PRESQUE ISLE BAY AREA OF CONCERN
Final Stage 3 Remedial Action Plan: Delisting

Pennsylvania Department of Environmental Protection
Office of the Great Lakes
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EXECUTIVE SUMMARY

This Stage 3 RAP presents specific targets and supports a petition to delist the final beneficial use impairment (BUI), fish tumors or other deformities, and the Presque Isle Bay Area of Concern (AOC). Pennsylvania's Department of Environmental Protection (PADEP) with the concurrence of the Presque Isle Bay Public Advisory Committee recommends delisting both fish tumor impairment and the AOC.

Development of the fish tumor delisting target for the Presque Isle Bay AOC was an iterative process. Based upon the recommendations of researchers and other experts during a series of workshops between 2003 and 2006, PADEP sampled a number of inland lakes and non-AOC locations in Lake Erie to identify a 'least-impacted" reference site for comparison. All of the candidate reference sites sampled were known to have brown bullhead populations but no known direct discharges of contaminants. In order to compare the sites over a period of years, a statistical methodology was developed that normalized the tumor rates to those of fish at age 7, the approximate mean age of the bullheads in the full data set. The surveys showed that neither the non-AOC locations in Lake Erie nor the inland Pennsylvania lakes were free of bullhead tumors. Additionally, locations where liver tumors were high had low external rates and vice versa. Long Point Inner Bay was identified as the least-impacted reference site for comparison against Presque Isle Bay. The delisting target selected for Presque Isle Bay is met when "the incidence rate of liver and external tumors is statistically equivalent or lower than the incidence rates at Long Point Inner Bay as confirmed by histopathology".

PADEP used data collected in the post-Recovery Stage to test the delisting target. Comparison of Presque Isle Bay to Long Point Inner Bay showed that the liver tumor rates were not statistically different. In fact, when statistically adjusted for age, it appears that the incidence of liver tumors in Presque Isle Bay bullhead may be a reflection of the broader Lake Erie background rate. The external tumor rate in Presque Isle Bay, however, was statistically significantly higher than Long Point but comparable to all but one of the potential Lake Erie "least-impacted" reference sites evaluated. Based on the limited sample sizes from the potential reference sites, it is difficult to determine whether or not the age-adjusted external tumor rate in Presque Isle Bay is significantly higher, lower, or the same as the background rate elsewhere in Lake Erie. It is true that similar to

Presque Isle Bay, incidences of unexplained external tumors are occurring in populations of brown bullheads in both AOC and non-AOC locations as well as inland Pennsylvania lakes.

PADEP turned its focus to investigating the cause of the external tumors and evaluating the appropriateness of using the tumors as an indicator of environmental degradation. A study designed to detect viruses in external bullhead tumors was inconclusive. A study evaluating whether the bay's bullheads were hybrids and, therefore, potentially predisposed to tumors found little evidence of atypical hybridization. An eighteen month laboratory exposure study did not find biomarkers signifying early stage cancer on any fish exposed to Presque Isle Bay sediment.

PADEP's recommendation to delist the fish tumors or other deformities BUI is grounded on the best science and technology available today. The decision is based on numerous investigations, sampling events, and consultation with the leading experts in brown bullhead investigations. While there is year-to-year variation, since the Recovery Stage designation in 2002 the incidence of liver and external tumors the bay's brown bullhead population has remained stable with little statistical difference in rates between sampling years. Incidence rates of both liver and external tumors remain well below the high levels seen in the early 1990s. Liver tumor rates, the end-point for which exposure to environmental contaminants is more clearly linked to sediment PAH contamination, are statistically indistinguishable from the Long Point Inner Bay reference site. The incidence of external tumors, however, remains elevated when compared to the reference site.

Because there are known legacy contaminants in the sediment regardless of their relationship to the bullhead tumors, PADEP commissioned ecological and human health risk assessments. Using appropriately conservative assumptions and existing data, both risk assessments concluded that cancer and noncancer risks posed by legacy contaminants in the Bay's sediment and fish are below targets for human health and ecosystem protection.

It may not be possible ever to fully restore this BUI due to the external tumors. Reviewing both the International Joint Commission and United States Policy Committee guidelines and principles, it seems clear that external tumors and, to some extent liver tumors, are a lakewide phenomenon. Based upon the data evaluated to select the Lake Erie "least-impacted" reference site and information from other AOCs, whatever is happening in Presque isle Bay is occurring elsewhere in both AOC and non-AOC locations. The rest of the bay's fishery, however, is diverse, abundant, and healthy, appearing unimpacted by whatever is affecting the bullheads.

In recommending the delisting of the AOC, PADEP determined that removal of sediment by dredging the bay is unnecessary, remedial measures with the greatest direct benefit to the bay are done, other watershed measures that positively impact the bay are ongoing, air and water discharges are permitted and monitored, no other species of fish or benthic organism appear to be impacted, and both the human health and ecosystem health assessment concluded that the existing conditions in the bay do not increase either cancer or noncancer risks to people or the environment.

The goal of the AOC program as defined under the Great Lakes Water Quality Agreement is to insure that AOCs, which have been defined as areas where human activities have caused or are likely to cause significant impairment of local beneficial uses of water resources, are improved to the point where their environmental conditions are equal to other non-AOC locations across the Great Lakes. Those conditions may not be pristine but are consistent with the ambient environmental conditions elsewhere in the Great Lakes.

PADEP believes that the RAP process has accomplished its goal to the maximum extent practicable and the ultimate identification of the causes of the external tumors needs to be addressed outside the scope of the AOC program. Based on the decreased and stable tumor rates, review of the available scientific evidence, and in close consultation with local and national experts and the concurrence of the Presque Isle Bay Public Advisory Committee, PADEP recommends delisting the Presque Isle Bay AOC.

1. INTRODUCTION

In 1984, the United States Fish and Wildlife Service received reports from local anglers of brown bullhead catfish (*Ameiurus nebulosus*) with external lesions and/or tumors caught in Presque Isle Bay, Erie, Pennsylvania. In 1991, due in part to concerns about these external anomalies, the United States Department of State designated Presque Isle Bay the 43rd Area of Concern (AOC) under the Great Lakes Water Quality Agreement. Over the next twenty years, federal, state, and local government and academic researchers carried out numerous surveys and investigations of Presque Isle Bay, looking at fish, sediment, water quality, and other indicators of ecosystem health.

As the lead agency for addressing the AOC, Pennsylvania's Department of Environmental Protection (PADEP) is responsible for developing quantifiable targets to measure progress towards restoring the AOC. Working closely with members of the Presque Isle Bay Public Advisory Committee (PAC) and research partners including Pennsylvania Sea Grant, Erie County Department of Health, United States Geological Survey's Leetown Science Center, Texas A & M University, and Pennsylvania State University, PADEP collected considerable evidence to determine whether targets are met and support delisting Presque Isle Bay as an AOC.

This document serves as the Stage 3 Remedial Action Plan (RAP) and provides the data and rationale to support the delisting decision. The focus of this RAP is on the one remaining beneficial use impairment - fish tumors or other deformities. The RAP presents specific targets, summarizes investigations and research, and provides the rationale for delisting the remaining impairment and Presque Isle Bay as an AOC.

2. BACKGROUND

2.1 Great Lakes Approach to Restoring Beneficial Uses

Two agreements between the United States and Canada form the governing framework for monitoring and improving the quality of Great Lakes water resources. First, the 1909 Boundary Waters Treaty set the tone with the creation of the International Joint Commission (IJC). The IJC is an independent, joint Canadian and American federal government agency that provides oversight of the two countries shared water resources. Second, the Great Lakes Water Quality Agreement (Agreement) signed in 1972 expresses the commitment of both countries to restore and maintain the chemical, physical, and biological integrity of the Great Lakes.

A 1987 amendment to the Agreement established criteria for identifying geographical AOCs based on the presence of conditions that "caused or are likely to cause impairment of the area's ability to support aquatic life" (United States and Canada, 1987). The Agreement further defined a beneficial use impairment (BUI) as a "change in the chemical, physical, or biological integrity" of the ecosystem that causes one or more of fourteen listed impairments. The impairments range from the loss of wildlife habitat and the presence of tumors or deformities on fish, to human health conditions related to water contact issues and drinking water standards. The amendment also established the RAP process for systematically restoring impaired beneficial uses in these areas.

The Agreement defines three stages for reporting progress at AOCs: (1) identification of BUIs; (2) selection of remedial and regulatory measures to address the cause(s) and source(s) of the BUIs; and (3) restoration of impairments. In 2001, the United States Policy Committee developed interim milestones to recognize progress between the three stages and provided a set of delisting principles to improve consistency across the Great Lakes basin. The Policy Committee created a "Recovery Stage" designation to acknowledge AOCs where implementation of remedial measures is complete and only time is needed for the ecosystem to respond prior to delisting the individual BUI and/or the AOC (USPC, 2001).

2.2 Presque Isle Bay Area of Concern

Located in northwestern Pennsylvania on the southern shore of Lake Erie, Presque Isle Bay is a 3718 acre shallow embayment with an average depth of 13 feet (Figure 1). It is 4.5 miles long and

1.5 miles across at its widest point. Presque Isle, a seven-mile long recurved sand spit, forms the bay. The southeastern end of the bay connects to Lake Erie through a narrow channel that is maintained by the United States Army Corps of Engineers for navigation. The City of Erie forms the southern and eastern borders of the bay.

The Presque Isle Bay drainage basin is approximately 25 square miles and includes much of the City of Erie as well as portions of Millcreek, Summit, Greene, and Harborcreek townships. The principal tributary streams are Mill Creek including Garrison Run, and Cascade Creek, which together account for two thirds of the water flowing into the bay. Approximately 80% of the watershed is urbanized. The bay is a relatively closed system, and exchange of water with the outer harbor and Lake Erie is restricted by the small harbor opening and low inflow to total volume ratio (PADEP, 1993).

In the 1980s, anglers reported external sores and tumors on brown bullhead catfish caught in Presque Isle Bay. These reports served as a catalyst for concerned citizens to petition for the inclusion of the bay as an AOC. Without citing specific reasons, the United States Department of State designated Presque Isle Bay as the 43rd AOC on January 30, 1991.

PADEP, as the lead regulatory agency for addressing the AOC, proceeded with the RAP process to identify BUIs and explore remedial and regulatory measures to address the cause(s) and source(s) of the BUIs. The evaluation described in the Stage 1 RAP used existing information to identify potential pollution sources and loadings. PADEP identified impaired uses by comparing available data with the fourteen beneficial use impairment guidelines developed by the IJC's Water Quality Board (IJC, 1991). To make these comparisons, PADEP used all relevant data and based impairments on the most compelling set of data or the collective weight of multiple data sets. Through this process, PADEP identified chemicals of potential concern (COPCs), including ten heavy metals, nutrients, chemical oxygen demand, cyanide, oil and grease, and polycyclic aromatic hydrocarbons (PAHs) and concluded that two of the fourteen beneficial use impairments existed: restrictions on dredging activities and fish tumors or other deformities (PADEP, 1993).

Additionally, PADEP noted a limited beach closing beneficial use impairment due to fecal coliform levels at the discharge of the Mill Creek Tube and other stormwater discharge points. A determination could not be made for the guideline addressing plankton populations as no data were available.

PADEP updated the RAP in 1995 to address the outstanding BUI determinations, respond to comments on the 1993 RAP, and clarify that the AOC did not include the outer harbor. Further investigation confirmed the impairment of the dredging and fish tumors beneficial uses and removed the beach closing and plankton population BUIs (PADEP, 1995).

Since the 1980s, PADEP and its partners collected information on fish tumor incidence rates and sediment quality conditions within the bay. Sediment chemistry samples were collected at a number of locations in the bay in 1982, 1986, 1990, 1992, 1993, 1994, 2000, and 2001 (PADEP, 2002). In addition, whole-sediment toxicity tests were conducted on samples collected within the AOC in 1982, 1986, 1994, and 2000 (PADEP, 2002). The sediments were found to contain low level contamination, primarily metals and PAHs, spread throughout the bay. The investigations also indicated that sediment quality conditions were improving in the bay. As a result, PADEP, in conjunction with the AOC's PAC, determined that monitored natural attenuation, rather than active remediation within the AOC, would provide the most cost-effective and practical method for restoring the restrictions on dredging beneficial use. PADEP and the PAC made continuing the reduction of sediment and contaminant loading to the bay a priority, focusing resources on restoration projects within the watershed.

State, federal, and local government agencies conducted numerous studies of the bay's brown bullhead catfish beginning in 1985. In the early 1990s, tumor rates were calculated as a percentage of the total fish collected without accounting for age. Rates were as high as 86% for grossly observable external tumors and 22% for liver tumors. Over the next ten years, tumor rates steadily declined to 19% for grossly observable external tumors and a reported zero percent for liver tumors. Investigators concluded that the overall health of the bay's brown bullhead population had improved dramatically and that external and liver tumor rates were comparable to inland reference lake sites in Pennsylvania. The bullhead population was stable and reproducing (PADEP, 2002).

The improvements in sediment quality, the decade-long downward trend in fish tumors, and the decision not to pursue active remedial measures within the AOC led to the redesignation of Presque Isle Bay to the Recovery Stage in 2002. The new status was a direct result of changes in the watershed, the most significant of which was the $100 million upgrade to the City of Erie's wastewater collection, treatment, and conveyance system. In 1985, the City undertook studies to determine and address the sources of pollution, added a parallel outfall into the Lake, and reduced the number of combined sewer overflows (CSOs) from more than 70 to five. Four of the CSOs discharge into the Mill Creek Tube which empties into the bay. All have screens and flow monitors. Additionally, the City of Erie maintains a litter trap at the end of Mill Creek that catches oil and debris from the CSOs and the stream. The City of Erie reports a CSO capture rate in excess of 99.9%. Additionally, there are no known unpermitted industrial waste discharges to the bay.

Other factors contributing to environmental improvements in the bay include the removal of a coal-fired power plant and wastewater discharge, the shift from industrial to more commercial activities along the bayfront and within the City, and restoration actions taken by local environmental groups throughout the watershed

2.3 Delisting Restrictions on Dredging Activities BUI

In 2005, a comprehensive sediment study began to assess the restrictions on dredging activities BUI. It incorporated a review of all existing sediment data, particularly data used to make dredging and disposal decisions, collection of surface and subsurface sediment samples, and identification of both delisting and ecosystem health targets. The assessment of the restrictions on dredging activities BUI included both practical and ecological perspectives. The practical restriction is based on Pennsylvania's laws and regulations, which preclude the disposal of the dredged material in the open lake regardless of contaminant presence or absence. This restriction is due to the fact that dredged material is defined as a solid waste and there are limitations associated with locating a disposal facility in waters of the Commonwealth. Disposal to the Confined Disposal Facility (CDF) or an upland site are the only allowable options. Because the restrictions on disposal of dredged material are not related to sediment contamination, but rather laws preventing the disposal of solid waste in waters of the Commonwealth, from the practical perspective the beneficial use is not considered impaired (PADEP, 2006).

From an ecological perspective, the sediment in the Presque Isle Bay AOC was evaluated against a delisting target based on discharges from the disposal of dredged material in the CDF (Table 1). The target takes into account the limitation on disposal options and current permitting practices by evaluating discharges from the CDF. The delisting target requires concentrations of chemicals of potential concern in the CDF's mixing zone to be below Pennsylvania Water Quality Standards at the 15-minute compliance point for acute criteria and the 12-hour compliance point for chronic criteria. Using elutriate data for areas routinely dredged within the AOC and calculations to predict concentrations in the CDF discharge based on concentrations in the sediment, it was determined that sediment dredged from any location within the AOC could be placed in the CDF.

The 2005 survey data was also used to evaluate sediment quality following the 2002 Recovery Stage designation. Ecosystem health targets were identified for benthic organisms, fish, and aquatic-dependent wildlife (Table 1). While concentrations of individual contaminants did, in limited locations, exceed sediment quality guidelines, and there is a potential for PAHs to be bioavailable to benthic organisms, actual toxicity tests did not confirm the predicted toxicity. The evaluation concluded that existing sediment quality conditions are sufficient to support benthic invertebrate communities and risks to fish and aquatic-dependent wildlife using habitats within the AOC are unlikely to be higher than that elsewhere in Lake Erie (PADEP, 2006). As a result of both the practical and ecological evaluations, the restrictions on dredging activities BUI was delisted in 2007.

In August and September 2009, to monitor ongoing compliance with the delisting target and ecosystem health targets, surficial sediment samples were collected from seven historical sampling locations within Presque Isle Bay, two historical sampling sites outside of the bay, and three locations within the bay where brown bullhead are routinely collected for tumor analysis. In addition, sediment samples were collected from the mouths of Mill Creek, Scott Run, and Cascade Creek in an effort to characterize the concentrations of contaminants deposited in the streams following rain events.

The sedimentation rate in the bay averages one centimeter per year, suggesting that approximately four centimeters of new sediment accumulated in the four years between sampling events. As a result, a significant change in sediment quality was not expected or observed. Analysis of data showed that the delisting target for the restrictions on dredging BUI continues to be met. There were no exceedences calculated for the discharge from the CDF (Rafferty and Boughton, 2012). Concentrations of Chemicals of Potential Concern (COPCs) varied between sampling events and the same PAH compounds were found to exceed Sediment Quality Guidelines (SQGs) in both events. Overall, sediment quality was seen to improve as evidenced by the fewer number of samples with contaminants exceeding SQGs in 2009. Pesticides, PCBs, and arsenic were not detected in concentrations exceeding SQGs in any of the 2009 samples, indicating that these compounds are not present at levels that would impact ecosystem health. The contaminant mixtures present did not contain COPCs in concentrations that would cause adverse impacts on benthic organisms. Metals present are binding to organic carbon and not bioavailable. There is a potential for PAHs to be bioavailable to benthic organisms. However, this measure has improved since 2005 where a higher number of sites exceeded ecosystem health targets. The ecosystem health target evaluating the potential of COPCs to be present at levels toxic to fish remained unchanged between the two sampling events.

Samples collected from the tributaries above the mixing zone with the bay had more exceedences of SQGs for PAHs than locations in the AOC. However, measures of bioavailability were similar to that found at the long-term monitoring sites, indicating that particle size and total organic carbon are limiting the availability of the contaminants to benthic organisms. The 2009 study confirmed that sediment quality continued to improve, the delisting target was being met, and the restriction on dredging activities beneficial use continued to be unimpaired (Rafferty and Boughton, 2012).

3.0 UPDATING THE BENEFICIAL USES EVALUATION

The 1993 Stage 1 RAP presents a detailed evaluation of the fourteen BUIs. That assessment is now more than twenty years old and conditions within the bay and its watershed have changed. While more than twenty years of data is available on the fish tumors or other deformities and restrictions on dredging activities impairments, the other twelve BUIs have not been reassessed. Over the years, surveys and studies for other purposes have collected data that can be used to re-evaluate the twelve BUIs not considered impaired in the 1993 RAP. PADEP reexamined those twelve beneficial uses using the most recent data available and confirmed that these BUIs remain unimpaired.

3.1 Restrictions on Fish and Wildlife Consumption

IJC Listing Criteria

When contaminant levels in fish or wildlife population exceed current standards, objectives, or guidelines, or public health advisories are in effect for human consumption of fish or wildlife. Contaminant levels in fish and wildlife must be due to contaminant input from the watershed.

Assessment

PADEP conducts routine analyses of fish flesh for the presence of PCBs, metals, and pesticides in both Presque Isle Bay and the open waters of Lake Erie as part of its base fish consumption advisory program. PADEP does not assess wildlife tissue. Consumption advisories based on elevated levels of PCBs and mercury are in place for fishes in both Presque Isle Bay and Lake Erie. While comparisons between the bay and open lake are difficult due to differences in species composition and migrations into and out of the bay by common species, there is no evidence that fish species in Presque Isle Bay are more contaminated than in Lake Erie.

Mercury levels in the bay's largemouth bass fell from over 0.3 ppm in 1996 to less than 0.25 ppm in samples taken in 2001, 2005, and 2006. Because of this trend, the consumption advisory was adjusted from two meals per month to one meal per week beginning in 2007 (Figure 2). Mercury and PCB concentrations in Presque Isle Bay yellow perch, believed to be a resident population, are comparable to concentrations in perch collected from Lake Erie (Figure 3). PCB concentrations in Presque Isle Bay common carp, believed to be resident population, are lower than concentrations in carp collected from Lake Erie. However, sample sizes are very small (two from each location) and the Lake Erie fish in particular need to be resampled due to data quality issues with the laboratory.

Generally, the species with the highest contaminant burdens (walleye, steelhead, lake trout, and smallmouth bass) reside either exclusively or primarily in the open lake.

Conclusion

While contaminant levels in fish do exceed current standards and there are consumption advisories, concentrations of PCBs and mercury in fish sampled from the bay are equal to or less than the same species sampled from the open waters of Lake Erie. The IJC's listing and delisting guidelines (IJC, 1991) specifically states "when a health advisory on fish in a localized area is no different from the health advisory for the whole lake and this area is not contributing to a whole lake problem, then it would not be recommended for identification of an AOC". Because the consumption advisories are not a result of bay-specific conditions, the Restrictions on Fish and Wildlife Consumption beneficial use is not considered impaired in Presque Isle Bay.

3.2 Tainting of Fish and Wildlife Flavor

IJC Listing Criteria

When ambient water quality standards, objectives, or guidelines, for anthropogenic substances(s) known to cause tainting, are being exceeded or survey results have identified tainting of fish or wildlife flavor.

Assessment

Impairment of this guideline is indicated if (1) water quality standards for tainting substances are being exceeded or (2) tainting of fish or wildlife flavor is determined through surveys. PADEP consulted the Pennsylvania Fish and Boat Commission (PFBC) regarding any complaints or notes of fish or wildlife with undesirable taste or odor. As of June 2012, the Commission's local Waterways Conservation Officer reported no public complaints in the thousands of angler surveys conducted during the past two decades.

In order to be consistent with the original assessment of this BUI, Presque Isle Bay Water Quality Network Station 632 trend data for copper and zinc levels were compared to PADEP's 25 Pa Code Chapter 16 Water Quality Criteria (WQC) . Copper was below analytical quantification levels for 40 network water samples collected between 2002 and 2011. Detectable levels of zinc were present in

15 of 40 samples for the same time period. PADEP's WQC for metals are calculated based on the hardness of the ambient water. Given the average $CaCO_3$ hardness of 117 mg/L in the bay for this period, Chapter 16 standards were exceeded for two of these samples (Figure 4).

Conclusion

Based on the ten most recent years of data from the Water Quality Network Station within the bay, there is no evidence of chronic or acute violation of taste and odor standards indicated by concentrations of copper and zinc. Therefore, the Tainting of Fish and Wildlife beneficial use is not impaired in Presque Isle Bay.

3.3 Degraded Fish and Wildlife Populations

IJC Listing Criteria

When fish and wildlife management programs have identified degraded fish or wildlife populations due to a cause within the watershed. In addition, this use will be considered impaired when relevant, field validated, fish or wildlife bioassays with appropriate quality assurance/quality controls confirm significant toxicity from water column or sediment contaminants.

Assessment

PADEP and PFBC fish survey data were used to ascertain the diversity and abundance of the fish populations in Presque Isle Bay (Figure 5). PADEP has documented 54 species of fish in the bay, most of which are minnows and other small forage fishes, including the Brook Silverside, a Pennsylvania endangered species that is uncommon outside the bay. Surveys have also found a number of other state-endangered species, including bigmouth buffalo, warmouth, and spotted gar—a species which occurs nowhere else in the Commonwealth. A 2012 survey by PADEP suggests that the state endangered Iowa Darters are increasing in relative abundance.

Evidence of the health of the bay's fishery is further demonstrated by the PFBC's 2008 black bass assessment. Over a three day period, a total of 693 bass were captured consisting of 675 largemouth bass and 18 smallmouth bass. Spring 2008 marked the highest number of bass ever sampled in Presque Isle Bay and the total number captured was a 65% increase over 2007. The occurrence of largemouth bass 12 inches and longer increased 150% from 2007 and the occurrence of largemouth bass 15 inches and longer increased 245%. The catch rate for largemouth over 15 inches was the highest observed in the last 18 years. The biggest risk to the bay's fishery is the continued introduction of non-native invasive species (Figure 6). In 2011, the tubenose goby, a cousin of the invasive round goby, was documented in the bay.

In 1992 and 1999 researchers conducted population studies on the bay's brown bullheads. Using mark-recapture methods, the 1992 estimate was 31,715 and the 1999 estimate was 30,950, suggesting a stable population.

Conclusion

There is no evidence of population-level impacts for any fish species found in Presque Isle Bay, including brown bullhead catfish. Therefore, the Degraded Fish and Wildlife Populations beneficial use is not considered impaired in Presque Isle Bay.

3.4 Bird or Animal Deformities or Reproductive Problems

IJC Listing Criteria

When wildlife survey data confirm the presence of deformities (e.g., cross-bill syndrome) or other reproductive problems (e.g., egg-shell thinning) in sentinel wildlife species.

Assessment

While no formal surveys have been conducted in the last twenty years, the bay and Presque Isle State Park are extensively visited by both amateur and professional nature watchers. The bay and park are part of an important migratory path for birds. Since 2008, the Presque Isle Audubon Society sponsors a one day bird count on Presque Isle State Park. Over three days, volunteers tally the number and species of birds on and over the park. In 2012, volunteers identified 146 species with 25 species of warblers. A running list of species identified at the park is posted on the Society's web site. There are no indications of either deformities or reproductive problems noted. A number of researchers from local and state academic institutions conduct research within the bay and on the park. No reports or other evidence of deformities or reproductive problems have been documented. Populations of other animals at the park are thriving, requiring a deer hunt every year to thin the herd.

Conclusion

There is no evidence of bird or animal deformities or reproductive health problems in the Presque Isle Bay AOC or surrounding watershed. Therefore, the Bird or Animal Deformities or Reproductive Problems beneficial use is not considered impaired.

3.5 Degradation of Benthos

IJC Listing Criteria

When the benthic macroinvertebrate community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when toxicity (as defined by relevant, field-validated, bioassays with appropriate quality assurance/quality controls) of sediment associated contaminants at a site is significantly higher than controls.

Assessment

In evaluating this beneficial use, it is important to note that benthic macroinvertebrate community composition in lakes and bays is very different from that in streams. As a result of habitat differences, even healthy lake communities will be dominated by midges and aquatic worms rather than the mayflies, stoneflies, and caddisflies that dominate healthy flowing streams. PADEP reviewed two different assessments of the bay's benthic macroinvertebrate community. The first (Diz, 2002), examined the benthic community structure, looked for chironomid mouthpart deformities, and conducted sediment toxicity bioassays using the benthic macroinvertebrates *Hyallela azteca* and *Chironomus tentans* and the planktonic crustacean *Daphnia magna.* The author concluded:

- The Presque Isle Bay benthic community is dominated by pollution-tolerant organisms, such as worms, midges, and snails, and is relatively lacking in those species which are known to be sensitive to stressful conditions, such as mayflies and caddisflies. However, the taxa found in Presque Isle Bay are typical of the benthic fauna found in northwestern Pennsylvania lakes.
- Bioassays showed no impact to the survival of the test organisms

- The occurrence of mouthpart deformities in midges is an indication of sediment toxicity. From each of nine sediment sample sites in Presque Isle Bay, 10 *chironomid* individuals were chosen at random. Of the 90 total chironomids examined, only one exhibited a mouthpart deformity.

In 2005, ten day and 28-day whole sediment toxicity tests were conducted with the midge *Chronomus dilutes* and the amphipod *Hyallela Azteca.* Thirty four surficial sediment samples were used to evaluate survival and growth endpoints. None of the samples were toxic to amphipods for either endpoint. One sample from the center of the AOC was toxic to midges when the survival endpoint was considered and three were designated toxic using the growth endpoint (PADEP, 2006). Three of the four samples toxic to midges did not have measured concentrations of contaminants expected to be toxic to the benthic organisms. At these locations, factors such as ammonia or hydrogen sulfide in the pore-water or other factors not related to the chemical contaminants in the sediment were believed to cause the observed toxicity.

Conclusion

Presque Isle Bay's benthic macroinvertebrate community is dominated by pollution-tolerant organisms, such as worms, midges, and snails which is typical for an environment of fine, organic-rich sediment. Direct testing found a limited number of sediment samples were toxic and it is believed the toxicity is due to non-contaminant related factors. Based on these studies, the Degradation of Benthos beneficial use is not considered impaired in the Presque Isle Bay AOC.

3.6 Eutrophication or Undesirable Algae

IJC Listing Criteria

When there are persistent water quality problems (e.g. dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, decreased water clarity, etc.) attributed to cultural eutrophication.

Assessment

PADEP conducts periodic trophic state index (TSI) assessments, annual summer plankton (zooplankton and phytoplankton/algae) sampling, and dissolved oxygen monitoring in Presque Isle Bay. TSI surveys involve collecting measures of plant productivity such as phosphorus levels, chlorophyll-a levels (the photosynthetic pigment in plants and algae), and the clarity of the water. TSI results are used to classify lakes and bays as either oligotrophic ("poorly fed"), mesotrohic ("moderately fed") to eutrophic ("well fed") to hypereutrophic ("beyond well fed").

The Carlson TSI score for the bay was last determined in 2005. The bay received a TSI score of 52 at that time, placing it in the *low eutrophic* range (Figure 7). This score suggests that the bay has good biological productivity but may be vulnerable to problems related to nutrient enrichment. This score does not suggest conditions are currently suitable for nuisance plant and algae growth.

One of the simplest ways to track the trophic state of a lake is to track the water clarity over time. Clearer water has less algae and suspended particles, while "cloudier" water tends to have more algae and suspended material. Trends indicate that water clarity has improved slightly during the past two decades (Figure 8).

Unlike most Pennsylvania lakes, Presque Isle Bay does not completely stratify into a warmer upper bay and cooler lower bay in the summer. While it functions somewhat independently of the rest of Lake Erie, Presque Isle Bay is actually part of the warmer "epilimnion", or upper lake, of Lake Erie. Therefore, unlike the central basin of Lake Erie proper, there is always some dissolved oxygen present in the bottom of the bay for fish and other aquatic life.

Bluegreen cyanobacteria blooms (especially *Microcystis aeruginosa*) typically comprise Harmful Algal Blooms in the Great Lakes. Nuisance levels of *Microcystis* have not been reported from Presque Isle Bay. Nonetheless, trend monitoring for the bluegreens *Anabaena* and *Microcystis* show that periodic blooms have occurred in Presque Isle Bay at levels approaching those in the western basin of Lake Erie. Most recently, elevated levels of *Microcystis* were noted in 2005, 2006, and 2011 (Figure 9).

Conclusion

This guideline evaluates whether there are persistent water quality problems due to nutrient enrichment. The bay is indicative of the larger lake and that there is not an excessive runoff of nutrients into the bay from Erie or the surrounding area. Algal blooms are the same in the bay as lakewide conditions. Annual assessments of water quality, algae, and oxygen levels in the bay have confirmed that cultural eutrophication is not occurring. In fact, there has been a trend of improving water clarity over the last two decades. Based on the trophic status of the bay, the increased water clarity, and lack of persistent algal blooms, Presque Isle Bay is not considered impaired for Eutrophication or Undesirable Algae.

3.7 Restrictions on Drinking Water, or Taste and Odor Problems

IJC Listing Criteria

When treated drinking water supplies are impacted to the extent that: 1) densities of disease-causing organisms or concentrations of hazardous or toxic chemicals or radioactive substances exceed human health standards, objectives, or guidelines; 2) taste and odor problems are present; or 3) treatment needed to make raw water suitable for drinking is beyond the standard treatment used in comparable portions of the Great Lakes which are not degraded (i.e., settling, coagulation, disinfection).

Assessment

Presque Isle Bay is not used as a source of drinking water. The City of Erie's drinking water intakes are both located in Lake Erie west and north of the Presque Isle Peninsula. Additionally, the City has an ordinance that prohibits the use of wells or springs located on a property to be used as a source of drinking water.

Conclusion

The Restrictions on Drinking Water Consumption, or Taste and Odor Problems beneficial use is not applicable or impaired in Presque Isle Bay.

3.8 Beach Closings

IJC Listing Criteria

When waters, which are commonly used for total-body contact or partial-body contact recreation, exceed standards, objectives, or guidelines.

Assessment

Although there are no designated public beaches within Presque Isle Bay, water samples are collected bi-weekly or weekly during the summer months, depending upon the location within the bay, and analyzed for *E.coli.* The monitoring began in 2007 with three sets of samples and has, in the last five years, expanded to sampling at twelve different locations. The Department of Conservation and Natural Resources (DCNR) issues beach advisories for Lake Erie Beaches when *E. coli* counts exceed 235 CFU/100 mL and restrict swimming when counts exceed 1000 CFU/100 mL. PADEP used these standards to compare the AOC to Lake Erie. Samples from Presque Isle Bay, collected at the mouth of Cascade Creek, south of the City of Erie's Wastewater Treatment Plant in Garrison Run, and at the mouth of Scott Run are consistently higher than other locations in the bay (Figure 10). Since 2009 the yearly average concentrations of *E. coli* in samples from these locations exceed the level for beach advisories but not the level for restricted swimming. No other locations exceeded either criterion. As a testament to the improved water quality conditions, the Presque Isle Partnership sponsors a one mile swim across the Bay from Presque Isle State Park to the Erie Yacht Club. Since its inception in 2008, every year 200 swimmers participate in the swim.

