish (Kidd *et al.* 2012). Past studies have shown a range of responses from watershed influences, such as wetlands, wildfire, and forest harvesting, on influencing mercury dynamics and concentrations in the water and biota in freshwater lake ecosystems. As landscape characteristic data are widely available through satellite mapping and GIS spatial layers, this wealth of knowledge provides an opportunity to determine the associations of watershed characteristics with mercury concentrations in fish across a large spatial scale.

Thesis Objectives

The purpose of this research was to evaluate the relationship between fish mercury concentrations and various lake and watershed scale characteristics. There are unanswered questions regarding why certain fish populations have higher mercury concentrations than those of neighbouring lakes. The variability in fish mercury concentrations should be explained by spatial attributes of the watershed, lake characteristics, and characteristics of the fish population.

Chapter 1 of this study will: 1) investigate the association of methylmercury in fish tissues with the amount of disturbance (natural or human caused) that has occurred within the watershed of a lake, as well as 2) investigate the association of methylmercury in fish tissue with lake and watershed scale variables that are believed to influence mercury bioavailabilty in the aquatic environment.

Chapter 2 examines the associations between walleye mercury concentrations and specific population characteristics.

Study Area

Lakes and watersheds in this study are located in Fisheries Management Zones (FMZs) 4, 6, 7, 8, 10, and 11 (Figure 0.1). The study area spans the Boreal Forest and Great Lakes-St. Lawrence Forest Regions of the Boreal Shield Terrestrial Ecozone which is where the majority of forest harvesting occurs within the Province's Area of the Undertaking (OMNR 2010b). The total area of the province managed for forest harvesting is 26.2 million hectares (OMNR 2010b). Between the years 2008 to 2009, approximately 123,387 hectares (ha) of Crown Forest Land were harvested where the clearcut silvicultural system was used for the vast majority of this area (OMNR 2010b).

Wiken (1986) described the Boreal Shield Ecozone as having abundant precipitation and well irrigated land with a multitude of lakes containing approximately 10% of Canada's freshwater. The topography, having been shaped by glaciation and postglacial deposition, is a massive rolling plain of ancient bedrock. The climate is continental with long cold winters and short warm summers. Numerous bogs, marshes and other wetlands cover this ecozone (Wiken 1986).

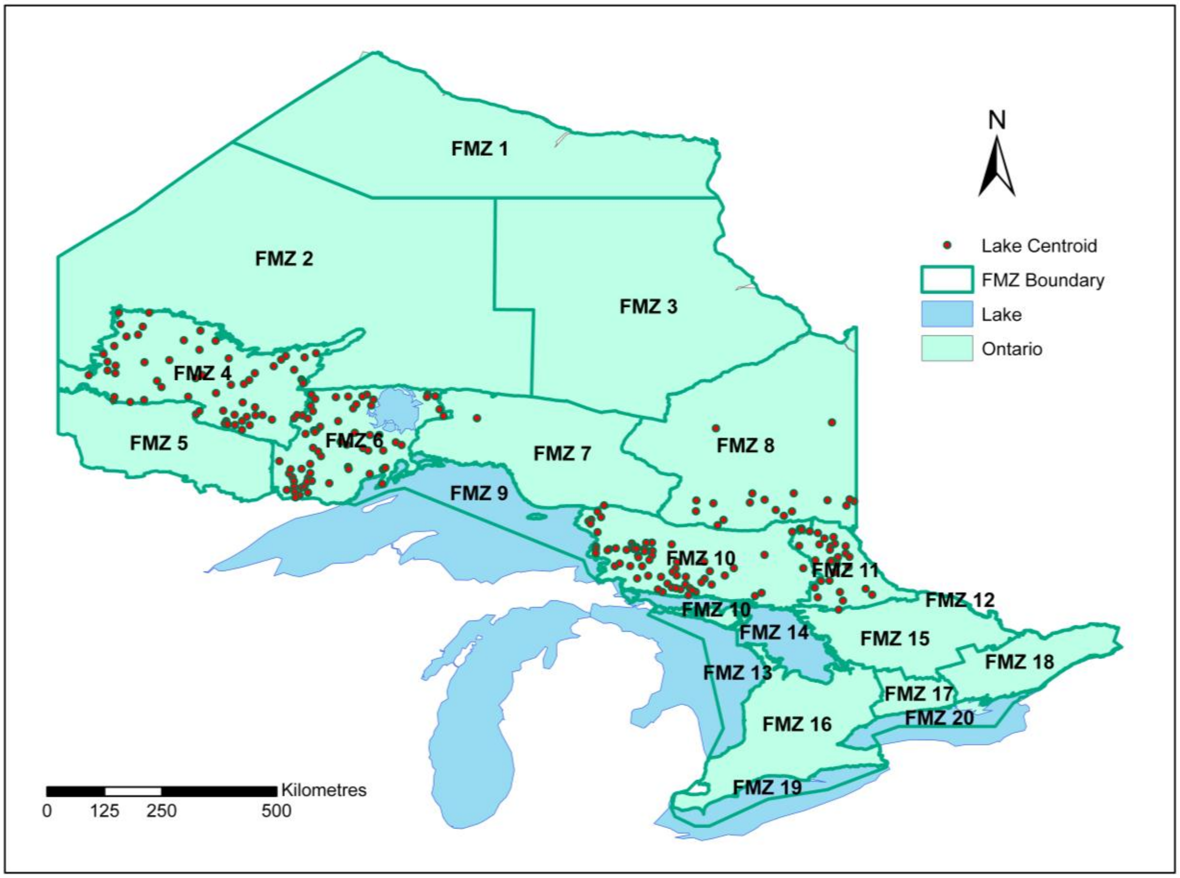


Figure 0.1. Study Lake Locations and Fisheries Management Zones (FMZ) of Ontario.

1. Mercury Associations with Catchment and Lake Scale Variables

Introduction

A significant fraction of the methylmercury in biota from remote or semi-remote regions is derived from anthropogenic mercury entering the aquatic ecosystem or its watershed from atmospheric deposition (Wiener *et al.* 2003). However, quantifying the relative contribution of natural and anthropogenic emissions to the methylmercury in aquatic life at remote and semi-remote locations is an enormous scientific challenge due to the spatial variation in natural sources of mercury and the biogeochemical transformations and transport of mercury within the landscape (Wiener *et al*. 2003). Fish methylmercury concentrations are largely influenced by a combination of watershed and lacustrine factors that exert controls on the production and bioaccumulation of methylmercury (Wiener *et al*. 2006).

Lake basin characteristics have shown associations with mercury concentrations in biota of Boreal lakes (Bodaly *et al.* 1993; Garcia and Carignan 1999; Garcia and Carignan 2000; Garcia and Carignan 2005; Garcia *et al.* 2007). Past studies have evaluated morphometric parameters (mean depth, maximum depth, lake surface area, lake volume, watershed area, ratios of watershed area to lake size or ratios of epilimnetic area to lake size) that relate to temperature and water chemistry as potential variables contributing to mercury variability in biota (Ramlal *et al.* 1993; Bodaly *et al.* 1993; Garcia and Carignan 1999). Lake size is an important variable influencing the mercury concentrations in fish (Bodaly *et al*. 1993). Bodaly *et al.* (1993) found an inverse relationship between lake size and mercury concentrations of planktivorous, omnivorous, and piscivorous fish in a study of six remote lakes in northwestern Ontario. Lake temperature, which is related to basin area, lake area and lake volume, influences rates of mercury methylation. Higher water temperatures and increased methylation in warm littoral sediments may explain the negative correlation between lake size (or volume) and fish mercury concentrations (Bodaly *et al.* 1993; Evans *et al.* 2005). Small lakes have a greater temperature variation than large lakes as they are generally shallower and typically respond faster to changes in atmospheric temperature, making them warmer in the summer and colder in the winter; a greater range in temperature could lead to higher rates of methylation compared to demethylation (Bodaly *et al.* 1993). Another explanation for the results found by Bodaly *et al.* (1993) may be that there is more efficient solute and sediment transport to lakes with smaller watersheds

(Gabriel *et al.* 2009). Likewise, the proportionate flux of allochthonous inputs of organic matter and complexed mercury from the watershed may have a greater influence on smaller lakes (Greenfield *et al.* 2001).

Studies of lake systems have found correlations between water chemistry and fish mercury concentrations. Water chemistry can influence the production, availability and bioaccumulation of methylmercury. Mercury speciation, a principal factor governing the methylation potential of a system, is strongly influenced by chemical conditions, notably redox, pH, organic ligands, and inorganic ligands (Wiener *et al*. 2003). Dissolved organic carbon (DOC) and pH are the two most documented water chemistry parameters associated with mercury biochemistry (Gabriel *et al.* 2009). Nutrients, primarily total phosphorus or total nitrogen, will control ecosystem productivity and may influence mercury accumulation in aquatic biota (Garcia and Carignan 1999; Rypel 2010).

The mercury and sulfur cycle are intimately linked in the aquatic ecosystem as both are mediated by bacterial controls. Sulfate reducing bacteria (SRB) and methanogenic bacteria methylate and demethylate mercury in freshwater ecosystems but the rate of methylation typically exceeds that of demethylation (Ullrich *et al*. 2001; Wiener *et al*. 2003). The speciation of sulfur is a major controller on the net methylation rate of mercury in many ecosystems (Munthe *et al.* 2007). The presence of sulfate stimulates the methylation of mercury by sulfate-reducing bacteria, while excess sulfide may reduce the bioavailability of mercury by forming mercury sulfide complexes which are immobilized in the sediments (Gilmour *et al.* 1992; O'Driscoll *et al.* 2005; Muthe *et al.* 2007). Under oxidized conditions in soil and water more than 99.9% of mercury and methylmercury in the aqueous phase is bound to organic sulfur groups (thiols or "–RSH" groups) forming metal-thiol complexes (Skyllberg p 40. In Bishop *et al.* 2009b). When oxygen is depleted because of microbial degradation of soil organic matter under water saturated conditions the redox-potential decreases to reduced conditions and the mercuric ion (Hg^{2+}) shifts its affinity from organic complexes to inorganic sulfide complexes (Morel *et al*. 1998; Gabriel and Williamson 2004; Skyllberg p 40. In Bishop *et al.* 2009b). Enhanced mobilization of mercury and formation of dissolved Hg-sulfides occurs when disturbance events alter the hydrologic cycle. Increased lateral transport of mercury and increased concentrations of dissolved inorganic sulfides and energy-rich organic matter will result in ideal conditions for the enhanced production of methylmercury by SRB.

The relationship between dissolved organic matter (DOM), commonly measured as dissolved organic carbon (DOC), and mercury is very complex with studies showing differences in the association between DOM and mercury accumulation in fish (Gabriel *et al.* 2009). Dissolved organic matter can bind trace metals and affect the speciation, solubility, mobility and toxicity of mercury in the aquatic environment (Ravichandran 2004). Dissolved organic matter may enhance or retard methylation of mercury, serve as a transport mechanism for mercury from terrestrial areas, or reduce the bioavaliability of inorganic and organic mercury species by binding to reaction sites (Ravichandran 2004). The enhanced mobility and transport of Hg-DOM complexes results in increased water column concentrations of mercury in otherwise pristine lakes and rivers (Ravichandran 2004). Dissolved organic matter also has the potential to enhance methylation of mercury by stimulating microbial growth (Ravichandran 2004). Dissolved organic carbon may have a positive influence on fish mercury concentrations (McMurty *et al*. 1989; Wren *et al*. 1991; Rencz *et al*. 2003; Belger and Forsberg, 2006; Driscoll *et al*. 2007). However, a negative correlation between THg in several species of fish and lake water DOC (or water colour used as an indicator of DOC) has frequently been documented (Grieb *et al*. 1990; Snodgrass *et al.* 2000; Greenfield *et al*. 2001; Gabriel *et al.* 2009). Complexation of mercury with DOC may limit the amount of inorganic mercury available to methylating bacteria since DOC molecules are typically too large to cross the cell membranes of the bacteria (Ravichandran 2004). Thus, dissolved organic carbon may prevent the biological uptake and methylation of ionic mercury within a waterbody (Munthe *et al.* 2007). Moreover, DOC is suspected to enhance photoreduction of Hg^{2+} to Hg^0 further reducing the bioavailability within an aquatic environment (O'Driscoll *et al*. 2003; Hall *et al*. 2008; Gabriel *et al.* 2009). Thus, contradictory results may be due to multiple ecological factors that complicate the net effect of lake water DOC concentrations on mercury concentrations in fish.

Fish mercury concentrations are often highly negatively correlated with pH (Cope *et al*. 1990; Sun and Hitchin 1990; Wiener *et al.* 1990). Acidic lakes with low buffering capacity typically have higher fish mercury concentrations due to multiple factors related to fish metabolism and molecular Hg bioavailability (Gabriel *et al.* 2009). Biochemical pathways for microbial production of MeHg are favoured at a low pH in the water column and at the sediment water interface. Methylation occurs more readily at the sediment-water interface since the pH and oxygen are lowest and mercury is more bioavailable with fewer chemical associations between inorganic mercury species and DOC (Ravichandran 2004). Additionally, under more acidic conditions, the production of monomethylmercury is favoured over dimethylmercury (Winfrey and Rudd 1990). Dimethylmercury has a higher volatility and is less stable in the water column; thus, at lower pH there is a decreased loss of volatile mercury from lake water (Winfrey and Rudd 1990). Acidic conditions favour methylation and sulfide production by SRB (Snodgrass *et al.* 2000). Moreover, bioaccumulation processes can influence the relationship between pH and fish mercury concentrations. Even if MeHg concentrations in the lakes are not high, a low pH can increase gill permeability and decrease the growth rates of aquatic biota resulting in elevated MeHg in fish (Winfrey and Rudd 1990). Greenfield *et al.* (2001) found pH to be the most important factor in determining THg in yellow perch. Similarly, Garcia and Carignan (2000) found pH to be the most important predictor of Hg concentrations in northern pike.

Differences in mercury transformations, cycling, and bioavailability can result from inter-lake differences in landscape features (Bodaly *et al.* 1993; Watras *et al.* 1995; Wiener *et al.* 2006). Munthe *et al.* (2007) suggested that watershed size and watershed to lake surface area ratios are the most important determinants for mercury delivery to aquatic systems. The magnitude of water chemistry response to disturbance in Boreal Shield lakes has been shown to be directly proportional to the ratio of disturbed area within the watershed to the lake's volume or area (Carignan *et al.* 2000; Garcia and Carignan 2005). Similarly, the concentration of mercury in aquatic biota has been correlated to forest-harvesting disturbance within the watershed of Boreal Shield lakes (Garcia and Carignan 1999; Garcia and Carignan 2005). However, larger watersheds may have lower mercury inputs from the watershed due to less efficient transport and increased loss processes (Munthe *et al.* 2007). Thus, watershed processes partly explain why mercury concentrations can vary five-fold among fish from neighbouring lakes receiving the same precipitation (Sorensen *et al*. 1990; Wiener *et al*. 2006; Munthe *et al*. 2007).

Wetlands play a significant role in the hydrologic cycle (Bullock and Acreman 2003) and are important sources of mercury to boreal aquatic ecosystems (Watras *et al.* 1995; St. Louis *et al.* 1996). The influence of contiguous wetlands on the mercury budget of a lake depends on the wetland location within the watershed and its hydrologic connectivity with the groundwater system and downstream channel network (Bullock and Acreman 2003). Ecosystems with abundant wetlands have shown elevated mercury concentrations in the aquatic biota (Wiener *et al*. 2006; Castro *et al.* 2007; Simonin *et al*. 2008; Rypel 2010). Although percent wetland area has been used in modeling mercury in biota, the relationships between wetlands and mercury in fish is still largely undefined.

Remote wetlands receive most of their mercury from atmospheric deposition. Newly deposited mercury complexes with the organic matter produced in wetlands within the top soil layer, enhancing the accumulation in soils (Driscoll *et al*. 2007). Atmospherically derived mercury is readily methylated in the organic-rich, anoxic sediments of boreal wetlands (St. Louis *et al.* 1996). From the wetland, mercury is easily mobilized and transported to hydrologically connected lakes by connected streams or shallow groundwater flow (Wiener *et al*. 2006). High rainfall, surface runoff, and human disturbance of wetlands promote the release and transport of mercury complexed with organic matter to lakes (Watras *et al*. 2005). A peatland study conducted at the Experimental Lakes Area Reservoir Project (ELARP) showed a four fold to fifteen fold difference in MeHg yield in rivers from watersheds containing peatlands compared to upland forested watersheds (St. Louis *et al*. 1994). Although yields of MeHg vary from one wetland to another, wetland areas in the watershed are sites of net methylmercury production (St. Louis *et al*. 1994). Yields of methylmercury from the wetland portion of the watershed were 26-79 times higher per unit area than from upland areas (St. Louis *et al*. 1994).

Wetlands differ in their vegetation and biogeochemical processes because of differences in hydrology; thus, there are large consistent differences among wetland types in relation to MeHg production and export (St. Louis *et al.* 1996). However, all wetland types have been shown to produce more MeHg during years of high water yield (St. Louis *et al*. 1996). St. Louis *et al*. (1996) concluded it is possible to model inputs of MeHg into lakes based on the percentage wetland area, the type of wetland, and the annual water yield of each watershed.

The influence of wetlands within the watershed has been shown to vary according to lake size. In a northern Wisconsin study, fish mercury levels in 43 lakes of different morphometry (drainage lakes, headwater lakes, and seepage lakes) were analyzed in relation to water chemistry, trophic ecology and spatial traits (watershed area, surrounding wetland abundance, and lake hydrologic position) (Greenfield *et al*. 2001). Wetland abundance only correlated with elevated fish mercury levels for small (<64 ha) drainage lakes with greater than 6% wetland in their watershed (Greenfield *et al*. 2001).

In remote environments, watershed characteristics such as wetlands may result in local areas, referred to as "methylmercury hot spots", which have increased mercury levels compared to the surrounding landscape as a consequence of increased mobility and methylation of atmospherically deposited mercury (Mills *et al*. 2009). These low-lying wetland hot spots may receive increased fluxes of mercury, associated with DOM or inorganic sulfides from the terrestrial area following increase lateral flow from clear-cutting disturbance (Bishop *et al.* 2009b). Spatial variability also exists within wetlands in terms of net MeHg production (Mitchell *et al.* 2008). Methylmercury concentrations were greatest around the fringes of peatlands where net production and accumulation of MeHg was higher. Hydrological mixing of solutes such as sulfate and labile carbon that accumulate in this fringe area around wetlands thereby enhances the net production of MeHg (Mitchell *et al.* 2008).

Boreal Canadian Shield watersheds serve as large reservoirs of mercury that shed their metal load when the soil, vegetation, and land hydrology are disrupted (Desrosiers *et al*. 2006). The podsolic soils of the boreal Canadian Shield forest readily adsorb and accumulate mercury from wet and dry atmospheric deposition (Desrosiers *et al*. 2006). Soils may also serve as a long-term source of mercury to surface waters (Hultberg *et al.* 1995; Gabriel and Williamson 2004). Soil cover may be an important watershed factor related to mercury variability in fish among lakes as consistent correlations between fish THg levels and upland THg levels in the soil A-horizons and O-horizons were found in lakes of the southern Boreal Shield (Gabriel *et al.* 2009). Moreover, forest canopy type and density have an important influence on mercury deposition (Witt *et al*. 2009). In the Boreal forest, the highest total mercury and methylmercury concentrations in throughfall were measured beneath dense conifer canopies with high leaf surface area (Witt *et al*. 2009). Studies at the Experimental Lakes Area show that forest canopies effectively collect mercury from the atmosphere resulting in up to 8 times more mercury deposited in a forested watershed compared to other types of open ecosystems (Environment Canada Presentation: St. Louis and Graydon 2011). Consequently, the effective scavenging abilities of the boreal forest to capture atmospheric mercury may increase the risk of mercury related water quality issues in conifer-dominated systems (Witt *et al*. 2009).

Land-use changes and anthropogenic disturbances may significantly increase mercury export from the watershed (Munthe *et al.* 2007). Disturbance within the watershed can alter the transport, transformation, and bioavailability of mercury in surface water (Driscoll *et al*. 2007).

Mercury has been shown to increase in bioavailability when water levels are raised for hydroelectric development (Hecky *et al.* 1991; Bodaly *et al.* 1997), forests are clear-cut (Garcia and Carignan 2005; Garcia *et al.* 2007), water levels are lowered for dewatering of muskeg for gem or mineral deposits (Lean 2007), and when water chemistry is affected (Lindqvist *et al.* 1991).

Water level manipulations within existing reservoirs and reservoir creation can elevate mercury levels in biota (Evers *et al*. 2007). The flooding of soils during reservoir creation yields a flux of mercury and detrital matter to the associated water in addition to forming an ideal methylating environment along the newly formed soil-water interface (Evers *et al*. 2007). Similar environments are created by water level fluctuations caused by damming, since the littoral zones are ideal environments for methylation because bacterial sulfate reduction is promoted under transitioning reduction-oxidation conditions (Evers *et al*. 2007). Schetagne and Verdon (1999) observed fish mercury increases of 1.5 to 4 times the natural background levels with concentrations peaking 10 to 15 years after dam construction. Once reservoirs are no longer manipulated or managed it is expected to take 20 to 40 years for fish mercury concentrations to return to initial background levels (Schetagne and Verdon 1999).

In much of the boreal climatic zone silviculture is the most widespread and important anthropogenic influence on watershed processes (Kreutzweiser *et al*. 2008). Researchers have only recently begun to investigate the effects of forest harvesting on the mercury dynamics in aquatic environments (Browne 2007). During the last century, forest harvesting disturbance has exceeded forest fire as the primary disturbance agent in vast areas of the Boreal forest (Carignan *et al*. 2000). Logging disturbance can alter the biogeochemical processes of boreal forest watersheds by changing the forest composition, plant uptake rates, soil conditions, moisture and temperature regimes, soil microbial activity and water fluxes (Carignan and Steedman 2000; Kreutzweiser *et al*. 2008). Forest harvesting alters the hydrologic cycle, soil processes, and plant communities thereby increasing the risk of altering the cycling and bioavailability of mercury in the terrestrial and aquatic ecosystem (Porvari *et al*. 2003; Garcia and Carignan 2005). Forest harvesting disturbance may severely change the soil structure by compaction from logging equipment and lead to hydrological flushing of nutrients and dissolved organic carbon from surficial organic soil layers into surface waters as a result of an elevated water table (Carignan and Steedman 2000).

Enhanced methylation of mercury and increased mobilization of mercury occurs once clear-cutting alters the forest hydrology and soil conditions. Reduced evapo-transpiration following fires and forest harvesting causes a rise in the groundwater table creating an anoxic environment in the flooded terrestrial soils (Garcia *et al*. 2007; Munthe *et al*. 2007). Decomposing organic material resulting from forest harvesting slash and increased sun exposure creates favourable warm and moist soil conditions where enhanced methylation occurs by sulfate reducing bacteria (Munthe *et al*. 2007). Increased DOC exports from logged watersheds can act as a vector for dissolved Hg from the terrestrial watershed to the lakes, thereby increasing the THg and MeHg concentrations in the aquatic ecosystem (Garcia *et al*. 2007). Increased levels of MeHg in runoff have persisted for greater than five years following logging and soil disturbance (Munthe *et al*. 2007). In southern Finland, significant increases of THg and MeHg were observed in the runoff from a small spruce forest after clear-cutting and soil treatment (Porvari *et al*. 2003). Five years following clear-cutting the median MeHg concentration in runoff was 1.9 times greater than before forest harvesting (Porvari *et al*. 2003). Comparable to forest harvesting, forest clearing after severe storm-fell events can lead to large increases in MeHg from forest soils to surrounding waters much to the same extent as clear-cutting (Munthe *et al*. 2007). Similarly, a temporary logging track inadvertently placed across a small brook in the Swedish long-term Gårdsjön reference watershed increased the annual outputs of MeHg by more than 3 fold for longer than half a decade (Munthe and Hultberg 2004).

Forest fire disturbance also impacts the cycling of mercury within the watershed ecosystem. The influence of fire on mercury dynamics depends on the watershed characteristics, the extent of damage to the soil organic layer, and associated changes in water chemistry (Carignan *et al*. 2000; Allen *et al*. 2005). Forest fires can promote mobilization of soil-bound Hg by increased leaching of DOC from the watershed. However, most of the mercury stored in fuel (forest litter) within the watershed is volatilized in the smoke from high intensity forest fires (Friedli *et al.* 2001; Sigler *et al.* 2003). Almost all (>95%) of the Hg volatilized by fire is elemental mercury (Friedli *et al*. 2001). The re-emitted mercury from fire is a significant component of the atmospheric mercury cycle as it represents an estimated 1.6 to 8% of all total mercury emissions worldwide (Friedli *et al*. 2001). Furthermore, forest fire has been shown to increase mercury accumulation by fish following nutrient export which caused increased productivity and restructuring of the food web (Kelly *et al.* 2006). The elevated Hg concentrations in fish following fire were a result of increased trophic position due to dietary alterations from *Hyalella* (detrivore) to increased piscivory and consumption of *Mysis* (zooplanktivore) (Kelly *et al.* 2006).

Major watershed perturbations can alter mercury export rates from the watershed and influence in-lake reactions that determine the fate of mercury within aquatic biota (Garcia and Carignan 2005). Fire and forest harvesting both affect runoff and DOC loading to lakes and have the potential to increase Hg-bound organic matter to lakes (Lamontagne *et al.* 2000; Kreutzweiser *et al.* 2008). Dissolved organic carbon concentrations were three-fold higher and lake water phosphorus and nitrogen concentrations increased by approximately two-fold in lakes with clear-cut watersheds (Carignan *et al.* 2000). Increased export of dissolved nutrients and DOC from the watershed caused subsequent short-term changes in the water quality resulting in implications for the ecological processes and biotic communities of receiving waters (Kreutzweiser *et al.* 2008). Following the increase in DOC after logging, mercury bioavailability was significantly higher at the base of the food chain and a concurrent increase in mercury levels was observed in the aquatic organisms (Garcia and Carignan 2005).

Piscivorous fish in lakes with harvested watersheds showed higher total mercury levels relative to fish in burned or reference lakes (Garcia and Carignan 2000). Garcia and Carignan (2000) observed significantly higher concentrations of mercury (expressed on a dry weight basis) in 560-mm northern pike of logged headwater lakes (3.4 µg/g, n=4) compared to northern pike of reference lakes (1.9 µg/g, n=8). The logged lakes had 11% to 72% of the watershed area affected by clear-cut disturbance (Garcia and Carignan 2000). Average mercury concentrations in fish of burned lakes were not significantly different than harvested or reference lakes (3.0 µg/g, n=7). Fire affected 50.1% of the watershed for one lake, and greater than 90% of the watershed area for six other lakes (Garcia and Carignan 2000). Wetland area was relatively consistent among lakes, and comprised an average of 1.8% (at the most 6%) of the drainage area for the studied lakes (Garcia and Carignan 2000). These Boreal Shield study lakes were relatively small, with a mean lake area of 0.4 km^2 (40 ha), and similar with respect to depth and watershed morphometry. All but one were headwater lakes and the variability in fish Hg explained by their statistical model increased from 79% to 92% when a second-order lake was removed from the analysis (Garcia and Carignan 2000). Thus explasizing the importance of scale as lake size was an important factor influencing mercury bioavailability within disturbed watershed ecosystems.

Additional research on the same set of lakes demonstrated that trophic position, characterized by the tissue ratios of stable isotopes ($\delta\ ^{15}N/\ \delta\ ^{14}N$), was an important factor to consider when comparing fish from different systems with differences in watershed and water characteristics (Garcia and Carignan 2005).The use of trophic position allowed the direct comparison of trends in mercury concentration from fish of lakes subject to different levels and types of watershed disturbance. The mercury concentrations in fish, once normalized to trophic position, were significantly related to the ratio of clear-cut area within the watershed to lake area or lake volume (Garcia and Carignan 2005). The mercury concentration of northern pike, walleye, and burbot was found to be 2 to 3 fold higher for disturbed compared to reference lakes (Garcia and Carignan 2005). Harvesting disturbance affected 9% to 72% of the total watershed area for the cut lakes (n=9), whereas reference lakes (n=20) had remained undisturbed for 70 years. Total mercury in top predator fish from lakes with partially burned watersheds was less than in cut lakes and did not differ from the undisturbed lakes (Garcia and Carignan 2005).

The effects of changes to lake ecosystems following forest disturbance on fish are not well studied but effects are likely dependent upon the characteristics of the lake and the surrounding landscape (Browne 2007). A synthesis of research available up to 2006 estimated that between 1/10 and 1/4 of the mercury in fish of high-latitude, managed forest landscapes could be attributed to forest management (Bishop *et al*. 2009). Since this estimate was made, large differences in the magnitude of the mercury response to different forest operation techniques have appeared (Bishop 2011). The watershed and lake ecosystems' response to forest management has ranged from very little observable change to manifold increases of upland leakages and bioaccumulation of methylmercury.

The objective of this chapter was to examine the associations between lake and watershed characteristics and mercury concentrations in common top predator species of fish in northern Ontario lakes. If forest harvesting generally results in an increase in the net export of mercury and production of MeHg within a watershed/lake ecosystem, then I expected lakes with forest harvesting disturbance would have elevated fish mercury concentrations. In addition, variability in fish mercury concentrations was expected to be associated with differences in the spatial attributes of the watershed and lake chemistry characteristics.

The specific objectives of this study were to:

1) Investigate the association between THg in fish tissues with the amount of disturbance (natural or human caused) within the waterbody catchments of the lakes.

I hypothesized that if forest harvesting within the waterbody catchment area increases the net transport of mercury into a lake, thereby increasing mercury methylation and bioavailability, then fish within lakes with forest harvesting within their waterbody catchment would have consistently elevated mercury concentrations relative to fish in lakes with undisturbed waterbody catchments. I predicted that the total amount of mercury in piscivorous fish was negatively related to lake size and positively related to the ratio of (clear cut area within the waterbody catchment / waterbody catchment area).

2) Investigate the association of THg in fish tissue with lake and waterbody catchment scale variables that are believed to influence methylation potential or transport of mercury to the aquatic environment.

At the lake scale, I hypothesized that if lake chemistry affects mercury methylation and accumulation in fish, then elevated fish mercury concentrations would be found in lakes with low pH, high DOC, and high sulfate concentrations.

At the waterbody catchment scale, I hypothesized that if wetlands are effective hotspots of methylmercury production and a major contributor to mercury loading to lakes, then lakes with abundant associated wetlands would have higher fish mercury concentrations compared to lakes with low wetland association, as a result of increased mercury bioavailability. I predicted the concentration of mercury in piscivorous fish would be positively related to the abundance of contiguous wetlands present within the waterbody catchment of a lake. I expected the total amount of mercury in piscivorous fish to be positively related to the ratio of (wetland area within the waterbody catchment / waterbody catchment area).

Methods

Study Lakes

In 2008, the Ontario Ministry of Natural Resources (OMNR) implemented the Broad-Scale Fisheries Monitoring (BSFM) program, a pillar of the new Ecological Framework for Fisheries Management, to monitor the health of Ontario's inland lakes (OMNR 2009). The province of Ontario was divided into 20 management zones among the three main districts (Northwest, Northeast, and Southern Region). The BSFM survey collected basic information on hundreds of lakes across a large geographic area in a short period of time. Survey lakes are representative examples of the Boreal lakes in northern Ontario as they were randomly selected within each management zone (OMNR 2010). For this study a subset of BSFM lakes were selected from Fisheries Management Zones (FMZs) 4, 6, 7, 8, 10, and 11. (Figure 1.1 and Appendix A). One of the main goals of the BSFM is to identify the connections between stresses (natural or human-induced) and condition of aquatic resources (OMNR 2010). Selected study lakes were those for which fish mercury contamination analysis had been conducted by Ontario Ministry of Environment (OMOE). Lakes ranged in size and were grouped according to lake surface area: 33 lakes in bin 1 (<100 ha), 79 lakes in bin 2 (100 to 500 ha), 56 lakes in bin 3 (500-1500 ha), 58 lakes in bin 4 (1500 to 5000 ha), and 17 lakes in bin 5 (>5000 ha). The location coordinates and lake surface areas of study lakes belonging to each FMZ are summarized in Appendix A (Tables A-1 to A-6).

Geospatial Characterization of Waterbody Catchments

The drainage area defined by the waterbody catchment is the local area that directly contributes rainfall or runoff to a waterbody which consists of all the associated shoreline catchments as well as the waterbody itself but excludes the sub-catchments of upstream lakes (Figure 1.2) (Furnans and Olivera 2000). The waterbody catchment is of particular importance to reservoir planners and water quality modelers (Furnans and Olivera 2000). The waterbody catchment of each lake was delineated from the OMNR's enhanced flow direction grid (version 2.0, 20 m resolution) (efdir) using the Hydrology Tools in the Spatial Analyst extension for ArcGIS (version 9.3.1, developed by the Environmental Systems Research Institute (ESRI)). The efdir is a D8 flow direction grid that was generated by the OMNR's Water Resources

Information Program following methods developed by Kenny and Matthews (2005) to incorporate mapped surface hydrology features (streams, lakes, and other waterbodies) and flow directions interpreted from the Ontario provincial digital elevation model (DEM) (OMNR 2005). The enhanced flow accumulation grid (efacc) was derived from the efdir using ESRI's Hydrology Tools. Waterbody catchment characteristics were quantified using ArcGIS by both analyses of DEM derivatives and geometric intersections with watershed polygons and other spatial data layers (Appendix B). Intersection outputs were written to ArcGIS Personal Geodatabases (PGDb) that were subsequently accessed through Microsoft Access (version 2003). Structured Query Language (SQL) queries were created to summarize waterbody catchment characteristics from geometric intersection output tables.

The spatial attributes of the lake and waterbody catchment (summarized in Table 1.1) were calculated in ArcGIS and summarized in MS Access for each study site (Appendix B: Table B.1). A summary of the provincial datasets that were used to calculate waterbody catchment scale characteristics as well as site specific waterbody catchment characteristics are summarized for each site in Appendix B. Since the effects of watershed disturbance on aquatic biota can often be long lasting, forest harvesting disturbance and natural disturbance (insect damage, blowdown, fire) were summarized from 1990-1999 and 2000-2009 for each waterbody catchment. Elevated mercury concentrations in fish can be observed for 20 to 30 years after severe watershed disturbances as a result of the long turnover time of fish populations due to long lifespans of piscivorous species and the long half-life of MeHg in fish (Bodaly *et al*. 1997). The climate data (average summer temperature and precipitation according to 1990-1999 and 2000-2009) were obtained in ArcGIS with the Historical Climate Analysis Tool (HCAT 2012).

Table 1.1. Acronyms and Units of Lake and Watershed Characteristics

Acronym	Description	Units
WBDY_area_ha	waterbody catchment surface area	hectares (ha)
Lake_area_ha	lake surface area	ha
WBDY_LAKE_RATIO	surface area ratio of lake to waterbody area	no units
Wetlandi_p	percentage of wetland cover in the waterbody catchment	percentage area of waterbody catchment (%)
Stream_km_ha	density of streams and rivers	km/ha
Virtual_Flow_km_ha	water virtual flow density	km/ha
Bedrock_p	proportion bedrock (surficial geology)	%
glaciofluvial_p	proportion glaciofluvial (surficial geology)	%
glaciolacustrine_p	proportion glaciolacustrine (surficial geology)	%
morainal_p	proportion morainal (surficial geology)	%
organic_p	proportion organic matter (surficial geology)	%
Roads_km_ha	road density	%
Harv1990-2000	forest harvesting between 1990-2000	%
Harv2000-2009	forest harvesting between 2000-2009	%
Dist1990-2000	natural disturbance between 1990-2000 (blowdown, insect damage, fire)	%
Dist2000-2009	natural disturbance between 2000-2009	%

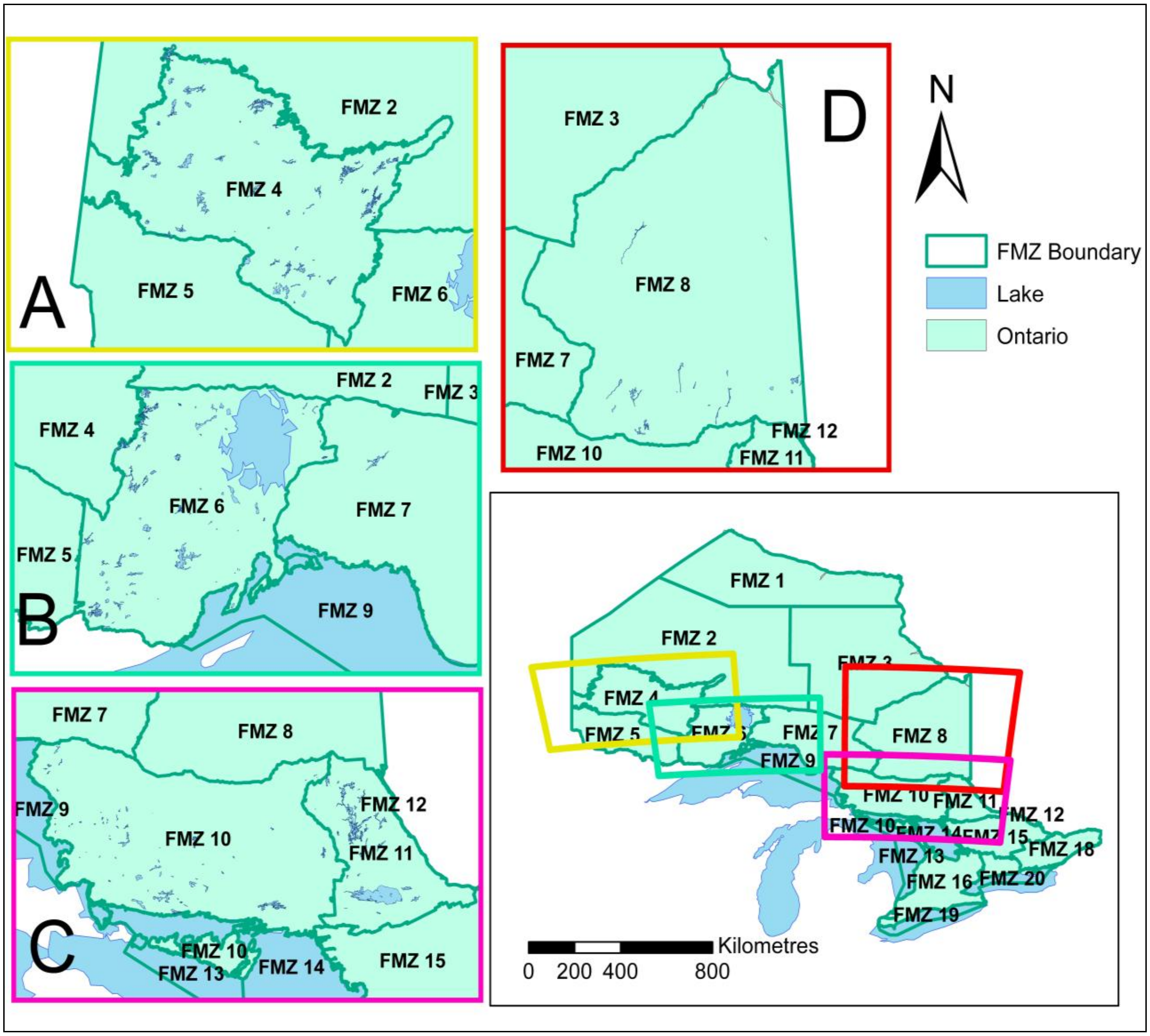
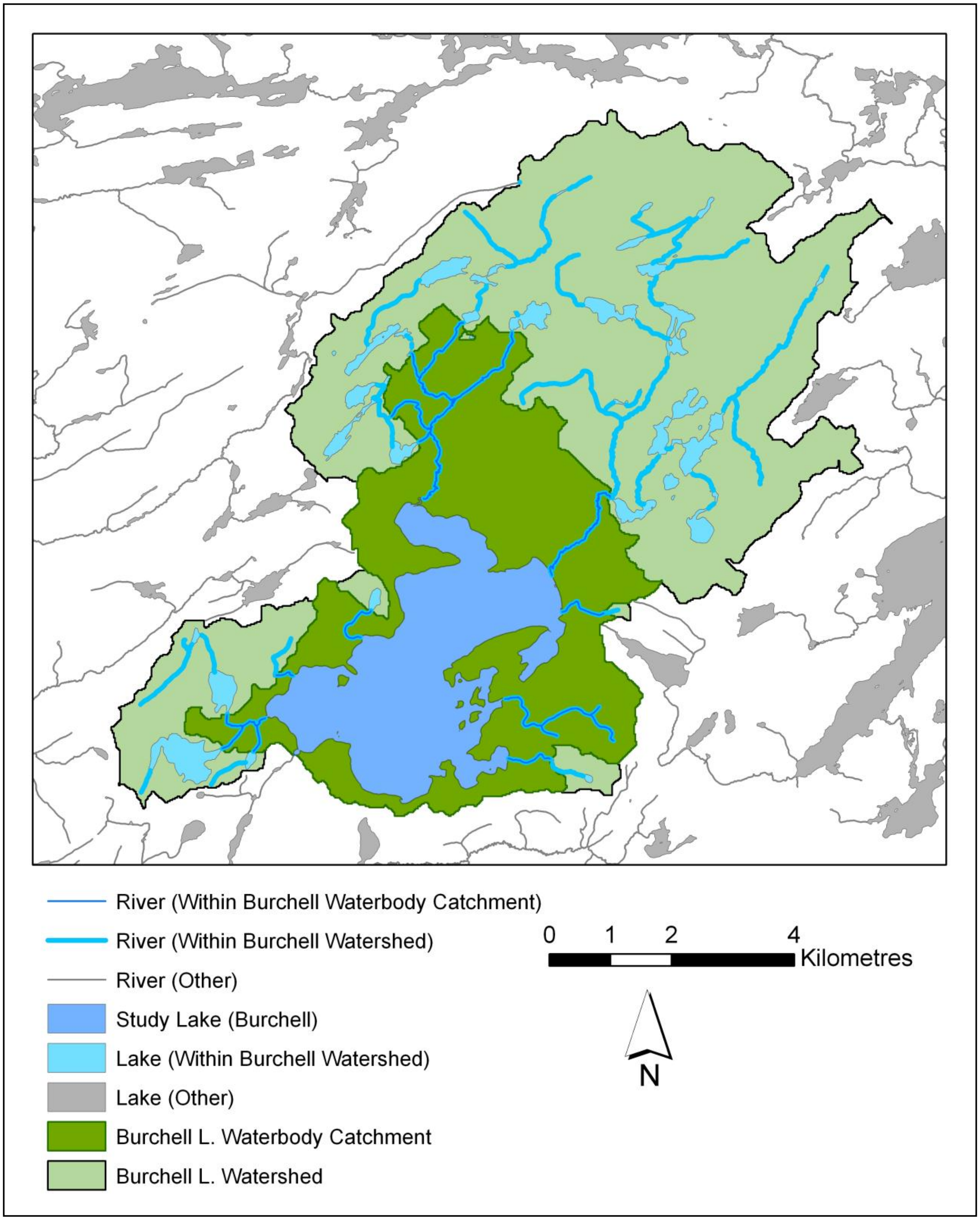


Figure 1.1. Study lakes in Fisheries Management Zones (FMZ) 4 (inset A), 6 and 7 (inset B), 8 (inset D), and 10 and 11 (inset C).


River (Within Burchell Waterbody Catchment)
River (Within Burchell Watershed)
River (Other)
Study Lake (Burchell)
Lake (Within Burchell Watershed)
Lake (Other)
Burchell L. Waterbody Catchment
Burchell L. Watershed
0 1 2 4
Kilometres
N

Lake Chemistry Collection and Analysis

Spring water quality data were collected by the OMNR and OMOE from select BSFM lakes (Ingram *et al.* 2006). The whole-lake composite samples were collected from the middle of each lake at a depth of 5 m. Collection of water samples occurred in 2008 and 2009 shortly after ice off while lakes were thermally mixed and a single sample is presumed to represent the entire lake's chemistry (Ingram *et al.* 2006). Within the perishability limit of 5 days, water samples were shipped and analysed at the OMOE Dorset Environmental Science Centre following standard analytical protocols (OMOE 1983). Water quality parameters that were measured are summarized in Table 1.2 and are presented for each lake in Appendix C (Table C.1).

Table 1.2. Acronyms and Units of Water Chemistry Variables

Acronym	Description	Units
ALKTI	gran alkalinity	mg/L as $CaCO_3$
CAUT	Calcium	mg/L
CLIDUR	chloride	mg/L
COLTR	true colour	True Colour Units (TCU)
COND25	Conductivity	µS/cm
DIC	dissolved inorganic carbon	mg/L
DOC	dissolved organic carbon	mg/L
KKUT	Potassium	mg/L
MGUT	Magnesium	mg/L
NAUT	Sodium	mg/L
NNHTUR	ammonia + ammonium	mg/L
NNOTUR	nitrate+nitrite	mg/L
NNTKUR	total Kjeldahl nitrogen	mg/L
pH	pH	pH units
PPUT	total phosphorus	mg/L
SIO3UR	reactive silicate	mg/L
SSO4UR	Sulphate	mg/L
FEUT	Iron	mg/L
MNUT	Magnesium	µg/L

Fish Sampling

Fish population information was gathered from the selected lakes during the summer BSFM netting program. Field methodology for the collection of fish involved the North American Large Mesh (NA1) and Ontario Small Mesh (ON2) gill netting procedures (Sandstrom *et al.* 2011). Fish collected for contaminant sample analysis were netted with the North American large mesh gillnet which has 8 different mesh sizes per gang (stretch measurements: 38, 51, 64, 76, 89, 102, 114, 127 mm) (Sandstrom *et al.* 2011). Equal allocation of effort was distributed across the depth strata and across all regions of the lake (Sandstrom *et al.* 2011). Upon fish collection, the length, weight, contaminant sample, and ageing structures were taken. From each lake, typically 10 to 20 fish tissue samples (consisting of approximately 50 g of lean, dorsal, skinless, boneless muscle tissue from above the lateral line) were collected from sport-fish species of various edible sizes (walleye, northern pike, lake trout, brook trout, and smallmouth bass [*Micropterus dolomieu*]) and frozen at -20°C until further analysis (Sandstron *et al.* 2011**).** Frozen fish tissue samples were sent to the OMOE Sport Fish and Biomonitoring Unit lab in Toronto, Ontario, for total mercury analysis by cold vapour-flameless atomic absorption spectroscopy at an accredited lab certified by the Canadian Association for Laboratory Accreditation. Total mercury (THg ppm wet weight (w.w.)) was determined for each fish following standard OMOE protocols (OMOE 2006). Ageing materials were sampled from individual fish according to protocols by Mann (2004). Structures were analyzed and interpreted at the OMNR's Regional Ageing Laboratory in Dryden, Ontario (Mann 2004).

