Volume 4, Number 2 Summer 1999

Great Lakes Research Review

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GREAT LAKES RESEARCH CONSORTIUM

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Sea Grant New York

Lake Ontario-St. Lawrence River Ecosystem

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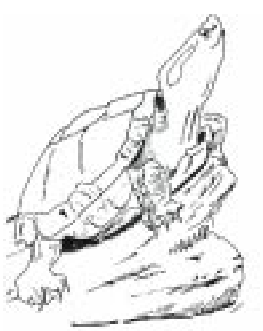

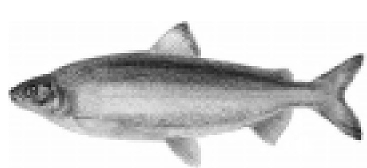

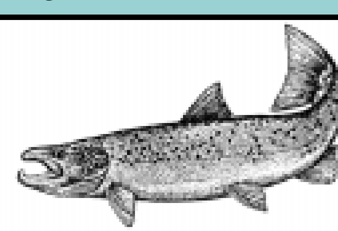

Great Lakes Research Review

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ABOUT THIS PUBLICATION:

The Great Lakes Research Review was created to provide an outlet for technically sophisticated information concerning important research being undertaken in the Great Lakes, particularly but not exclusively by researchers affiliated with the New York Great Lakes Research Consortium and its member institutions. It is designed to fill a gap between newsletter-type general information articles and articles that are suited for peer-reviewed technical journals. We hope to provide a substantive overview of Great Lakes research.

The Review is a joint publication of the Great Lakes Research Consortium, the Great Lakes Program of the University at Buffalo and the New York Sea Grant Institute. Editing responsibilities alternate between the Consortium and the Great Lakes Program. Each issue has a special theme. Past issues focused on the fate and transport of toxic substances and the effects of toxics, Great Lakes fisheries issues, and exotic species and their impact on the Great Lakes. If you would like copies of previous issues you may contact the Great Lakes Research Consortium or the Great Lakes Program.

THE UPCOMING ISSUE:

The present issue is the second of two volumes describing some of the work related to the Lake Ontario and St. Lawrence River ecosystem. These issues have been prepared in conjunction with our St. Lawrence River-Lake Ontario (SLRLO) Research Initiative which is organizing research teams to improve understanding of these linked systems and their relationships and to support the Lake Ontario Lakewide Management Plan and the restoration and protection of the St. Lawrence River. For more information about SLRLO, contact Jack Manno at *jpmanno@mailbox.syr.edu* or David Lean at *dlean@oreo.uottawa.ca.*

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Great Lakes Research Review

Lake Ontario-St. Lawrence River Ecosystem

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Introduction

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I was asked by the publishers of the Great Lakes Research Review to take this opportunity to formally introduce myself as the Great Lakes Programs Coordinator for the New York State Department of Environmental Conservation. In this role I serve as the Department's primary contact for Great Lakes activities and planning initiatives. I would like to use this space to share with you more about myself, the position and a mechanism to support research in the New York Great Lakes Basin.

As many of you know, I have been working with various Great Lakes programs for the past seven years in the Division of Water. Through this work I have gained experience in and respect for the challenges presented by multi-party environmental planning and management initiatives. Specifically, I have served as the chairperson for binational committees responsible for providing information about Lake Ontario and Niagara River programs and I have worked closely with DEC's regional and local partners to better coordinate water quality improvements in NYS's Great Lakes Basin. I have also served as a professional facilitator for meetings and workshops dealing with Great Lakes issues as well as other topics.

As I step into the position of NYSDEC's Great Lakes Programs Coordinator my duties will include: managing work on Lake Erie and Lake Ontario Lakewide Management Plans and the Niagara River Toxics Management Plan; serving on several basin-wide and binational Great Lakes committees and commissions and acting as the Department's liaison to the New York State Great Lakes Basin Advisory Council and the New York State Great Lakes Legislative Coalition. My predecessor, Mr. Gerald Mikol, held the Great Lakes Coordinator position from 1994 until his appointment to Regional Director of the Regional Office in Buffalo, New York in 1996. For those of you who know and have worked with Mr. Mikol, you can appreciate the high standards of professionalism and wealth of knowledge he brought to this position; standards I plan to uphold as I serve in this position.

I am an alumna of the State University of New York College at Oswego with a degree in Public Policy / Communication Studies. I have also completed several graduate courses in Environmental and Great Lakes Public Policy at the SUNY College at Buffalo. I currently reside in the Saratoga Springs area north of Albany, New York and work out of NYSDEC's Central Office. However, I will be relocating to the Buffalo, New York area and working at the Regional Headquarters in coming months.

Having a Great Lakes Programs Coordinator is important news for New York ... especially in a time when there seems to be growing concern about how the Great Lakes States are continuing to support their individual Great Lakes efforts. New York's Great Lakes Programs are diverse and multi-faceted involving both the natural resource and environmental quality aspects of the agency. I will be making efforts to improve the cross-program and cross-regional links so that the Great Lakes programs are better coordinated and working towards compatible and sustainable goals.

Each of us, from our own perspective must recognize that significant strides have been made in improving the NYS Great Lakes Basin. For example, the agencies involved with the Niagara River Toxics Management Plan (NYSDEC, USEPA, Environment Canada and Ontario Ministry of the Environment) can now document a 50% or greater reduction in most of the priority chemicals of concern in the ambient water of the Niagara River. Also, the NYS Department of Health just recently downgraded the fish consumption advisory for Gill Creek (a tributary to the Niagara River) for the portion of the creek from the Hyde Park dam to the mouth. These improvements have been realized and documented due to the dedicated work of scientists and researchers studying and analyzing the environment. Other improvements worth noting are: the full recovery of herring gull populations in Lake Ontario; the number of Bald Eagle nesting territories in the Lake Ontario basin has steadily grown from 2 nests in 1984 to 8 nests in 1999; and there is increasing evidence that Lake Trout are reproducing naturally in Lake Ontario.

As NYSDEC continues working with our partner agencies to define and achieve sustainable ecosystem health, we must be able to determine risk, make management choices and implement policies and programs. Making sensible choices and putting effective policies in place, requires the Department to have an accurate understanding of what is actually happening in the environment. This in turn requires monitoring and research efforts by Department staff and by partner agencies and organizations. I am pleased to announce that NYSDEC is now in a position (since there is a Great Lakes Programs Coordinator) to revitalize the New York State Great Lakes Protection Fund which provides a perpetual source of funds for regional and statewide research projects aimed at protecting and conserving the health of New York's Great Lakes ecosystem.

The Fund, since its inception, has been administered for NYSDEC's Commissioner by the Great Lakes Programs Coordinator with input from the New York State Great Lakes Basin Advisory Council. The Council works with NYSDEC to ensure the ecological health, economic growth and recreational enjoyment of NY's Great Lakes resources now and in the future.

The NYS Great Lakes Protection Fund supports collaborative projects between academia, government, non-government, industry and environmental interest groups. Projects should be conducting research and exchanging or applying information about remediating and sustaining the health of the plant, animal and human elements of the ecosystem. Also considered are projects that build on those that have received seed money from the Great Lakes Small Grant process administered by the Great Lakes Research Consortium. A Request-For-Proposals will be distributed in the Fall of 1999 and I anticipate grant awards to be announced in the early part of 2000. The priority categories for this cycle will focus on:

- Environmental Effects of Contamination in the Great Lakes
- Collection and Analysis of Data on the Great Lakes
- Development of new or improved environmental clean-up technologies / policies applicable to the Great Lakes
- Health of Great Lakes Biota

As I close this introduction, I encourage each of you to take a moment to reflect on and celebrate the work that has been done and the achievements that have been made over the past decade. Let these thoughts be encouraging and energizing as we gather ourselves up for taking on the challenges of the new millennium. I encourage each of us to be introspective as we determine how we can best fit into the enormous puzzle of improving and protecting our Great Lakes. It takes each of us playing different roles ... researchers, policy-makers, advocates, educators, business and citizens to meet these challenges and to make a difference. Please feel free to call me with questions or concerns at (518) 457-6610. Thank you-

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Mercury in Lake Ontario and the St. Lawrence River

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Mercury (Hg) is a contaminant of global concern. Large predatory fish in thousands of North American and European lakes, including the Great Lakes and St. Lawrence River, display Hg levels above governmental advisories for safe consumption. Fortunately, a decrease in Hg concentrations in fish has been observed during the past 20 years (Borgmann and Whittle, 1992; Lalonde, 1998; Fig. 1). However, Hg remains an important issue and Health Canada recently stated that moderate to heavy fish consumption by residents of the Great Lakes basin could result in body burdens warranting concern (Health Canada, 1998). Methyl mercury (MeHg) is particularly dangerous, because it is easily bioaccumulated and biomagnified along the food web. The effects of MeHg are selective within organisms and the developing nervous system is particularly vulnerable. Fetuses exposed to large amounts MeHg in utero through maternal ingestion of MeHg contaminated food during pregnancy display clinical manifestations of cerebral palsy with severe loss in motor functions and mental retardation. Autopsies of exposed fetuses show that several regions of the brain were underdeveloped due to lack of cell proliferation, cell migration and alteration of microtubules.

Since many predatory fish from the Great Lakes and St. Lawrence River still contain high levels of Hg, one would expect that Hg levels in the water should be high. However, this is not the case. Recent studies by our team and others (e.g. Amyot et al., submitted; Mason et al., 1997; Centre Saint-Laurent, 1996) have shown that levels of total Hg in the water of these aquatic systems are very low in most sampling sites not located directly near a point source. In fact, the Hg levels measured were similar to those found in the open ocean.

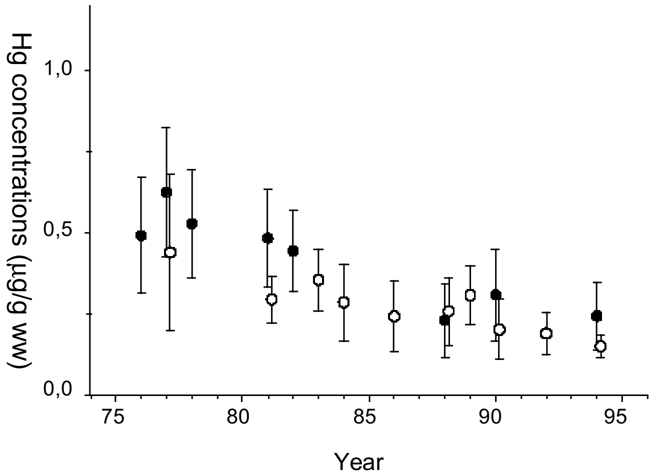
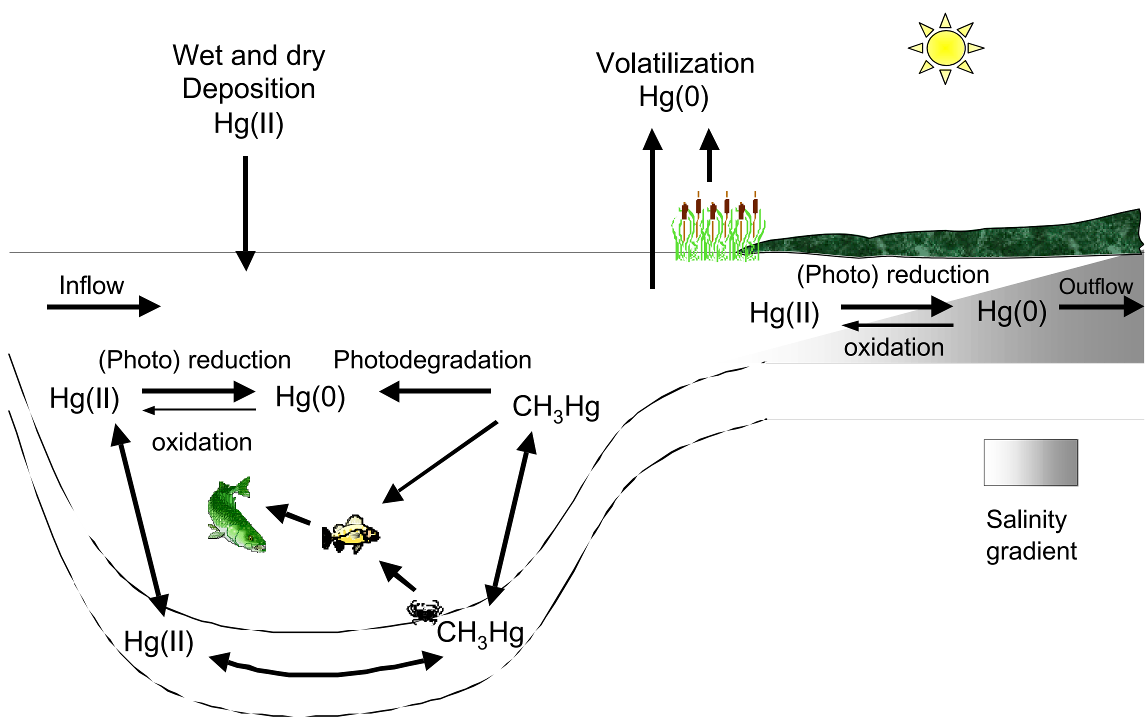


Figure 1. Decrease of mean Hg concentrations with time in white sucker (full circles) and yellow perch (open circles) in Lake St. Francis.