Conclusion

There are no public beaches within the Presque Isle Bay AOC and therefore, this beneficial use does not apply. However, comparison of the last five years of *E. Coli* sampling with the criteria for restricting swimming at the public beaches on Presque Isle State Park show that the beach closing beneficial use would not be impaired in Presque Isle Bay.

3.9 Degradation of Aesthetics

IJC Listing Criteria

When any substance in water produces a persistent objectionable deposit, unnatural color, or turbidity, or unnatural odor (e.g., oil slick, surface scum).

Assessment

PADEP is occasionally called upon to investigate an unusual odor, color, or plume within the bay. In many cases, the conditions are natural due to weather (i.e., seiche, heavy rain storm or high wind) which may cause turbid conditions. There are also occasions when a surface sheen is noted due primarily to boater use, marinas, and inputs from the watershed.

Conclusion

Temporary impacts to aesthetics typical of urban embayments are noted within Presque Isle Bay. Because the conditions are not persistent and do not significantly impact the bay, the degradation of aesthetics beneficial use is not considered impaired.

3.10 Added Costs to Agriculture or Industry

IJC Listing Criteria

When there are additional costs required to treat the water prior to use for agricultural purposes (i.e., including, but not limited to, livestock watering, irrigation and crop-spraying) or industrial purposes (i.e., intended for commercial or industrial applications and noncontact food processing).

Assessment

Water from Presque Isle Bay is not used by agricultural or industrial operations.

Conclusion

The beneficial use associated with costs to agriculture and industry is not impaired in Presque Isle Bay.

3.11 Degradation of Phytoplankton and Zooplankton Populations

IJC Listing Guideline

When phytoplankton or zooplankton community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when relevant, field-validated, phytoplankton or zooplankton bioassays (e.g. Ceriodaphnia; algal fractionation bioassays) with appropriate quality assurance/quality controls confirm toxicity in ambient waters

Assessment

PADEP samples the plankton community at WQN monitoring stations in Presque Isle Bay (WQN 632) and Lake Erie (WQNs 601 and 622). It is difficult to compare plankton trends due to natural annual and seasonal variation in the community composition. The plankton communities at all three monitoring sites have been degraded by the establishment of non-native species. These include the cyanobateria (bluegreen algae) *Lyngbya* which is known to cause harmful algal blooms in the western basin in Lake Erie as well as the planktonic larvae of zebra and quagga mussels. The mean number of plankton taxa in Presque Isle Bay (25.25) is not significantly different than Lake Erie WQNs 601 (25.5) or 622 (25.75) and the qualitative list of taxa present is virtually identical to Lake Erie. However, Presque Isle Bay is warmer, shallower, and more productive than the open waters of Lake Erie. As a result, the plankton in the bay is more abundant than in the open lake. The annual timing and succession of phytoplankton blooms in the bay is also somewhat accelerated relative to the open lake (Figure 11). *Microcystis* blooms are larger in the warmer waters of Presque Isle Bay than in the open lake. However, *Microcystis* is present at all the monitored sites (Figure 12). Plankton communities in the both bay and lake are degraded to a degree due to non-native species. Despite the presence of non-native species, the plankton communities in Presque Isle Bay are sufficient to support one of the most diverse and abundant fisheries in the Commonwealth.

In 2002, the planktonic crustacean *Daphnia magna* along with the benthic macroinvertebrates *Chironomus tentans and Hyallela azteca* were used in a bioassay of the sediment quality in Presque Isle Bay. Nine replicate toxicity tests were conducted. There was no significant difference in survival between *Daphnia* exposed to Presque Isle Bay sediment and the control. However, reproduction was significantly less for Daphnia exposed bay sediment in 7 of the 9 bioassays.

Conclusion

Despite some evidence of reduced *Daphnia* reproduction in the bioassay by Diz (2002), Presque Isle Bay plankton populations are as taxonomically rich as in Lake Erie and abundance/biomass is greater. Therefore, the Degradation of Phytoplankton and Zooplankton Populations BUI does not occur in Presque Isle Bay.

3.12 Loss of Fish and Wildlife Habitat

IJC Listing Delisting Criteria

When fish and wildlife management goals have not been met as a result of loss of fish and wildlife habitat due to a perturbation in the physical, chemical, or biological integrity of the Boundary Waters, including wetlands.

Assessment

The 1993 Stage 1 RAP states that the PFBC is the agency involved in setting fish and wildlife management goals. The PFBC manages Presque Isle Bay as a sport fishery and conducts periodic surveys. As discussed under the Degraded Fish and Wildlife Populations BUI, the bay's fishery is very diverse and high quality, meeting management goals as a sport fishery.

Conclusion

Based upon the most recent survey data from PFBC and PADEP, the Loss of Fish and Wildlife Habitat beneficial use is not considered impaired in Presque Isle Bay.

4. FISH TUMORS OR OTHER DEFORMITIES BUI

4.1 Historical Perspective

Beginning in the late 1980s, Great Lakes researchers attempted to define quantifiable delisting targets for the fish tumors or other deformities BUI based primarily on the incidence rate of liver tumors and external deformities. Researchers considered fish tumors an indicator of both environmental degradation and a measure of health impairment to fish populations (Baumann, 1992a). Of the original 42 AOCs, 17 had fish with readily identifiable tumors or deformities.

The first attempt to define the fish tumor impairment in "precise set of scientifically defensible" criteria resulted in an IJC recommendation that "the incidence rate of neoplastic and preneoplastic liver tumors in bottom-feeding fishes not exceed 2 percent" (Hartig et al, 1990). The basis for this recommendation was the assumption that certain contaminants cause tumors in fish and a presumption that fish from uncontaminated locations should have a zero liver tumor incidence rate. A review of existing data from the Great Lakes and Puget Sound; however, showed that liver tumors develop in fish from uncontaminated sites. A two percent tumor rate accounts for this fact as well as uncertainties in fish movement and factors other than exposure to contaminants that promote tumors (Michael J. Mac, 2009 personal communication).

The IJC modified this recommendation, publishing guidelines that suggest the fish tumor or other deformities beneficial use impairment exists when "the incidence rates of fish tumors or other deformities exceeds rates at unimpacted control sites or when survey data confirm the presence of neoplastic or preneoplastic liver tumors in bullheads or suckers" (IJC, 1991). The IJC listing/delisting guidelines were developed to assist in making recommendations for listing new AOCs and in reviewing Remedial Action Plans. The intention was to establish a "set of yardsticks" that could be applied throughout the basin and keep the Remedial Action Plan program focused. They are written as guidelines to serve as a common starting point for each AOC. The IJC constructed the guideline to allow each AOC to adapt it to site-specific conditions in setting delisting targets (e.g., selection of unimpacted control site and which fish species to use as an indicator).

Subsequent studies throughout the 1990s, by both American and Canadian researchers recommended specific numeral targets for liver and external tumor incidence rates as indicators of environmental degradation. For example, a review of data collected between 1984 and 1993 from ten contaminated and three reference locations from across the Great Lakes concluded that liver tumor incidences above 5% and external tumor incidences in excess of 25% were evidence of impairment (Baumann et al., 1996). A comparison of "least impacted control sites" and contaminated embayments, river mouths, and nearshore areas within Lake Erie defined impairments when liver tumor incidences were above 5-7% and external tumor incidence were above 13-15% (Baumann et al., 2000).

4.2 Delisting Target for the Presque Isle Bay AOC

Development of the fish tumor delisting target for the Presque Isle Bay AOC has been an iterative process. Following the recommendations of Great Lakes researchers, delisting criteria were based on having a liver and external tumor incidence rate on brown bullhead below a specified target number. The 1993 RAP concluded that the fish tumor or other deformities beneficial use was impaired because liver tumor rates exceeded the IJC's 2% benchmark and external abnormalities were in excess of 10-12% (PADEP, 1993). The 2002 Recovery Stage redesignation was due in part to the decreasing liver and external tumor rates. Additionally, the tumor rates were below the recommended indicators of environmental degradation of 25% for external tumors and 5% for liver tumors. While these targets represented good starting points for developing delisting criteria, a number of questions were raised regarding the comparability of data from different years and locations and whether contaminants in the sediment were the cause of the tumors. Through a series of workshops between 2003 and 2006, PADEP and the PAC sought advice from experts including fishery and wildlife biologists, pathologists, representatives from other AOCs, and researchers. Consensus was reached on a broad range of sampling and analysis issues, including the following recommendations:

- Samples should include only brown bullheads that are a minimum length of 250 mm (9.9 inches) to exclude younger specimens. Length and age studies show that brown bullheads greater than 250 mm are at least age three and likely to be reproductively mature (Maceina and Sammons, 2006).

- Ages should be determined for all necropsied bullhead using otoliths rather than pectoral spines because otolith-based ages are more accurate.
- Given the strong positive correlation between bullhead age and tumor development, it is important to compare fish of the same age to evaluate temporal trends and differences between locations.
- Examination of the fish should include both gross visual observation and histopathology.
- Both external and liver tumor incidence rates should be determined for beneficial use restoration purposes. However, special studies may look at other internal organs as well.
- It is important to examine multiple sections from each liver to ensure that any tumor present in the organ will be detected.

One of the major outcomes of the workshops was the decision to compare tumor incidence rates in Presque Isle Bay to that of reference sites using comparable fish collection (e.g., Rafferty and Grazio, 2006) and histopathology methods (e.g., Blazer et al., 2006). Unfortunately, the majority of data from past studies at the reference locations could not be used as the collection and histopathology methods employed in those studies are not comparable to those used in Presque Isle Bay. In particular, the majority of past studies used pectoral spine-based age determinations and these age determinations are not comparable to the otolith-based age determinations used in the bay. In addition, histopathological methods varied among studies. A consequence of this decision was the need to determine new, more realistic delisting tumor incidence targets.

Following expert recommendations, PADEP decided not to select any of the specific incidence rates recommended historically because those rates did not account for important factors such as the age of the fish population. Instead, PADEP chose to focus the delisting target on a comparison of the liver and external tumor incidence in Presque Isle Bay to an appropriate Lake Erie reference or "least-impacted control site". In order to identify the least-impacted reference site, PADEP sampled a number of candidate sites from across Lake Erie. All candidate sites were non-AOCs that lacked point-source discharges of pollutants or known sediment contamination and had a resident bullhead population.

Based on this premise, PADEP collected samples in 2004, 2005, and 2007 from Dunkirk Harbor, NY; Long Point Inner Bay, ON; Old Woman's Creek, OH; and Sandusky Bay, OH (Figure 13). Sample evaluation included gross visual observation for all fish collected and histopathological analysis of any raised external or mouth lesions and all livers for a subset of 30-50 fish at each location.

In evaluating the data, PADEP incorporated recommendations from the expert workshops, specifically the need to compare fish of the same age to evaluate temporal trends and differences between locations. Historically, PADEP reported tumor incidence rates based on dividing the number of fish with tumors by the number of fish sampled. This approach did not take into account the demographics (e.g., age, length, gender, etc.) of the sample. A statistical model was developed that used logistic regression which allowed for the comparison of tumor incidence between sites by taking the age and length of each bullhead into consideration when determining the probability of a tumor (Rutter, 2010). Applying a Bayesian hierarchical model, the results of the logistic regression can be easily presented as point estimates and intervals of biological terms (i.e., the probability that a bullhead has a tumor).

In Presque Isle Bay, tumor rates increase with fish age. PADEP used logistic regression to measure tumor incidence rates as a function of age and Bayesian statistical analysis was used to compare incidence rates between ages, and account for multiple sampling locations and dates. This approach does not directly compare cohorts of fish, but rather allows for the determination of a point estimate of tumor incidence rates for fish of a given age. Age seven was chosen because this was the approximate mean age of bullhead in the full dataset (Rutter, 2010). The 95% confidence interval describes the "certainty" of the point estimate with narrow intervals indicating a more accurate point estimate for the tumor incidence rate.

The candidate site with the lowest liver tumor incidence (0.0%) was Dunkirk Harbor and the candidate site with the lowest external tumor incidence was Long Point Inner Bay (6.4%; Table 2). A closer examination of the Dunkirk Harbor tumor incidence indicated a high level of uncertainty in the estimate based on the 95% confidence intervals around the mean incidence rate (0.0%, 56.0%). Dunkirk Harbor also had the highest incidence of external tumors (22.5%). The second lowest liver tumor incidence among the reference sites was Long Point Inner Bay (1.2%) and the 95% confidence interval (0.0%, 14.9%) was much narrower indicating less uncertainty in the estimate. Based on combined external and liver tumor incidence rates, Long Point Inner Bay was selected as the least-impacted Lake Erie reference site and the appropriate reference site for Presque Isle Bay

Proposed Presque Isle Bay Delisting Target

The fish tumor or other deformities beneficial use is no longer considered impaired when the incidence rate of liver and external tumors is statistically equivalent or lower than the incidence rates at Long Point Inner Bay as confirmed by histopathology.

4.3 Testing the Delisting Target

Following the 2002 Recovery Stage designation, PADEP conducted annual monitoring surveys through 2010 to assess tumor trends. The assessment included both gross visible observation and histopathology of raised external lesions and livers. Applying the same logistic regression and Bayesian statistical analysis developed in evaluating the Lake Erie potential reference sites, the data were normalized to age seven years for consistency in reporting and comparability between sampling years and with Long Point Inner Bay. .

Both the liver and external tumor incidence rates were found to be stable throughout the Recovery Stage monitoring period (Table 3; Figures 14 and 15). The incidence rate of liver tumors ranged from 1.1% in 2002 to 3.9% in 2007 with a median liver tumor incidence rate for this period of 2.8 % (Table 3). It is important to note that the five years of data does not provide enough data points to determine a trend, rather it shows inter-annual variation which is most likely an artifact of the random sampling methodology than a true difference in tumor incidence rates. Also, when the yearly estimates are examined, there is no statistically significant trend. Using the Bayesian 95% confidence intervals to measure certainty, there were no statistical differences in the liver tumor incidence rates among years or among the various Presque Isle Bay collection sites.

Brown bullhead from Presque Isle Bay sites had similar grossly observed external lesion incidence rates to those collected at the Lake Erie potential reference sites. External tumor rates confirmed by histopathology ranged from 11.9% in 2005 to 18.9% in 2004 with a median external tumor incidence rate for this period of 15.4 %.(Table 3). As was the case with the bay's liver tumor rates, there was not enough data to report a downward or upward trend and there were no statistical differences in the external tumor rates among years or among collection sites.

PADEP conducted gross visual observations of the bullheads collected in the AOC following the Recovery Stage designation through 2010 (Figure 16). While not confirmed by histopathology, looking at the combined external and mouth tumors, the data shows a relatively stable, downward trend in external tumors over time. Looking first at the post-Recovery Stage between 2002 and 2007, the median external tumor rate grossly observed for bullheads estimated by length to be age seven was 20.5%. This is considerably higher than the rate as confirmed by histopatholgy, illustrating how gross visual observation overestimates the actual tumor incidence rate.

Over the five years in which Presque Isle Bay was sampled, 222 brown bullheads from the bay were necropsied and analyzed for liver and external tumors. Sampling of Long Point Inner Bay occurred in 2004, 2005, and 2007 with a total of 193 brown bullheads collected for analysis. Statistically, there were enough samples from each location to evaluate whether the tumor rates were equivalent. Rather than compare the two medians, the confidence interval estimating the difference in true tumor rates was determined (Rutter, 2010). When the confidence interval contained zero and was small, the two tumor rates could be considered statistically equivalent. The narrowness of the confidence interval was also important in determining whether the tumor rates were equivalent or different. If the confidence interval describing the difference in tumor rates was too large or did not contain zero, then the tumor rates were considered statistically significantly different (Rutter, 2010).

Looking first at the median liver tumor rates and confidence intervals on Tables 2 and 4, Presque Isle Bay and Long Point Inner Bay's 95% confidence intervals were narrow and overlap. The results of the statistical analysis indicated that the distribution of liver incidence on a standardized brown bullhead (300 mm and age 7) in Presque Isle Bay was almost identical to Long Point Inner Bay's rate when the confidence interval describing the difference in tumor rates was examined. This means the liver tumor incidence rates at these locations were statistically equivalent and the delisting target was being met.

The same was not true for the external tumors. The confidence intervals for external tumor incidence in Presque Isle Bay and Long Point Inner Bay overlap, but the interval for Presque Isle Bay was much wider and the median tumor incidence rate at 15.4% was much higher. When the confidence interval describing the difference in tumor rates between Presque Isle Bay and Long Point Inner Bay was examined it was too large to indicate that two tumor rates were statistically equivalent. So, the differences in external tumor rates in Presque Isle Bay and Long Point Inner Bay were not within the range of values that would be considered statistically equivalent (Rutter, 2010). Therefore, the delisting target for external tumors was not being met.

A further examination of the confidence intervals for liver and external tumor incidence showed that the uncertainty in the external tumor incidence on standardized bullheads was greater than for liver tumor incidence. The wider confidence interval observed for external tumors could not be explained by differences in sample size, as the same fish were used for both analyses. Therefore this increase in uncertainty may be attributed to the hypothesis that the relationship between external tumor presence and the covariates age and length is not as strong as it is for liver tumors.

5. APPROACH TO DELISTING THE FISH TUMOR BUI

Assessing the fish tumor or other deformities BUI required examination of more than twenty years of data from sampling, analysis, research, and discussions. To organize the information and findings, PADEP developed a decision tree (Figure 17) based on the United States Policy Committee's *Delisting Principles and Guidelines* (USPC, 2001). The guidelines recommend delisting a BUI when one of the following conditions is demonstrated:

- A delisting target has been met through remedial actions which confirm that the beneficial use has been restored.
- It can be demonstrated that the impairment is not limited to the local geographic extent, but rather is typical of lakewide, region-wide, or area-wide conditions
- The impairment is caused by sources outside the AOC.
- It can be demonstrated that the beneficial use impairment is due to natural rather than human causes.

PADEP evaluated Presque Isle Bay's fish tumors through the filter of each of these conditions to determine whether or not to recommend delisting the BUI.

5.1 Has the Delisting Target Been Met?

The delisting target for the Fish Tumors or Other Deformities BUI in Presque Isle Bay states that:

The fish tumor or other deformities beneficial use is no longer considered impaired when the incidence rate of liver and external tumors is statistically equivalent or lower than the incidence rates at Long Point Inner Bay as confirmed by histopathology.

As noted previously, PADEP monitored the incidence of tumors and other deformities in the bay's brown bullhead population annually throughout the Recovery Stage for comparison against the delisting target. Based on the statistical analysis, liver tumor incidence in Presque Isle Bay is statistically equivalent to Long Point Inner Bay while external tumor incidence is elevated compared to Long Point Inner Bay. The delisting target is only partially met.

5.2 Is the Impairment Widespread?

The guidelines recognize that certain use impairments may, in fact, be regional or lake-wide in nature rather than confined within the boundaries of the AOC. PADEP investigated the geographic extent of the bullhead tumor problem by sampling bullhead from both inland Pennsylvania lakes and sites throughout Lake Erie. PADEP sampled brown bullhead from three inland Pennsylvania lakes, Canadohta Lake, Sugar Lake, and Eaton Reservoir, in 2002, 2003, and 2004 (Table 5). Both liver and external tumor rates varied over time and between lakes, however, all were consistently below Presque Isle Bay for those same years. Integrating data from the three inland lakes, the median liver tumor rate was 1.5% and the median external tumor rate was 2.3% compared to the bay which had a median liver tumor rate of 1.9% and an median external tumor rate of 18.1% during this same time frame.

As discussed in Section 4, PADEP also sampled sites across Lake Erie sites as part of the effort to identify a reference site or background rate. Liver tumor rates ranged from a low of zero percent in Dunkirk Harbor to a high of 28.7% in Sandusky Bay. External tumor rates ranged from 6.4% in Long Point Inner Bay to 22.5% in Dunkirk Harbor (Table 2). Although some bullhead populations in Lake Erie experience elevated incidences of liver and external tumors at levels equal to or exceeding levels found in Presque Isle Bay, the incidence rates in other populations are quite low. It is also noteworthy that Dunkirk Harbor had both the lowest incidence of liver tumors and the highest incidence of external tumors, underscoring the poor correlation between these BUI listing criteria at certain sites.

The Department's findings are consistent with those reported by others. Poulet et al. (1994) documented the presence of external tumors on bullhead collected from 17 water bodies (both contaminated and uncontaminated) throughout New York State. Spitzbergen and Wolfe (1995) similarly investigated nine protected reservoirs and ponds in New York State where there was no reported evidence of elevated levels of anthropogenic contamination confirmed by sediment sampling but over 30% of mature brown bullheads had liver tumors and up to 100% exhibited external tumors.

While tumors on brown bullhead occur in various locations throughout Lake Erie and inland Pennsylvania reference lakes, it is not clear whether the tumor rates in Presque Isle Bay are a reflection of some lakewide or basinwide background rate due anthropogenic activities or indicative of a locally degraded environment. Interestingly, none of the non-AOC locations, which were chosen because there were no known discharges of contaminants, had a zero tumor rate for both external and liver tumors. Additionally, the three inland lakes sampled do not have discharges or contaminated sediment and one is a drinking water reservoir. Whatever is happening in Presque Isle Bay appears to be occurring elsewhere in both AOC and non-AOC locations to a greater and lesser extent.

5.3 Is the Source of the Impairment Outside the AOC?

Presque Isle Bay's sediment contains organic contaminants and heavy metals in concentrations similar to other urban harbors. Given the moderate levels of contaminants present, it is reasonable to consider that bullhead may seasonally migrate into and out of the bay and are exposed to sources of contamination located elsewhere in Lake Erie. In 1994 PADEP conducted a large-scale mark-recapture study of Presque Isle Bay bullhead that suggested limited movement with only two fish migrating into the bay and one out (Obert, 1994). Building on that work, the United States Geologic Survey (USGS) conducted an updated radio-telemetry study of Presque Isle Bay bullhead migration (Millard et al., 2009). Forty-nine brown bullhead were collected from various sites within the bay, radio tagged, and released unharmed at the point of capture. Both fixed-station receivers and manual tracking were used to relocate tagged fish. As was the case with in the previous study, USGS found little evidence of migration out of Presque Isle Bay. The telemetry study also supported the conclusion that some bullheads do move among sites within Presque Isle Bay, although most tagged specimens return to the same sites repeatedly. The lack of migration suggests that, the factors influencing or causing the development of the liver and external tumors are present in Presque Isle Bay.

5.4 Is the Impairment Due to Natural Causes?

The occurrence of tumors in brown bullhead catfish is most frequently attributed to exposure to environmental carcinogens—in particular PAHs. Nonetheless, bullheads with tumors are found in both contaminated and uncontaminated waterbodies throughout the northeastern United States

(Pinkney and Harshbarger, 2005, Poulet et al., 1994, Spitzbergen and Wolfe, 1995). Perhaps it is not surprising, then, that tumored bullheads were found during PADEP's sampling in presumed uncontaminated sites like Old Woman's Creek in Ohio and Pennsylvania's inland Eaton Reservoir. The occurrence of tumored bullhead in unpolluted waters calls into question the cause-and-effect relationship between contaminants and tumors. In fact, tumors in many species of fish are known to have natural causes (c.f., Baumann 1992a). Certain hybrid fishes (e.g., carp-goldfish hybrids and swordtail-platyfish hybrids) are known to have elevated incidences of "spontaneous" tumors. In addition, certain tumors in several fish species (northern pike, muskellunge, walleye, and drum. are known to be caused by viruses. PADEP and its research partners investigated the potential role of genetic predisposition and viruses as causes of the bullhead tumors in the bay.

5.4.1 Genetic predisposition to tumors through hybridization

Studies of hybrid fishes have shown that hybrids and succeeding backcross generations are highly sensitive to pollutants (Setlow et al, 1989). Given that brown bullhead are known to hybridize with black bullhead (*Ameiurus melas*), and that certain hybrids are known to have elevated incidences of spontaneous tumors, the extent of potential hybridization among these species in Presque Isle Bay was investigated by Cingolani et al. (2007). Samples were collected from Dunkirk Harbor, NY, Old Woman Creek, OH, Long Point Bay, ON, Tamarack Lake, PA, and Presque Isle Bay. Reference brown bullhead samples were obtained from a reservoir in Huntington County, PA and black bullhead reference specimens were obtained from Clear Lake, IA. More than 20 specimens from each location were included in the study.

Researchers compared aspects of the outward appearance (shape, structure, color, and pattern) as well as the form and structure of the internal parts like bones and organs of the two species. Any external deformities or tumors were noted. Additionally, the genetic make-up was compared using nuclear DNA microsatellites to identify differences among these two species. Looking at the outward appearance and internal structure, the study concluded that the majority of Presque Isle Bay brown bullhead matches the reference brown bullhead population and not the reference black bullhead. Evidence of black bullhead genes in the brown bullhead samples and vice versa was found in the Presque Isle Bay bullheads as well as in fish from other locations in Lake Erie. However, the bay's fish are not different from brown bullhead collected in other Lake Erie locations. Based on this study, hybridization is not valid as a causal explanation for tumors in Presque Isle Bay brown bullhead. (See Appendix A for the full report).

5.4.2 The Role of Viruses

The USGS's Leetown Science Center used molecular techniques to investigate the role of viruses as a causal agent for external tumors in brown bullhead. The analysis included samples from both Presque Isle Bay and the South River in the Chesapeake Bay watershed. No viral DNA or RNA was directly detected. While definitive viral sequences were not identified, a number of gene transcripts associated with cellular responses to viral infection were observed. The investigators found insufficient evidence of viral involvement in the tumors (Iwanowicz et al., 2012). However, it should be noted that the RNA quality of both the Presque Isle Bay and walleye tumor samples used as positive controls was very low and sample sizes were small. Furthermore, the report suggested sampling brown bullhead tumors during multiple seasons would increase the likelihood of detecting a viral pathogen, since retroviruses (one virus type that commonly causes external tumors in other fish species) cannot be detected unless they are in their replication phase, which may occur during a "narrow window" of the annual cycle (See Appendix B for the full report). The results of the study are inconclusive due primarily to the poor quality of the small sample of fish evaluated.

5.4.3 Exposure to Contaminants

Numerous field studies have suggested a correlation between exposure to chemicals, most frequently PAHs, in the sediment of lakes and rivers and an increased prevalence of liver tumors in brown bullhead (Baumann et al., 1987, 1991; Baumann and Harshbarger, 1995, 1998; Brown et al., 1973; Harshbarger et al., 1984; Leadley et al., 1998; Pinkney et al., 2001, 2004a; Pyron et al., 2001; Smith et al., 1994). There is less field evidence linking chemicals in the environment to external tumors in brown bullhead (Bowser et al., 1991; Poulet et al., 1994; Spitsbergen and Wolfe, 1995). While there is experimental evidence linking PAH exposure to tumors in other fish (Bunton 1996), relatively little experimental work has been done with the brown bullhead and the strength of correlation between PAHs and tumors has varied among studies. In general, the evidence linking PAH-contaminated sediment with liver tumors is much stronger than the evidence associating PAHs with external tumors (Rafferty et al 2009).

Recognizing the limitations of the research to date, PADEP undertook a whole sediment exposure study in order to better understand the causal relationship between exposure to Presque Isle Bay sediment and the development of tumors in brown bullhead. In a laboratory setting, fifty-six juvenile brown bullhead were exposed to sediment collected from either Presque Isle Bay (19.41 mg/Kg total PAHs) or Canadohta Lake, the sediment control condition (1.49 mg/Kg total PAHs). Ten additional bullhead were held in aquaria containing laboratory water only (water control condition). The experiment continued for 556 days. Periodically, the fish were grossly observed for the development of visible tumors and other lesions and liver samples were obtained and analyzed for biomarkers of early-stage carcinogenesis (DNA adduct) and histopathological evaluations for tumors and pre-cancerous cells.

None of the bullhead developed grossly observable raised tumors and fish in all conditions appeared to thrive. A single liver tumor developed in a fish exposed to the sediment control condition. There were no other differences in the histopathological evaluation among the fish exposed to Presque Isle Bay sediment, Canadohta Lake sediment control, or the water control conditions. DNA adduct results were similarly negative. No PAH-DNA adducts formed in any experimental condition, indicating that the PAH carcinogens present in Presque Isle Bay sediment are not bioavailable to bullhead, or the experimental regimen was not able to adequately represent the exposure scenario that may operate in Presque Isle Bay. The bay's sediment did contain higher concentrations than Canadohta Lake for seven of eight detected PAHs, yet exposure to Presque Isle Bay sediment had no detected adverse effects on brown bullhead.

From a BUI delisting standpoint, the critical dependent variable in this study is the development of tumors. None of the bullhead developed raised external lesions. The only specimen diagnosed with liver tumors was from the low-PAH control sediment condition. The most sensitive biomarkers of early-stage cancer also failed to indicate that carcinogenesis had been initiated in exposed fish. Livers of specimens in all conditions had a heavy parasite burden, but this burden did not vary among conditions. Even with some of the experimental limitations, this work strongly suggests that simple exposure to Presque Isle Bay sediment is not responsible for the tumors and other deformities seen in the brown bullhead population. (Experiment Results in Appendix C).

5.5 Evaluating the Fish Tumor BUI

Using the United States Policy Committee's guidelines, PADEP consolidated all known information about the fish tumor or other deformities BUI in Presque Isle Bay (Table 6). PADEP used data collected in the post-Recovery Stage to identify and test a delisting target, which incorporated both liver and external tumors. Comparison to a selected "least impacted control site" showed that the liver tumor incidence in Presque Isle Bay met the delisting target. This is not the case for the external tumors. Data collected from Lake Erie sites did indicate that the liver tumor rate in Presque Isle Bay may be a reflection of a background rate for this species in the Great Lakes. The incidence of external tumors across the Lake Erie sites fluctuated more, with Presque Isle Bay incidence rates in the middle of the spectrum for those sites evaluated. The bullheads do not appear to routinely or consistently migrate outside the bay, which suggests that there is something in the bay's ecosystem causing the tumors. Because bullhead tumors are found in varying incidence rates across Lake Erie, it is clear that the conditions causing the tumors in the bay are present elsewhere at inland lakes and both AOC and non-AOC locations. PADEP evaluated the possibility that the bay's bullhead are hybrids between two species and thus, potentially predisposed to tumor formation, or that the tumors are caused by a naturally occurring virus. Studies did not support the genetic hybrid hypothesis and the viral study while limited by the small number of samples, did not identify any viral sequences. The exposure study did not establish a cause and effect relationship between chemicals in the sediment and tumors, even at the earliest detectable stage. It is possible that the tumors are a result of multiple factors, including naturally occurring viruses and chemical contaminants that interact to produce tumors. These facts call into question the validity of the external tumors as an indicator of environmental degradation.

6.0 Risk Assessment

It was the external tumors found on the bay's brown bullheads in the late 1980s that galvanized the public and resulted in the listing of Presque Isle Bay as an AOC. Despite the absence of scientific data to support a causal relationship between contaminants in the bay's sediment and external tumors, it is known that PAHs, metals, and other legacy COPCs are present. More than 20 years of studies document the concentrations of contaminants in the sediment, water, and fish, yet there is no clear understanding of the risk posed by these contaminants. Prior to recommending any delisting action, PADEP wanted to ensure that the contaminants in the bay's sediment do not pose an unacceptable level of risk to the bay's ecosystem or to the health and welfare of the people who enjoy it.

6.1 Ecological Risk Assessment

PADEP commissioned an ecological risk assessment to determine whether contaminants within the bay pose a significant risk to the benthic macroinvertebrates, fish, birds, and other animals in the food web. The assessment used a mix of existing data, conclusions, and recommendations from sediment, fish tumor, and other environmental studies conducted in the bay over the past twenty years. A conceptual site model identified potential ecological receptors and the sources and exposure paths for contaminants (Figure 18). COPCs are the legacy contaminants, including heavy metals and specific PAHs selected because of their frequency in exceeding toxicity thresholds in surficial sediment.

The assessment was built around the question "Do legacy contaminants continue to pose a risk to ecosystem receptors within Presque Isle Bay"? The evaluation focused on three objectives: (1) to maintain and protect the benthic invertebrate community; (2) to maintain a quality fishery; and (3) to protect and improve the near-shore habitat in support of aquatic-dependent wildlife. These objectives were originally identified by the PAC as part of the 2005 sediment survey. Because the available data on Presque Isle Bay was not collected to support a formal risk assessment of exposure pathways, a weight-of-evidence approach was taken as a screening level ecological risk analysis. The risk characterization integrated the exposure and effects characterizations to assess whether COPCs are sufficiently high to pose unacceptable risks to ecological receptors. The weight-of-evidence concluded:

- Surface sediment COPCs appear to be the primary chemical stressor in this system, although habitat (substrate) and invasive species may be additional stressors on the ecological community that may be challenging to tease apart.
- The potential risk of COPC exposure to benthic invertebrates across the AOC is generally low based on the whole sediment toxicity test. Isolated areas may pose a moderate to high risk of exposure.
- Benthic invertebrate exposure risk decreased through time and is generally meeting toxicity targets.
- The probable effect concentration (PEC) targets are generally met across the AOC for most COPCs. Exceedences do occur for metals like barium and cadmium and for some PAHs. Studies focused on high concentration areas tend to exceed PEC in most cases but skew the AOC-wide results.
- Metal bioavailability across the AOC appears to be decreasing through time, with recent samples meeting low toxicity thresholds.
- The quality fishery objective within the AOC is supported by good water quality, a low risk of prey base (benthic invertebrates) exposure to COPCs, and fish tissue concentrations of monitored compounds that are similar to background levels.
- Water quality conditions are based on qualitative evaluations and fish tissue concentrations for monitored contaminants (e.g., mercury and PCBs) and are similar to or better than Lake Erie.
- Near-shore sediment habitats suggest that ingestion exposure risks to wildlife are moderate to low, and the elevated surface sediment concentrations of PAHs and metals in the AOC tend to be in the vicinity of the docks and shipping channel.