Standardizing Mercury in Fish Tissue

In order to have comparable contaminant information among lakes, total mercury concentrations in fish of a standard length were used for inter-lake comparisons of fish populations. The THg concentrations standardized to a mean total length exhibit important inter-lake variability for all species of fish even within the same region (Tremblay *et al*. 1998; Simoneau *et al*. 2005). The fish dataset, compiled from OMNR and OMOE databases, contained information on 2250 walleye (121 lakes), 1191 northern pike (106 lakes), 940 lake trout (60 lakes), 241 brook trout (18 lakes), and 518 smallmouth bass (37 lakes) (see Appendix D: Table D.2). Standardized lengths were chosen based on the OMOE standard lengths calculated from the population size distributions of various fish species (Gewurtz *et al.* 2010) as well as from analysis of the mean total length of each species in the dataset (Appendix D: Figure D.1). Standards for brook trout and smallmouth bass were based on the population means from this dataset alone, as no standard was available from the OMOE. The standard total lengths by species were: walleye (500 mm), northern pike (650 mm), lake trout (600 mm), brook trout (300 mm), and smallmouth bass (400 mm). The standard mercury level (µg/g wet weight of skinless boneless dorsal muscle = ppm w.w.) for each fish population was calculated by using the power function

$$\text{Equation 1.} \quad (C_L=aL^b) = (\log C_L=\log(a)+b^* \log(L))$$

C_L is the concentration at length L (cm), a is a constant, and b is the power of the relationship between concentration and length on fish data, specific to one netting season with more than 3 data points, (Gewurtz *et al.* 2011) (Appendix D). The power-series (log-log) regression is used by the OMOE Sports Fish Contaminant Monitoring Program to calculate the relationship between contaminant concentration and length (Gewurtz *et al*. 2011). The fit of the relationship between fish total mercury concentration and total length was evaluated in order to determine the most appropriate model. Compared to the polynomial or linear regression, the log-log regression was the most appropriate model used to describe the relationship between mercury concentration and length (Gewurtz *et al*. 2011).

Statistical Procedure

A correlation analysis was performed and inter-correlated variables (Pearson correlation coefficients (r) > 0.9) were removed from the dataset in order to minimize multi-collinearity in ordination analyses. Nonmetric multidimensional scaling ordinations were used to examine the association between spatial attributes of the waterbody catchment, lake chemistry characteristics and total mercury concentrations for each species. Nonmetric multidimensional scaling is considered the most robust unconstrained ordination for use in community ecology as it makes few assumptions about the nature of the data (Minchin 1987; Holland 2008). Ordinations were performed in R (version 2.13.1) using the package *vegan* (version 2.0-2) which was designed for ecological data analysis (Okasanen 2011). The same set of waterbody catchment and lake scale variables were used in ordinations of lakes and watersheds for each species. A subset of the 243 lakes was included in separate ordinations characterized by fish species and lakes without water quality data were excluded from the sample set (walleye = 107 lakes, northern pike = 97 lakes, lake trout = 54 lakes, smallmouth bass = 35 lakes, brook trout = 15 lakes). Ordination analysis was also used to show the associations between the physical watershed and lake scale variables and their relationship with spring lake water chemistry variables.

Dissimilarity was measured as Bray-Curtis distances and random starting configurations (the maximum number is given by *trymax*) were used to reach a stable solution. The metaMDS function uses the monoMDS function which implements Kruskal's NMDS using monotone regression and weak treatment of ties (Kruskal 1964 a, b). The dimensionality of each ordination was determined using the scree plot of stress vs. dimensionality for each individual ordination. The goodness of fit was determined by the non-metric r^2 value based on stress *S* from the Shepard plot. Once the dimensionality was determined, the data were fitted into the dimensions with no hidden axes of variation. In order to avoid local minima (*trymax*=1000), random starting configurations were used until a convergent solution was reached. Scores were scaled, centred and rotated. A square root transformation and Wisconsin double standardization of the data matrix was used to improve the results of the ordination. The final solutions were compared by using Procrustes analysis (root mean squared error [rmse]). The solution was regarded as a convergent stable solution if two solutions were very similar in their Procrustes rmse and the largest residual was very small.

Based on the ordinations, individual lake or waterbody catchment scale variables that were associated with THg concentrations in fish were selected for further analyses. The correlations (r) between watershed scale variables, lake scale variables, and standardized fish THg concentrations were examined to further refine the subset of variables for further analyses. The relationship between selected lake chemistry parameters and fish THg concentrations for all species was examined with a general linear model analysis. Similarly, the relationships between harvesting or natural disturbance and fish THg concentrations for all species were examined using general linear model analyses. To determine if the relationship between THg concentrations and lake chemistry or waterbody catchment scale parameters differed among the species of fish an analysis of covariance (ANCOVA) was used.

Three categories of contamination for mercury concentrations in fish tissue were created to summarize the variability in mercury concentrations for further analysis of the associations between fish standardized mercury concentrations and watershed characteristics for the five species of fish. The low category is based on the Canadian standard for the commercial sale of fish and represents the group of lakes with less than 0.5 ppm w.w. THg in the standard fish. The moderate category represents lakes having standard fish of intermediate contamination ranging in concentrations from 0.5 to 1.0 ppm w.w. THg. The elevated category represents lakes with standard fish having concentrations that exceed the American FDA "action level" of 1.0 ppm w.w. THg for commercially sold fish. The difference in waterbody catchment disturbance from 1990-2009 and 2000-2009 or wetland percentages were compared among the three categories of contamination. Differences among groups were evaluated for each species with the non-parametric Kruskal-Wallis analysis of variance (ANOVA).

The relationship between fish THg concentrations with the lake to waterbody catchment area ratio or lake surface area was determined with general linear model analysis. Analysis of covariance (ANCOVA) was used to determine if the relationship between fish THg concentrations with either the lake to waterbody catchment area ratio or lake size differed according to species. Lake surface area was divided into 5 size classes to summarize the variability in lake size in order to make additional comparisons to fish mercury concentrations among lakes. Differences in fish THg concentrations related to lake size were evaluated among the five lake surface area classes with the non-parametric Kruskal-Wallis analysis of variance (ANOVA) and Dunn's pair-wise comparison test.

Results

Catchment Analysis

Waterbody catchment area for study lakes ranged from 69 ha (Beak Lake) to 104,362 ha (Lake Nipissing) (mean 5362±9573 ha, n=243). Surface area ranged from 7 ha (Beak Lake) to 72861 ha (Lake Nipissing) (1739±5182 ha). Lake surface area/waterbody catchment area ratio ranged from 0.01 (Bear Lake, a small lake with a large direct drainage area) to 0.70 (Lake Nipissing, a large lake with a relatively small direct drainage area). Spatial waterbody catchment variables were summarized for each lake in Appendix B (Table B.1). Between 2000 and 2009, forest harvesting ranged from 0% to 40% of the total waterbody catchment area and natural forest disturbance ranged from 0% to 100% of the total waterbody catchment area for the 243 lakes used in this study. In total, 138 lakes (approximately 57%) had forest harvesting in the catchment area and 42 lakes (approximately 17%) had natural disturbance events between 2000 and 2009. The percentage cover of wetlands was also variable, covering 0% to 34.9% of the total waterbody catchment area.

Fish Mercury Concentrations

Species specific ranges in THg concentration, based on fish of standardized lengths are presented in Figure 1.3. The standardized fish THg concentrations are variable among species within and among lakes (Appendix D: Table D.2). On average, the length-standardized mercury THg concentration in walleye (mean 0.78 ± 0.36 ppm w.w. for TLEN=500 mm), northern pike (mean 0.73. ± 0.39 ppm w.w for TLEN=650 mm) and lake trout (mean 0.66 ± 0.52 ppm w.w. for TLEN=600 mm) exceeded the Canadian standard of 0.5 ppm w.w. THg in fish tissue.

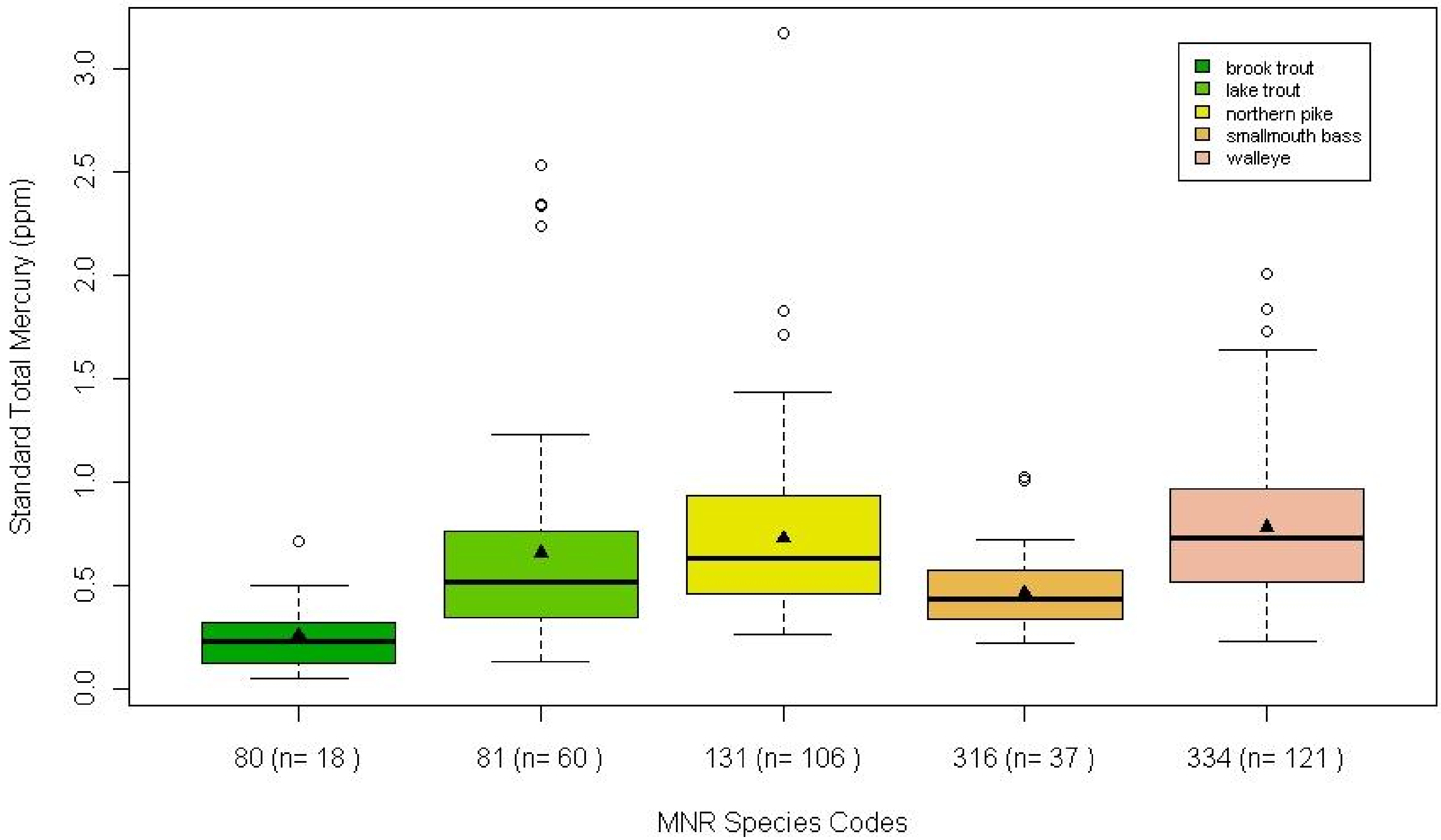


Figure 1.3 Boxplot of standardized total THg concentrations in fish tissue (ppm w.w.) according to species. Horizontal black lines represent the median, the boxes indicate the inter-quartile range (IQR) and the whiskers represent 1.5 times the IQR from the first and third quartile, hollow circles represent data outside of the(Q1-1.5*IQR, Q3+1.5*IQR), black triangle represents mean, n is number of lakes

Multivariate Analysis of the Associations between Fish THg concentrations and Lake and Catchment Scale Characteristics

Walleye THg Associations with Catchment and Water Chemistry Characteristics

The subset of 107 study lakes that contained walleye varied in their waterbody catchment and water chemistry characteristics but showed no clear groupings of different lake types. Waterbody catchments ranged from those dominated by bedrock to catchments with higher proportions of glaciolacutrine and morainal deposits (axis 1, Figure 1.4). Lakes with higher amounts of DOC and nutrients (total Kjeldahl nitrogen (NNTKUR) and total phosphorus (PPUT)) tended to have catchments with higher values for stream density (km/ha), a greater proportion of catchment area classified as organic surficial material, and wetland areas and were distinct from lakes with high secchi depth, large lake surface area and large waterbody catchment area (axis 2, Figure 1.4). Overall, the variability summarized by the ordination is consistent with expectations of inland lakes of northern Ontario which range from small dystrophic lakes rich in humic materials where the majority of organic matter originates from the watershed (allochthonous) to large clear oligotrophic lakes. The vast amount (94.5%) of among lake variability is captured by this 2-dimensional solution. The fit of the ordination is satisfactory as a convergent solution was found after 365 iterations where the final stress = 0.234 (procrustes: rmse 0.0012, max resid 0.0085).

Total mercury in walleye was only weakly associated with waterbody catchment and lake water chemistry characteristics. Only 5.4% of the variability in walleye THg concentration was explained by the ordination (r^2=0.054, p=0.066), however, the trend in walleye THg concentration illustrated by the vector (Figure 1.4) was consistent with predicted patterns. As indicated by the correlation vector in the ordination, walleye THg concentrations were positively associated with the proportion of surficial material classified as organic matter within the waterbody catchment, stream density (km/ha), wetland area, DOC, and nutrients; characteristics of small dystrophic lakes. There was also a positive association between the walleye THg and forest harvest (2000-2009 and 1990-2000) and road density (km/ha). Conversely, walleye THg concentrations were negatively associated with lake volume, lake surface area, max depth of the water column, secchi depth, sulfate concentrations, the lake:catchment area ratio, and waterbody catchment area.

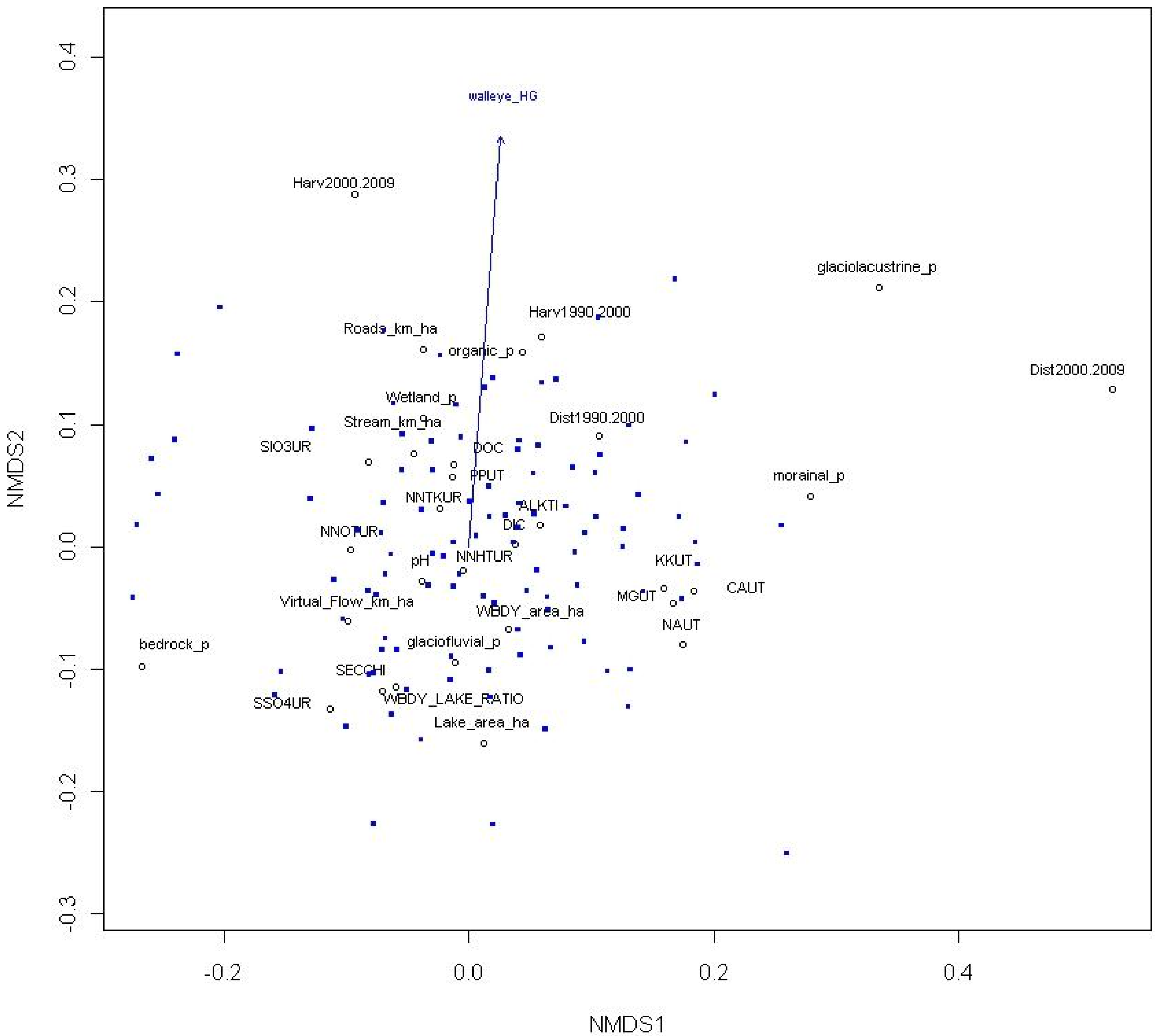


Figure 1.4. Non-metric multidimensional scaling ordination based on catchment and water chemistry characteristics of 107 walleye lakes. Blue squares represent individual lakes and open circles represent the variables used in the ordination. The correlation (r = 0.23) between lake characteristics and walleye THg concentrations is illustrated by the vector.

Northern Pike THg Associations with Catchment and Water Chemistry Characteristics

Similar to the lakes containing walleye, the 97 lakes that contained northern pike varied according to waterbody catchment and lake scale variables and no distinct clustering of different lake types was apparent. Surficial geology appears to influence major differences among lakes, as the catchments range from bedrock to finer glaciofluvial and glaciolacustrine deposits (Axis 1, Figure 1.5). Lake water alkalinity and base cation (magnesium (Mg^{2+}), calcium (Ca^{2+}), potassium (K^{+}), and sodium (Na^{+})) concentrations were positively associated with harvesting disturbance from 2000-2009 and road disturbance within the waterbody catchment and negatively associated with lake surface area (Figure 1.5). Approximately 94% of the differences among sites are explained by the 2-dimensional solution. A convergent solution was found after 260 iterations where the final stress = 0.245 (procrustes: rmse 0.0007, max resid 0.0036). This ordination is suitable for interpretation as the non-metric r^2 and final stress value are satisfactory.

Northern pike THg concentrations were very weakly associated with waterbody catchment and lake scale characteristics with only 1.2% of the variability in northern pike THg concentration explained by the ordination (r2=0.012, p=0.579). As indicated by the correlation vector in the ordination, northern pike THg concentrations showed positive associations with base cation concentrations, alkalinity, road density (km/ha), and the proportion of the waterbody catchment classified as glaciofluvial and glaciolacustrine surficial geology. Northern pike THg concentrations were negatively associated with the lake:catchment area ratio, lake surface area, and waterbody catchment area. There was no apparent association between northern pike THg concentrations and lake water DOC concentration, nutrients, or secchi depth .

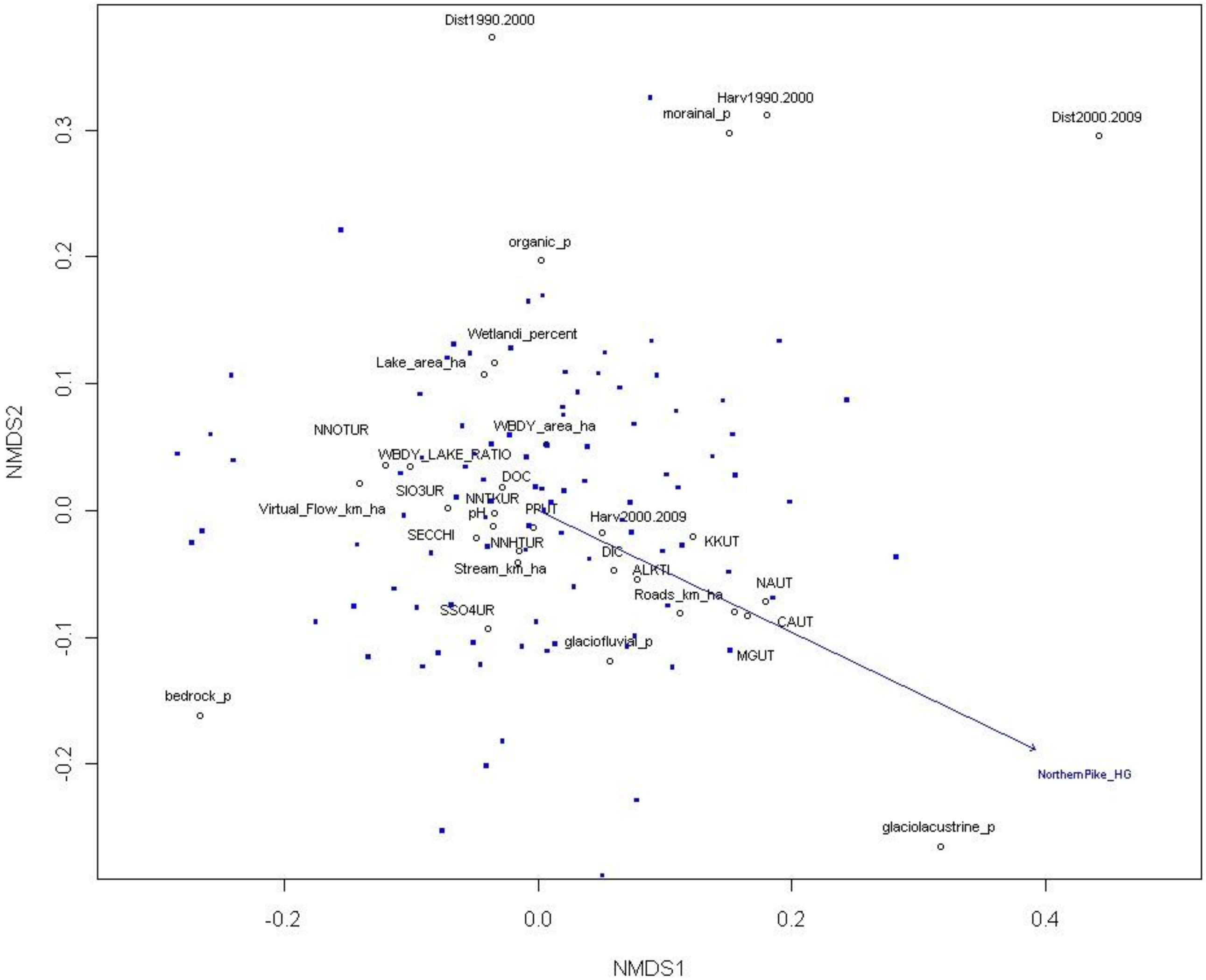


Figure 1.5. Non-metric multidimensional scaling ordination based on waterbody catchment and lake scale characteristics of 97 northern pike lakes. Blue squares represent individual lakes and open circles represent the variables used in the ordination. The correlation (r = 0.11) between lake characteristics and northern pike THg concentrations is illustrated by the vector.

Lake Trout THg Associations with Catchment and Water Chemistry Characteristics

The 54 lakes containing lake trout were relatively closely grouped in the ordination and are fairly similar in terms of waterbody catchment and lake scale characteristics (Figure 1.6). Similar to the subset of lakes containing walleye, lakes containing lake trout range from large clear lakes to small stained lakes (axis 2, Figure 1.6). Approximately 96.2% of the differences among lakes were explained by the 2-dimensional ordination solution. A convergent solution was found after 113 iterations where the final stress = 0.1942 (rmse 0.0013, max resid 0.0061).

Lake trout THg concentrations were also very weakly associated with waterbody catchment and lake scale variables. Only 1.0% of the variability in lake trout THg concentration was explained by the variables used in the ordination (r^2=0.01, p=0.778). Interpreting the relationship between fish THg concentrations with waterbody catchment and lake scale variables is risky since lake trout mercury concentrations were not significantly correlated to the ordination. However, lake trout THg concentrations showed positive associations with harvesting during the years 2000-2009 and 1990-2000, the amount of wetlands within the waterbody catchment area, the proportion of the surficial geology classified as organic material, stream density (km/ha), reactive silicate (SIO3UR), nutrients, DOC, and the percentage of glaciofluvial and glaciolacustrine surficial geology within the waterbody catchment. Lake trout THg concentrations show negative associations with base cations, sulfate, secchi depth, the waterbody catchment area, lake surface area, and the lake:catchment area ratio. Lake water pH does not appear to be associated with lake trout THg concentrations.

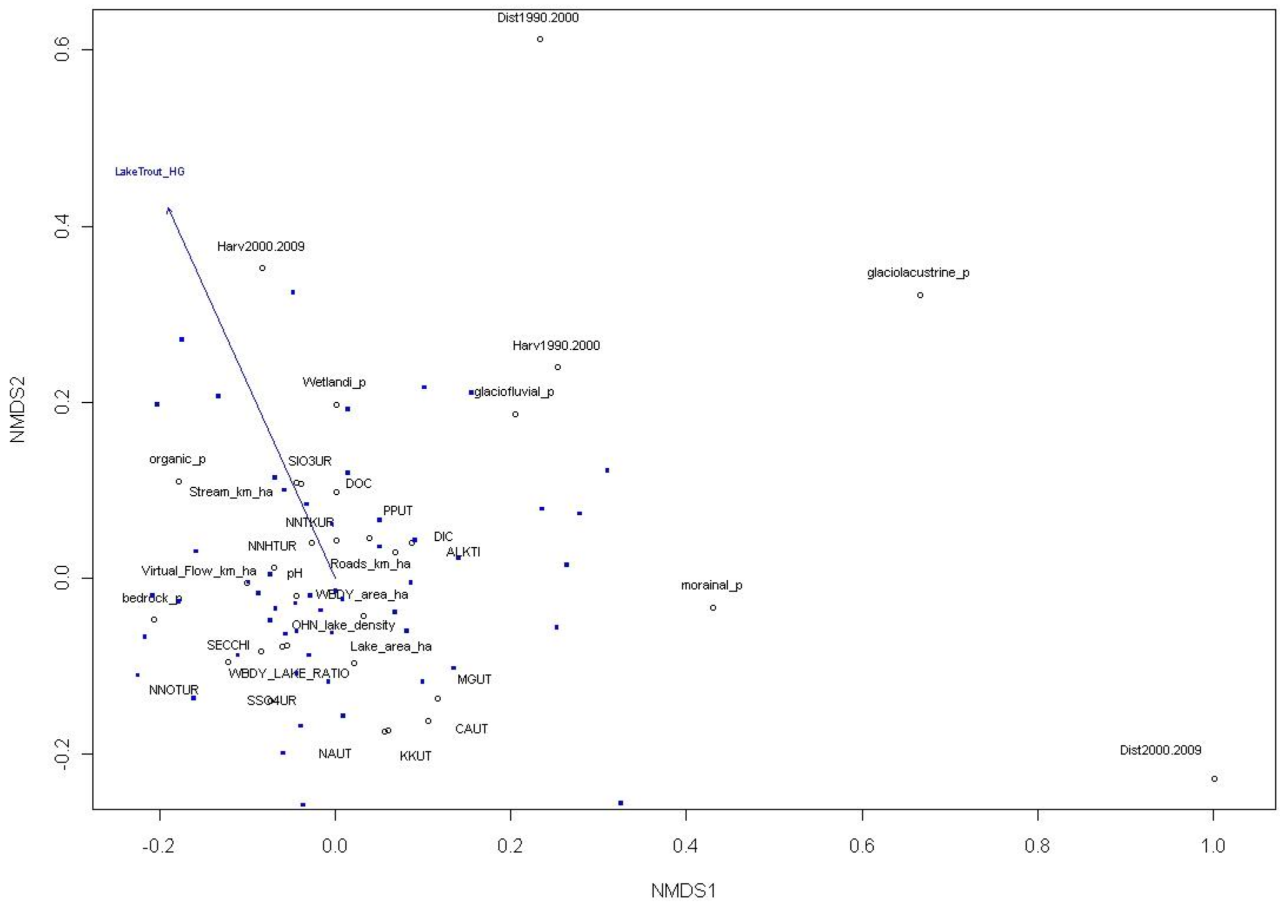


Figure 1.6. Non-metric multidimensional scaling ordination based on waterbody catchment and lake scale characteristics of 54 lake trout lakes. Blue squares represent individual lakes and open circles represent the variables used in the ordination. The correlation (r = 0.1) between lake characteristics and lake trout THg concentrations is illustrated by the vector.

Smallmouth Bass THg Associations with Catchment and Water Chemistry Characteristics

The 35 lakes containing smallmouth bass were more similar to each other based on waterbody catchment and lake scale characteristics compared to the preceding groups of lakes. Similar to the groups of lakes containing walleye or lake trout, lakes containing smallmouth bass range from large clear lakes to small stained lakes (axis 2, Figure 1.7). Forest harvesting and natural disturbance variables are distinctly separate from the other lake and waterbody catchment scale variables and these occurrences were associated with only a few lakes. Approximately 96.7% of the differences among sites were explained by the 2-dimensional solution. A convergent solution was found after 15 iterations where the final stress = 0.1807 (procrustes: rmse 0.0003, max resid 0.0008).

Waterbody catchment and lake variables explained 37.7% of the variability in smallmouth bass total THg concentration (r^2=0.377, p=0.001). Smallmouth bass THg concentrations are not closely associated with harvesting disturbance within the waterbody catchment between the two time intervals of 1990-2000 and 2000-2009 (Figure 1.7). However, natural disturbance within these two time periods appears to be more clearly associated with fish THg concentrations. There is a positive association with THg and the percentage of morainal and glaciolacustrine surfical geology. As indicated by the correlation vector in the ordination, smallmouth bass THg concentrations show negative associations with sulfate concentration, secchi depth, the percentage of bedrock surficial geology, lake surface area, waterbody catchment area, the lake:catchment area ratio. No observed association exists between lake water pH and smallmouth bass THg concentrations.

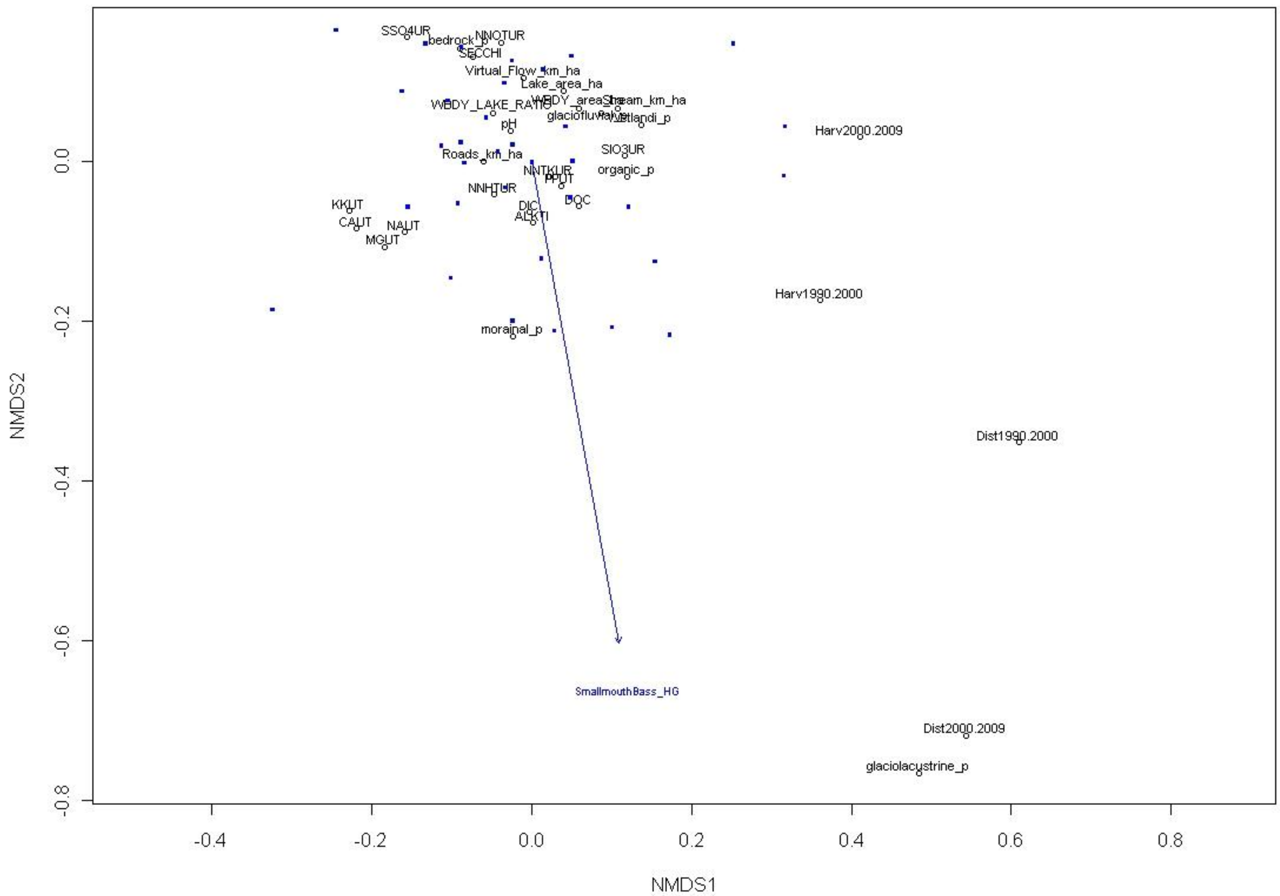


Figure 1.7. Non-metric multidimensional scaling ordination based on the waterbody catchment and lake scale characteristics of 35 lakes containing smallmouth bass. Blue squares represent individual lakes and open circles represent the variables used in the ordination. The correlation (r = 0.61) between lake characteristics and smallmouth bass THg concentrations is illustrated by the vector.

Brook Trout THg Associations with Catchment and Water Chemistry Characteristics

The 15 brook trout lakes were very similar in waterbody catchment and lake scale characteristics and are closely grouped in the ordination with no clear pattern of association with any environmental variables (Figure 1.8). The relationships among variables noted in previous groups of lakes are not as apparent for this dataset. The 2-dimensional ordination explains 98% of the differences among sites. A convergent solution was found after 10 iterations where the final stress = 0.1412 (procrustes: rmse 0.0024, max resid 0.0068).

There was a significant relationship between brook trout THg concentrations and waterbody catchment and lake scale variables (r^2=0.535, p=0.011). As indicated by the correlation vector in the ordination, brook trout THg concentrations showed positive associations with forest harvesting and natural disturbance which occurred between 1990-2000. The amount of wetlands within the waterbody catchment, road density (km/ha), DOC, and nutrients are also positively associated with brook trout THg concentrations. The lake:catchment area ratio and secchi depth are negatively associated with THg concentrations in brook trout. There is no apparent association between lake water pH and brook trout THg concentrations.

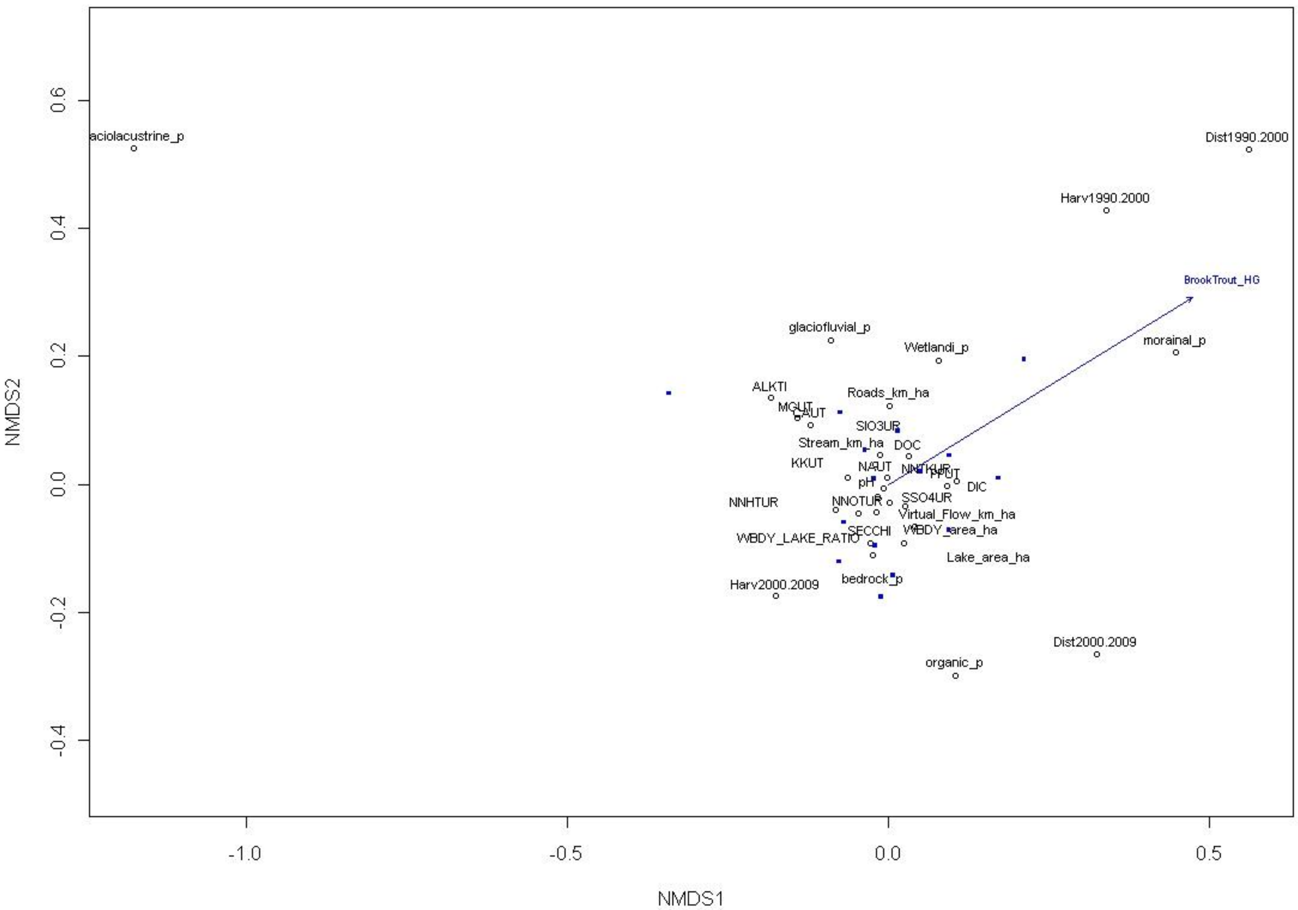


Figure 1.8. Non-metric multidimensional scaling ordination based on the waterbody catchment and lake scale characteristics for 15 lakes. Blue squares represent lakes. Blue squares represent individual lakes and open circles represent the variables used in the ordination. The correlation (r = 0.73) between lake characteristics and brook trout THg concentrations is illustrated by the vector.

Multivariate Associations between Water Chemistry and Waterbody Catchment and Lake Scale Characteristics

A NMDS ordination was used to assess the associations between waterbody catchment and lake scale factors and their relationship to spring lake water chemistry. No distinct grouping of lake type exists and the 175 lakes differ mainly by surficial geology characteristics of the waterbody catchments. The HCAT average decadal summer temperature and average decadal precipitation from 1990-1999 and 2000-2009 were not variable among lakes. This ordination summarized approximately 95.8% of among lake variability with the 2-dimensional solution. A convergent solution was found after 103 iterations where the final stress = 0.2143 (procrustes: rmse 0.0016, max resid 0.0089).

Nutrients, reactive silicate (SIO3UR), dissolved organic carbon, and base cations were correlated vectors that were positively associated with higher percentages of wetlands, finer glacial sediments and organic matter dominating the surficial geology of the waterbody catchment as well as disturbance from forest harvesting and roads ($p<0.05$, Figure 1.9). Dissolved organic carbon and total Kjeldhal nitrogen concentrations were the water chemistry variables with the strongest association with the variables used in the ordination (r^2=0.28, p=0.001 and r^2= 0.27, p=0.001 respectively). Forest harvesting disturbance during the 1999-2000 decade is distinctly separate from the 2000-2009 decade with respect to water chemistry as indicated by the distance between variables in ordination space. Secchi depth was positively associated with the lake:catchment area ratio and stream density (km/ha) and was significantly related to the ordination (r^2=0.18, p=0.001). Sulfate concentrations showed a weaker correlation to the ordination (SS04UR: r^2=0.05, p=0.015) but were closely associated with the percentage of bedrock within the waterbody catchment. Alkalinity (r^2=0.06, p=0.003) and base cation concentrations (CAUT: r^2= 0.05, p=0.007; KKUT: r^2=0.12, p=0.001; MGUT: r^2=0.09, p=0.001; NAUT: r^2=0.06, p=0.006) were correlated to one another and showed weak positive relationships with the percentage surficial sediments classified as organic material and disturbance within the waterbody catchment. The HCAT average decadal summer temperature and average decadal precipitation from 1990-1999 and 2000-2009 had no apparent associations with the lake water chemistry variables.

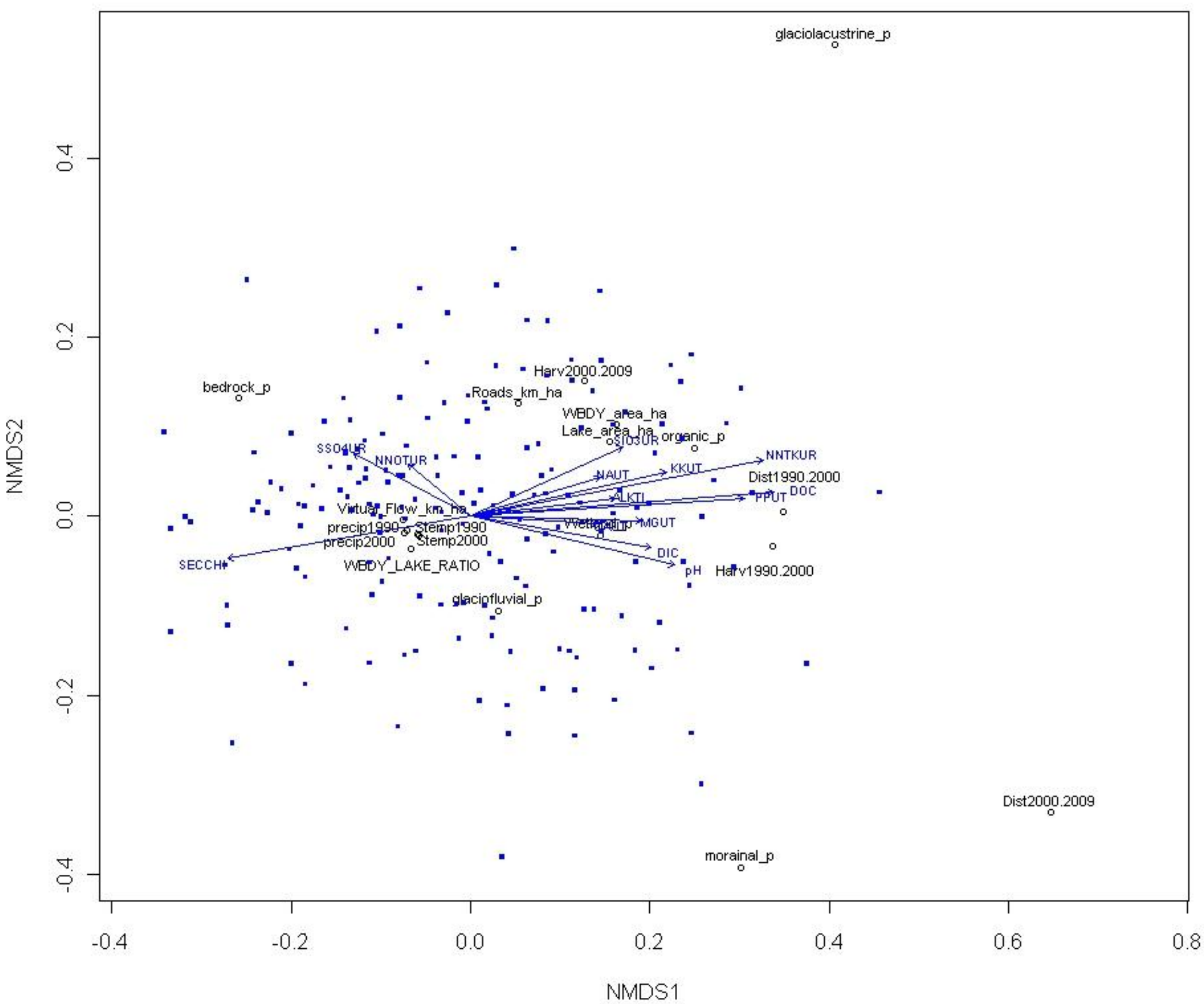


Figure 1.9. Non-metric multidimensional scaling ordination showing the associations between waterbody catchment features and correlated lake chemistry characteristics (n=175 lakes). Blue squares represent individual lakes and open circles represent the variables used in the ordination. The correlations between lake characteristics and water chemistry variables are illustrated by the vectors.

Univariate Relationships between Waterbody Catchment Scale or Lake Chemistry with Fish Mercury Concentrations

The correlations between THg concentrations in all fish species and characteristics of the waterbody catchment and lake which appeared most influential in the ordination analyses are shown in Table 1.3. The strongest and most consistent correlation was between DOC and THg concentrations in all species. Dissolved organic carbon concentrations were significantly positively related to THg concentrations ($F_{(1,\ 314)}$ =21.596, p<0.0001). and the relationship between fish THg concentrations and DOC concentrations was the same for the 5 species of fish examined as indicated by the ANCOVA analysis of the homogeneity of regression (intercept: $F_{(4, 314)}$ =0.684, p=0.603; slopes: $F_{(4, 314)}$ =0.463, p=0.763) in the subset of 175 lakes (Figure 1.10 a).

Secchi depth (3.28 ± 1.79 m) was consistently correlated to THg concentrations for all fish species (Table 3) and is a negatively associated covariate of DOC (7.69 ± 3.74 mg/L) (r=-0.736, p=0.0001). Secchi depth was significantly negatively related to THg concentrations ($F_{(1, 313)}$ = 21.67, p<0.0001). The relationship between fish THg concentrations and secchi depth was not the same for the 5 species of fish examined as indicated by the ANCOVA analysis of the homogeneity of regression (intercept: $F_{(4,\ 313)}$ =3.208, p=0.013; slopes: $F_{(4,\ 313)}$ =0.776, p=0.542). In this particular analysis, the intercept is significantly different amongst the 5 species of fish due to differences in the mean mercury concentrations (walleye 0.78 ± 0.36 ppm w.w., smallmouth bass 0.47 ± 0.19 ppm w.w., northern pike 0.73 ± 0.39 ppm w.w., lake trout 0.66 ± 0.52 ppm w.w., and brook trout 0.26 ± 0.16 ppm w.w.).

The DOC concentration in lake water was significantly positively correlated with the percentage of wetland area in the catchment (r^2=0.116, p<0.001) but was not significantly related to forest harvesting disturbance (r^2= 0.014, p=0.125) (Figure 1.11 a, b). The percentage of wetland area in a catchment was positively correlated to true colour (r = 0.309, p <0.001), secchi depth (r = -0.268, p<0.001), phosphorus (r = 0.245, p <0.01), total Kjeldahl nitrogen (r = 0.300, p <0.001), reactive silicate (r = 0.155, p <0.05), proportion of organic surficial material within the waterbody catchment (r = 0.387, p <0.001) and negatively correlated to secchi depth (r = -0.268, p<0.001).