The paradox between low Hg levels in the water and high levels in fish can be resolved when we take into account the transformations that Hg deposited via precipitation must undergo before being accumulated in fish. Most of the atmospheric Hg is in the reduced form, Hg(0), and must be oxidized into Hg(II), in order to be deposited by wet and dry precipitation (Fig. 2). Once in the water column, Hg(II) can be transferred to the sediments where it can be methylated by bacteria and transferred to higher organisms. Alternatively, deposited Hg(II) in the water can be reduced back to Hg(0) which is volatile and will eventually return to the atmosphere. It is apparent from the above discussion that a simple relationship between total Hg in the water and Hg in fish cannot be easily drawn. However, a number of studies, mainly conducted on smaller systems, have concluded that anthropogenic inputs from the atmosphere are often the main source of Hg to fish. Recently, Mason et al. (1997) have presented a modelled budget of Hg fluxes for Lake Michigan. This budget clearly shows the importance of air/water fluxes for this Great Lake: atmospheric deposition accounted for 75% of the Hg inputs and gas exchange accounted for 44% of the outputs.

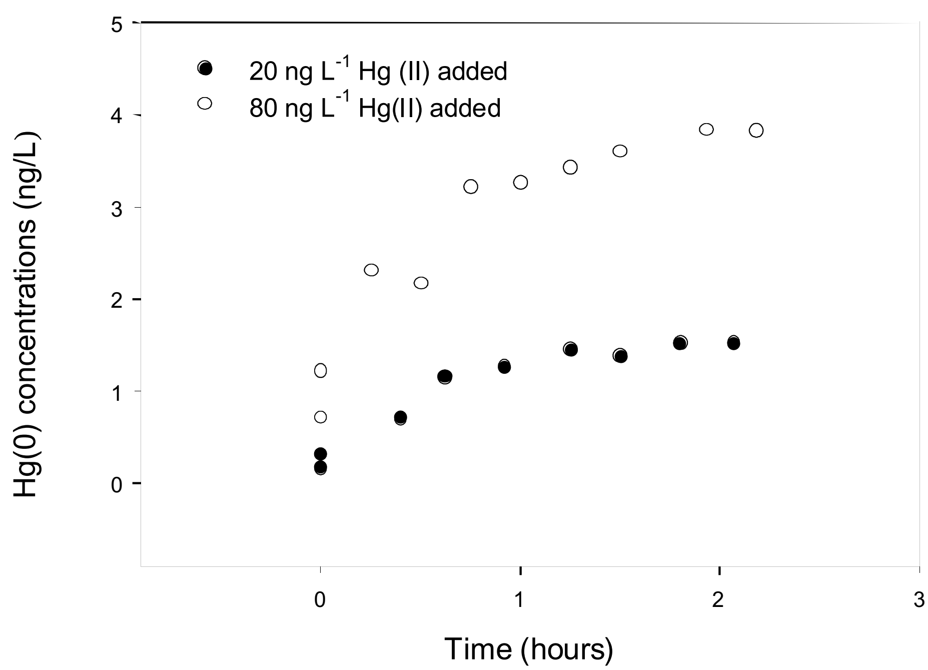
Figure 2. *Mercury Cycling in Lake Ontario and the St. Lawrence River.*



The measurement of air / water fluxes of Hg and a good understanding of the factors determining the reaction rates between Hg(0) and Hg(II) are an important prerequisite for a successful modelling of Hg cycling. We recently dedicated much effort in this field of research. Laurier Poissant's team and colleagues at Environment Canada have focused part of their research on the precise measurement of Hg fluxes above water surfaces and soil. They found that mercury evasion rates from the St. Lawrence River were correlated with solar radiation. They also proposed that redox reactions occurring in the air over the water may influence Hg evasion rates (Poissant and Casimir, 1998). Furthermore, we have recently conducted two cruises on Lake Ontario and the St. Lawrence River where we simultaneously measured Hg(0) concentrations in the surface water and at different heights above the surface, using real-time high resolution techniques. These Hg fluxes will represent the first direct estimates for Lake Ontario (Poissant et al., in prep.). During these cruises, we also performed experiments on Hg reduction and oxidation in surface waters (Amyot et al., submitted). Temporal trends in Hg(0) levels were identified with higher levels being found during the summer, probably as a result of increased Hg(0) production. Hg(0) can be either produced by (photo)chemical reactions, or by bacterial and phytoplanktonic reduction. Hg(0) levels in the St. Lawrence were lower than in Lake Ontario, a trend similar to the one observed for total Hg. This may result from a settling of particles in Lake Ontario which decreases total Hg concentration in the water column. We also conducted field and laboratory experiments which showed that photoreduction of Hg(II) to Hg(0) followed first-order kinetics. However, this photoreduction was difficult to observe in the field since the photoreducable Hg had been depleted. In the very clear waters of Lake Ontario, the capacity to photoreduce Hg exceeds the supply.

The observed rate is much less than Vmax.

Figure 3. *Production of Hg(0) in water samples spiked with different concentrations of Hg(II) and incubated under light.*



When spiking St. Lawrence water samples with different Hg(II) concentrations, we obtained, after 1 hour of incubation, different Hg(0) concentrations (Fig. 3). These Hg(0) concentrations represented about 5% of the Hg(II) initially added. Therefore, only a small fraction of the Hg(II) in the system was available for reduction. Future studies of our team will focus on the identification of these photoreducible and inert complexes. Oxidation of Hg(0) was also observed and increased with salinity near Quebec City (Fig. 2). This observation is in agreement with earlier laboratory and field experiments (Amyot et al., 1997) and may represent a source of Hg not previously included in our mercury budgets.

These studies demonstrate the dynamic nature of Hg cycling in the upper layer of the water column of these systems. They provide important estimates of key parameters used in regional modelling of Hg in the Great Lake Basin.

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Changes in Lake Whitefish (*Coregonus clupeaformis*) Stocks in Eastern Lake Ontario Following Dreissena Mussel Invasion

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INTRODUCTION

On-going fish community and fisheries sampling programs in eastern Lake Ontario and the Bay of Quinte have been conducted by the Ontario Ministry of Natural Resources based at the Glenora Fisheries Station for over 40 years. These programs have provided the information needed to relate fish community responses to major ecosystem perturbations such as the re-introduction of lake trout (Christie et al. 1987) and phosphorus input control (Minns et al. 1986). The impact on eastern Lake Ontario lake whitefish (Coregonus clupeaformis) following the invasion of Dreissena (zebra and quagga mussels) is the subject of the current investigation.

The eastern Lake Ontario lake whitefish population reached historically high levels of abundance after a dramatic resurgence that started in the early 1980s (Casselman et al. 1996). There are two major spawning stocks in eastern Lake Ontario—"lake" (south shore Prince Edward County) and "bay" (Bay of Quinte). Both stocks were recovered by the early 1990s after having existed only as a remnant population for nearly two decades, during the 1960s and 1970s. By 1996, the two stocks supported over 50% of the total Lake Ontario commercial fish harvest for all species. Casselman et al. (1996) suggested that factors controlling abundance were favorable, and the reproductive potential of the recovered stocks was large, such that the eastern Lake Ontario lake whitefish population would be sustained at high levels. One caution about this hypothesis was that impacts on the lake whitefish population following Dreissena mussel invasion remained to be determined (Casselman et al. 1996).

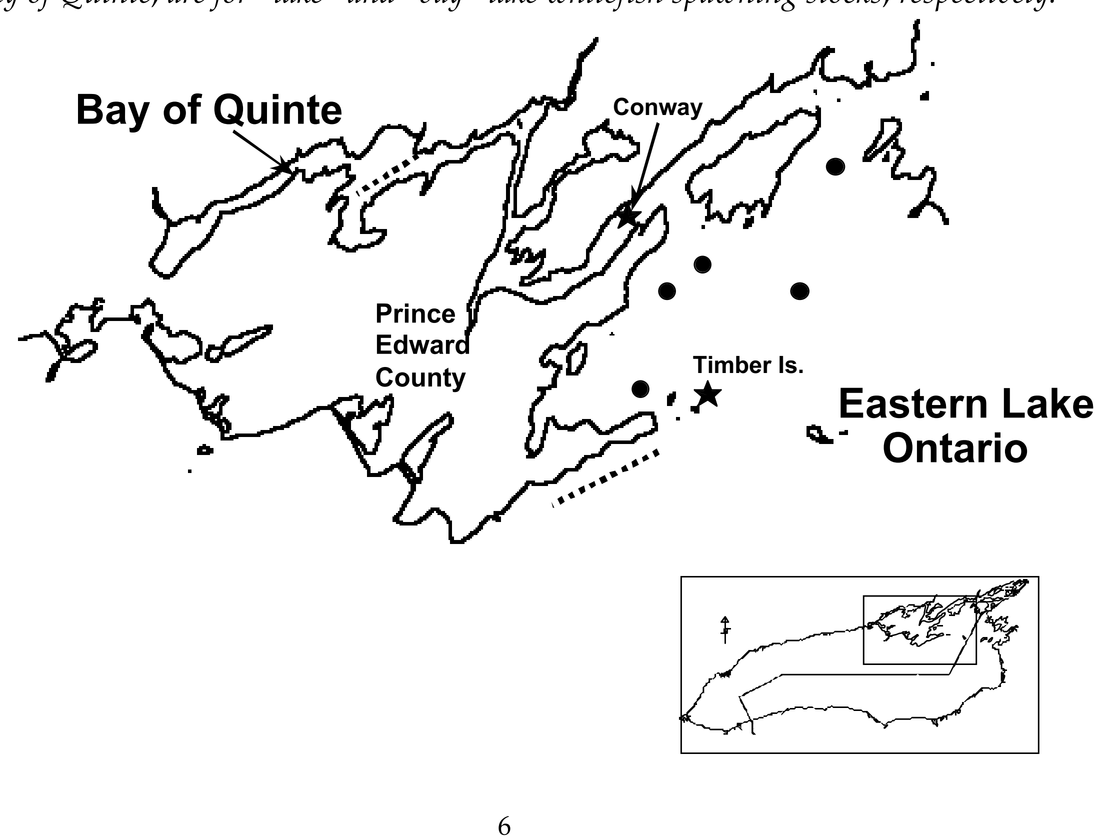
Dreissena invaded eastern Lake Ontario as early as 1991, and measurable effects on water quality were produced by 1993 (e.g., water clarity, Johannsson et al. 1998). Dreissena invaded the Bay of Quinte in 1993, were fully established by 1994 (Schaner 1998), and significant changes in Bay of Quinte water quality (e.g., phosphorus, chl a, water clarity, E. S. Millard, Department of Fisheries and Oceans, personal communication) and phytoplankton communities (K. H. Nichols, Ontario Ministry of Environment and Energy, personal communication) were observed by 1995.

This study shows changes in lake whitefish abundance, recruitment, body condition, and diet coincident with changes in the eastern Lake Ontario ecosystem following Dreissena invasion.