The weight-of-evidence indicates that targets supporting the Presque Isle Bay ecosystem are being met. While gaps in data do exist, this evaluation suggests that the risk to ecosystem receptors with the AOC is improving through time and currently rates low to moderate risk (LimnoTech, 2012).

More specifically, the assessment drew three conclusions: first, that the PAHs and metals within the bay do not appear to pose a significant risk to receptors in the ecosystem; second, that liver tumors may be a better indicator of sediment conditions than external tumors, as liver tumors have been shown to correlate with PAHs and metal concentrations in surface sediment; and third, that the presence of external tumors does not appear to be a health threat to fish or to humans or wildlife that consume them. (See Appendix D for the full report.)

6.2 Human Health Risk Assessment

PADEP also commissioned a Human Health Risk Assessment. This assessment estimated human health risks due to contact with COPCs in bay sediments and from eating fish caught in Presque Isle Bay. Consistent with United States Environmental Protection Agency's (USEPA) protocols, estimating the risk to human health followed a four stage process: hazard identification, exposure assessment, toxicity assessment, and risk characterization (Figure 19). Data collected between 2004 and 2010 for metals, PAHs, PCBs, and pesticides in the bay's surface sediment and fish were first compared to the USEPA screening levels to determine which contaminants should be included in the risk estimate process. Screening identified arsenic, lead, total PCBs, and six PAH compounds as COPCs for the sediment and mercury, selenium, mirex, pesticides, and PCBs as the COPC in fish. Estimation of the human health risks was conservative in terms of the exposure scenarios and estimates of exposure. Both cancer and non-cancer (e.g., toxicity) risks were calculated for adults and children exposed to sediment through dermal contact or ingestion and eating fish. The evaluation included fourteen separate fish species collected in Presque Isle Bay and Lake Erie.

The main exposure route for contaminants in Presque Isle Bay is through fish consumption. These risks were several orders of magnitude greater than those associated with direct contact with contaminated sediments. On the other hand, contaminant levels in bay fishes were generally found to be comparable to or lower than those found in Lake Erie fishes. The cancer and non-cancer risk estimates generated from consumption of fish tissue were highly dependent on the fish species. Based on the dataset, lake trout and smallmouth bass represented the species with the highest cancer and noncancer risk estimates. However, these species occur either exclusively (lake trout) or primarily (smallmouth bass) in Lake Erie proper rather than Presque Isle Bay. The contaminants with the highest contribution to the non-cancer and cancer risk estimates for lake trout and smallmouth bass were PCBs. The assessment of cancer and noncancer risks included the assumption of a single species diet and that all fish consumed originated from the AOC. These assumptions are conservative and likely to overestimate the risks from consumption of fish (Homan, 2012).

The cancer and non-cancer risk estimates for direct contact with contaminated sediments were generally below the target risk levels set by USEPA for all exposure groups evaluated. All chemical-specific and cumulative excess lifetime cancer risk estimates were below 1×10^{-5} and all chemical-specific and cumulative hazard indices were below 1.0. Again, these estimates are conservative and likely to overestimate the risk (Homan, 2012). (See Appendix E for the full report).

7. RECOMMENDATION TO DELIST

7.1 Delisting Guidelines

The goal of the AOC program as defined under the Great Lakes Water Quality Agreement is to insure that AOCs, which have been defined as areas where human activities have caused or are likely to cause significant impairment of local beneficial uses of water resources, are improved to the point where their environmental conditions are equal to other non-AOC locations across the Great Lakes. Those conditions may not be pristine but are consistent with the ambient environmental conditions elsewhere in the Great Lakes.

The International joint Commission (IJC) issued listing/delisting criteria for Great Lakes Areas of Concern in 1991 (IJC, 1991). The criteria serve as guidelines for the fourteen beneficial use impairments (BUIs) contained in Annex 2 of the Great Lakes Water Quality Agreement. The IJC listing/delisting guidelines were developed to assist in making recommendations for listing new AOCs and in reviewing RAPs. The intention was to establish a "set of yardsticks" that could be applied throughout the basin and keep the RAP program focused. They are written as guidelines to serve as a common starting point for each AOC with the clear expectation that specific delisting goals and targets are derived locally to address BUIs. In 2001, the United State Policy Committee (USPC) provided a set of "Delisting Principles and Guidelines" to update the IJC's general criteria for American AOCs. The USPC's guidelines state explicitly that delisting targets are locally derived, premised on local goals and related environmental objectives for the watershed, and consistent with federal and state regulations and policies, when available.

Both the IJC's guidance and the USPC's principles state that RAPs are intended to address use impairments of local, geographical extent and cause, rather than lakewide or basinwide phenomena. The USPC principles provide more explicit direction regarding delisting either an AOC or individual BUI. According to those principles, RAPs can only address impairments caused by local sources and impacts from outside the AOC should not preclude delisting. Under these circumstances, an external impairment and its sources should be addressed by another environmental program such as the Lakewide Management Plan (LaMP). Additionally, both the IJC and USPC guidances note that it may not be possible to fully restore some beneficial uses even though all remedial actions are implemented. For example, there may be natural factors or social or economic factors that prevent full restoration of the BUI. Under these circumstances, delisting can and should proceed. PADEP used both the IJC and USPC guidelines in evaluating potential BUIs in the Stage 1 RAP, setting delisting targets in the Stage 2 and 3 RAPs, and determining whether to delist in the Stage 3 RAP.

7.2 Status of the BUIs

The Presque Isle Bay AOC had two BUIs: restrictions on dredging activities and fish tumors or other deformities. Both BUIs were believed to be a result of the PAHs found in sediment throughout the bay. For the restrictions on dredging activities, the IJC criterion for delisting is "when contaminants in sediments do not exceed standards, criteria, or guidelines such that there are restrictions on dredging or disposal activities". Pennsylvania's laws regulate dredged material as a solid waste and place restrictions on disposal options. To address the BUI within the context of Pennsylvania's laws, a delisting target was developed based on the process Pennsylvania uses to determine whether material can be disposed of in Erie's Confined Disposal Facility. Because dredging only occurs in limited areas within the bay, ecosystem health targets were added to ensure environmental improvements could be monitored throughout the AOC. PADEP delisted the restrictions on dredging BUI in 2007 after a comprehensive sediment survey in 2005 showed that the delisting and ecosystem health targets were being met. A 2009 sediment survey also confirmed that the delisting target is being met and samples from the majority sites from the AOC and its tributaries met the ecosystem health targets.

For the fish tumors or other deformities BUI, the IJC delisting guideline is "when the incidence rates for fish tumors or other deformities do not exceed rates at unimpacted control sites or when survey data confirm the absence of neoplastic or pre-neoplastic liver tumors in bullheads or suckers". PADEP chose a delisting target based on comparison of external and liver tumors from Presque Isle Bay to a selected "least-impacted" reference site.

Based upon the recommendations of researchers and other experts during a series of workshops between 2003 and 2006, PADEP sampled a number of inland lakes and non-AOC locations in Lake Erie to identify a "least-impacted" reference site. All of the candidate reference sites sampled were known to have brown bullhead populations but no known direct discharges of contaminants. In order to compare the sites over a period of years, a statistical methodology was developed that normalized the tumor rates to those of fish at age 7, the approximate mean age of the bullheads in the full data set. Interestingly, the surveys showed that neither the non-AOC locations in Lake Erie nor the inland Pennsylvania lakes were free of bullhead tumors. Additionally, locations where liver tumor rates were high often had low external rates and vice versa. For example, between 2004-2007 median external tumor rates in Dunkirk Harbor were 22.5% while median liver tumor rates were 0% and Sandusky Bay had a 9.3% median external tumor rate and a 28.7% median liver rate. The exception was Long Point Inner Harbor which had both low external (6.4%) and liver (1.2%) tumor rates. As a result, Long Point Inner Bay was identified as the least-impacted reference site for comparison against Presque Isle Bay. The delisting target selected for Presque Isle Bay is met when "the incidence rate of liver and external tumors is statistically equivalent or lower than the incidence rates at Long Point Inner Bay as confirmed by histopathology".

Comparison of Presque Isle Bay to Long Point Inner Bay showed that the liver tumor rates were not statistically different. In fact, when statistically adjusted for age, it appears that the incidence of liver tumors in Presque Isle Bay bullhead may be a reflection of the broader Lake Erie background rate. The same was not true for the external tumors where Presque Isle Bay was statistically significantly higher than Long Point. Still the external tumor rate in the Presque Isle Bay bullhead was comparable to all but one of the potential Lake Erie "least-impacted" reference sites evaluated. Based on the limited sample sizes from the potential reference sites, it is difficult to determine whether or not the age-adjusted external tumor rate in Presque Isle Bay is significantly higher, lower, or the same as the background rate elsewhere in Lake Erie. It is true that similar to Presque Isle Bay, incidences of unexplained external tumors are occurring in populations of brown bullheads in both AOC and non-AOC locations as well as inland Pennsylvania lakes.

7.3 Causes of Fish Tumors

PADEP turned its focus to investigating the cause of the external tumors and evaluating the appropriateness of using the tumors as an indicator of environmental degradation. PADEP's approach included investigating pathogens as potential natural causes of the tumors; evaluating the role that the genetics of the bay's bullheads may play; conducting its own experimental investigation into the relationship between exposure to bay sediment and the development of tumors; and conducting an extensive review of the scientific literature. A study designed to detect viruses in external bullhead tumors was inconclusive. A study evaluating whether the bay's bullheads were hybrids and, therefore, potentially predisposed to tumors found little evidence of atypical hybridization. An eighteen month laboratory exposure study did not find biomarkers signifying early stage cancer on any fish exposed to Presque Isle Bay sediment.

As was the case in its own investigations, a review of the scientific literature revealed inconsistent relationships between exposure to environmental contaminants and the development of tumors in bullhead. The preponderance of the published literature focuses on the role of PAHs as the cause of tumors in brown bullhead. While there is a sound scientific basis for the role of PAHs as fish carcinogens in general, the causal role of PAHs in bullhead tumors remains unclear. Studies such as the work of Baumann in the Black River and other locations in the Great Lakes have shown an apparent relationship between sediment PAHs and bullhead tumors, particularly liver tumors, while others such as the work of Pinkney in the Chesapeake Bay tributaries and Spitsbergen in New York ponds and reservoirs have shown inconsistent associations between PAHs and tumor rates or no correlations at all. PADEP's own work in sampling potential non-AOC reference sites showed inconsistencies between external and liver tumor rates within the same locations. If the hypothesis is that exposure to contaminated sediment causes external and liver tumors, then bullheads collected from locations without known sources of contamination should have few, if any external or liver tumors and locations with contaminated sediment should have elevated levels of both external and liver tumors. That was not the case. Bullhead from the site with the highest external tumor rate, Dunkirk Harbor, had a zero percent liver tumor rate. Conversely, the site with the highest liver tumor incidence rate, Sandusky Bay, had one of the lowest external tumor rates.

The scientific literature supports a stronger causal relationship between PAH exposure and liver tumors than external tumors. It seems apparent, based on the recent work reported by Pinkney (2011) and Baumann (2010), that if PAHs play a causal role in bullhead tumors they are a subset of a more complicated and multifactorial etiology. While the exposure route for liver tumors is primarily thought to be via ingestion of contaminated sediments and aquatic organisms, external tumors have been attributed to factors including various direct and indirect-acting carcinogens in the water column and sediment, solar radiation, abrasions, viruses, or some combination of all of these.

The inconsistencies in incidence rates coupled with the lack of a direct link with exposure to PAHs or the ability to isolate the factors resulting in the formation of external tumors makes the external tumor rate an unreliable measure of environmental degradation.

Despite the expenditure of considerable resources, there are still tumors on bullheads. The rest of the bay's fishery, however, is diverse, abundant, and healthy, appearing unimpacted by whatever is affecting the bullheads. Additionally, the 2005 sediment survey included direct toxicity tests to two benthic organisms and found only limited toxicity. Of 34 samples tested, one was toxic to midges using the survival endpoint and three using the growth endpoint. Based on the results of the sediment toxicity tests, it is apparent that contaminants in the bay's sediment are not particularly bioavailable and are not adversely impacting the benthic community. The fact that the bullheads have tumors does not appear to indicate any negative consequences for other fish species or benthic organisms, in fact yellow bullheads residing in Presque Isle Bay appear quite healthy. Thirty years after the discovery of external tumors on the bay's brown bullhead catfish, the environment has become cleaner, supporting a diverse fishery with both threatened and endangered species thriving, and yet the brown bullheads still have tumors. Based on the data collected in Presque Isle Bay and elsewhere, the tumors on brown bullhead do not appear to be a good indicator of an unhealthy ecosystem.

While there is stronger evidence correlating liver tumors with PAHs in sediment, the question of what is causing the tumors on this one species of fish may never be answered. Other AOCs are also struggling with this issue. In the December 2011 Stage 2 RAP for the Sheboygan River AOC, Wisconsin focused its delisting target for this BUI on neoplastic liver tumors as factors other than contamination such as viral infection and parasites have been shown to elicit external and preneoplastic tumor responses.

Comparison of Presque Isle Bay to Long Point Inner Bay showed that the liver tumor rates were statistically equivalent. In fact, when statistically adjusted for age, it appears that the incidence of liver tumors in Presque Isle Bay bullhead may be a reflection of the broader Lake Erie background rate. The same was not true for the external tumors where Presque Isle Bay was statistically significantly higher than Long Point. Based on limited sample sizes, it is unclear at present whether or not the age-adjusted external tumor incidence rate in Presque Isle Bay is significantly higher than the background rate elsewhere in Lake Erie.

7.4 Rationale for Delisting the Fish Tumor BUI

PADEP's recommendation to delist the fish tumors or other deformities BUI is grounded on the best science and technology available today. The decision is based on numerous investigations, sampling events, and consultation with the leading experts in brown bullhead investigations. While there is year-to-year variation, since the Recovery Stage designation in 2002 the incidence of liver and external tumors the bay's brown bullhead population has remained stable with little statistical difference in rates between sampling years. Incidence rates of both liver and external tumors remain well below the high levels seen in the early 1990s. Liver tumor rates, the end-point for which exposure to environmental contaminants is more clearly linked to sediment PAH contamination, are statistically indistinguishable from the Long Point Inner Bay reference site. The incidence of external tumors, however, remains elevated when compared to the reference site.

Because there are known legacy contaminants in the sediment regardless of their relationship to the bullhead tumors, PADEP commissioned ecological and human health risk assessments. Using appropriately conservative assumptions and existing data, both risk assessments concluded that cancer and noncancer risks posed by legacy contaminants in the Bay's sediment and fish are below targets for human health and ecosystem protection.

It may not be possible ever to fully restore this BUI due to the external tumors. Reviewing both the IJC and USPC guidelines and principles, it seems clear that external tumors and, to some extent liver tumors, are a lakewide phenomenon. Whatever is happening in Presque isle Bay is occurring elsewhere in both AOC and non-AOC locations. The best course of action for the Presque Isle Bay AOC is to delist with continued monitoring of the sediment and fish; work with the Lake Erie LaMP to include other AOCs in determining the cause(s) of the fish tumors and identification of possible remedial measures at a lakewide scale; focus on restoration projects in the bay's watershed to continue reducing sediment loading to the bay; and use existing regulatory and statutory authority to require permits, cleanup, monitoring, and restoration. P ADEP, therefore, recommends delisting of the fish tumor or other deformities BUI for the Presque Isle Bay AOC.

7.5 Rationale for Delisting the Presque Isle Bay AOC

One goal of the AOC program is to address the source or sources of the beneficial use impairment. In the absence of an identified source to remediate, PADEP and its partners have taken action to address contaminant loading to the bay through permitting, infrastructure improvements, and restoration projects.

There are no wastewater treatment discharges to the bay as a result of more than $100 million in upgrades to the City of Erie's wastewater treatment system in the 1980s and 1990s. The upgrades included the reduction of the number of combined sewer overflows from more than seventy to five. Four of the CSOs discharge into the Mill Creek Tube which empties into the bay. All have screens and flow monitors. Additionally, the City of Erie maintains a litter trap at the end of Mill Creek that catches oil and debris from the CSOs and the stream. The City of Erie reports a CSO capture rate in excess of 99.9%. Additionally, there are no known unpermitted industrial waste discharges to the bay.

In 2002, when Presque Isle Bay was designated in the Recovery Stage a decision was made not to dredge the bay. Extensive sediment sampling failed to identify contaminant "hot spots" in the bay where limited dredging could occur to remove contaminated sediment and at 3,655 acres, remedial dredging of the entire bay is cost-prohibitive and unnecessary. Instead, the remedial measure selected in 2002 was natural attenuation, allowing cleaner sediment to form a cap over contaminated sediment. With a sedimentation rate averaging 1 cm per year, this is a slow process but both the 2005 and 2009 sediment surveys confirmed it is happening.

In considering future remedial measures, there is still work to be done to mitigate the impacts of contaminant loading from stormwater runoff. This work is being done through the Integrated Water Resources Management Plan for Lake Erie and restoration projects funded under the Great Lakes Restoration Initiative (like the work done on Cascade Creek), Coastal Zone grants, and Growing Greener.

In recommending the delisting of the AOC, PADEP determined that removal of sediment by dredging the bay is unnecessary, remedial measures with the greatest direct benefit to the bay are done, other watershed measures that positively impact the bay are ongoing, air and water discharges are permitted and monitored, no other species of fish or benthic organism appear to be impacted, and both the human health and ecosystem health assessment concluded that the existing conditions in the bay do not increase either cancer or noncancer risks to people or the environment.

PADEP believes that the RAP process has accomplished its goal to the maximum extent practicable and the ultimate identification of the causes of the external tumors needs to be addressed outside the scope of the AOC program. Based on the decreased and stable tumor rates, review of the available scientific evidence, and in close consultation with local and national experts and its own PAC, PADEP recommends delisting the Presque Isle Bay AOC. This Stage 3 RAP provides the data to 1) show that 14 measures of water quality listed in the Agreement are not impaired in the AOC; 2) support PADEP's assertion that the fish tumor or other deformities beneficial use is no longer impaired; and 3) show that the Agreement's delisting criteria h*ave* been achieved for the Presque Isle Bay AOC. The removal of the final BUI indicates that environmental conditions in Presque Isle Bay are comparable to non-AOC locations in the Great Lakes. PADEP, with the concurrence of the Presque Isle Bay Public Advisory Committee, recommends delisting the AOC.

8. ROLE OF THE PAC

Beginning in 1983 with the formation of the Erie County Environmental Coalition, Erie's citizens have focused their efforts and attention on restoring Presque Isle Bay. It was members of the Coalition along with the Erie Harbor Improvement Council that petitioned for the inclusion of the bay on the list of AOCs. In 1991, the bay became the 43rd AOC and members of the Coalition and Council became the Presque Isle Bay Public Advisory Committee (PAC). Over the next twenty years, the PAC met quarterly providing advice to PADEP on priorities, studies, delisting targets, and other matters impacting the AOC. With the decision to delist the AOC, the role of the PAC becomes even more critical to ensure beneficial uses remain restored.

After twenty years of focusing on contaminants in the sediment and tumors on brown bullhead, the time has come for the PAC to broaden its involvement beyond the bay. The PAC will continue to meet regularly and provide insight and advice to PADEP on the post-delisting monitoring of the bay. It will also focus attention on the monitoring and restoration work needed in the watershed by assisting PADEP in setting priorities and communicating problems and progress to the public.

Future research, studies, and monitoring conducted in the bay will be reported through the Lakewide Management Plan for Lake Erie. It is PADEP's intention to seek the PAC's input and advice on Lake Erie issues such as strategies to reduce nutrient and other contaminant loading to the Lake, addressing invasive species, monitoring lakewide fish consumption advisories, and investigating the presence and impact of emerging contaminants.

9. POST-DELISTING RESPONSIBILITIES AND MONITORING

Annex 2 of the Great Lakes Water Quality Agreement requires formal monitoring of the recovery of impaired beneficial uses in AOCs only to the point at which the BUIs are no longer considered to be impaired. Even though this point has been attained, PADEP and the Presque Isle Bay PAC recognize that it is important to document the sustained recovery of the AOC, to continue to work to improve water quality in the Presque Isle Bay watershed, and to proactively address new environmental threats as these issues are identified.

The objective of post-delisting monitoring is to ensure that bullhead tumor rates remain stable over time and sediment quality objectives related to the delisting and ecosystem health targets continue to be met. Monitoring will continue in the bay's watershed to document sediment and contaminant loading and the health of fish and macroinvertebrate populations. Activities related to the BUI monitoring will be reported through the Lake Erie Lakewide Management Plan (LaMP). The Lake Erie LaMP is issued every three years with yearly updates in the form of fact sheet. A citizens' forum assists in the selection of priority and focus areas as well as outreach and education on the LaMP. The Triennial LaMP report includes the status and milestones for all of the Lake Erie AOCs. PADEP will continue to report through the LaMP on the environmental status of the bay as well as efforts to restore, protect, and monitor the watershed. Should data trends indicate the delisting and ecosystem health targets are not being met, PADEP will use its existing statutory and regulatory authorities (e.g., Clean Air Act, Clean Water Act, Dam Safety and Encroachments Act, and Clean Streams Law) to ensure sources of pollution are addressed.

In addition, PADEP will continue to monitor water quality and fish tissue contaminant trends in Presque Isle Bay and in Pennsylvania's open waters of Lake Erie through its Water Quality Network sampling program. Both Presque Isle Bay and Lake Erie are currently on the 303(d) list of impaired waters. The bay's listing is a result of fish consumption advisories which are not related to either the restrictions on dredging activities or fish tumors or other deformities BUIs. Monitoring and advisories will continue under the PADEP and PFBC's fish consumption advisory program.

PADEP intends to turn the focus to non-AOC issues, emerging contaminants, and supporting further research into the non-contaminant related factors playing a role in fish tumors. The post-delisting monitoring plan spans a ten year period and is considered a "living document" that will be periodically reviewed by the PADEP and PAC. Monitoring activities may be expanded, revised, or deleted over time. Specific activities and timeframes may be modified following consultation with the PAC due to resource constraints, advances in analytical methods, or new scientific research findings from other studies.

9.1 Beneficial Use Impairments

9.1.1 Restrictions on Dredging Activities

Question to answer: Is the primary delisting target for the restrictions on dredging beneficial use being met?

Target: In at least 90% of samples, the concentrations of chemicals of potential concern in the confined disposal facility mixing zone are below Pennsylvania's Water Quality Standards at the 15-minute compliance point for acute criteria and the 12-hour compliance point for chronic criteria.

To evaluate the delisting target, PADEP will use elutriate data from sediment samples collected by parties permitted under PA's Chapter 105 program to perform dredging within the AOC. The frequency of monitoring will depend on when permitted dredging activities occur. Monitoring data and the status of dredging activities will be reviewed annually.

Question to answer: Is ecosystem health showing any change?

A. Benthos

Target: In at least 90% of sediment samples, the concentrations of chemicals of potential concern are below levels that are associated with acute or chronic toxicity in sediment-dwelling organisms.

Whole sediment chemistry and whole sediment toxicity tests will be used to evaluate ecosystem health. Sampling locations will include sites within the AOC, the study area, and areas adjacent to the AOC. Specifically, samples will be collected from up to eight locations within the AOC. The locations include the areas adjacent to the mouths of Scott Run (SR-25), Mill Creek (MC-23/MC-27), and Cascade Creek (CC-26); one location in the center of the Bay (PIB-07), and one in Misery Bay (PIB-46); an additional sample will be collected from the ponds within Presque Isle State Park (i.e., study area); a sample will also be collected from the Outer Harbor and one from Thompson Bay; and a reference sample (TBD). Samples will be analyzed for PCBs, 34 PAHs, metals, AVS/SEM, total organic carbon, and grain size. Toxicity testing using the freshwater amphipod *Hyallela azteca* or the midge *Chironomus dilutus* will also be done. Monitoring will occur every three years beginning in 2008 until 2014 and then the schedule for additional monitoring re-evaluated.

B. Fish and Wildlife Health

Target: In at least 90% of samples, the concentration of six or more chemicals of potential concern do not exceed Effects Range Median.

Whole sediment chemistry will be used to evaluate this ecosystem health target.

Target: The concentration of mercury and PCBs in tissues of fish from Presque Isle Bay should not be significantly higher than levels in fish tissue from Lake Erie.

PADEP's fish consumption advisory sampling program will be used to evaluate this target.

9.1.2 Fish Tumors or Other Deformities

Question to answer: Is the primary delisting target for the fish tumors or other deformities beneficial use being met?

Target: The bay's fish tumor or other deformities beneficial use is no longer considered impaired when the incidence rate of liver and external tumors is statistically equivalent or lower than the incidence rates at Long Point Inner Bay as confirmed by histopathology.

To evaluate the target, PADEP's Post-delisting bullhead monitoring will be consistent with the methods recommended by PADEP (2002) and Rafferty and Grazio (2007). Histopathology of liver and external tumors will be consistent with the methods described by Blazer et al, 2009(a) and (b). Beginning in 2013, monitoring for grossly observable lesions will be conducted in 2013, 2016, 2019, and 2022. Necropsies and histopathological analyses will be conducted in 2013, 2019, and 2022.

Long Point Inner Bay, the reference site for Presque Isle Bay, will be sampled using the methods described above the same years as Presque Isle Bay.

- *Target population-* Presque Isle Bay resident brown bullhead catfish (*Ameiurus nebulosus*) with a minimum total length of 250 mm. A minimum total length of 250 mm is used to increase the likelihood that sexually mature specimens will be collected for analysis.
- *Minimum sample sizes-* The minimum sample size shall consist of 200 specimens (or the total sample if n<200) for gross observation of external lesions and other deformities. The minimum sub-sample size for histopathological tumor analysis shall be 30 randomly sub-sampled individuals (or the total sample if n<30).
- *Necropsy and histopathology-* A minimum of 30 bullhead will be randomly sub-sampled and subjected to general necropsy. Internal organs will be observed for the presence of gross pathology. Abnormal conditions will be photographed and recorded on the field data sheet. Histopathological tumor analysis will be performed on all liver/gall bladder samples and raised external lesions. Specimens will be humanely euthanized prior to necropsy.
- *Specimen Age-* Both otoliths will be removed from each necropsied specimen for aging. In the event that neither otolith can be recovered, pectoral spines will be used for age estimation.

9.2 Presque Isle Bay Watershed Monitoring

Monitoring of both legacy and emerging contaminants in the Presque Isle Bay watershed is essential to ensure that the bay ecosystem continues to be protected. Sampling will be consistent with the QAPP entitled: GLRI State Capacity Grant – Presque Isle Bay Watershed Restoration, Protection, and Monitoring Plan (PADEP, 2011). Sixteen locations identified in the Presque Isle Bay Watershed Restoration, Protection, and Monitoring Plan (http://pib.psu.edu) will be sampled every five years beginning in 2015. Analysis includes the following:

- Sediment samples will be analyzed for legacy contaminants including metals, oil and grease, PAHs, PCBs, pesticides, nitrogen, phosphorus, AVS, and SEM. Particle size distribution will also be determined.

- Water samples will be analyzed for temperature, dissolved oxygen, conductivity, 5-day biological oxygen demand, total organic carbon, total nitrogen, total phosphorus, and dissolved phosphorus.
- Fish habitat and population health as well as macroinvertebrate community distribution will also be evaluated at each sampling location.

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Appendix A

Investigation into the Hybridization of *Ameiurus* Catfish in Presque Isle Bay, Erie, PA

Appendix B

Investigating the Possible Association of Virus with External Papillomas in Brown Bullhead

Appendix C

Whole-sediment exposure of brown bullhead (*Ameiurus nebulsos)* to industrially contaminated sediment

Appendix D

Presque Isle Bay Area of Concern Screening-Level Ecological Risk Assessment

Appendix E

An Evaluation of Human Health Risks from Contaminants in Presque Isle Bay

TABLES

Table 1. Proposed Delisting and Ecosystem Health Targets for the Restrictions on Dredging Beneficial Use Impairment At the Presque Isle Bay Area of Concern

Ecosystem Goal for Presque Isle Bay Sediment: Maintain and/or restore sediment quality conditions such that human health is protected and the human uses of the aquatic ecosystem (e.g., fish and wildlife consumption; navigation and shipping, etc.) are protected and, where necessary, restored.

Ecosystem Objective for Presque Isle Bay Sediment: Maintain and protect the benthic invertebrate, fish, and wildlife communities of Presque Isle Bay.

Beneficial Use Impairment	Management Objective, Indicators, Metrics, and Targets
Restrictions On Dredging	
Sediment Management Objective	Protect human uses of the aquatic ecosystem (e.g., navigation, shipping, and recreation) and minimize the impact of dredged material discharge on water quality.
Sediment Quality Indicator	Whole sediment chemistry Elutriate test data
Metrics	Concentrations of COPCs in the confined disposal facility mixing zone as determined by application of the USACE's CDFate model using elutriate data or other model using whole-sediment chemistry data from Presque Isle Bay sediment samples.
Narrative Delisting Target	The concentrations of COPCs in the CDF mixing zone are below Pennsylvania Water Quality Standards at the 15-minute compliance point for acute criteria and the 12 hour compliance point for chronic criteria.
Numeric Delisting Target	Pennsylvania Chapter 16 and Chapter 93 Water Quality Standards.
Assumptions	No more than 10% of samples will exceed the target.

Ecosystem Health Target	Management Objective, Indicators, Metrics, and Targets
Ecosystem Health for Benthos	
Sediment Management Objective:	Maintain and/or restore sediment quality conditions such that benthic communities, including epibenthic and infaunal species, are protected and, where necessary, restored.
Sediment Quality Indicator:	Whole-sediment chemistry Whole-sediment toxicity
Metrics:	Concentrations of COPcs in whole-sediment samples Whole sediment toxicity tests 1. 28-d *Hyallela azteca* survival and growth 2. 10-d *Chironomus dilutus* survival and growth
Narrative Ecosystem Health Target:	The concentrations of COPCs (metals, PAHs, and PCBs) are below the levels that are associated with acute or chronic toxicity in sediment-dwelling organisms; The survival and growth of freshwater amphipods, *H.azteca* and midges, *C. dilutus*, exposed to sediment samples from Presque Isle Bay should be greater than or equal to the normal range of survival rates observed for appropriately selected control or reference sediment samples.
Numeric Ecosystem Health Target:	At least 90% of the sediment samples from Presque Isle Bay have the conditions necessary to support healthy benthic invertebrate communities, as indicated by: mean PEC-Q I<1.0; SEM-AVS<0.0; SEM-AVS/f_{oc} <3,000; ESB-TUs <1.0; toxicity to the freshwater amphipod *Hyalella azteca* or the midge *Chironomus dilutus* for the survival or growth endpoints: - Control-adjusted survival of amphipods $\geq$ 75% - Control-adjusted growth of amphipods $\geq$90% - Control-adjusted survival of midges $\geq$ 75% - Control-adjusted growth of midges $\geq$70%

Ecosystem Health for Fish and Wildlife

Sediment Management Objective	Maintain and/or restore sediment quality conditions such that fish and wildlife communities, including aquatic dependent amphibians, reptiles, birds, and mammals, are protected and, where necessary, restored.
Sediment Quality Indicator	Whole-sediment chemistry Fish health Fish tissue chemistry
Metrics	Concentrations of COPCs in whole-sediment samples Concentrations of bioaccumulative COPCs in fish tissue
Numeric Ecosystem Health Target	The concentrations of five or more COPCs in a sample do not exceed Effects Range Median as calculated by Long, et al. (1996); or The concentrations of bioaccumulative COPCs in tissues of fish from Presque Isle Bay should not be significantly higher than the levels in fish tissue from Lake Erie; if COPC concentrations are elevated in PIB fish, then the levels should be lower than the toxicity thresholds for fish and aquatic-dependent wildlife..
Assumptions	No more than 10% of samples will exceed the target.

Site	Liver Tumor Incidence		External Tumor Incidence	
	95% Confidence Interval	Median	95% Confidence Interval	Median
Long Point Inner Bay	(0.0%, 14.9%)	1.2%	(0.0%, 32.0%)	6.4%
Dunkirk Harbor	(0.0%, 56.0%)	0.0%	(0.0%, 73.3%)	22.5%
Old Woman Creek	(0.0%, 61.6%)	3.0%	(0.0%, 83.0%)	20.9%
Sandusky Bay	(11.4%, 47.8%)	28.7%	(0.9%, 22.0%)	9.3%

Table 2. Liver and External Tumor Rates for Potential Lake Erie Reference Sites (2004-2007).