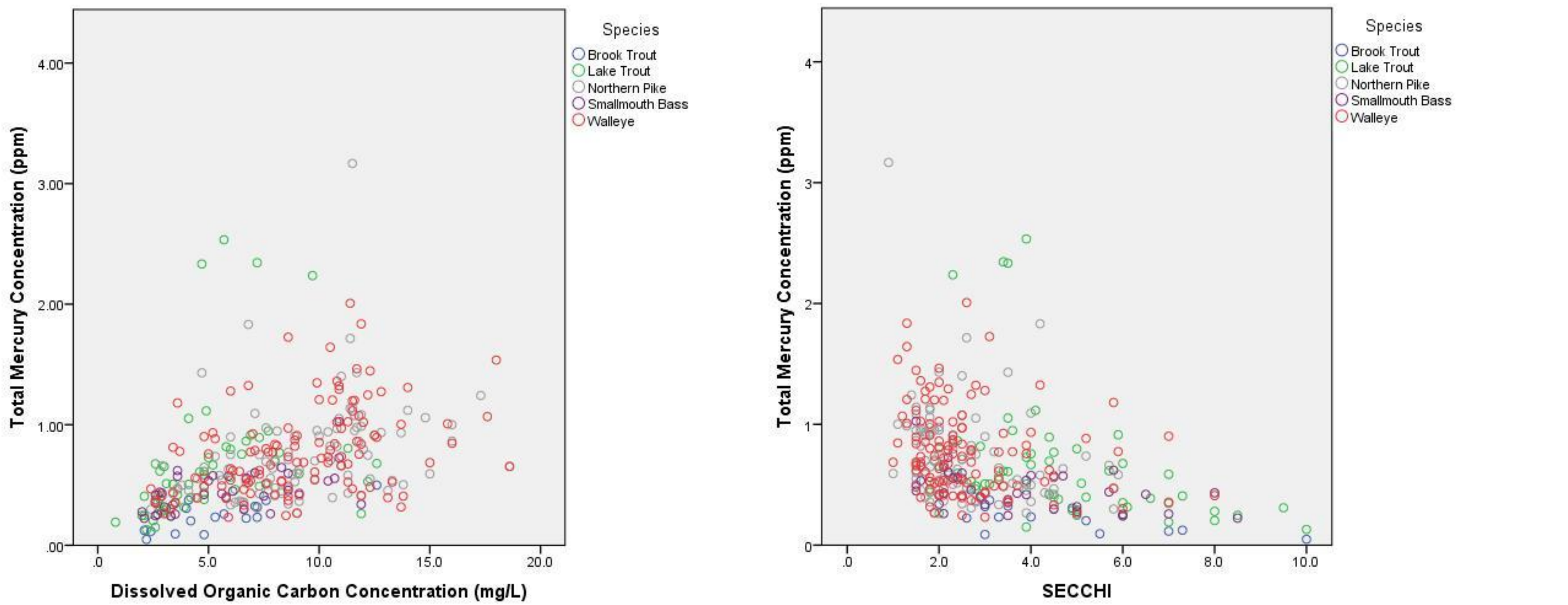
Further analysis of other water chemistry variables showed weak relationships with fish mercury concentrations. Total Kjeldahl nitrogen (323.3 ± 103.5 µg/L) and total phosphorus (9.2

± 5.2 μg/L) in lake water were correlated to THg concentrations in some but not all species of fish (Figure 1.12). When species were analysed separately, total Kjeldahl nitrogen was positively related to fish THg concentrations in brook trout (r = 0.526, p < 0.05), lake trout (r = 0.278, p <0.05), northern pike (r = 0.245, p <0.05), and smallmouth bass (r =0.485, p <0.01) but not walleye. Total phosphorus was only significantly related to fish THg concentrations in brook trout (r =0.608, p <0.05) and northern pike (r =0.309, p <0.01). Overall, fish THg concentrations were significantly related to total Kjeldahl nitrogen ($F_{(1, 314)}$ =7.297, p=0.007) but not to total phosphorus ($F_{(1, 314)}$ =2.196, p= 0.139) concentrations in lake water (Figure 1.12). The relationship between fish THg concentration and total Kjeldahl nitrogen concentration in lake water does not differ significantly as a function of species (ANCOVA analysis of the homogeneity of regression intercept: $F_{(4, 314)}$ =2.017, p=0.092); slopes: $F_{(4, 314)}$ =1.298, p=0.271)). Lake water pH (7.00 ± 0.40) and sulfate concentrations (3.64 ± 4.31 mg/L) did not span across a wide range values. Fish THg concentrations were not significantly related to lake water pH ($F_{(1, 313)}$ =2.477, p=0.117) nor sulfate concentrations ($F_{(1, 314)}$ =0.409, p= 0.523) (Figure 1.13).

Table 1.3. Pearson correlation coefficients for waterbody catchment and lake scale variables significantly associated with fish THg concentrations.

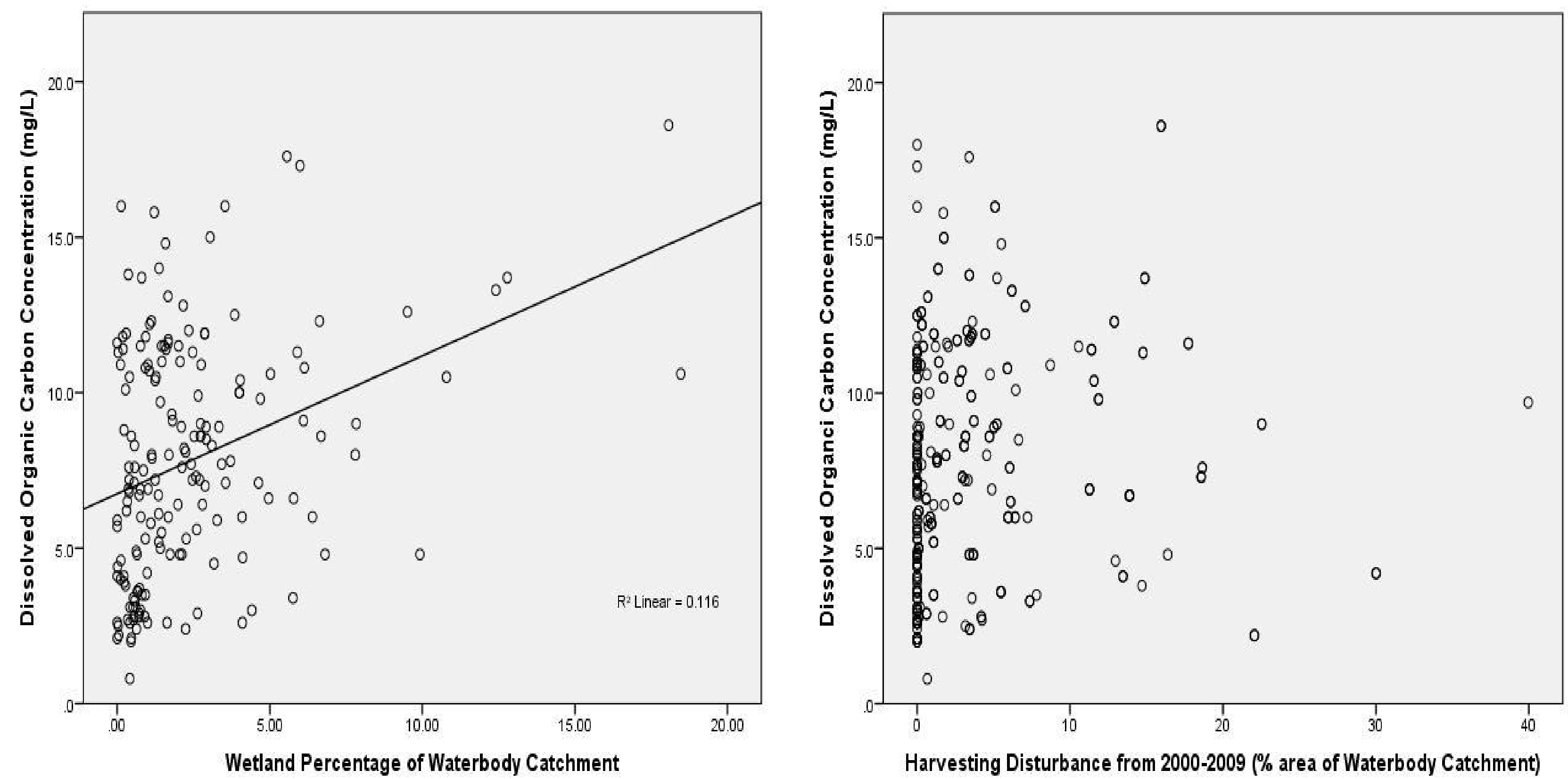
Variable	Walleye		N. Pike		Lake Trout		Brook Trout		Sm. Bass	
Bedrock_p									-0.432	**
Dist1990-2000							0.563	*		
Dist2000-2009			0.206	*						
DOC	0.381	***	0.370	***	0.347	**	0.786	***	0.544	***
Glaciolacustrine_p			0.213	*					0.611	***
Harv1990-2000							0.534	*	0.426	*
Harv2000-2009									0.405	*
Morainal_p							0.560	*		
NNHTUR							-0.523	*	0.344	*
NNOTUR	0.208	*								
NNTKUR			0.245	*	0.278	*	0.526	*	0.485	**
pH	-0.250	**								
PPUT			0.309	**			0.608	*		
SECCHI	-0.262	**	-0.292	**	-0.377	**	-0.717	**	-0.447	**
SSO4UR	-0.295	**							-0.407	*
WBDY_LAKE_RATIO	-0.294	**			-0.273	*				
Wetlandi_p							0.551	*		

Level of significance: * if $p \leq 0.05$, ** if $p \leq 0.01$, *** if $p \leq 0.001$



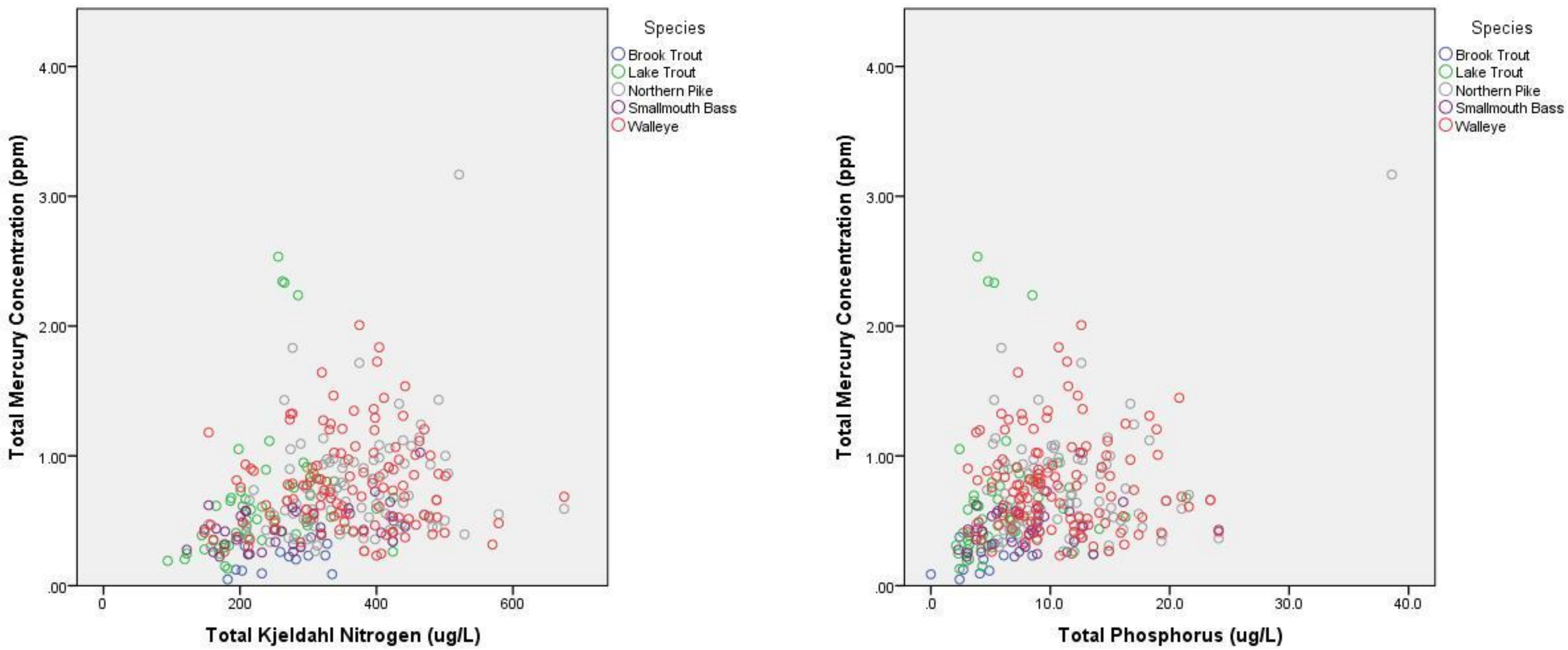
A. DOC concentration (mg/L) **B. Secchi Depth (m)**

Figure 1.10. The relationship between fish THg concentrations and: A) DOC concentrations in lake water, and B) Secchi depth.



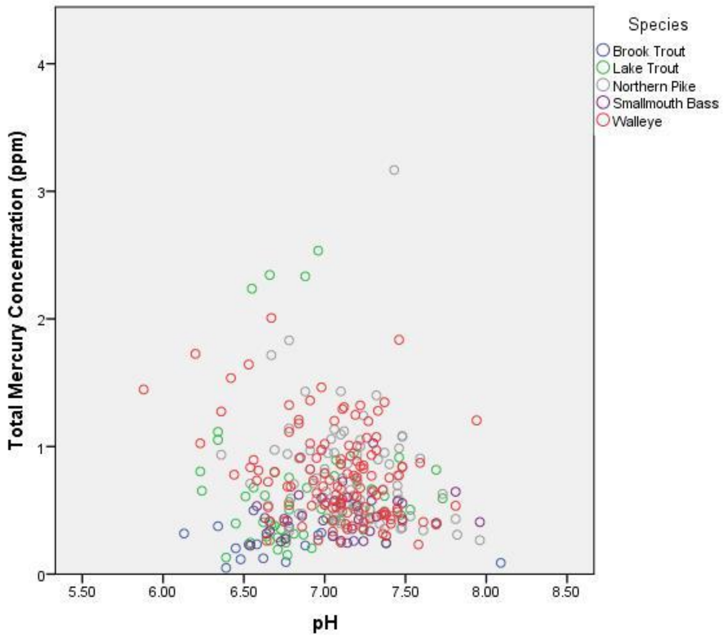
A. Wetlands **B. Forest Harvesting**

Figure 1.11. Relationship between dissolved organic carbon and A) percentage of wetlands in waterbody catchment (r^2=0.116, p<0.001), and B) percentage of waterbody catchment disturbed by harvesting disturbance during the last decade (r^2= 0.014, p=0.125).

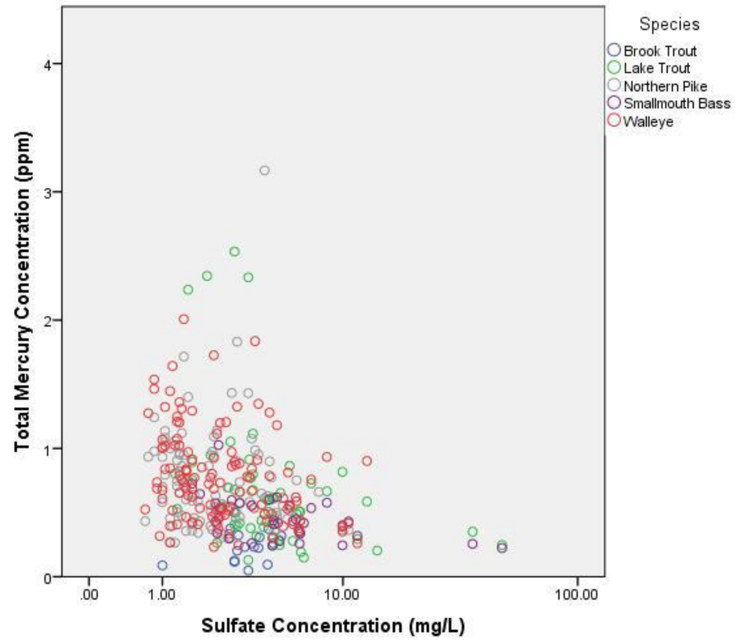


A. Total Kjeldahl Nitrogen **B. Total Phosphorus**

Figure 1.12. The relationship between fish THg concentrations and: A) total Kjeldahl nitrogen, and B) total phosphorus.



A. pH



B. Sulfate

Figure 1.13. The relationship between fish THg concentrations and: A) pH, and B) sulfate concentrations.

Forest Harvesting and Natural Disturbance Associations with Fish Mercury Concentrations

For all species, the THg concentration in fish from lakes with waterbody catchment areas affected by forest harvesting or natural disturbance fell within the observed range of THg concentrations in fish from lakes with no catchment disturbance (Figs. 1.14-1.17). Fish THg concentrations were not significantly related to percent area of the catchment disturbed by forest harvesting between 2000 and 2009 or to disturbance occurring between 1990 and 2009 (Figure 1.14: $F_{(1, 334)} = 1.683$, $p= 0.195$, and Figure 1.15: $F_{(1, 334)} = 2.257$, $p= 0.134$ respectively). Similarly, fish THg concentrations were not related to the percentage of the waterbody catchment area disturbed by natural disturbance events during the 2000-2009 or 1990-2009 time periods (Figure 1.16: $F_{(1, 335)} = 0.022$, $p= 0.881$, and Figure 1.17: $F_{(1, 335)} = 0.463$, $p= 0.497$ respectively).

The percentage of waterbody catchment area disturbed by harvesting and natural disturbance was highly variable among the different lakes (Table 1.4). Harvesting disturbance occurred within the waterbody catchment area of 81 walleye lakes (76% of sites), 65 northern pike lakes (67% of sites), 35 lake trout lakes (65% of sites), 5 brook trout lakes (33% of sites) and 21 smallmouth bass lakes (60% of sites). Natural disturbance occurring between 2000-2009 ranged from 0% to 100% of the waterbody catchment area for all sites, however the majority of lakes had very little natural disturbance. The waterbody catchment area of 83 walleye lakes (78% of sites), 75 northern pike lakes (77% of sites), 49 lake trout lakes (91% of sites), 15 brook trout lakes (100% of sites), and 28 smallmouth bass lakes (80% of sites) had no natural disturbance.

Table 1.4. Percentage of the waterbody catchment area disturbed by forest harvesting and natural disturbance events during the years 2000-2009 for groups of lakes organized by species presence.

Lakes grouped by species occurrence (number of lakes)	Range of Harvesting Disturbance (% area)	Average Harvesting Dist. mean (± SD) (% area)	Range of Natural Disturbance (% area)	Average Natural Dist. mean (± SD) (% area)
Walleye (n=121)	0 - 22.5	3.2 (± 4.7)	0 - 97.4	3.1 (± 12.8)
Northern Pike (n=106)	0 - 22.5	3.6 (± 5.2)	0 - 100	4.3 (± 16.4)
Lake Trout (n=60)	0- 39.9	4.3 (± 7.9)	0 - 97.4	2.8 (± 13.9)
Brook Trout (n=18)	0 - 30.0	3.5 (± 8.5)	0	0
Smallmouth Bass (n=37)	0 – 13.89	1.8 (± 3.0)	0 - 46.2	1.8 (± 7.9)

SD=standard deviation

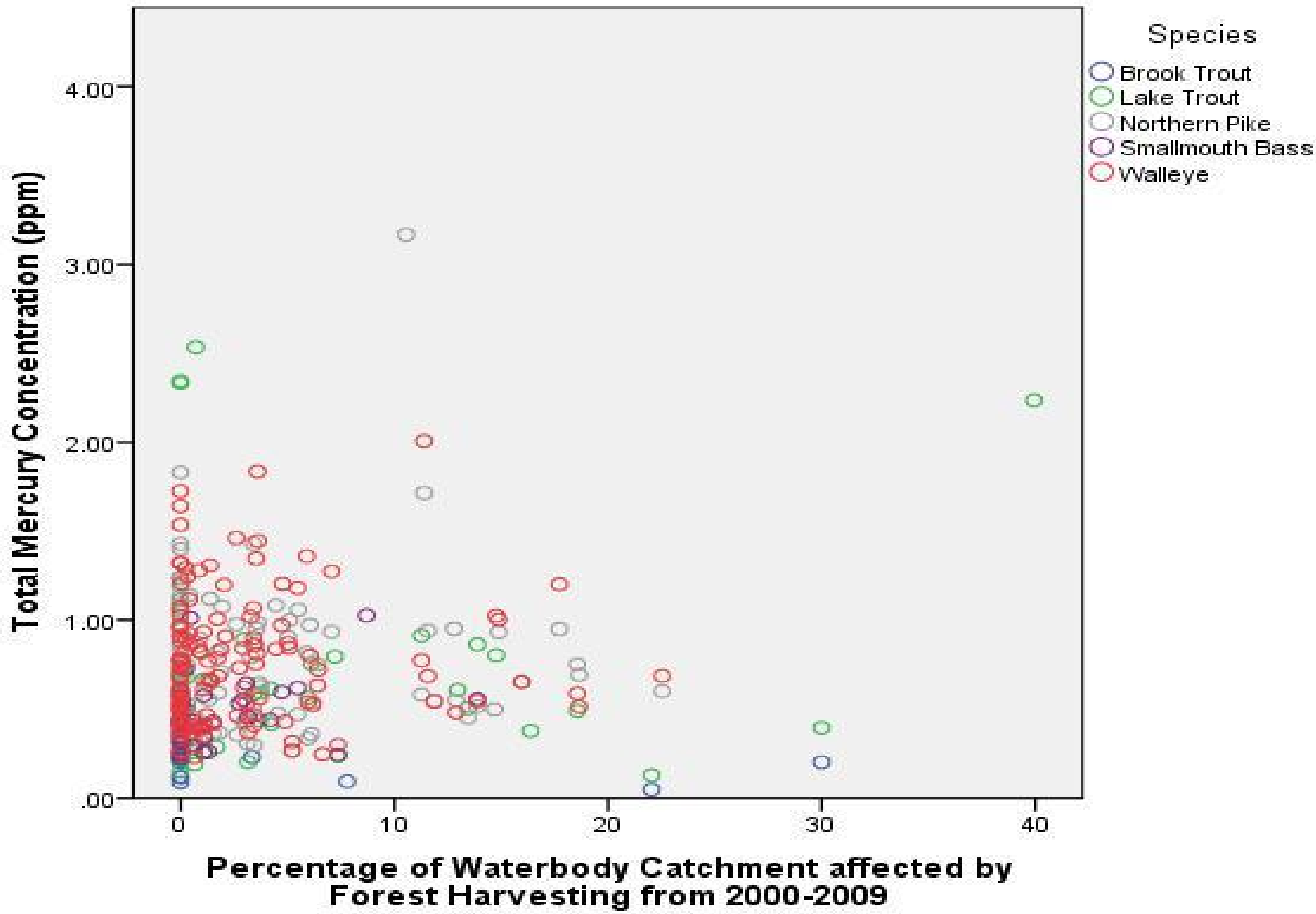


Figure 1.14. Relationship between harvesting disturbance from 2000-2009 and standardized THg concentration according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

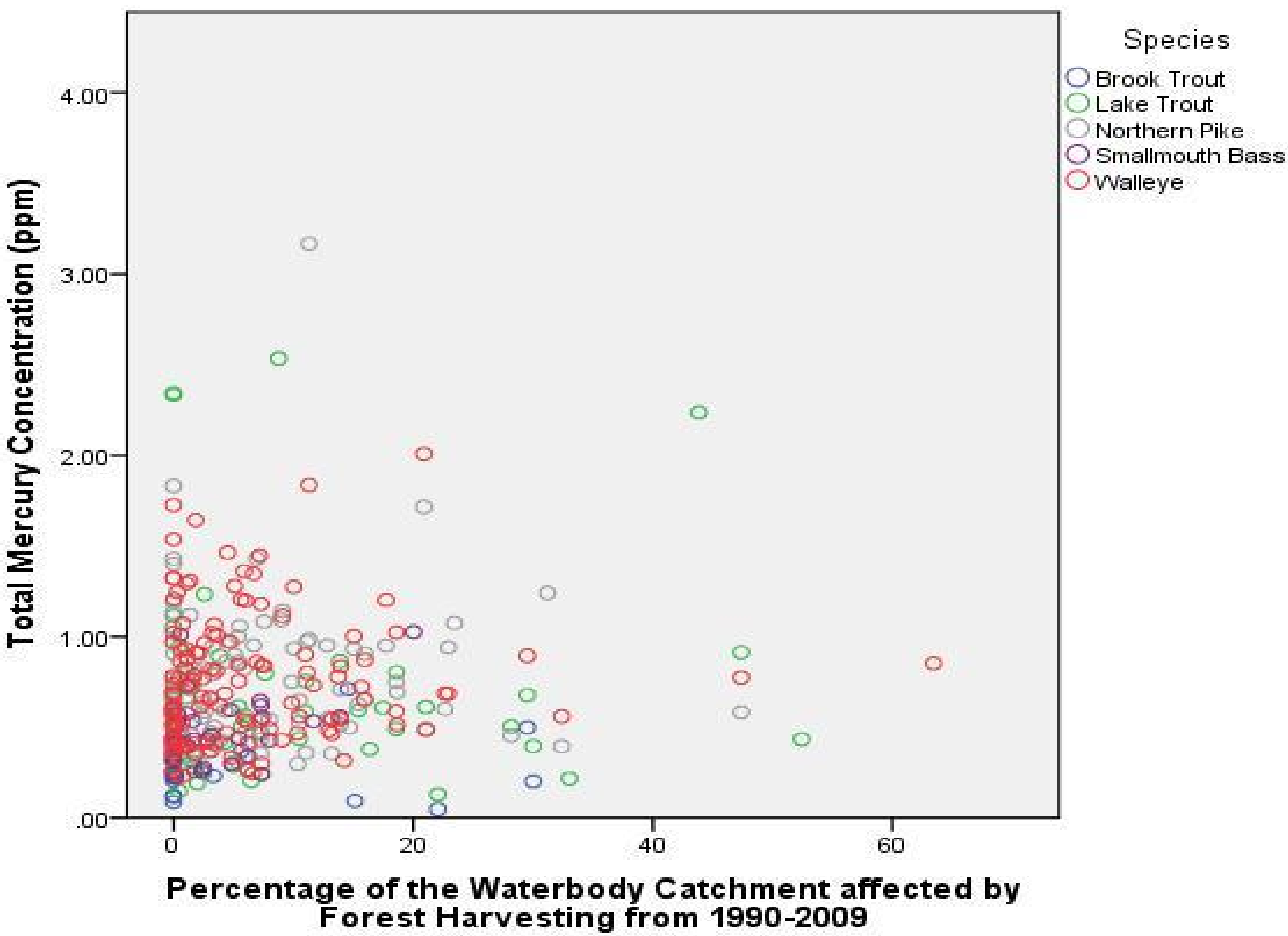


Figure 1.15. Relationship between harvesting disturbance from 1990-2009 and standardized THg concentration according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

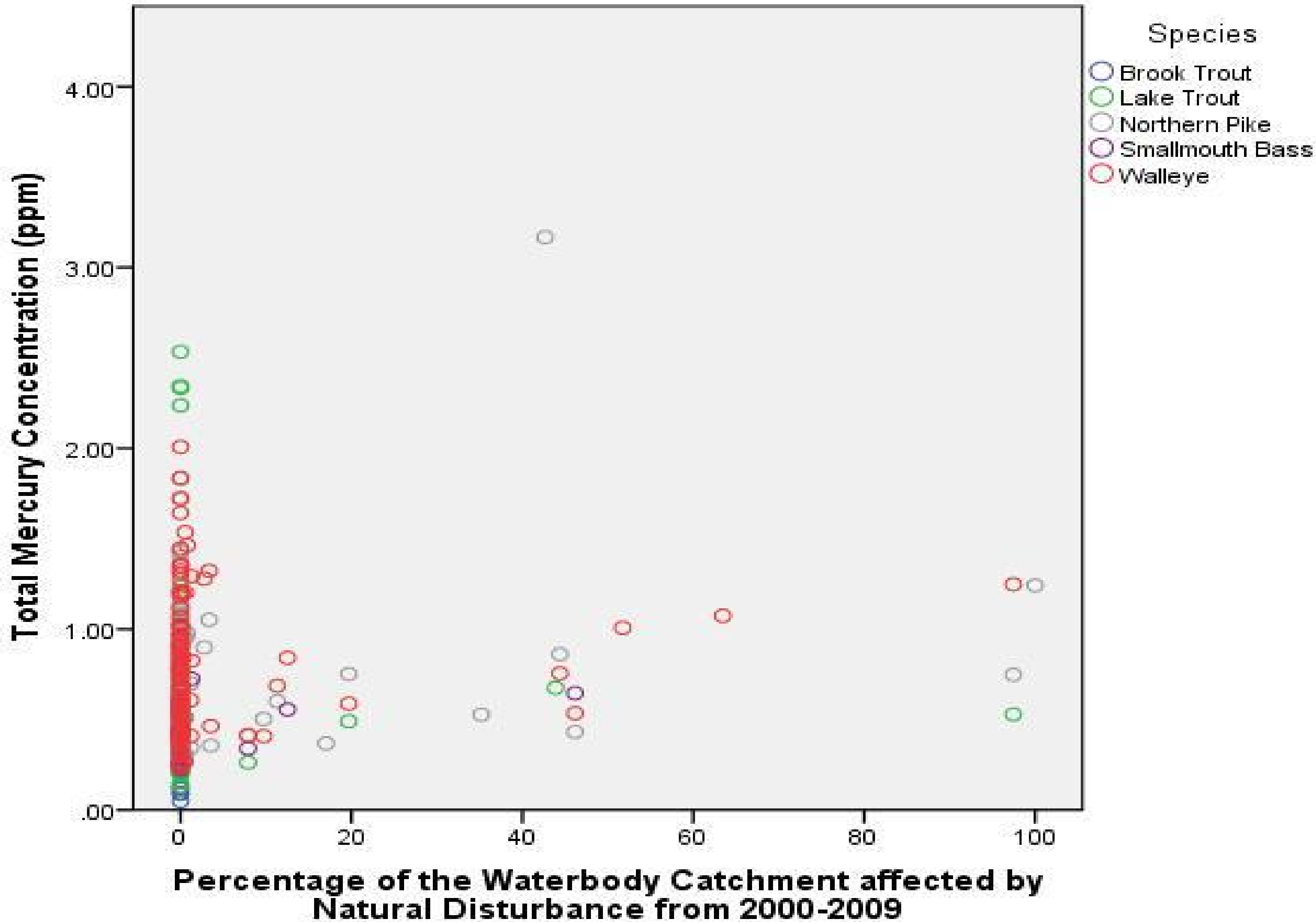


Figure 1.16. Relationship between natural disturbance between 2000-2009 and standardized THg concentration according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

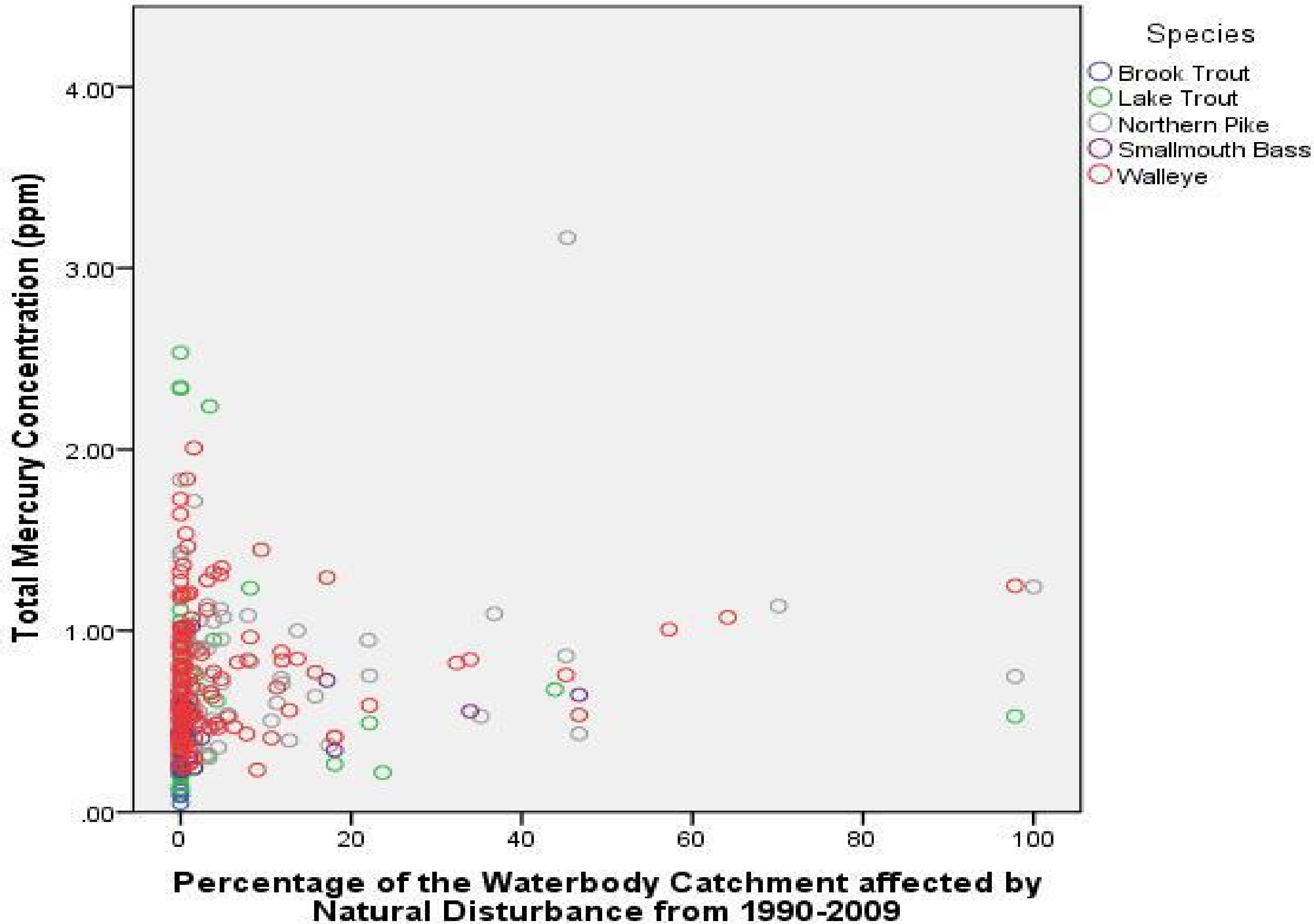


Figure 1.17. Relationship between natural disturbance from 1990-2009 and standardized THg concentration according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

Relationships between Wetlands and Fish Mercury Concentrations

The percentage of waterbody catchment area with wetlands was variable among the different lakes (Table 1.5). The THg concentration for all species of fish from lakes with waterbody catchment areas containing wetlands was also within the observed range of THg concentrations in fish from lakes with very few to no associated wetlands (Figure 1.18). Fish THg concentrations were not significantly related to percent area of the waterbody catchment having wetlands (Figure 1.18: $F_{(1, 334)} = 2.380$, $p = 0.124$).

Table 1.5. Percentage of Wetlands within the Waterbody Catchment for groups of lakes organized by species presence.

Lakes grouped by species occurrence (number of lakes)	Range of Wetland Area within Waterbody Catchment (% of total area)	Average Wetland Area within Waterbody Catchment (± SD) (% of total area)
Walleye (n=121)	0 – 18.1	2.8 (± 2.9)
Northern Pike (n=106)	0 – 18.5	2.71 (± 3.2)
Lake Trout (n=60)	0- 9.9	1.54 (± 2.2)
Brook Trout (n=18)	0 – 9.5	1.6 (± 2.4)
Smallmouth Bass (n=37)	0 – 5.1	1.6 (± 1.4)

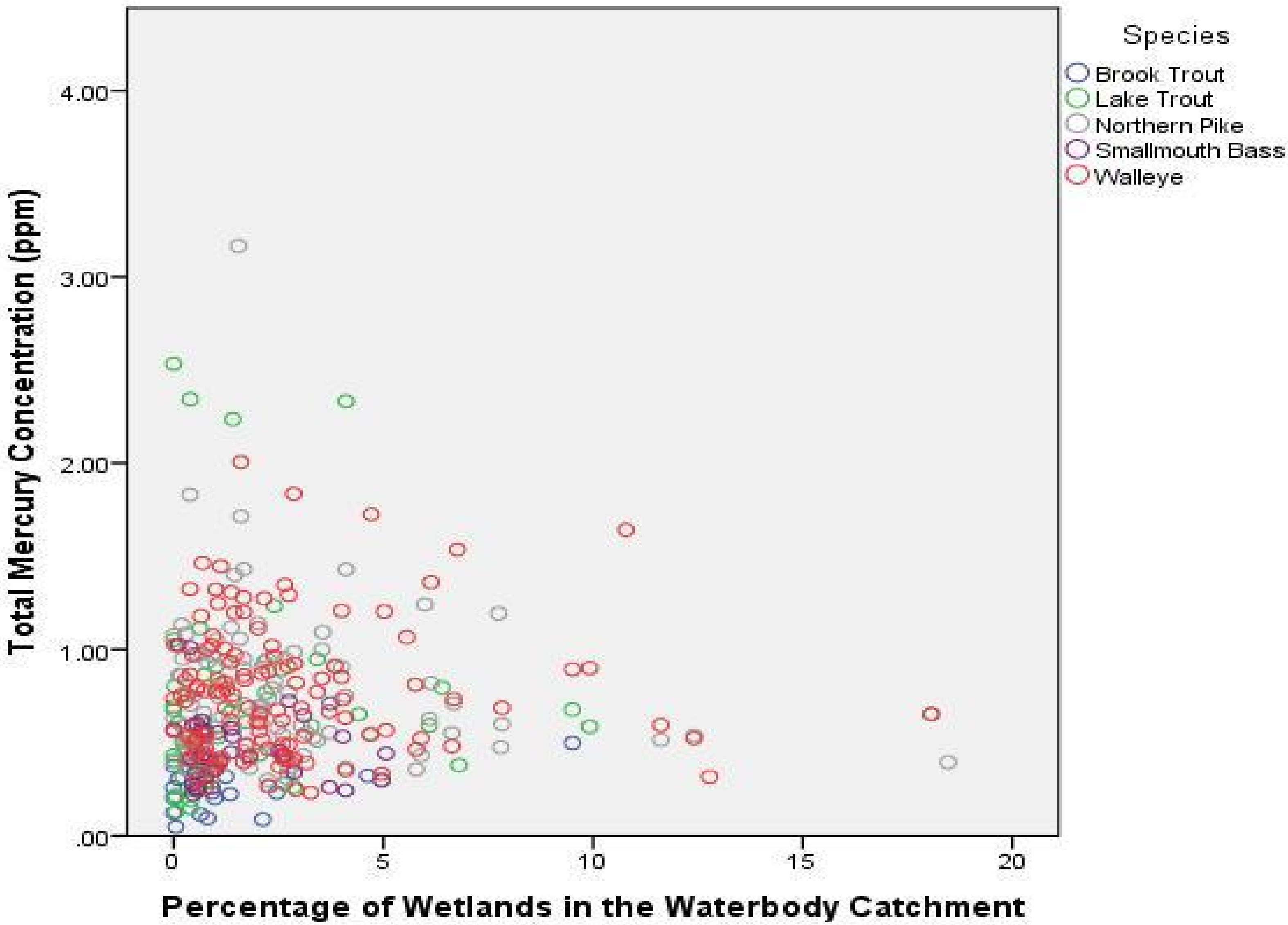


Figure 1.18. Relationship between wetland percentage within waterbody catchment area and standardized THg concentration according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

Waterbody Catchment Associations with Categories of Contamination

Waterbody catchment characteristics were not significantly different amongst the three groups of lakes classified by the standard mercury concentration in walleye. The percentage of waterbody catchment disturbed by forest harvesting between 2000-2009 or 1990-2009 did not differ significantly among the mercury contamination categories of walleye (Figure 1.19: 2000-2009 ($H=0.936$, $df=2$, $p = 0.626$) or Figure 1.20: 1990-2009 ($H=0.765$, $df=2$, $p = 0.730$)). Similarly, the percentage of natural disturbance occurring in waterbody catchments between 2000-2009 or 1990-2009 did not differ significantly among mercury contamination categories of walleye (2000-2009 ($H=5.540$, $df=2$, $p = 0.063$) Figure 1.21 or 1990-2009 ($H=1.237$, $df=2$, $p = 0.539$) Figure 1.22). The percentage of wetland area of the total waterbody catchment was not significantly different between the categories of fish mercury contamination ($H=0.186$, $df=2$, $p = 0.911$; Figure 1.23). The same analysis conducted on lakes containing northern pike, lake trout, and smallmouth bass showed similar trends with no significant difference among the categories of contamination.

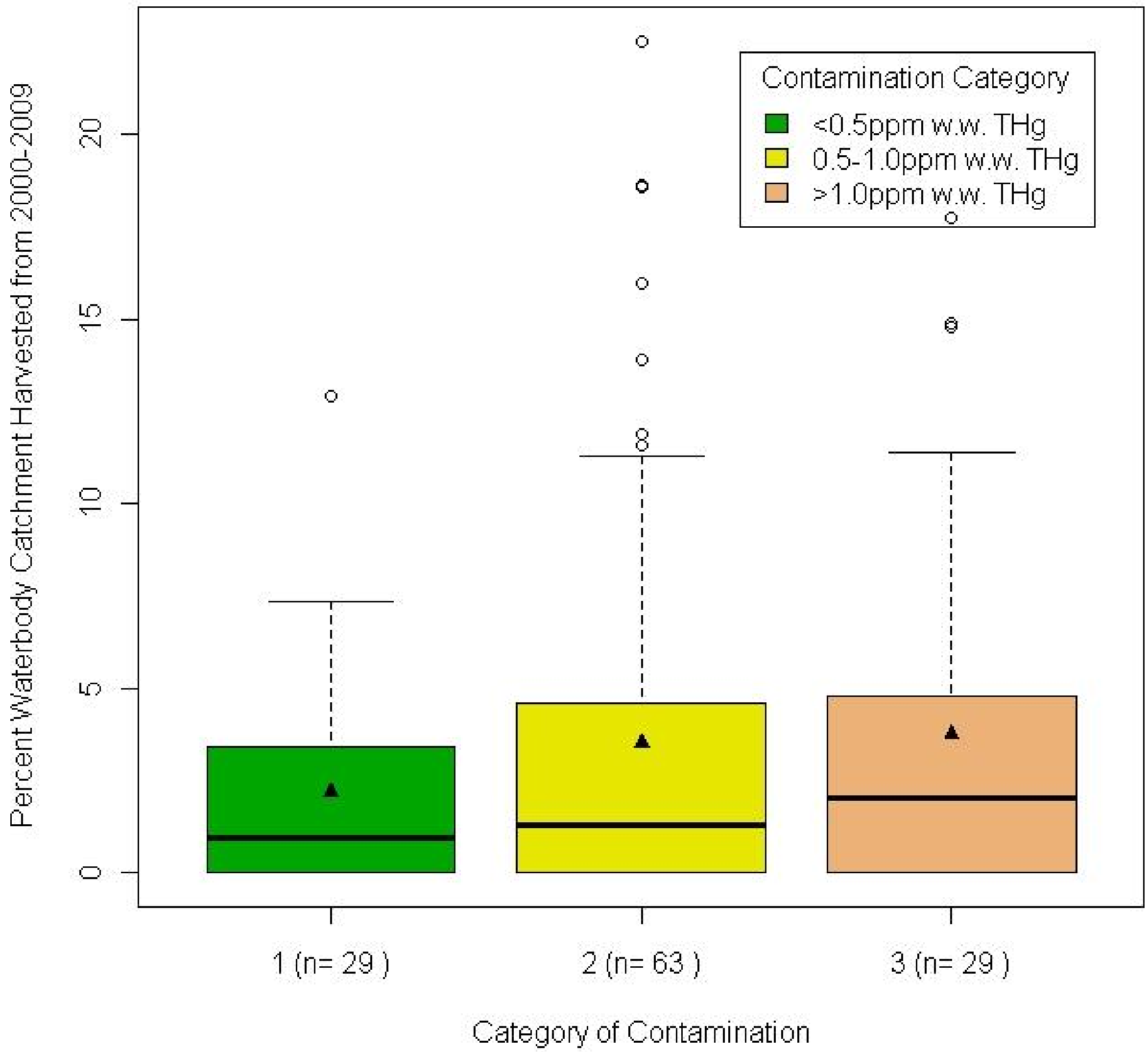


Figure 1.19. The comparison between walleye THg contamination categories and percentage of forest harvesting disturbance within the waterbody catchment from the years of 2000-2009. The horizontal black line represents the median, the boxes indicate the inter-quartile range (IQR) and the whiskers represent 1.5 times the IQR from the first and third quartile, hollow circles represent data outside of the (Q1-1.5* IQR, Q3+1.5*IQR). The mean is represented by the black triangle and the number of samples (n) represents the number of lakes in that category.

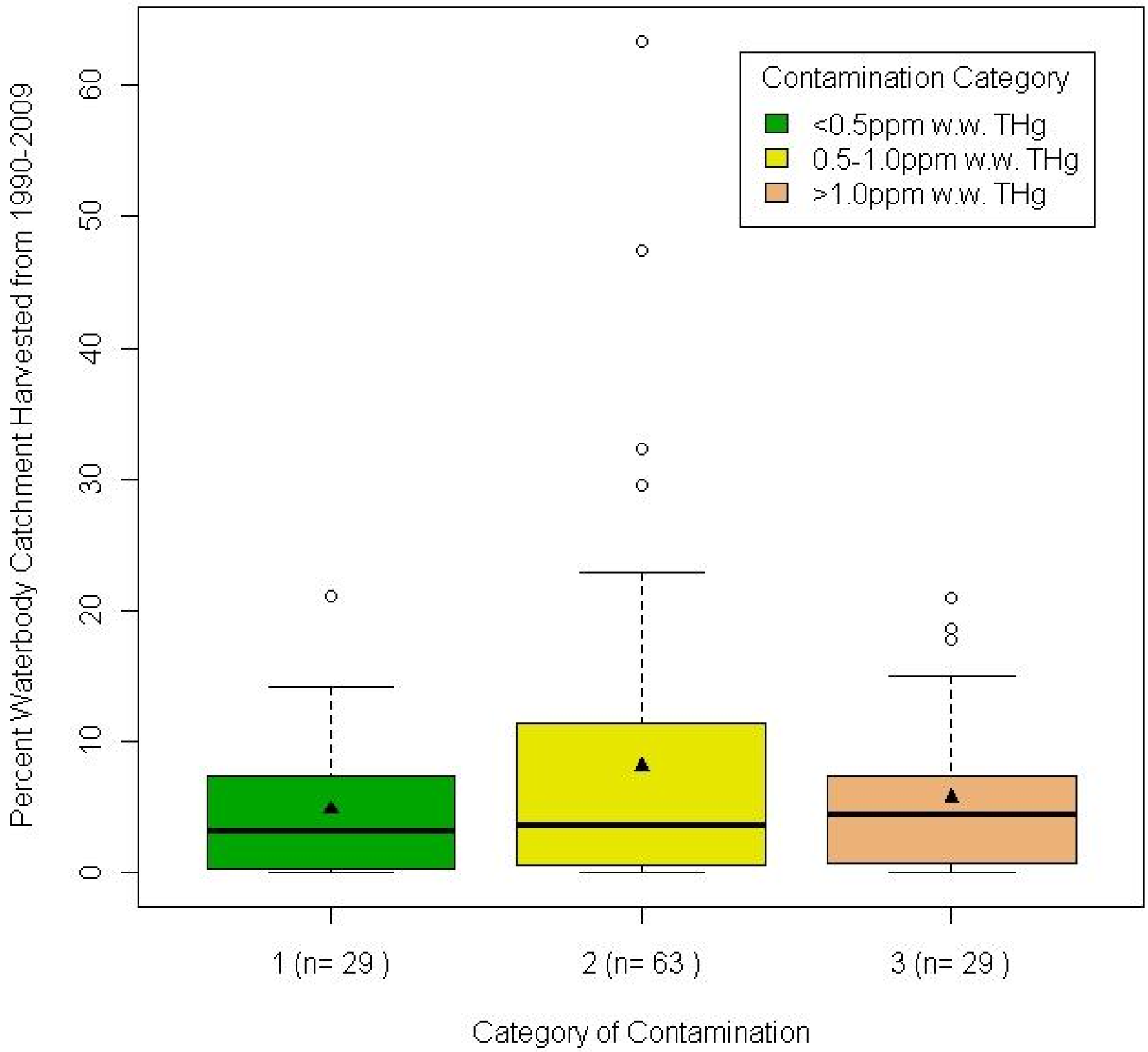


Figure 1.20. The comparison between walleye THg contamination categories and percentage of forest harvesting disturbance within the waterbody catchment from the years of 1990-2009. Details of plot are the same as the earlier Figure 1.19.