DATA SOURCES

Long-term index gillnetting and bottom trawling at fixed sites as well as commercial harvest sampling conducted on eastern Lake Ontario and the Bay of Quinte (Figure 1) allowed examination of changes in lake whitefish population dynamics and biological characteristics associated with Dreissena invasion. The relative abundance of juvenile (1 and 2 yr-olds) and

Figure 1. Map of eastern Lake Ontario and the Bay of Quinte showing index gillnetting (circles), bottom trawling (stars), and approximate locations of lake whitefish samples collected from the commercial harvest, during the fall spawning run (dotted lines). The trawling sites at Timber Island and Conway, and the locations of samples collected from the commercial harvest on the south shore of Prince Edward County and in the Bay of Quinte, are for "lake" and "bay" lake whitefish spawning stocks, respectively.



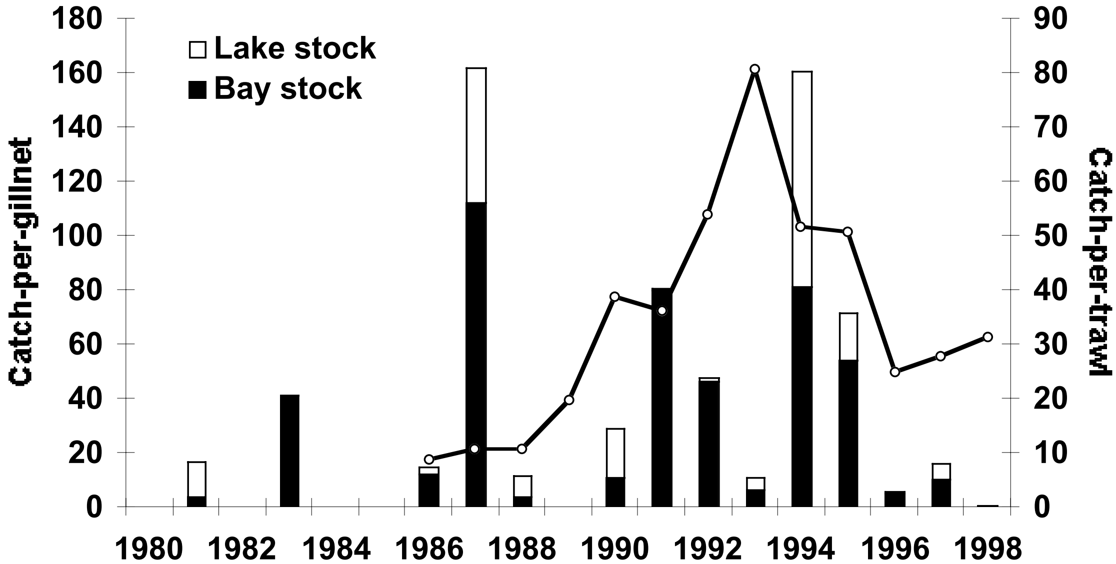


Figure 2. *Female lake whitefish body condition (least squares mean eviscerated weight adjusted for differences in length among years) for samples collected during fall spawning runs for "lake" and "bay" spawning stocks, 1990 to 1998. Error bars are +/- 95% C.I.*

adult lake whitefish (mixed "lake" and "Bay" stocks) was monitored annually in eastern Lake Ontario bottom set, multi-meshed gillnets. Young-of-the-year lake whitefish abundance—as an indicator of year-class strength—was assessed using bottom trawls in areas known to be "lake" and "bay" spawning stock nursery habitat. Adult lake whitefish body condition (corrected for gonad development) was measured using samples collected from the commercial harvest, during the fall spawning run, for both "lake" and "bay" stocks (Figure 1).

Previous accounts indicated that eastern Lake Ontario lake whitefish consumed mainly the amphipod, Diporeia hoyi, prior to Dreissena invasion (Christie et al. 1987, Ihssen et al. 1981). In 1998, a study was undertaken to contrast the traditional lake whitefish diet with their diet following Dreissena invasion. Trends in Diporeia abundance were examined using data collected during long-term benthic fauna monitoring conducted by the Department of Fisheries and Oceans at Burlington, Ontario.

RESULTS AND DISCUSSION

Lake Whitefish Abundance

Having peaked in 1993, juvenile and adult lake whitefish abundance declined through the mid-1990s (Figure 2). Juvenile fish from the 1991 and 1992 year-classes contributed very strongly (e.g., 58% of the catch in 1993), but the large numbers of young-of-the-year fish produced in 1994 and 1995 did not contribute strongly to the juvenile/adult population (e.g., 29% of the catch in 1996). Low numbers of young-of-the-year fish were produced in the last three years, (1996, 1997 and 1998) for both stocks (Figure 2). Therefore, additional declines in adult lake whitefish abundance are anticipated in the near future.

In 1997, five lake whitefish carcasses were observed in bottom trawls and 3 dead/dying fish were caught in gillnets. This was the first such occurrence in 40 years of index netting activity. In 1998, seven more carcasses were observed. The cause of death could not be determined. The fish ranged in size from 250 to 350 mm total length and represented young, immature fish 2 to 3 years of age.

Lake Whitefish Body Condition

Lake whitefish body condition declined significantly after 1993 in both spawning stocks (Figure 3). Estimates of amphipod (Diporeia hoyi) density in the areas inhabited by the two stocks (i.e., lower Bay of Quinte and eastern Lake Ontario) indicated that this formerly important food source of the lake whitefish also declined dramatically over the same time period (Figure 4). Diporeia declined by 90% in 1993 compared with the 1990 to 1992 average in the two areas, and further declined to negligible numbers thereafter. Therefore, the decline in lake whitefish condition may be related to the decline in amphipod abundance.

The decline in Diporeia abundance may be due to lake whitefish predation, and / or Dreissena impacts (e.g., direct competition for phytoplankton). The relative importance of these two potential factors may be difficult to determine.

Lake whitefish abundance peaked in 1993 at historically high levels (see above), the same year that impacts on amphipod abundance was first observed. Predation by white perch (Morone americana), was hypothesized to be responsible for maintaining low Diporeia abundance in the lower Bay of Quinte during the mid-1970s (Johnson and McNeil 1986). Diporeia density increased following the collapse of the white perch population in 1978 (Johnson and McNeil 1986). Sly and Christie (1992) suggested that the resurgence of lake whitefish in Lake Ontario would decrease the density of Diporeia. Dermott and Munawar (1993) indicated Diporeia abundance in the profundal zone of Lake Erie was limited by competition for space and food with Dreissena. Given that Dreissena only fully colonized the Bay of Quinte in 1994, that they were at lowest densities in the lower bay (approximately 1000 m-2, Schaner 1998), and that the major decrease in Diporeia occurred in 1993, the decline of Diporeia must have occurred while Dreissena densities were still relatively low (e.g., compared to densities observed in 1997, approximately 10,000 to 100,000 m-2, Schaner 1998).

Figure 3. *Female lake whitefish body condition (least squares mean weight adjusted for differences in length among years) for samples collected during fall spawning runs for "lake" and "bay" spawning stocks, 1990 to 1998. Error bars are +/- 95% C.I.*

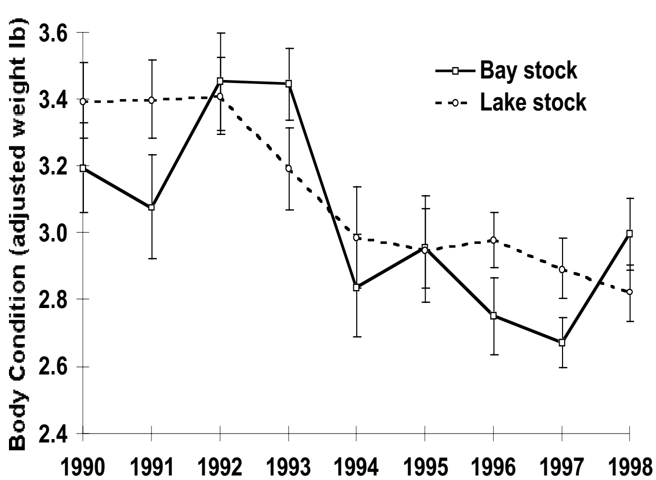
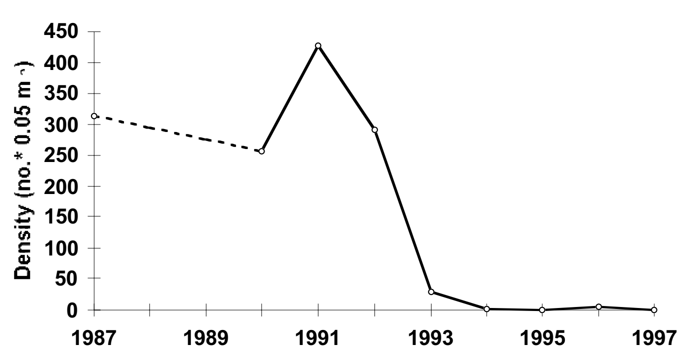


Figure 4. *Amphipod (Diporeia hoyi) density (no.*0.05 m-2) averaged for two sites in the lower Bay of Quinte and eastern Lake Ontario, 1987 to 1997 (no sampling in 1988 or 1989).*



Taxa			Number of Fish	Frequency of Occurrence
Crustacea	Decapoda		1	1%
Insecta	Diptera		14	10%
	Trichoptera		7	5%
Mollusca	Gastropoda		7	5%
	Pelecypoda	*Dreissena*	128	90%
	Pelecypoda	Other[1]	34	24%
Number of food items			5,814	
Number of stomachs containing items			142	84%
Number of fish examined			169	

Table 1. *Frequency of occurrence of items in the diet of juvenile and adult lake whitefish from eastern Lake Ontario, 1998 (FL = 352 mm, range 180 to 516 mm).*

Juvenile and Adult Lake Whitefish Diet

Results of the 1998 lake whitefish diet study are shown in Table 1; 5,814 prey items were identified from 142 lake whitefish stomachs containing food items. Dreissena mussels were present in 90% of the stomachs. The majority of Dreissena consumed by lake whitefish were young-of-the-year (95% of the mussels measured 2 to 7 mm in length). Other common prey items included other pelecypods (24%, mainly Pisidium and Sphaerium) and dipteran larvae (10%). Notably absent from the stomachs samples was the amphipod, Diporeia hoyi—not a single individual was identified. Of particular interest is whether poor lake whitefish body condition will persist while on a diet predominated by Dreissena mussels.

SUMMARY

Significant impacts on the eastern Lake Ontario and Bay of Quinte lake whitefish stocks were observed coincident with and following the invasion of Dreissena (zebra and quagga) mussels. These impacts included decreased juvenile and adult abundance, poor survival of juvenile fish, a significant decline in adult body condition, and reduced production of young-of-the-year fish. A dramatic decline in Diporeia abundance, formerly the major lake whitefish diet item, was coincident with both historically high lake whitefish abundance and the early stages of Dreissena colonization. Juvenile and adult lake whitefish diet changed primarily to Dreissena mussels after Diporeia declined to negligible levels.

ACKNOWLEDGEMENTS

The on-going contributions of staff at the Glenora Fisheries Station are gratefully appreciated. We also acknowledge U.S. Fish and Wildlife Restoration Act Funding, administered by the Great Lakes Fishery Commission, which supported the lake whitefish diet study.

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Interspecific Competition in Tributaries: Prospectus for Restoring Atlantic Salmon in Lake Ontario

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INTRODUCTION

Historically, Lake Ontario may have supported the world's largest freshwater population of Atlantic salmon (Salmo salar) (Webster 1982). However, by the late 1800's, salmon were virtually extinct in the lake due to the damming of tributaries, overharvest, deforestation, and pollution (Parsons 1973). Of these factors, the building of dams on tributaries, which precluded access by the salmon to natal spawning streams, was probably the most detrimental. Since the extirpation of Atlantic salmon in the Lake Ontario watershed over a century ago, considerable change has occurred throughout the lake and tributary ecosystem. The changes within the ecosystem that may have the most profound effect on Atlantic salmon restoration include the presence of exotic species, including other salmonines, and reduced habitat quality, especially in tributaries. These changes must be taken into account when considering Atlantic salmon restoration.