Monitoring Year	**Liver Tumor Rate**	**95% Bayesian Credibility Interval**	**External Tumor Rate**	**95% Bayesian Credibility Interval**
2002	1.1%	(0.0%, 22.2%)	18.6%	(0.0%, 87.6%)
2003	2.4%	(0.0%, 8.6%)	16.7%	(4.4%, 33.2%)
2004	2.1%	(0.0%, 21.0%)	18.9%	(0.0%, 66.6%)
2005	2.3%	(0.0%, 32.4%)	11.9%	(0.0%, 49.3%)
2007	3.9%	(0.0%, 27.0%)	17.3%	(0.4%, 54.3%)

Table 3. Liver and External Tumor Rates for Presque Isle Bay during the Recovery Stage Monitoring. Tumors are histologically verified for an age 7 bullhead.

Site	Liver Tumor Incidence		External Tumor Incidence	
	Median	95% Bayesian Credibility Interval	Median	95% Bayesian Credibility Interval
Presque Isle Bay	2.8%	(0.0%, 18.3%)	15.4%	(0.8%, 45.8%)
Long Point Inner Bay	1.2%	(0.0%, 14.9%)	6.4%	(0.0%, 32.0%)

Table 4. Median Tumor Rates for Presque Isle Bay and Selected Lake Erie Reference Site, Long Point Inner Bay. Tumors are histologically verified for an age 7 bullhead.

Site	Liver Tumor Incidence	External Tumor Incidence
Canadohta Lake	0.8%	3.2%
Eaton Reservior	1.5%	0.5%
Sugar Lake	1.5%	3.3%

Table 5. Inland Lake Tumor Estimates for the Period 2002-2004. Incidence rates were determined using logistic regression based on a 7 year old bullhead.

Table 6. Evaluation of the Fish Tumor BUI in terms of the USPC (2001) Guidelines.

US Policy Committee Delisting Guideline (2001)	How is the Guideline evaluated wrt bullhead tumors? (Methods)	Is Delisting Guideline Met?	Current Status
1. A delisting target has been met through remedial actions which confirms that the beneficial use has been restored.	Delisting targets developed based on tumor rates at least impacted reference site (Long point Inner Bay). Incidence rates at PIB AOC compared to reference site.	*Liver Tumors*-Yes *External Tumors*-No	*Liver Tumors*- Not significantly different than reference sites (Approx. 2.80% v. 1.20%) *External Tumors*- Significantly higher than reference sites (Approx. 15.4% v. 6.4%)
2. It can be demonstrated that the beneficial use impairment is due to natural rather than human causes.	Investigated the role of pathogens (viruses) in the induction and promotion of tumors.	*No*	Insufficient evidence of viral involvement.
3. It can be demonstrated that the impairment is not limited to the local geographic extent, but rather is typical of lakewide, region-wide, or area-wide conditions (under this situation, the beneficial use may not have been originally needed to be recognized as impaired).	Compared the incidence of liver and skin tumors in PIB AOC to lakewide/background rates.	*Liver Tumors*-Yes *External Tumors*-unclear	*Liver Tumors*- Can be conceptualized as a "regional problem" or simply as the background rate for this species in the Great Lakes Region *Skin Tumors*- Incidence rate elevated in PIB compared to Lake Erie reference site but comparable to all but one of the potential reference sites evaluated. Based on the limited sample sizes from the potential reference sites, it is difficult to determine whether or not PIB external tumor rates are higher, lower, or the same as background rates elsewhere in Lake Erie.
4. The impairment is caused by sources outside the AOC. The impairment is not restored but the impairment classification can be removed or changed to "impaired-not due to local sources". Responsibility for addressing "out of AOC" sources is given to another party (i.e., LaMPs).	Investigate potential for bullhead to migrate out of PIB and be exposed to remote contaminants. Determine sources and loads for contaminants of concern.	*No*	Migration studies show that bullhead are resident to PIB. While the cause(s) of bullhead tumors in PIB have never been definitively identified, similar to PIB incidences of unexplained external tumors are occurring in populations of brown bullheads in both AOC and non-AOC locations.

Figures

Figure 1. Map of the Presque Isle Bay Area of Concern (AOC) Boundary

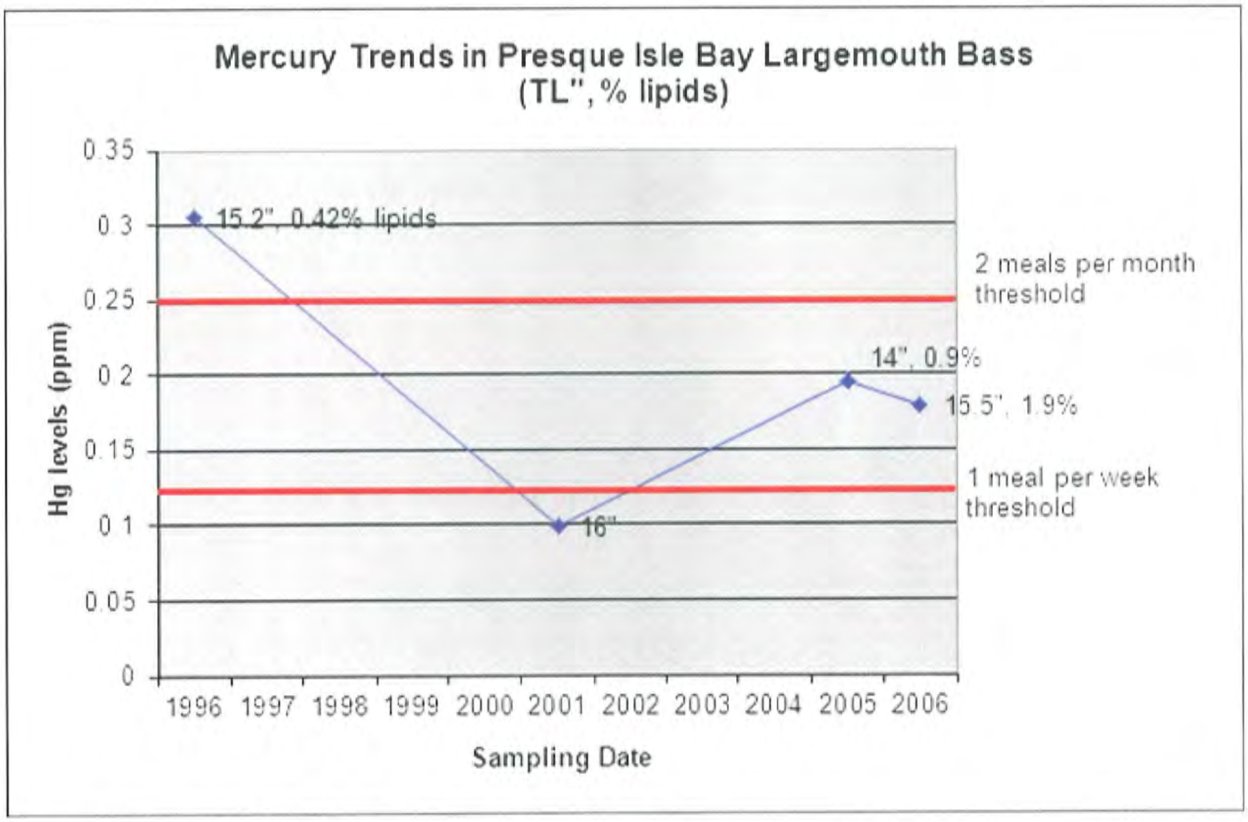


Figure 2. Mercury Trends in Presque Isle Bay Largemouth Bass.

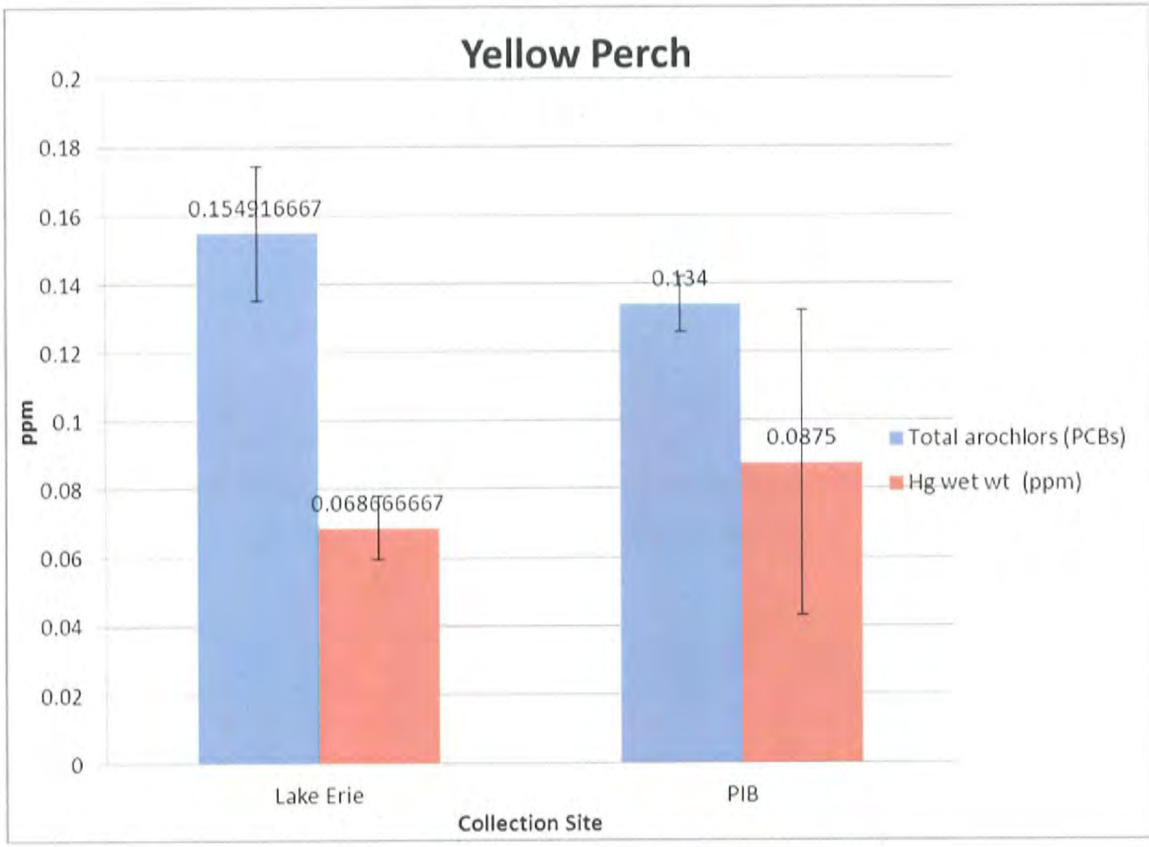


Figure 3. Mercury and PCB concentrations in Presque Isle Bay Yellow Perch.

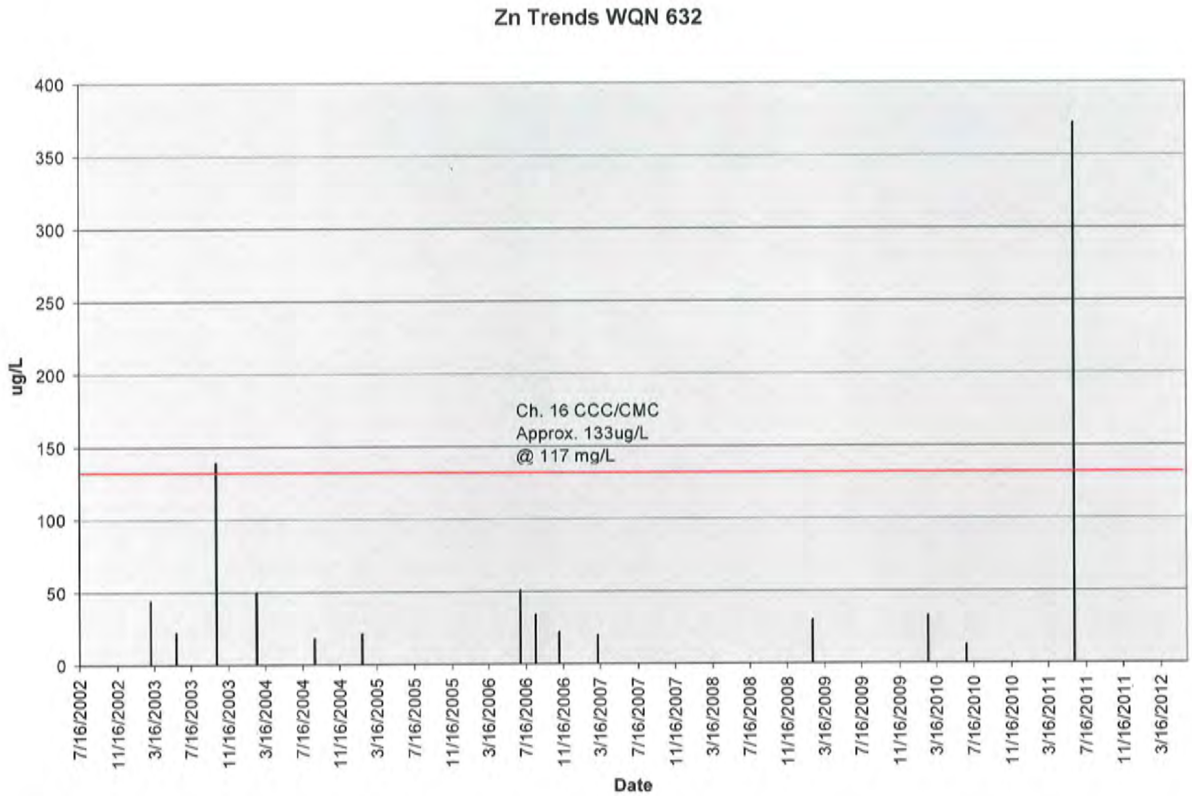


Figure 4. Detectable Levels of Zinc in Water Qualtiy Network Samples Collected between 2002 and 2011.

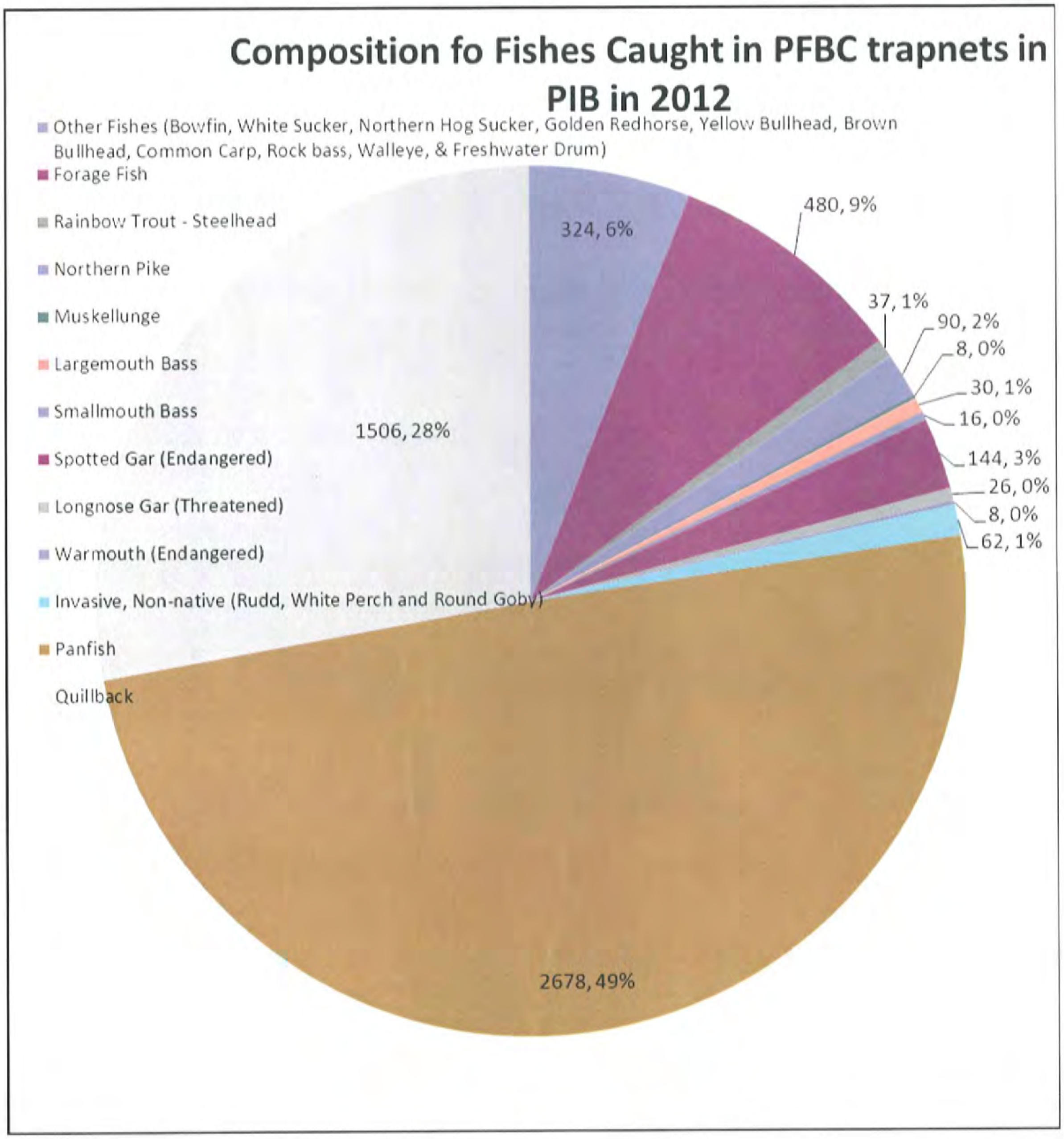


Figure 5. Diversity and Abundance of Fish Populations in Presque Isle Bay.

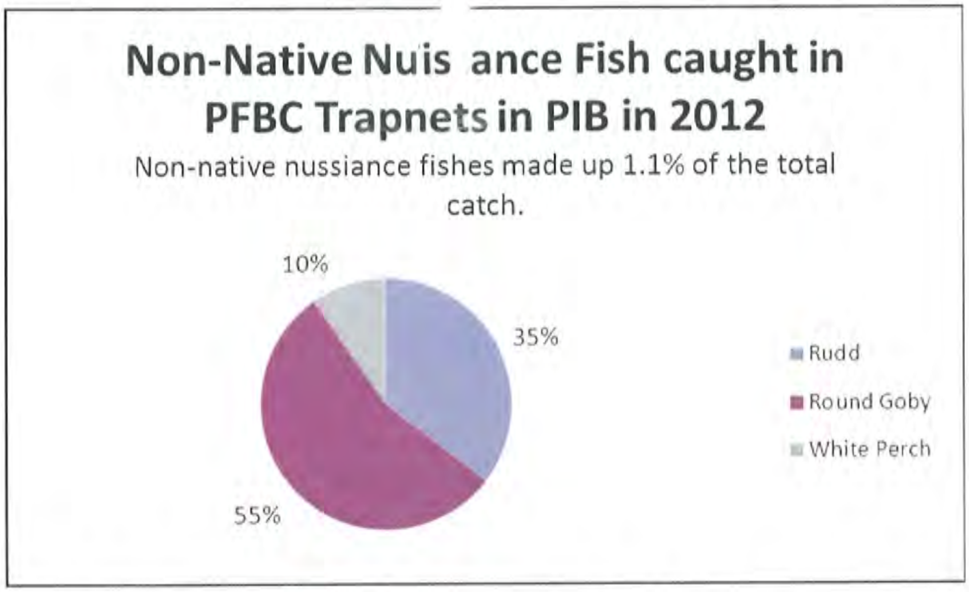


Figure 6. Non-Native Fish Found in Presque Isle Bay 2012 Survey.

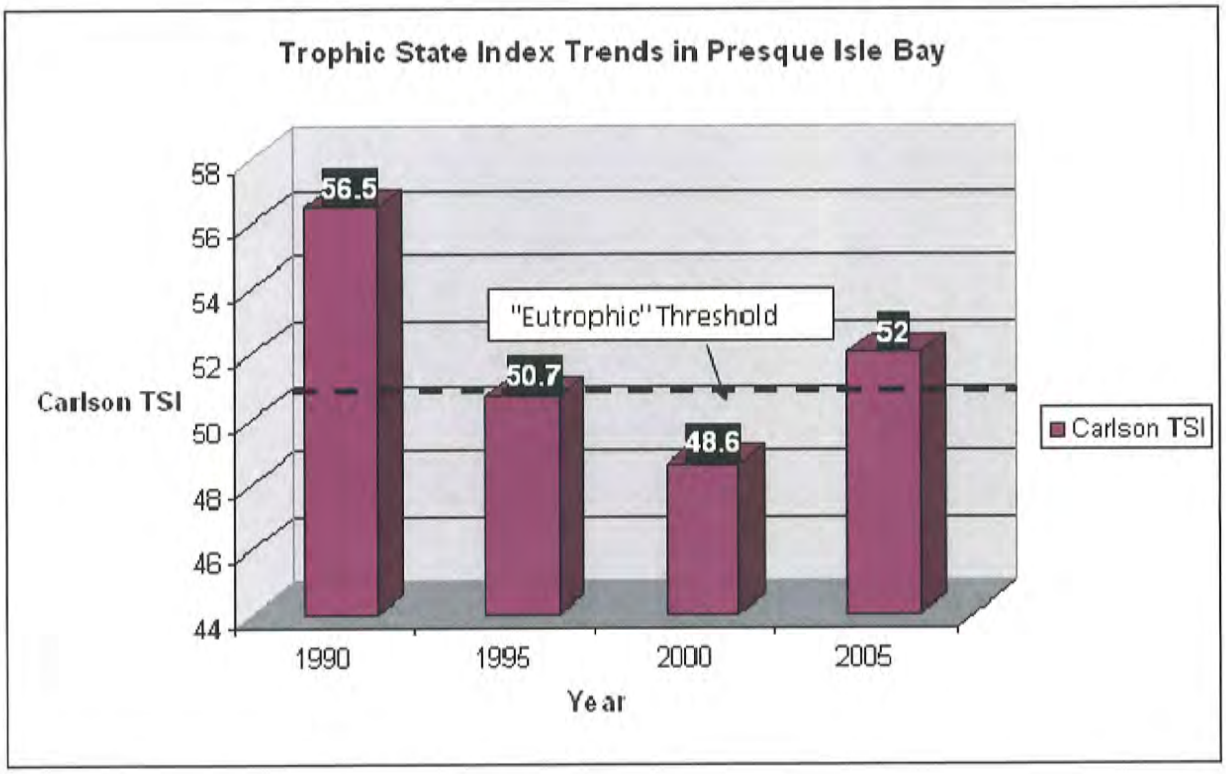


Figure 7. Trophic State Scores for Presque Isle Bay.

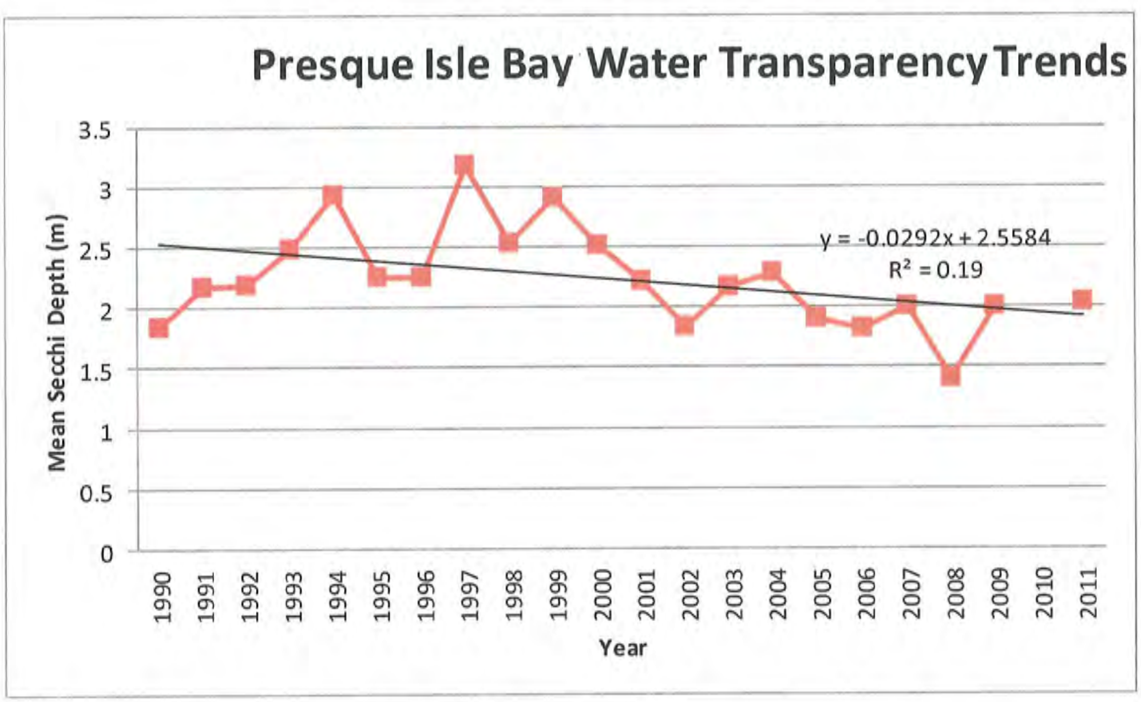


Figure 8. Water Clarity over Time in Presque Isle Bay.

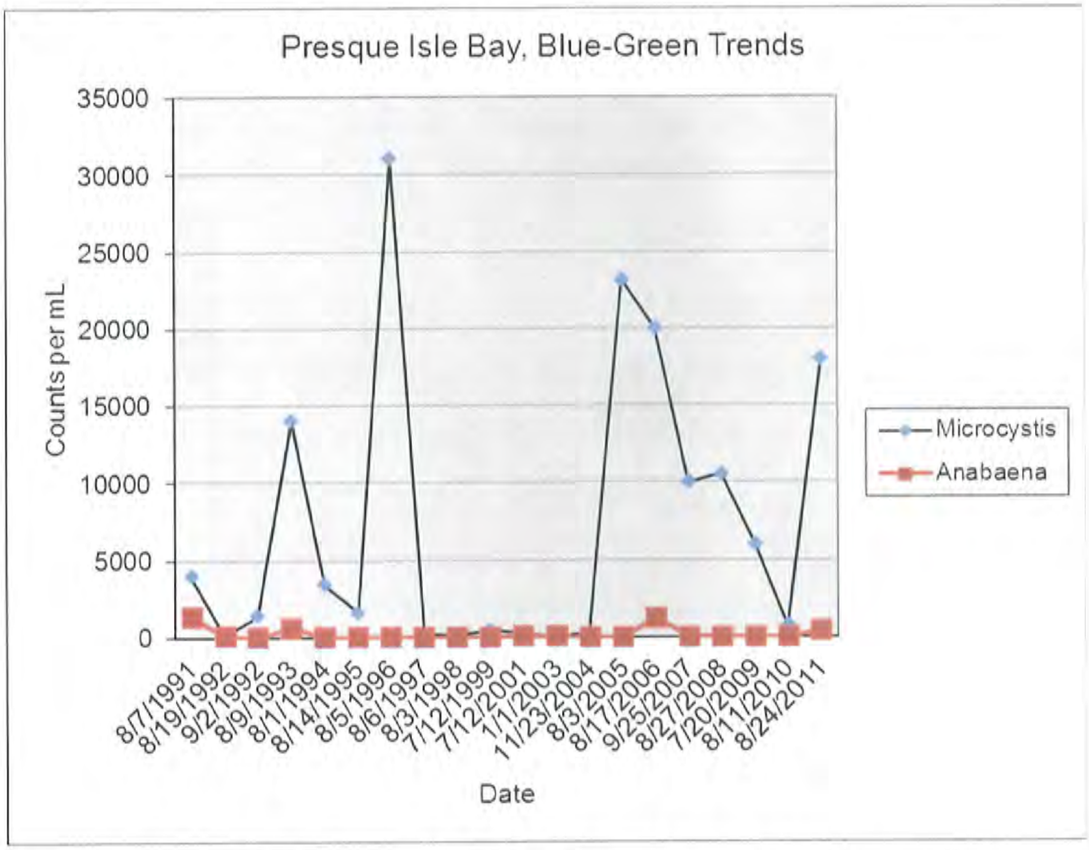


Figure 9. Counts of Blue Green Algae (*Anabaena* and *Microcystis)* in Presque Isle Bay from 1991 – 2011.

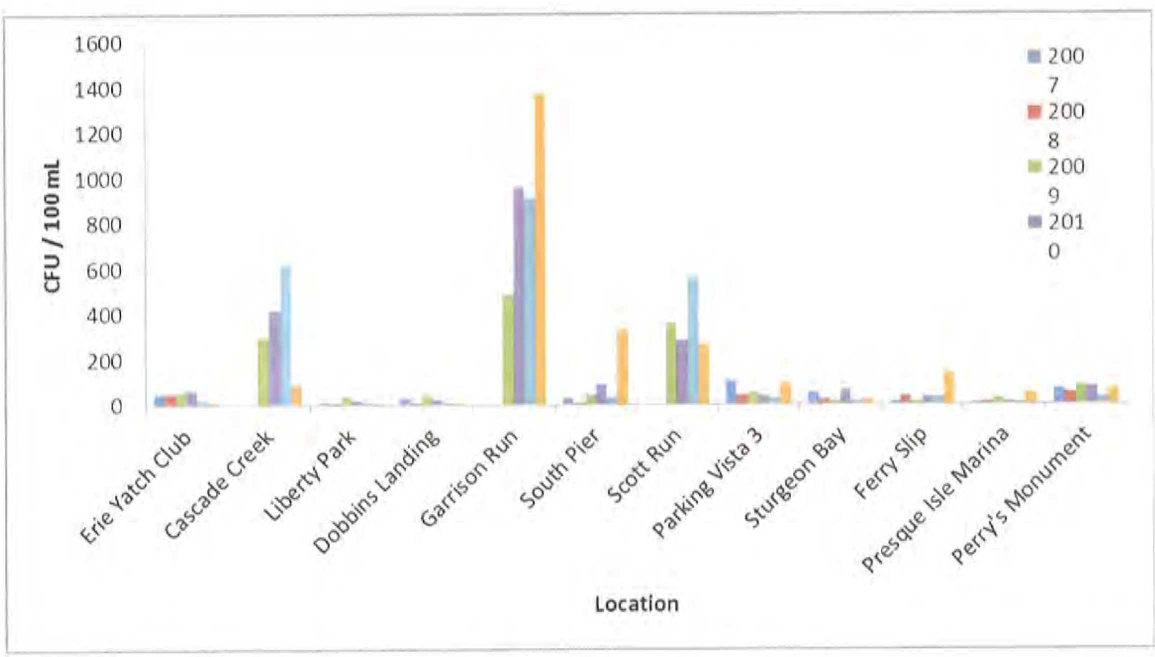


Figure 10. *E.coli* Counts in Presque Isle Bay Samples (2007-2010).