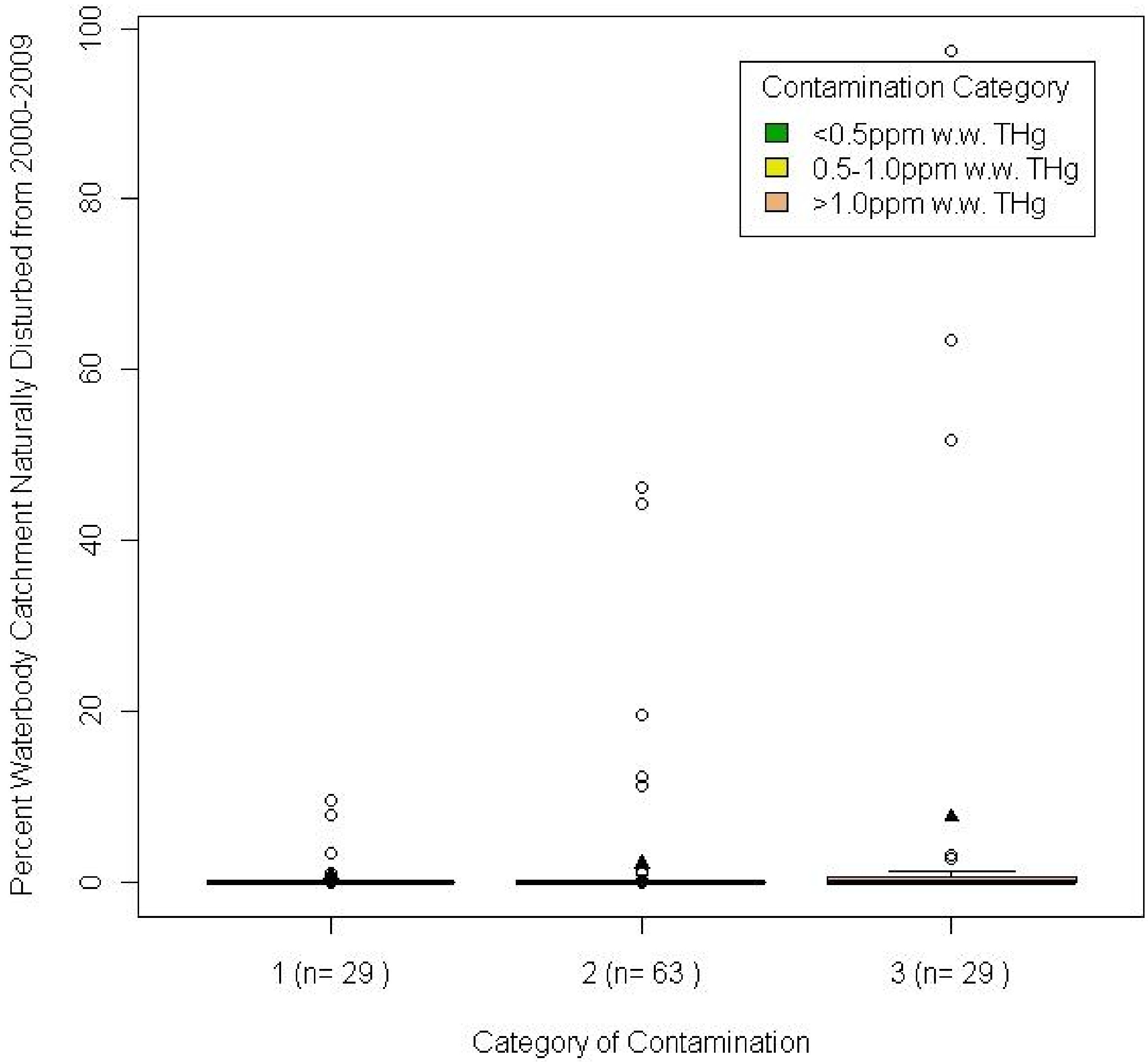


Figure 1.21. The comparison between walleye THg contamination categories and percentage of natural disturbance within the waterbody catchment during the years 2000-2009. Details of plot are the same as the earlier Figure 1.19.

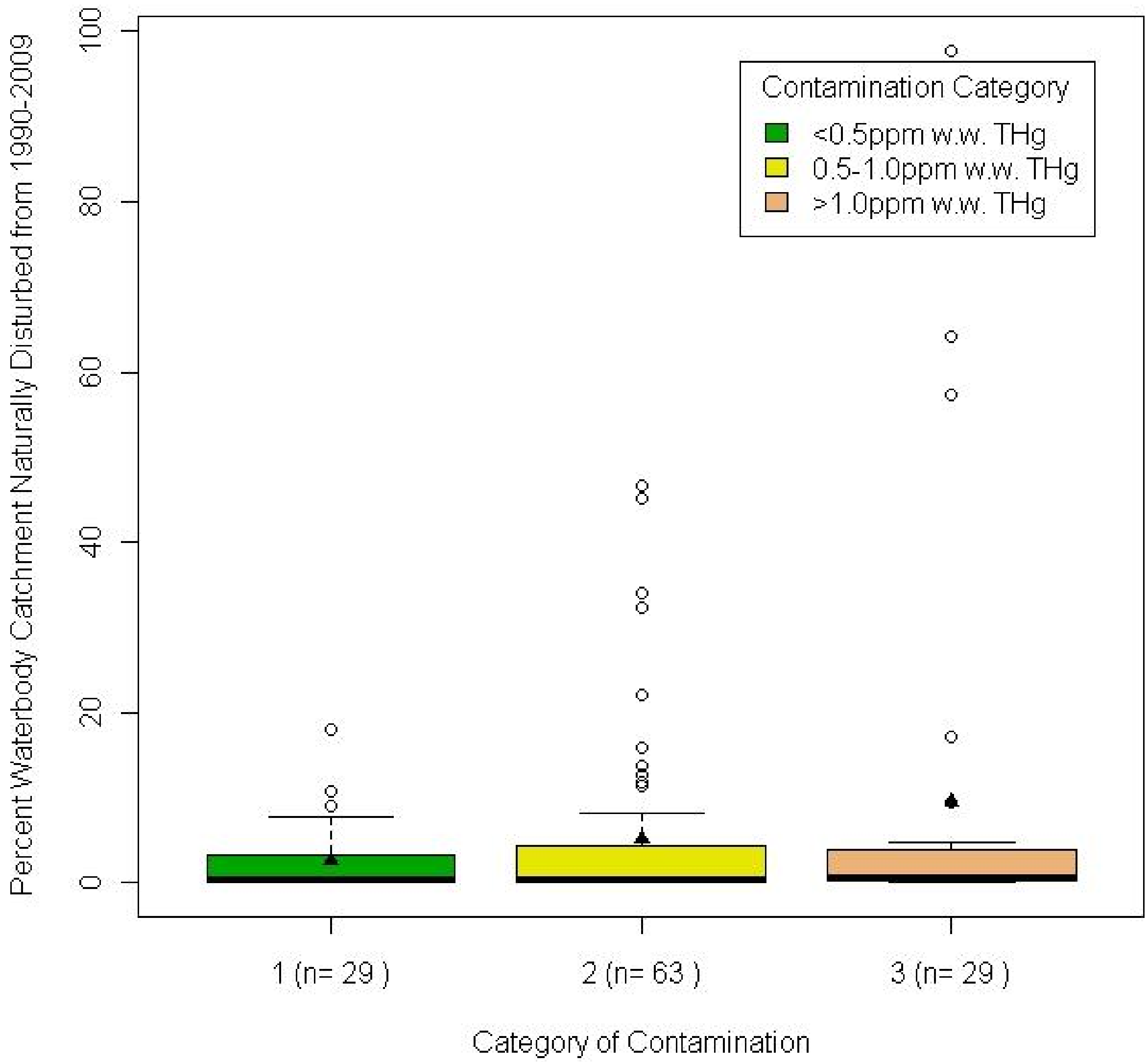


Figure 1.22. The comparison between walleye THg contamination categories and percentage of natural disturbance within the waterbody catchment during the years 1990-2009. Details of plot are the same as the earlier Figure 1.19.

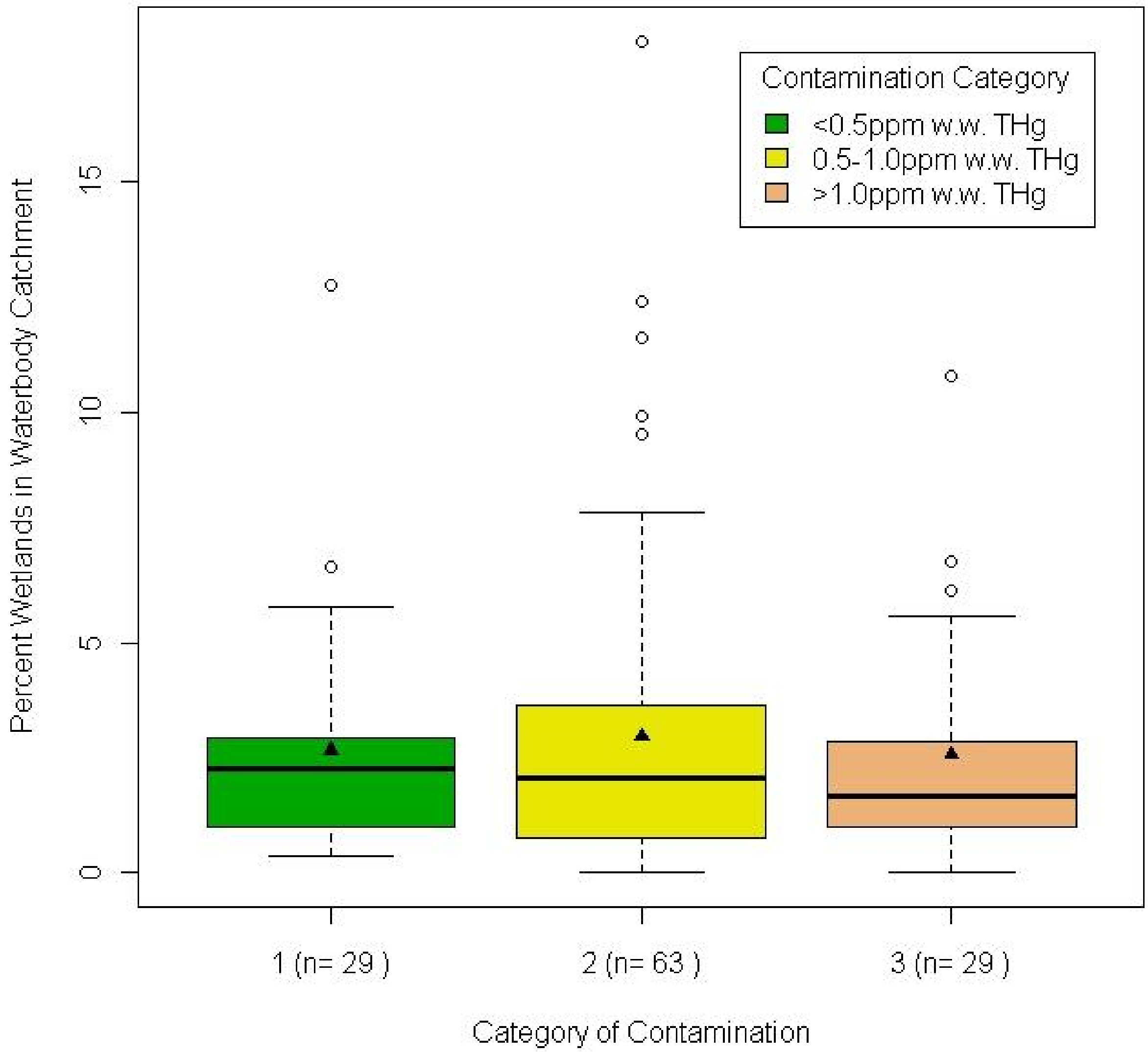


Figure 1.23. The comparison between walleye THg contamination categories and percentage of wetland area from within the waterbody catchment. Details of plot are the same as the earlier Figure 1.19.

Lake Size and Lake to Waterbody Catchment Area Ratio Influences

Overall, fish THg concentrations were significantly related the ratio of lake area within the waterbody catchment (Figure 1.24: $F_{(1, 332)}$ =8.074, p=0.005). The relationship between fish THg concentration and the lake to waterbody catchment ratio does not differ significantly as a function of species (ANCOVA analysis of the homogeneity of regression (intercept: $F_{(4, 332)}$ = 1.203 , p= 0.309; slopes: $F_{(4, 332)}$ = 0.314 , p= 0.868)).

Fish THg concentrations tended to be negatively associated with lake area but the relationship was variable among species and was not significant (Figure 1.25: $F_{(1, 332)}$ =0.088 , p= 0.766). When species were examined separately, all species showed a negative relationship between THg levels and lakes size, analysed in size classes defined by surface area. Northern pike (H=9.671, df=4, p=0.046) and lake trout (H=10.778, df=4, p= 0.029) had significantly higher standardized THg concentrations in small (size bin 1: < 100 ha) lakes (Dunn's pairwise comparison, p<0.05, Figure 1.27 and Figure 1.28 respectively). Walleye (H=3.034, df=4, p=0.552), brook trout (H=0.009, df=4, p=0.925) and smallmouth bass (H=3.949, df=3, p=0.267) also showed a negative relationship with lake size class although THg differences among the size classes were not significant as indicated by the non-parametric Kruskal-Wallis test (Figure 1.26, Figure 1.29, and Figure 1.30 respectively).

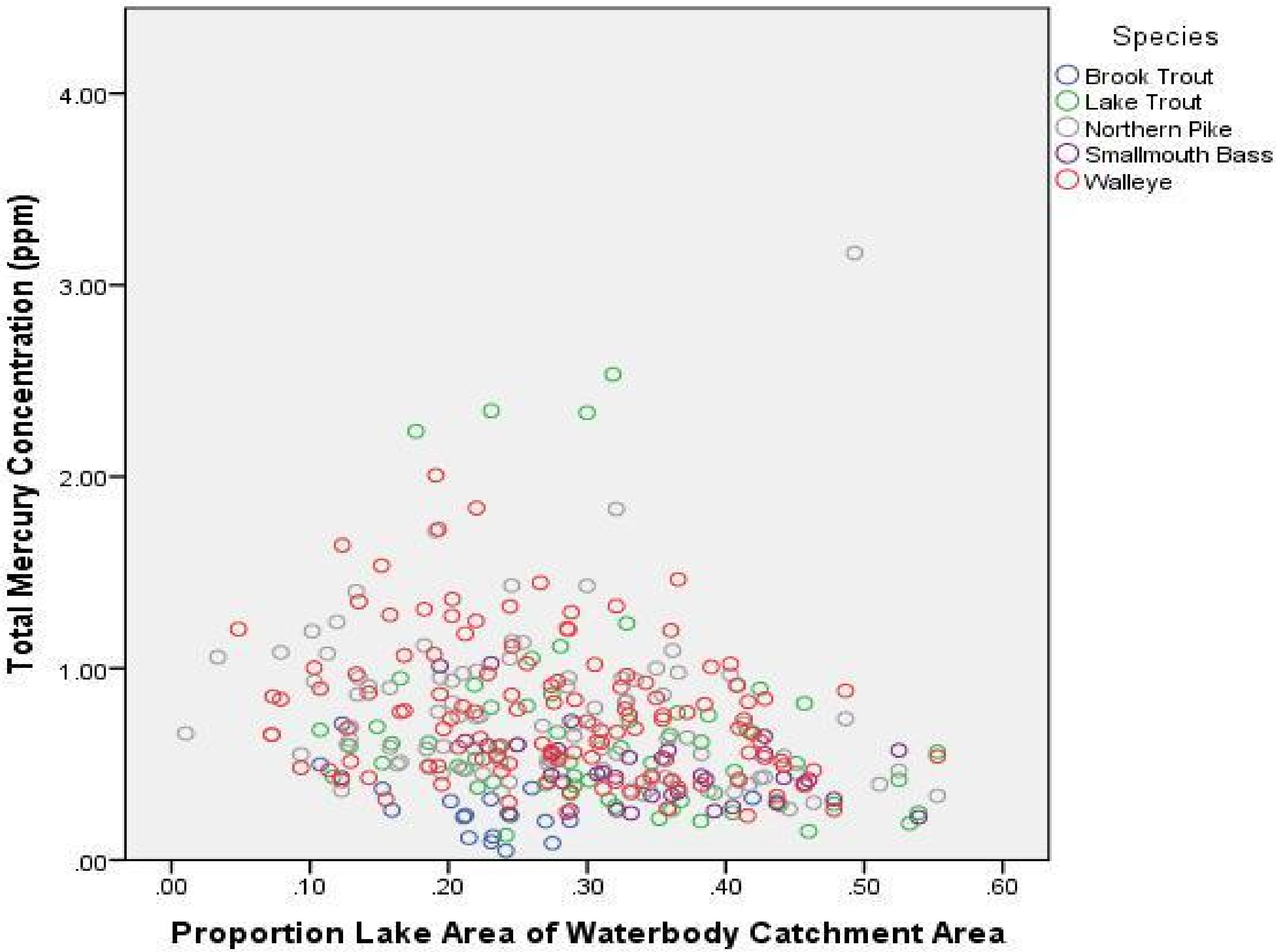


Figure 1.24. The relationship between the lake to waterbody catchment surface area ratio and THg concentrations (ppm w.w.) according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

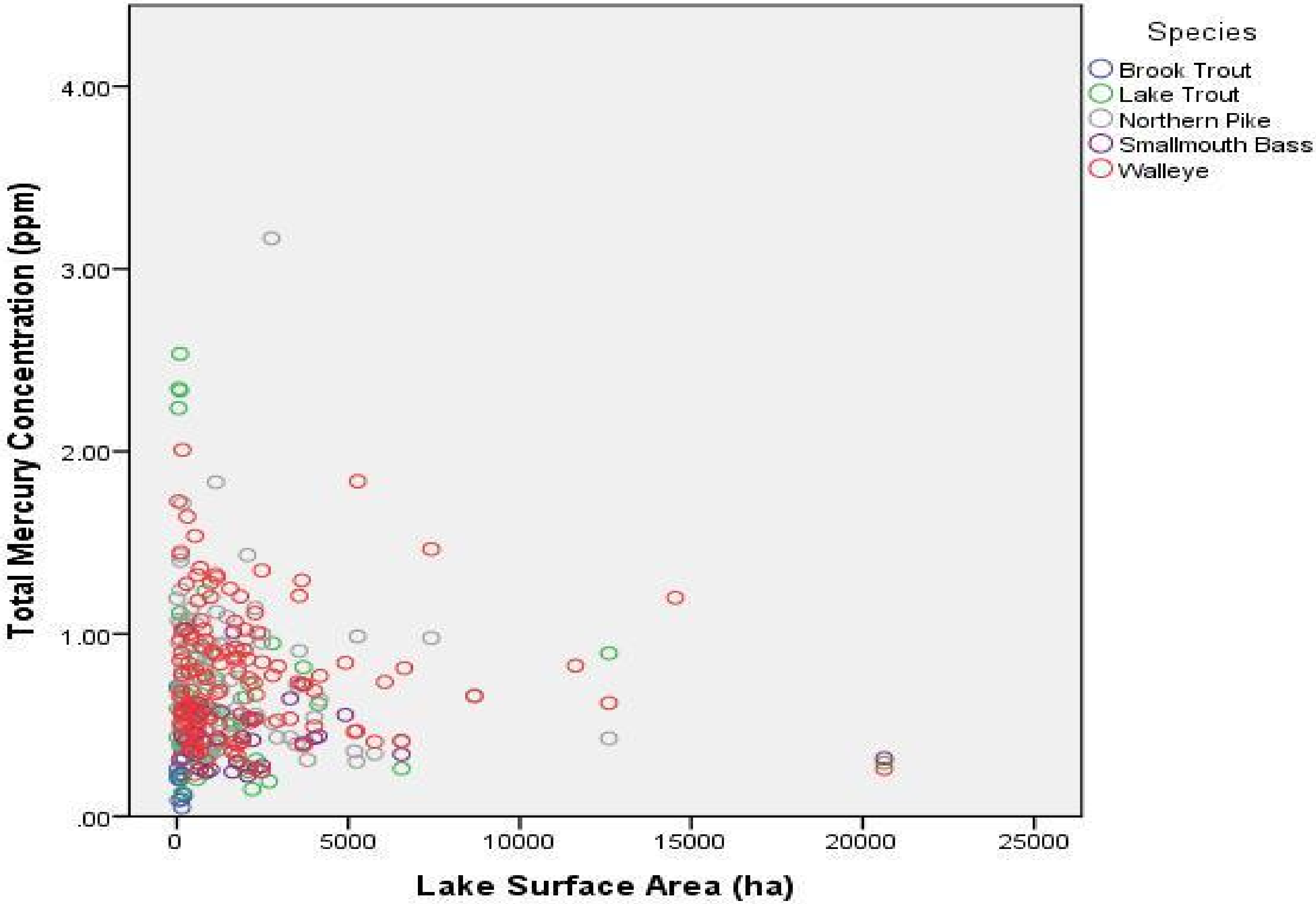


Figure 1.25. The relationship between fish THg concentration (ppm w.w.) and lake area according to species (brook trout: n=18, lake trout: n=60, northern pike: n=106, smallmouth bass: n= 37, walleye: n=121).

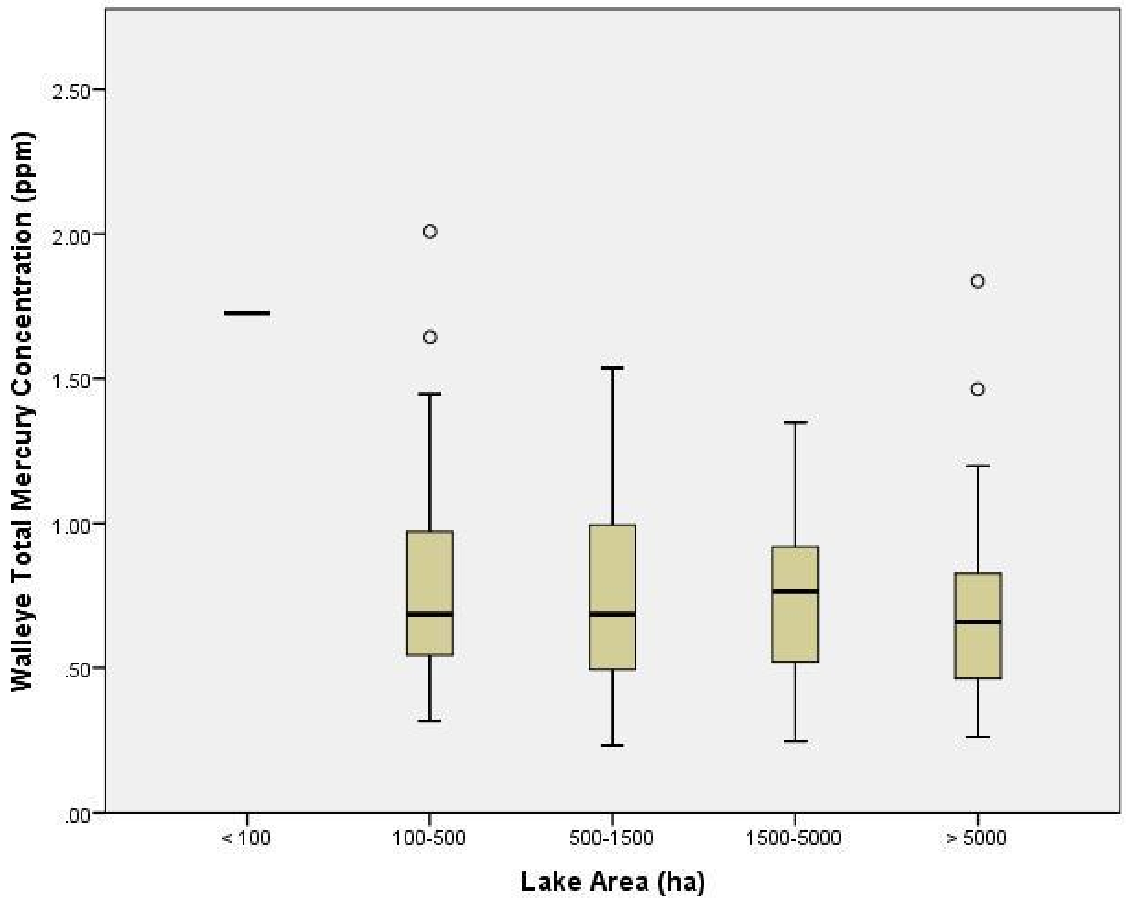


Figure 1.26. THg concentrations of walleye grouped according to lake surface area size bin category (Bin 1 [<100 ha]: n=1, Bin 2 [100-500 ha]: n=29, Bin 3 [500-1500 ha]: n=39, Bin 4 [1500-5000 ha]: n=40, Bin 5 [>5000 ha]: n=13). The box indicates the inter-quartile range (IQR), the dark horizontal line indicates the median and the "whiskers" extending above and below the box indicate 1.5 times the IQR. Outliers are represented as circles and extreme values (greater than 3 times the IQR) are indicated by asterisks.

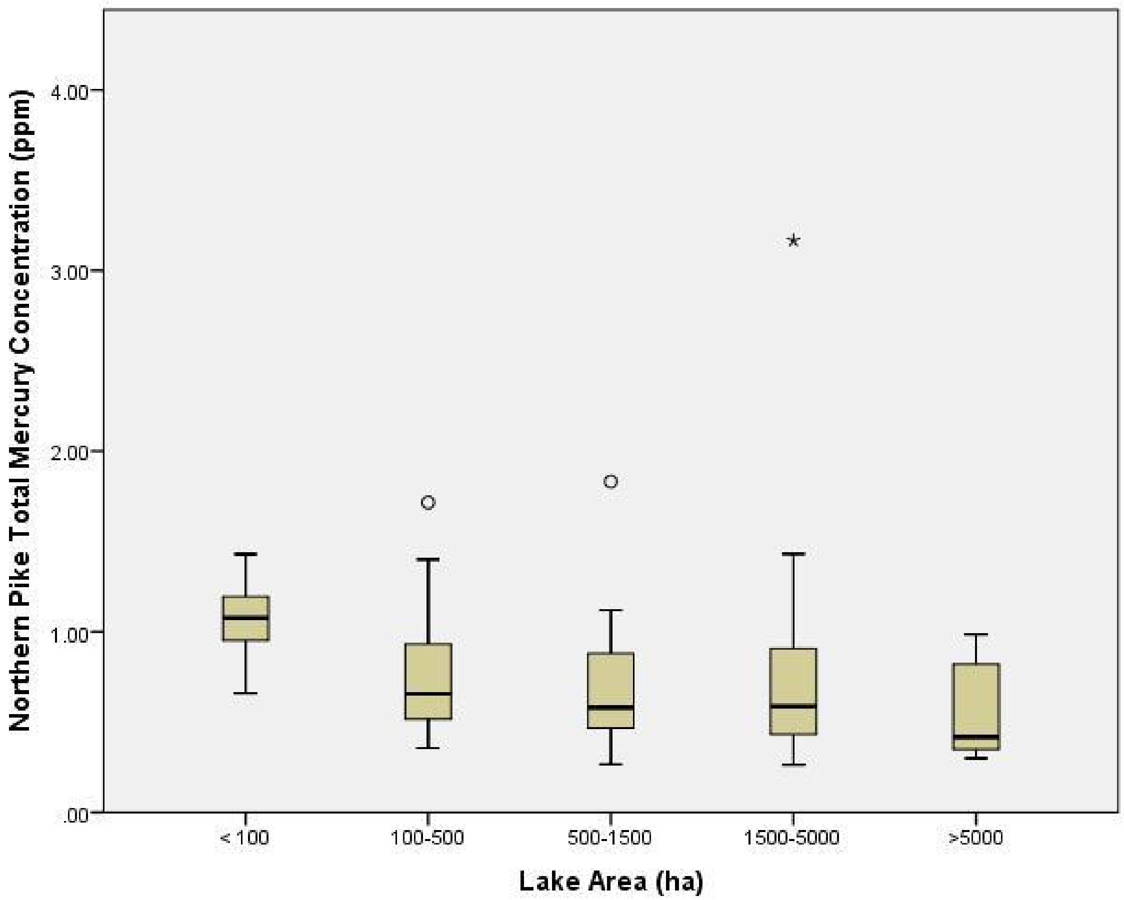


Figure 1.27. THg concentrations of northern pike grouped according to lake surface area size bin category (Bin 1 [<100 ha]: n=5, Bin 2 [100-500 ha]: n=29, Bin 3 [500-1500 ha]: n=35, Bin 4 [1500-5000 ha]: n=29, Bin 5 [>5000 ha]: n=8). Details of the plot are the same as the previous Figure 1.26.

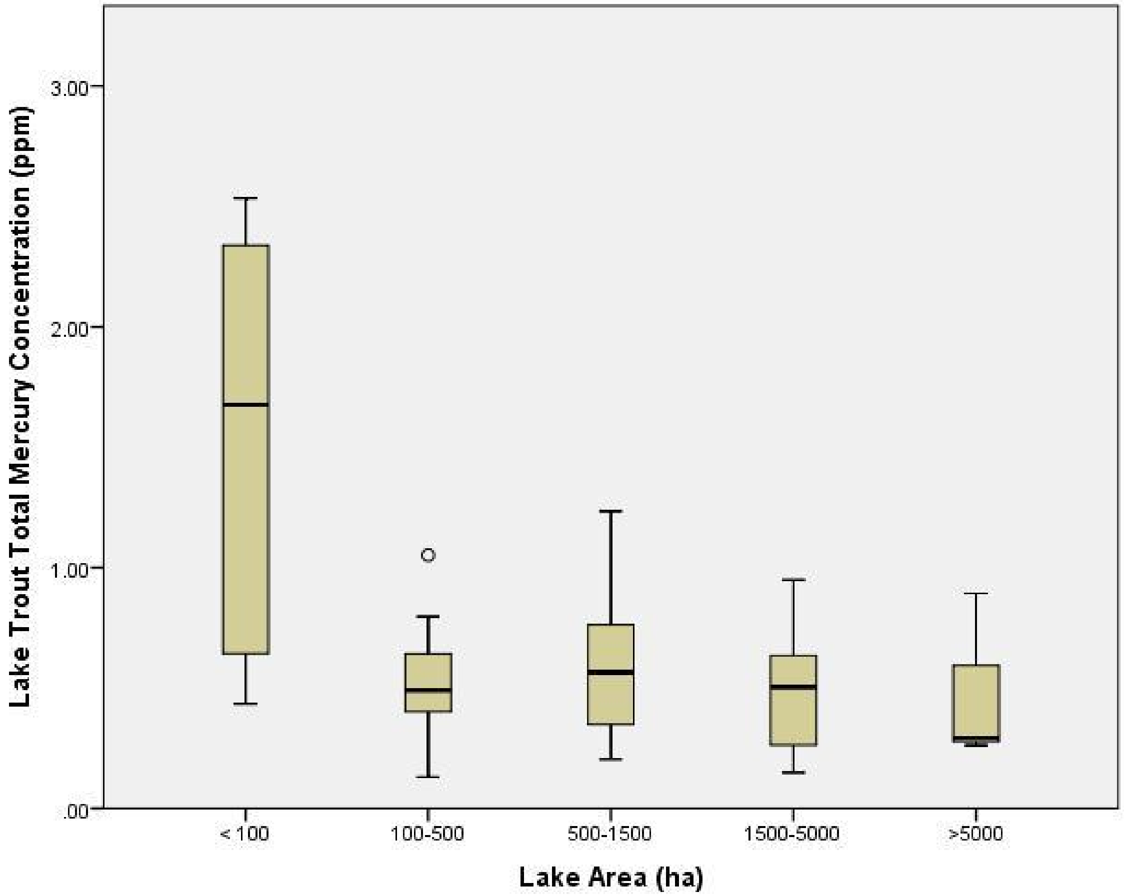


Figure 1.28. THg concentrations of lake trout grouped according to lake surface area size bin category (Bin 1 [<100 ha]: n=8, Bin 2 [100-500 ha]: n=15, Bin 3 [500-1500 ha]: n=19, Bin 4 [1500-5000 ha]: n=15, Bin 5 [>5000 ha]: n=3). Details of the plot are the same as the previous Figure 1.26.

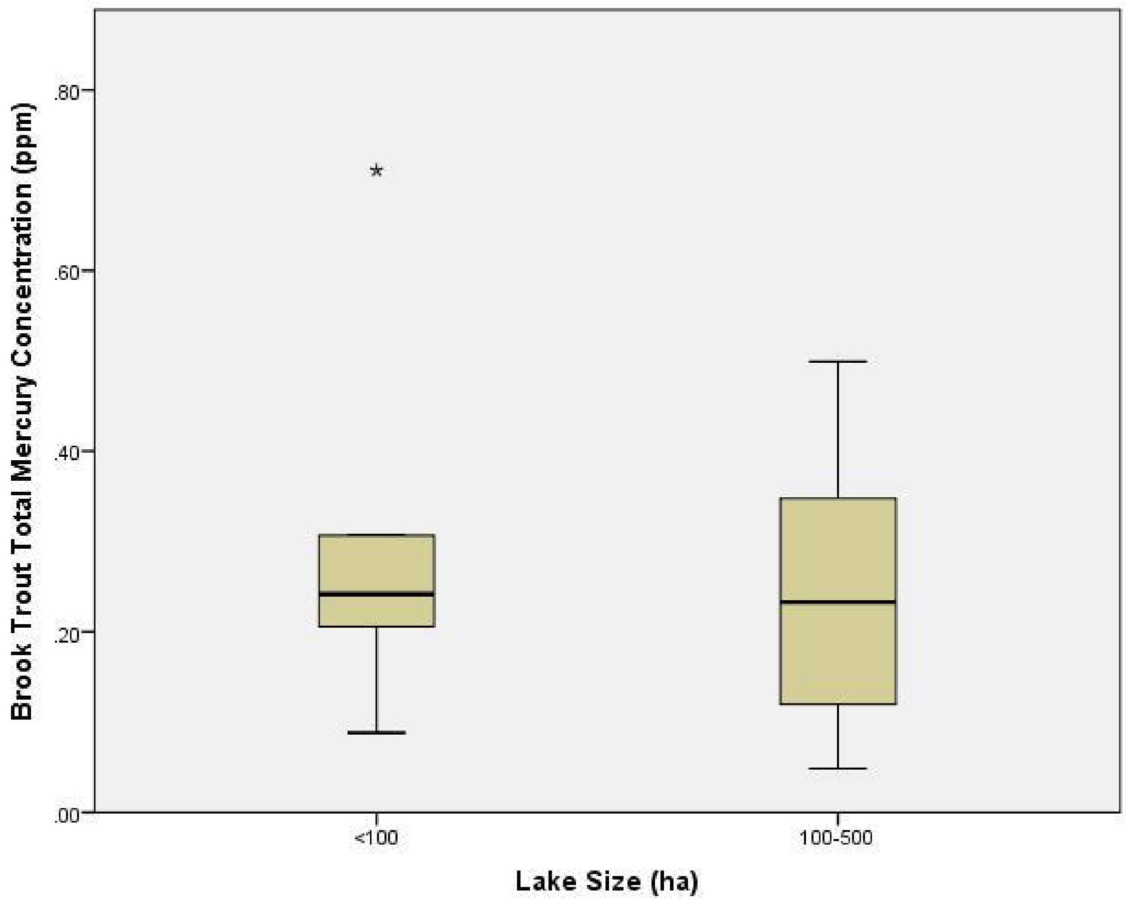


Figure 1.29. THg concentrations of brook trout grouped according to lake surface area size bin category (Bin 1 [<100 ha]: n=6, Bin 2 [100-500 ha]: n=12). Details of the plot are the same as the previous Figure 1.26.

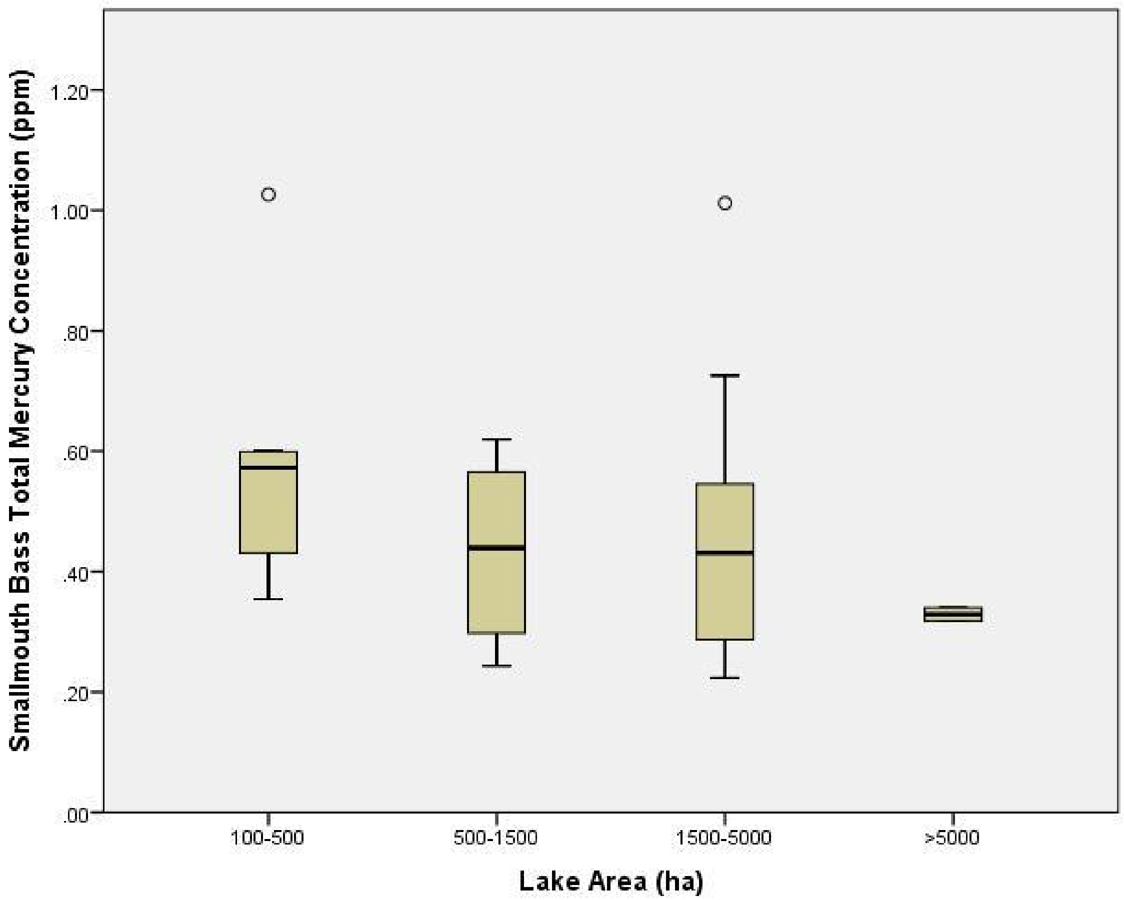


Figure 1.30. THg concentrations of smallmouth bass grouped according to lake surface area size bin category (Bin 2 [100-500 ha]: n=7, Bin 3 [500-1500 ha]: n=12, Bin 4 [1500-5000 ha]: n=16, Bin 5 [>5000 ha]: n=2). Details of the plot are the same as the previous Figure 1.26.

Discussion

For the majority of lakes in this study, mercury contamination in fish is a significant problem as the average total mercury concentration for a standard sized lake trout, northern pike, or walleye exceeded Health Canada's guideline of 0.5 ppm w.w. THg content in fish (Appendix D: Table D.2). All species of fish are highly variable in terms of THg concentrations among lakes which may differ by an order of magnitude or more between adjacent lakes (Figure 1.3). These results are comparable to the James Bay region of Quebec where length-standardized THg concentrations in fish varied by factors of 3-4 between neighbouring lakes (Schetagne and Verdon 1999). Similarly, populations of walleyes in lakes of four different regions of Quebec (Saint Lawrence Valley, Chibougamau, Abitibi, and Témiscamingue) standardized to length of 350 mm ranged from 0.17 to 0.79 ppm (Simoneau *et al.* 2005).

Overall, fish mercury concentrations for the different species were weakly correlated to the waterbody catchment and lake scale factors. The associations between THg concentrations of different fish species with waterbody catchment or lake scale environmental factors showed similar trends among a wide variety of lakes occurring in northern Ontario (Figs. 2.4-2.8). Although no apparent grouping of lakes existed based on their waterbody catchment or lake characteristics, the surficial geology of the waterbody catchment was important for distinguishing the differences among lakes which varied in terms of water chemistry and ranged from small darkly stained dystrophic lakes to larger clear oligotrophic lakes. Common to all species, the highest mercury concentrations in fish were generally found in small darkly stained lakes having recent forest harvesting disturbance and greater percentages of wetlands and organic material within the waterbody catchment. The density of roads (km/ha) was also associated with the percentage of forest harvesting disturbance as would be expected for many of the waterbody catchments where forest harvesting disturbance was the single anthropogenic disturbance present. Water chemistry parameters such as DOC and total phosphorus were associated with one another and positively associated with fish mercury concentrations.

The ordinations (Figure 1.4 and Figure 1.5) for lakes containing walleye or northern pike have high stress values (>0.2) which may lead to a misinterpretation of the results. Kruskal (1964 a, b) and later authors suggested that the stress values in NMDS be used as guidelines for interpreting ecological data. A stress value > 0.2 is considered poor and the ordination may not reliably summarize variability in the data set resulting in misleading interpretations. However, stress increases with the number of variables and the number of samples and a larger dataset will result in a higher stress value (Holland 2008). Higher stress values for the walleye and northern pike ordinations could be attributed to larger sample sizes (n=107 for walleye, and n=97 for northern pike). The northern pike and walleye datasets are largely the same as both species coexist in 70 lakes which contributes to the similar associations observed between watershed and lake scale characteristics among the ordinations of lakes containing walleye or northern pike.

The number of lakes available for analysis was also limited for some species. The number of lakes with smallmouth bass was relatively small and the ordination violates the rule for adequate sample size where the number of variables used in the ordination should be less than five times the number of sites (Tabachnik and Fidell 1989). Likewise, the number of brook trout lakes (n=15) is less than the number of variables (31 different waterbody and lake scale characteristics) used in the ordination, violating the rule for adequate sample size in ordination analysis. Therefore, there is a possibility that the interpretation of these ordinations may be misleading and the small sample size may misrepresent the associations between sites and variables. With a small sample size, the results of the ordination may be strongly influenced by outliers and fail to represent the reality of an ecological relationship. However, a smaller sample size may be adequate if there are strong reliable correlations between a distinct set of factors (Tabachnik and Fidell 1996). Although the small sample size could lead to misinterpretation of either the smallmouth bass or brook trout ordination, the associations between watershed and lake scale variables with fish THg concentrations are generally consistent with the weak relationships found for walleye and lake trout.

Water chemistry and nutrients, variables which are influenced by the vegetation, soil, and bedrock composition as well as anthropogenic activities within the watershed, can affect the bioaccumulation of mercury through the aquatic ecosystem. Lake chemistry was significantly linked to the physical attributes of the lake and waterbody catchment. Nutrients, reactive silicate, DOC, DIC, and base cations were associated with one another and were higher in waterbody catchments with higher percentages of wetlands, proportion of the waterbody catchment classified as glacial and organic surficial material, and forest harvesting during the 1999-2000 and 2000-2009 decades (Figure 1.9).

Disturbances that alter groundwater flow and surface runoff can lead to the release of nutrients and DOC to receiving water bodies by overland flow and subsurface flow paths (Browne 2007; Kreutzweiser *et al.* 2008). The metabolism of oligotrophic boreal lakes is highly dependant on allochthonous supplies of nutrients and DOC from the watershed. The origin of DOC in lake water is primarily allochthonous at concentrations <10 mg/L (Driscoll *et al.* 1995; Garcia and Carignan 1999). Thus, the low concentrations of DOC (7.69 ± 3.74 mg/L) in these study lakes would suggest lakes are largely influenced by allochthonous inputs from the catchments.

The results of this study have shown that fish mercury concentrations were positively correlated with DOC concentrations (and negatively correlated to secchi depth) for all species of fish (Figure 1.10). These results are consistent with other studies by Schetagne and Verdon (1999), Garcia and Carignan (2000), Driscoll *et al.* (2007), and Simonin *et al.* (2008). Previous studies have reported conflicting results concerning the relationship between DOC and fish THg accumulation. Numerous studies have shown a positive correlation between lake water DOC and fish species THg concentration (McMurty *et al*. 1989; Wren *et al*. 1991; Garcia and Carignan 2000; Rencz *et al*. 2003; Belger and Forsberg 2006; Driscoll *et al*. 2007). In contrast, Grieb *et al.* (1990), Snodgrass *et al.* (2000) and Greenfield *et al.* (2001) documented a negative relationship between DOC and several different fish species. Finally, Simonin *et al.* (2008) found no association between DOC and yellow perch or bass THg accumulation. The biochemistry of DOC in each lake may be variable in terms of quality or quantity of DOC and mixed results could be explained by the influence of DOC on differences in mercury speciation, solubility, mobility and toxicity (Ravichandran 2004). However, dissolved organic carbon is likely an important vector for mercury transport from the waterbody catchment to the lakes in this study as well as an important determinant of mercury bioavailability.

Lake colour and DOC concentration are both functions of the wetland influence on the lake (D'Arcy and Carignan 1997). The DOC-mediated transport of mercury species from the catchment is likely partially responsible for the positive correlation between percentage of wetlands and mercury in lake inflows which may indirectly lead to an increase in mercury bioavailability at the base of the food web and subsequent bioaccumulation in fish and wildlife (Driscoll *et al.* 2005). Much higher mercury levels have been found in fish from coloured lakes that received greater inputs from wetlands (Rudd 1995). In this study, a significant positive relationship was found between DOC concentrations in lake water and the percentage of wetlands in the waterbody catchment (Figure 1.11: a). Although wetlands are important sources of MeHg to boreal lakes, a high amount of variability exists for the relationship between DOC concentrations and associated wetland area. The type of wetland influences differences in biogeochemical processes and the source strength of MeHg yields to the lake (St. Louis *et al.* 1996). Additionally, the degree of connectivity between the wetland and the lake is also influenced by year to year variability in water yields (St. Louis *et al.* 1996). Wetland type, annual water yield, and hydrological connectivity of wetlands associated with each lake were not measured in the current study, consequently limiting our understanding of wetland influence on water chemistry.