Within the Lake Ontario ecosystem, habitat quality has been most severely degraded in the tributaries (Smith 1995). Impassable dams still block hundreds of kilometers of historic salmon stream habitat (e.g., Oswego River system) and most habitat that is still accessible is not in pristine condition. A listing of Lake Ontario tributaries in New York with historic and current potential Atlantic salmon spawning and nursery habitat is provided in Table 1. Where suitable, accessible stream habitat for Atlantic salmon occurs within the Lake Ontario watershed, it is likely that it is now being used by naturalized Pacific salmonids including chinook salmon (Oncorhynchus tshawytscha), coho salmon (O. kisutsch), and steelhead (rainbow) trout (O. mykiss) (Johnson 1980, Johnson and Ringler 1981, Wildridge 1990, Wisniewski 1990). Of these species, steelhead which has been naturalized in Lake Ontario for over a century (Parsons 1973) and which is considered a close "ecological equivalent" to Atlantic salmon (Gibson 1981), may pose a biological impediment in terms of stream competition for Atlantic salmon restoration.

Little information exists regarding the resource utilization of Atlantic salmon in large lakes. In Lake Ontario, because of the low recovery of Atlantic salmon in the lake, almost no information is available on the habitat use or diet of salmon. Consequently, the potential for competition between Atlantic salmon and other salmonines in Lake Ontario is unknown.

STREAM COMPETITION IN SALMONIDS

Competition in stream salmonids occurs in two forms, intraspecific (Egglishaw and Shackley 1977, Symons and Heland 1978, Kennedy and Strange 1986, Gibson 1988) and interspecific (Gibson 1981, Fausch and White 1986, Kennedy and Strange 1986, Hearn 1987). According to Birch (1977) "competition occurs when a number of animals (of the same or different species) use common resources which are in short sup-

Table 1. *Lake Ontario tributaries in New York with historic use and current potential for Atlantic salmon spawning and nursery habitat. Historic use (H) from Parsons, 1973.*

Stream	Salmonid Habitat Established Above/Below Dams		Pacific Salmonines Above/Below Dams		Comments
Twelvemile Creek (H)	no	(no dams)	no	-	Warm, extensive channelization.
Johnson Creek	yes	no	no	no	Hazardous waste site present in drainage. Temperatures are generally warm.
Oak Orchard Creek (H)	no	no	no	no	Hydroelectric power dam, warm.
Genesee River (H)	no	no	no	no	Natural falls 90 ft. high precluded passage historically. Mount Morris dam warms river.
Irondequoit Creek	yes	(no dams)	yes		High quality stream.
Salmon Creek (H)	no	(no dams)	no		Warm.
Ninemile Creek (H)	yes	no	unknown	no	Pacific salmonines are allowed past dam at Hannibal, stream is warm below dam.
Rice Creek	yes	no	no	no	At least three dams between lake and trout habitat.
Oswego River System (H)	yes	no	yes	no	Massive habitat alteration due to navigation, hydropower development, flood control, urban development, pollution. Wild rainbow trout are established in Finger Lakes.
Grindstone Creek (H)	yes	marginal	no	no	One dam. Possible water quality problem in upper reaches.
Little Salmon River (H)	yes	no	no	no	Four dams between lake and trout habitat.
Salmon River (H)	yes	yes	yes	yes	Excellent tributary habitat with established populations of naturalized Pacific salmonines. Potential for Atlantic salmon spawning/nursery habitat in river.
Deer Creek (H)	yes	(no dams)	yes	-	Limited salmonid habitat
Little Sandy Creek (H)	yes	(no dams)	yes	-	Excellent quality stream. Atlantic salmon reproduction documented in the 1980's. Naturalized Pacific salmonines present.
Lindsey Creek	yes	(no dams)	yes	-	High quality but small stream. Naturalized Pacific salmonines present
Skinner Creek	yes	yes	no	yes	High quality but small stream. Naturalized Pacific salmonines present. One small dam.
South Sandy Creek	yes	no	no	no	Large stream with extensive trout habitat above dam.
North Sandy Creek (H)	yes	yes	no	yes	Large stream with extensive trout habitat above dam, some potential below dam.
Stony Creek	no	no	no	no	Warm
Black River (H)	yes	yes	yes	yes	Fish passage at lower two dams. Several dams on system for hydropower. Steelhead established below 2nd dam. Habitat in trib in this section has been rehabilitated, now potentially suitable for salmonids.
Chaumont River (H)	no	(no dams)	no	no	Warm.

ply; or if the resources are not in short supply, competition occurs when animals seeking that resource harm one another in the process." In stream salmonids competition usually occurs over food and space where individual fish establish and defend territories or maintain dominance hierarchies when resources are limiting (Hearn 1987, Gibson 1988). In juvenile salmonids where competition is often most intense, territories are generally established in riffles and dominance hierarchies in pools (Chapman 1966).

Intraspecific competition has been shown to regulate populations of juvenile Atlantic salmon in streams. Egglishaw and Shackley (1977) considered intraspecific competition to play the greatest part in regulating the number of Atlantic salmon fry in a Scottish stream. Gibson and Dickson (1984) concluded that in areas of rapids (the preferred habitat of Atlantic salmon fry), intraspecific competition within year classes was more severe than interspecific competition from brook trout (Salvelinus fontinalis). Symons and Heland (1978) found that yearling Atlantic salmon reduced the number of Atlantic salmon fry (<60 mm) by chasing them and occasionally eating them. They also reported that territorial defense did not occur between yearling salmon and fry until fry exceeded 65 mm. Similarly, Kennedy and Strange (1986) found that intraspecific competition from older Atlantic salmon significantly reduced the survival and growth of salmon fry.

Atlantic salmon occur naturally in streams with brook trout in North America and brown trout (Salmo trutta) in Europe. Although they found no evidence that Atlantic salmon fry stocking had a detrimental effect on brown trout, Egglishaw and Shackley (1980) reported that some stream sections with fewer salmon often had higher densities of trout, suggesting some competitive interactions between the two species. Kennedy and Strange (1986) concluded that high interspecific competition from brown trout was responsible for restricting the distribution of Atlantic salmon fry to shallow habitats. Brook trout can have a negative effect on young salmon when the species occur sympatrically. MacCrimmon et al. (1983) found that Atlantic salmon populations were suppressed in the presence of brook trout and that salmon fry had a size disadvantage relative to trout fry due to later emergence. Gibson and Dickson (1984) reported that brook trout had a negative effect on juvenile Atlantic salmon, especially yearling salmon parr, which use deeper and slower areas than salmon fry. Because of morphological and physiological adaptations (less buoyant and enlarged pectoral fins - adaptations to more effectively exploit riffles), Atlantic salmon fry have been reported to displace brook trout fry from riffles (Gibson 1973). Longitudinal zonation within stream systems with brook trout occurring in tributaries and smaller streams and Atlantic salmon utilizing riffles in main rivers is thought to be a mechanism that reduces competition between these species where they occur sympatrically (Gibson et al. 1987).

Competition in stream salmonids is well documented and will undoubtedly play some role if Atlantic salmon restoration in Lake Ontario is attempted. Consequently, from a management perspective, two ecological theorems related to competition should be considered. First is the "competitive exclusion principle" which maintains that competition between two species which have identical resource requirements will eventually lead to the exclusion of one of the species (Pianka 1981). The second is "interactive segregation" (Nilsson 1967) which asserts that in sympatry two ecologically similar species will reduce competitive interactions by segregating into the habitats they are best suited to exploit. It is most likely that re-introduction of Atlantic salmon in Lake Ontario tributaries will create partitioning of the available food and space resource beyond that which is currently occurring between steelhead, chinook salmon, coho salmon, brown trout, and brook trout. Bjornn (1978) found that although resource partitioning between steelhead and chinook salmon juveniles resulted in lower densities of each species in sympatry, the total juvenile salmonid density was greater than for either species in allopatry.

Juvenile steelhead are the most abundant salmonid present in most high quality streams that are accessible to adult fish migrating from Lake Ontario. The abundance of juvenile steelhead in these streams coupled with this species' ecological similarity to Atlantic salmon (Gibson 1981, Close et al. 1988), suggest competition in streams may occur if salmon restoration is attempted. Gibson (1988) found steelhead juveniles to be more aggressive than Atlantic salmon (as well as coho salmon and brook trout) and that steelhead dominated Atlantic salmon even when the salmon had prior residence and were larger. Similarly, Hearn and Kynard (1986) observed that stocking Atlantic salmon fry had no effect on the downstream movement of rainbow trout fry.

In sympatry with other salmonids, juvenile Atlantic salmon, especially fry, select riffle habitats (Gibson 1973, Gibson and Dickson 1984, Hearn and Kynard 1986, Close et al. 1988). The utilization of shallow, fast water when faced with competition from other salmonid species presumably allows Atlantic salmon to occupy riffle habitats which they are better suited to exploit than the other salmonid species because of their specific morphological and physiological adaptations. Reduced buoyancy allows Atlantic salmon to remain close to the bottom, thus reducing exposure to fast currents in riffles (Hearn 1987). Furthermore, enlarged pectoral fins allow Atlantic salmon to retain their position in fast currents (Arnold et al. 1991). In allopatry, although still frequently found in riffles, Atlantic salmon juveniles also occur in deeper, slower habitats (Symons and Heland 1978, Hearn and Kynard 1986, Kennedy and Strange 1986).

Even though juvenile steelhead are more aggressive than Atlantic salmon and have been found to dominate even larger salmon, because of their morphological and physiological adaptations, it is thought that juvenile salmon may out-compete steelhead in riffles. Gibson (1981) predicted that severe competition would occur between Atlantic salmon fry and steelhead fry in riffles but gave the edge to salmon. Conversely, Hearn and Kynard (1986) found no evidence of competition between Atlantic salmon fry and rainbow trout fry, but thought that there was some evidence of competition between rainbow trout juveniles and yearling Atlantic salmon.

PROSPECTUS FOR LAKE ONTARIO

Atlantic salmon juveniles will compete for available food and habitat resources with other salmonids, especially steelhead, in Lake Ontario tributaries. However, predicting the potential outcome of competitive interactions between steelhead and Atlantic salmon in streams is complicated by the fact that the steelhead will be wild, and at least initially, the Atlantic salmon will be produced in hatcheries. There is substantial evidence indicating that Atlantic salmon produced in hatcheries behave differently than wild fish (Fenderson and Carpenter 1971, Johnson et al. 1996, McDonald et al. 1998). Consequently, since most studies of interspecific competitive interactions of juvenile Atlantic salmon used wild salmon, very little useful information is available to help predict how hatchery produced salmon will perform in these circumstances.

Atlantic salmon fry would likely have an initial size advantage over steelhead fry since salmon spawn in the fall and steelhead during the winter and spring. However, the larger size of Atlantic salmon fry may not necessarily confer a competitive advantage over steelhead (Gibson 1988). It is possible that because of their larger initial size and morphological and physiological adaptations to shallow, fast water, Atlantic salmon fry may out-compete the more aggressive steelhead fry in riffles. Consequently, as suggested by Gibson (1981) severe competition between Atlantic salmon fry and steelhead fry will likely occur in riffles. Jones and Stanfield (1993) reported that Atlantic salmon growth and survival significantly increased when other salmonid species (i.e., steelhead, coho salmon, brown trout) were excluded from a section of Wilmot Creek, a tributary of Lake Ontario.

Although competition between Atlantic salmon fry and steelhead fry may be severe in riffles, it may not be the major factor limiting the abundance of either species. Competition at the late fry (fall/winter transition) (Rimmer et al. 1983, 1984, Johnson and Kucera 1985) or yearling (Hearn and Kynard 1986) stage may be more critical in limiting the abundance of these species. Wiens (1977) has proposed that competition has the most profound effect on a population during times of "ecological crunch." In many stream salmonid populations (apart from high temperature limitations during summer) the ecological crunch occurs during winter (Reeves et al. 1991). Movement patterns of Atlantic salmon fry (Rimmer et al. 1983, 1984) and steelhead fry (Johnson and Kucera 1985) suggest that fall/winter is often a crunch period for these species and over-winter rearing habitat may be limiting. It is possible that Atlantic salmon restoration will increase competition for over-winter habitat that is probably already occurring intraspecifically for steelhead. If over-winter habitat is limiting and substantial downstream movement of parr occurs in smaller tributaries during autumn, the potential survival of these migrants is questionable if suitable rearing habitat does not exist downstream. Consequently, systems such as the Salmon River in New York which may possess suitable mainstem rearing habitat may provide appropriate conditions for parr migrating from smaller tributaries in the autumn. Furthermore, competition between Atlantic salmon and steelhead may be reduced if salmon utilize larger tributaries such as the Salmon River for both spawning and rearing as they do in Canada (Gibson et al. 1987).