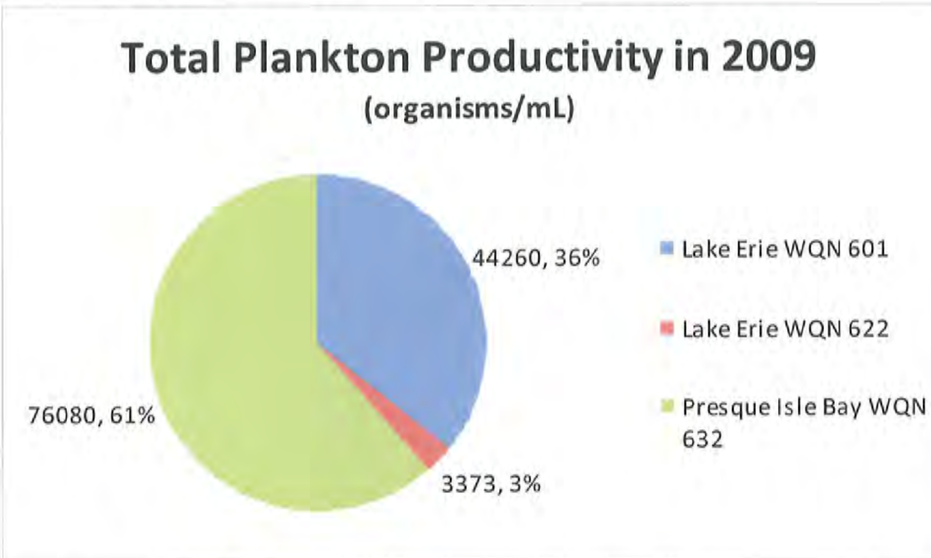


Figure 11. Relative plankton productivity at Lake Erie Water Quality Network Stations

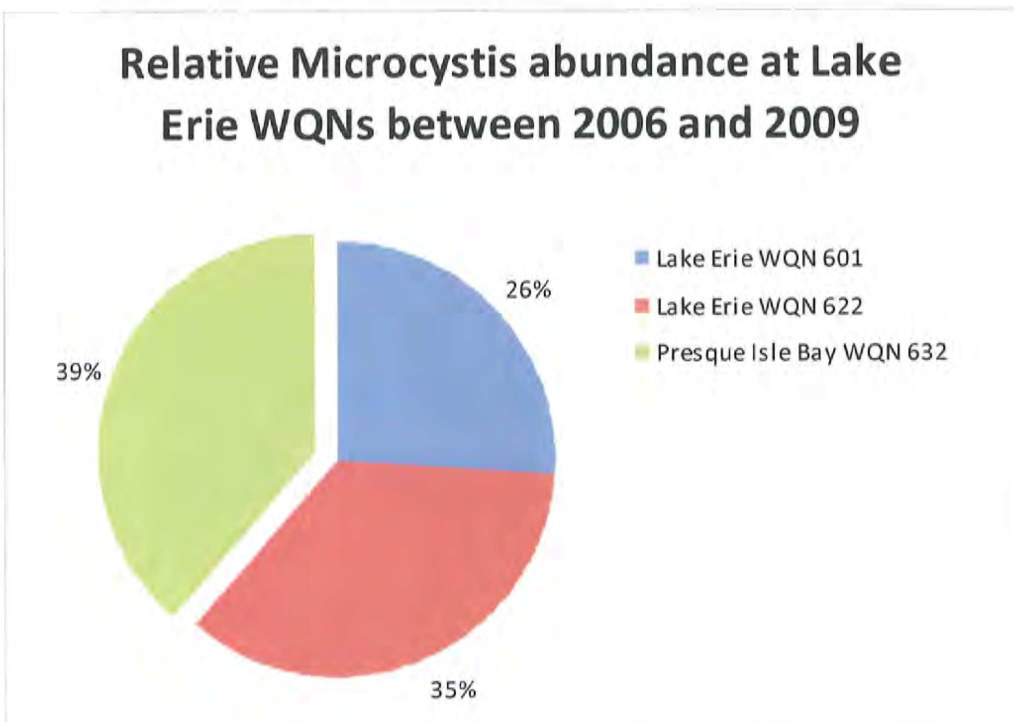
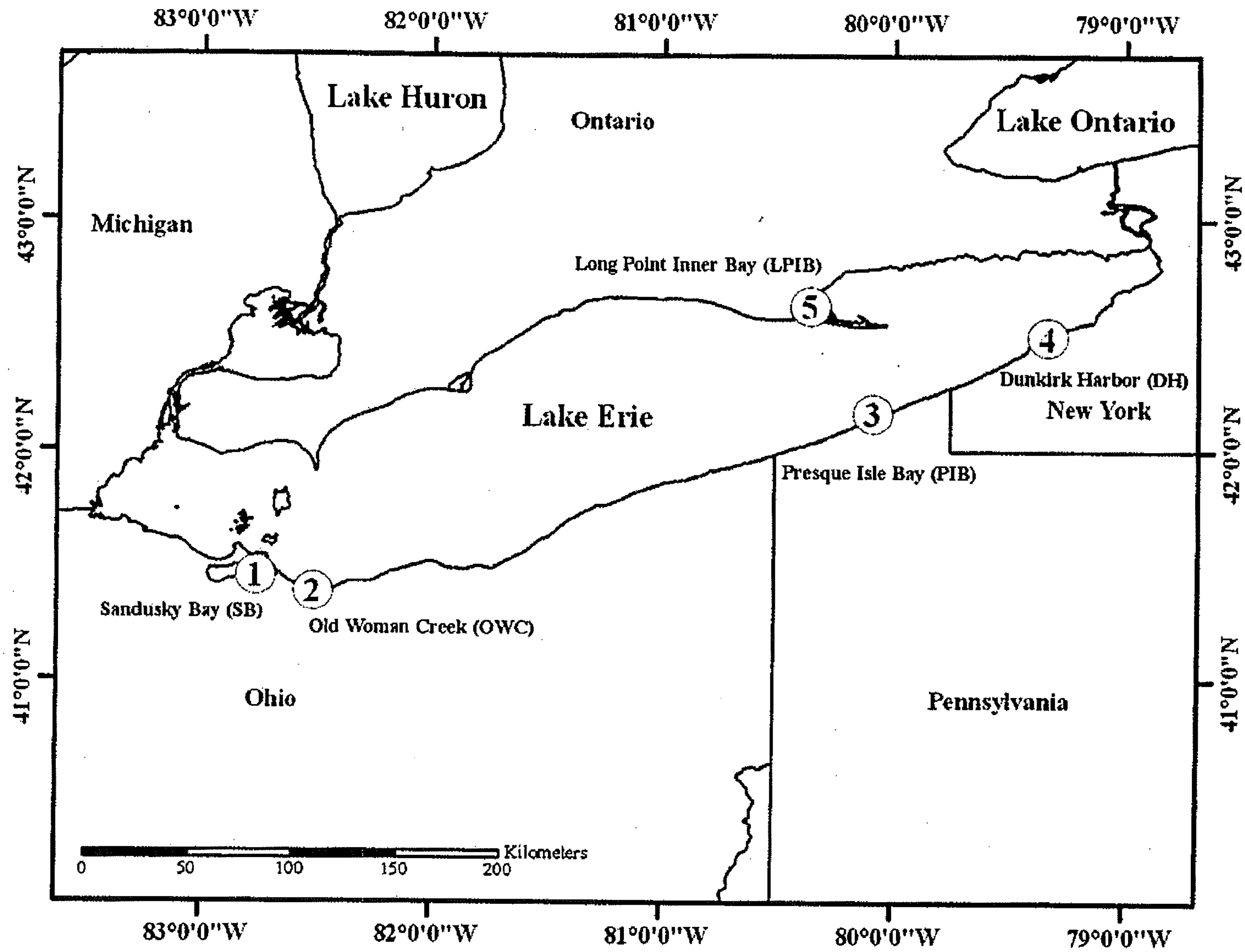


Figure 12. Relative abundance of the bluegreen *Microcystis* at Lake Erie Water Quality Network Stations, 2006-2009

Figure 13. Sampled Lake Erie Reference Locations.



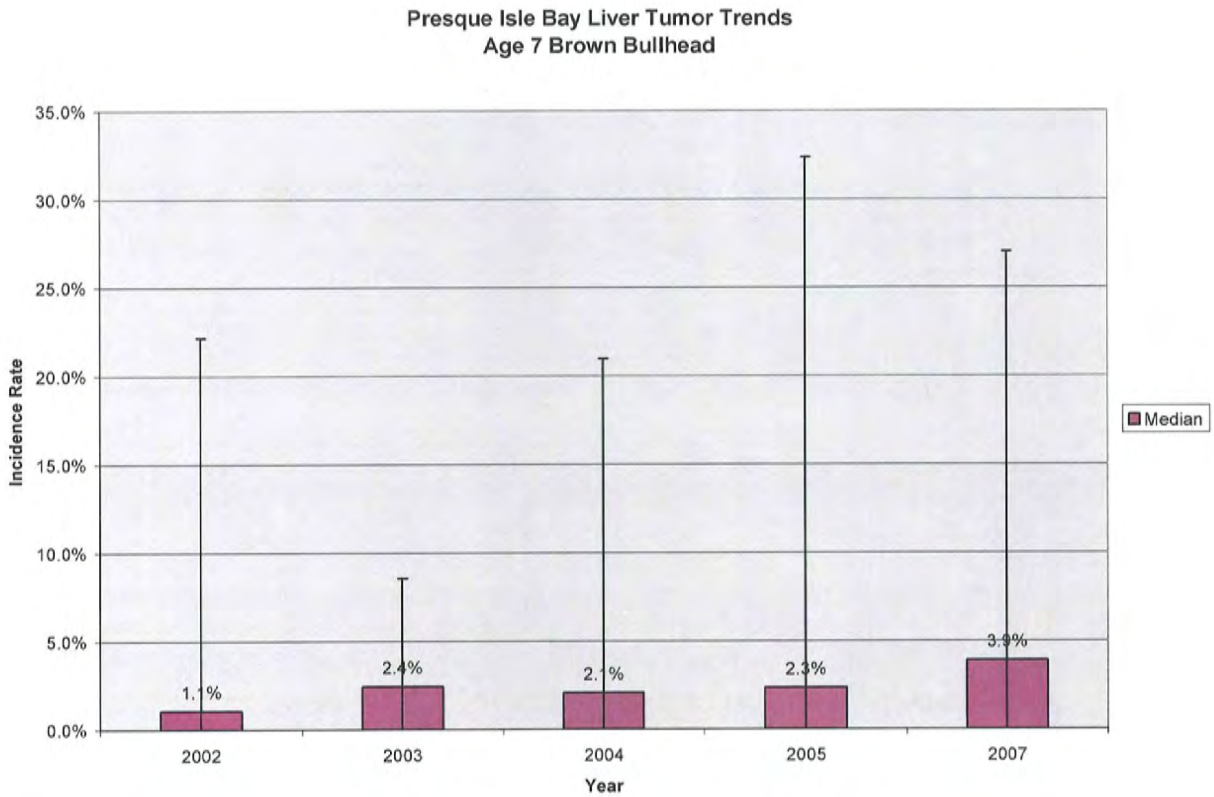


Figure 14. Recovery Period Liver Tumor Trends in Presque Isle Bay Brown Bullhead. Error Bars are ± Bayesian 95% Credible Intervals.

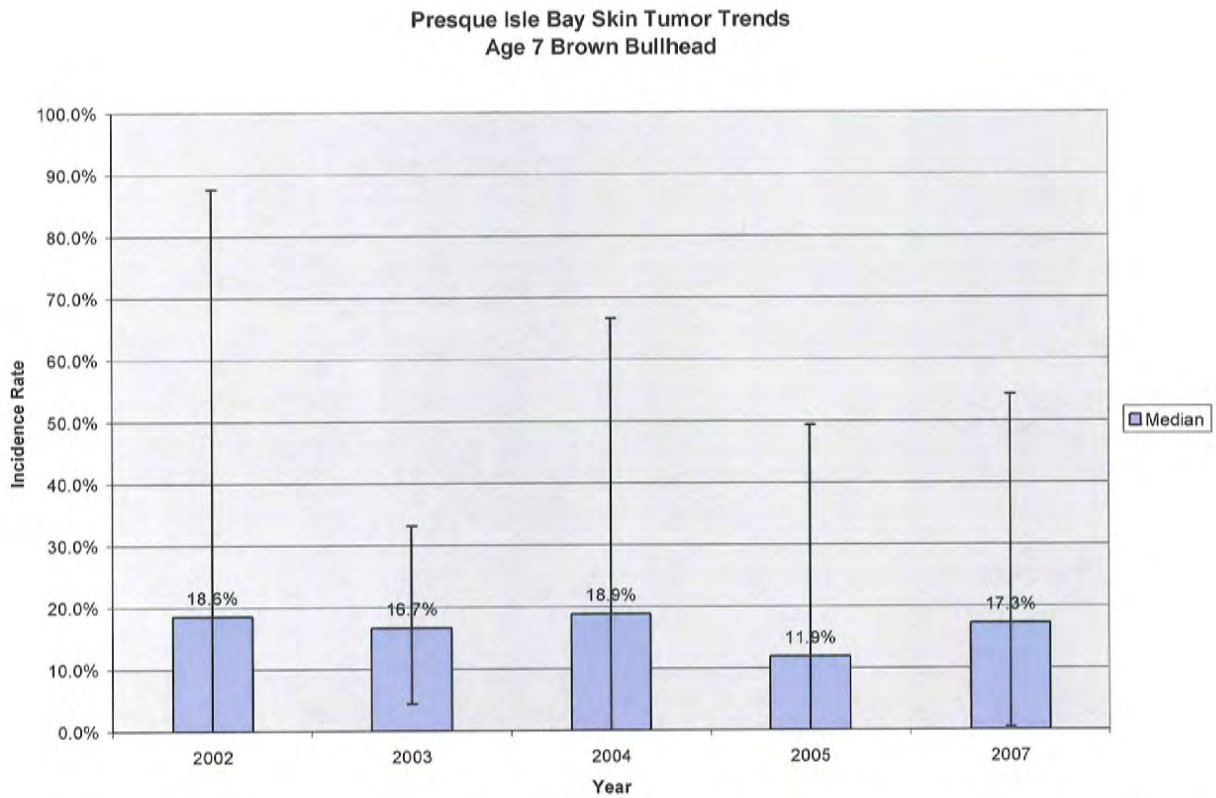


Figure 15. Recovery Period External Tumor Trends in Presque Isle Bay Brown Bullhead. Error Bars are ± Bayesian 95% Credible Intervals.

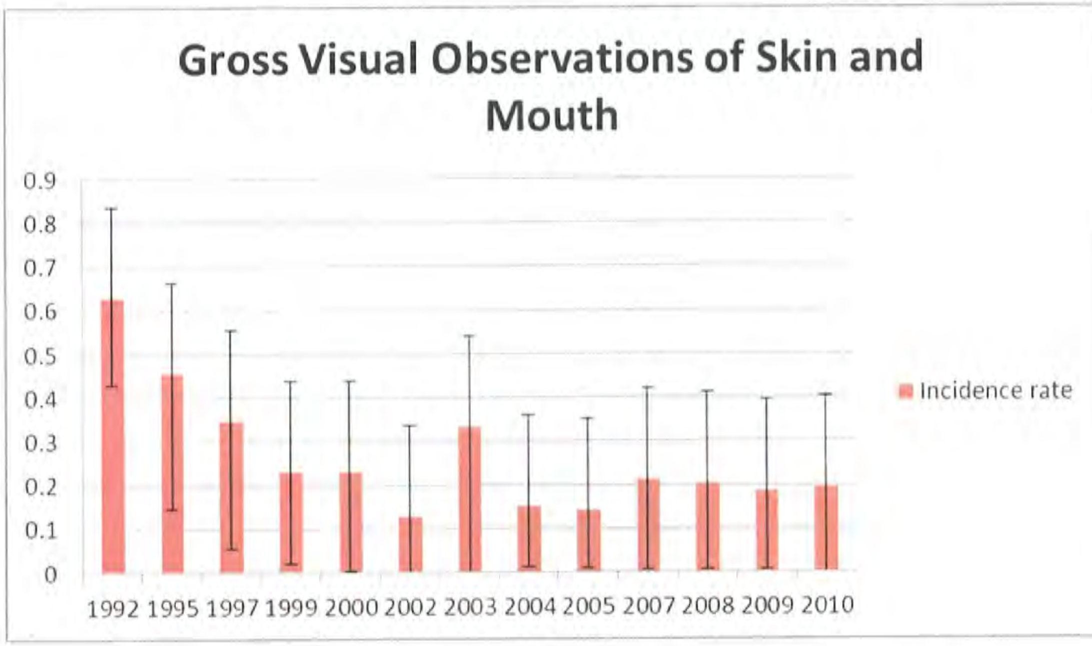
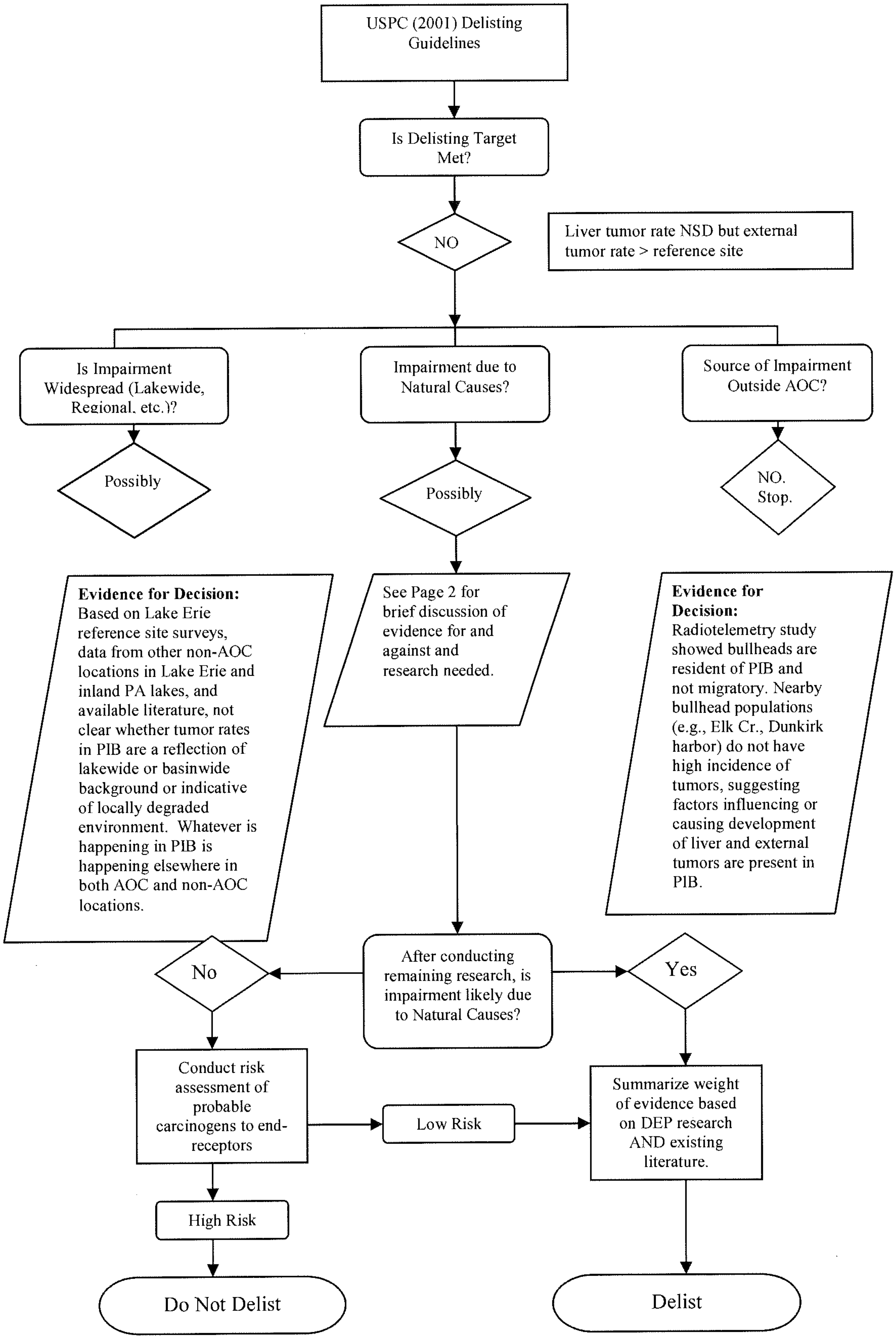


Figure 16. Incidence of Grossly Observed External Tumors on Presque Isle Bay Brown Bullhead (not confirmed with histopathology)

Figure 17. PADEP Delisting Process Decision Tree



Impairment due to Natural Causes?

Evidence For Natural Causes:

- High tumor rates documented in bullhead populations in some uncontaminated sites
- Lack of correlation between contaminants and bullhead tumors in several field studies
- Poor correlation between liver tumor and skin tumor incidence rates
- Natural and/or multiple causes *surmised* for external tumors in several recent studies
- 30-d sediment exposure study by Grady et al. (1992) was negative showed no effect
- DEP sediment exposure study (>450-d; ongoing) has been negative for grossly visible lesions
- Viruses, parasites, and other pathogens known to cause tumors in other species

Evidence Against Natural Causes:

- Numerous field studies showing a correlation between tumors and contaminants in bullhead and other species.
- One lab study (J. Black) showed relationship between application of contaminated sediment and skin tumor formation
- Lab studies with other species show association between contaminants and tumors (esp. liver tumors)
- Hybridization (and resulting genetic predisposition to tumors) has been ruled out.
- Attempts to identify viruses to date have been negative (mainly electronmicroscopy; molecular pilot study (PEER) by USGS also negative
- Attempts to isolate parasites to date have been inconclusive

Future Research Needed:

- Funding is needed to test for biomarkers of early-stage carcinogenesis (DNA adducts and histopathology) in bullhead in the DEP sediment exposure study. Otherwise, several additional years of exposure will be necessary before negative results can be accepted as valid.
- Viral analysis via contemporary molecular techniques is essential for definitive answers on this potential cause of tumors. We have partnered with USGS Leetown to conduct viral research this year.
- Microsporidian and Myxosporidian parasites are known to cause tumors in some fish species. We believe that additional work needs to be done to rule in or out these organisms as potential causes of the tumors.

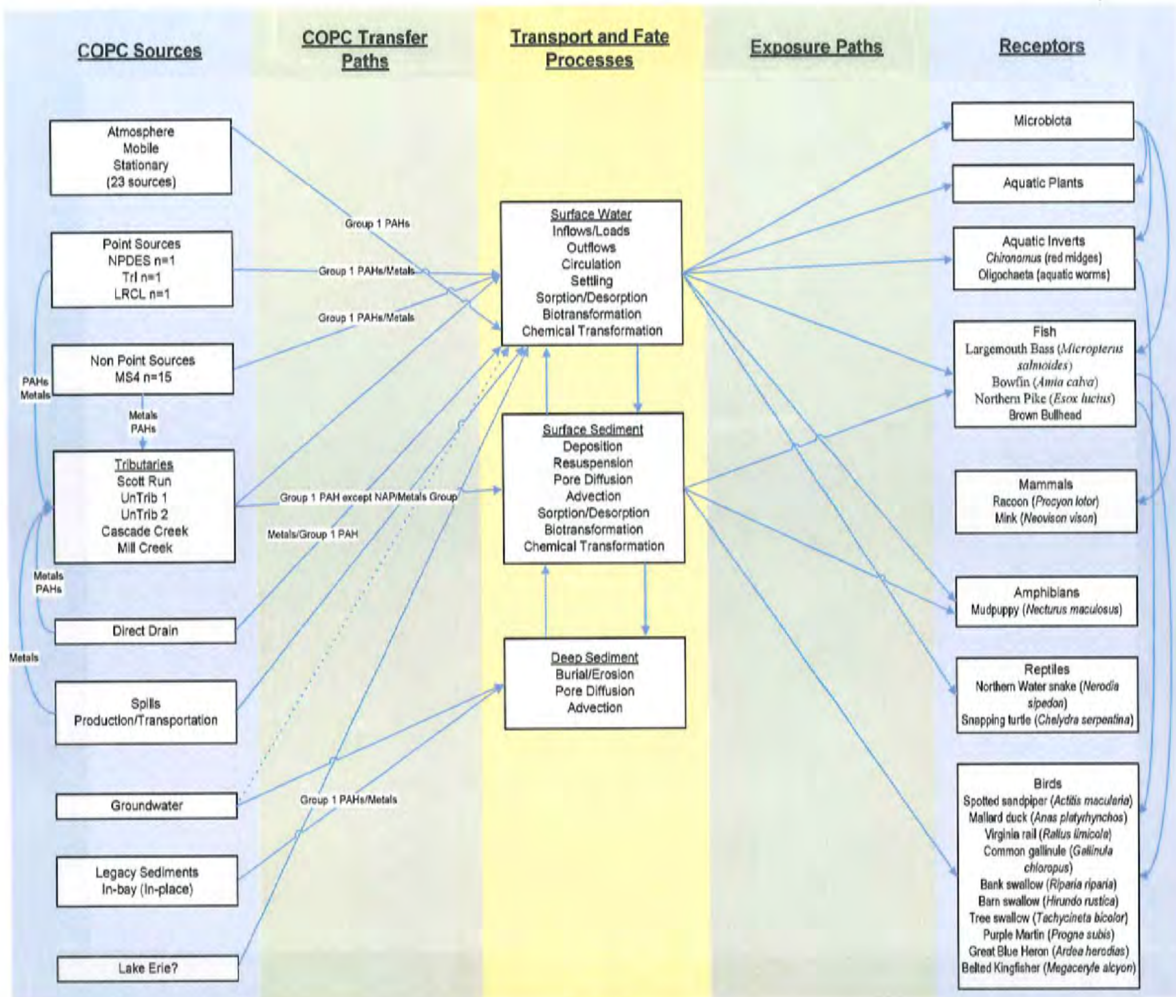


Figure 18. Conceptual site model for the Presque Isle Bay Ecological Risk Assessment.

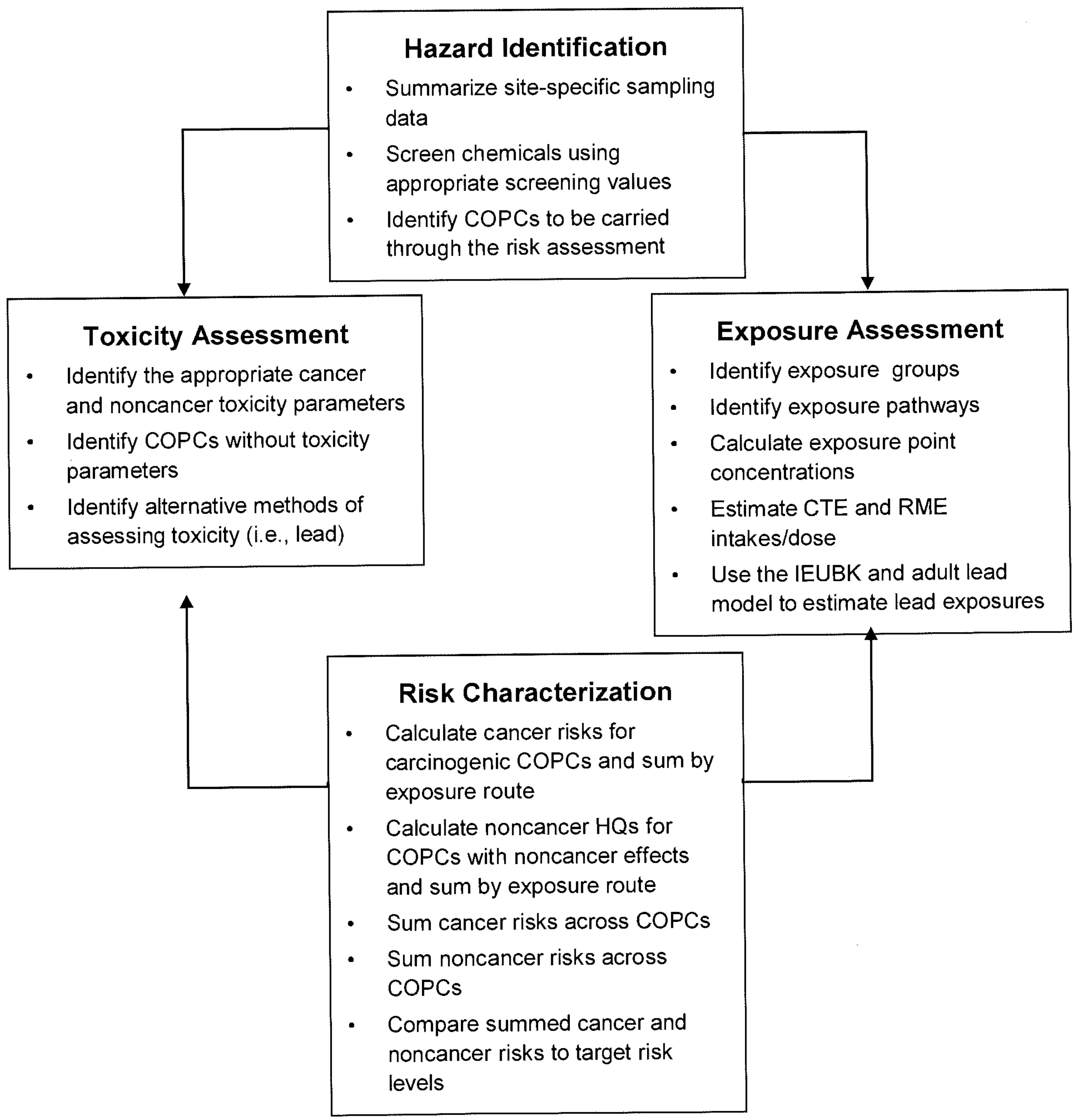


Figure 19. Overview of the four stage risk assessment process for the Presque Isle Bay Human Health Risk Assessment.

Final Report

Investigation into the Hybridization of *Ameiurus* catfish in Presque Isle Bay, Erie, Pennsylvania

GLNPO Project no. GL2003-340

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ABSTRACT

The Brown Bullhead, *Ameiurus nebulosus* (Lesueur, 1819), is a bottom-dwelling fish native to the Great Lakes that is commonly used to determine tumor prevalence in degraded aquatic environments. Brown Bullheads are in constant contact with benthic sediments due to their feeding habitats which may naturally expose them to industrial wastes and other contaminants trapped in bottom sediments. In 1991, the United States Department of State listed Presque Isle Bay, Lake Erie, Erie, Pennsylvania, as an Area of Concern of aquatic habitat for the primary impairments of sediment contamination and high incidences of epidermal and hepatic tumors in Brown Bullheads. Studies conducted in Presque Isle Bay found skin and liver tumor rates of Brown Bullheads have decreased between 1992 and 1999. It was proposed by Eric Obert, extension director of Pennsylvania Sea Grant that the Brown Bullhead population of Presque Isle Bay may contain some hybrids within the genus *Ameiurus*. Studies of hybrid fishes have shown that hybrids and succeeding backcross generations are highly sensitive to pollutants, while the parental wild species are less susceptible. The purpose of this study was to determine morphological and genetic variation within and among populations of Brown Bullheads and Black Bullheads in Presque Isle Bay, compared to other Brown Bullheads in other sites in Lake Erie. Morphological and meristic analysis indicates the majority of Brown Bullheads from Presque Isle Bay group with the reference Brown Bullhead population and not the reference Black Bullhead collection morphologically using principal component analysis. Collections from the Lagoons and Thompson's Bay each include an individual which maybe a hybrid, but what is likely being collected as a Brown Bullhead for the tumor studies in Presque Isle Bay is morphologically a Brown Bullhead. Genetically, over half of the Bullheads sampled and examined using microsatellite DNA were identified as having all *Ameiurus nebulosus* alleles, but multi-locus nuclear genotypes suggest the presence of extensive backcrossing between *Ameiurus nebulosus* and *Ameiurus melas* in Presque Isle Bay.

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Chapter 1

Introduction

The Brown Bullhead *Ameiurus nebulosus* (Lesueur, 1819), is a bottom-dwelling fish native to the Great Lakes that is commonly used to determine tumor prevalence in degraded aquatic environments (Baumann et al. 1996, Lesko et al. 1996, Smith et al. 1994). Brown Bullheads are in constant contact with benthic sediments due to their feeding habitats which may naturally expose them to industrial wastes and other contaminants trapped in bottom sediments (Lesko et al. 1996). In 1991, the United States Department of State listed Presque Isle Bay, Lake Erie, Erie, Pennsylvania, as an Area of Concern of aquatic habitat for the primary impairments of sediment contamination and high incidences of epidermal and hepatic tumors in Brown Bullheads. Studies conducted in Presque Isle Bay found skin and liver tumor rates of Brown Bullheads have decreased between 1992 and 1999 (Pyron et al. 2001). It was proposed by Eric C. Obert extension director of Pennsylvania Sea Grant (personal comm.) that the Brown Bullhead population imposed with tumors in Presque Isle Bay may be a hybrid within the genus *Ameiurus*. Studies of hybrid fishes have shown that hybrids and succeeding backcross generations are highly sensitive to pollutants (Setlow et al. 1989), while the parental wild species are less susceptible (Harshbarger and Clark 1990). If, in fact, the bullhead population in Presque Isle is comprised of hybrids and/or back-crossed individuals, then the tumor rate in this population may be exacerbated.

1.1 Presque Isle Bay – Area of Concern

Presque Isle Bay is located north of the city of Erie, Erie County, in the northwest corner of Pennsylvania. Presque Isle Bay is formed by a 1,295 hectare sandy, crescent peninsula reaching in a northeast direction on Lake Erie from the western portion of the city, and is Pennsylvania's only port on the Great Lakes. The bay is a relatively sheltered body of water and a closed system with a flushing time of almost 2.5 years. Presque Isle Bay is roughly 7.24 kilometers long with a maximum width of 2.41 kilometers and connects with Lake Erie through a narrow channel at the eastern end. The land use within the Presque Isle Bay watershed is approximately 80 precent urban and spans roughly 41 kilometers. Its primary tributaries are Cascade Creek and Mill Creek, which together account for two-thirds of the water flowing into the bay. Presque Isle Bay has suffered from the accumulation and degradation of contaminants discharged by point and nonpoint sources.

Presque Isle Bay was declared the Great Lakes' 43rd Area of Concern by the United States Department of State as recommended by the International Joint Commission in January of 1991. Great Lakes Areas of Concern (AOC) are severely degraded geographic areas within the Great Lakes Basin. Areas of Concern are defined by the United States-Canada Great Lakes Water Quality Agreement (Annex 2 of the 1987 Protocol) as geographic areas that fail to meet the general or specific objectives of the agreement where such failure has caused or is likely to cause impairment of beneficial use of the area's ability to support aquatic life. Currently, there are forty identified Areas of Concern, twenty-five are located completely within the United States, ten exclusively in Canada, and five are shared by both countries along river systems.