Wetlands are not the only important source of DOC and MeHg to lakes. Although the results of this study showed DOC concentrations in lake water were not significantly correlated with recent harvesting disturbance (Figure 1.11: b), previous research has shown higher DOC loadings in watersheds that have been logged compared with those disturbed by fire (Lamontagne *et al.* 2000; Carignan *et al*. 2000). Large quantities of easily leached or decomposed organic material remaining after forest harvesting can increase the DOC concentration in lakes. Results from the Gouin Reservoir in Haute-Maurice, Quebec showed dissolved organic carbon concentrations three-fold higher in cut lakes than in reference or burnt lakes (Carignan *et al.* 2000). Fresh DOC derived from harvesting disturbance may be less coloured than humic DOC derived from soil horizons (Carignan *et al.* 2000). Hence, variations in DOC structure and dissolved ions between harvested and non-harvested watersheds could lead to differences in the biogeochemical reactions involved with organic and inorganic mercury (O'Driscoll *et al*. 2005).

Lake productivity and nutrient inputs may also be important for mercury bioaccumulation processes. Overall, nutrients had a weak positive influence on fish THg mercury concentrations (Figure 1.12). This weak relationship is supported by findings of Hayer *et al.* (2011) that water quality attributes such as nutrients are poor predictors of mercury concentrations in fish. Following forest harvesting disturbance, nitrogen mineralization and nitrification often increase resulting in increased nitrogen availability and exports to receiving waters (Kreutzweiser *et al.* 2008). Phosphorus undergoes similar processes and responses and may increase by a smaller degree than nitrogen (Kreutzweiser *et al.* 2008). Subsequent changes in primary productivity as a result of nutrient inputs may alter food web interactions by changing the abundance of primary consumers, thus influencing bioaccumulation of mercury in the food web (Planas *et al.* 2000).

The range of pH, 5.9 to 8.6, for my study lakes was within the range found in lakes in the north-eastern United States by where pH ranged from 5.0 to 9.1 (Greenfield *et al.* 2001; Simonin *et al.* 2008). The relationship between pH and fish mercury levels is typically negative across a wide range of lake types and species (Suns and Hitchin 1990; Wren *et al.* 1991; Bodaly *et al.* 1993; Scheuhammer and Graham 1999; Garcia and Carignan 2000; Greenfield *et al.* 2001; Essington and Houser 2003; Burgess and Hobson 2006; Simonin *et al.* 2008; Scudder *et al.* 2009). However, the present study did not support the theory that a lower pH increases the bioavailability of mercury thereby making more mercury available for bioaccumulation for the different species of fish analysed. Only walleye lakes showed a significant negative correlation between pH and walleye THg concentrations ($r = -0.250$, $p<0.01$). Although the pH of lake water has been used as an important predictor of mercury levels in fish at the lanscape scale (Greenfield *et al.* 2001), many inconsistencies exist in the literature which suggests a significant degree of interaction among lake water pH and other confounding variables (Watras *et al.* 1998; Sonesten 2003). It is plausible that the observed relationship between methylmercury and lake pH is a result of the covariance between lake water colour and productivity (Meili 1994). Thus, slower growth of fish in dystrophic lakes with high concentration of humic substances may also be attributed to the limited productivity in darkly stained waters that have low pH, low light, and low oxygen concentrations.

The results of this study found that sulfate concentrations in lake water were not correlated to standardized mercury concentrations in fish tissue, comparable to research by Simonin *et al.* (2008). The range of sulfate concentrations in water chemistry of my study lakes was relatively small, with only two lakes north of Elliot Lake having elevated sulfate concentrations (Quirke Lake: 48.4 mg/L sulfate, and Whiskey Lake: 36.4 mg/L sulfate) due to the presence of mining activity (Stanrock Mine). Sulfate availability has been shown to influence inorganic Hg bioavailability and microbial activity related to methylation of mercury (Gilmour *et al.* 1992; Heyes *et al.* 2000). Studies have shown a positive relationship between sulfate concentration and mercury concentrations in surface water (Gilmour *et al.* 1998; Wiener *et al.* 2006), periphyton (Desrosiers *et al.* 2006), zooplankton (Garcia *et al.* 2007), and northern pike (Garcia and Carignan 2000). However, finding a relationship between fish THg and surface water sulfate concentrations is difficult as sulfate reduction and bioaccumulation are two co-occurring processes.

The type and amount of dissolved ions in lake water will either aid or inhibit the methylation of mercury (Gabriel and Williamson 2004). Increases in water hardness as measured by dissolved ions (conductivity, Ca, Mg, Na, and K) could possibly reduce the bioavailabilty of neutral species of mercury (e.g., $Hg(HS)_2$, $HgCl_2$, $Hg(OH)_2$, HgS) that are important in methylation (Benoit *et al.* 1999; Kelly *et al.* 2003; Gabriel *et al.* 2009). Previous research has shown the acid neutralizing capacity (ANC) and concentrations of calcium, magnesium, and potassium base cations in the water were highly negatively correlated with mercury concentration in smallmouth bass, largemouth bass, walleye and yellow perch (Grieb *et al.* 1990; Spry and Wiener 1991; Simonin *et al.* 2008). Also, Sonesten (2003) found a negative relationship between dissolved ions and the mercury concentration in perch. Cations such as Ca^{2+} and Mg^{2+} released and exported to receiving waters after forest harvesting may coagulate and increase the precipitation of chelated humic substances thereby decreasing the bioavailability of mercury (Sonesten 2003). However, base cation concentrations were not significantly correlated to standardized mercury concentrations in any fish species in the lakes analysed in this study.

Compared to previous research, the spring water chemistry measurements used in this study were weakly correlated to mercury concentrations in fish. The inherent variability in water chemistry measurements (i.e. not measured on fine enough time or spatial scales) and the complex interactions between water chemistry variables and methylation rates and bioaccumulation are reasons why this may be the case. Furthermore, sampling lake chemistry during dry or wet years may misrepresent the true associations between lake chemistry variables and mercury concentrations in piscivorous fish. In order to draw inferences on the relationships between lake chemistry and mercury concentrations in piscivorous fish, water chemistry should be sampled during the different seasons and over multiple years. Similarly, the average decadal weather conditions were measured on a coarse temporal scale. The average annual weather conditions as well as specific conditions occurring during each summer period should be considered in studies evaluating changes in watershed hydrology and mercury biogeochemistry as suggested by St. Louis *et al.* (1996).

The studies by Garcia and Carignan (1999, 2000, 2005) are amongst the few studies that have evaluated the effects of forest harvesting and fire disturbance on mercury in aquatic biota of boreal forest ecosystems. Contrary to my predictions based on these studies I did not find significant associations between disturbance as percentage area of waterbody catchment and water chemistry or fish mercury concentrations (Figure 1.14-Figure 1.17). However, lakes with fish having standardized concentrations <0.5 ppm w.w THg had consistently lower amounts of forest harvesting and natural disturbance within their waterbody catchments. The average percentage of the waterbody catchment affected by forest harvesting increased across the categories of contamination for walleye, northern pike, lake trout, and smallmouth bass. This suggests that forest harvesting may be one of many contributing factors responsible for the elevated mercury concentrations in certain species. Although lakes in this study varied with respect to lake size and catchment scale influences, a detailed study of small lakes which are likely the most sensitive may show differences in the response of fish mercury concentrations to catchment scale influences. Compared to the small headwater lakes considered in the studies by Garcia and Carignan (2000, 2005), the lakes in my study varied with respect to catchment scale characteristics as well as across a larger spatial scale. Differences in scale between studies may have influenced differences in the strength of waterbody catchment scale influences on mercury concentrations in piscivorous fish.

Similar results were found by Rask *et al.* (1998); the limnological response to clearcutting within the waterbody was modest. Moreover, the area disturbed by clear-cutting may have a variable influence on the associated waterbody depending on the duration and areal extent of the disturbance impact, the intensity of mechanical site preparation, the extent of tree removal, and the pre- and post-logging treatment of the site. The connection between the lake and the disturbed area may also vary due to changes in hydrology (water table elevations and subsurface flow paths). Since the upland soils are only occasionally inundated by water, surface runoff is typically only generated in close connection to a waterbody. Yet, much of the mercury from disturbed watersheds is transported during storm events and is associated with high loads of soil-derived suspended sediment (Engstrom *et al.* 2007). The magnitude of the forestry impacts may vary with position in the waterbody catchment but the intensity of the land-water linkage is generally the most intense near a body of water and decreases with the distance from the water body (Steedman *et al.* p. 59 in Gunn *et al.* 2003).

Forest harvesting has attracted international attention from the scientific community as a potentially significant driver in mercury contamination of aquatic ecosystems (Bishop *et al.* 2009). However, my study was not able to support this concept. Water chemistry responses to logging are highly variable and often site specific (Kreutzweiser *et al.* 2008). The probability and magnitude of logging impacts on soil nutrient cycling and exports in boreal forest watersheds is dependent on many factors such as soil type, stand and site conditions, hydrological connectivity, post-logging weather patterns, and type and timing of harvest activities (Kreutzweiser *et al.* 2008). Logging can change the hydrologic cycle of a watershed, increasing the total runoff and influx of nutrients, minerals, trace metals, and organic matter to the associated waterbody. Since the aquatic response to watershed disturbance was likely site specific it was difficult across a large landscape to connect mercury concentrations in fish to forest harvesting disturbance. There may have been differences in the sensitivity of the lakes included in this study towards forest harvesting disturbance within the waterbody catchment. The response of disturbed ecosystems to increased terrestrial inputs and within lake cycling of mercury may vary on a lake by lake basis. In future work, the influence of certain physical scale characteristics, such as forest harvesting, should be considered for lakes that are of similar waterbody catchment types.

Numerous studies have shown a significant positive relationship between associated wetlands and fish THg concentrations (Rudd 1995; Shanley *et al.* 2005; Castro *et al.* 2007; Simonin *et al.* 2008). Rypel (2010) found mercury concentrations in (bluegill [*Lepomis macrochirus*], largemouth bass [*Micropterus salmoides*], northern pike, walleye, and muskellunge [*Esox masquinongy*]) were significantly and positively related to the wetland area index. For my lakes the percentage area of the waterbody catchment covered by wetlands did not show a significant relationship with fish mercury concentrations for any species nor was a trend observed across the different lakes categorized by level of contamination. The lack of relationship between fish mercury concentrations and wetland percentage could be due to the lake/watershed-wetland size relationship. Large lakes may dilute the effects of wetland influence. Moreover, the highly variable influence of wetlands as sources of MeHg to associated waterbodies is related to differences in the internal hydrology of the wetland, the water yield of the catchment, the hydrological connectivity and the percentage of wetland areas within a catchment (St. Louis *et al.* 1996; Harris *et al.* 2007). The strength of the wetland area as a source of MeHg to the lake and contributing factor to mercury in fish would be better understood if differences in wetland hydrology and the annual water yield were determined.

Interestingly, walleye and lake trout lakes with higher amounts of wetlands and forest harvesting disturbance in their watersheds shared similarities in water chemistry characteristics (Figure 1.4 and Figure 1.6). Watersheds with greater amounts of wetlands are at higher risk of logging impacts on nutrient export and receiving water chemistry (Kreutzweiser *et al.* 2008). However, it is possible that the relatively low sample size makes it difficult to distinguish the confounding watershed interactions and the study design did not allow for sites to be selected based on separate wetland or disturbance characteristics.

Overall, my results were consistent with previous research on the affect of lake size on mercury concentrations in fish. Small lakes (size bin 1: <100 ha = 1 km^2) tended to show higher fish mercury concentrations compared to larger lakes for northern pike, lake trout, and smallmouth bass. No trend in the relationship between lake size and THg concentration was observed for walleye or brook trout populations. There are several ecological explanations for these results. Mercury concentrations tended to be higher in fish of smaller lakes than larger lakes in the Mackenzie River Basin as a probable consequence of higher summer epilimnion temperatures (Evans *et al.* 2005). The effect of lake size on mercury concentration may be stronger for some fish species than others, in part because of differences in their habitat use within a lake and growth rates as a function of lake size (Evans *et al.* 2005). Mercury concentrations were predicted to exceed the 0.5 ppm threshold guideline for commercial sale of fish in 600 mm lake trout living in lakes smaller than 6500 ha, 450 mm walleye living in lakes smaller than 2000 ha, and 600 mm northern pike living in lakes less than 100 ha (Evans *et al.* 2005). Lakes showed a strong negative relationship between mercury concentrations of length-adjusted fish and lake surface area, especially for lake trout ($r^2 = 0.71$) (Evans *et al.* 2005). The optimal thermal range and dissolved oxygen habitat can influence growth rates of lake trout (Evans *et al.* 2005). The mercury concentrations of walleye and northern pike are less likely to be correlated with lake size because they primarily inhabit littoral habitat which does not expand as appreciably with increasing lake size (Evans *et al.* 2005).

Aside from habitat, lake size has been shown to be negatively correlated with mercury concentrations due to higher epilimnion temperatures and presumably higher ratios of methylation (Bodaly *et al.* 1993). Smaller lakes are generally shallower and respond more quickly to changes in atmospheric temperature. The greater variation in surface water temperature of small lakes could lead to higher rates of methylation relative to demethylation (Bodaly *et al.* 1993). As lake size increases the pelagic zone becomes an increasingly large proportion of the lake area or volume and efficiently dilutes and cools surface water inflows from the watershed during the summer months (Evans *et al.* 2005).

Watershed characteristics are also likely to exert a relatively greater influence on small lakes (Suns and Hitchin 1990; Bodaly *et al.* 1993). The higher mercury concentrations in fish in small lakes could be a result of a greater influence from allochthonous inputs of organic matter from the watershed (Greenfield *et al.* 2001). Differences in study lake size may be one of the reasons my results were not consistent with those of Garcia and Carignan (2000, 2005) who found that the ratio of clear-cut area to lake area was significantly related to mercury concentrations in fish. Their study lakes from the Réservoir Gouin in Haute-Mauricie, Québec ranged from 20 to 230 ha with watershed areas ranging from 50 to 1970 ha whereas my study lakes had an average surface area of 1739 ha and an average waterbody catchment of 5362 ha. Additionally, I observed no significant relationship between lake surface area and mercury concentrations in fish. My results are consistent with a study of 161 lakes by Rypel (2010) which found no significant relationship between lake area and mercury concentrations in the five species of piscivorous fish. Differences in lake morphometries, drainage ratios and water renewal times may explain the different impacts of major watershed perturbations, such as forest harvesting, on the water quality and aquatic biota (Carignan and Steedman 2000).

Since MeHg is biomagnified through the food chain, among lake variability in THg for a given species can be due to inter-lake differences in trophic position (Cabana and Rasmussen 1994). Muscle concentrations of mercury are highly related to the trophic position of fish (Kidd *et al.* 1995; Sharma *et al.* 2008). Fish trophic position explained more variation in fish mercury concentrations than watershed and lake characteristics (Garcia and Carignan 2005). Accounting for trophic position is particularly useful for among-lake comparisons of omnivorous fish, such as lake trout, which have highly variable methylmercury concentrations even within a single life stage due to inter-lake differences in trophic position (Wiener *et al*. 2003).

When comparing fish mercury concentrations from different lakes, it is important to take lake trophic position into account in combination with lake and catchment characteristics (Cabana and Rasmussen 1994; Vander Zanden and Rasmussen 1996; Garcia and Carignan 2005). As trophic position increases, so does the mercury bioaccumulation in piscivorous fishes (Cabana and Rasmussen 1994; Kidd *et al.* 1995). Trophic position explained most of the variability in mercury concentrations of piscivorous fish species standardized to an average length (Garcia and Carignan 2005). In addition, Garcia and Carignan (2005) were able to determine the relationship between fish mercury concentrations and amount of watershed disturbance once the fish were normalized to trophic position. Evaluating the effect of watershed catchment scale influences on mercury levels in fish from different lakes in my study was challenging without knowledge of the among-lake differences in trophic position. Although lake-to-lake differences in fish mercury concentrations did not appear to be influenced by forest harvesting disturbance, there may have been a different relationship observed if mercury concentrations in fish were standardized to length as well as the lake specific δ^{15}N-defined trophic position. This approach would have allowed for a direct comparison of trends in mercury concentrations in fish of lakes having different trophic structure and degrees of watershed perturbation (Garcia and Carignan 2005).

Conclusion

Mercury concentrations in fish of neighbouring lakes are known to vary because of differences in internal MeHg production within a lake, differences in MeHg inputs from dissimilar types of watersheds surrounding lakes, and/or differences in the bioavailability and trophic transfer of methylated mercury. Fish mercury concentrations varied greatly among the different lakes in this study and showed weak associations with waterbody catchment and lake chemistry characteristics. This study does not support the hypothesis that elevated fish mercury concentrations are associated with lakes having recent forest harvesting disturbance within the catchment. Future work should continue to evaluate differences in lake sensitivity towards forest harvesting disturbance at a finer spatial scale. Although mercury concentrations in piscivorous fish of these study lakes did not appear to be associated with forest harvesting disturbance, the lakes may have had differences in their sensitivity to forest harvesting disturbance. Even if forest harvesting increased MeHg bioavailability within a lake, there are many dynamic processes that occur within the environment that lead to the elevated concentrations found in piscivorous fish. The association between forest harvesting and elevated mercury concentrations in piscivorous fish may have been masked by numerous biological scale factors that also influence mercury cycling within the food web. This finding may support other research that suggests that biological considerations, such as trophic structure, may exert a greater role on the concentration of mercury in piscivorous fish.

2. Walleye Mercury Relationships with Population Characteristics

Introduction

Elevated methylmercury burdens in piscivorous fish of freshwater lakes is largely a consequence of biological controls that affect the bioaccumulation of Hg (Watras *et al.* 1998; Simoneau *et al.* 2005; McIntyre and Beauchamp 2007; Lavigne *et al.* 2010). Included among the suite of biological factors affecting mercury concentration are food web structure, trophic status, fish population structure, growth rates of individual fish, and physiological controls on uptake (Grieb *et al*. 1990; Wiener *et al*. 2003; McIntyre and Beauchamp 2007; Munthe *et al.* 2007; Gabriel *et al*. 2009). Biological controls are often not considered in studies attempting to explain contaminant levels in biota due to the difficulty in accurately quantifying biological characteristics but are hypothesized to explain a fraction of the between-lake variability in THg concentrations in fish (Vander Zanden and Rasmussen 1996). Although many physical, chemical, and biological scale factors are known to influence mercury concentrations in piscivorous fish, investigations that consider the relative importance of these scales are currently lacking.

The dynamics of the food web is an important biophysical control on methylmercury bioaccumulation and biomagnification by wildlife (Munthe *et al.* 2007). The bioaccumulation of mercury ultimately originates at the lowest level of the food web and variations in the standing biomass, productivity rate, or composition of the lower trophic levels may cascade through the food web and result in alterations to the bioaccumulation rate and levels of mercury contamination in higher trophic level species among different lakes (Allen *et al.* 2005). Methylmercury produced in lake sediments can be directly incorporated into the benthic food web by periphyton (Desrosiers *et al.* 2006; Bell and Scudder 2007) and benthic invertebrates (Wong *et al.* 1997) or may enter the water column and become incorporated into pelagic food webs by microseston (phytoplankton, bacterioplankton, and cellular debris) and zooplankton uptake (Watras *et al.* 1998; Munthe *et al.* 2007). Once fish become large enough to shift their diet from planktivory to benthivory and eventually piscivory the mercury accumulation rate accelerates abruptly (Power *et al.* 2002; Wiener *et al.* 2003). Furthermore, the length of the underlying food chain significantly affects the concentration of mercury in top predator species of fish (Cabana *et al.* 1994).

Growth rates have been generally overlooked, compared to other factors such as length (Schetagne and Verdon 1999) or trophic level (Tremblay *et al*. 1998), in studies attempting to predict fish mercury levels in specific lake environments (Lavigne *et al*. 2010). The influence of growth rate on fish mercury levels has been shown to supersede the influence of all other environmental factors including point source pollution from mine tailings within the immediate vicinity of study lakes (Simoneau *et al.* 2005).

Decreased fish abundance from intensive fishing may lead to increased growth rates of the remaining fish, due to decreased inter- and intra-specific competition, resulting in more efficient growth following fishing. Verta (1990) measured growth rates before and after an intensive fishing operation in which half of the fish biomass, including piscivorous fish, was removed from a small (17 ha) remote lake in Finland. Northern pike growth rates doubled and mercury concentrations were significantly decreased after the intensive fishing operation. The decrease in mercury concentrations in fish was hypothesized to be a result of a couple different scenarios including increased growth following reduced competition for food (Verta 1990). Manipulations of fish growth were proposed as a means to manage Hg contamination in fisheries (Verta 1990).

In a northern Quebec intensive fishing experiment on two lakes Surette *et al*. (2005) found decreased walleye mercury concentrations corresponded to increased growth of fish. The decline in fish mercury was unrelated to changes in fish diet, structural alterations of the food web, reductions of methylmercury in forage fish, or reductions in the methylmercury content of the lake by fish removal (Surette *et al*. 2005). Similar results were found for a study in Norway, where significant declines in total mercury were observed in northern pike following intensive fishing in a 120 ha lake (Sharma *et al.* 2008). Larger fish that grow faster have had lower concentrations of Hg compared to smaller slower growing fish due to somatic growth dilution (SGD) whereby fish accumulate more biomass relative to Hg (Ward *et al.* 2010). Studies have shown that SGD is a factor influencing variability in Hg concentrations in fish since populations of fast growing fish have proportionately greater gains in biomass relative to the amount of mercury bioaccumulated in somatic cells than slow growing fish (Simoneau *et al.* 2005;

Lavigne *et al.* 2010). Among lake variation in activity costs of foraging and predator avoidance influences contaminant bioenergetics in freshwater fish (Rennie *et al.* 2005). Active fish that allocate energy into survival (i.e. predator avoidance) rather than growth or reproduction have been observed to grow slower resulting in higher mercury concentrations in fish due to lower growth efficiency (Rennie *et al.* 2005). Moreover, if food availability is limited there are greater costs associated with obtaining energy for growth and reproduction (Rennie *et al.* 2005). Thus, fish bioenergetic processes that influence fish growth are indirectly linked to mercury bioaccumulation.

This study examined the relationship between growth rates and mercury concentrations in walleye from a subset of lakes in northern Ontario. The specific objectives of this study were to:

1) Determine if mercury concentrations are related to growth rates of walleye.

If growth rate influences mercury bioaccumulation in piscivorous fish, then slower growing walleye would have elevated fish mercury concentrations compared to faster growing walleye as a result of higher energetic costs to increase in body size.

2) Determine if growth rate in walleye is related to abundance of individuals in a lake.

If the density of walleye individuals is higher in certain lakes than others, then I would expect that growth rates would be slower as a result of increased competition.

Methods

Study lakes were chosen from those of the Ontario Ministry of Natural Resources (OMNR) Broad-Scale Fisheries Monitoring Program (OMNR 2009). The walleye lakes included in this analysis are a subset of lakes from Chapter 1 that also had information on fish age (analysis conducted by the Ontario Ministry of Natural Resources, Mann 2004). This study was limited to walleye for 99 lakes (summarized in Appendix E). Available sample sizes for lake trout, northern pike, smallmouth bass, and brook trout populations were insufficient to perform analysis because there were too few fish or too few lakes where ageing structures were collected.

An indicator of growth rate, age at standardized length, was used to determine the influence of growth on mercury concentrartions in walleye similar to methods used by Simoneau *et al.* 2005 and Lavigne *et al.* 2010. Since it was not feasible to calculate the growth rate of individual fish throughout their lifetime, this study involved one season of fish collection that represents different age-classes of fish. Several functions or models have been used to model the mean length or weight of fishes with age (Ogle 2011). The von Bertalanffy (VB) growth model provides a good description of somatic growth of mature fish (Lester *et al.* 2004). For each lake, the VB model (Equation 1) was used to fit the non-linear relationship between fish length and age for walleye individuals using the NCStats and Fisheries Stock Assessment (FSA) package in R (Ogle 2011; R 2011). The pattern of declining growth rate with age is based on the change in energy allocation from somatic growth to reproduction as the fish matures (Lester *et al.* 2004). Thus, the walleye growth rate proxy (Age_{L500mm}) was calculated for a mature fish having a standard total length of 500 mm, which is assumed to be a mature size. Walleye in Ontario's inland lakes typically have reached maturity a total length of 500 mm. Separate studies of Ontario and Quebec lakes have shown the mean length when female walleye reach maturity is approximately 400-450 mm and the mean length for male fish to reach maturity is 324-350 mm (Morton 2006; Venturelli *et al.* 2010).

$$\text{Equation 1.} \quad L_T = L_\infty \, (1\text{-}e^{-K\,[t-t_0]})$$

Where:

- L_T is the total length (mm) of fish at time t (years; which represents the fish age as determined with ageing structures)

- L_∞ is the asymptotic length (mm)
- K is the growth coefficient, and t_0 is the hypothetical fish age (years) at a length of 0mm.

For each lake, the fit of the VB growth model was evaluated based on visual assessement of the VB growth trajectory, the plot of the model residuals, and the histogram of the residuals (Ogle 2012) (Appendix E: Table 1). The assumptions underlying nonlinear regression models of homoscedasticity and normally distributed measurement errors were validated by the plot of the model residuals and the histogram of the model residuals (Ogle 2012). Graphical procedures were sufficient to validate the model fit, but were supplemented by statistical tests (Ritz and Streibig 2008). Nonlinear model diagnostics included the lack of fit test, F-test, likelihood ratio test, and plot of residuals (Appendix E: Table E.2). However, the statistical tests used in assumption checking are hyper-sensitive (Ogle 2012). Fish were collected with a random sampling design and independence was ensured by lethal sampling which eliminated the chance for repeated measurements on the same fish. If the VB growth model failed to converge to fit the data, linear regression was used to describe the relationship between length and age for the fish population (Appendix E: Table E.3). Lakes were excluded from analysis if the walleye growth trajectory reached an asymptote before 500 mm total length (i.e. the maximum size of fish in the lake is less than 500 mm) or if the Age_{L500mm} was greater than the maximum age of fish sampled from the lake.

Walleye mercury concentrations ($[THg]_{L500mm}$ ppm w.w) were calculated for each walleye population in the previous chapter for a standard fish of 500 mm total length using a power-series regression (refer to methods section in the previous chapter for details and Appendix D) (Gewurtz *et al*. 2011). The influence of growth on mercury bioaccumulation was evaluated by the relationship between total mercury concentration at the standard length ($[THg]_{L500mm,}$) and the average growth rate (Age_{L500mm}) which was determined by linear regression analysis.

The relationship between Age_{L500mm} and $[THg]_{L500mm}$ concentration with the abundance of walleyes was evaluated with linear regression. A subset of 95 lakes with population data were included in this analysis. A measure of walleye abundance was calculated as catch per unit effort (CPUE) by determining the average number of walleye caught in the North American (NA1) large mesh nets (stretch size 38, 51, 64, 76, 89, 102, 114, 127 mm) per unit effort. Effort for large mesh gill nets was calculated by a random sampling design based on surface area and depth strata to ensure equal efforts among all lakes (Sandstrom *et al.* 2011).

Results

Walleye Growth Rates

The age of walleye at the standard total length of 500 mm (Age_{L500mm}) ranged from 3.44 years to 20.56 years suggesting that walleye growth rates are highly variable amongst lakes. Mature fish that take a long time to reach a standard total length of 500 mm tended to have higher THg concentrations than faster growing fish of other lakes; illustrated by the relationship between the growth rate (Age_{L500mm}) and standardized mercury concentration ($[THg]_{L500mm}$) for 99 walleye lakes (Figure 2.1; r^2=0.333, p< 0.001). Thus, mercury concentrations in walleye were typically higher in lakes with slower growing walleye compared to lakes with faster growing walleye.

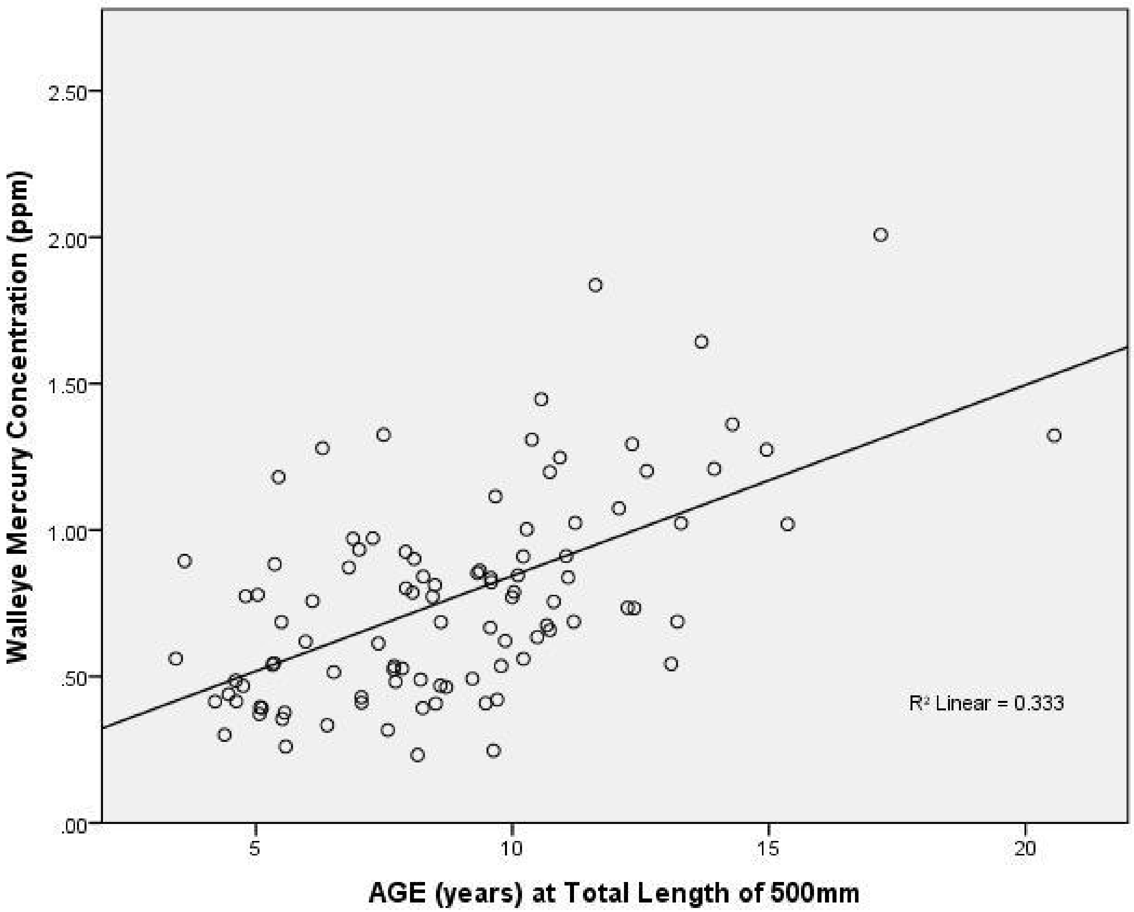


Figure 2.1. The relationship between walleye growth rate (Age_{L500mm}) and standard mercury concentrations ($[THg]_{L500mm}$) for 99 lakes.

Associations between Walleye Abundance and Growth Rates

Walleye standard mercury concentrations were not significantly associated with the density of walleye (Figure 2.2; n=95, r^2 =0.001, p= 0.807). However, the time required for walleye to grow to 500 mm (Age_{L500mm}) was significantly positively related to the density of walleye (Figure 2.3; n=95, r^2 =0.136, p<0.0001) indicating that walleye growth rates are lower in more dense populations.

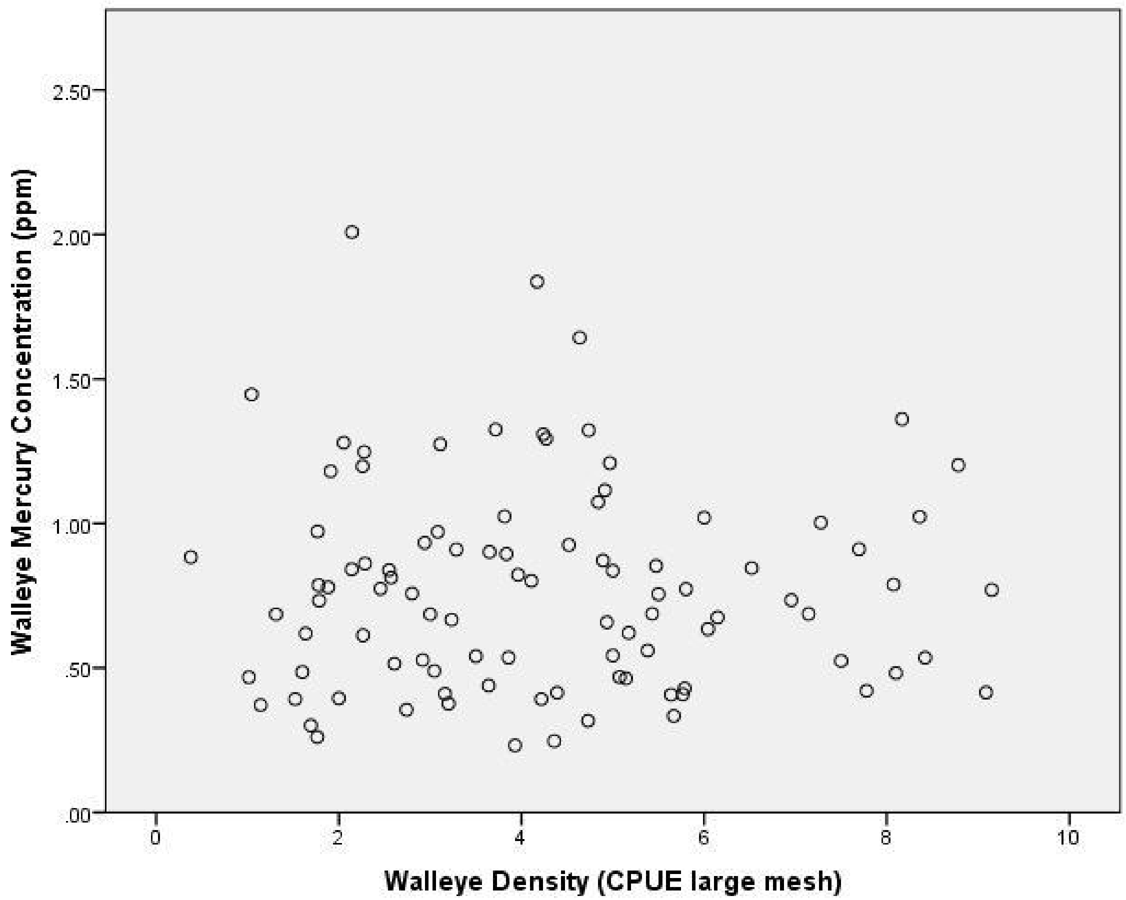


Figure 2.2. The relationship between walleye density and walleye standardized total mercury concentration for 95 walleye lakes.

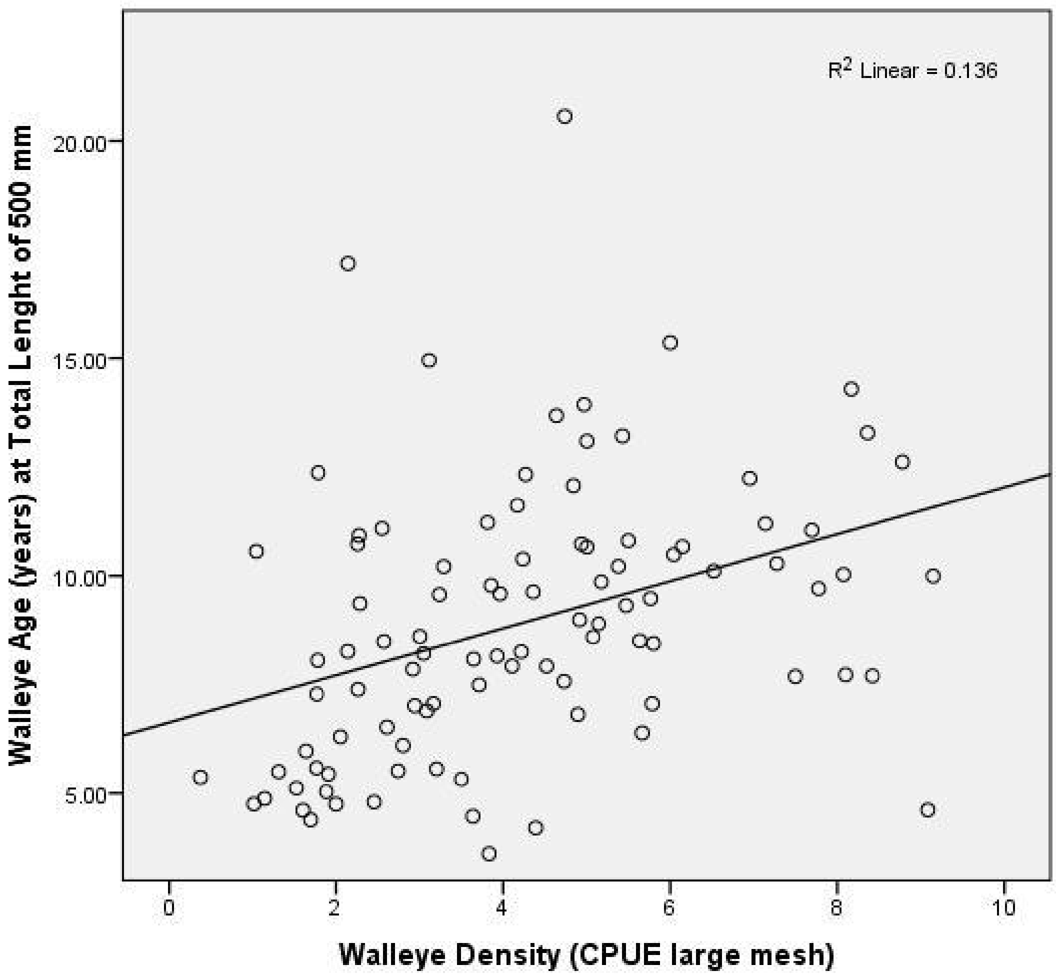


Figure 2.3. The relationship between walleye density and growth rate (Age_{L500mm}) for 95 walleye lakes.

Discussion

The results of this study suggest that the mercury concentrations of walleye are significantly influenced by growth rate with slower growing fish tending to have higher Hg levels (Figure 2.1). The positive relationship observed between standardized mercury levels and the age that fish reached 500 mm for the 99 walleye populations studied (Figure 2.1) was similar to the strong positive relationship between walleye standard mercury levels and age found by Simoneau *et al.* (2005) for 12 walleye populations ($r = 0.92$, $p<0.001$) and Lavigne *et al.* (2010) for 54 walleye populations ($r^2=0.55$, $p<0.001$). The strong positive relationship between growth and mercury concentrations supports the conclusion of other studies that growth efficiency influences the process of mercury bioaccumulation by walleye (Simoneau *et al.* 2005; Lavigne *et al.* 2010; Ward *et al.* 2010).

Walleye growth rates are the by-product of an optimization process of energy allocation which is influenced at a fish community level. The growth rates of walleye were significantly affected by the walleye abundance (CPUE) (Figure 2.3). These results could suggest that slow growth rates and higher mercury concentrations in fish of certain lakes could be related to fish density in those lakes. Lakes with dense fish populations may exhibit reduced growth rates in fish causing fish that grow less efficiently to have higher tissue Hg concentrations relative to faster growing fish (Stafford and Haines 2001). Greater intra-specific competition for food sources exists in less intensely fished smaller lakes compared to larger lakes (Stafford and Haines 2001).

Slow growth rates of walleye found in certain lakes may be indicative of relatively low fishing pressures. Lakes that are less accessible or lakes with low fishing pressure relative to lake size may have old, slow growing fish with high mercury levels as a result of mercury bioaccumulation over a longer lifespan. Similar results were found in lakes of the Mackenzie River Basin (Evans *et al.* 2005) where fish populations tend to be dominated by older individuals where fishing pressure is low or nonexistent (Evans *et al.* 2005). Increased fishing pressure from sport-fishing or commercial fishing may remove older, larger fish from the population thereby reducing energy costs associated with predator avoidance and competition for food amongst the remaining fish. A sustainable amount of fishing pressure may influence fish bioenergetics and contaminant accumulation by enhancing growth rates and subsequently decreasing the burden of contaminants in piscivorous fish. Although fishing pressure was not measured for the lakes in this study, it may be related to growth rates of piscivorous fish and should be considered in future work.

The growth rate is an integrative indicator of multiple environmental conditions and varies in the significance on its effect on fish THg concentrations (Lavigne *et al.* 2010). Environmental conditions that promote slow growth, such as low light, low temperature, and low nutrient concentrations, can affect bioaccumulation and biomagnification rates of mercury (Muir *et al.* 2001). The foraging efficiency of fish, such as walleye, that rely on sight for hunting is related to water turbidity (Abrahams and Kattenfeld 1997). Thus, environmental factors that influence growth rates are also likely responsible for the elevated mercury in walleye observed in smaller dystrophic lakes.

Growth rate is a reflection of the surrounding environment, including the climate and geography, which in turn affects fish metabolism, feeding ecology and lake productivity (Schindler 1995). However, because the lakes of this study were sampled across a broad region of Ontario at approximately the same latitude I do not expect that climate variability is a primary contributor to observed differences in growth rates. This conclusion is further supported by the results of Chapter 1 which showed climate had very little association with variability in lake characteristics. Often, fish of more northern regions are slower growing than those from lakes in central Canada (Colby and Nepszy 1981; Galarowicz and Wahl 2003). However, further examination of the environmental and biological parameters that control the growth of fish could provide insight to the causes of inter-lake differences in fish mercury concentrations.

The growth of walleye may also be negatively affected by the toxicological effects of mercury. Fish exposed to dietary methylmercury have shown loss of coordination, diminished swimming activity, starvation, reduced growth, impaired reproduction and mortality (Friedmann *et al.* 1996; Mahaffey 2006; Weis 2009; Sandheinrich and Wiener 2011). A review by Sandheinrich and Wiener (2011) concluded that adverse health effects such as altered behaviour, development, growth and reproduction are associated with mercury concentrations of 0.30 ppm w.w. or greater in freshwater fish. As the average mercury concentrations in fish exceeded this threshold for the majority of lakes in this study, mercury is likely negatively impairing the growth of piscivorous fish to some degree. Future studies could examine the effects of mercury toxicity on fish growth to determine if a feedback mechanism between growth, toxicity, and mercury concentrations in fish exists.

Conclusion

Walleye growth rates, which were highly variable among lakes, are able to explain a portion of the variability in mercury concentrations among lakes. The variability in mercury concentrations in walleye may be explained by differences in growth efficiency with respect to costs associated with obtaining and processing food. Abundance of individuals in a population may indirectly influence mercury concentrations in walleye by decreasing growth rates. As mercury bioaccumulation in fish is influenced by growth, further research would benefit from a better understanding the biological and environmental influences on growth. Further work could investigate the influence of fishing pressure on lake ecology and the link to bioenergetics and mercury bioaccumulation of piscivorous fish.

3. General Discussion and Conclusion

In Chapter 1, standardized THg concentrations were shown to differ by an order of magnitude or more between populations of fish in lakes across FMZs 4, 6, 7, 8, 10 and 11 in northern Ontario. No point source of mercury pollution existed for the vast majority of lakes in this study yet their piscivorous fish populations are highly contaminated with mercury and human consumption restrictions are advised. The variability in THg concentrations was not associated with forest harvesting nor many of the waterbody catchment or lake characteristics used as predictors of THg concentrations in fish in similar studies. For all species, higher THg concentrations in fish were associated with smaller, darkly stained dystrophic lakes. For all species, fish THg concentrations showed the strongest relationship with dissolved organic carbon concentrations in lake water.

Although the present study did not show significantly higher THg levels in piscivorous fish of lakes affected by clear-cutting, the disturbance may have altered the cycling of mercury in the watershed ecosystem resulting in an indirect influence on THg bioavailability and concentrations in lower trophic level biota. The lack of relationships between watershed and lake scale factors and THg concentrations in piscivorous fish could be due to the limitations of the available dataset, rather than a true absence of influence from factors such as forest harvesting disturbance or wetlands. Future attempts to relate landscape features with mercury in piscivorous fish of this region could focus on similar watershed/lake ecosystems with a greater range of disturbance area impacted by forest harvesting and fewer confounding factors.

Studies that attempt to evaluate the effects of forest harvesting would benefit from using a before-and-after design. However, my study would have likely been more useful in demonstrating watershed impacts on water chemistry and lower trophic level organisms. Studies which evaluate the spatial associations with THg contamination in piscivorous fish would benefit by considering characteristics such as trophic structure and growth rates.

In Chapter 2, the variability in THg concentrations of walleye was partially explained by differences in growth rates. Thus, biological characteristics of the fish population and aquatic community likely exert a strong influence on the high levels of

THg found in piscivorous fish. Understanding the complex relationships between landscape, limnological, and ecological factors that synergistically affect the mercury cycle will subsequently lead to a better understanding of THg concentrations that bioaccumulate and biomagnify to dangerously high levels in piscivorous fish. This study illustrated the difficulty of determining the associations between mercury concentrations in fish across a broad landscape, suggesting mercury concentrations in fish vary on a lake by lake basis due to differences in the relative importance of watershed, lake and biological scale factors. Although mercury concentrations in piscivorous fish were weakly associated with characteristics of the waterbody catchment, further investigation of the importance of scale on lake sensitivity to disturbances is required. Future work could consider additional analysis of lake ecosystems that have similar population and community ecology in order to further evaluate the importance of catchment scale characteristics.

Further research of mercury sensitive ecosystems across northern Ontario is required as much of the boreal forest is a likely target for development (i.e. mining) in the near future. The development of new road networks will likely increase sport-fishing opportunities in once remote lakes which may have elevated concentrations of mercury in the fish. Increased fishing pressures may alter both fish populations and community dynamics, thus indirectly affecting how mercury is bioaccumulated and biomagnified. Therefore, due to the dynamics of lake ecosystems, constant monitoring of mercury contaminant levels in fish of Ontario's lakes is recommended.

5.0. References

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Appendix A. Study Lakes

Table A.1. Location of Zone 4 Lakes (Lambert Conformal Conic) and Lake and Waterbody Catchment Size.

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
434593	12600814	Amik Lake	1162	6371	3
512960	12692672	Arc Lake	596	2439	3
459929	12617103	Bawden Lake	376	3652	2
530912	12712328	Bertaud Lake	413	1630	2
402629	12602779	Big Sandy Lake	3808	8715	4
418463	12776799	Birch Lake	11623	27923	5
371292	12717211	Bluffy Lake	2487	7108	4
476406	12663780	Bury Lake	707	3731	3
310071	12631720	Canyon Lake	1698	4962	4
485854	12680963	Carling Lake	1556	7080	4
468858	12565537	Cecil Lake	1561	3461	4
324990	12637738	Clay Lake	2755	5587	4
333752	12777247	Coli Lake	2114	5901	4
284167	12708670	Confusion Lake	1463	4044	3
293902	12697574	Conifer Lake	1138	2339	3
487129	12585430	Crystal Lake	116	3468	2
285845	12645985	Delaney Lake	1278	2434	3
450241	12656413	Expanse Lake	864	2436	3
539395	12680867	Fitchie Lake	1148	3704	3
572464	12714847	Greenbush Lake	2927	8207	4
436694	12577734	Gustauson Lake	144	409	2
435325	12753625	Hailstone Lake	531	3505	3
299174	12756352	Hammell Lake	831	2528	3
395542	12595854	Hartman Lake	518	1246	3
468545	12656165	Hik Lake	127	417	2
448189	12569117	Indian Lake (z4)	4000	9051	0
413261	12734793	Jubilee Lake	978	3413	3
313310	12803651	Kirkness Lake	2145	4876	4
450083	12594018	Kukukus Lake	4168	11196	4
457574	12556611	Little Sandbar Lake	218	776	2
316403	12773119	Little Vermilion Lake	5489	28772	5
285749	12723508	Longlegged Lake	2794	6332	4
438189	12570252	Mameigwess Lake	5242	11311	5
500299	12580447	Mattawa Lake	1788	5459	4
557332	12706430	McCrea Lake	4014	9797	4
520305	12703813	Miniss Lake	6921	20617	5
388208	12635537	Mold Lake	48	272	1
433773	12572574	Mud Lake	128	307	2
339198	12793996	Nungesser Lake	7417	20278	5
281749	12740892	Onnie Lake	165	865	2
450188	12713407	Otatakan Lake	1696	10094	4
332524	12713786	Pakwash Lake	8678	20682	5

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
352993	12660943	Perrault Lake	3302	7716	4
312079	12833992	Pikangikum Lake	6060	30050	5
394412	12758157	Premier Lake	152	734	2
465218	12594113	Press Lake	3646	12631	4
427603	12639455	Richardson Lake	195	544	2
547474	12666317	Savant Lake	12599	29659	5
354004	12823745	Silcox Lake	874	3264	3
550849	12656399	Silver Lake	153	501	2
551382	12651650	Smye Lake	284	1403	2
403705	12674783	Spruce Lake	115	963	2
283978	12639975	Tom Lake	57	247	1
480092	12603620	Towers Lake	102	765	2
458919	12575438	Victoria Lake	926	2690	3
348311	12673012	Wabaskang Lake	5766	16932	5
411866	12683134	Wapesi Lake	2351	6036	4
476992	12586188	Wintering Lake	1654	6088	4
256537	12697810	Wyder Lake	265	935	2

Table A.2. Location of Zone 6 Lakes (Lambert Conformal Conic) and Lake and Waterbody Catchment Size.