RESEARCH NEEDS

Research is needed to determine if Atlantic salmon and steelhead interactions in tributaries may impact salmon restoration. Specifically, experiments designed to determine juvenile Atlantic salmon growth, survival and habitat use in allopatry and in sympatry with steelhead need to be carried out in tributaries. Conducting experiments in allopatric and sympatric situations with steelhead should provide the necessary information to determine which situation may have the best chance of success for restoring Atlantic salmon. The identification of quality habitat that presently does not support steelhead (presumably inaccessible areas above barriers) is a top priority for this experimental design. It may also be beneficial to examine growth, survival, and habitat use of juvenile steelhead in the absence of Atlantic salmon to better understand what the impacts of salmon restoration may be on this non-native but, none-the-less, valuable species.

Other, research needs that should be considered include the evaluation of the performance of different strains of Atlantic salmon in tributaries. There are presently four strains of Atlantic salmon (i.e., Lake Clear, Grand Lake, Penobscot, Sebago) that are readily available for restoration purposes. It is possible that specific adaptations unique to one or more of the strains may allow them to survive better in competition with steelhead based on the habitat conditions present in Lake Ontario tributaries. Research is also warranted examining the potential use of egg (versus fry) planting to enable salmon to more successfully compete with steelhead. In larger tributary systems such as the Salmon River, research will be needed to detect fall movement of salmon and steelhead parr from smaller tributaries to mainstem rearing habitats. In addition, determination of mainstem riverine habitats that may support Atlantic salmon reproduction and juvenile rearing will be a priority. Finally, research will be needed to better understand Atlantic salmon ecology in Lake Ontario including food habits and habitat use to help assess the potential for competition with other salmonines.

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Identification of the Polychlorinated Terphenyl Formulation Aroclor 5432 in the St. Lawrence River Area of Concern

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INTRODUCTION

Polychlorinated biphenyls (PCB) and polychlorinated terphenyls (PCT) are compounds with similar chemical and physical properties, differing only in an additional phenyl ring for PCT. The Monsanto Company produced, under the trade name Aroclor®, approximately 115 million pounds of PCT between 1959 and 1972, compared with approximately 760 million pounds of PCB during the same period (DeKok et al., 1982). Peak production of PCT occurred in 1971 at about 20 million pounds, and production was discontinued in 1972 by the Monsanto Company due to environmental concerns and effects similar to PCB (Federal Register, 1984). The "pure" PCT Aroclors (CAS No. 61788-33-8) were designated by a four-digit number (5432, 5442 and 5460), similar to the PCB nomenclature (Figure 1). The first two digits designated terphenyl, with the last two digits representing the weight percentage of chlorination.

PCT have not received the extensive regulatory attention afforded to PCB. PCT are indirectly regulated under section 6(e) of the Toxics Substances Control Act (TSCA), by the fact that PCT Aroclors are usually contaminated with 1-5% of PCB (Federal Register, 1984). In 1979, the Interagency Testing Committee (ITC) recommended that PCT be tested for environmental and health effects under section 4(a) of TSCA. PCT were later exempted from environmental and health effects testing by the United States Environmental Protection Agency (USEPA) since PCT are no longer utilized in the United States (Federal Register, 1984).

Pydraul® was a trade name for a fire-resistant, high temperature hydraulic fluid produced by the Monsanto Company utilized extensively by many heavy industrial users for high-temperature applications. During the production life

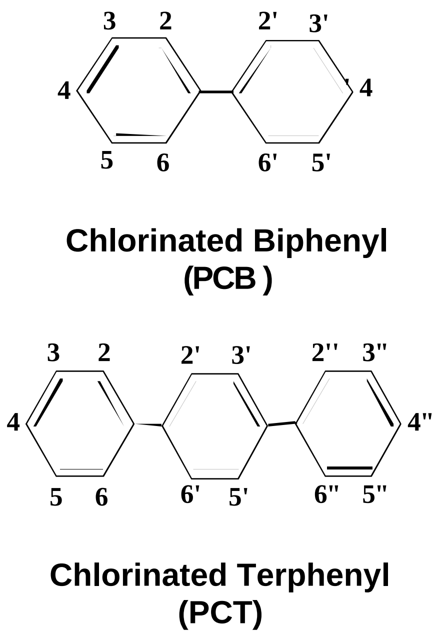


Figure 1. *Chemical structure of polychlorinated biphenyl and terphenyl.*

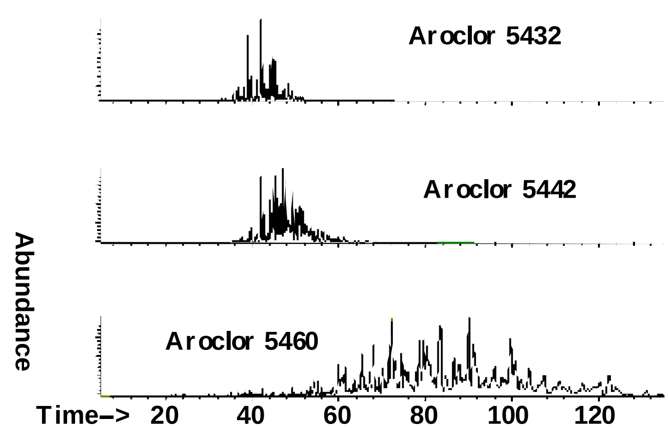
of Pydraul, various formulations were produced differing only in the combination of PCB and PCT Aroclor(s) used. For example, Pydraul 280 included Aroclor 1260 (16%) and Aroclor 1248 (62%). Pydraul 312 included Aroclor 1242 (47%), whereas Pydraul 312A only contained Aroclor 5432 (31%) (Fowlkes, 1970). The remaining percentages of all the various Pydraul formulations were composed of petroleum oils and phosphate esters. During the latter stages of the production life of Pydraul in the early 1970's, PCT were directly substituted for PCB in the Pydraul 312A formulation and marketed by the Monsanto Company as a "chlorinated biphenyl free" product (Fowlkes, 1970).

The area of study is immediately adjacent to the 165-acre General Motors (GM - Central Foundry Division) aluminum casting facility on the St. Lawrence River, in Massena, St. Lawrence County, NY. The GM site was listed as a National Priority List (NPL) "Superfund Site" in September 1983 due to the landfilling of an estimated 30,000 cubic yards of PCB-contaminated hydraulic oil sludge (USEPA, 1999). Over the past ten-years, the AOC and the St. Regis Mohawk Indian Reservation (Akwesasne Nation) have been the subject of extensive environmental, remediation and human health studies by researchers at SUNY Albany, SUNY Oswego, and Syracuse University as part of the National Institute of Environmental Health Science's - Superfund Basic Research Program (SBRP) directed by David Carpenter.

MATERIALS AND METHODS

The snapping turtle has been suggested by the International Joint Commission (IJC) as an ecosystem health indicator for the Great Lakes. Numerous facets of the snapping turtle's natural history make it a desirable species for Great Lakes pollution research. Snappers are common throughout the Great Lakes, non-migratory, and have limited home ranges (see review by Bishop and Gendron, 1998). This animal occupies a unique position as an environmental sentinel because it's omnivorous diet and long-life expectancy make it the only biomonitor similar to humans in these combined respects. The snapping turtle has both the extended life expectancy and varied diet that enable it to bioconcentrate toxic compounds that may be present in such low concentrations as to be below detection limits by other monitoring techniques.

Figure 2. *Gas chromatography of the commercially produced Aroclor mixtures.*



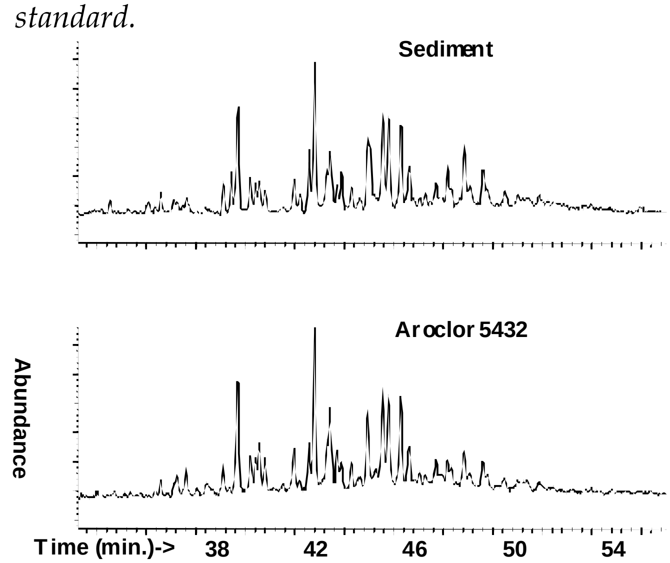
Surficial sediment and a gravid snapping turtle were collected from the area adjacent to the GM NPL site, with the assistance of the St. Regis Mohawk Tribe Environmental Division. The detection, separation and quantitation of PCT in environmental samples are complicated because their chemical and physical properties are similar to PCB and other chlorinated aromatic compounds. A method based on alumina adsorption column chromatography was utilized which separates PCB from PCT permitting the accurate identification and semi-quantitative determination of PCT Aroclors utilizing conventional gas chromatography and electron capture detection (Pagano et al., 1998). All analyses were confirmed by gas chromatography/electron impact - mass spectrometric (GC/EI-MS) methods. The developed method has been effectively used for several environmental matrices, including water, air, sediment, and biota.

RESULTS AND DISCUSSION

Identification and confirmation of PCT in the surficial St. Lawrence River sediment is illustrated in Figure 3. The concentration of PCT in the sediment was determined to be 0.8 mg/kg (dry weight), approximately 6.5% of the sediment bound PCB measured. As you will note in Figure 3, the St. Lawrence River sediment chromatographic pattern is a near identical match to that of the Aroclor 5432 standard used for comparison. Several studies emanating from the SUNY Albany SBRP have characterized and utilized St. Lawrence River sediment (Sokol et al., 1994; Pagano et al., 1995; Chiarenzelli et al., 1997; Scrudato et al., 1999). These studies have noted that the sediment-bound PCB found in the St. Lawrence River have been extensively dechlorinated as measured by the reduction in the average number of chlorines/biphenyl and relative increase of ortho-substituted chlorines. The microbially induced alterations to the St. Lawrence River sediment significantly influence the environmental mobility, bioaccumulation potential, and toxicity of the biologically modified PCB remaining.

Several inferences regarding the recalcitrance and lack of mobility of the A5432 found in the St. Lawrence River sediment are offered. The production life of the Pydraul 312A formulation was only a few years in the early 1970's, thus the PCT found in the sediment have remained nearly unchanged for almost 30 years. The bioaccumulation potential of the Aroclor 5432 mixture has not been altered (nor reduced) by reductive dechlorination and/or physico-chemical environmental transformations. Little or no information is available on the solubility, vapor pressure, or partition coefficients of individual PCT congeners and Aroclor 5432 formulation. Compared to the PCB, the molecular weight and size of the PCT molecule would suggest reduced environmental mobility resulting in high concentrations ("hot spots") near point sources. Conversely, the bioaccumulation potential and environmental half-life of PCT may be greater than PCB.