The International Joint Commission lists fourteen beneficial use impairments to be used by Areas of Concern as criteria for the listing and delisting process. In Presque Isle Bay, the impaired beneficial uses are restrictions on dredging of sediments; and fish tumors and other deformities. Sediments in Areas of Concern are often contaminated with industrial or agricultural pollutants released in the environment long ago such as polychlorinated biphenyl (PCB), polycyclic aromatic hydrocarbons (PAH), nitrosamines, and many heavy metals including: arsenic, barium, cadmium, chromium, copper, lead, nickel, and zinc (Diz 2002). Other contaminants continue to enter the environment though the burning of fossil fuels and runoff from agricultural and urban areas (International Joint Commission 1989). By restricting dredging activities in an Area of Concern, contaminated sediments are thus less likely to be disturbed and dispersed. A fish tumor or deformity impairment occurs when incidence rates of fish tumors or other deformities exceed rates at unimpacted control sites that are locally relevant and when survey data confirm the presence of neoplastic or preneoplastic liver tumors in Brown Bullheads or White Sucker (*Catostomus commersoni*). Unimpacted sites are areas where there is a lack of industrial or municipal pollution discharges located upstream or in the immediate areas where neighboring land uses have not disrupted ecosystem function or structure.

1.2 Indicator Organism

Brown Bullheads are frequently used in environmental contaminant studies because they are a scaleless, benthic fish in constant contact with the sediments, and have a known sensitivity to environmental carcinogens (International Joint Commission 1989).

Studies on the Brown Bullhead in Presque Isle Bay (PADEP 1992, PADEP 1995, PADEP 1997) showed rates of orocutaneous tumors decreased from 64 percent to 22 percent and liver tumors decreased from 10 percent to 3percent from 1992 to 1997. It was noted in the 1997 study that the age distribution of bullheads collected in the 1992 study were markedly older than bullheads in the 1995 study, which in turn were older than bullheads in the 1997 study. The oldest population, 1992, has the highest tumor rates while the youngest study population, 1997, had the lowest tumor rates. In the 1997 study however, tumors were shown in bullheads aged fifteen years or older, including the reference population. In a study of Presque Isle Bay Brown Bullheads conducted in 1999 (Pyron et al. 2001) a decrease in skin and liver tumor rates was not associated with the losses of larger, older individuals or declining reproduction rates. Their data provide evidence that the population is not losing older individuals; therefore the decline in tumor rates cannot be attributed to a younger population. It is suspected that hybridization between Brown and Black Bullheads may be a factor in the decrease in tumor rates (Eric Obert, Personal comm.). Studies of hybrid fishes have shown that hybrids and succeeding backcross generations are highly sensitive to pollutants, while the parental wild species are less susceptible (Harshbarger and Clark 1990, Setlow et al. 1989). If the Brown

Bullhead population in Presque Isle Bay is comprised of hybrids, their quantitative value may in fact be compromised.

In December 2002, with respect to the tumor rates decrease in bullheads, Presque Isle Bay was upgraded from an Area of Concern and designated to be an Area of Concern in the Recovery Stage, as the result of significant environmental improvement in the bay since the early 1990s. It became the first Great Lake Area of Concern in the United States to be upgraded to the recovery status. However, tumors are still present on bullheads in Presque Isle Bay and it is still unclear what is causing the tumors and deformities in the fishes.

1.3 Taxonomic status

Taylor (1954), while assembling the records of fishes collected by John N. Lowe in the Upper Peninsula of Michigan, placed the generic name *Ameiurus* in synonymy with *Ictalurus* and proposed to use the name Ictaluridae for the North American catfishes and bullheads. This submission had been generally followed until Lundberg (1992) separated *Ameiurus* from *Ictalurus*.

The catfish family Ictaluridae contains about sixty living and extinct species. Modern genera of Ictaluridae share several synapomorphies, including extensive jaw adductor muscle origin from the skull roof that is known to have evolved in the early Oligocene (Lundberg 1992). In the genus *Ameiurus*, seven extant species are recognized and seven extinct species are known from their fossilized remains. The oldest of these fossils provides a minimum age estimate for the genus of approximately thirty million years (Lundberg 1992).

Ameiurus is divisible into two morphological species groups, the *natalis* group and the *catus* group. The *catus* group is comprised of four species, including three "flat-head" bullheads not found in Pennsylvania: *A. platycephalus* (Flat Bullhead), *A. brunneus* (Snail Bullhead), and *A. serracanthus* (Spotted Bullhead). They usually have a flatted head, large eye, emarginated tail, and a large dark blotch in the basal portion of the dorsal fin. *Ameiurus catus* (White Bullhead) also has a relatively large eye, but has a more convex head, lacks dorsal fin blotch, and is somewhat intermediate between *Ictalurus* and the bullheads in having a moderately forked tail (Jerkins and Burkhead 1994). In Pennsylvania, the geographic range of *A. catus* has included the Susquehanna and Delaware river systems, and it has been introduced into parts of the Ohio River watershed.

The *natalis* group is comprised of three species: *Ameiurus melas* (Black Bullhead) *A. natalis* (Yellow Bullhead), and *A. nebulosus*. Of the three species, *A. natalis* and *A. nebulosus* commonly occur in Pennsylvania, whereas *A. melas* has an endangered status in Pennsylvania. The Black Bullhead's most eastern distribution occurs in western Pennsylvania and as a result, is rarely found. The last documented collection in Presque Isle Bay took place during the late spring of 1972 (AEA 1973) and was reported to be in a 1987 checklist from the Pennsylvania Fish Commission (PADEP 1991).

The native distribution of *Ameiurus* catfishes ranged from southern Canada, the St. Lawrence River, all the Great Lakes except Lake Superior and the Red River of the North in Ontario and Manitoba, south to the Gulf of Mexico and northern Mexico, in the streams of the Atlantic Coast from New York to Lake Okeechobee in Florida, to their westernmost point in central Montana (Smith 1985, Page and Burr 1991, Hubbs and Lagler 2004). Introductions have extended the range west of the Rockies in isolated pockets including areas of British Columbia, Alberta, Mexico, California, Arizona, Nevada, and Idaho.

1.4 Identification

Fishes belonging to the genus *Ameiurus* are medium sized, lack scales and have a large and flattened head. The teeth of the upper and lower jaws are minute and sharp, and arranged in broad pads. The swim bladder is connected with the Weberian ossicles, and is involved in the reception and production of sound. All members possess an often elongated adipose fin free at the posterior edge, four pairs of paired barbels, and a spinous ray in the dorsal fin and in each pectoral fin (Becker 1983).

Ameiurus melas (Rafinesque, 1820) Black Bullhead: *Ameiurus* –"primitive" or "curtailed" in reference to the slight notch in the caudal fin, *melas* - black.

Black Bullheads have a robust body, rounded anteriorly, compressed posteriorly (Figure 1). Snout is bluntly pointed in lateral view and broadly rounded in dorsal view; with elongated barbels on the snout just anterior to posterior nostrils. Black Bullheads have a mouth that is short but wide, terminal and horizontal. Black Bullheads have very long barbels sweeping posteriorly from upper jaw at each corner of the mouth and four shorter barbels attached in a transverse line on the lower chin. The fish has numerous minute needlelike teeth in broad bands on upper and lower jaws. Dorsal fin origin about midway between pectoral and pelvic fins; dorsal fin with a stout spine and 5-6 rays; dorsal adipose fin free at posterior end. Anal fin rays including rudimentaries are 15-21 (Becker 1983), sometimes 17-21 (Smith 1985, Trautman 1981). The pectoral fin has a stout spine without sharp teeth on the posterior edges that catch the finger (Trautman 1981). The caudal fin is somewhat square and slightly notched at midpoint, and the lateral line is complete (Becker 1983). Trautman (1981) notes the body of an adult Black Bullhead is usually bi-colored with a sharp demarcation between the darker lower sides and the lighter ventral sides and a light, ventral, caudal bar, usually conspicuous in large young and adults that connects with the light color of the ventral surface.

Ameiurus nebulosus (Lesueur, 1819) Brown Bullhead: *nebulosus* – clouded, in reference to mottled coloring.

The Brown Bullhead has a stout body, compressed posteriorly (Figure 2). The head of the Brown Bullhead is depressed and the profile of the dorsum is straight in juvenile to distinctly convex in some adults (Jenkins and Burkhead 1994); they have a small eye, and the mouth is slightly subterminal with jaws equal or with the upper jaw slightly longer. The caudal fin is usually slightly emarginated, sometimes straight in small young. Chin barbels are gray, black, or black-spotted by their base. The anal fin usually has 22-23 (extremes 21-24) rays, counting rudimentaries; its distal margin usually slightly rounded. The posterior edges of pectoral spines have many sharp teeth, which may become blunted in large individuals (Trautman 1981). The dorsal fin has a stout spine and 6-7 soft rays. The body of adult Brown Bullheads is often conspicuously mottled, especially on the sides, and there is no sharp demarcation line between ventral surface of the body and lower sides (Trautman 1981).

Brown Bullheads and Black Bullheads are often difficult to distinguish but have been reported to be separable by the character of the serrae on the posterior edge of the pectoral spine: moderate serrae in Brown Bullheads (Figure 2) and weak serrae in Black Bullheads (Figure 1) (Trautman 1981, Hubbs and Lager 1991, Jenkins and Burkhead 1994). The posterior spine serrae in Black Bullheads are variable, being absent to moderately developed. Although most often weakly developed in adult Black Bullheads, the pectoral serrae are unreliable for consistently distinguishing Black Bullheads from Brown Bullheads (Burkhead et al. 1980).

Fin pigmentation differences have also been reported. Of these characters, only the depigmented “bar” at the caudal base of Black Bullhead is consistently present, and then only in larger juveniles and adults. However, it is often evident only when directly compared to specimens of Brown Bullheads (Burkhead et al. 1980). Black Bullheads are best distinguished from Brown Bullheads by higher and rarely overlapping gill raker counts. Brown Bullheads have 3 or 4 gill rakers on the epibranchial limb, and 8 or 9 gill rakers on the first ceratobranchial limb. Black bullheads will have 5 to 7 gill rakers on the epibranchial limb and 10 to 15 gill rakers on the first ceratobranchial limb.

1.5 Biology

Ameiurus spawn in late spring to early summer in Pennsylvania. Spawning takes place in open excavations in sand and gravel, and in the shelter of logs, rocks, or vegetation (Becker 1983, Cooper 1983). Both males and females may contribute to nest construction but this is primarily the female's duty (Smith 1985). The spawning act takes place by the pair facing in opposite direction with their bodies in close contact and the female depositing from 50 to 10,000 or more eggs in the nest. Generally, the male or both parents guard the nest and protect the young for a time (Trautman 1981). When the young rise off the nest, the parents swim about them in circles to keep them in a compact school, and strays are caught in their parents' mouth and returned to the school (Becker 1983). Adult bullheads are most active at night. When they are active in daytime, it is generally in muddy, clouded water. They have poor vision and use their sense of smell and the taste buds on the skin, lips and barbels to find food. Bullheads are opportunistic feeders that eat whatever food is available, including carrion (Becker 1983).

The Black Bullhead seems to prefer silty waters and soft mud bottoms, and is highly tolerant of many types of industrial and domestic pollutants, as well as warm water temperatures (Trautman 1981). It appears incapable of invading in abundance the deeper, cooler, clearer waters, with or without some vegetation, which is the habitat of the Brown Bullhead, or the very clear, heavily vegetated habitat of the Yellow Bullhead (Trautman 1981).

1.6 Hybridization

The offspring between one full species and another full species are called F1 hybrids. The offspring between an F1 hybrid and an individual of either parent species are called backcrosses. In fish hybrids a blending of the parent characters normally occur, and the identification of F1 hybrids can be difficult or impossible (Trautman 1981).

Brown and Black Bullheads are known to naturally hybridize and Trautman (1981) reports that there had been considerable hybridization and backcrossing between Black and Brown Bullheads in western Lake Erie. Trautman (1981) goes on to note that “when mass hybridization occurs in the small, silty, largely vegetationless impoundments, the majority of the population resembles Black Bullheads, and that a large number of “typical” Black may be present, but there may be few or no “typical” Browns. In deeper waters the situation appears to be reversed, and backcrosses usually favor the Brown rather that the Black Bullhead. Both bullheads are spring spawners and rely on thigmotactic and chemosensory clues to modify their spawning behaviors and recognize individuals in a population (Cooper 1983, Page and Burr 1991).

Other freshwater fishes, such as sunfish in the genus *Lepomis* readily hybridize in polluted waters, where conditions hinder species recognition (Page and Burr 1991). Stauffer et al. (1979) attributed natural hybridization to the overcrowding of spawning fishes, abiotic stress, and cohabitation of rare and abundant fishes.

Studies of hybrid fishes have shown that hybrids and succeeding backcross generations are highly sensitive to pollutants, while the parental wild species are less susceptible. Certain hybrids such as the Platyfish-Swordtail hybrid (*Xiphophorus maculates* x *Xiphophorus helleri*) and succeeding backcross generations are highly sensitive to carcinogens, while the parental wild species are not susceptible to neoplasia (Setlow et al. 1989). Also, the hybrid of European Carp (*Cyprinus carpio*) x Goldfish hybrid (*Carassius auratus*) is believed to have a genetic predisposition to neoplasia, unlike its parental species (Harshbarger and Clark 1990). A better understanding of the bullhead population in Presque Isle Bay is essential for the continued use of this population as an indicator species for the Great Lakes.

1.7 Purpose

The purpose of this study is to determine morphological and genetic variation within and among populations of Brown Bullheads and Black Bullheads in Presque Isle Bay, compared to other Brown Bullheads in other sites in Lake Erie. Further study maybe warranted to determine if hybridization of the Lake Erie or Presque Isle Bay Brown Bullheads promote higher tumor rates than areas outside the Great Lakes' basin.

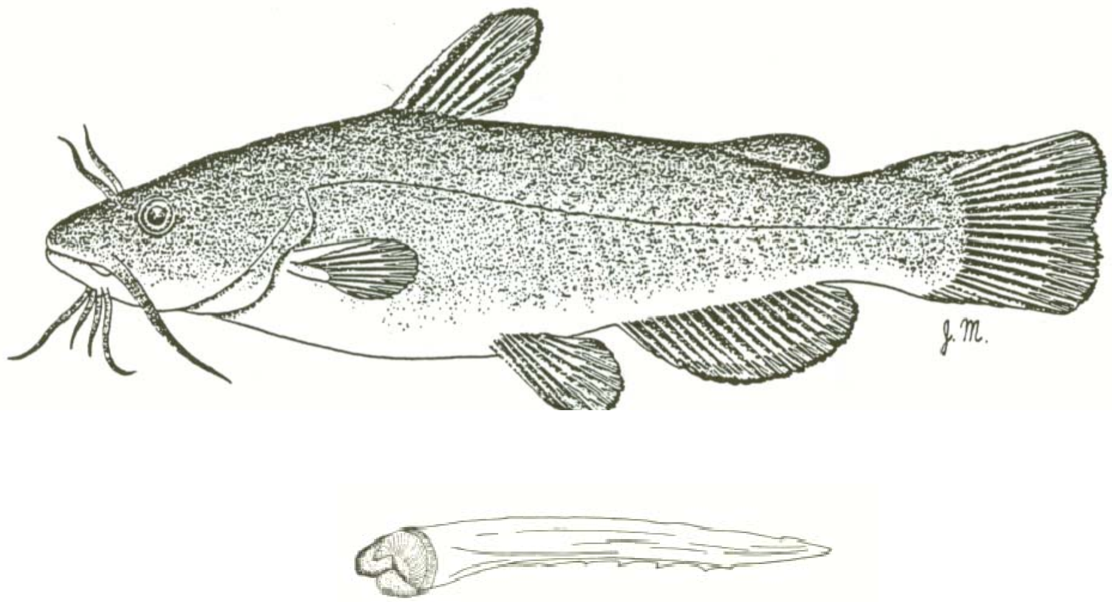

Figure 1. Lateral view of *Ameiurus melas* and pectoral spine serrae (Cooper 1983)

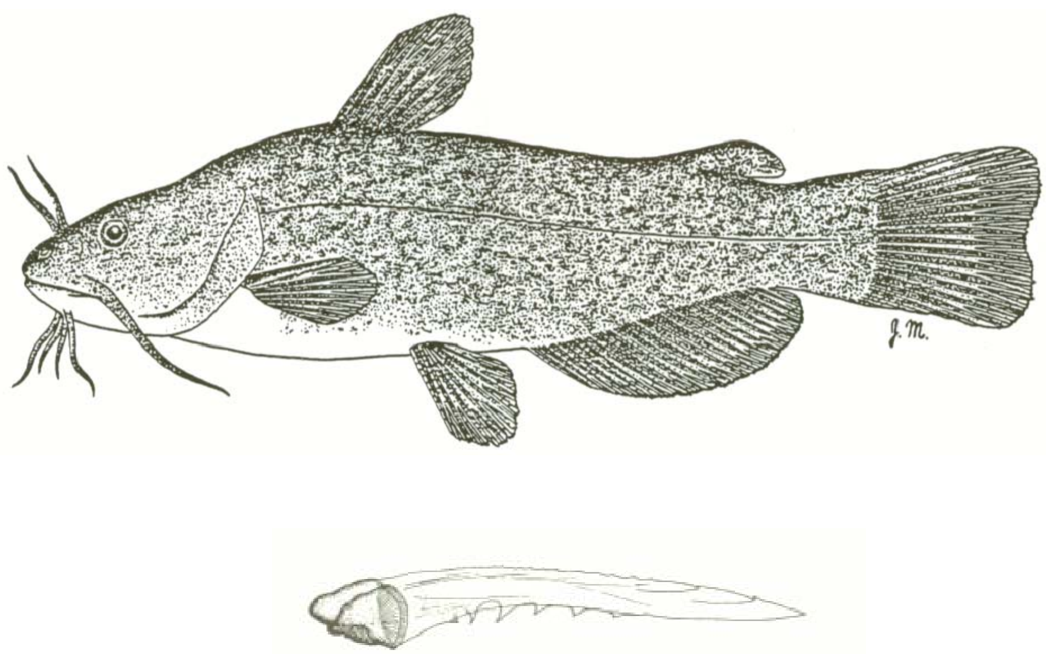

Figure 2. Lateral view of *Ameiurus nebulosus* and pectoral spine serrae (Cooper 1983)

Chapter 2

Methods and Materials

2.1 Samples and Collections

Ameiurus nebulosus specimens for this study from Presque Isle Bay, Lake Erie, Pennsylvania *(Latitude = 42°09'N; Longitude = 80°04'W)* (Figure 3); Dunkirk Harbor, Dunkirk, New York *(Latitude = 42°49'N; Longitude = 79°34'W)*; and Old Woman Creek, Ohio *(Latitude = 41°21'N; Longitude = 82°30'W)* (Figure 4) were collected by electrofishing. After capture, all individuals were anesthetized with sodium benzocaine (MS-222). A piece of the right pectoral fin was removed and placed in a numbered 1.5 ml microcentrifuge tube containing 95% ethanol for later genetic analysis. All individuals were tagged with a number corresponding to the genetic analysis, preserved in 10% formalin for one week, washed in water for two days, and later stored in 70% ethanol. Additionally, the Pennsylvania Department of Environmental Protection provided thirty *A. nebulosus* specimens from Long Point Bay, Ontario, Canada *(Latitude = 42°58'N; Longitude = 82°36'W)* (Figure 3); and the Pennsylvania Fish and Boat Commission collected thirty *A. nebulosus* in trap nets from Tamarack Lake, Crawford County, Pennsylvania *(Latitude = 41°35'N; Longitude = 80°05'W)*. Twenty-eight *A. nebulosus* specimens were collected by rod and reel from a reservoir in Petersburg, Huntingdon County, Pennsylvania *(Latitude = 40°57' N; Longitude = 78°05' W)* and served as reference specimens for the Brown Bullhead. These individuals were collected from outside the historic range of the Black Bullhead in Pennsylvania. All collections at a particular site were obtained in one sampling trip.

Thirty specimens of *Ameiurus melas* were obtained from Clear Lake in Clear Lake, Iowa *(Latitude = 42°56’ N; Longitude = 93°63’W)* by the Iowa Department of Natural Resources. Eleven samples of *A. melas* were provided by the Wisconsin Department of Natural Resources but were used only for comparisons to the Iowa samples. The Iowa specimens served as reference specimens for Black Bullheads. The *A. melas* specimens were shipped frozen and later thawed, when the fish were tagged and the right pectoral fin was removed, placed in 1.5 ml microcentrifuge tubes containing 95% ethanol, and stored in the laboratory prior to DNA preparation leaving the left side of the fish intact for morphological analysis.

All specimens were assessed for the presence of external lesions and gross deformities using the specifications of the Pennsylvania Fish and Boat Commission’s Deformities, Erosions, Lesions, and Tumors (DELTs) index. Bullheads used in previous tumor studies from Long Point Bay, Ontario were a reference population located on Lake Erie with no known point-source of contaminants and Old Woman Creek, Ohio is a reference site having only low-level PAH contamination at railway and highway bridges (Baumann et al. 1996).

Twenty-eight specimens from each site were included in this study with the exception of the Lagoons, Presque Isle Bay (thirty specimens) and Dunkirk, New York, which consisted of twenty-two specimens.

All specimens were deposited in the permanent collections of the Pennsylvania State University Fish Museum.

2.2 Mensural and Meristic Characters Examined

A total of six meristic (count) and twenty-five mensural (measurement) characters (Table 1) was examined on each individual from all nine sites totaling 7,688 separate measurements or counts. All measurements were made with Fowler Promax 150 mm digital calipers and recorded to the nearest .01 mm.

Counts followed methods as outlined in Hubbs and Lager (1958). Morphometric distances were measured also as described by Hubbs and Lager (1958) with the exception noted. All measurements were taken point to point except head depth, which was measured from the point equidistant and dorsal to the midline of each eye to a point directly vertical on the base of the fish. All counts and measurements were made on the left side of the fish, except gill raker counts which required incisions to the dorsal and ventral junction of the operculum to expose the gill rakers on the right side of the fish. All gill rakers on the right arch were counted including rudiments on the first ceratobranchial limb with the exception of the raker straddling the angle of the arch. Counting was aided by the use of a variable magnification stereo dissecting microscope.

2.3 Mensural and Meristic Character Analysis

Meristic differences were analyzed using principal component analysis in which the correlation matrix was factored. Head, body, and fin shape variation were assessed by analyzing the mensural data using sheared principal component analysis, in which the covariance matrix was factored. This procedure restricts size variation to the first principal component; and subsequent components are strictly shape related (Humphries et al. 1981). Comparisons among species were made by plotting the first principal component (PC1) from the meristic variation and the sheared second principal component (SPC2) from the mensural variation. Minimum polygon clusters were drawn to encompass the points of a species or a population on the principal components plots. A multivariate analysis of variance (MANOVA) was used to test differences among the minimum polygon clusters formed by each species in the plots.

Principal component analysis (PCA) was chosen for this study over discriminant analysis (DA) to uncover unknown trends in the data. Principal component analysis does not attempt to *a priori* group data by user-specified criteria or presume multiple groups and thus allow for their discovery (Humphries et al. 1981). Principal component analysis is a way of identifying patterns in data, and expressing the data in such a way as to emphasize their similarities and differences. The use of PCA allows the number of variables in a multivariate data set to be reduced, while retaining as much as possible of the variation present in the data set (McGarigal et al. 2000). Principal component analysis organizes entities along continuous gradients defined by the principal components and seeks to describe the sources of greatest variation among the entities, where entities are generally assumed to represent a single random sample of a known of unknown number of populations. For PCA to work, the data set must consist of a single set of two or more continuous, categorical and/or count variables, and no distinctions exists between independent and dependent variables (McGarigal et al. 2000).

The main purpose of discriminat analysis is to describe the differences among two or more well-defined groups and predict the likelihood that an entity of unknown origin will belong to a particular group based on a suite of discriminating characteristics. Discriminate analysis assumes the variables are independent. Classification is a part of discriminate analysis and classifies entities into groups using a classification criterion that, in general, maximizes correct classification of entities into prespecified groups (McGargal et al. 2000). Principal component analysis was performed using the SAS® system for windows, version 8.02 and MINITAB®, release 14.

2.4 Fin Digestion and DNA Extraction

Fin clips were blotted dry of ethanol, minced into small pieces using a clean razor blade and placed into 1.5 ml microcentrifuge tubes. 500µL lysis buffer (0.1M Tris, 4M urea, 0.2M NaCl, 0.01M CDTA, 0.5% lauroyl sarcosine) with 5µL proteinase K solution (0.1mg/ml concentration) was added, and the samples were incubated overnight at 55°C.

500µL equilibrated Phenol:Chroloform:Isoamyl alcohol (25:24:1) was added and inverted seven times, and spun in a microfuge for 10 minutes. The top layer was transferred to a new tube and 500µL Chloroform:Isoamly alcohol (24:1) was added, inverted and centrifuged for two minutes. The top layer was removed to a new tube and 1000µL of cold 95% EtOH was added and inverted and centrifuged for 20 minutes at 4°C. The ethanol was removed by decanting and the DNA pellet was washed with 200µL cold 70% EtOH. The ethanol was again removed by decanting and the pellet was dried over night. The extracted DNA was resuspended in 100µL HPLC grade water and stored at 4°C.

2.5 Polymerase Chain Reaction Recovery of Enriched DNA

Eight samples of extracted *Ameiurus nebulosus* DNA were initially sent to Dr. Travis Glenn (Savannah River Ecological Laboratory, Aiken, South Carolina) for construction of a genomic library enriched for microsatellite loci (Glenn and Schable 2005). Extracted DNA was enriched for $(AAAG)^{6}$, $(ACAG)^{6}$, $(AGAT)^{8}$, $(ATCC)^{5}$ and $(ACAT)^{8}$ following a protocol available from Travis Glenn (glenn@srel.edu). In brief, the DNA was digested with *Rsa*I, ligated to Super- SNX linkers, hybridized to biotinylated microsatellite oligonucleotides, captured on Dynabeads (Dynal Biotech Inc.) and unwanted DNA was washed away.

The enriched DNA fragments were amplified using polymerase chain reaction (PCR) using, 1.67µM SuperSNX-f (5′-GTTTAAGGCCTAGCTAGCAGAATC-3′), 10X PCR buffer, 250µg/ml BSA (Bovine Serum Albumin), 0.3125 mM of each dNTP, 4.17 mM $MgCl_2$, 0.5 units/µL *Taq* DNA polymerase (Fisher Brand), and HPCL $H_2$0 in a total volume of 12 µL. PCR was conducted in a DNA Dyad Thermalcycler (MJ Technologies) with the following profiles: 2 minute hot start at 95°C, followed by 25 cycles of 20 seconds at 95ºC, 20 seconds at 60°C, and 1.5 minutes at 72ºC, with a final extension step of 30 minutes at 72°C. Electrophoresis was conducted with 5µL of PCR product using a 3% SB agarose gel containing ethidium bromide and SB buffer (Brody et al. 2004) at 300 V for 10 minutes for verification of successful enrichment and DNA recovery.

2.6 Ligating Enriched DNA into Plasmids and Sequencing of MiniPrep Clones

The enriched DNA library was ligated into the PCR 4-TOPO cloning vector by TA cloning using Invitrogen's TOPO-TA cloning kit and following the manufacture's protocol. The ligated cloning vectors were transformed into One-Shot TOP10 chemically competent *E. coli* cells (Invitrogen) following the manufacture's protocol. Ampicillin (*amp*) sensitive bacteria and a vector that carries a gene conferring *amp* resistance were used to incorporate the enriched/recovered DNA + cloning vector into a bacterial host. Colonies were plated on LB plates containing *amp* antibiotic, to permit screening of successful transformants. One hundred-twenty clones were picked and swabbed into 3 mL tubes of LB medium with *amp* antibiotic and incubated overnight at 37°C. Plasmid DNAs were purified using a S.N.A.P. MiniPrep Kit (Invitrogen). Colonies were screened for inserts by PCR as following: each 10µL reaction contained miniprepped 1.5 µL plasmid DNA as template, along with 250µg/mL BSA (Bovine Serum Albumin), 10X PCR reaction buffer, 10 mM each T3 and T7 primers, 4.17mM $MgCl_2$, 0.5 mM of each dNTP, 0.5 units/µL *Taq* DNA polymerase (Fisher Brand), and dH_20. PCR was conducted in a DNA Dyad Thermalcycler (MJ Technologies) with the following profiles: 2 minute hot start at 95°C, followed by 35 cycles of 20 seconds at 95ºC, 20 seconds at 50°C, and 1.5 minutes at 72ºC, with a final extension step of 10 minutes at 72°C.

Electrophoresis was conducted with 5µL aliquots of PCR product using a 3% SB agarose gel containing ethidium bromide and SB buffer at 300 V for 10 minutes. PCR products containing an insert were cleaned by column centrifugation using Princeton separation columns with Sephadex® G-50 (Sigma). The clean PCR products were quantified by spectrophotometer and saved for cycle sequencing. Sequencing reactions were conducted using ¼ reactions with BigDye® v 3.1 cycle sequencing kit (ABI). Reactions consisted of 2µL BigDye® master mix, 2µL 10 µM T7 sequencing primer, 6µL 2.5x sequencing buffer, and ~40-80ng of clean PCR product + HPLC water to make a total volume of 20µLs. Samples were cycled 55-75 times in a DNA Dyad Thermalcycler (MJ Technologies) according to manufacturer's suggestion.

Following the cycle sequencing reaction, products were again cleaned by Sephadex® G-50 column centrifugation and placed into a DNA SpeedVac on medium heat for about 30 minutes or until dry. The dry samples were reconstituted with 10µL of DI formamide, transferred to a 96 well plate, denatured for 2 minutes at 95°C, and snap cooled on ice. Sequences were analyzed on an ABI PRISM® 3100-Avant Genetic Analyzer following the manufacturer's settings.

2.7 Primer design and selection

Twenty microsatellite primers were designed using Oligo 6.6 (Molecular Biology Insights, Cascade, CO) and ordered from the Penn State Nucleic Acid Facility (Penn State University, University Park, PA). Each microsatellite locus was screened in six *A. nebulosus* and three *A. melas* "pure" parental type specimens by PCR and electrophoresis. PCR conditions were optimized by altering $MgCl_2$ concentrations and/or annealing temperatures. To check for amplification, 5 µL of PCR product was loaded onto a 2% SB agarose gel and electrophoresed in 1X SB buffer for 45 minutes at 150 V. Primers were then chosen to be fluorescently labeled for genotyping based on non-overlapping allele sizes between the parental type specimens.

2.8 Fluorescent Primer optimization, selection, multi-plexing, and genotyping

Nine fluorescently labeled microsatellite primer sets for *A. nebulosus* were designed (Table A2). The alleles ranged in size from 160-300 base pairs in length.

PCR conditions were optimized by altering $MgCl_2$ concentrations and/or annealing temperatures. The optimized PCR conditions for each individual locus can be found in the appendix. Each PCR reaction used 12 ng of DNA. To check for amplification, 6 μL of PCR product was loaded onto a 2% SB agarose gel and electrophoresed in 1X SB buffer for 15 minutes at 300 V.

Of the nine loci, five were selected to be the primary markers for this study (Aneb16, Aneb37, Aneb61, Aneb63, and Aneb64). These diagnostic loci were then applied to the remaining reference, Presque Isle Bay, Lake Erie, Tamarack Lake, and Wisconsin specimens. Multiplexing was performed with the Aneb37 and Aneb64 primer pairs in one reaction and Aneb61 and Aneb63 in another (Table 2 A-B). Aneb16 was performed separately (Table 2 C). The optimized multiplex and single PCR reactions were used to genotype a total of 248 bullheads.

Fluorescently labeled PCR product was then prepared for fragment analysis on the ABI PRISM® 3100-Avant Genetic Analyzer. A size standard of GeneScan-500 LIZ (Applied Biosystems) was run with each sample. Samples for fragment analysis consisted of 0.5 µL of LIZ size standard, 9.5 µL of formamide and 0.5 µL of fluorescently labeled PCR product and were loaded into each well of a 96-well plate. Once all PCR products were added, the plate was denatured at 95^{o} C for 2 minutes and snap cooled on ice. The plate was mounted on the ABI PRISM® 3100-Avant Genetic Analyzer and programmed for fragment analysis according to the manufacturer. The data were analyzed using GENESCAN, and genotypes were recorded. Genotypic data of the five loci were run through Hardy-Weinberg exact tests, linkage disequilibrium tests, allele frequency tests and the F-statistics F_{IS} and F_{ST} using the population genetics software GENEPOP (Raymond and Rousset 2004).

Figure 3. Presque Isle Bay collection sites; 1) Sara's Cove located at the head of the bay, 2) the lagoons, a series of connected ponds, and 3) Thompson's Bay located in the outer harbor.

Figure 4. Lake Erie collection sites: 4) Old Woman Creek, Ohio, 5) Long Point Bay, Ontario, 6) Dunkirk Harbor, New York, and 7) Presque Isle Bay, Pennsylvania.