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
527619	12408457	Addie Lake	117	904	2
627303	12506420	Arrowroot Lake	177	1318	2
548475	12474291	Athelstane Lake	1765	4039	4
548687	12437396	Batwing Lake	615	2616	3
742768	12569255	Beatty Lake	652	1462	3
502487	12488265	Bedivere Lake	2280	9270	4
654257	12460317	Bisect Lake	32	321	1
650129	12532218	Black Sturgeon Lake	4912	11470	4
541491	12436133	Blunder Lake	126	642	2
543249	12581757	Brightsand Lake	1345	4135	3
629572	12617682	Bukemiga Lake	795	3098	3
516518	12456147	Burchell Lake	1045	3220	3
639075	12534051	Circle Lake	387	3401	2
654686	12498032	Cliff Lake	41	222	1
518301	12466946	Crayfish Lake	541	2573	3
616820	12591318	Crevasse Lake	118	641	2
642746	12596863	Cry Lake	245	589	2
599433	12471768	Dog Lake	14536	40328	5
732429	12616189	Elbow Lake	346	4389	2
552989	12579347	Empire Lake	681	3056	3
722768	12618833	Frank Lake	512	2624	3
673988	12519139	Frazer Lake	1968	5777	4
616762	12538661	Gennis Lake	124	590	2
559779	12543818	Grew Lake	280	1179	2
557494	12604054	Harmon Lake	2947	10674	4

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
603038	12461177	Hawkeye Lake	430	1884	2
593479	12569483	Holinshead Lake	1958	4796	4
562281	12547723	Holly Lake	303	740	2
536736	12428466	Jacob Lake	173	815	2
605926	12514561	Jolly Lake	101	1380	2
534879	12468369	Kashabowie Lake	2249	6346	4
561055	12631708	Kawaweogama Lake	3525	8532	4
568255	12554732	Kearns Lake	932	3426	3
546699	12445961	Kekekuab Lake	546	5839	3
595350	12519097	Lac des Iles	1558	5832	4
564413	12490393	Lac du Milieu	121	782	2
531912	12575966	Little Metionga Lake	682	3364	3
520225	12404065	Little North Lake	849	3486	3
559880	12548479	Loganberry Lake	427	3362	2
549942	12456029	Lower Shebandowan Lake	2295	7565	4
611615	12617992	Maggotte Lake	115	432	2
649387	12425091	Marie Louise Lake	772	2733	3
573203	12431627	Marks Lake	39	337	1
537114	12581376	Metionga Lake	1994	4942	4
561823	12500437	Muskeg Lake	3494	6836	4
651623	12456690	Nalla Lake	46	310	1
522575	12441511	Nelson Lake	653	2338	3
510734	12420168	Northern Light Lake	6550	18133	5
620585	12603032	Obonga Lake	3730	12430	4
548057	12543327	Pakashkan Lake	5177	21728	5
632646	12448317	Penassen Lakes	41	384	1
738459	12584864	Pinel Lake	86	509	1
681258	12508959	Purdom Lake	244	1058	2
555191	12509718	Ricestalk Lake	266	1643	2
595591	12620562	Sandison Lake	310	2514	2
542161	12414673	Sandstone Lake	725	5090	3
559622	12589875	Sparkling Lake	1267	4352	3
521061	12447817	Squeers Lake	370	735	2
521232	12415603	Sunbow Lake	549	1763	3
521783	12429695	Titmarsh Lake	968	1751	3
656008	12460031	Upper Hunters Lake	25	289	1
645162	12613595	Wabinosh Lake	1730	10965	4
633100	12499425	Walotka Lake	94	295	1
565359	12616607	Wapikaimaski Lake	3569	12497	4
637918	12619165	Waweig Lake	1152	2972	3
720873	12611745	Weewullee Lake	55	200	1
532859	12425689	Weikwabinonaw Lake	1247	3727	3

Table A.3. Location of Zone 7 Lakes (Lambert Conformal Conic) and Lake and Waterbody Catchment Size.

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
794919	12565608	Kenogamisis Lake	4218	24858	4
969518	12376582	Whitefish Lake - Expanded Reservoir	1650	8516	4

Table A.4. Location of Zone 8 Lakes (Lambert Conformal Conic) and Lake and Waterbody Catchment Size.

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
1291424	12564925	Burntbush Lake	128	1771	2
1221851	12369436	Dumbell Lake	174	904	2
1232217	12357829	Duncan Lake	985	6249	3
1130393	12382665	Horwood Lake	5274	23939	5
1182659	12385186	Indian Lake	54	480	1
1109973	12392621	Ivanhoe Lake	1758	12343	4
1131585	12566909	Kapuskasing River - ds Lost River	1850	38188	4
1327664	12399096	Larder Lake	3688	8073	4
1186896	12408109	Mattagami River	2478	18326	4
1246433	12378936	Mistinikon Lake	1229	7279	3
1204220	12392270	Muskasenda Lake	480	3731	2
1137338	12338050	Opeepeesway Lake	2062	8397	4
1244399	12407971	Radisson Lake	540	2909	3
1334747	12395884	Raven Lake	573	2087	3
1144824	12346548	Rice Lake	2466	8640	4
1105510	12365818	Rollo Lake	808	3695	3
1298682	12389471	Round Lake	1183	9643	3
1323481	12385465	St. Anthony Lake	495	1218	2
1182659	12385186	Dungaree, or Indian Lake	54	480	1

Table A.5. Location of Zone 10 Lakes (Lambert Conformal Conic) and Lake and Waterbody Catchment Size.

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
961120	12359140	Anjigami Lake	1134	3532	3
1042850	12273706	Anvil Lake	90	300	1
1094394	12218649	Astonish Lake	79	392	1
1120712	12257617	Bark Lake	1884	6485	4
1071149	12288840	Beak Lake	7	69	1
1054212	12190918	Big Basswood Lake	2689	5048	4
1060424	12185052	Bright Lake	1218	4996	3
1261497	12328341	Carmen Lake	21	132	1
1079737	12194212	Chiblow Lake	1996	3356	4
1066116	12203059	Constance Lake	119	729	2
1010358	12240615	Devils Lake	196	826	2
978310	12276723	Dick Lake	100	436	1
1079872	12250470	Doehead Lake	33	143	1
1078418	12235919	Duval Lake	166	751	2

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
1080580	12221329	Endikai Lake	620	2924	3
1011761	12289607	Galloway Lake	26	211	1
947766	12337776	Gamitagama Lake	194	836	2
1028576	12239431	Garden Lake	140	920	2
1004594	12276646	Gavor Lake	125	298	2
1122372	12215979	Geiger Lake	89	317	1
1040854	12273885	Gong Lake	377	1634	2
1030606	12283378	Goulais Lake	241	2222	2
1019376	12281448	Graham Lake	110	408	2
975178	12274413	Griffin Lake	160	1157	2
1033060	12291303	Gull Lake	137	1277	2
1014419	12278020	Hanes Lake	137	1183	2
1261142	12331310	Island Lake	31	147	1
1075410	12228452	Kirkpatrick Lake	1110	3816	3
1152372	12223081	Klondyke Lake	199	773	2
1269069	12245067	Kukagami Lake	1858	3973	4
961699	12313928	Kwagama Lake	210	978	2
965523	12345630	Lake 34	353	2994	2
1097683	12179694	Lauzon Lake	2198	4779	4
1073573	12195214	Little Chiblow Lake	642	1462	3
959132	12268276	Mamainse Lake	148	613	2
1087083	12197715	Matinenda Lake	4128	10796	4
1104464	12190120	McGiverin Lake	275	1729	2
1021554	12213732	McMahon Lake	224	743	2
1041669	12293094	Megisan Lake	613	1679	3
951894	12341503	Mijinemungshing Lake	498	1649	2
1093189	12202350	Moon Lake	517	1534	3
1248524	12325148	Okinada Lake	94	772	1
951084	12333902	Old Woman Lake	266	1023	2
1210343	12186891	Panache Lake	8006	17134	5
1018975	12278318	Point Lake	77	487	1
958897	12282899	Queminico Lake	111	437	2
1013429	12283052	Quinn Lake	137	834	2
987995	12278881	Quintet Lake	160	738	2
1117226	12210611	Quirke Lake	2065	3829	4
1038912	12255202	Ranger Lake	2311	6283	4
1209878	12271332	Rome Lake	227	985	2
1100679	12193913	Rossmere Lake	120	571	2
1165222	12240037	Rushbrook Lake	173	631	2
1022346	12260122	Saddle Lake	112	528	2
1042513	12263813	Saymo Lake	843	2671	3
1258936	12330212	Shack Lake	165	674	2
1037549	12220022	Shelden Lake	141	611	2
987027	12239835	Sill Lake	42	146	1
1030895	12273374	South Branch Lake	200	728	2
1129767	12232149	Spinweb Lake	96	378	1
1109659	12186698	Turtle Lake	150	891	2
958830	12276463	Upper Pancake Lake	102	378	2

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
1055703	12217727	Wakomata Lake	2479	6121	4
1198802	12182054	Walker Lake	349	804	2
1021862	12263616	Ward Lake	50	233	1
994683	12245847	Weckstrom Lake	21	180	1
1133655	12205493	Whiskey Lake	989	2524	3

Table A.6. Location of Zone 11 Lakes (Lambert Conformal Conic) and Lake and Waterbody Catchment Size.

X centroid	Y centroid	Lake Name	Lake Surface Area (ha)	Waterbody Catchment Area (ha)	Size Bin
1308726	12263605	Aileen Lake	157	573	2
1313579	12306907	Anima Nipissing Lake	1920	5992	4
1285823	12322822	Anvil Lake	232	966	2
1294456	12201941	Bear Lake	70	6858	1
1327319	12158655	Cadden Lake	42	218	1
1330039	12286332	Cassels Lake	731	2623	3
1332705	12178394	Clear Lake	265	687	2
1312580	12263713	Cross Lake	1623	4894	4
1296514	12218722	Deer Lake	300	1335	2
1289592	12298679	Diamond Lake	950	3309	3
1318365	12274910	Driftwood Lake	89	760	1
1285643	12266232	Emerald Lake	581	1521	3
1318761	12257820	Hangstone Lake	365	1462	2
1309656	12314990	Kittson Lake	81	1366	1
1305679	12286856	Kokoko Lake	543	1567	3
1295501	12315264	Lady Evelyn Lake	6632	17241	5
1333658	12197958	Lake Nipissing Lake	72861	104362	5
1375346	12193749	Lake Nosbonsing	1765	4616	4
1273504	12324691	Makobe Lake	2007	5579	4
1294748	12183429	Mercer Lake	59	304	1
1309454	12220516	Muskosung Lake	322	1038	2
1336443	12278662	Rabbit Lake	2110	6388	4
1315693	12246280	Red Cedar Lake	2308	7178	4
1305915	12294821	Red Squirrel Lake	394	1077	2
1328906	12302420	Rib Lake	675	2345	3
1259610	12293372	Rodd Lake	32	136	1
1303850	12275533	Temagami Lake	20628	43149	5
1364926	12205438	Trout Lake	1885	6557	4
1317690	12270344	Wasaksina Lake	587	1911	3
1335195	12253348	Wicksteed Lake	1476	5387	3

Appendix B. Software Used, Data Sources, and Waterbody Catchment Variables

Software Used:

- ArcGIS (version 9.3.1) [2]
- Microsoft Excel 2003
- Microsoft Access 2003
- R (version 2.13.1)[3] and vegan package[4]
- IBM SPSS Statistics 19 [5]
- Historical Climate Analysis Tool[6]
- NCStats and FSA package- Fisheries Stock Assessment Methods Version 0.2-6 [7]

Data from the Ontario Geospatial Data Exchange (OGDE) Land Information Ontario (LIO) Used:

- Digital Elevation Model (version. 2.0.0)[8]
- Enhanced Flow Direction Grids for Zones 15, 16n, 16se, 17ns, 17nn
- Provincial Land Cover 2000 – 27 Classes (PLC2000)
- Bedrock Geology of Ontario (1:250,000), Ontario Geological Survey Data
- Ontario Hydrographic Network: Waterbody
- Water Virtual Flow – Seamless Provincial Data Set
- Surficial Geology from the digitized version of the Northern Ontario Engineering Geology Terrain Study (NOEGTS)
- Wetland Interim
- Natural Resources Values Information System (NRVIS) 2009 Roads layer

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[2] Environmental Systems Research Institute (ESRI), Inc. ArcGIS. Version 9.3.1. Environmental Systems Research Institute, Inc. Redlands, California.
[3] R. 2011. version 2.13.1. The R Foundation for Statistical Computing. <http://www.R-project.org>
[4] Okasanen, J. 2012. Vegan: Community Ecology Package. Version 2.0-2. <http://cran.r project.org/web/packages/vegan/index.html>
[5] IBM SPSS Statistics. Version 19.
[6] Historical Climate Analysis Tool (HCAT). 2012. Northwest Science and Information Section, Ontario. Ministry of Natural Resources.
[7] Ogle, D. 2011.Fisheries Stock Assessment Methods. Northland College. < http://www.rforge.net/FSA/>
[8] Ontario Ministry of Natural Resources (OMNR). 2005. Provincial Digital Elevation Model., version 2.0.0 [computer file] Land Information Ontario (LIO). Peterborough, ON. URL: http://www.mnr.gov.on.ca/en/Business/LIO/index.html[vd26]

Forest harvesting and natural disturbance data were available from the forensic forestry layer provided by Larry Watkins, Forest Analyst for the Forest Evaluations and Standards Section of the Ministry of Natural Resources. The forensic forestry layers were created from combining Ontario Forest Resources Information, Silvicultural Effectiveness Monitoring, Provincial fire, Provincial blowdown, and Annual Report GIS data.

Data from Larry Watkins, Forest Analyst for the Forest Evaluations and Standards Section, Forests Branch of the Ministry of Natural Resources (Sault Ste. Marie, Ontario).

- Forensic Forestry layers: yrdep_90 (1990-1999) and yrdep_00 (2000-2009)
- Harvesting Data from Annual Report 2002-2009
- Provincial Depletion Blowdown and Fire 2000-2009

Table B.1. Waterbody Catchment Area Characteristics per Lake (Harv = forest harvesting disturbance, Dist = natural disturbance).

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Addie Lake	12.9	2.1	0.0	18.7	1.3	0.0	0.0062
Aileen Lake	27.4	5.1	0.0	0.0	0.0	0.0	0.0031
Amik Lake	18.2	1.4	0.0	1.4	4.6	0.0	0.0032
Anima Nipissing	32.3	2.6	1.1	0.6	0.0	0.0	0.0019
Anjigami Lake	32.1	0.4	0.0	0.0	0.0	0.0	0.0089
Anvil Lake	30.1	4.1	0.0	0.0	0.0	0.0	0.0020
Anvil Lake	24.0	12.4	0.0	0.0	0.0	0.0	0.0052
Arc Lake	26.1	1.0	0.0	0.0	0.5	3.3	0.0000
Arrowroot Lake	13.4	0.1	4.4	0.0	0.0	0.0	0.0175
Astonish Lake	20.1	0.1	0.0	0.0	0.0	0.0	0.0096
Athelstane Lake	43.7	5.0	4.3	0.6	1.1	0.0	0.0029
Bark Lake	29.3	2.0	6.8	3.7	0.0	0.0	0.0026
Batwing Lake	23.5	4.7	1.4	11.9	0.4	0.0	0.0063
Bawden Lake	12.9	0.8	0.2	14.9	0.7	0.0	0.0073
Beak Lake	9.7	7.7	0.0	0.0	0.0	0.0	0.0000
Bear Lake	1.1	0.7	0.0	0.0	0.0	0.0	0.0137
Beatty Lake	44.6	0.5	0.0	0.0	0.0	0.0	0.0000
Bedivere Lake	24.6	2.0	8.7	0.4	3.1	0.0	0.0037
Bertaud Lake	25.3	0.2	0.0	0.0	70.1	0.0	0.0000
Big Basswood	53.3	0.4	1.3	0.7	0.0	0.0	0.0076
Big Sandy Lake	43.7	2.7	2.2	3.1	2.4	0.5	0.0053
Birch Lake	42.0	1.2	0.0	0.9	5.4	1.3	0.0001
Bisect Lake	9.9	0.0	0.0	0.0	0.0	0.0	0.0000
Black Sturgeon Lake	42.8	1.0	4.5	2.9	21.4	12.5	0.0066
Bluffy Lake	35.8	3.5	0.3	5.1	13.4	0.3	0.0060
Blunder Lake	39.0	3.0	2.5	1.8	1.4	0.0	0.0076
Bright Lake	24.4	0.5	0.1	0.0	0.0	0.0	0.0117
Brightsand Lake	33.0	2.2	3.3	0.0	22.0	0.0	0.0008
Bukemiga Lake	25.7	0.0	3.8	14.8	0.3	0.0	0.0118
Burchell Lake	32.5	9.9	7.6	3.4	0.0	0.0	0.0045
Burntbush Lake	7.3	18.1	0.0	16.0	0.0	0.0	0.0082
Bury Lake	19.0	0.9	0.8	0.0	0.7	63.5	0.0019

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Cadden Lake	19.1	4.7	0.0	0.0	0.0	0.0	0.0007
Canyon Lake	34.2	2.9	0.2	0.4	0.4	0.0	0.0072
Carling Lake	22.0	1.1	0.0	0.3	0.4	97.4	0.0000
Carmen Lake	16.1	0.0	0.0	0.0	0.0	0.0	0.0000
Cassels Lake	32.8	1.4	0.0	1.1	0.0	0.0	0.0051
Cecil Lake	45.1	0.2	14.7	13.5	0.2	0.0	0.0068
Chiblow Lake	59.5	0.0	5.2	1.3	1.1	0.0	0.0016
Circle Lake	11.4	0.8	3.2	1.2	6.3	0.0	0.0176
Clay Lake	50.3	1.5	0.8	10.6	2.6	42.7	0.0023
Clear Lake	38.6	0.6	0.0	0.0	0.0	0.0	0.0032
Cliff Lake	18.5	0.5	1.6	1.8	0.0	12.9	0.0056
Coli Lake	35.8	2.3	1.0	5.2	0.5	0.5	0.0071
Confusion Lake	36.2	3.6	9.0	0.0	36.8	0.0	0.0040
Conifer Lake	48.7	2.3	1.2	0.0	11.8	0.0	0.0002
Constance Lake	16.3	0.3	0.0	14.7	0.0	0.0	0.0120
Crayfish Lake	21.0	0.6	5.1	6.1	0.5	0.0	0.0003
Crevasse Lake	18.3	1.4	0.0	0.0	0.0	0.0	0.0000
Cross Lake	33.2	4.1	0.0	0.0	0.0	0.0	0.0029
Cry Lake	41.6	0.0	0.0	0.0	0.0	43.9	0.0000
Crystal Lake	3.3	1.6	0.0	5.5	2.3	0.0	0.0108
Deer Lake	22.7	0.6	0.7	0.0	0.0	0.0	0.0146
Delaney Lake	52.5	0.7	0.4	0.0	0.7	0.0	0.0000
Devils Lake	23.7	1.1	8.3	21.6	0.0	0.0	0.0133
Diamond Lake	28.7	4.8	0.0	0.0	0.0	0.0	0.0019
Dick Lake	23.0	0.1	0.0	0.0	0.0	0.0	0.0000
Doehead Lake	23.1	1.3	0.0	0.0	0.0	0.0	0.0000
Dog Lake	36.2	1.5	4.0	2.0	0.1	0.2	0.0037
Driftwood Lake	11.7	1.3	0.0	0.0	0.0	0.0	0.0016
Dumbell Lake	19.3	2.7	0.0	0.0	0.0	0.0	0.0003
Duncan Lake	15.9	1.7	4.2	0.9	0.4	2.8	0.0026
Duval Lake	22.1	6.8	0.0	16.4	0.0	0.0	0.0070
Elbow Lake	7.9	0.3	3.1	4.5	7.9	0.0	0.0092
Emerald Lake	38.2	0.0	3.3	3.2	0.0	0.0	0.0089

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Empire Lake	22.3	4.1	3.4	6.4	3.7	0.0	0.0132
Endikai Lake	21.2	0.7	1.8	5.5	0.0	0.0	0.0004
Expanse Lake	35.5	0.2	1.9	3.5	0.8	44.4	0.0049
Fitchie Lake	31.0	2.5	0.0	0.0	0.0	0.0	0.0005
Frank Lake	19.5	1.7	0.0	0.7	0.0	0.0	0.0080
Frazer Lake	34.1	2.1	1.7	2.6	21.3	4.3	0.0073
Galloway Lake	12.1	3.7	14.6	0.0	0.0	0.0	0.0157
Gamitagama Lake	23.2	0.0	0.0	0.0	0.0	0.0	0.0000
Garden Lake	15.2	0.4	5.7	0.0	0.0	0.0	0.0176
Gavor Lake	42.1	4.6	0.0	0.0	0.0	0.0	0.0012
Geiger Lake	28.0	0.6	0.0	0.0	0.0	0.0	0.0000
Gennis Lake	21.1	7.8	1.5	4.6	0.0	0.0	0.0041
Gong Lake	23.1	6.4	0.4	7.2	0.0	0.0	0.0067
Goulais Lake	10.8	7.5	0.3	4.4	0.3	0.0	0.0000
Graham Lake	27.0	5.7	0.0	0.0	0.0	0.0	0.0058
Greenbush Lake	35.7	5.9	0.0	0.0	0.6	0.0	0.0000
Grew Lake	23.8	11.6	0.0	0.0	0.0	0.0	0.0000
Griffin Lake	13.9	0.3	0.0	7.4	0.0	0.0	0.0001
Gull Lake	10.7	9.5	29.3	0.3	2.0	0.0	0.0079
Gustauson Lake	35.1	0.1	33.0	0.0	23.7	0.0	0.0016
Hailstone Lake	16.2	6.8	0.0	0.0	0.1	0.5	0.0000
Hammell Lake	32.9	2.4	2.6	0.0	8.2	0.0	0.0009
Hanes Lake	11.6	6.5	7.6	0.0	0.4	0.0	0.0078
Hangstone Lake	25.0	0.5	0.0	0.0	0.0	0.0	0.0148
Harmon Lake	27.7	2.9	3.0	0.2	32.4	0.0	0.0049
Hartman Lake	41.5	3.3	0.1	0.7	9.0	0.0	0.0112
Hawkeye Lake	22.9	0.5	0.0	4.7	0.3	0.0	0.0114
Hik Lake	30.4	0.4	0.0	0.0	0.0	0.0	0.0000
Holinshead Lake	40.8	3.8	1.9	0.0	0.0	0.0	0.0002
Holly Lake	40.8	1.1	0.0	0.0	0.0	0.0	0.0000
Horwood Lake	23.5	2.9	7.7	3.6	0.7	0.0	0.0066
Indian Lake (FMZ4)	44.3	2.6	8.0	0.0	0.2	0.2	0.0042
Island Lake	20.8	1.3	0.0	0.0	0.0	0.0	0.0000

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Ivanhoe Lake	14.5	2.1	10.9	5.0	2.4	0.0	0.0095
Jacob Lake	21.3	2.0	13.0	0.0	1.8	6.3	0.0056
Jolly Lake	7.3	4.0	62.6	0.8	0.5	0.0	0.0114
Jubilee Lake	28.6	1.7	0.0	17.7	0.2	0.6	0.0041
Kapuskasing River - ds Lost R.	6.8	5.0	0.9	4.8	0.0	0.0	0.0124
Kashabowie Lake	35.8	4.0	8.9	2.8	1.0	0.0	0.0023
Kawaweogama Lake	41.4	6.7	1.4	0.0	4.8	0.0	0.0002
Kearns Lake	27.5	3.3	10.5	0.0	0.0	35.2	0.0026
Kekekuab Lake	9.3	6.6	0.1	12.9	2.2	0.0	0.0062
Kenogamisis Lake	17.5	8.4	0.1	3.5	0.0	0.0	0.0066
Kirkness Lake	44.0	0.5	0.0	0.0	0.0	0.0	0.0004
Kirkpatrick Lake	29.1	0.5	0.5	0.1	0.0	0.0	0.0019
Kittson Lake	6.0	7.0	0.0	0.0	0.0	0.0	0.0000
Klondyke Lake	25.8	4.8	0.0	0.0	0.0	0.0	0.0000
Kokoko Lake	34.7	0.5	0.0	0.0	0.0	0.0	0.0052
Kukagami Lake	46.8	1.6	0.0	0.5	0.0	0.0	0.0062
Kukukus Lake	37.2	1.1	1.1	1.3	15.8	0.0	0.0024
Kwagama Lake	21.6	0.6	0.0	0.0	0.0	0.0	0.0009
Lac des Iles	26.8	2.0	14.5	5.6	0.2	0.0	0.0041
Lac du Milieu	15.4	12.8	9.0	5.2	0.0	0.0	0.0041
Lady Evelyn	38.7	5.8	0.0	3.6	0.0	0.0	0.0007
Lake 34	11.8	0.2	1.5	0.0	2.5	0.0	0.0029
Lake Nipissing	74.7	1.1	0.2	0.3	0.0	0.0	0.0046
Lake Nosbonsing	38.3	3.2	0.0	0.0	0.0	0.0	0.0154
Larder Lake	45.7	1.1	0.2	0.9	0.0	0.0	0.0055
Lauzon Lake	46.0	0.4	0.5	0.0	0.0	0.0	0.0057
Little Chiblow Lake	43.9	0.9	3.3	1.7	0.0	0.0	0.0041
Little Metionga Lake	21.4	6.1	0.0	5.9	0.3	0.0	0.0048
Little North Lake	24.4	0.6	0.0	7.4	1.6	0.0	0.0009
Little Sandbar Lake	28.2	2.4	1.3	0.3	3.5	0.0	0.0012
Little Vermilion Lake	19.4	1.1	11.1	11.4	1.2	0.8	0.0079
Loganberry Lake	13.0	7.8	0.1	22.5	0.0	11.3	0.0000
Longlegged Lake	44.1	3.4	0.5	0.0	3.9	0.0	0.0019

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Lower Shebandowan Lake	51.6	0.4	0.1	0.0	1.7	0.0	0.0098
Maggotte Lake	26.7	1.1	3.6	3.6	9.5	0.0	0.0121
Makobe Lake	37.4	4.4	0.0	0.0	0.0	0.0	0.0006
Mamainse Lake	24.1	0.1	0.0	22.1	0.0	0.0	0.0032
Mameigwess Lake	46.3	2.2	6.9	3.4	3.3	0.0	0.0038
Marie Louise Lake	28.3	0.9	0.0	0.0	2.5	0.0	0.0087
Marks Lake	11.7	1.4	4.1	0.0	8.1	10.1	0.0019
Matinenda Lake	38.4	0.7	1.2	4.2	0.0	0.0	0.0036
Mattagami River	13.5	2.6	3.2	3.5	4.8	0.0	0.0122
Mattawa Lake	32.8	1.3	0.0	1.7	0.3	0.0	0.0032
McCrea Lake	41.2	1.8	0.0	0.0	0.2	0.0	0.0000
McGiverin Lake	15.9	0.1	4.5	13.0	1.1	0.0	0.0009
McMahon Lake	30.1	0.3	0.0	4.2	0.0	0.0	0.0047
Megisan Lake	36.5	2.2	0.0	0.0	1.0	0.0	0.0029
Mercer Lake	19.9	0.2	0.0	12.8	0.0	0.0	0.0091
Metionga Lake	40.4	0.9	0.0	0.0	0.7	0.0	0.0002
Mijinemungshing Lake	30.6	0.0	0.0	0.0	0.0	0.0	0.0004
Miniss Lake	33.6	0.8	0.0	0.0	20.2	0.0	0.0000
Mistinikon Lake	17.5	1.5	2.4	4.0	0.0	0.0	0.0044
Mold Lake	17.6	1.4	3.9	40.0	3.4	0.0	0.0098
Moon Lake	33.7	1.5	1.1	0.0	0.0	0.0	0.0006
Mud Lake	41.7	0.0	32.4	0.0	12.8	0.0	0.0007
Muskasenda Lake	12.9	6.1	11.7	3.7	0.4	0.0	0.0065
Muskeg Lake	51.1	18.5	4.0	0.6	1.0	0.0	0.0057
Muskosung Lake	31.0	2.0	1.1	1.1	0.0	0.0	0.0051
Nalla Lake	14.9	0.0	0.0	0.0	0.0	0.0	0.0036
Nelson Lake	27.9	12.4	0.0	6.2	5.6	0.0	0.0125
Northern Light Lake	37.0	2.9	5.1	1.1	10.2	7.9	0.0041
Nungesser Lake	36.6	0.7	1.9	2.6	0.0	0.8	0.0020
Obonga Lake	30.0	0.3	9.2	6.5	0.2	0.0	0.0061
Okinada Lake	12.1	0.0	0.0	0.0	0.0	0.0	0.0000
Old Woman Lake	26.6	0.0	0.0	0.0	0.0	0.0	0.0000
Onnie Lake	19.1	1.6	9.5	11.4	1.6	0.0	0.0051

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Opeepeesway Lake	24.6	1.7	3.6	3.4	0.1	0.0	0.0030
Otatakan Lake	17.0	5.6	0.0	3.4	1.3	0.0	0.0004
Pakashkan Lake	23.9	5.8	10.5	2.7	0.9	3.5	0.0066
Pakwash Lake	50.8	2.1	1.7	1.4	3.4	0.1	0.0051
Panache Lake	46.9	1.6	2.0	0.1	0.0	0.0	0.0049
Penassen Lakes	10.7	0.6	0.0	0.0	0.0	0.0	0.0000
Perrault Lake	42.8	3.1	4.2	3.1	0.5	46.2	0.0093
Pikangikum Lake	20.3	0.0	0.0	0.0	0.0	0.0	0.0021
Pinel Lake	16.8	7.7	0.0	26.4	0.0	0.0	0.0011
Point Lake	15.9	1.4	0.0	0.0	0.0	0.0	0.0051
Premier Lake	20.7	0.5	0.0	18.6	2.5	19.7	0.0000
Press Lake	29.5	2.7	0.9	0.2	15.9	1.3	0.0012
Purdom Lake	23.1	0.1	11.3	8.7	1.3	0.0	0.0015
Queminico Lake	25.5	0.3	0.0	14.3	0.0	0.0	0.0171
Quinn Lake	16.5	6.1	0.0	0.0	0.0	0.0	0.0016
Quintet Lake	21.6	1.1	0.0	0.0	0.0	0.0	0.0075
Quirke Lake	54.2	0.5	0.2	0.0	0.0	0.0	0.0123
Rabbit Lake	33.1	1.4	1.5	0.1	0.1	0.0	0.0059
Radisson Lake	18.6	1.7	21.1	0.0	4.2	0.0	0.0048
Ranger Lake	36.8	0.6	1.5	0.2	0.0	0.0	0.0058
Raven Lake	27.5	0.7	0.0	13.9	0.0	0.0	0.0018
Red Cedar Lake	34.4	3.7	1.1	1.3	0.0	0.0	0.0032
Red Squirrel Lake	36.6	1.0	0.0	0.0	0.0	0.0	0.0068
Rib Lake	28.8	0.9	0.7	1.1	0.1	0.0	0.0053
Rice Lake	28.7	2.9	0.0	6.6	0.4	0.0	0.0000
Ricestalk Lake	16.2	34.9	0.0	2.1	2.4	0.0	0.0136
Richardson Lake	35.8	0.0	0.0	0.0	0.0	0.0	0.0000
Rodd Lake	23.3	3.3	0.0	0.0	0.0	0.0	0.0000
Rollo Lake	21.9	1.0	36.1	11.3	1.5	0.0	0.0073
Rome Lake	23.0	1.2	0.0	0.0	0.0	0.0	0.0002
Rossmere Lake	21.0	0.0	0.6	23.0	0.0	0.0	0.0098
Round Lake	12.3	1.8	1.3	1.5	0.0	0.0	0.0142
Rushbrook Lake	27.5	3.5	0.0	0.0	0.0	0.0	0.0034

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Saddle Lake	21.2	0.9	0.0	0.0	0.3	0.0	0.0144
Sandison Lake	12.4	10.8	1.9	0.0	0.0	0.0	0.0094
Sandstone Lake	14.3	0.7	4.2	4.9	7.6	0.1	0.0068
Savant Lake	42.7	2.6	0.9	3.0	1.1	0.0	0.0011
Saymo Lake	31.6	0.2	0.0	0.0	3.3	0.0	0.0124
Shack Lake	24.5	2.5	0.0	3.3	0.0	0.0	0.0021
Shelden Lake	23.1	0.8	7.3	7.8	0.0	0.0	0.0074
Silcox Lake	26.8	2.1	0.0	0.0	0.0	1.2	0.0000
Sill Lake	28.8	0.0	0.0	0.0	0.0	0.0	0.0113
Silver Lake	30.5	2.3	0.0	3.3	0.0	0.0	0.0051
Smye Lake	20.2	2.2	3.0	7.1	0.0	0.0	0.0052
South Branch Lake	27.4	2.0	0.0	0.0	0.8	0.0	0.0066
Sparkling Lake	29.2	1.7	12.1	1.9	11.9	0.0	0.0064
Spinweb Lake	25.5	1.7	0.0	0.0	0.9	0.0	0.0104
Spruce Lake	11.9	6.0	31.2	0.0	7.3	100	0.0136
Squeers Lake	50.3	1.0	0.4	0.9	0.7	0.0	0.0031
St. Anthony Lake	40.6	0.3	7.2	0.1	0.0	0.0	0.0018
Sunbow Lake	31.3	2.5	0.0	3.2	0.6	0.0	0.0011
Temagami Lake	48.4	0.8	0.0	0.0	0.0	0.0	0.0029
Titmarsh Lake	55.3	0.8	0.0	6.0	1.3	0.0	0.0028
Tom Lake	23.1	0.4	0.0	0.0	0.0	0.0	0.0020
Towers Lake	13.3	1.5	0.0	0.0	0.0	0.0	0.0069
Trout Lake	28.8	0.4	0.0	0.0	0.0	0.5	0.0142
Turtle Lake	16.9	0.7	13.7	0.0	0.0	0.0	0.0010
Upper Hunters Lake	8.7	2.1	0.0	0.0	0.0	0.0	0.0010
Upper Pancake Lake	26.9	1.0	0.0	30.0	0.0	0.0	0.0065
Victoria Lake	34.4	2.8	0.0	1.8	0.4	17.0	0.0010
Wabaskang Lake	36.5	2.7	0.3	0.1	0.4	1.2	0.0048
Wabinosh Lake	15.8	0.2	2.5	0.1	0.2	0.0	0.0078
Wakomata Lake	40.5	0.5	2.5	0.0	0.0	0.0	0.0038
Walker Lake	43.5	0.0	1.6	1.0	0.0	0.0	0.0037
Walotka Lake	31.8	0.0	8.0	0.7	0.0	0.0	0.0028
Wapesi Lake	39.0	1.2	1.9	1.7	5.5	51.7	0.0012

Lake Name	Lake Area (%)	Wetland Area (%)	Harv1999-2000 (%)	Harv2000-2009 (%)	Dist1999-2000 (%)	Dist2000-2009 (%)	Road Density km_ha
Wapikaimaski Lake	29.9	4.0	0.0	0.0	1.1	0.0	0.0023
Ward Lake	21.5	0.8	0.0	0.0	0.0	0.0	0.0174
Wasaksina Lake	31.4	1.4	0.0	0.0	0.0	0.0	0.0025
Waweig Lake	38.8	0.3	4.9	6.1	1.7	0.0	0.0073
Weckstrom Lake	11.6	0.0	52.4	0.0	0.0	0.0	0.0161
Weewullee Lake	27.6	2.1	0.0	0.0	0.0	0.0	0.0062
Weikwabinonaw Lake	33.5	1.2	11.4	11.6	0.3	0.1	0.0012
Whiskey Lake	39.2	0.5	0.0	0.0	0.0	0.0	0.0040
Whitefish Lake - Expanded Reservoir	20.0	0.4	0.1	0.5	0.1	0.0	0.0031
Wicksteed Lake	27.5	2.7	0.1	2.1	0.0	0.0	0.0020
Wintering Lake	27.2	0.4	0.0	3.4	0.9	9.7	0.0040
Wyder Lake	28.4	4.4	0.0	0.0	1.4	0.0	0.0000
Dungaree, Indian Lake	22.4	0.0	21.5	1.9	5.0	0.0	0.0247

Table B.2. Presence of Waterpower Generating Station [1]

Lake Name	Object ID	Feature
Chiblow Lake	800494993	Waterpower Generating Station
Dog Lake	150901352	Waterpower Generating Station
Kapuskasing River - ds Lost River	301085944	Waterpower Generating Station
Mattagami River	950283143	Waterpower Generating Station
Whitefish Lake – Expanded Reservoir	51473894	Waterpower Generating Station

Table B.3. Presence of Mining Activity [10]

Lake Name	Object ID	Count of Mining Features
Anima Nipissing Lake	700944883	2
Big Basswood Lake	800495003	4
Big Sandy Lake	1300269298	1
Birch Lake	1100449655	1
Bright Lake	800495087	3
Burchell Lake	150485798	1
Cross Lake	700945513	1
Emerald Lake	700945497	3
Endikai Lake	800494484	2
Hammell Lake	1100449877	1
Horwood Lake	950283324	18
Kenogamisis Lake	750778091	14
Kukagami Lake	500958245	7
Lady Evelyn	200258020	2
Lake Nipissing	700946217	4
Larder Lake	200414151	18
Lauzon Lake	800495169	2
Lower Shebandowan Lake	151326646	2
Mamainse Lake	800493561	3
Mattagami River	950283143	1
Mistinikon Lake	200295614	2
Moon Lake	800494883	2
Muskasenda Lake	950283305	1
Opeepeesway Lake	450604606	8
Panache Lake	500960023	5
Quirke Lake	800494782	5
Ranger Lake	800493788	1
Rib Lake	700945042	1
Round Lake	200413019	11
Savant Lake	1300268877	1
Temagami Lake	700945140	7
Wakomata Lake	800494532	1
Whiskey Lake	500959450	2

[9] Hydroelectric Generating Station (WatPowGenStn.shp) Data from Land Information Ontario

[10] Old Mine (MNDMMINE.shp) Data from Land Information Ontario

Appendix C. Water Chemistry Variables

Table C.1. Spring Water Chemistry Data for each lake.

Lake Name	Year Sampled	SECCHI (m)	ALKTI (mg/L as $CaCO_3$)	CAUT (mg/L)	CLIDUR (mg/L)	COLTR (TCU)	COND25 (µS/cm)	DIC (mg/L)	DOC (mg/L)	KKUT (mg/L)	MGUT (mg/L)	NAUT (mg/L)	NNHTUR (µg/L)	NNOTUR (µg/L)	NNTKUR (µg/L)	pH	PPUT (µg/L)	SIO3UR (mg/L)	SSO4UR (mg/L)	FEUT (µg/L)	MNUT (µg/L)
Addie Lake	2008	1.9	23.3	8.32	1.74	51.4	60.6	5.86	7.6	0.415	2.18	1.16	26	14	447	7.45	11.7	3	2.75		
Aileen Lake	0																				
Amik Lake	2009	1.8	18.4	6.26	0.25	99.2	45	4.42	14	0.62	1.79	1.19	14	80	439	7.12	18.3	1.88	1.4	323	
Anima Nipissing	2009	8	5.74	3.28	0.21	9.6	30.8	1.8	2.9	0.2	0.935	0.705	10	38	148	6.75	3.1	0.54	5.85		
Anjigami Lake	2009	4.2	7.3	3.48	0.14	48.4	27.8	2	6.8	0.34	0.73	0.77	22	156	277	6.78	5.9	1.92	3.05	77	
Anvil Lake	2009	3.5	7.62	3.48	0.22	23.8	26.6	2.16	4.7	0.19	0.68	0.665	14	62	265	6.88	5.3	1.12	3.5	93	34
Anvil Lake	2008	2	2.03	1.76	0.14	26	20	0.78	3.8	0.285	0.645	0.715	18	166	213	6.3	5.4	1.98	4.9	82	
Arc Lake	2009	2.8	19	6.46	0.16	70	44.4	4.68	10.9	0.535	1.68	0.665	18	64	274	7.22	7.6	1.62	1.05	106	
Arrowroot Lake	2008	1.5	19.6	6.54	1.53	100	54.4	4.66	16	0.49	2.51	1.43	18	76	505	7.14	8.2	2.84	2.1		
Astonish Lake	2009	4.9	14	6.68	0.18	17.6	40	4.24	4	0.175	0.615	0.76	16	70	207	6.96	4.8	1.7	4	43	
Athelstane Lake	2008	4.5	19.4	0	0.51	21.6	51.8	5.18	6.6	0	0	0	8	42	289	7.05	7.9	3.26	2.75		
Bark Lake	2010	1.8	9.33	3.6	0.26	28.8	30.6	2.76	4.8	0.305	1.03	0.995	20	12	355	7.01	14.1	2.36	4.1	175	
Batwing Lake	2008	2.5	12.4	4.06	0.37	47.6	36.6	3.06	9.8	0.38	1.67	1.01	16	56	470	7	9	1.58	2.8		
Bawden Lake	2009	1.8	39.8	7.14	0.27	89.6	47.2	4.92	13.7	0.605	1.76	0.84	30	54	479	7.2	14.7	1.76	1	171	
Beak Lake	0																				
Bear Lake	2010	5.7	10.8	5.26	2.01	7.2	47.8	3.16	2.9	0.575	1.32	1.73	4	2	214	7.16	6.7	0.48	7.75		
Beatty Lake	2008	3.9	97.4	29.5	0.1	7	189	23.3	3.1	0.325	5.17	0.6	30	40	310	7.96	11.1	1.46	1.25	212	
Bedivere Lake	2008	1.5	10.7	0	0.3	75.4	33.8	2.4	11.5	0	0	0	20	96	463	6.78	14.8	3.22	2.35		
Bertaud Lake	2009	1.8	16.5	5.94	0.13	66.4	39.8	4.2	11.4	0.57	1.34	0.57	12	74	322	7.06	5.4	1.2	1.05	110	
Big Basswood	2009	7	4.78	3.36	1.66	2.6	33.4	1.48	0.8	0.27	0.82	1.38	2	288	94	6.71	3	0.16	6.4		
Big Sandy Lake	2009	2.7	65.2	20.8	2.33	24.6	136	8.32	7.2	1.16	4.37	2.59	14	42	313	7.82	14.9	1.1	2.8	76	
Birch Lake	2009	3.9	22.7	8.32	0.15	29	52.4	5.82	8.1	0.5	1.26	0.645	22	10	312	7.24	9	0.9	1.5		
Bisect Lake	2008	2.6	8.33	1.28	0.19	44	27.2	2.42	7.3	2.54	0.655	0.21	16	72	303	6.73	7.1	1.98	2.2	108	
Black Sturgeon Lake	2008	2.3	36.6	15.6	12	56.2	125	8.9	10.7	0.88	3.54	8.92	8	158	361	7.48	5.5	3.56	3.65	84	
Bluffy Lake	2009	1.1	21.6	7.92	0.18	129	50.2	5.12	16	0.625	1.84	0.735	16	54	501	7.22	15	1.84	1.15	331	34
Blunder Lake	2008	1	9.61	4.04	0.4	114	31.8	2.46	15	0.285	1.34	0.875	24	74	675	6.79	21	2.72	1.65	116	
Bright Lake	0																				