Figure 3. *GC/MS-EI chromatographic comparison of St. Lawrence River sediment and Aroclor 5432 standard.*

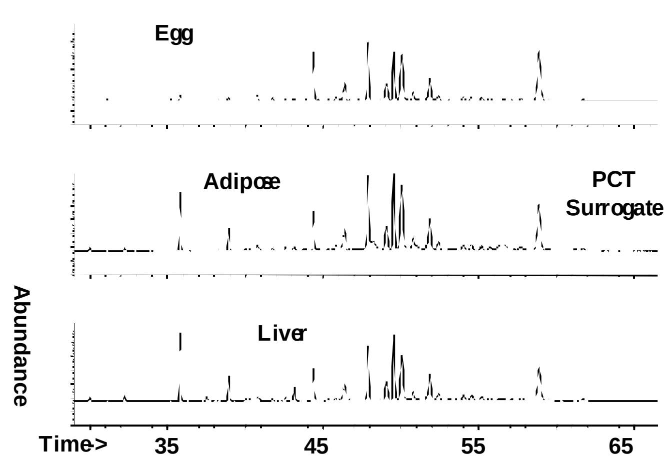


Results from the analysis of the turtle tissues and eggs indicate that PCT found in the St. Lawrence River sediment are biologically available and accumulated into the highest environmental trophic levels (Figure 4). Concentrations of PCT in snapping turtle adipose, liver and eggs, were 42.2, 20.2, and 6.5 mg/kg - lipid basis, respectively. Concentrations of PCT were 2-5% of the PCB found in the snapping turtle tissues and eggs. The gas chromatographic patterns indicate that PCT were selectively metabolized, eliminated, and bioaccumulated from the contaminated sediment profile. Aroclor 5432 consists of PCT homologues with a range of 1-5 chlorines, the majority being comprised of the trichloro- homologue group. The congeners found in the turtle tissues and eggs consist mainly of higher-chlorinated PCT homologues (Cl 4 and Cl 5) found in the original A5432 mixture at small proportions. Similar to the PCB profile found in turtles, the lower chlorinated PCT congeners are metabolized and eliminated, whereas the higher-chlorinated, recalcitant congeners are selectively accumulated and transferred to the higher trophic levels. Research by Gallagher and co-workers (1995) has demonstrated that the PCT formulation Aroclor 5432 was a more potent inducer of hepatic cytochrome (P4501A) and associated ethoxyresorufin *O*-deethylase (EROD) activity in teleosts than the PCB Aroclor 1254. Compared with PCB, the available information on the toxicological properties of PCT is limited due to the complex nature of the multiple PCT molecule configurations (*ortho-, meta-, and para-*), and lack of individual "pure" congeners for laboratory study. Previous reports of Aroclor 5432 accumulation in biota are limited to white-tailed eagles and gray seals in Sweden (Renberg et al., 1978), oysters from the Back River, Virginia (Hale et al., 1990), and several biological compartments in Tabbs Creek, Virginia (Gallagher et al., 1993).

CONCLUSIONS

Polychlorinated terphenyls have been identified in the sediment and tissues of the common snapping turtle (*Chelydra serpentina serpentina*) within the St. Lawrence River AOC for the first time. PCT were identified as Aroclor 5432 in the surficial sediment at 0.8 mg/kg (dry weight), approximately 6.5% of the sediment-bound PCB concentration. The most probable source of the PCT being the hydraulic fluid Pydraul® 312A utilized by many heavy industrial users for high-temperature applications. The sediment-bound PCT showed no biological or physico-chemical alterations, chromatographically matching an Aroclor 5432 technical standard after approximately 30 years of environmental exposure. Concentrations of PCT in the snapping turtle adipose, liver and eggs, were 42.2, 20.2, and 6.5 mg/kg - lipid basis, respectively. Analysis chromatographic congener profile indicates that PCT are selectively metabolized and bioaccumulated at concentrations ranging between 2-5% of PCB found in the snapping turtle tissues and eggs. Additional research is required to identify the source, mobility, environmental fate, and toxicity of PCT in the St. Lawrence River AOC.

Figure 4. *GC/EI-MS chromatographic comparison of snapping turtle egg, adipose, and liver from the St. Lawrence River AOC.*



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Declining Contaminant Levels in Herring Gull Eggs from Toronto Harbour, Lake Ontario, 1974-1998

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INTRODUCTION

Eggs of Herring Gulls (*Larus argentatus*) have been used to monitor annual concentrations of organochlorine contaminants in the aquatic food web of the Great Lakes since the early 1970s (Mineau et al. 1984). That monitoring scheme, maintained by the Canadian Wildlife Service (CWS) of Environment Canada, is now comprised of 15 sites, two or three per lake and one in most connecting channels (Pekarik and Weseloh 1998).

To date most of the results of this monitoring scheme have been presented on a Great Lakes-wide basis (Pekarik and Weseloh 1998), a lake-wide basis (Ewins et al. 1992) or on a compound basis (Hebert et al. 1997, Norstrom et al. 1982); perhaps because of the breadth of the database, very few results have been presented on an individual colony basis. In an attempt to give the monitoring program a wider audience, we have undertaken to present results from individual colonies in local journals or in special geographically focused issues of other journal(s) (e.g. this paper). Here we present the results of 24 years of contaminants monitoring data, 1974-1997, for four compounds: PCBs, DDE, mirex and 2,3,7,8 tetra-chlorodibenzo-dioxin (TCDD) in Herring Gull eggs from Toronto Harbour. We discuss these results in light of studies of breeding biology parameters of colonial waterbirds, in general, and Herring Gulls in particular from the same site.

METHODS

One fresh Herring Gull egg has been collected from each of 10-13 completed clutches annually from Mugg's Island (1974-1988) or Tommy Thompson Park (TTP; 1989-1998) in Toronto Harbour during early May. The two sites are approximately 2 kms apart; the later site was selected after Herring Gulls were forced to abandon Mugg's Island in 1988 (DVW, pers. obs.) The eggs were analyzed at the CWS National Wildlife Research Centre, Hull, Quebec, following the methods of summarized in Pekarik et al. (1998). Prior to 1986 all eggs were analyzed individually, after that they have been analyzed as site pools. Seventy-six different organochlorine compounds are analyzed for, including up to 59 PCB congeners. Only the four most prevalent or best known compounds are presented here: DDE, total PCBs (1:1 Aroclor 1254:1260), mirex and 2,3,7,8 dibenzo-p-dioxin (TCDD). Data for all compounds are available in Pekarik et al. (1998) and references therein. For statistical analysis, the chemical concentrations measured in the eggs were mathematically transformed when necessary and analyzed by change-point regression to determine temporal trends. The change point regression determined if there was a statistically significant change in contaminant concentration over time and if that rate of change was constant or if and when it had changed (see Pekarik and Weseloh [1998] for more details on statistical analysis).

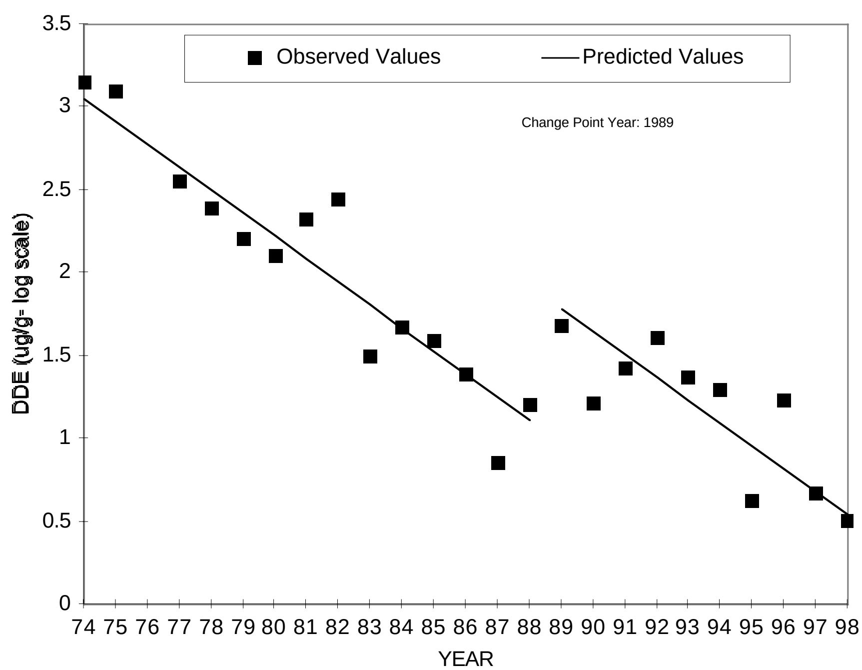
RESULTS

All four compounds declined significantly during the study: DDE and PCBs both declined 93.5% from 1974 to 1997; DDE from 23.32 ug/g to 1.95 and PCBs from 165.6 ug/g to 10.66. Both compounds showed a significant increase (a positive change point) in 1989 but after that continued to decrease at the same rate before and after the change point (Fig. 1). Mirex declined 94.2% from 7.44 ug/g to 0.43. It showed a positive change point in 1979 and declined at a slower rate after the change point than previously. TCDD declined 80.2% from 60 ug/kg to 11.9. It did not show a change point but rather declined at a constant rate from 1984, when it was first quantified, to 1997.

DISCUSSION

The results of this study, i.e. continued declining contaminant levels in Herring Gull eggs in western Lake Ontario through 1998, extend the earlier data presented for this site by three years (cf. Pekarik and Weseloh 1998). The change point years for DDE and mirex remain the same, while that for PCBs is now two years later. The rates of decline for DDE and mirex did not

***Figure 1**. Results of change point regression analysis of DDE concentrations in Herring Gull eggs from Toronto Harbour, 1974-1998. Significant decline in concentrations before and after the change point (1989); no change in rate of decline.*



change from previously relative to before and after their change points but PCBs are now declining faster after their change point than previously. Other interpretations of the Herring Gull database, as well as the comparability of contaminant trends in fish are discussed in Pekarik and Weseloh (1998).

For the contaminants presented here, the concentrations found in Herring Gull eggs at Toronto Harbour were above the Great Lakes-wide averages for TCDD (Toronto Harbour = 11.89 pg/g; Great Lakes-wide average = 7.34) and mirex (TH = 0.43 ug/g; GL = 0.10). They were below the Great Lakes-wide averages for DDE (TH = 1.95 ug/g; GL = 2.50) and total PCBs (TH = 10.66 ug/g; GL = 15.72) (data computed from 1997 values found in Pekarik et al. (1998)).

Contaminants can affect wildlife at many different levels, e.g. community, population, individual, etc. and studies of contaminant levels and effects on wildlife often go hand in hand. During this study, various breeding biology parameters were measured in Herring Gulls, and other colonial waterbirds, which nested in Toronto Harbour. For example, censuses at Toronto Harbour showed that, among obligate piscivores, breeding populations of Double-crested Cormorants increased dramatically and those of Common Terns greatly decreased from 1976/77 to the present day (Table 1). These patterns are consistent with their respective population trends on the Great Lakes as a whole: cormorants have increased Lakes-wide as a result of decreased contaminant levels and increased abundance of forage fish (Weseloh et al. 1995); Common Terns have declined as a result of competition for nest sites with the larger, more aggressive and earlier spring arriving Ring-billed Gulls (Blokpoel and Tessier 1996, Haymes and Blokpoel 1978).

For Herring Gulls, which also eat primarily fish, their population nearly doubled between 1976/77 and 1990 but appear to have been unchanged during the 1990s (Table 1). Ring-billed Gulls, which are more omnivorous than Herring Gulls, increased more than 3-fold between 1976 and 1990; no other data are available for them for

Table 1. *Numbers of nests (i.e. breeding pairs) of selected colonial waterbirds in Toronto Harbour. Data for 1976/77 and 1990 are from Blokpoel and Tessier (1996), those for 1997/98 are from S. Jarvie (Toronto Region Conservation Authority, pers. comm.).*

	YEAR		
SPECIES	1976/77	1990	1997/98
Double-crested Cormorant	0	3	1598
Herring Gull	57	103	>98
Ring-billed Gull	14,267	49,106	NA
Common Tern	1,246	128	500/230+
Caspian Tern	7	0	NA

the 1990s. Both of these species are showing a growth in numbers that is consistent with the occupation and exploitation of newly available habitat, such as has been the case Tommy Thompson Park (Haymes and Blokpoel 1978, Blokpoel and Fetterolf 1978).

In terms of avian reproductive parameters, data are available from 1977, 1978-1981, 1992 and 1998. In 1977 hatching success for four major species nesting in Tommy Thompson Park was as follows: Herring Gulls (65.1%), Ring-billed Gulls (72.4-89.5%), Common Terns (83.9-95.2%) and Caspian Terns (81.8%). This was similar to or above that reported for other colonies on Lake Ontario and other Great Lakes The overall reproductive success of Ring-billed Gulls and Common Terns at Toronto Harbor in 1977 was normal, i.e. 1.35-1.76 and 1.56-2.00 young fledged/nest, respectively; specific figures are not available for Herring Gulls or Caspian Terns (Haymes and Blokpoel 1978). During 1978-1981 reproductive success of Herring Gulls at Mugg's Island ranged from 1.40 to 1.56 yg/pair (Mineau et al. 1984). However, in 1992, hatching success was reported at 53% and fledging success at 0.28 yg per nest (B. McMartin, unpubl.). The reasons for these lowered values are not known. In 1998, hatching success had returned to normal levels, 72.2% (N=144, CWS unpubl.).