Table 1. Morphological characters recorded from specimens of *Ameiurus nebulosus* and *Ameiurus melas.*

Mensural variable	**Mnemonic**	**Corrected by ***
Standard length	SL	
Head length	HL	A
Head width	HW	B
Postorbital head length	POHL	B
Interorbital width	HED	B
Interorbital height	VED	B
Preorbital length	PRE	B
Cheek depth	CD	B
Lower jaw length	LJL	B
Head depth	HD	B
Body depth	BD	A
Distance from snout to dorsal fin insertion	SNDOR	A
Distance from snout to pelvic fin insertion	SNPEL	A
Dorsal fin base length	DFBL	A
Distance from anterior dorsal fin to anterior anal fin	ADAA	A
Distance from anterior dorsal fin to posterior anal fin	ADPA	A
Distance from posterior dorsal fin to anterior anal fin	PDAA	A
Distance from posterior dorsal fin to posterior anal fin	PDPA	A
Distance from posterior dorsal fin to ventral point of least caudal peduncle	PDVC	A
Distance from posterior anal fin to dorsal point of least caudal peduncle	PADC	A
Distance from anterior dorsal fin to insertion of pelvic fin	ADP2	A
Distance from posterior dorsal fin to insertion of pelvic fin	PDP2	A
Caudal peduncle length	CPL	A
Least caudal peduncle length	LCPD	A
Anal fin base length	AFBL	A
Meristic variable	**Mnemonic**	
Dorsal fin rays	drays	
Anal fin rays	arays	
Pectoral fin rays	P1rays	
Pelvic fin rays	P2rays	
Epibranchial gill raker	EGR	
Ceratobranchial gill raker	CGR	

* All measurements except SL were corrected for the size of the fish by either dividing by SL denoted as A or by dividing by HL denoted as B.

Table 2. Optimized Multiplexing Conditions for Selected Loci

A. Aneb37 and Aneb64 primer sets multiplex

multiplex	**1 rxn**	**Thermocycling**	**conditions**	
10X buffer B	1.92 μl	**Step**	**Temp °C**	**Time**
dNTP [1.25 mM]	2.0 μl	Denature	95	30 s
$MgCl_2$ [1.50 mM]	0.792 μl	Annealing	57	30 s
Aneb37F-PET [0.01mM]	0.3 μl	Extension	72	1 m
Aneb37R [0.01 mM]	0.3 μl	Cycles	32	
Aneb64F-FAM [0.01mM]	0.3 μl	Final Extension	72	2 m
Aneb64R [0.01 mM]	0.3 μl	Incubate	15	forever
Taq [5U/μl]	0.11 μl			
HPLC water	3.998 μl			
DNA template	2 μl			
Total	12 μl			

B. Aneb61 and Aneb63 primer sets multiplex

mulitplex	**1 rxn**	**Thermocycling**	**conditions**	
10X buffer B	1.92 μl	**Step**	**Temp °C**	**Time**
dNTP [1.25 mM]	2.0 μl	Denature	95	30 s
$MgCl_2$ [1.50 mM]	0.792 μl	Annealing	57	30 s
Aneb61F-FAM [0.01mM]	0.3 μl	Extension	72	1 m
Aneb61R [0.01 mM]	0.3 μl	Cycles	32	
Aneb63F-NED [0.01mM]	0.3 μl	Final Extension	72	2 m
Aneb63R [0.01 mM]	0.3 μl	Incubate	15	forever
Taq [5U/μl]	0.11 μl			
HPLC water	3.998 μl			
DNA template	2 μl			
Total	12 μl			

C. Aneb16 primer

	1 rxn	**Thermocycling**	**conditions**	
10X buffer B	1.20 μl	**Step**	**Temp °C**	**Time**
dNTP [1.25 mM]	2.0 μl	Denature	95	30 s
$MgCl_2$ [2.25 mM]	1.19 μl	Annealing	50	30 s
Aneb16F-FAM [0.01 mM]	0.3 μl	Extension	72	1 m
Aneb16R [0.01 mM]	0.3 μl	Cycles	32	
Taq [5 U/μl]	0.11 μl	Final Extension	72	2 m
HPLC water	4.004 μl	Incubate	15	forever
DNA template	2 μl			
Total	12 μl			

Chapter 3

Meristics and Morphometrics

3.1 Meristics - Principal Component Analysis

Meristic differences were analyzed using principal component analysis (PCA). Principal component analysis is a multivariate ordination technique commonly used for examining morphological variables and to differentiate closely related species (Stauffer et al. 1997). Principal component analysis identifies patterns in a data set and eliminates redundancy in univariate analysis when mulitcollinear data are involved (Iezzoni and Pritts 1991). The main purpose of PCA is to convert a number of correlated variables into a smaller set of components of the original variables called principal components with minimum loss of information. Each set is uncorrelated with any other set, but components with in the set are related. This is done by creating linear combinations of the original variables, which are oriented in directions along continuous gradients defined by the principal components and seeks to describe the sources of greatest variation among entities (McGarigal et al. 2000). The first principal component accounts for as much of the variability in the data as possible, and each succeeding component accounts for as much of the remaining variability as possible (McGarigal et al. 2000). Principal component analysis compares the sources of greatest variation in the data set and produces scores for each individual. Morphological relationships are determined by comparing these scores to the scores of other individuals also contained in the data set.

3.2 Morphometrics - Sheared Principal Components Analysis

Sheared principal component analysis (SPCA) is effective in identifying shape differences among the populations independent of size (Reyment et al. 1984) and was used to assess the head, body and fin variation. Sheared principal component analysis ordinates morphometric data independently of a main ordination, allowing for the mensural variables to be analyzed independent of size (Reyment et al. 1984). The first principal component identifies size differences while succeeding sheared principal components, being independent of size, detect shape (Brookstein et al. 1985, Humphries et al. 1981). Sheared principal component analysis was used by Stauffer (1991) to distinguish among *Pseudotropheus pursus* Stauffer, *P.lanisticola* Burgess, and *P. livingstonii* (Boulenger) from Lake Malawi and by Stauffer et al. (1997) to describe a new genus of North American minnows *Pararhinicthhys*, which arose from intergentic hybridization events between *Rhinichthys cataractae* and *Nocomis micropogon*.

3.3 Reference Specimens

Principal component analysis was conducted on the meristic data. The clusters were formed by plotting the second principal component of the meristic data, PC2, against the first principal component of the meristic data, PC1 (Figure 5). The non-overlapping of the two minimum polygon clusters, generated from the principal components plots, illustrates the difference in the meristic data that distinguish *A. nebulosus* and *A. melas*. Variable loadings on these two factors are listed in Table 3. The three factors accounting for 2.3 %, 1.16 % and 0.99 % of the variability follow respectively. Gill rakers on first epibranchial, gill rakers on first ceratobranchial, and anal fin rays, account for almost all of the variability in PC1, while pectoral fin (p1) rays and dorsal fin rays account for the majority in PC2.

Sheared principal component analysis was then conducted on the mensural variables of the reference specimens. The first principal component of the SPCA (SPC2) of the mensural data was plotted against the second sheared principle component of the SPCA (SPC3) of the mensural data (Figure 6) to assess the ability of the mensural data in detecting shape differences. Minimum polygon clusters were then made for the reference specimens to determine the differences for *Ameiurus nebulosus* and *A. melas*. A minimum polygon cluster is a closed figure on the two dimensional plot that includes the spatial data points of all individuals belonging to a particular sample. The non-overlapping of the two minimum polygon clusters, generated from the principal components plots demonstrates the difference in the mensural data that distinguish *A. nebulosus* and *A. melas*. The variable loadings SPC2 and SPC3 are listed in Table 4. The first principal component of the morphometric data, which is a size component, accounted for 89.9 % of the total variance while first and second sheared principal components accounted for 2.3% and 1.6% of the remaining 10.1 % variance, respectively. Variables that had highest loadings on the first sheared principal component were cheek depth (0.258), preorbital distance (0.244) and lower jaw length (0.241). Snout to pelvic fin distance accounted for the most variability on the second sheared principal component.

Minimum polygon clusters were formed by plotting the first principal component (PC1) of the meristic data against the first sheared principal component (SPC2) of the morphometric data (Figure 7). This plot yields no overlap between the minimum polygon clusters of the reference specimens of *A. nebulosus* and the reference specimens of *A. melas* and show differences between the two taxa. This technique was used for the remaining collections including the fifty-six reference specimens and either one collection or as all collections in Presque Isle Bay, Lake Erie, or Tamarack Lake.

3.4 Presque Isle Bay Collections

A plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from a data set including reference specimens of *A. nebulosus* and *A. melas* and the collections from Presque Isle Bay were plotted individually and pooled against the reference specimens.

Collections from Sara's Cove contained individuals very similar morphologically to the reference specimens of *A. nebulosus* along both the PC1 (meristic data) and SPC2 (mensural data) axes (Figure 8). Almost all individuals from these collections fell in close proximity to the cluster or in the minimum polygon cluster for reference specimens of *A. nebulosus*. An individual is considered as falling within a minimum polygon cluster if a data point is within the boundaries of the polygon or touching any side of the polygon. Those individuals falling above and outside the range of PC1 and SPC2 for *A. nebulosus* may suggest heterosis in Sara’s Cove. Collections from the lagoons (Figure 9) and Thompson’s Bay (Figure 10) each include one individual whose PC1 value fall outside of the range for *A. nebulosus* and within the range of *A. melas*. When all the Presque Isle Bay specimens are pooled together and plotted (Figure 11), most all the individuals fall in close proximity to the cluster or in the minimum polygon cluster for the reference specimens of *A. nebulosus*, but does contain an individual whose PC1 value falls outside of the range for *A. nebulosus* and within the range of *A. melas*.

3.5 Lake Erie Collections

A plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data form a data set including reference specimens of *A. nebulosus* and *A. melas* and all of the collections from Lake Erie were plotted individually and pooled against the reference specimens. Collections from Old Woman Creek, Ohio, contain a few individuals whose PC1 value falls outside of the range for *A. nebulosus* and within the range of *A. melas*. Most individuals fell in close proximity to the cluster or in the minimum polygon cluster for reference specimens of *A. nebulosus* (Figure 12). Collections from Long Point Bay, Ontario Canada have individuals very similar morphologically to the reference specimens of *A. nebulosus* along both the PC1 and SPC2 axes (Figure 13). Collections from Dunkirk Harbor, New York includes one individual whose PC1 values fall outside of the range for *A. nebulosus* and within the range of *A. melas*. Most individuals fell in close proximity to the cluster or in the minimum polygon cluster for reference specimens of *A. nebulosus* (Figure 14).

When all the Lake Erie specimens are pooled and plotted along with the reference specimens (Figure 15), there are many more individuals whose PC1 values fall outside of the range for *A. nebulosus* and within the range of *A. melas*.

3.6 Tamarack Lake – Inland Brown Bullhead Collection

A plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data form a data set including reference specimens of *A. nebulosus* and *A. melas* and the collection from Tamarack Lake is shown in Figure 16. Almost all individuals from these collections fell in close proximity to the cluster or in the minimum polygon cluster for reference specimens of *A. nebulosus* along both the PC1 and SPC2 axes.

3.7 Wisconsin – Black Bullhead Population

A plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data form a data set including reference specimens of *A. nebulosus* and *A. melas* and the collections of *A. nebulosus* from Tamarack Lake and *A. melas* from Wisconsin is shown in Figure 17. Almost all brown bullheads from Tamarack Lake fell in close proximity to the cluster or in the minimum polygon cluster for reference specimens of *A. nebulosus*. In contrary, all Black Bullheads but one from the Wisconsin population has SPC2 value that fell outside of the range for *A. melas* and within the range of *A. nebulosus*.

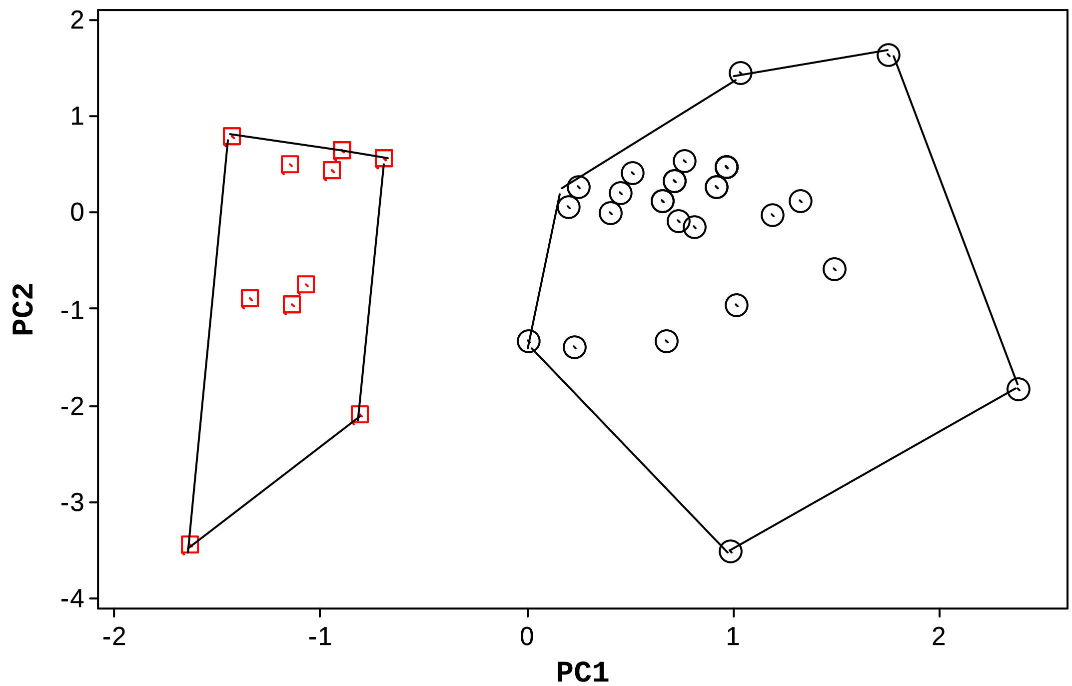

Figure 5. Plot of the first and second principal components, PC1 and PC2 respectively, derived from the principal component analysis of the reference specimens of *Ameiurus nebulosus* (squares) and *A. melas* (circles) using the meristic data.

Table 3. Variance loadings of the meristic characters on the first two principal components describing variation in fin ray counts and gill rakers of the reference specimens of *Ameiurus nebulosus* and *A. melas*.

Character	**PC1**	**PC2**
Dorsal fin rays	0.32465	0.56980
Anal fin rays	-0.76364	-0.20202
Pelvic fin rays	-0.32276	0.26438
Pectoral fin rays	0.20522	0.77254
Gill raker count on first epibranchial	0.87536	-0.33094
Gill raker count on first ceratobranchial	0.83773	-0.14656

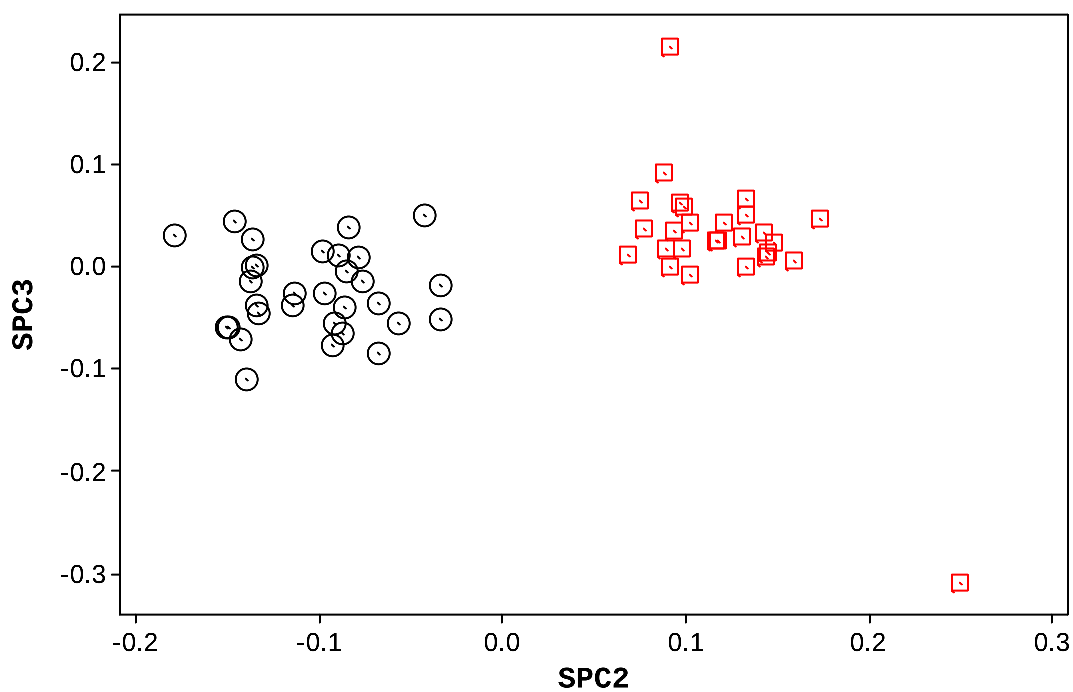

Figure 6. Plot of the first and second sheared principal component, SPC2 and SPC3 respectively, derived from the sheared second principal component analysis of the reference specimens of *Ameiurus nebulosus* (squares) and *A. melas* (circles) using the mensural data.

Table 4. Variance loadings of the mensural characters on the first two sheared principal components describing variation in shape of the reference specimens of *Ameiurus nebulosus* and *A. melas.*

Character	SPC2	SCP3
SL	0.04750	-0.12354
HL	-0.04957	-0.05888
HW	-0.13415	-0.05297
POHL	-0.04070	-0.21031
HED	0.01585	0.38523
VED	0.01700	0.19233
PRE	-0.22750	0.70155
CD	-0.37289	0.04866
LJL	-0.27757	-0.27279
HD	0.04163	0.06032
BD	0.19364	0.07696
SNDOR	-0.07245	-0.08943
SNPEL	0.67590	0.12456
DFBL	0.09837	-0.23889
ADAA	0.08872	0.01544
ADPA	0.08603	-0.12868
PDAA	0.09118	0.03520
PDPA	0.11655	-0.11772
PDVC	0.07988	-0.06405
PADC	0.12900	0.02364
ADP2	0.20695	0.04013
PDP2	0.21529	0.11873
CPL	0.04502	-0.10634
LCPD	0.17280	0.04409
AFBL	0.12244	-0.15568

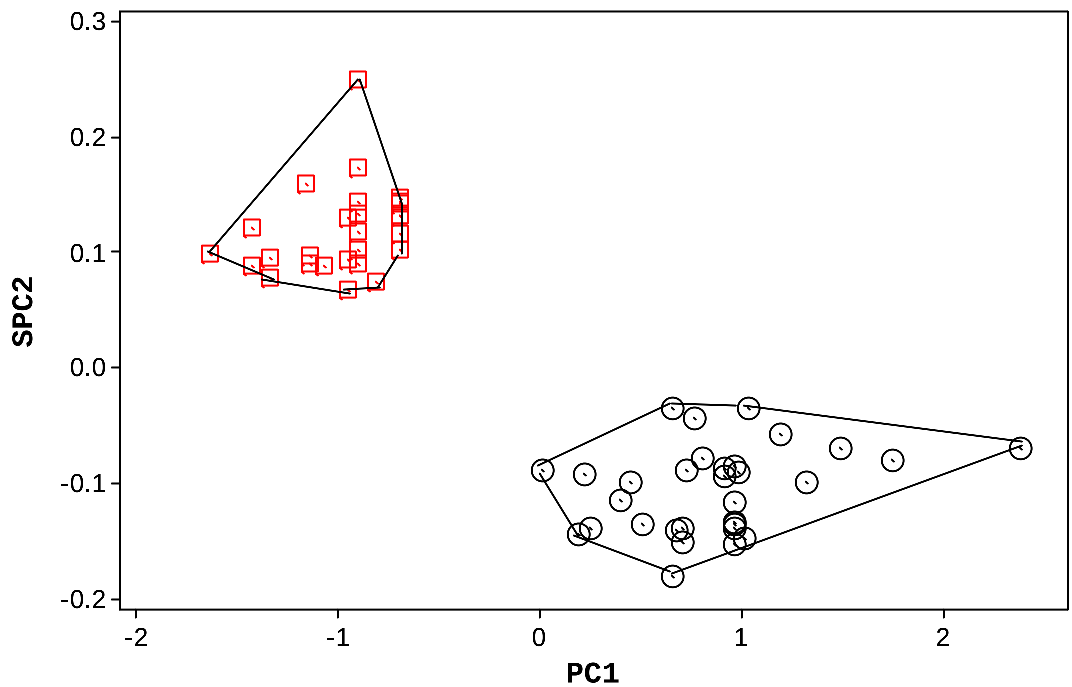

Figure 7. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares) and *A. melas* (circles).

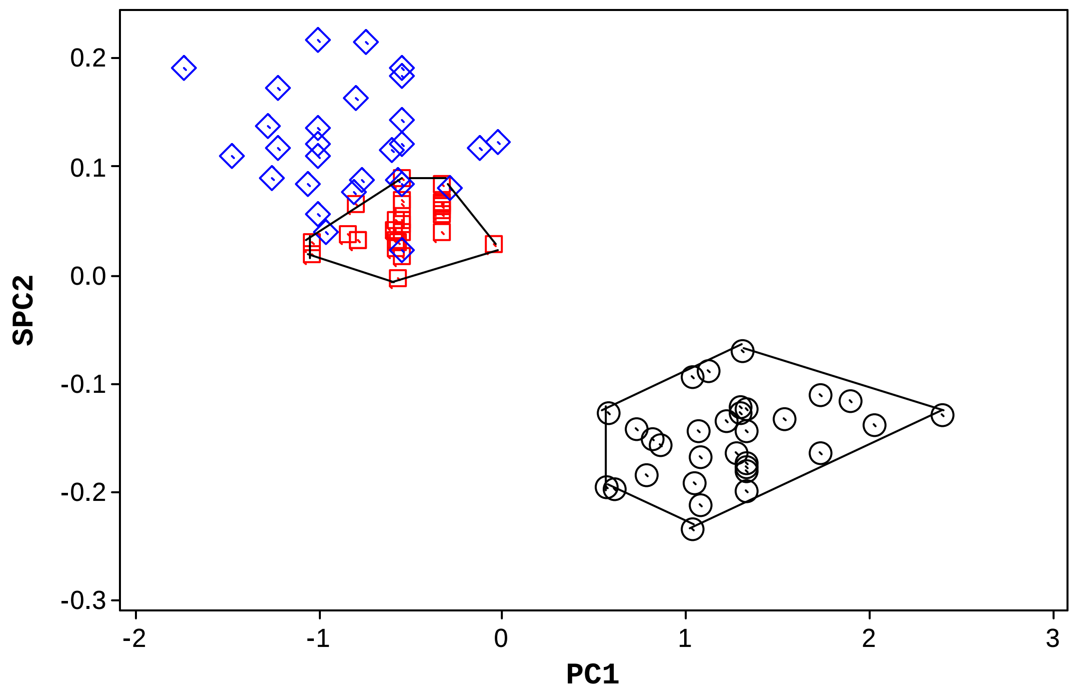

Figure 8. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles) and Sara's Cove (diamonds).

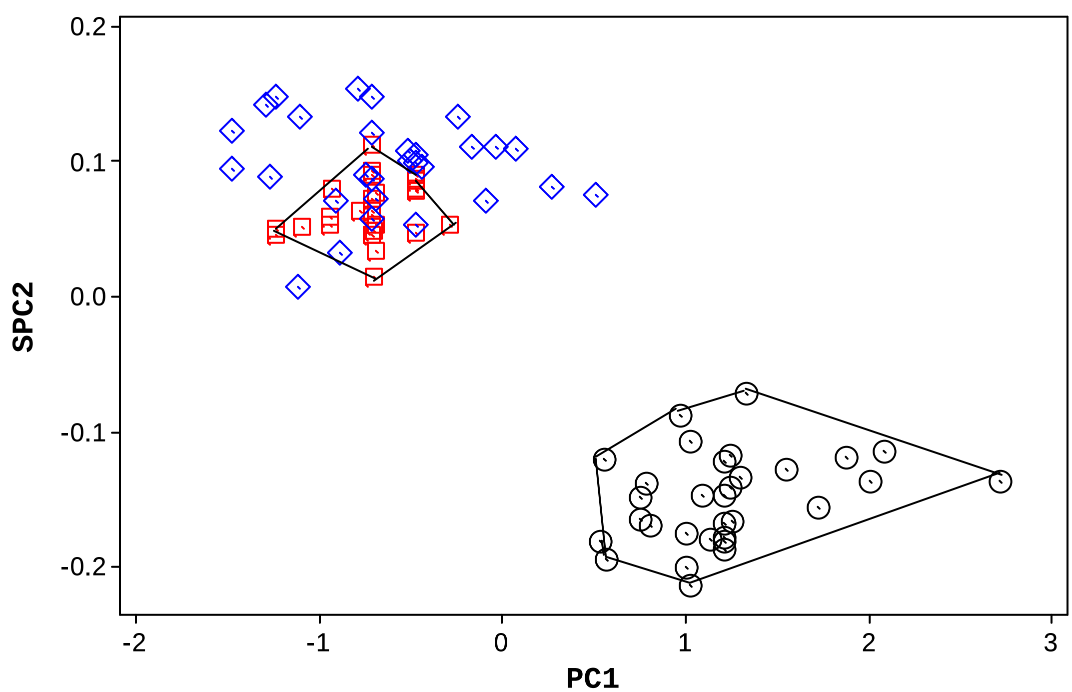

Figure 9. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles) and lagoons (diamonds).

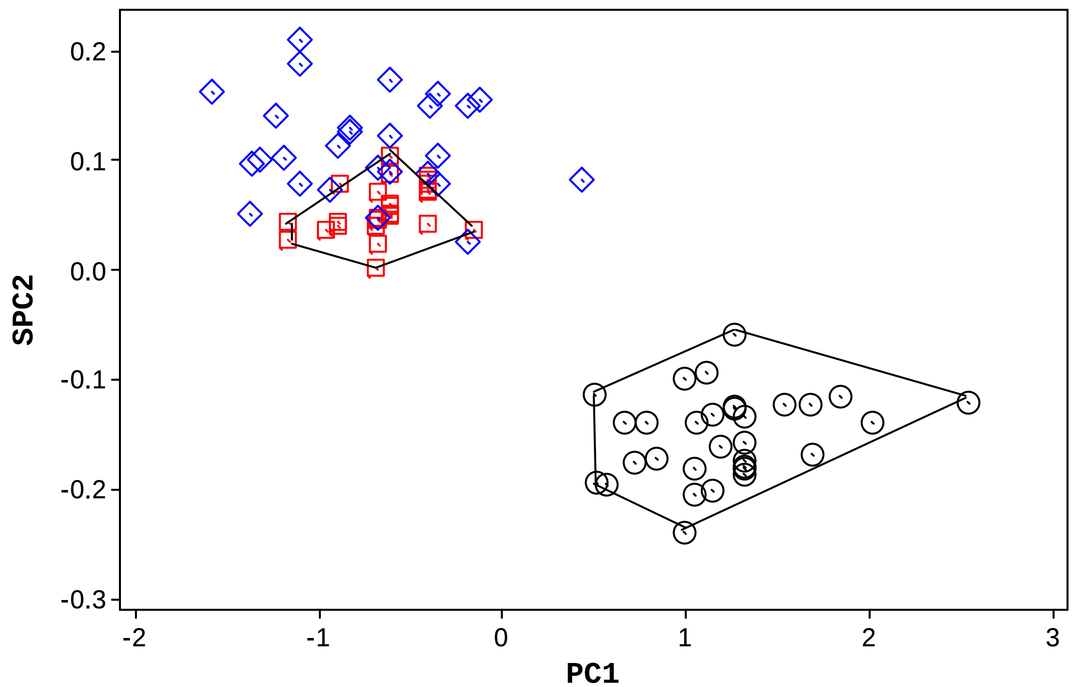

Figure 10. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles) and Thompson's Bay (diamonds).

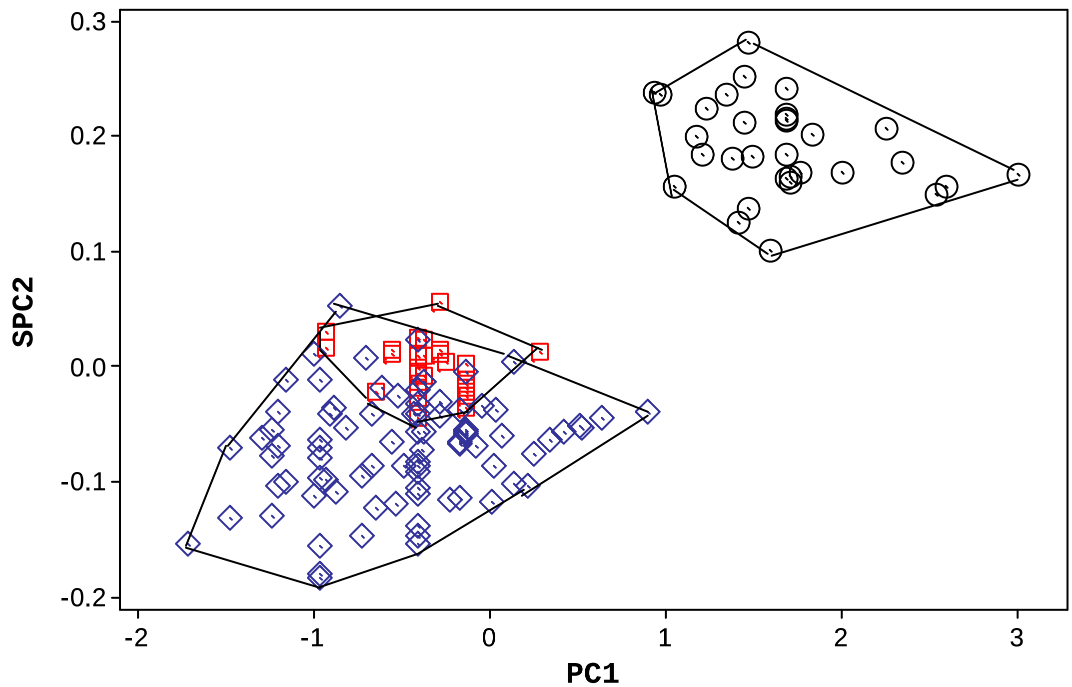

Figure 11. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), and Presque Isle Bay collections (diamonds).

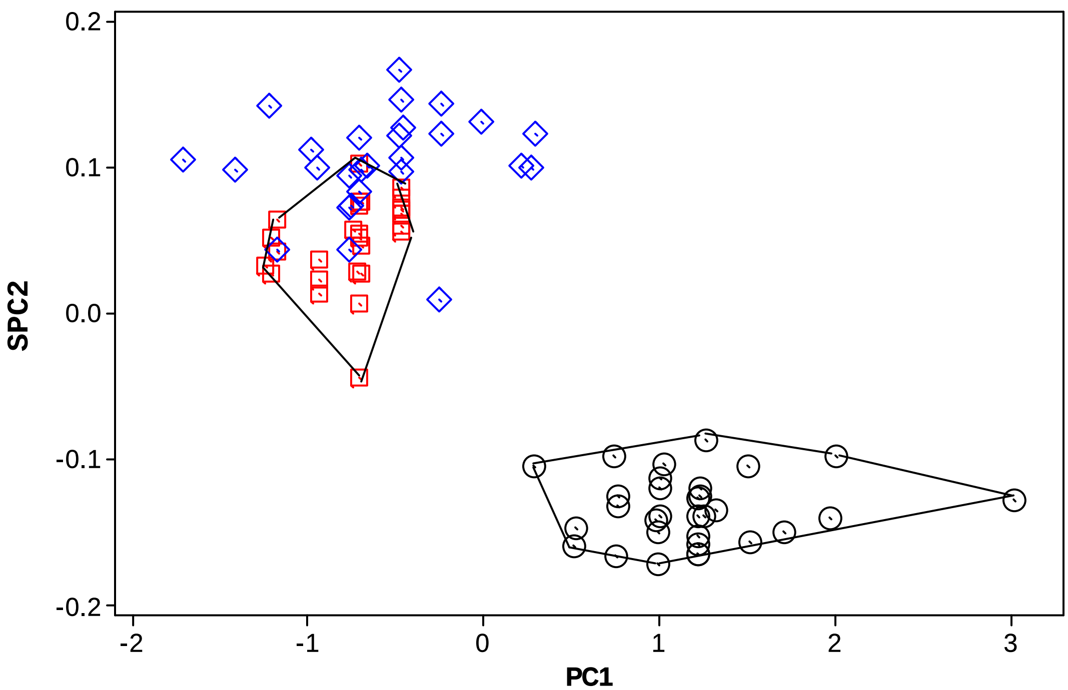

Figure 12. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), and Old Woman Creek, Ohio collections (diamonds).