Lake Name	Year Sampled	SECCHI (m)	ALKTI (mg/L as $CaCO_3$)	CAUT (mg/L)	CLIDUR (mg/L)	COLTR (TCU)	COND25 (µS/cm)	DIC (mg/L)	DOC (mg/L)	KKUT (mg/L)	MGUT (mg/L)	NAUT (mg/L)	NNHTUR (µg/L)	NNOTUR (µg/L)	NNTKUR (µg/L)	pH	PPUT (µg/L)	SIO3UR (mg/L)	SSO4UR (mg/L)	FEUT (µg/L)	MNUT (µg/L)
Brightsand Lake	2008	1.5	13.1	4.28	0.25	75.4	34.8	3.3	8.1	0.56	1.29	0.975	24	68	353	7.02	10.8	3.12	1.45	193	
Bukemiga Lake	2008	2	6.84	2.78	0.26	94.4	23.6	1.84	11.3	0.515	0.94	0.855	14	74	338	6.23	8.8	2.92	1.25	69	
Burchell Lake	2008	7	23.8	0	3.91	13.2	97.8	6.42	4.8	0	0	0	8	124	216	7.1	3.1	2.72	12.8	495	
Burntbush Lake	2010	1.6	28.1	9.64	0.32	128	62	7.48	18.6	0.33	2.69	0.78	8	8	269	7.3	19.7	0.76	1.35	183	
Bury Lake	2009	2.5	30.2	9.76	0.22	80.6	65.4	7.66	11.8	0.705	2.38	0.775	26	74	369	7.32	13	1.92	1.3	110	
Cadden Lake	2009	3.1	2.96	2.3	13.7	50.4	65	1.2	8.6	0.43	0.635	8.56	18	2	401	6.2	11.4	0.36	2.25		
Canyon Lake	2009	3.4	14.8	4.68	0.36	24.8	39.6	3.84	7	0.615	1.38	1.01	26	30	313	7.16	10.2	0.92	2.25	76	
Carling Lake	2009	2.7	24.2	8.22	0.23	77.8	55	6.04	12.2	0.635	1.89	0.705	14	80	333	7.19	16.3	1.72	1.3	84	
Carmen Lake	2009		14.8	5.94	0.14	33.4	42	3.64	5.7	0.255	1.26	0.86	34	4	259	7.18	11.4	2.24	4.7	103	52
Cassels Lake	2009	4	20.1	7.94	3.38	27	74	5.34	5.2	0.325	2.35	2.43	18	80	208	7.29	5.6	1.76	8.45	53	
Cecil Lake	2009	3.9	37.8	12	2.43	13.8	90.6	9.64	4.1	0.76	2.74	2.19	8	2	210	7.53	6.6	1.28	3		
Chiblow Lake	2009	6.1	4.6	2.74	0.39	6.2	25	1.36	2.4	0.235	0.67	0.82	10	92	148	6.62	3.7	1.02	5.15		
Circle Lake	2008	2.1	30.3	8.68	0.84	63.6	70.4	7.68	11.5	0.49	2.6	2.43	20	88	458	7.35	13.2	4.66	2.2	182	
Clay Lake	2009	0.9	38.6	12	3.82	94.8	99.6	9.78	11.5	1.28	2.89	4.97	36	40	521	7.43	38.6	0.92	4.25	601	41
Clear Lake	2009	6.5	7.2	2.7	0.84	10.6	32	1.58	4.8	0.47	1.02	1.12	10	2	261	6.99	4.3	0.44	4.95		
Cliff Lake	2009	3.8	34.1	10.4	2.52	17.2	83	8.64	6.4	0.81	2.75	2.47	18	6	302	7.54	9.7	0.48	2.25		
Coli Lake	2009	1.9	11.3	4.28	0.13	45.6	32	3	9	0.515	0.87	0.78	36	32	382	6.96	14.9	2.72	1.15	244	
Confusion Lake	2009	4	17.2	4.7	0.23	28.6	40.2	3.88	7.1	0.835	1.6	1.06	8	118	289	7.1	5.2	0.68	2.25		
Conifer Lake	2009	5.2	16.8	4.96	0.24	17.8	44.4	4.44	5.3	1.04	1.79	1.01	6	110	220	7.2	4.7	0.5	3.15		
Constance Lake	2010	4	12.2	4.46	1.72	8.6	41.8	3.48	3.8	0.425	1.31	1.97	16	8	302	7.11	7.7	2.64	4.85		
Crayfish Lake	2008	1.8	8.16	0	1.55	47.4	32	2.24	7.6	0	0	0	12	84	327	6.69	8.6	2.18	2.2	233	
Crevasse Lake	2008	4.2	35.4	10.3	0.26	30.8	81.8	9	5.5	0.625	3.09	2.41	8	86	266	7.07	6.9	3.36	3.15	165	
Cross Lake	2008	3.5	10.5	5.7	0.68	9.4	51.4	2.84	2.6	0.29	1.82	1.03	20	44	214	7.14	8.9	0.52	9.95	28	
Cry Lake	2008	6	27.9	9.16	0.51	16.4	69	7.32	2.6	0.5	2.14	1.37	8	30	187	6.89	21.4	1.64	3.7	48	
Crystal Lake	2009	1.8	34.3	6.92	0.36	123	41.4	4.32	14.8	0.505	1.31	0.8	16	32	419	7.04	10.3	1.62	1.25	228	
Deer Lake	2008	1.8	15	5.62	1.41	64.8	55.2	3.9	8.3	0.57	2.04	1.11	24	58	489	7.07	17.7	1.54	5.5	165	
Delaney Lake	2009	4.5	13.3	4.18	0.47	8.4	37.2	3.64	3.6	0.59	1.25	1.11	10	18	209	7.06	7.1	0.24	3.15		
Devils Lake	2009	7.1	5.14	2.98	0.25	9.4	25.8	1.66	3.7	0.38	0.635	0.76	10	64	268	6.65	3.5	0.84	4.5		
Diamond Lake	2008	3.5	3.19	2.24	0.22	12.6	23.8	0.98	2.4	0.265	0.71	0.755	18	38	211	6.63	3.6	0.94	5.65		
Dick Lake	2009	4.2	8.84	4.66	0.16	24.6	29.6	2.8	4.7	0.21	0.46	0.525	18	206	246	6.69	5.5	1.32	3.2	100	36
Doehead Lake	2009	3.1	9.41	4.14	0.26	35.8	32	2.54	5.8	0.16	1.05	0.84	6	62	258	6.81	8.2	2.66	4.05	137	
Dog Lake	2008	2.1	23.9	7.92	0.62	69.4	51.2	5.96	11.5	0.635	2.04	4.14	8	190	397	7.27	4.1	3.48	2.45	181	
Driftwood Lake	0																				

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Dumbell Lake	2008	3.2	28.7	8.08	0.49	33	71.8	7.68	8.6	0.215	3.4	0.73	36	8	353	7.45	8.8	1.04	4.6	90	
Duncan Lake	2008	3	21.3	7.8	0.21	35.4	57.6	5.44	6	0.23	1.96	0.95	20	74	273	7.33	6.5	2.44	4.5	69	
Duval Lake	2009	4.6	6.76	3.58	0.28	26.8	27.2	2.16	4.8	0.155	0.655	0.655	6	88	212	6.69	4.8	1.8	3.6	111	
Elbow Lake	2008	1.5	50.6	15.6	0.64	96.2	105	11.8	11.9	0.575	3.29	0.52	18	68	404	7.48	10.4	1.68	1.05	213	
Emerald Lake	2009	8	8	6.32	0.85	5.2	55.8	2.32	2.5	0.305	1.59	0.725	10	26	119	6.92	3.2	0.92	14.2		
Empire Lake	2008	2	11.3	3.64	0.38	48.2	31.2	3.02	6	0.56	1.08	0.96	24	64	342	6.95	10.8	1.92	1.55	225	
Endikai Lake	2009	5.8	8.6	4.12	0.32	18.8	33.6	2.4	3.6	0.21	0.885	0.815	2	118	154	6.84	3.8	2.24	4.9	52	
Expanse Lake	2009	2.3	29.2	9.3	0.2	78.2	64.4	7.16	11.8	0.675	2.23	0.83	34	54	410	7.4	14.2	1.66	1.4	149	
Fitchie Lake	2009	2.2	23.2	8.6	0.64	76.4	54.2	6	11.3	0.505	1.63	0.78	20	78	333	7.22	8.1	1.76	1	175	
Frank Lake	2008	1.6	39.8	11	0.27	71.2	85	9.66	13.1	0.355	2.58	0.535	28	28	482	7.41	12.5	1.44	1.15		
Frazer Lake	2008	2.5	38.1	9.34	1.06	25.8	72.4	7.82	6.4	0.245	2.6	1.71	26	34	348		11.4	3.2	2.7	189	
Galloway Lake	0																				
Gamitagama Lake	2009	7.3	3.19	1.94	0.12	8.8	16.4	1.08	2.1	0.165	0.33	0.5	40	64	194	6.62	2.7	0.36	2.95		
Garden Lake	2009	3.1	8.57	4.22	0.23	50	32.8	2.48	7.6	0.285	1.02	0.865	10	126	304	6.78	6.7	2.86	4.45	141	
Gavor Lake	2009	3.5	16	7.34	0.29	33.6	43.6	4.44	7.1	0.26	0.535	0.525	16	100	328	6.98	5.3	1.2	3.1	110	30
Geiger Lake	2009	4.1	3.34	2.3	0.42	35.6	20.4	1.22	4.9	0.16	0.455	0.745	8	54	243	6.34	6.3	1.48	3.7	129	
Gennis Lake	2008	2.6	29.3	8.1	0.46	25.4	65.6	7.4	8	0.4	2.77	0.94	20	112	437	7.36	8.1	1.04	1.4	328	
Gong Lake	2009	5	6.96	3.46	0.2	32.8	26.4	2.04	6	0.22	0.74	0.765	14	82	302	6.69	5.1	2.08	3.7	95	
Goulais Lake	2009	3	7.25	3.96	0.19	49.8	27.6	1.84	12.8	0.19	0.9	0.645	12	90	391	6.54	9.5	2.36	2.9	173	
Graham Lake	2009	2.4	4.64	2.88	0.18	71.8	22.4	1.48	9.4	0.2	0.7	0.67	24	84	416	6.38	8.3	2.12	3.05	147	
Greenbush Lake	2009	4.3	25.4	9.28	0.15	49.2	58	6.44	11.3	0.415	1.8	0.44	56	172	433	7.25	6.6	0.5	0.7	145	
Grew Lake	0																				
Griffin Lake	2009	10	5.77	3.32	0.16	8.6	25.8	1.78	2.7	0.215	0.45	0.565	6	346	173	6.69	2.8	1.56	3.55		
Gull Lake	2009	1.5	8.56	4.22	0.17	98.4	29.8	2.32	12.6	0.22	1.03	0.685	14	82	425	6.56	9	2.44	2.95	189	
Gustauson Lake	0																				
Hailstone Lake	2009	1.1	6.16	3.36	0.11	134	22.4	1.52	18	0.25	0.79	0.685	22	44	442	6.42	11.5	2.48	0.85	488	
Hammell Lake	0																				
Hanes Lake	2009	2.5	7.41	3.9	0.27	48.8	28.6	2.04	8.5	0.215	0.72	0.615	76	68	328	6.63	5.8	2.04	3.75	133	35
Hangstone Lake	2009	2.2	15	5.12	0.57	41.4	45.2	3.84	7.1	0.26	1.93	0.81	24	28	277	7.14	9	1.64	4.5	76	
Harmon Lake	2008	2.2	10.6	3.62	0.25	58.4	30.2	2.72	8.9	0.525	1.12	0.905	12	80	316	6.91	8.3	2.24	1.5	303	
Hartman Lake	2009	3	67.8	11.5	12.8	16	114	8.5	5.9	0.975	2.63	7.51	18	2	400	7.58	10.8	0.7	2.25	53	
Hawkeye Lake	2008	2.5	16.1	5.08	1.18	46.6	46.2	4.48	8.6	0.525	2	1.24	18	94	359	6.98	6	3.48	2.85	181	
Hik Lake	2009	1.9	7.31	2.94	0.13	70.8	23	1.8	10.5	0.32	0.78	0.69	26	28	336	6.81	11.8	1.32	1.3	184	

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Holinshead Lake	2008	1.8	11	4.08	0.26	109	31.4	2.76	12.5	0.58	1.27	0.775	18	68	418	6.85	14.3	3.04	1.65	439	
Holly Lake	2008	2	10.6	3.42	0.25	42.4	27.8	2.66	8	0.355	1.13	0.795	16	36	366	7.05	9.1	1.7	1.65	322	
Horwood Lake	2008	1.3	28.8	11.2	0.3	72.6	69.6	6.98	11.9	0.35	2.46	0.73	18	38	404	7.46	10.7	1.6	3.8		
Indian Lake (z4)	2009	3.7	21	7.4	2.71	22.8	58.2	5.36	5.6	0.625	1.58	2.44	16	12	250	7.21	8.4	1.74	2.3	69	25
Island Lake	2009	2.6	9.28	4.82	0.13	45.4	32.8	2.6	6.7	0.26	0.935	0.72	42	80	272	6.88	7	2.12	3.95	87	25
Ivanhoe Lake	2009	2.3	56.7	16.8	1.53	49.4	115	2.08	8.9	0.675	3.94	1.52	22	72	381	7.59	9.8	2.36	3	74	
Jacob Lake	2008	1.9	15.4	5.54	0.55	52	42.6	4.18	9	0.37	1.57	1.08	26	60	460	7.13	15.8	3.46	1.95	884	
Jolly Lake	2008	1.7	24.8	8.28	0.27	60.2	57	6.4	10	0.4	1.9	0.865	30	50	455	7.24	12.2	3.6	1.9	189	
Jubilee Lake	2009	1.9	29.4	5.44	0.23	74.8	37.4	3.58	11.6	0.445	1.25	0.9	10	72	331	7.06	6.2	1.82	1.35	138	32
Kapuskasing River - ds Lost River	2010	1.3	66.8	21	1.86	39	135	19	10.6	0.75	4.73	1.76	36	2	470	7.94	18.9	1.24	2.65	133	
Kashabowie Lake	2008	1.6	12.7	4.5	0.29	67.4	38	3.12	10.4	0.59	1.57	0.945	14	54	424	6.58	9.5	2.62	2.45	277	
Kawaweogama Lake	2008	2.5	8.51	3.04	0.21	57.4	25.2	1.92	8.6	0.5	0.885	0.75	24	42	323	6.93	7.8	1.36	1.45	307	
Kearns Lake	2008	1.8	17.5	5.92	0.28	62.8	44.4	4.7	8.9	0.7	1.59	0.935	18	66	389	7.1	15.3	2.92	2.4	197	
Kekekuab Lake	2008	1.5	15.8	5.84	0.3	79.2	42	3.8	12.3	0.425	1.52	0.91	24	24	579	7.14	14.6	2.88	2.6	368	
Kenogamisis Lake	0																				
Kirkness Lake	2009	2.6	12.2	4.02	0.44	36.2	31.8	3.1	6.8	0.445	0.92	0.965	24	24	298	7.05	13.1	0.72	1.2	266	
Kirkpatrick Lake	2009	6.6	6.03	3.14	0.32	8	26.4	2.04	2.8	0.175	0.685	0.66	2	78	144	6.67	3.2	1.48	4.2		
Kittson Lake	0																				
Klondyke Lake	2009	4	3.94	1.62	0.2	27.6	17.4	0.74	3.9	0.205	0.395	0.825	10	54	175	6.3	3.9	2.18	4.1		
Kokoko Lake	2008	3	17.3	7.22	0.25	14.4	54	4.6	3.4	0.25	1.74	0.765	24	36	252	7.28	6.8	1.06	6.35		
Kukagami Lake	2010	5.5	4.31	3.68	0.81	9.6	32.8	1.5	2.6	0.275	0.915	0.825	10	8	199	6.74	4.3	0.54	8.2		
Kukukus Lake	2009	3.3	19.6	6.08	0.23	33.6	44.8	4.92	7.9	0.62	1.52	1.23	24	20	439	7.29	12.2	0.7	1.7	115	30
Kwagama Lake	2009	7	2.45	1.46	0.12	7.2	15	0.86	2.4	0.22	0.305	0.595	48	80	203	6.48	4.9	0.94	2.95	41	70
Lac des Iles	2008	2	22.2	5.96	1.43	51	59.6	5.68	9.9	0.245	3.02	0.715	18	54	396	7.19	12.2	1.88	3.7	129	
Lac du Milieu	2008	1.8	23.2	7.92	0.41	103	54	5.52	13.7	0.48	2.11	0.985	32	56	570	7.36	16.2	3.74	0.95	94	
Lady Evelyn	2008	2	3.8	2.48	0.19	17.2	25.4	1.16	3.4	0.275	0.79	0.775	18	58	195	6.59	3.9	1.48	5.55	88	
Lake 34	2009	2	2.82	2.18	0.1	84.8	19	0.88	11	0.225	0.475	0.74	32	110	374	6.17	4.8	2.2	2.65	136	
Lake Nipissing	2008	2.2	17.8	6.68	5.34	22.6	72.8	4.46	4.3	0.63	2.2	4.29	32	32	336	7.32	15.2	0.6	6.35	126	
Lake Nosbonsing	2008	2.5	13	4.64	2.94	30.6	53.4	3.3	4.5	0.895	1.49	0.1	22	4	414	7.13	17.4	3.1	5.25	45	
Larder Lake	2008	2.9	33.5	11.1	4.75	19.2	111	8.32	5.8	0.625	3.33	4.19	34	164	319	7.69	8.2	1.64	9.95	93	
Lauzon Lake	2009	3.9	6.36	3.7	1.54	10.4	36.8	1.8	2.6	0.365	0.985	1.32	2	100	178	6.77	4.3	0.7	6.6		
Little Chiblow Lake	2009	4.9	4.57	2.62	0.42	8.8	24.8	1.4	2.8	0.23	0.65	0.81	4	66	168	6.71	4.2	0.72	5		
Little Metionga Lake	2008	1.6	11.9	4.06	0.24	99.6	33.2	3.08	10.8	0.585	1.25	0.95	22	92	396	6.91	12.7	3.4	1.35	179	

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Little North Lake	2008	6	43.8	0	1.01	7.6	102	10.6	3.3	0	0	0	10	46	212	7.38	13.6	3.96	4.65	258	
Little Sandbar Lake	2009	2	15.8	5.8	2.72	52.8	47.4	3.88	7.7	0.565	1.19	2.09	24	12	309	7.22	12.8	3.4	1.7	185	
Little Vermilion Lake	2009	1.3	16.1	5.42	0.25	107	39.2	3.88	13.3	0.655	1.43	0.895	26	30	403	7.07	20.1	2.28	1.15	405	33
Loganberry Lake	2008	1.5	17.7	5.32	0.21	69.8	42.2	5	9	0.64	1.59	0.96	22	46	378	6.99	15.1	3.4	1.5	244	
Longlegged Lake	2009	3.6	16.5	4.94	0.3	26.8	43.2	4.28	7.7	0.76	1.66	1.13	20	24	293	7.18	10.6	0.48	2.15	50	
Lower Shebandowan Lake	2008	2.2	20.4	7.78	2.48	30.4	62.8	5.08	6.9	0.545	1.71	2.02	14	4	338	6.92	7.8	2.58	3.75	65	
Maggotte Lake	2008	1.5	3.98	2.02	0.21	113	17.2	1.22	12.3	0.44	0.67	0.68	14	48	411	5.88	20.8	2.98	1.15	202	
Makobe Lake	2008	2	1.36	1.64	0.2	14.8	19.2	0.42	3	0.245	0.545	0.67	18	30	186	6.24	3.6	0.9	5.1		
Mamainse Lake	2009	10	3.11	2.1	0.22	6.8	19.2	1.16	2.2	0.24	0.325	0.535	36	244	182	6.39	2.4	1.12	3.5		
Mameigwess Lake	2009	5.8	24.8	7.86	0.34	8.4	55.8	6.38	2.4	0.65	1.65	1.39	8	2	157	7.38	5.7	1.44	2.45		
Marie Louise Lake	2008	2.5	91.9	23.8	2	14.8	192	21.7	7.5	0.705	2.38	1.42	8	6	407	7.96	8.6	2.6	4.75	101	
Marks Lake	2008	1.8	18.5	8.1	0.3	81.2	57.4	4.58	11.9	0.675	1.63	0.925	40	102	561	7.09	25.1	2.6	5.85	100	
Matinenda Lake	2009	5.7	4.53	2.58	0.28	10.2	24.2	1.36	2.8	0.22	0.63	0.8	6	92	165	6.63	3.9	1.28	4.7		
Mattagami River	2008	2	22.7	9.02	2.08	72.8	65	5.44	9.9	0.315	2.05	1.83	22	68	367	7.37	9.8	2.18	3.95	129	
Mattawa Lake	2009	2.7	10.1	3.84	0.31	83.2	27.6	2.5	10.5	0.505	0.86	0.895	14	54	394	7	9.2	2.8	1.35	295	30
McCrea Lake	2009	2.7	18.6	6.76	0.12	52.4	44	4.9	9.3	0.415	1.34	0.53	26	88	319	7.08	7.6	0.96	0.9	128	
McGiverin Lake	2009	3.5	3.93	2.46	0.32	26.8	23.8	1.12	4.6	0.28	0.625	0.845	8	108	232	6.51	6.4	1.8	4.65	67	28
McMahon Lake	2009	4.4	12.1	5.26	0.33	7.2	39.4	3.16	2.7	0.345	0.905	0.83	18	54	260	7.1	6.9	0.44	4.65		
Megisan Lake	2009	4.5	12.4	5.3	0.23	42.2	37.4	2.92	8.2	0.23	1.18	0.78	8	84	273	7.06	5.8	1.72	3.6	66	
Mercer Lake	0																				
Metionga Lake	2008	1.6	11.9	4.06	0.24	99.6	33.2	3.08	10.8	0.585	1.25	0.95	22	92	396	6.91	12.7	3.4	1.35	179	
Mijinemungshing	2009	4.5	3.74	2.54	0.12	34.6	20	1.12	5.7	0.205	0.395	0.565	28	128	260	6.51	5.6	0.94	3.1	58	
Miniss Lake	2009	3.4	20	6.74	0.19	52.2	46.8	4.92	9.6	0.535	1.65	0.65	12	64	336	7.24	4.8	1.48	1.3	71	
Mistinikon Lake	2008	2.2	20.3	8.38	0	57	56.6	5.2	10.4	0.22	1.78	0.895	24	52	365	7.25	9.2	2.08	0	79	
Mold Lake	2009	2.3	5.62	1.98	0.17	59	19.6	1.62	9.7	0.52	0.72	0.765	18	44	285	6.55	8.5	2.1	1.55	245	
Moon Lake	2009	5	4.31	2.58	0.28	11.4	23	1.56	2.6	0.195	0.5	0.71	12	80	164	6.6	4.6	1.12	4.6		
Mud Lake	2009	4.5	13.5	4.34	0.27	12.8	34.6	3.56	4.4	0.515	1.03	1.21	8	2	346	7.1	7.3	0.88	2.15	53	
Muskasenda Lake	2008	2.4	45.8	16.6	0.26	55.6	105	10.6	9.1	0.33	3.45	0.805	40	18	400	7.73	11.1	1.7	4.4	133	
Muskeg Lake	2008	1.7	27.1	7.9	0.26	60	60.4	6.94	10.6	0.525	2.74	0.965	28	90	529	7.34	19.4	2.46	1.2	709	
Muskosung Lake	2008	3.3	14.2	5.28	0.76	31.4	48.4	3.44	6.4	0.465	1.6	0.105	16	54	350	7.18	7.5	1.12	5.25	194	
Nalla Lake	2008	4.4	0	3.42	0.22	29.6	29.4	0	8	0.325	1.32	0.775	8	8	303		3.6	2.84	2.7		
Nelson Lake	2008	2	9.65	0	0.39	81	34	2.38	13.3	0	0	0	12	46	477	6.65	12.4	2.82	2.5	168	
Northern Light Lake	2008	2	10.1	0	0.56	63.8	35.4	2.62	11.9	0	0	0	12	88	424	6.66	12	2.28	2.35	86	

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Nungesser Lake	2009	2	12.3	4.32	0.12	84.6	31.6	3.04	11.7	0.46	1.05	0.88	12	48	337	6.98	12.3	2.12	0.85	314	
Obonga Lake	2008	2.1	18.7	6.86	0.31	77.2	49	4.9	10.1	0.595	1.88	0.855	8	108	332	6.65	6.9	3.06	2.1	179	
Okinada Lake	2009	5.3	0.94	1.36	0.11	17.4	17.2	0.7	3.8	0.21	0.44	0.72	14	24	160	5.88	5.2	2.24	4.45		42
Old Woman Lake	2009	3.5	2.41	1.86	0.11	20	16.4	0.88	4.1	0.185	0.31	0.535	26	170	198	6.34	2.4	1.16	2.8	56	30
Onnie Lake	2009	2.6	8.6	2.9	0.19	75.8	26.4	2.28	11.4	0.63	1.02	0.81	14	82	375	6.67	12.6	1.76	1.45	432	50
Opeepeesway Lake	2009	2	19.8	7.44	0.3	78.8	50	4.64	11.7	0.34	1.67	0.73	26	60	491	7.1	9	2.24	2.85	137	
Otatakan Lake	2009	1.2	23	8.48	0.2	134	50.4	5.36	17.6	0.7	1.99	0.62	16	46	428	7.26	11.8	2.06	1	249	26
Pakashkan Lake	2008	1.8	25.7	7.48	0.21	45.2	57.4	6.74	6.6	0.775	2.1	1.12	14	84	398	7.37	12	4.28	1.5	65	
Pakwash Lake	2009	1.5	29.5	9.84	0.99	73.2	70.6	7.26	11	1.01	2.35	1.46	26	66	488	7.36	23.4	1.84	3.15	265	
Panache Lake	2010	5.2	10.2	5.52	6.3	8.8	66.6	3.04	3.3	0.64	1.76	4.39	4	50	202	7.08	4.4	1.24	9.4		
Penassen Lakes	2008	2.8	0	2.36	0.1	33.8	19.8	0	7.6	0.135	0.87	0.57	16	8	350		4	1.68	1.4		
Perrault Lake	2009	2.1	54.3	16.9	2.29	30	119	12.7	8.3	0.96	4.21	2.41	22	6	420	7.81	16.1	1.12	1.85	72	29
Pikangikum Lake	2009	1.9	22.6	6.64	0.21	75.6	51.2	5.94	11	0.53	2.02	0.99	14	76	365	7.03	17	2.34	0.9	302	38
Pinel Lake	2008	1.7	38.3	12.8	0.25	105	81.8	9.38	14	0.29	2.71	0.47	34	34	512	7.55	8.9	1.38	1.15	280	
Point Lake	2009	2.9	11.3	3.16	0.17	61.8	24.4	1.56	8.6	0.2	0.765	0.675	20	96	347	6.57	7.2	2.12	3.2	133	
Premier Lake	2009	2.8	16.7	7.92	0.18	32.2	51.2	4.32	7.3	0.635	1.1	0.665	22	44	297	7.11	8.8	0.72	6.1	45	
Press Lake	2009	2.2	14.8	5.36	0.98	86.8	40.2	3.6	10.9	0.57	1.25	1.33	16	82	398	7.11	9.7	2.92	1.65	401	40
Purdom Lake	2008	1.5	20.4	6.72	0.32	65.6	50.2	4.98	10.9	0.49	2.06	0.85	30	76	464	7.3	12.5	3.04	2.4	161	
Queminico Lake	2009	6	20.9	6.84	0.21	9.2	52	5.36	4.1	0.46	1.61	1.03	78	66	335	7.4	6.7	1.08	3.15		27
Quinn Lake	2009	2.3	4.09	2.6	0.22	50.2	22.4	1.64	8.3	0.2	0.565	0.67	40	72	372	6.25	6.3	2.06	3.5	169	38
Quintet Lake	2009	8	3.76	2.44	0.21	11.4	19.8	1.48	3.4	0.27	0.365	0.505	12	150	184	6.38	3	1	3.25	53	38
Quirke Lake	2009	8.5	4.06	19.3	3.29	5	142	1.5	2.1	1.89	1.47	2.44	8	760	170	6.54	3	1.04	48.4		
Rabbit Lake	2009	3.9	10	7.68	3.14	20.8	71.6	5.56	5	0.325	2.26	2.41	12	90	201	7.2	5.1	1.56	7.15	42	
Radisson Lake	2008	3.4	23.3	8.4	0.22	22.4	62.2	6.14	4.8	0.26	1.89	0.875	8	86	204	7.37	4	1.32	4.25	117	
Ranger Lake	2009	9.5	7.75	3.76	0.56	7.8	31.2	2.12	3.1	0.27	0.955	0.76	6	50	184	6.87	2.1	1	4.7		
Raven Lake	2008	2.4	11.8	4.64	1.31	32.4	48.2	2.9	6.7	0.42	1.55	1.92	18	130	308	7.1	7.4	1.78	5.65	92	
Red Cedar Lake	2009	2.1	15.9	5.74	2.39	40.6	54.8	4.28	7.8	0.4	1.9	2.09	20	72	278	7.1	7.4	1.64	3.65	77	
Red Squirrel Lake	2008	3.5	10.2	4.54	0.24	11	39.4	2.62	2.6	0.215	1.27	0.8	18	44	195	7.17	3.8	0.96	6.25		
Rib Lake	2009	7	17.2	6.6	12	12.4	88.4	4.58	3.5	0.265	1.93	7.46	10	50	161	7.24	4.2	1.38	6.3		
Rice Lake	2009	2.5	13.7	5.06	0.22	40.6	38.6	3.14	8.5	0.275	1.36	0.67	44	26	407	7.1	11.8	1.6	3.1	69	
Ricestalk Lake	2008	1.3	12.3	4.24	0.17	125	31.8	3.42	16.1	0.49	1.56	0.705	18	4	546	6.76	14.4	2.44	0.9	377	
Richardson Lake	2009	4.7	33.4	9.86	0.34	18	72.4	8.46	5.9	0.725	2.37	1.11	32	10	282	7.45	11.6	0.84	2.3		
Rodd Lake	0																				

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Rollo Lake	2009	5.9	42	14.5	0.31	27.8	92	10.6	6.9	0.35	2.66	0.85	18	84	298	7.46	7.2	1.72	3.55		
Rome Lake	2009	3	3.76	2.2	0.19	45	21.2	0.8	7.2	0.2	0.65	0.715	8	56	278	6.13	7.7	1.68	5.15	199	56
Rossmere Lake	2009	1.8	4.33	2.64	0.36	37.4	25	1.08	6.4	0.3	0.695	0.885	8	108	367	6.57	8.9	1.84	4.45	112	32
Round Lake	2008	2.1	39.3	14.5	8	41.8	132	10	9.1	0.87	3.54	6.34	12	204	381	7.48	24.1	1.64	10.6	48	
Rushbrook Lake	2010	4.5	7.43	3.38	0.26	10.4	29.2	2.4	3	0.415	0.855	1.04	8	2	191	6.99	8.3	0.84	5.05	74	
Saddle Lake	2009	4	4.81	2.88	0.23	22.8	23.2	1.4	5.3	0.23	0.56	0.725	42	68	325	6.58	6.1	1.06	3.75	80	30
Sandison Lake	2008	1.3	4.62	1.94	0.16	90.6	16.6	1.2	10.5	0.465	0.66	0.615	18	50	320	6.53	7.3	2.16	1.2	108	
Sandstone Lake	2008	3	32	9.44	2.71	33.2	85.8	8.22	6.9	0.4	1.89	1.4	10	96	346	7.41	10.1	2.86	4.35	154	
Savant Lake	2009	4.4	17.5	6.52	0.18	22.4	42.2	4.68	7.3	0.445	1.01	0.545	10	40	238	7.07	6.1	0.72	1.65		
Saymo Lake	2009	6.1	7.36	3.4	0.25	12	27.8	1.96	3.9	0.225	0.765	0.71	8	56	179	6.81	3.2	1.12	3.85		
Shack Lake	2009	3.3	4.53	2.66	0.11	56.4	23.8	1.36	7.2	0.255	0.65	0.65	68	64	300	6.53	8.5	2	3.35	164	70
Shelden Lake	2009	5.5	7.34	3.72	0.33	16.4	31	2.08	3.5	0.315	0.775	0.93	12	164	232	6.76	4.1	1.66	4.4	58	26
Silcox Lake	2009	1.7	24.2	7.04	0.13	66.2	53.4	6.24	9.8	0.56	2	0.895	34	46	404	7.2	21.6	1.2	1	392	56
Sill Lake	0																				
Silver Lake	2009	2.3	14.2	6.66	0.11	79	36.4	3.58	12	0.355	0.63	0.415	24	38	348	7	9	1.3	1.05	94	
Smye Lake	2009	1.7	4.44	2.16	0.13	90.4	17.6	1.24	12.8	0.515	0.575	0.51	20	60	322	6.36	7.7	2.02	0.75	327	
South Branch Lake	2009	3.5	5.38	3.02	0.15	47	24	1.44	7.2	0.17	0.655	0.685	24	90	324	6.55	5.9	1.92	3.45	122	
Sparkling Lake	2008	2.5	5.54	2.3	0.32	47.2	21.8	1.64	8	0.43	0.765	0.75	12	56	303	6.54	6.2	2.24	1.5	144	
Spinweb Lake	2009	3.5	6.08	3.14	0.24	14.8	26.4	1.86	3.6	0.32	0.615	0.82	30	112	232	6.82	4.2	1.36	4.55	45	28
Spruce Lake	2009	1.4	23.6	8.2	0.14	124	53.6	5.4	17.3	0.805	2.03	0.715	34	58	465	7.24	17	1.56	0.85	229	
Squeers Lake	2008	7	13.5	0	1.97	9	42.8	3.98	3.6	0	0	0	12	30	271	6.87	13.5	0.48	3.9		
St. Anthony Lake	2008	2.8	17.9	6.22	0.18	17.4	52.4	4.24	6.2	0.39	1.83	0.885	16	4	299	7.47	5.6	0.32	4.75	64	
Sunbow Lake	2008	2.7	13.4	0	0.74	37.6	41.6	3.76	8.6	0	0	0	14	102	442	6.86	12.7	1.84	2.4		
Temagami Lake	2008	5	11.8	6.7	0.93	5	59.6	3.08	3	0.335	1.88	1.15	12	48	177	6.64	5.1	0.46	11.6		
Titmarsh Lake	2008	3.3	10	0	0.29	20.8	35.2	3.18	6	0	0	0	8	76	244	6.78	6	1.84	2.5	184	
Tom Lake	2009	3.4	5.57	2.48	0.21	27.8	23	1.6	7.2	0.33	0.695	0.835	16	38	262	6.66	4.8	1.52	2.05	43	
Towers Lake	2009	2.5	56.3	9.82	0.16	57.4	61.6	7	11	0.62	1.75	0.925	20	56	433	7.32	16.7	2	1.55	97	28
Trout Lake	2008	5.1	11.9	5.2	14.8	11.6	99.2	3.36	3.1	0.71	1.54	5.89	6	166	225	6.98	4.3	0.96	6.25	229	
Turtle Lake	2010	2.7	2.18	1.74	0.32	13.6	15.2	0.92	3.7	0.235	0.43	0.545	12	22	270	6.44	7.4	0.82	3.45	60	
Upper Hunters Lake	2008	4	5.25	2.18	0.15	59.8	20.2	1.46	6.9	0.155	0.835	0.495	32	22	325	6.23	8.3	1.56	1.75	87	
Upper Pancake Lake	2009	5.2	4.24	2.5	0.27	18	20.6	1.4	4.2	0.225	0.38	0.645	46	166	282	6.45	4.4	1.28	3.05	103	39
Victoria Lake	2009	2.8	17.6	6.02	0.27	28.4	42.8	4.44	6.4	0.51	1.3	1.04	12	20	271	7.23	12.6	0.84	2.2		
Wabaskang Lake	2009	2.5	52	16.5	1.81	26.8	113	12.5	8.6	0.975	4.01	2.1	22	4	424	7.61	19.3	0.78	1.8	87	36

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Wabinosh Lake	2008	1.7	24.3	8.08	0.8	62.6	58.4	6.08	8.8	0.695	2.1	1.24	18	88	306	6.95	9.2	3.18	1.8	167	
Wakomata Lake	2009	5	6.3	3.4	0.61	3.4	30.2	1.74	2	0.235	0.76	0.805	2	156	122	6.76	2.3	1.18	5.05		
Walker Lake	2009	3.5	20.9	5.74	5.46	11.8	66.2	2.76	3.4	0.605	1.66	3.96	14	74	194	7	5.7	1.06	9.5		
Walotka Lake	2008	3.9	11.3	3.86	0.23	25.6	34.8	3.2	5.7	0.315	1.55	0.695	8	116	256	6.96	3.9	1.72	2.95	161	
Wapesi Lake	2009	1.3	21.7	7.82	0.13	137	50.4	5.2	15.8	0.62	1.83	0.715	28	52	457	7.15	19	2.04	1	291	
Wapikaimaski Lake	2008	1.7	8.57	3.22	0.22	81.4	26	2.12	10	0.535	0.96	0.78	22	68	350	6.84	8.6	2.2	1.3	137	
Ward Lake	2009	5	14.6	2.62	0.18	26.6	22	1.08	5.5	0.24	0.52	0.67	24	86	284	8.67	4.3	1.24	4	111	35
Wasaksina Lake	2008	1.5	16.7	6.48	0.25	23.8	51	4.06	6.1	0.225	1.97	0.735	30	12	318	7.32	8.9	0.8	6.05		
Waweig Lake	2008	4	21.7	6.86	1	38.8	54	5.64	6.5	0.79	1.87	1.39	14	68	279	6.78	11.6	3.12	1.65	184	
Weckstrom Lake	0																				
Weewullee Lake	2008	3	119	33.4	0.13	12.8	229	0	4.8	0.665	6.3	0.64	50	140	335	8.09	0	2.16	1	255	
Weikwabinonaw Lake	2008	1.6	10.1	3.98	0.63	69.2	34	2.84	10.4	0.34	1.26	1.05	16	98	441	6.77	10.1	2.6	2.25	188	
Whiskey Lake	2009	6	5.04	15.4	2.48	11.6	112	1.56	2.7	1.56	1.23	2.07	10	556	232	6.73	3.1	1.08	36.4	44	
Whitefish Lake - Expanded Reservoir	0																				
Wicksteed Lake	2009	2.3	8.67	3.32	0.17	60.4	28.6	2	9	0.355	1.08	0.545	22	62	306	6.95	7.2	1.68	3.9	106	
Wintering Lake	2009	2.3	21.7	7.86	1.07	79.6	52.6	5.28	13.8	0.57	1.84	1.09	32	38	500	7.24	15.9	1.2	1.3	186	
Wyder Lake	2009	3.6	7.17	2.66	0.23	28.2	24.6	2.06	6.7	0.565	0.92	0.75	8	66	243	6.65	5.8	0.9	2.3	80	
Dungaree, Indian Lake	2008	2.5	32.8	13.2	20.7	67.8	144	7.92	11.6	0.27	2.99	12.1	22	26	451	7.48	10.1	1.78	3.65	122	
AVERAGE		**3.28**	**17.66**	**6.05**	**1.08**	**45.24**	**49.64**	**4.17**	**7.69**	**0.44**	**1.47**	**1.23**	**19.36**	**71.72**	**323.31**	**7.00**	**9.20**	**1.77**	**3.64**	**166.21**	**36.92**
STANDARD DEVIATION		**1.79**	**16.15**	**4.67**	**2.49**	**32.23**	**32.13**	**3.24**	**3.74**	**0.29**	**0.98**	**1.47**	**11.79**	**74.74**	**103.49**	**0.40**	**5.20**	**0.88**	**4.31**	**124.14**	**11.50**

If year sampled = 0, then water chemistry was not sampled for this lake.

Appendix D. Fish Standardized Total Mercury Concentrations

The standard total length according to species (Table 1) was determined by evaluating the population averages for all species (Fig. 1) as well as the standard used by the Ministry of the Environment (MOE) (Gewurtz *et al*. 2010). Standards for brook trout and smallmouth base were based on the data collected from the lakes of this study as no standard was available from the MOE. The standardized mercury concentrations for each species from individual lakes are presented in Table 2.

Table D.1. Fish Names and Standard Lengths

Species Common Name (MNR CODE)	Species Scientific Name	Standard Length (mm)
Walleye (SPC 334)	*Sander vitreus*	500
Northern Pike (SPC 131)	*Esox lucius*	650
Lake Trout (SPC 081)	*Salvelinus namaycush*	600
Brook Trout (SPC 080)	*Salvelinus fontinalis*	300
Smallmouth Bass (SPC 316)	*Micropterus dolomieu*	400

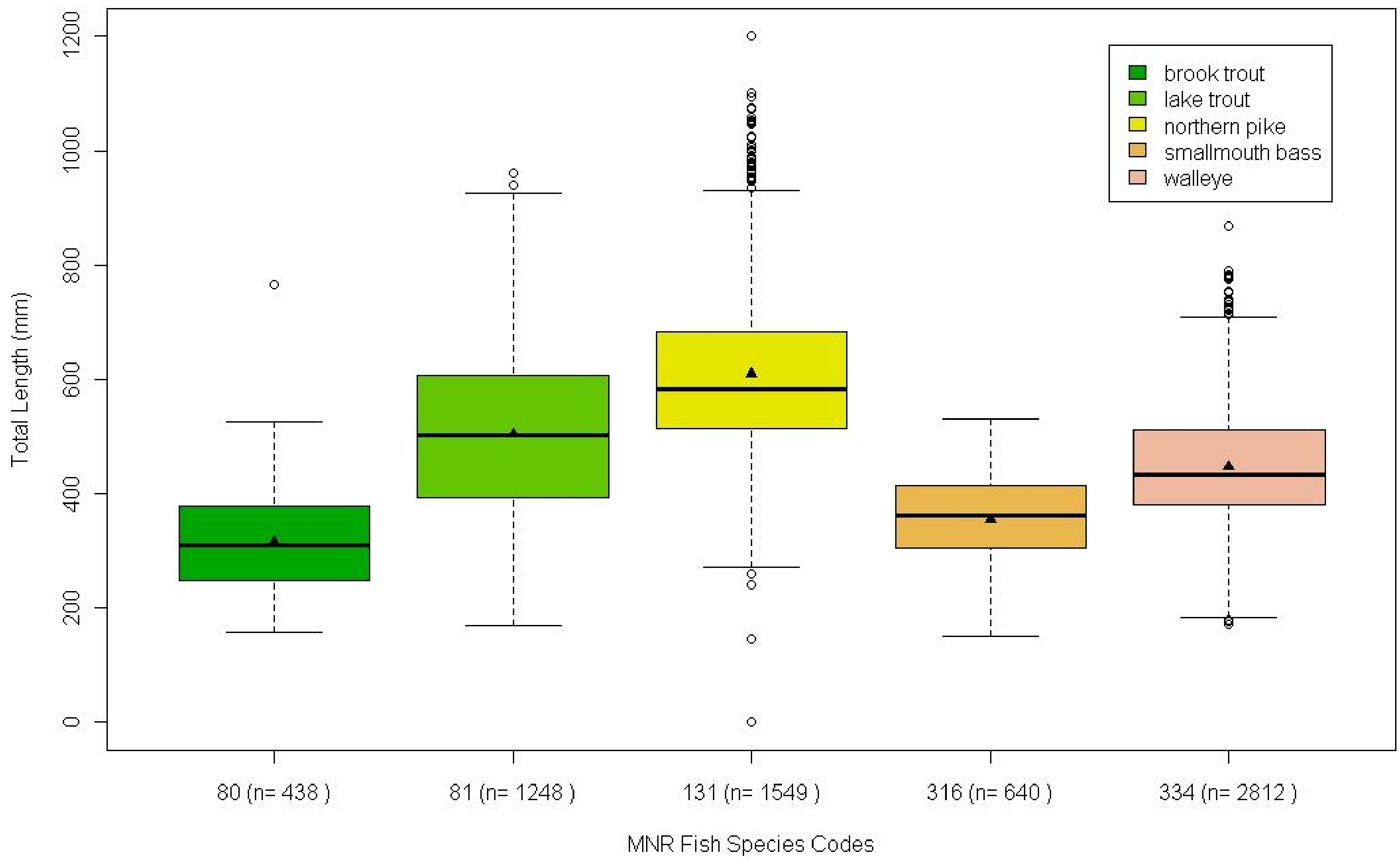

Figure D.1.Total Lengths of Sampled Fish according to Species. The horizontal black line represents the median, the boxes indicate the inter-quartile range (IQR) and the whiskers represent 1.5 times the IQR from the first and third quartile, hollow circles represent data outside of the (Q1-1.5* IQR, Q3+1.5*IQR). The mean is represented by the black triangle and the number of samples (n) represents the number of lakes in that category.