The data presented here (and in associated references) show that contaminant levels have been decreasing for most of the years of this study and have declined in the range of 80-90% since egg monitoring was begun. At the same time, populations of most colonial waterbird species for which there are data have been increasing or, if not, their decreases have probably been due to non-contaminant related causes, e.g. competition between species and habitat change through vegetative growth. Similarly, most measures of reproductive parameters show normally reproducing birds. Although the immediate outlook suggests few, if any, population level impairments due to contaminants, there are several possible, more subtle impacts which have not been investigated and which lie outside the scope of this study, e.g. longevity, age at first breeding, origin of recruitment into the population, etc. Mora (1993) details how these may impact Great Lakes waterbird populations and gives possible examples.

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Biotransformation of PCBs in St. Lawrence River Sediments and Dechlorinating Microorganisms

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INTRODUCTION

The environmental fate of natural and man-made organochlorine pollutants has undergone a dramatic change in recent years due to the discovery of nobel microorganisms which can dehalogenate them through co-metabolic process or energy-yielding halorespiration in reduced environments. The substrates for which dechlorination has been demonstrated by pure cultures fall into two categories: chlorinated aromatic compounds such as chlorobenzoates and chlorophenols (e.g., Cole *et al*., 1994; DeWeerd *et al*., 1991; Christiansen and Ahring, 1996; Sanford *et al*., 1996), and chlorinated aliphatic compounds such as chloroethenes (e.g., Holliger *et al*., 1998; Wild *et al*., 1996; Krumholz, 1997; Maymo-Gatell *et al*., 1997; Sharma and McCarty, 1996). However, there are other organochlorine contaminants which are dehalogenated by undefined mixed natural microorganisms. Polychlorinated biphenyls (PCBs) are typical of such contaminants. PCB dechlorination is believed to be "halorespiration" which utilizes the free energy that could be gained from the exergonic dechlorination reaction (Lee *et al*., 1998; Holliger *et al*., 1999).

The evidence for PCB dechlorination is wide-spread in natural environments such as freshwater sediments (Brown *et al*., 1984,1987a,b; Rissatti, 1992; Beurskens *et al*., 1993; Sokol *et al*., 1994; David *et al*., 1994; Bedard and May, 1996) as well as of estuarine and coastal sediments (Brown and Wagner, 1990; Lake *et al*., 1992; Wu *et al*., 1998). PCB dechlorination involves preferential removal of Cls' from *meta* and *para* positions, although *ortho* dechlorination of Aroclor mixtures has recently been demonstrated by microorganisms in estuarine sediments (Berkaw *et al*., 1996; Wu *et al*., 1998). Thus, the final products of PCB dechlorination are congeners with fewer Cl substitutions. They are largely *ortho*-substituted, but also contain a large number of *meta* and *para* substitutions as well. Since aerobic degradation of PCBs is limited only to congeners with low Cl substitutions, reductive dechlorination may serve as a crucial initial step which removes the barrier to aerobic breakdown leading ultimately to mineralization.

PCB dechlorinating microorganisms have so far eluded isolation and remain unidentified. In recent years, however, we have been able to gain information on their growth requirements and on the kinetics of growth and dechlorination, using statistical methods of population measurements, the most probable number and dilution fractionation techniques (Kim and Rhee, 1997, 1999; Sokol *et al.*, 1998). These studies have shown that microbial dechlorination of PCBs is driven by the growth of dechlorinating populations.

REDUCTIVE DECHLORINATION

Microbial reductive dechlorination is the only biological process known to degrade highly chlorinated PCB congeners. Although it preferentially removes *para*- and *meta*-chlorines, not all Cl's at these positions are removed, because their removal is determined by the pattern of chlorination on the biphenyl ring rather than the substitution position *per se* (Rhee *et al.*, 1993a,b). Therefore, the final dechlorination products as well as the extent of dechlorination can vary widely even for the same Aroclor, depending on the competence of the dechlorinating microorganisms present (Sokol *et al.*, 1994). In laboratory dechlorination studies of Aroclor 1248 by St. Lawrence River microorganisms, the highest extent of dechlorination, measured as the average number of Cl per biphenyl, was about 2.5. This represents approximately 40% Cl removal from the original mixture although the total *meta* and *para* Cls' account for 60% (Liu *et al.*, 1996; Kim and Rhee, 1997; Sokol *et al.*, 1998). The extent of dechlorination appears to be less in Aroclors with higher Cl content, such Aroclor's 1254 and 1260 (Rhee *et al.*, 1993b; Quensen *et al.*, 1990; Kim, 1997).

Dechlorination is concentration-dependent. However, our recent investigations of Aroclor 1248 have shown that it has a relatively high threshold concentration (40 ppm) below which no dechlorination occurs. More importantly, the maximum extent of dechlorination by sediment microorganisms from the St. Lawrence River was also shown to be concentration-dependent and the % Cl removed was found to increase as a saturation function of PCBs in our two kinetic studies (Sokol *et al.*, 1998; Rhee *et al.*, 1999). This concentration dependence was due to the accumulation of residual products which are resistant to further dechlorination. These results indicate that the maximum extent of dechlorination reflects the concentration-dependent selection of dechlorination competence and that natural dechlorination is limited by the type of dechlorinating microorganisms present.

Recent high-resolution studies of Hudson River sediments also showed the concentration dependence of the maximum level of dechlorination as well as the existence of threshold concentration (US EPA, 1997). A regression analysis of more than 250 sediment samples for the degree of dechlorination against sediment PCB concentrations showed a linear relationship with an intercept at a mean PCB concentration of 30 ppm, suggesting a threshold level. However, there was no correlation between sediment age and the level of dechlorination. Therefore, the observed PCB composition was considered to represent residual or final products after maximum dechlorination.

DECHLORINATING MICROORGANISMS

(a) Dechlorinating Microorganisms and PCB Requirement

The microbiology of PCB dechlorination is still poorly understood, mainly because of the failure to isolate the microorganisms responsible for the process. Recently we, for the first time determined the population dynamics of dechlorinators and their interactions with other anaerobic respirators in sediments using the most probable number (MPN) technique (Kim and Rhee, 1997, 1999). When PCB-free sediment spiked with Aroclor 1248 was inoculated with microorganisms eluted from PCB-contaminated

St. Lawrence River sediments, the time course of dechlorination mirrored the growth of dechlorinators. In the control sediments without PCBs, the population declined with time from the initial inoculum level. These results show that PCB dechlorinating microorganisms require PCBs for growth. Once the "dechlorinatable" congeners were exhausted, the population began to decline, although the time of decline might not coincide with the end of rapid phase depending on culture conditions (Kim and Rhee, 1997, 1999).

(b) Dechlorinating Competence and its Selection

It appears that dechlorinating populations or competence can vary widely when judged by dechlorination patterns. Recently, we have been able to identify some of the conditions which are responsible for the selection of different competence. When methanogens were inhibited with 2-bromoethanesulfonate (BES) in sediments amended with Aroclor 1248, the extent of dechlorination by St. Lawrence River sediment microorganisms was reduced by about half (Kim and Rhee, 1999). These results were mainly due to the absence of further dechlorination for some *meta*-rich products of the initial dechlorination. However, the total number of dechlorinators in the sediments was not different from that in BES-free sediments (Kim and Rhee, 1999). These results indicate that either some dechlorinators require methanogens to dechlorinate these congeners or some methanogens are also dechlorinators. Inhibition of sulfate reducers did not have any effect on dechlorination (Kim and Rhee, 1999). We have also found that at low PCB concentrations, certain dechlorination competence was absent, indicating that PCB concentrations can also be a factor in selecting dechlorinating populations.

When dechlorinating populations were fractionated and quantified using a combination of the dilution fractionation and MPN techniques, we were able to further characterize the population which was involved in the further dechlorination of *meta*-rich congeners. Their population size was two orders of magnitude less than that of other populations which was recognizable by its own distinct dechlorination pattern (3.5 _ 10^4 *vs* 2.4 _ 10^6 cells • g dry weight sediment^{-1}). Despite the low number, these *meta*-dechlorinators increased the overall dechlorination by almost two fold (Cho *et al*., 1999). Therefore, it may be possible to select specific competence to maximize dechlorination by exploiting differences in specific requirements.

(c) Dechlorinating Populations and Dechlorination Kinetics

Although earlier kinetic studies demonstrated concentration-dependence of PCB dechlorination, they failed to identify its underlying mechanism, mainly because the biomass of dechlorinating microorganisms were not measured. Since dechlorination is a slow process which takes place over a period of weeks, it is not possible to determine whether the concentration dependence of dechlorination rate and the threshold concentration simply reflect a difference in biomass or enzyme affinity. It is also unclear whether the concentration effect on the maximum level of dechlorination is related to the viability of dechlorinating microorganisms. Therefore, we investigated dechlorination of Aroclor 1248 at 10 concentrations ranging from 0 to 900 ppm with concurrent measurement of dechlorinating microorganisms using the MPN technique (Rhee *et al*., 1999).

Dechlorination rate without biomass normalization (nmol Cl removed • g sediment^{-1} • day^{-1}) was a linear function of Aroclor concentrations, with an intercept at 40 ppm (dry weight) as in the previous study (Sokol *et al*., 1998). Below the threshold level, dechlorinating microorganisms did not grow. Therefore, it is clear that the threshold represents the minimum concentration necessary for growth.

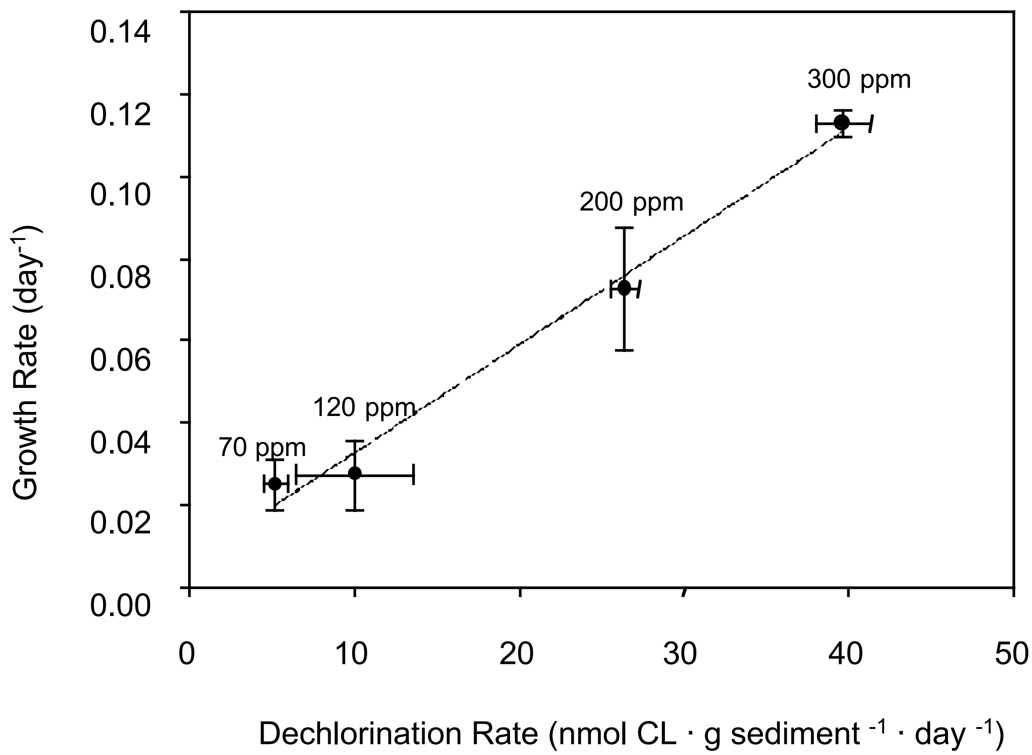


Figure 1. *Dechlorination rate of Aroclor 1248 vs. the specific growth rate of dechlorinating microorganisms below the inhibitory concentration.*

Above the threshold concentration, the specific growth rate of dechlorinating microorganisms increased with Aroclor concentrations up to 300 ppm and then decreased at higher concentrations of 600 and 900 ppm, probably due to inhibition. [It is also possible that at those high concentrations, different dechlorinating populations are selected.] Similarly, the yield of dechlorinating microorganism per mol Cl removed also showed a peak at 300 ppm and a decrease at higher concentrations, suggesting toxic effects. Because of the exponential manner of population growth, chlorine removed per dechlorinating microorganism over time, or the biomass-normalized dechlorination rate, was first order, and the rate constant appears to be a saturation function of Aroclor concentrations.

It is interesting, however, that within the non-toxic concentration range (below 300 ppm), there were significant correlations between the dechlorination rate (nmol Cl removed • g sediment^{-1} • day^{-1}) and the growth rate ($P = 0.007$) (Fig. 1), and between the maximum amount of Cl removed (nmol Cl • g sediment^{-1}) and the highest number of dechlorinating microorganisms at each PCB concentration ($P = 0.002$). It appears, therefore, that in non-inhibitory range of concentrations, dechlorination is tightly linked to the growth of dechlorinators.

DECHLORINATORS IN NATURAL SEDIMENTS AND DECHLORINATION POTENTIAL

In light of the importance of population growth in dechlorination in laboratory sediments, we investigated historically PCB contaminated sediments (primarily with Aroclor 1248) from the St. Lawrence River (six cores and five sections of one core) to examine any correlation between dechlorinating microorganisms and PCB dechlorination (Cho and Rhee, unpub-

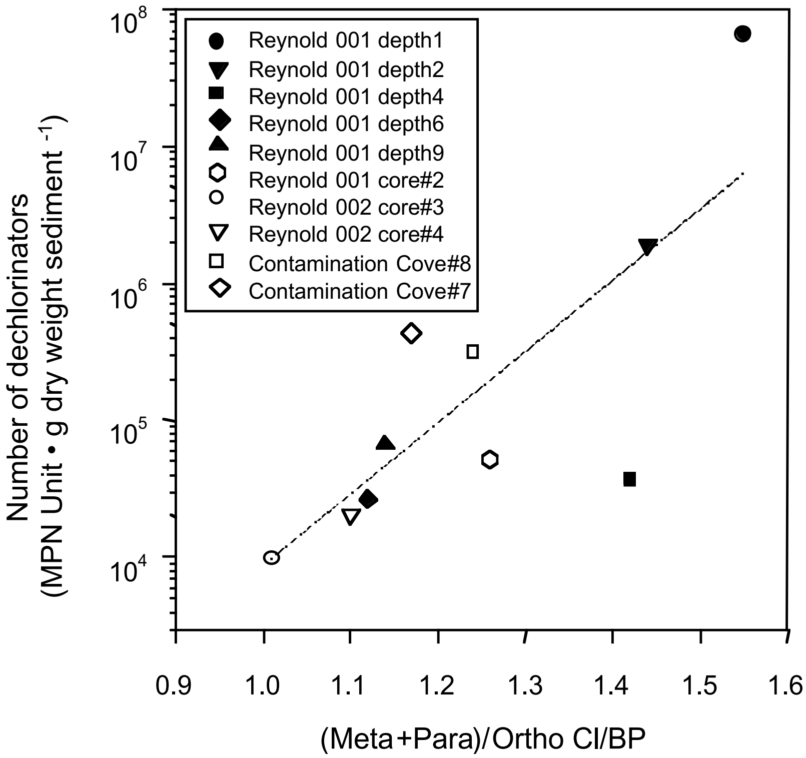


Figure 2. *The number of dechlorinating microorganisms and extent of dechlorination in historically contaminated sediments in the St. Lawrence River.*

lished). The total PCB concentration and the number of dechlorinators ranged from 14 to 1336 ppm and 9.8 _ 10^3 to 6.5 _ 10^7 MPN units • g dry weight sediment^{-1}, respectively. There was no correlation between total PCB concentrations and dechlorinator numbers. However, the population size was significantly correlated ($P = 0.007$) to the extent of dechlorination (expressed as the average number of Cl per biphenyl or the ratio of *meta + para* Cl to *ortho* Cl) of sediment PCBs (Fig. 2). More significantly, the initial number of dechlorinators was highly correlated ($P = 0.0002$) to the extent of subsequent dechlorination in the same sediments after 1 year incubation in the laboratory (Fig. 3).

No additional dechlorination was found in two sediment samples which had fewer than 2.04 _ 10^4 MPN units • g dry weight sediment^{-1}. Although these investigations are continuing at additional sites, the results so far suggest that the number of dechlorinating microorganisms may be used to predict dechlorination potential in PCB contaminated St. Lawrence River sediments.

DECHLORINATION IN DREDGED SEDIMENTS

The remediation of PCB contaminated sediments *in situ* poses many technological and logistical problems. Most often heavily contaminated sediments are dredged and disposed of in contained landfills, or CDFs, as the remediation method (Gullbring and Hammar, 1996; Fraser, 1993). Many contaminated areas including the St. Lawrence Superfund Site (General Motors and Reynold sites) have been dredged or are scheduled for dredging.

However, this method does not remediate the toxicants; it simply sequesters them in a "safe place" where they remain as a potential danger

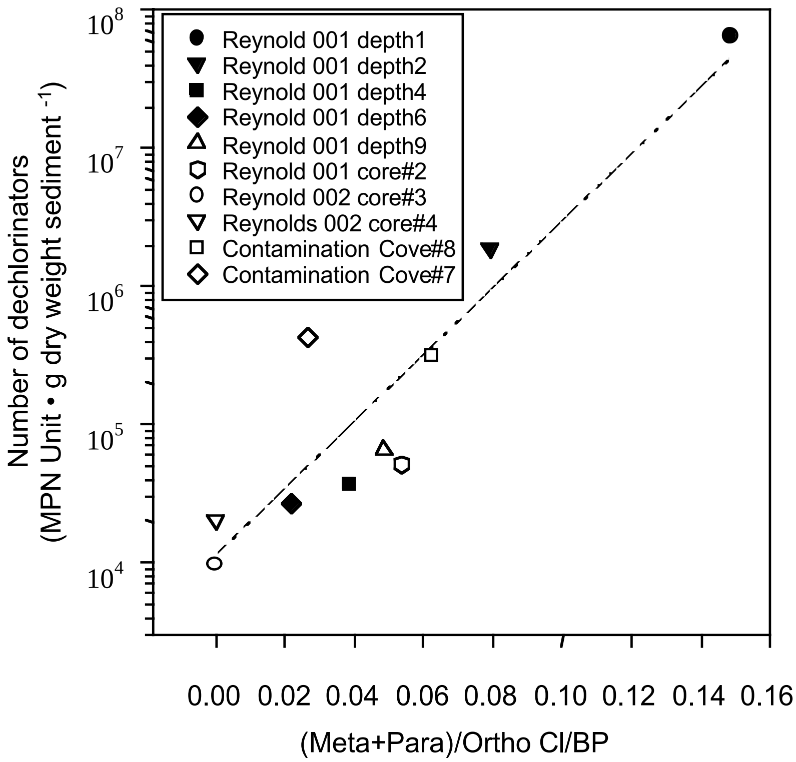


Figure 3. *The initial number of dechlorinating microorganisms in contaminated sediments from the St. Lawrence River and the extent of dechlorination after one year under laboratory incubation.*

for the future. When we investigated the state of PCB biotransformation in Hudson River sediments which were dredged and encapsulated in 1978 at Moreau, NY, they showed little evidence of dechlorination since dredging; compared with the PCBs in Hudson River sediments today, the extent of dechlorination in the dredged sediments was much less, with only a 9% reduction in total Cl content, compared to a 41% reduction in the contemporary River sediments. The extent of dechlorination in the encapsulated sediments was similar to that found in sediments exposed to aerobic conditions which prevents any dechlorination. When microorganisms eluted from encapsulated sediments were tested for their ability to dechlorinate in sediments spiked with Aroclor 1242, no dechlorination activity was found (Rhee and Sokol, unpublished).

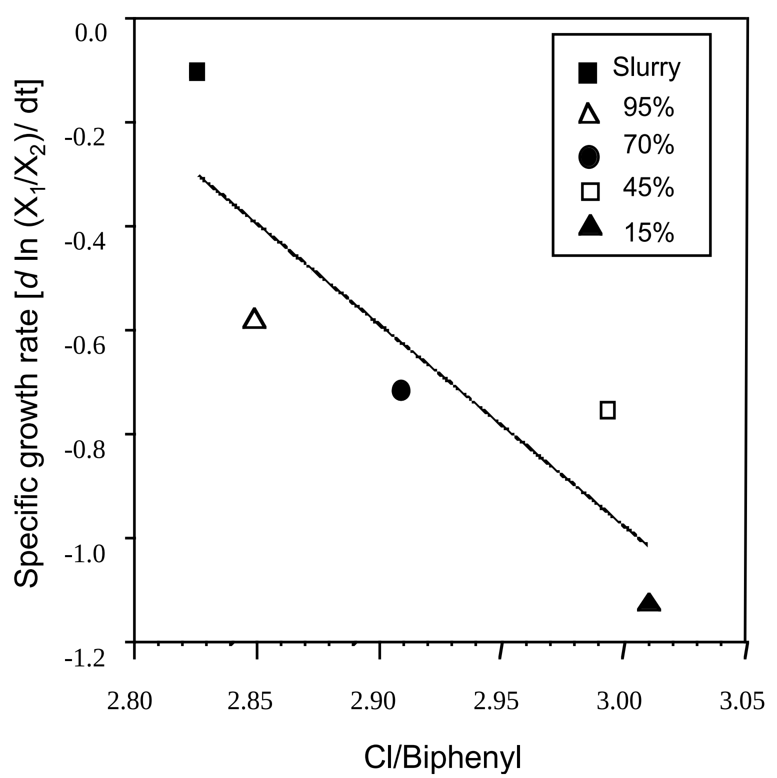
Visual observation indicated a very low moisture level in the dredged sediments at the Moreau site. The importance of moisture content in biodegradation is well known; as the moisture content was reduced below an optimum level, both microbial activity and community size decreased (Barros *et al.*, 1995), and the degradation of various pesticides also slowed (Choi *et al.*, 1988; Davis and Madsen, 1996). Therefore, we carried out investigations using clean sediments spiked with Aroclor 1248 to determine the effects of "dewatering" dredged sediments on PCB dechlorination. The sediments were inoculated with microorganisms eluted from PCB-contaminated sediments from the St. Lawrence River. When active dechlorination was in progress after 8 weeks with 9% reduction of the number of Cls', these sediments were divided into five portions and the moisture content was adjusted to 95, 70, 45, and 15%. Although dechlorination continued at all moisture levels, the final extent of dechlorination after 30 weeks was lower at lower moisture levels. Interestingly, the maximum extent of dechlorination was correlated with the specific rate of decline in dechlorinator populations ($P < 0.05$) (Fig. 4).

Therefore, it appears that at lower moisture levels, dechlorinator populations declined faster, which resulted in lower overall dechlorination.

CONCLUSIONS

There are many lines of evidence that microbial dechlorination of PCBs, the critical first step toward complete degradation by aerobic microorganisms, is driven by the growth of dechlorinating populations: (a) no dechlorination found below the threshold concentration was due to the lack of population growth, (b) in the non-inhibitory concentration range, dechlorination rate without biomass normalization (nmol Cl removed • g sediment^{-1} • day^{-1}) was correlated to the specific growth rate (day^{-1}), (c) the extent of dechlorination in historically contaminated St. Lawrence River sediments after 1 year laboratory incubation appears to be correlated to the initial size of dechlorinator population, and (d) the effect of moisture level on the extent of dechlorination was due to differences in the death rate of dechlorinating microorganisms. There also appears to be various dechlorinating subpopulations with different competence which have different growth requirements. Therefore, the growth enhancement of dechlorinator populations appears to be the key to removing the barrier to aerobic degradation which leads to the ultimate detoxification of PCBs.

Figure 4. *The specific rate of decline of dechlorinating microorganisms and the extent of dechlorination in sediments with various moistur e contents.*



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