Table D.2. Standard Mercury Levels according to Lake and Species determined for a standard length by power regression analysis

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Addie Lake	150496910	131	0.7601	0.7258	0.0885	0.002	9	0.6935205
Addie Lake	150496910	334	0.8718	0.8611	0.1005	0.000	14	0.514687
Aileen Lake	700945566	316	0.5552	0.5235	0.1506	0.001	16	0.4423357
Aileen Lake	700945566	334	0.7947	0.7833	0.1248	0.000	20	0.5671018
Amik Lake	400315396	131	0.4733	0.4074	0.1356	0.028	10	1.1194573
Amik Lake	400315396	334	0.5389	0.5132	0.1041	0.000	20	1.308949
Anima Nipissing Lake	700944883	334	0.8010	0.7930	0.1014	0.000	29	0.4106146
Anima Nipissing Lake	700944883	81	0.7660	0.7400	0.1327	0.000	11	0.2796177
Anima Nipissing Lake	700944883	316	0.6770	0.6590	0.0853	0.000	20	0.433169
Anjigami Lake	51474072	131	0.9344	0.9213	0.0814	0.000	7	1.8317707
Anjigami Lake	51474072	334	0.8565	0.8206	0.0590	0.008	6	1.3250926
Anvil Lake	800493548	81	0.9144	0.9037	0.1258	0.000	10	2.3333677
Anvil Lake	800493548	131	0.8164	0.8023	0.1152	0.000	15	1.4303722
Arc Lake	1300268867	131	0.6885	0.6625	0.1053	0.000	14	1.0514309
Arc Lake	1300268867	334	0.6407	0.6207	0.1108	0.000	20	1.3227733
Arrowroot Lake	750779260	131	0.7137	0.6779	0.1232	0.002	10	0.8640284
Astonish Lake	800494606	80	0.2210	0.1720	0.1183	0.049	18	0.3065966
Athelstane Lake	151208180	131	0.8647	0.8421	0.1037	0.001	8	0.3042688
Athelstane Lake	151208180	316	0.6057	0.5728	0.1166	0.001	14	0.2969926
Athelstane Lake	151208180	334	0.7773	0.7649	0.1546	0.000	20	0.333821
Bark Lake	500957559	81	0.7721	0.7265	0.0810	0.009	7	0.4372039
Bark Lake	500957559	131	0.5676	0.5059	0.0910	0.019	9	0.6479315
Bark Lake	500957559	334	0.7532	0.7121	0.1222	0.005	8	0.5611084
Batwing Lake	150491421	131	0.8803	0.8683	0.0915	0.000	12	0.5491211
Batwing Lake	150491421	334	0.8333	0.8250	0.0996	0.000	22	0.542987
Bawden Lake	1300844677	131	0.4924	0.4290	0.1355	0.024	10	0.9329485
Bawden Lake	1300844677	334	0.7494	0.7355	0.1023	0.000	20	1.0028461
Beak Lake	800345235	131	0.4391	0.3690	0.1390	0.037	10	1.1940519
Bear Lake	700946278	131	0.6158	0.5677	0.1304	0.007	10	0.6603719
Beatty Lake	750778227	131	0.7630	0.7037	0.0898	0.023	6	0.2656312
Bedivere Lake	150473650	131	0.6050	0.5878	0.1415	0.000	25	1.1412887

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Bedivere Lake	150473650	334	0.6562	0.6382	0.1473	0.000	21	1.1146209
Bertaud Lake	1300268823	131	0.7648	0.7060	0.1404	0.023	6	1.135526
Big Basswood Lake	800495003	81	0.7424	0.7356	0.0896	0.000	40	0.1917845
Big Sandy Lake	1300269298	131	0.4247	0.3527	0.2105	0.041	10	0.3085838
Birch Lake	1100449655	334	0.6017	0.5655	0.1333	0.002	13	0.8257912
Black Sturgeon Lake	750778612	316	0.8283	0.7854	0.0643	0.012	6	0.5555932
Black Sturgeon Lake	750778612	334	0.7872	0.7606	0.0804	0.001	10	0.8413176
Bluffy Lake	1100550507	131	0.8220	0.7997	0.1346	0.000	10	1.0003669
Bluffy Lake	1100550507	334	0.6782	0.6603	0.1221	0.000	20	0.8456351
Blunder Lake	151208182	131	0.9605	0.9473	0.0607	0.003	5	0.5924075
Blunder Lake	151208182	334	0.8976	0.8919	0.1037	0.000	20	0.6856699
Bright Lake	800495087	131	0.8262	0.8128	0.1005	0.000	15	0.4073917
Bright Lake	800495087	334	0.6326	0.6043	0.1463	0.000	15	0.5026995
Brightsand Lake	150901325	131	0.6727	0.6181	0.2196	0.013	8	0.9475719
Bukemiga Lake	150901230	81	0.7368	0.7040	0.1294	0.001	10	0.8041889
Bukemiga Lake	150901230	334	0.3721	0.3525	0.1232	0.000	34	1.0244276
Burchell Lake	150485798	81	0.5547	0.5251	0.1557	0.001	17	0.5861812
Burchell Lake	150485798	334	0.5738	0.5501	0.1173	0.000	20	0.9014529
Burntbush Lake	1000537723	131	0.8612	0.8414	0.1112	0.000	9	0.6559886
Burntbush Lake	1000537723	334	0.5846	0.5016	0.1437	0.045	7	0.6528109
Bury Lake	1300269028	334	0.3162	0.2782	0.1311	0.010	20	1.0738145
Cadden Lake	700947355	334	0.8824	0.8768	0.0876	0.000	23	1.7259846
Canyon Lake	1150542356	334	0.9033	0.8979	0.0704	0.000	20	0.9255677
Carling Lake	1300268905	81	0.5210	0.4611	0.1329	0.018	10	0.5284432
Carling Lake	1300268905	131	0.7372	0.6934	0.1440	0.006	8	0.7476931
Carling Lake	1300268905	334	0.5924	0.5610	0.1317	0.001	15	1.2474667
Carmen Lake	200296197	80	0.6960	0.6790	0.1155	0.000	20	0.2594699
Cassels Lake	67920422	81	0.6225	0.6053	0.1185	0.000	24	0.665577
Cassels Lake	67920422	316	0.7892	0.7775	0.1230	0.000	20	0.5763863
Cassels Lake	67920422	334	0.8990	0.8930	0.1025	0.000	19	0.9331344
Cecil Lake	400315735	81	0.3797	0.3453	0.1339	0.004	20	0.5048195
Cecil Lake	400315735	131	0.7459	0.7290	0.1013	0.000	17	0.4536236

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Circle Lake	750778738	334	0.6890	0.6696	0.0834	0.000	18	0.4687393
Clay Lake	1150542289	131	.547	.490	0.1058	0.014	10	3.1672279
Clear (Watt) Lake	700946921	316	0.7558	0.7422	0.0617	0.000	20	0.4199347
Clear (Watt) Lake	700946921	334	0.6953	0.6735	0.1913	0.000	16	0.5970944
Coli Lake	1100449750	131	0.6896	0.6275	0.0729	0.021	7	0.2632787
Coli Lake	1100449750	334	0.7257	0.6952	0.0763	0.001	11	0.2685495
Confusion Lake	1100450204	131	0.5785	0.5317	0.1441	0.007	11	1.0931867
Conifer Lake	1150542006	131	0.9557	0.9446	0.0567	0.001	6	0.7373695
Conifer Lake	1150542006	334	0.7390	0.7153	0.1363	0.000	13	0.8835285
Constance Lake	800494861	131	0.7252	0.7002	0.1345	0.000	13	0.4992101
Crayfish Lake	150481682	131	0.3757	0.3063	0.1821	0.045	11	0.9735991
Crayfish Lake	150481682	334	0.7844	0.7701	0.1321	0.000	17	0.8011185
Crevasse Lake	150450375	131	0.8896	0.8758	0.0692	0.000	10	0.5790648
Cross Lake (Torrington-Yates)	700945513	131	0.9748	0.9664	0.0379	0.002	5	0.3653191
Cross Lake (Torrington-Yates)	700945513	316	0.7338	0.7005	0.1132	0.002	10	0.24432
Cross Lake (Torrington-Yates)	700945513	334	0.7068	0.6905	0.1604	0.000	20	0.35208
Cry Lake	150448814	81	0.6780	0.6579	0.1377	0.000	18	0.6760015
Crystal Lake	400315552	131	0.7355	0.7025	0.1204	0.002	10	1.0585655
Deer Lake	700946030	334	0.4710	0.4416	0.1409	0.001	20	0.5272213
Deer Lake	700946030	131	0.5329	0.5070	0.1006	0.000	20	0.4509069
Delaney Lake	1150542280	81	0.6657	0.6471	0.1367	0.000	20	0.4184906
Delaney Lake	1150542280	131	0.5140	0.4600	0.1586	0.013	11	0.4650781
Delaney Lake	1150542280	316	0.5222	0.4881	0.0752	0.002	16	0.5727192
Dog Lake	150901352	334	0.4284	0.4012	0.1286	0.001	23	1.1979381
Dumbell Lake	200295827	334	0.3947	0.3591	0.0927	0.004	19	0.4894125
Dumbell Lake	200295827	131	0.7745	0.7626	0.0744	0.000	20	0.7729461
Duncan Lake	200295858	131	0.7671	0.7205	0.0826	0.010	7	0.898964
Duncan Lake	200295858	334	0.8926	0.8819	0.1068	0.000	12	1.279337
Duval Lake	800494168	81	0.8770	0.8700	0.1091	0.000	20	0.378935
Elbow Lake	750777599	131	0.6423	0.5912	0.1010	0.009	9	1.0834354
Elbow Lake	750777599	334	0.8407	0.8208	0.0801	0.000	10	0.8385038
Emerald Lake	700945497	81	0.7012	0.6846	0.1273	0.000	20	0.203675

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Empire Lake	150901331	131	0.9770	0.9712	0.0672	0.000	6	0.7519724
Empire Lake	150901331	334	0.8871	0.8791	0.0876	0.000	16	0.6344114
Endikai Lake	800494484	131	0.8594	0.8125	0.1691	0.023	5	0.4728291
Endikai Lake	800494484	271	0.5269	0.4905	0.0949	0.002	15	1.168442
Endikai Lake	800494484	316	0.5718	0.5388	0.1030	0.001	15	0.6197818
Endikai Lake	800494484	334	0.8723	0.8648	0.1067	0.000	19	1.1806564
Expanse Lake	1300269085	131	0.6872	0.6480	0.0936	0.003	10	0.8609836
Expanse Lake	1300269085	334	0.2971	0.2580	0.1266	0.013	20	0.7553738
Fitchie Lake	1300268891	131	0.6195	0.5719	0.1369	0.007	10	0.580964
Fitchie Lake	1300268891	334	0.6789	0.6611	0.1114	0.000	20	0.6747039
Frank Lake	750777552	131	0.4293	0.3580	0.1209	0.040	10	0.4539845
Frank Lake	750777552	334	0.9064	0.8947	0.0639	0.000	10	0.3950951
Galloway Lake	800493200	80	0.5502	0.5237	0.1103	0.000	19	0.7111211
Gamitagama Lake	51474236	80	0.6880	0.6360	0.0733	0.011	8	0.1238435
Gamitagama Lake	51474236	81	0.7520	0.7107	0.0595	0.005	8	0.4065818
Garden Lake	800494073	80	0.8411	0.8014	0.0452	0.010	6	0.3716971
Garden Lake	800494073	81	0.7918	0.7501	0.0954	0.007	7	0.5050012
Gavor Lake	800493445	80	0.3262	0.2908	0.0911	0.007	21	0.3237622
Geiger Lake	800366788	81	0.7838	0.7672	0.0693	0.000	15	1.1155241
Gennis Lake	150462792	131	0.4490	0.4239	0.1174	0.000	24	0.47583
Gong Lake	800493500	81	0.6952	0.6675	0.0885	0.000	13	0.796896
Greenbush Lake	1300268813	131	0.6507	0.6070	0.1003	0.005	10	0.4326364
Greenbush Lake	1300268813	334	0.4185	0.3862	0.1190	0.002	20	0.5244613
Grew Lake	150901347	131	0.9032	0.8911	0.0615	0.000	10	0.5163556
Grew Lake	150901347	334	0.7495	0.7363	0.1098	0.000	21	0.5955531
Gull Lake	800493167	80	0.7960	0.7450	0.0364	0.017	6	0.4991095
Gull Lake	800493167	81	0.2176	0.1742	0.0629	0.038	20	0.6783
Gull Lake	800493167	334	0.8813	0.8747	0.0616	0.000	20	0.8947558
Gustauson Lake	400315672	81	0.3648	0.3296	0.1341	0.005	20	0.2171255
Hailstone Lake	1100449821	334	0.7303	0.7154	0.0935	0.000	20	1.5365568
Hammell Lake	1100449877	81	0.8694	0.8368	0.0956	0.007	6	1.234413
Hammell Lake	1100449877	131	0.9939	0.9919	0.0209	0.000	5	0.8283071

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Hammell Lake	1100449877	334	0.5610	0.5272	0.1501	0.001	15	0.9643339
Hangstone Lake	700945634	131	0.7592	0.7351	0.0809	0.000	12	0.6023931
Hangstone Lake	700945634	316	0.7117	0.6829	0.0639	0.001	12	0.6016186
Hangstone Lake	700945634	334	0.8127	0.8023	0.0777	0.000	20	0.7863681
Harmon Lake	150446690	334	0.7940	0.7825	0.1031	0.000	20	0.8221342
Hartman Lake	400315535	334	0.6956	0.6787	0.0868	0.000	20	0.2320487
Hawkeye Lake	151326631	316	0.5188	0.4707	0.1120	0.008	12	0.5962035
Hawkeye Lake	151326631	334	0.6579	0.6416	0.1211	0.000	23	0.971988
Hik Lake	1300269089	131	0.7799	0.7524	0.0957	0.001	10	0.7049521
Holinshead Lake	150455227	131	0.9318	0.9232	0.0762	0.000	10	0.9148321
Holinshead Lake	150455227	334	0.8247	0.8150	0.0711	0.000	20	0.9106923
Holly Lake	150901343	131	0.6294	0.5764	0.1951	0.011	9	0.4109512
Holly Lake	150901343	334	0.2961	0.2642	0.0920	0.006	24	0.4209554
Horwood Lake	950283324	131	0.4278	0.3758	0.1299	0.015	13	0.9862243
Horwood Lake	950283324	334	0.9135	0.9087	0.0769	0.000	20	1.8362907
Indian Lake (Zone4)	400704878	131	.510	.449	0.1038	0.020	10	0.5426
Indian Lake (Zone4)	400704878	316	0.8078	0.7597	0.0692	0.015	6	0.4292638
Indian Lake (Zone4)	400704878	334	0.5266	0.4902	0.1264	0.002	15	0.4922718
Indian Lake (Zone8)	950283369	131	.928	.924	0.0534	0.000	20	1.0765603
Island Lake	200296178	80	0.5814	0.5581	0.1272	0.000	20	0.2241909
Ivanhoe Lake	450603811	131	0.6136	0.5908	0.1215	0.000	19	0.9067597
Ivanhoe Lake	450603811	334	0.8723	0.8652	0.0912	0.000	20	0.8719762
Jolly Lake	151209168	334	0.7742	0.7420	0.0614	0.002	9	0.8532158
Jubilee Lake	1100449943	131	0.8808	0.8659	0.1027	0.000	10	0.9508576
Jubilee Lake	1100449943	334	0.5941	0.5715	0.1501	0.000	20	1.2017435
Kapuskasing River - ds Lost River	301085944	334	0.7018	0.6843	0.0784	0.000	19	1.2047177
Kashabowie Lake	150479364	316	0.7094	0.6609	0.1144	0.009	8	0.5332891
Kashabowie Lake	150479364	334	0.4627	0.4359	0.1379	0.000	22	0.7331023
Kawaweogama	150901125	131	0.7212	0.6863	0.1342	0.002	10	0.7108897
Kawaweogama	150901125	334	0.5664	0.5423	0.1099	0.000	20	0.7348156
Kearns Lake	150901339	131	0.8288	0.8002	0.1184	0.002	8	0.5270191
Kekekuab Lake	150489325	131	0.5015	0.4391	0.1903	0.022	10	0.5514919

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Kekekuab Lake	150489325	334	0.8010	0.7900	0.1407	0.000	20	0.4819657
Kenogamissi Lake	950283143	131	0.6673	0.6371	0.1274	0.001	13	0.951954
Kenogamissi Lake	950283143	334	0.7856	0.7744	0.1132	0.000	21	1.3475079
Kirkness Lake	1100449589	334	0.4993	0.4715	0.0842	0.000	20	0.517479
Kirkpatrick Lake	800494290	81	0.7539	0.7386	0.1654	0.000	18	0.387832
Kokoko Lake	700945293	81	0.6937	0.6746	0.1278	0.000	18	0.5070613
Kokoko Lake	700945293	131	0.8344	0.7930	0.1109	0.011	6	0.426078
Kokoko Lake	700945293	316	0.8016	0.7906	0.0699	0.000	20	0.3366626
Kokoko Lake	700945293	334	0.7594	0.7460	0.0776	0.000	20	0.4391319
Kukagami Lake	500958245	271	0.8189	0.7736	0.0691	0.013	6	0.1359091
Kukukus Lake	400315402	131	0.5385	0.4808	0.1444	0.016	10	0.638546
Kukukus Lake	400315402	334	0.3640	0.3287	0.1212	0.005	20	0.7700513
Kwagama Lake	51474393	80	0.5110	0.4734	0.1362	0.003	15	0.1157405
Lac du Milieau	150474466	334	0.7584	0.7398	0.1231	0.000	15	0.3171948
Lady Evelyn Lake	200258020	334	0.7363	0.7217	0.1590	0.000	20	0.8124642
Lake Temagami	700945140	81	0.5815	0.5606	0.1662	0.000	22	0.2929736
Lake Temagami	700945140	316	0.7265	0.7113	0.1250	0.000	20	0.3179869
Lake Temagami	700945140	334	0.5555	0.5308	0.1456	0.000	20	0.2609378
Larder Lake	200414151	81	0.8273	0.8182	0.0998	0.000	21	0.8176385
Larder Lake	200414151	131	0.7688	0.7110	0.0753	0.022	6	0.387384
Larder Lake	200414151	316	0.8082	0.7975	0.0951	0.000	20	0.4012629
Larder Lake	200414151	334	0.8216	0.8117	0.0968	0.000	20	0.3920747
Lauzon Lake	800495169	81	0.6630	0.6208	0.1724	0.004	10	0.149583
Lauzon Lake	800495169	316	0.9565	0.9517	0.0787	0.000	11	0.4182812
Little Chiblow Lake	800494974	81	0.7490	0.7131	0.1046	0.003	9	0.2875751
Little Metionga	150454298	131	0.6607	0.6183	0.1537	0.004	10	0.8205079
Little Metionga	150454298	334	0.7229	0.6537	0.1151	0.032	6	1.3609804
Little North Lake	151208288	81	0.5996	0.5774	0.0876	0.000	20	0.240787
Little North Lake	151208288	316	0.7223	0.6876	0.0927	0.002	10	0.2435646
Little North Lake	151208288	334	0.6273	0.6066	0.1590	0.000	20	0.3009873
Little Sandbar Lake	400315863	131	0.6999	0.6624	0.0987	0.003	10	0.4572127
Loganberry Lake	150901340	131	0.6261	0.5845	0.2225	0.004	11	0.6011643

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Loganberry Lake	150901340	334	0.5062	0.4803	0.0924	0.000	21	0.6870551
Longlegged Lake	1100550520	81	0.6859	0.6617	0.1520	0.000	15	0.9482679
Longlegged Lake	1100550520	131	0.4551	0.3869	0.1115	0.032	10	0.5115136
Longlegged Lake	1100550520	334	0.4132	0.3806	0.1257	0.002	20	0.7728408
Lower Shebandowan Lake	151326646	334	0.9688	0.9643	0.0773	0.000	9	0.5357653
Maggotte Lake	150901245	334	0.7042	0.6878	0.0942	0.000	20	1.4465571
Makobe Lake	700944596	81	0.4663	0.4367	0.0945	0.001	20	0.6528728
Mamainse Lake	800493561	80	0.6700	0.5876	0.0699	0.046	6	0.048808
Mamainse Lake	800493561	81	0.7175	0.6469	0.0815	0.033	6	0.130236
Mameigwess Lake	400315677	131	0.5959	0.5453	0.1688	0.009	10	0.2993317
Mameigwess Lake	400315677	334	0.7382	0.7244	0.1428	0.000	21	0.4680095
Marie Louise Lake	150493619	316	0.7246	0.6971	0.0599	0.000	12	0.4081617
Matinenda Lake	800494887	81	0.6348	0.6133	0.1525	0.000	19	0.6156317
Matinenda Lake	800494887	316	0.8615	0.8538	0.0922	0.000	20	0.4399646
Mattawa Lake	150452866	131	0.7940	0.7425	0.1351	0.017	6	0.8228783
Mattawa Lake	150452866	334	0.7935	0.7820	0.0745	0.000	20	0.7884171
McCrea Lake	1300268818	334	0.4889	0.4524	0.0939	0.003	16	0.68761
McGiverin Lake	800495078	81	0.5311	0.4724	0.1409	0.017	10	0.6084622
McMahon Lake	800494627	81	0.7824	0.7551	0.1772	0.001	10	0.4157175
Megisan Lake	800493139	81	0.7513	0.7400	0.1089	0.000	24	0.7687703
Mercer Lake	700946662	131	0.7347	0.7207	0.0678	0.000	21	0.9523443
Metionga Lake	150901327	131	0.8072	0.7430	0.1986	0.038	5	0.9675559
Metionga Lake	150901327	334	0.7150	0.7075	0.1082	0.000	40	1.0233725
Mold Lake	400315160	81	0.8385	0.8270	0.1200	0.000	16	2.2371534
Mud Lake	400315734	131	0.8049	0.7805	0.1308	0.000	10	0.3944589
Mud Lake	400315734	334	0.8409	0.8320	0.1035	0.000	20	0.5602018
Muskasenda Lake	950283305	81	0.5859	0.5169	0.2036	0.027	8	0.5936692
Muskasenda Lake	950283305	131	0.3852	0.3511	0.1575	0.003	20	0.628318
Muskeg Lake	150471011	131	0.8227	0.7932	0.1359	0.002	8	0.3953783
Muskosung Lake	700946020	334	0.5992	0.5792	0.1795	0.000	22	0.6131119
Nalla Lake	150485050	81	0.8624	0.8395	0.1229	0.001	8	0.6953571
Nelson Lake	150490430	131	0.7211	0.7056	0.1425	0.000	20	0.5383332

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Nelson Lake	150490430	334	0.8195	0.8088	0.0688	0.000	19	0.5242444
Northern Light Lake	150901585	81	0.7878	0.7453	0.1095	0.008	7	0.2622756
Northern Light Lake	150901585	131	0.6325	0.6121	0.1446	0.000	20	0.4086377
Northern Light Lake	150901585	316	0.6339	0.6034	0.0927	0.001	14	0.3401339
Northern Light Lake	150901585	334	0.4089	0.3778	0.1140	0.002	21	0.4144833
Nosbonsing Lake	700946387	131	0.7025	0.6777	0.1413	0.000	14	0.5541588
Nosbonsing Lake	700946387	334	0.5318	0.4983	0.0994	0.001	16	0.3920536
Nungesser Lake	1100449608	131	0.8055	0.7812	0.1549	0.000	10	0.9775165
Nungesser Lake	1100449608	334	0.8193	0.8092	0.1161	0.000	20	1.4637679
Obonga Lake	150901291	334	0.4349	0.3946	0.0948	0.005	16	0.7232618
Old Woman Lake	51474249	80	0.7110	0.6949	0.1485	0.000	20	0.3753901
Old Woman Lake	51474249	81	0.6283	0.6065	0.1254	0.000	19	1.0519333
Onnie Lake	1100450017	131	0.7942	0.7256	0.1471	0.042	5	1.7157154
Onnie Lake	1100450017	334	0.5664	0.5423	0.1071	0.000	20	2.0078043
Opeepeesway Lake	450604606	131	0.5049	0.4430	0.1676	0.021	10	1.4318076
Opeepeesway Lake	450604606	334	0.2745	0.2187	0.1392	0.045	15	0.8613019
Otatakan Lake	1300268831	334	0.3886	0.3546	0.1291	0.003	20	1.0672378
Pakashkan Lake	150458169	131	0.8253	0.7903	0.1259	0.005	7	0.356492
Pakashkan Lake	150458169	334	0.5852	0.5621	0.1332	0.000	20	0.4638044
Pakwash Lake	1100450118	131	0.7405	0.7080	0.1262	0.001	10	0.6637516
Pakwash Lake	1100450118	334	0.5670	0.5430	0.0805	0.000	20	0.6579983
Perrault Lake	1150542158	131	0.8859	0.8732	0.0827	0.000	11	0.4320122
Perrault Lake	1150542158	316	0.7919	0.7688	0.0642	0.000	11	0.645628
Perrault Lake	1150542158	334	0.5791	0.5528	0.1733	0.000	18	0.5352131
Pikangikum Lake	225893826	334	0.4617	0.4318	0.0910	0.001	20	0.736889
Premier Lake	1100449826	81	0.8204	0.7844	0.1354	0.005	7	0.4903364
Premier Lake	1100449826	131	0.8188	0.7929	0.1659	0.001	9	0.7516963
Premier Lake	1100449826	334	0.8414	0.8326	0.1006	0.000	20	0.5878452
Press Lake	400734886	316	0.9853	0.9804	0.0284	0.001	5	0.7258152
Press Lake	400734886	334	0.6656	0.6470	0.1281	0.000	20	1.2932946
Purdom Lake	750779165	316	0.5721	0.5186	0.1107	0.011	10	1.0263455
Quirke Lake	800494782	81	0.6162	0.5949	0.1434	0.000	20	0.2466641

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Quirke Lake	800494782	316	0.6882	0.6642	0.1444	0.000	15	0.2232908
Rabbit Lake	700945359	81	0.3322	0.2951	0.1330	0.008	20	0.7273997
Rabbit Lake	700945359	316	0.6980	0.6803	0.1258	0.000	19	0.5351457
Rabbit Lake	700945359	334	0.7779	0.7655	0.0921	0.000	20	0.757623
Radisson Lake	200414063	81	0.6399	0.6199	0.0992	0.000	20	0.6127059
Radisson Lake	200414063	131	0.8898	0.8530	0.1115	0.016	5	0.492962
Radisson Lake	200414063	334	0.8578	0.8436	0.0713	0.000	12	0.4856322
Ranger Lake	800493788	81	0.8310	0.8236	0.0661	0.000	25	0.3091037
Raven Lake	200295580	81	0.5778	0.5356	0.2159	0.004	12	0.8642318
Raven Lake	200295580	131	0.8809	0.8570	0.1076	0.002	7	0.5159349
Raven Lake	200295580	271	0.3451	0.2947	0.1037	0.021	15	0.5495692
Raven Lake	200295580	316	0.7289	0.7080	0.1139	0.000	15	0.5583897
Raven Lake	200295580	334	0.5817	0.5495	0.1547	0.001	15	0.5452579
Red Cedar Lake	700945716	131	0.8108	0.8003	0.1174	0.000	20	0.5557651
Red Cedar Lake	700945716	316	0.4229	0.3909	0.1663	0.002	20	0.2611261
Red Cedar Lake	700945716	334	0.7572	0.7437	0.1291	0.000	20	0.6667298
Red Squirrel Lake	700945170	316	0.5383	0.5075	0.1225	0.001	17	0.3543517
Red Squirrel Lake	700945170	334	0.6194	0.5982	0.0906	0.000	20	0.3767765
Rib Lake	700945042	81	0.7286	0.7126	0.1030	0.000	19	0.3468939
Rib Lake	700945042	316	0.6556	0.6365	0.0957	0.000	20	0.258343
Rib Lake	700945042	334	0.9112	0.9063	0.0955	0.000	20	0.3552293
Rice Lake	450604487	334	0.5239	0.4975	0.1913	0.000	20	0.2470584
Richardson Lake	1300190797	131	0.5279	0.4689	0.1087	0.017	10	0.6321833
Richardson Lake	1300190797	316	0.4628	0.3957	0.0910	0.030	10	0.5725634
Rodd Lake	500645939	81	0.9038	0.8964	0.0752	0.000	15	0.5900343
Rollo Lake	450604142	81	0.8208	0.7760	0.1705	0.013	6	0.9135474
Rollo Lake	450604142	131	0.5688	0.5072	0.1975	0.019	9	0.5830157
Rollo Lake	450604142	334	0.9298	0.9259	0.0744	0.000	20	0.7741929
Rome Lake	500957430	80	0.6045	0.5551	0.0726	0.008	10	0.3183321
Round Lake	200413019	131	0.7746	0.7621	0.1088	0.000	20	0.3639739
Round Lake	200413019	271	0.6516	0.6248	0.1367	0.000	15	0.3392301

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Round Lake	200413019	316	0.8160	0.7976	0.0884	0.000	12	0.4299489
Round Lake	200413019	334	0.6993	0.6826	0.1011	0.000	20	0.4155493
Saddle Lake	800493763	80	0.7331	0.7174	0.1574	0.000	19	0.2336312
Sandison Lake	150901227	334	0.6811	0.6634	0.1029	0.000	20	1.6427272
Sandstone Lake	150901634	334	0.8197	0.7971	0.0856	0.000	10	0.4295142
Savant Lake	1300268877	81	0.9339	0.9174	0.0553	0.002	6	0.8934991
Savant Lake	1300268877	131	0.9010	0.8845	0.0792	0.000	8	0.4269233
Savant Lake	1300268877	334	0.7904	0.7780	0.1007	0.000	19	0.6213155
Saymo Lake	800493673	81	0.7175	0.7018	0.0905	0.000	20	0.3155273
Shack Lake	200296181	80	0.3342	0.2972	0.1694	0.008	20	0.2318357
Shelden Lake	800731173	80	0.7230	0.6923	0.1738	0.001	11	0.0942502
Silcox Lake	225893835	131	0.6846	0.6395	0.1838	0.006	9	0.6997286
Silcox Lake	225893835	334	0.4874	0.4589	0.1371	0.001	20	0.6074887
Sill Lake	800494047	80	0.7830	0.7287	0.1139	0.019	6	0.2056637
Silver Lake	1300269036	131	0.7374	0.6717	0.1588	0.029	6	0.7942098
Silver Lake	1300269036	334	0.5266	0.5003	0.1043	0.000	20	1.0201599
Smye Lake	1300269065	131	0.7993	0.7706	0.1131	0.001	9	0.9349905
Smye Lake	1300269065	334	0.2198	0.1765	0.1229	0.037	20	1.2742382
Sparlking Lake	150901317	131	0.5765	0.5160	0.1376	0.018	9	0.7089997
Sparlking Lake	150901317	334	0.6773	0.6594	0.1147	0.000	20	0.8363148
Spruce Lake	1300185772	131	0.8772	0.8618	0.1071	0.000	10	1.2420889
St. Anthony Lake	200295700	81	0.6283	0.5997	0.0956	0.000	15	0.4636496
St. Anthony Lake	200295700	131	0.8902	0.8841	0.1044	0.000	20	0.3558446
Sunbow Lake	150901627	131	0.9420	0.9347	0.0629	0.000	10	0.4717886
Sunbow Lake	150901627	316	0.8523	0.8338	0.0833	0.000	10	0.4552321
Sunbow Lake	150901627	334	0.6477	0.6225	0.0945	0.000	16	0.3714618
Titmarsh Lake	150493480	81	0.3248	0.3037	0.2806	0.000	34	0.565103
Titmarsh Lake	150493480	131	0.6963	0.6529	0.1120	0.005	9	0.3350087
Titmarsh Lake	150493480	334	0.8544	0.8453	0.1246	0.000	18	0.5407057
Tom Lake	1150542347	81	0.7677	0.7548	0.1164	0.000	20	2.3441471
Towers Lake	400315346	131	0.3860	0.3476	0.1207	0.006	18	1.4007245

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Towers Lake	400315346	334	0.8277	0.8181	0.0494	0.000	20	0.9707566
Trout Lake	700946259	81	0.6490	0.6296	0.1818	0.000	20	0.5115056
Turtle Lake	800495116	334	0.8917	0.8763	0.1090	0.000	9	0.7789721
Upper Pancake Lake	800493380	80	0.7190	0.6628	0.0577	0.016	7	0.2026312
Upper Pancake Lake	800493380	81	0.7090	0.6508	0.1447	0.017	7	0.3963746
Victoria Lake	400315659	131	0.6710	0.6490	0.0942	0.000	17	0.3677241
Wabaskang Lake	1150968820	131	0.7984	0.7732	0.1040	0.000	10	0.3431729
Wabaskang Lake	1150968820	334	0.5528	0.5279	0.0923	0.000	20	0.4086177
Wabinosh Lake	150901259	131	0.6410	0.6134	0.1361	0.000	15	0.7666723
Wabinosh Lake	150901259	131	0.6410	0.6134	0.1361	0.000	15	0.7661642
Wakomata Lake	800494532	81	0.8425	0.8313	0.1135	0.000	16	0.2473944
Wakomata Lake	800494532	316	0.7489	0.7357	0.1685	0.000	21	0.2770178
Walotka Lake	750779398	81	0.6868	0.6694	0.1046	0.000	20	2.5341586
Wapesi Lake	1300268945	334	0.6004	0.5783	0.1115	0.000	20	1.0073288
Wapikaimaski Lake	151326590	131	0.5956	0.5148	0.1507	0.042	7	0.9065895
Wapikaimaski Lake	151326590	334	0.6187	0.6022	0.1439	0.000	25	1.2087722
Wasaksina Lake	700945476	316	0.9961	0.9941	0.0159	0.002	4	0.4488403
Wasaksina Lake	700945476	334	0.5015	0.4659	0.1411	0.002	16	0.6190372
Waweig Lake	150901214	81	0.6064	0.5857	0.1152	0.000	21	0.7556515
Waweig Lake	150901214	131	0.7819	0.7456	0.0614	0.004	8	0.3601822
Weckstrom Lake	800730291	81	0.9285	0.9106	0.0660	0.002	6	0.4343253
Weewullee	750777689	80	0.6149	0.5599	0.0896	0.012	9	0.0882448
Weikwabinonaw Lake	150493911	131	0.6180	0.6006	0.1090	0.000	24	0.9411347
Weikwabinonaw Lake	150493911	334	0.8609	0.8502	0.0605	0.000	15	0.6860392
Whiskey Lake	500959450	81	0.7311	0.7105	0.1314	0.000	15	0.3506365
Whiskey Lake	500959450	316	0.7626	0.7468	0.0880	0.000	17	0.2555687
Whitefish Lake - Expanded Reservoir	51473894	316	0.8970	0.8841	0.0758	0.000	10	1.0122365
Whitefish Lake - Expanded Reservoir	51473894	334	0.4914	0.4278	0.0756	0.024	10	0.86573
Wicksteed Lake	700945679	334	0.6708	0.6525	0.0752	0.000	20	0.9098426
Wintering Lake	400315521	131	0.7496	0.7357	0.1131	0.000	20	0.5029664

Lake Name	OBJECT_ID	Species Code	R2	Adjusted R2	std.error	p-value (Sig)	N	THg (ppm w.w.)
Wintering Lake	400315521	334	0.6284	0.6077	0.1094	0.000	20	0.4075528

Appendix E. Standard Age Calculations for Walleye

The standard age estimates (Age_{Lstd}) of walleye at a total length of 500 mm was calculated for 99 lakes (Table 1). Overall, the non-linear model assumptions were adequately met with the Von Bertalanffy Growth curve. The fit of the nonlinear model was evaluated by determining certain statistical parameters; for most fish populations, the model appeared to fit the data appropriately with slight heteroscedasticity and approximately normal residuals (Table 2). Linear regression analysis was used to estimate Age_{L500mm} for 9 lakes where the Von Bertalanffy (VB) growth model did not converge to fit the data (Table 3).

Table E.1. Walleye age at standard total length of 500 mm (Age_{Lstd}) for each lake and 95% upper and lower confidence intervals. Linear regression analysis was used for 9 lakes.

GRIDCODE	Lake Name	FMZ	Age_L500mm	UCI	LCI	n
150496910	Addie Lake	6	6.520	5.973	7.286	213
400315396	Amik Lake	4	10.382	10.011	10.723	99
700944883	Anima Nipissing Lake	11	7.060	(linear regression)		
51474072	Anjigami Lake	10	7.490	7.075	8.135	100
1300268867	Arc Lake	4	20.563	15.997	28.912	112
151208180	Athelstane Lake	6	6.389	6.128	6.656	136
500957559	Bark Lake	10	3.440	3.058	3.846	13
150491421	Batwing Lake	6	13.098	11.659	15.531	82
1300844677	Bawden Lake	4	10.279	9.821	10.726	100
150473650	Bedivere Lake	6	8.990	8.450	9.630	92
750778612	Black Sturgeon Lake	6	8.262	7.829	9.473	32
1100550507	Bluffy Lake	4	10.107	9.667	10.616	138
151208182	Blunder Lake	6	8.606	8.128	9.120	44
150901230	Bukemiga Lake	6	11.225	10.451	13.187	89
150485798	Burchell Lake	6	8.084	7.411	9.300	68
1300269028	Bury Lake	4	12.077	11.310	13.014	94
1150542356	Canyon Lake	4	7.919	7.608	8.240	140
1300268905	Carling Lake	4	10.925	10.167	11.728	76
67920422	CASSELS Lake	11	7.013	6.538	7.530	49
750778738	Circle Lake	6	8.596	8.191	9.081	63
1150542006	Conifer Lake	4	5.368	3.935	7.127	17
150481682	Crayfish Lake	6	7.924	7.368	8.523	81
700946030	Deer Lake	11	7.851	6.682	16.000	30
150901352	Dog Lake	6	10.730	9.813	12.715	121
200295827	Dumbell Lake	8	8.211	7.656	9.025	51
200295858	Duncan Lake	8	6.301	5.917	6.753	39
750777599	Elbow Lake	6	11.087	9.849	14.208	34
150901331	Empire Lake	6	10.485	9.895	10.920	67
800494484	Endikai Lake	10	5.443	5.025	5.856	19

GRIDCODE	Lake Name	FMZ	Age_L500mm	UCI	LCI	n
1300269085	Expanse Lake	4	10.806	10.381	11.343	145
1300268891	Fitchie Lake	4	10.669	10.172	11.305	118
750777552	Frank Lake	6	4.750	4.450	5.110	28
800493167	Gull Lake	10	3.610	2.538	4.660	23
700945634	Hangstone Lake	11	8.055	6.842	10.786	31
150446690	Harmon Lake	6	9.588	9.097	10.160	105
400315535	Hartman Lake	4	8.154	7.650	8.595	76
151326631	Hawkeye Lake	6	7.279	6.140	10.740	40
150455227	Holinshead Lake	6	11.048	10.683	11.429	202
150901343	Holly Lake	6	9.706	8.980	10.716	49
950283324	Horwood Lake	8	11.620	(linear regression)		
400704878	Indian Lake (z4)	4	9.221	8.816	9.670	227
450603811	Ivanhoe Lake	8	6.813	6.429	7.240	50
151209168	Jolly Lake	6	9.320	(linear regression)		
1100449943	Jubilee Lake	4	12.620	(linear regression)		
150479364	Kashabowie Lake	6	12.375	9.887	20.376	41
150901125	Kawaweogama Lake	6	12.245	11.832	12.727	211
150489325	Kekekuab Lake	6	7.723	7.089	8.470	76
700945293	Kokoko Lake	11	4.470	(linear regression)		
400315402	Kukukus Lake	4	9.996	9.640	10.378	201
150474466	Lac du Milieu	6	7.573	6.720	8.632	40
200258020	Lady Evelyn	11	8.490	7.667	9.367	60
700946387	Lake Nosbonsing	11	8.253	7.617	9.015	98
200414151	Larder Lake	8	5.119	4.898	5.350	32
150454298	Little Metionga Lake	6	14.290	13.411	15.773	169
151208288	Little North Lake	6	4.390	3.863	4.868	40
150901340	Loganberry Lake	6	11.196	10.565	12.013	97
1100550520	Longlegged Lake	4	8.444	8.139	8.769	221
151326646	Lower Shebandowan Lake	6	9.780	9.056	10.670	30
150901245	Maggotte Lake	6	10.563	9.426	11.665	23
400315677	Mameigwess Lake	4	4.751	4.347	5.091	54
150452866	Mattawa Lake	4	10.030	(linear regression)		
1300268818	McCrea Lake	4	13.217	12.251	14.861	157
150901327	Metionga Lake	6	13.287	12.766	13.917	272
400315734	Mud Lake	4	10.216	9.578	11.132	86
700946020	Muskosung Lake	11	7.389	6.704	8.184	82
150490430	Nelson Lake	6	7.682	7.399	7.995	105
150901585	Northern Light Lake	6	4.204	4.059	4.354	227
1100450017	Onnie Lake	4	17.180	11.718	21.886	72
450604606	Opeepeesway Lake	8	9.363	8.581	10.740	154
150458169	Pakashkan Lake	6	8.900	8.620	9.210	167
1100450118	Pakwash Lake	4	10.730	9.939	11.799	328
1150542158	Perrault Lake	4	7.696	7.384	8.022	248
400734886	Press Lake	4	12.335	11.274	14.608	216
700945359	Rabbit Lake	11	6.102	(linear regression)		
200414063	Radisson Lake	8	4.606	4.337	4.926	20
200295580	Raven Lake	8	5.356	5.011	5.768	42
700945716	Red Cedar Lake	11	9.569	8.140	14.830	100
700945170	Red Squirrel Lake	11	5.562	5.183	6.214	110

GRIDCODE	Lake Name	FMZ	Age_L500mm	UCI	LCI	n
700945042	Rib Lake	11	5.515	5.079	6.017	98
450604487	Rice Lake	8	9.632	8.570	14.486	100
450604142	Rollo Lake	8	4.799	4.583	5.043	70
200413019	Round Lake	8	4.617	4.353	5.027	104
150901227	Sandison Lake	6	13.685	12.944	14.736	136
150901634	Sandstone Lake	6	7.058	6.578	7.553	80
1300268877	Savant Lake	4	9.861	9.478	10.265	578
1300269036	Silver Lake (2009)	4	15.365	13.061	22.230	114
1300269065	Smye Lake	4	14.956	13.653	18.069	108
150901317	Sparkling Lake	6	10.660	6.560	7.540	126
150901627	Sunbow Lake	6	4.880	(linear regression)		
700945140	Temagami Lake	11	5.583	5.174	6.063	55
150493480	Titmarsh Lake	6	5.324	5.004	5.693	62
400315346	Towers Lake	4	6.894	6.476	7.376	118
800495116	Turtle Lake	10	5.037	4.831	5.286	32
1150968820	Wabaskang Lake	4	9.480	9.012	9.953	528
151326590	Wapikaimaski Lake	6	13.935	13.041	15.143	178
700945476	Wasaksina Lake	11	5.970	(linear regression)		
150493911	Weikwabinonaw Lake	6	5.496	4.949	6.070	24
700945679	Wicksteed Lake	11	10.212	9.100	12.511	116
400315521	Wintering Lake	4	8.506	8.192	8.812	290

Table E.2. Statistical Tests used to Evaluate Model Fit

GRIDCODE	Lake Name	Lack of Fit	Likelihood Ratio	Levene's Test	Shapiro Wilk W	Shapiro Wilk p-value
150496910	Addie Lake	0.7129	0.4510	0.2831	0.9784	0.5694
400315396	Amik Lake	0.1664	0.1124	0.0012	0.9634	0.0075
700944883	Anima Nipissing Lake			(linear regression)		
51474072	Anjigami Lake	0.3032	0.2715	0.1024	0.9274	0.0000
1300268867	Arc Lake	0.0291	0.0115	0.9108	0.9795	0.0821
151208180	Athelstane Lake	0.0030	0.0009	0.0007	0.9755	0.0150
500957559	Bark Lake	0.5870	0.3390	0.8430	0.8874	0.0899
150491421	Batwing Lake	0.0051	0.0014	0.2802	0.9247	0.0001
1300844677	Bawden Lake	0.0007	0.0001	0.0166	0.9682	0.0195
150473650	Bedivere Lake	0.5051	0.3707	0.0174	0.9906	0.7622
750778612	Black Sturgeon Lake	0.5734	0.4654	0.6840	0.9462	0.1126
1100550507	Bluffy Lake	0.0144	0.0071	0.0043	0.9942	0.8486
151208182	Blunder Lake	0.0096	0.0021	0.8639	0.9536	0.0750
150901230	Bukemiga Lake	0.1091	0.0684	0.0048	0.9918	0.8568
150485798	Burchell Lake	0.1978	0.0970	0.0928	0.9842	0.5438
1300269028	Bury Lake	0.0047	0.0014	0.0634	0.9763	0.0857
1150542356	Canyon Lake	0.0039	0.0020	0.0078	0.9271	0.0000
1300268905	Carling Lake	0.0868	0.0251	0.8715	0.9658	0.0381
67920422	CASSELS Lake	0.2984	0.2056	0.0952	0.9461	0.0259
750778738	Circle Lake	0.0058	0.0014	0.2735	0.9890	0.8580
1150542006	Conifer Lake	0.2863	0.0254	0.0549	0.9560	0.5578
150481682	Crayfish Lake	0.0124	0.0032	0.7359	0.9726	0.0797
700946030	Deer Lake	0.1814	0.0585	0.4853	0.9409	0.0960
150901352	Dog Lake	0.0000	0.0000	0.1779	0.9731	0.0165
200295827	Dumbell Lake	0.4738	0.3215	0.4067	0.9873	0.8572
200295858	Duncan Lake	0.0327	0.0100	0.3482	0.9802	0.7118
750777599	Elbow Lake	0.2982	0.1645	0.0852	0.9733	0.5567
150901331	Empire Lake	0.6587	0.4944	0.4938	0.9812	0.4037
800494484	Endikai Lake	0.1147	0.0429	0.1820	0.9650	0.7524
1300269085	Expanse Lake	0.9295	0.9085	0.0002	0.9941	0.8190
1300268891	Fitchie Lake	0.1191	0.0791	0.0056	0.9921	0.7447

GRIDCODE	Lake Name	Lack of Fit	Likelihood Ratio	Levene's Test	Shapiro Wilk W	Shapiro Wilk p-value
750777552	Frank Lake	0.9110	0.8518	0.8229	0.8006	0.0001
800493167	Gull Lake	0.0000	0.0000	0.2207	0.9375	0.1585
700945634	Hangstone Lake	0.5948	0.4259	0.3149	0.9574	0.2486
150446690	Harmon Lake	0.0008	0.0002	0.4145	0.9589	0.0025
400315535	Hartman Lake	0.0000	0.0000	0.0000	0.0000	0.0000
151326631	Hawkeye Lake	0.1222	0.0542	0.3896	0.9764	0.5913
150455227	Holinshead Lake	0.0000	0.0000	0.0014	0.9648	0.0001
150901343	Holly Lake	0.0182	0.0054	0.3460	0.9785	0.5168
950283324	Horwood Lake			(linear regression)		
400704878	Indian Lake (z4)	0.2810	0.2320	0.6009	0.8925	0.0000
450603811	Ivanhoe Lake	0.3041	0.1710	0.4489	0.9793	0.5237
151209168	Jolly Lake			(linear regression)		
1100449943	Jubilee Lake			(linear regression)		
150479364	Kashabowie Lake	0.0334	0.0071	0.3987	0.9679	0.3077
150901125	Kawaweogama Lake	0.0013	0.0006	0.0046	0.9880	0.0732
150489325	Kekekuab Lake	0.0000	0.0000	0.7143	0.9826	0.3823
700945293	Kokoko Lake			(linear regression)		
400315402	Kukukus Lake	0.0047	0.0023	0.0000	0.9827	0.0141
150474466	Lac du Milieu	0.3045	0.1729	0.1065	0.9288	0.0147
200258020	Lady Evelyn	0.3596	0.1717	0.0283	0.9781	0.3546
700946387	Lake Nosbonsing	0.0000	0.0000	0.0469	0.9577	0.0031
200414151	Larder Lake	0.5731	0.4914	0.0482	0.9842	0.9073
150454298	Little Metionga Lake	0.1487	0.1094	0.0000	0.9697	0.0010
151208288	Little North Lake	0.2996	0.1525	0.4567	0.9723	0.4244
150901340	Loganberry Lake	0.0000	0.0000	0.0000	0.9688	0.0206
1100550520	Longlegged Lake	0.0040	0.0021	0.0000	0.9884	0.0703
151326646	Lower Shebandowan Lake	0.0011	0.0000	0.5153	0.9669	0.4585
150901245	Maggotte Lake	0.0939	0.0291	0.5873	0.9823	0.9411
400315677	Mameigwess Lake	0.0205	0.0025	0.1511	0.9731	0.2626
150452866	Mattawa Lake			(linear regression)		
1300268818	McCrea Lake	0.4931	0.4168	0.0005	0.9917	0.4972
150901327	Metionga Lake	0.0000	0.0000	0.0012	0.9911	0.0998

GRIDCODE	Lake Name	Lack of Fit	Likelihood Ratio	Levene's Test	Shapiro Wilk W	Shapiro Wilk p-value
400315734	Mud Lake	0.0007	0.0002	0.0001	0.9734	0.0731
700946020	Muskosung Lake	0.0000	0.0000	0.0889	0.9601	0.0120
150490430	Nelson Lake	0.1863	0.1314	0.5149	0.9816	0.1549
150901585	Northern Light Lake	0.0001	0.0000	0.0000	0.9884	0.0643
1100450017	Onnie Lake	0.1819	0.1050	0.0000	0.9438	0.0030
450604606	Opeepeesway Lake	0.0000	0.0000	0.0134	0.9657	0.0007
150458169	Pakashkan Lake	0.0000	0.0000	0.0007	0.9785	0.0106
1100450118	Pakwash Lake	0.0000	0.0000	0.0000	0.9588	0.0000
1150542158	Perrault Lake	0.0000	0.0000	0.0000	0.9651	0.0000
400734886	Press Lake	0.0007	0.0004	0.9786	0.0023	0.0000
700945359	Rabbit Lake			(linear regression)		
200414063	Radisson Lake	0.0024	0.0006	0.0274	0.9608	0.5599
200295580	Raven Lake	0.0077	0.0010	0.6796	0.9737	0.4512
700945716	Red Cedar Lake	0.0000	0.0000	0.0000	0.8804	0.0000
700945170	Red Squirrel Lake	0.0007	0.0004	0.0009	0.9459	0.0002
700945042	Rib Lake	0.0141	0.0099	0.0000	0.9446	0.0004
450604487	Rice Lake	0.0001	0.0000	0.1070	0.9826	0.2112
450604142	Rollo Lake	0.0012	0.0006	0.0001	0.9243	0.0004
200413019	Round Lake	0.0042	0.0028	0.0183	0.7962	0.0000
150901227	Sandison Lake	0.0000	0.0000	0.0138	0.9777	0.0251
150901634	Sandstone Lake	0.0543	0.0257	0.0000	0.9696	0.0565
1300268877	Savant Lake	0.0000	0.0000	0.0000	0.9759	0.0000
1300269036	Silver Lake (2009)	0.0000	0.0000	0.0002	0.9183	0.0000
1300269065	Smye Lake	0.0003	0.0001	0.0005	0.9203	0.0000
150901317	Sparkling Lake	0.0125	0.0077	0.0305	0.9945	0.9072
150901627	Sunbow Lake			(linear regression)		
700945140	Temagami Lake	0.0024	0.0003	0.0214	0.9759	0.3336
150493480	Titmarsh Lake	0.3252	0.1647	0.0002	0.9654	0.0774
400315346	Towers Lake	0.0641	0.0498	0.0001	0.9796	0.0908
800495116	Turtle Lake	0.1046	0.0354	0.8318	0.8589	0.0007
1150968820	Wabaskang Lake	0.0000	0.0000	0.0000	0.9527	0.0000
151326590	Wapikaimaski Lake	0.0019	0.0006	0.0002	0.9782	0.0071

GRIDCODE	Lake Name	Lack of Fit	Likelihood Ratio	Levene's Test	Shapiro Wilk W	Shapiro Wilk p-value
700945476	Wasaksina Lake			(linear regression)		
150493911	Weikwabinonaw Lake	0.0739	0.0164	0.2117	0.9360	0.1326
700945679	Wicksteed Lake	0.0000	0.0000	0.0430	0.9782	0.0558
400315521	Wintering Lake	0.0005	0.0003	0.0000	0.9810	0.0007

Table E.3. Walleye age at standard total length of 500 mm (Age_{Lstd}) calculated by linear regression analysis

Lake Name	p	r^2	Age_L500mm	n
Anima Nipissing Lake	2.20E-16	0.7540	7.06	54
Hornwood Lake	2.59E-15	0.7260	11.62	50
Jolly Lake	2.20E-16	0.8413	9.32	74
Jubilee Lake	2.20E-16	0.8380	12.62	116
Kokoko Lake	2.20E-16	0.9072	4.47	52
Mattawa Lake	2.20E-16	0.8628	10.03	148
Rabbit Lake	1.19E-15	0.8219	5.75	78
Sunbow Lake	0.005978	0.3870	5.07	16
Wasaksina Lake	0.0001383	0.5826	5.97	36