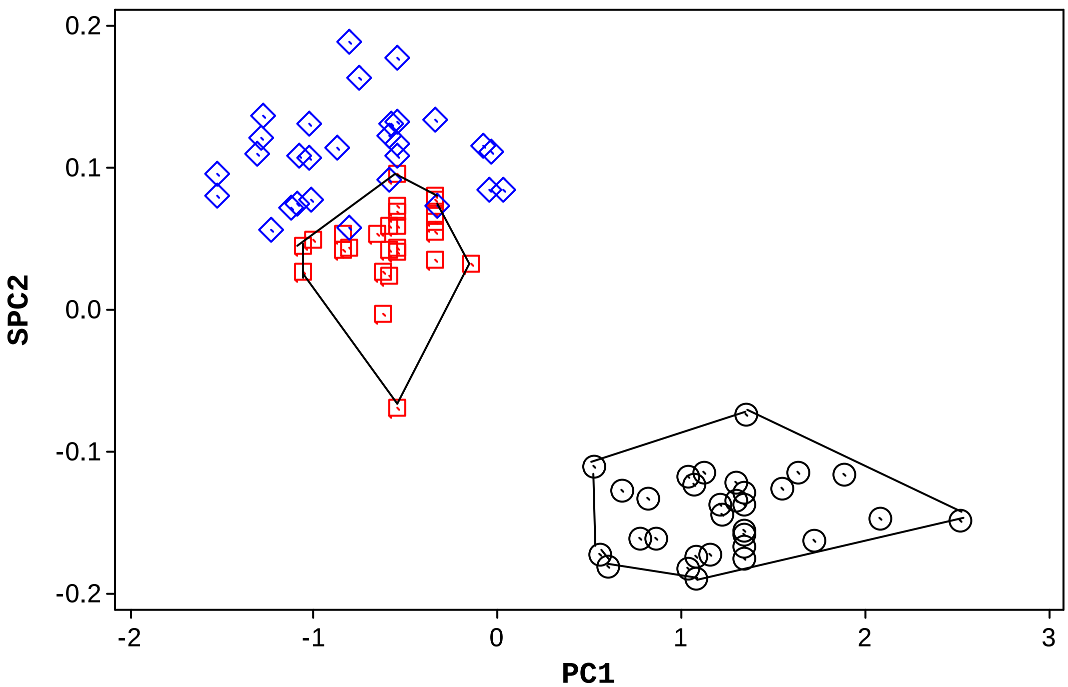

Figure 13. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), and Long Point Bay, Ontario, Canada collections (diamonds).

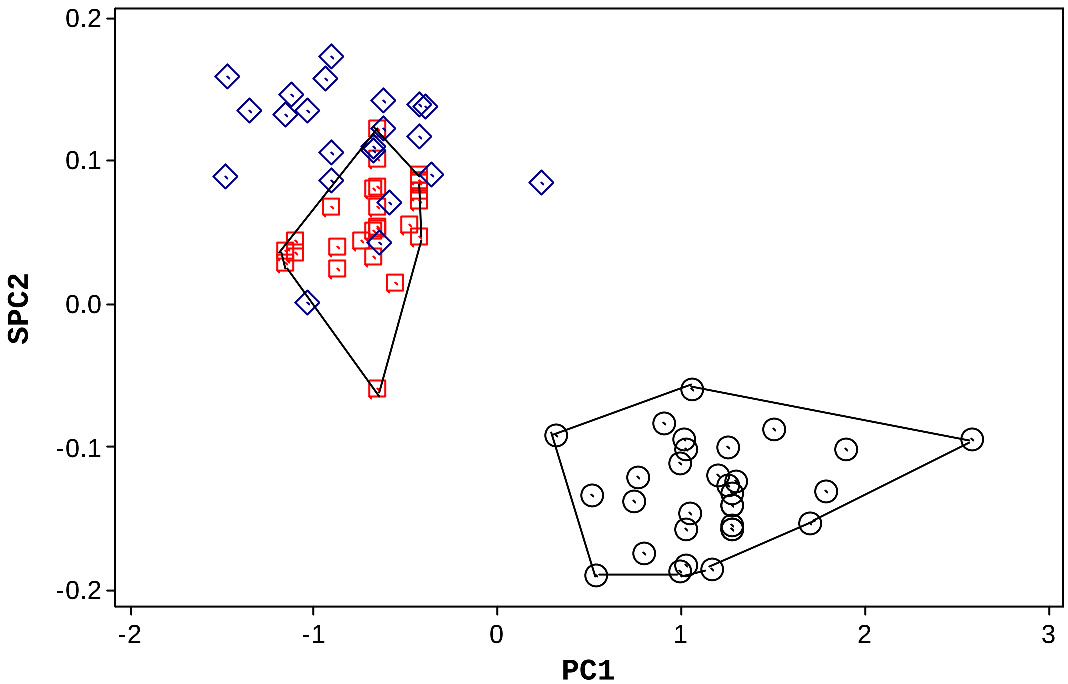

Figure 14. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), and Dunkirk Harbor, New York collections (diamonds).

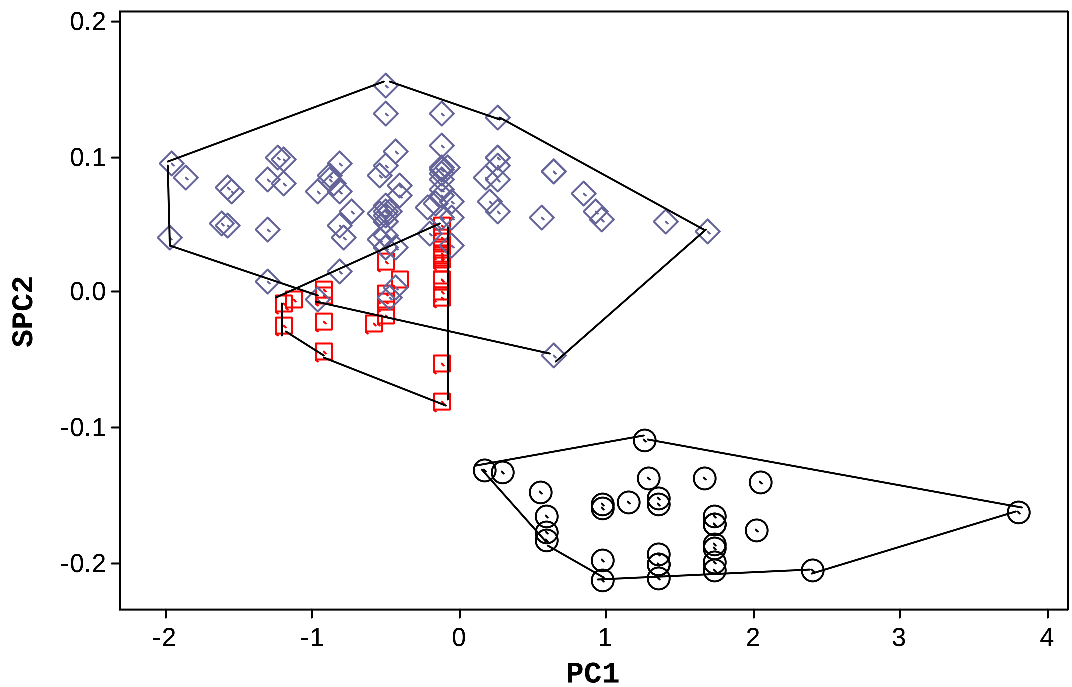

Figure 15. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), and Lake Erie collections (diamonds).

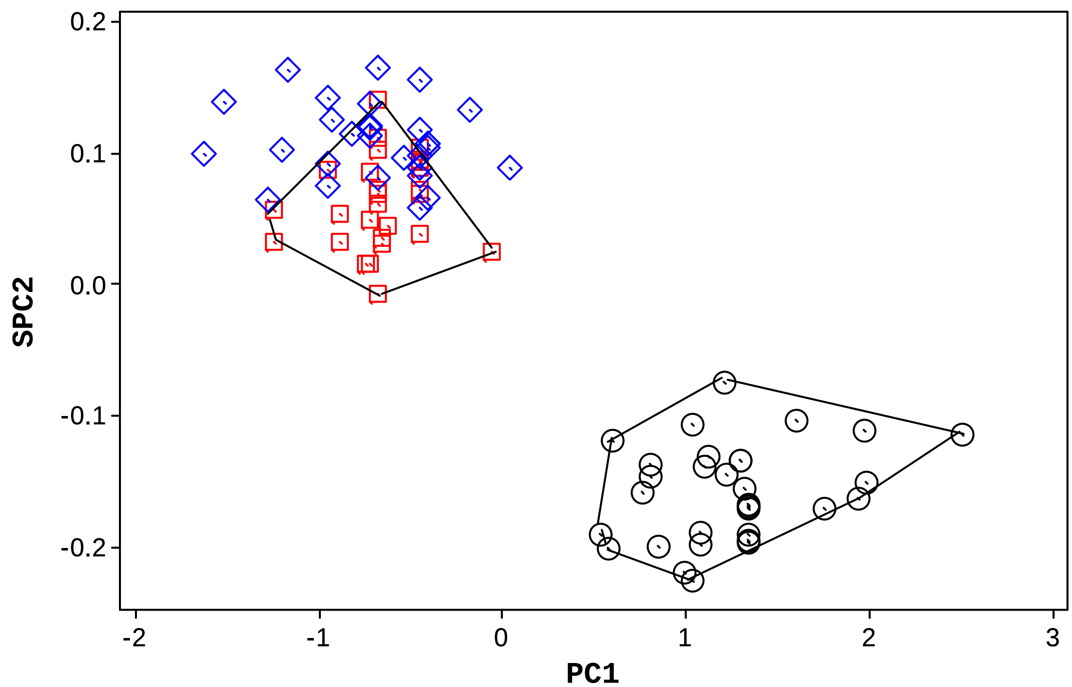

Figure 16. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), and Tamarack Lake (diamonds).

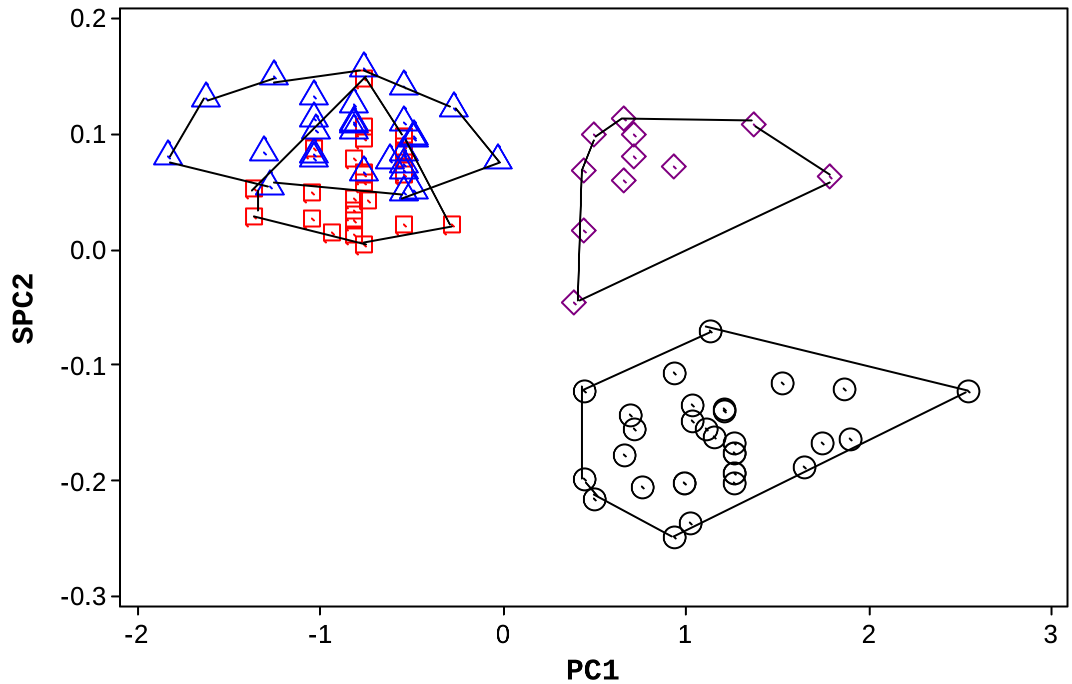

Figure 17. Plot of the first principal component (PC1) of the meristic data and the first sheared principal component (SPC2) of the mensural data from the reference specimens of *Ameiurus nebulosus* (squares), *A. melas* (circles), Tamarack Lake (triangles) and Wisconsin (diamonds).

Chapter 4

Microsatellites

4.1 Introduction to Microsatellites

Microsatellites were used to estimate the genetic structure of the two bullhead species and characterized the extent of potential hybrid populations. Microsatellites are extremely important markers for revealing genetic variation at population levels and between closely related species because of their high polymorphism, distribution across the genome, abundance, co-dominant inheritance pattern, and their short length, which facilitates genotyping by polymerase chain reaction (Lui et al. 1999a). Alleles are distinguished by size through electrophoresis. Prior studies have used microsatellites as genetic markers to estimate gene flow, effective population size, and inbreeding, as well as in parentage determination (Lui et al. 1999b, and Waldibieser and Bosworth 1997).

Microsatellites are tandem repeats of 1-6 nucleotides found at high frequency in the nuclear genomes of most taxa (Lai and Sun 2004). A microsatellite locus usually varies in length between 5 and 40 repeat units long, but longer strings of repeats are possible. Dinucleotide, trinucleotide, and tetranucleotide repeat motifs are the most common choices for molecular genetic studies. The differing numbers of repeats observed at polymorphic loci between two homologous chromosomes within an individual and between different individuals represent microsatellite alleles.

The DNA adjacent to a microsatellite locus is termed the flanking region. Because the sequence of flanking regions for a given locus are generally conserved (i.e. identical) across individuals of the same species and sometimes different species, a particular microsatellite locus can often be identified by its unique flanking sequences. Short stretches of synthesized DNA, called oligonucleotides or primers, can be designed to bind to specific flanking regions and guide the amplification of a microsatellite locus with the polymerase chain reaction (PCR). These ideal markers allows for the use of small tissue samples and can be amplified with PCR despite some DNA degradation (Selkoe and Toonen 2006).

Microsatellites DNA loci are highly unstable and mutate at high rates compared to other genetic markers (Goldstein and Pollock 1997). While the exact mechanism of mutation at such loci is still not well characterized at a molecular level, it is generally believed that the process and patterns of mutation at different loci may differ from locus to locus, depending on the motif as well as the size of alleles at each locus (Xu and Fu 2004). The instability of these DNA regions may result from DNA polymerase slippage as well as unequal recombination. During replication, dissociation, and subsequent reannealing of the DNA strands, one or more of the repeats is unpaired and forms a single stranded loop; a process called slippage. This can result in either the addition or deletion of a repeat unit, depending on whether the looped strand is located on the template or replicating strand. If this mistake is not corrected by the proof reading mechanism of DNA polymerase, it will remain as a mutation at that locus, and the alleles will differ in size by having different numbers of repeat units. It is not uncommon for an individual to have two different-sized microsatellite alleles between its two homologous chromosomes, making it a heterozygote for that locus. One strength of microsatellite markers is that they are co-dominant, such that both alleles of a heterozygote are visualized under normal conditions.

Experimental and theoretical studies indicate that for most microsatellite loci, mutations lead to stepwise changes of the repeat size of alleles through the rate of mutation leading to expansion may not be equal to that of contraction of allele size. The stepwise mutation model, originally proposed for the study of protein charge changes in a more generalized form may be more suitable for the study of most microsatellite loci (Xu and Fu 2004). The stepwise mutational model adds or subtracts one or more repeat units from the string of repeats at some constant rate to mimic the process of errors during DNA replication that generates mutations, creating a Gaussian-shaped allele frequency. However, non-stepwise mutation processes are also known to occur, including point mutation and recombination events such as unequal crossing over and gene conversion. While debate continues about the prevalence of non-stepwise mutations for microsatellites, the current consensus is that the frequency and effects are usually low, and the stepwise mutation appears to be the dominant force creating new alleles in the few model organisms studied to date (Selkoe and Toonen 2006).

When analyzing microsatellite data it is important to determine if microsatellite allele frequencies fall within Hardy-Weinberg equilibrium and do not violate the assumptions of random mating, no genetic drift, no mutation, no migration, and no natural selection. Non-random mating tends to reduce genetic variation. Random mating means that alleles as carried by the gametes come together strictly in proportion to their frequencies in the population as a whole. Situations where the random mating assumption does not hold include: inbreeding, geographic structures, assortative mating, rare allele advantages, and mating system effects (Graur and Li 2000). Random genetic drift removes genetic variation from the population at a rate inversely proportional to population size. Mutation is the process that produces a new allele that is different from the ancestral allele. Mutation restores genetic variation to a population by producing novel alleles. Mutation is difficult to measure or observe directly, and rates of mutation can vary between loci. Genetic migration (gene flow) is the permanent movement of genes from one population into another. Migration can restore genetic variation into isolated and differentiated populations or homogenize allele frequencies between populations when it occurs frequently (Graur and Li 2000). Selection is the differential survival and reproduction of phenotypes that are better suited to the environment or to obtaining mating success and is the evolutionary force responsible for adaptation to the environment. Microsatellites are not usually considered to be under positive or negative selection, since they are non-coding regions and different sized alleles are believed to be effectively neutral.

Deviations from Hardy-Weinberg equilibrium may also be caused by the presence of null alleles. One method to detect such deviations is to compare the expected levels of heterozygosity to the observed levels of heterozygosity of alleles at a locus within a population. A null allele is any allele at a locus that consistently fails to amplify to detected levels through PCR (Dakin and Avise 2004). When null alleles occur, any genotype observed as a homozygote may contain one observable allele and one null allele and the genotypes observed may therefore be scored as a homozygote when in effect it is a heterozygote. This can lead to observed heterozygosity that is lower than expected by

Hardy-Weinberg equilibrium. One may either choose to ignore the problem, drop the affected loci from consideration, or redesign and optimize the primers to eliminate null alleles (Dakin and Avise 2004). Although null alleles lead to underestimated heterozygosity within samples, it is a minor source of error in estimating heterozygosity excess (Dakin and Avise 2004). The occurrence of null alleles is widely acknowledged and many papers report the results of diagnostic tests for the presence of null alleles (Dakin & Avise 2004), but options for dealing with null alleles are limited.

The program GENEPOP (Raymond and Rousset 1995) was used to calculate observed (H_o) and expected (H_e) heterozygosity, linkage disequilibrium, and p-values for the exact Hardy-Weinberg test associated with H_o and Hardy-Weinberg equilibrium. The program ML-Relate was used to test for the presence and frequency of null alleles (Kalinowski and Taper 2006).

4.2 Genetic Characteristics

When selecting microsatellite loci for a hybridization study, it is often possible and desirable to identify specific loci that have alleles that do not overlap in size between pure reference populations of the two species under study. It is therefore possible to come up with a set of alleles that are unique to each species. It is possible to distinguish between the types of hybridization present in Presque Isle Bay and the Lake Erie sites because each type will leave a characteristic genetic signature. Because microsatellite loci with non-overlapping allele size ranges were selected for this study (Figures 18 – 22), each genotyped individual can be scored for the frequency and pattern of "*melas*" alleles and “*nebulosus*” alleles. If an individual is the F1 progeny of *A. nebulosus* and *A. melas*, all loci will be heterozygous; with one *nebulosus* allele and one *melas* allele, with the net frequency of 0.5 “*nebulosus*” alleles and 0.5 “*melas*” alleles.

If the F1 hybrids are breeding with themselves to produce F2 progeny, their offspring will be a random combination of the two species’ alleles, and therefore some loci may be homozygous for “*nebulosus*” alleles, while others will be homozygous for “*melas*” alleles, and still others will be heterozygous for the two species. These three allelic categories should be likely with 0.25 of the loci being all *nebulosu*s, 0.25 being all *melas*, and 0.50 being heterozygous for the two species. F2 individuals created by two F1 hybrids will have an overall frequency across all loci of 0.5, because they are a random recombination of F1 individuals that have overall frequency of 0.5 for their pooled alleles across all loci. F2 individuals are distinguished form F1 individuals by having some loci that are homozygous for one or both parental species, but an overall frequency of approximately 0.5

If the hybrids have backcrossed with the pure parental species, they will be skewed in the direction of the species in which they backcrossed. For example, if an F1 hybrid backcrossed with a pure *nebulosus*, approximately half of the loci will be homozygous for “*nebulosus*” alleles but a few loci will have “*melas*” alleles in the heterozygous state. Overall, a backcrossed individual will have 75% or more of its alleles from its parental species.

4.3 Polymorphism, Heterozygosity, and Hardy-Weinberg Equilibrium- Reference specimens

Nine microsatellite markers were considered for this study. Five of these nine original loci were used (Aneb16, Aneb37, Aneb61, Aneb63, and Aneb64) by screening the fluorescent fragments generated by PCR against the two pure populations analyzed with GENESCAN software (Applied Biosystems). According to the results from GENEPOP v. 3.4 (Raymond and Rousset 1995) none of the loci showed linkage disequilibrium. Observed heterozygosity for *Ameiurus nebulosus* ranged from 0.077 (locus Aneb64) to 0.778 (Aneb16 and Aneb37), with an average of 0.483. Hardy-Weinberg equilibrium was tested with GENEPOP v. 3.4 (Raymond and Rouset 1995) with departure from Hardy-Weinberg equilibrium in two loci (Aneb64 and Aneb63). Observed heterozygosity for *Ameiurus melas* ranged from 0.143 (locus Aneb61) to 0.793 (Aneb16), with an average of 0.495. Hardy-Weinberg equilibrium was tested with GENEPOP v. 3.4 (Raymond and Rouset 1995) with departure from Hardy-Weinberg equilibrium in one locus (Aneb64).

The loci were then analyzed for the presence of null alleles using the software program ML-Relate (Kalinowski and Taper 2006) and two loci (Aneb63 and Anb64) had an estimated frequency of null alleles in the population of over 0.10. Because the null alleles had a higher frequencies in *A. nebulosus* for both of these loci, a frequency of *A. melas* alleles may have been overestimated (Table 5). Aneb63 and Aneb64 were used for the scoring, and then omitted from data set for the rescoring of specimens for the Presque Isle Bay and Lake Erie populations.

4.4 Polymorphism, Heterozygosity, and genetic distances- Presque Isle Bay and Lake Erie Collections

All five microsatellite loci were polymorphic in the Presque Isle Bay and Lake Erie populations (Table 6 and 7). The Lake Erie populations had fewer alleles per locus than the Presque Isle Bay collections although it was not significant (one-tailed test, P = 0.82). Of the sixty-eight alleles detected in the Presque Isle Bay population, forty-six were present in the Lake Erie samples, with an additional fifteen alleles from Lake Erie not being found in the Presque Isle Bay samples. The distance estimates show a small level of genetic differentiation between populations at each locus, but high levels within the populations in comparisons of F_{IS} and F_{ST} values. F_{IS} estimates ranged from 0.2198 to 0.2668 for the Presque Isle Bay specimens (Table 9) and 0.155 to 0.303 for the Lake Erie and Presque Isle Bay specimens (Table 10). Pair-wise F_{ST} estimates for Presque Isle Bay ranged from 0.0064 to 0.0319 and 0.0020 to 0.0199 for the Lake Erie and Presque Isle Bay collections.

4.5 Genotypes – Presque Isle Bay and Lake Erie Collections

A genetic hybrid index score was developed by assigning a value of 1 for each *Ameiurus nebulosus* allele and 0 for each *A. melas* allele and dividing by the total number of alleles for the specimen. This score was used to characterize the individual as *A. nebulosus* or as having some genetic material from *A. melas*. Before adjustment for suggested presence of null alleles, the data from Brown Bullheads collected in Presque

Isle Bay shows over 40 percent of the bullheads sampled from Sara's Cove and Thompson's Bay contain some genetic material from Black Bullheads. Twenty-seven percent of the Brown Bullheads in the lagoons had Black Bullhead genetic material in their DNA (Figure 23). These numbers were reduced to 22 percent of the bullheads in Thompson's Bay and 10 percent of the bullheads in the lagoons having some Black Bullhead alleles after adjustment of null alleles. Sara's Cove still had over 40 percent of the bullheads sampled containing some Black Bullhead alleles (Figure 24).

Twenty-nine percent of the Brown Bullhead specimens from Old Woman Creek, Ohio contain some black bullhead alleles, while Long Point Bay, Ontario, Canada and Dunkirk Harbor, New York had 32 and 38 percent respectively before adjustment of null alleles (Figure 25). After adjustment, Old Woman Creek had 25 percent and Long Point Bay and Dunkirk both had 29 percent (Figure 26).

With the adjustment for the suggestion for the presence of null alleles, the multi-locus nuclear genotypes suggest the presence of advanced-generation hybrids or backcrosses between *A. nebulosus* and *A. melas* in Presque Isle Bay and Lake Erie.

4.6 Polymorphism, Heterozygosity, and Genetic distances- Tamarack Lake and Wisconsin Collections

All five loci were polymorphic for the Tamarack Lake Brown Bullheads except Aneb61, which only had one allele. All loci were polymorphic for the Wisconsin Black Bullhead specimens however Loci Aneb61 and Aneb63 had only two alleles (Table 7). Tamarack Lake and Wisconsin had fewer alleles but also had fewer specimens in the collection compared to Presque Isle Bay and Lake Erie collections. Observed heterozygosity ranged from 0.1111 (Aneb63 and Aneb64) to 0.8333 (Aneb16) in Tamarack Lake, and 0.1818 (Aneb61 and Aneb63) to 1.0 (Aneb37) in the Wisconsin specimens. The estimates of F_{IS} for Tamarack Lake ranges between -0.027 and +0.71 (Weir and Cockerham 1984). The value of F_{IS} for Wisconsin ranges between -0.202 and +0.608 (Weir and Cockerham 1984). Negative F_{IS} values indicate heterozygote excess (outbreeding) and positive values indicate heterozygote deficiency (inbreeding) compared with Hardy-Weinberg equilibrium expectations.

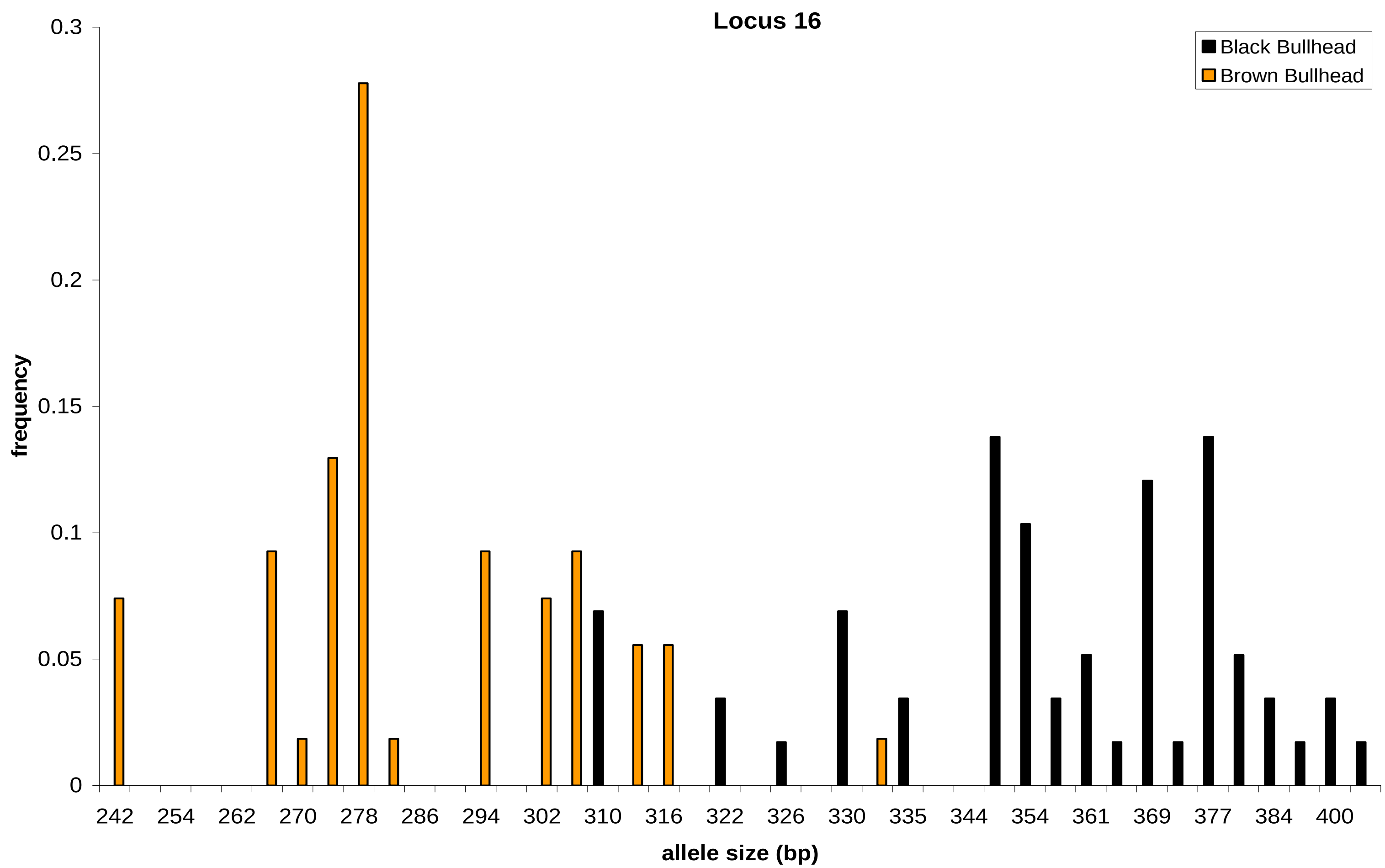

Figure 18. Allele frequencies at microsatellite locus 16 from samples of the reference specimens

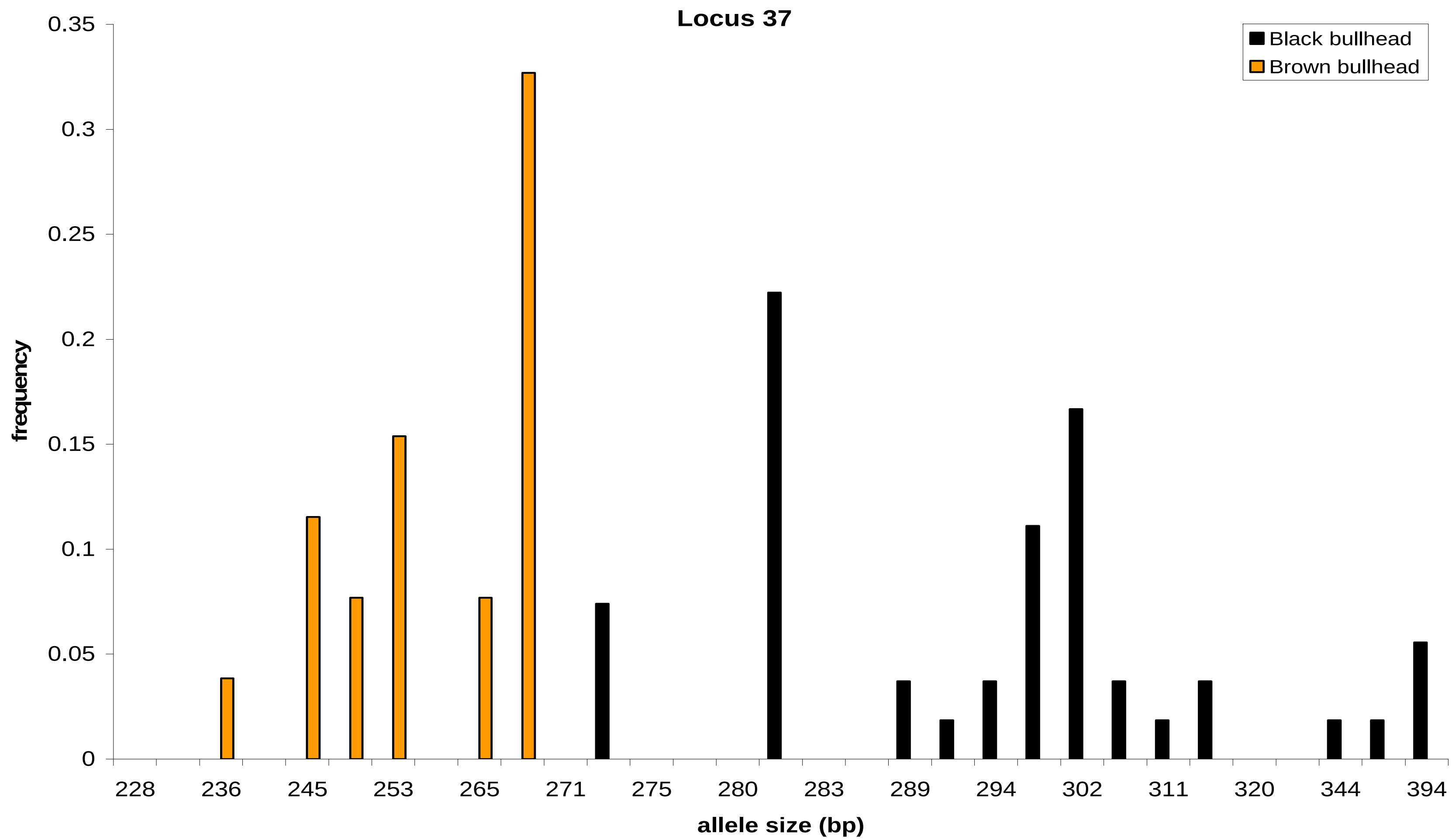

Figure 19. Allele frequencies at microsatellite locus 37 from samples of the reference specimens

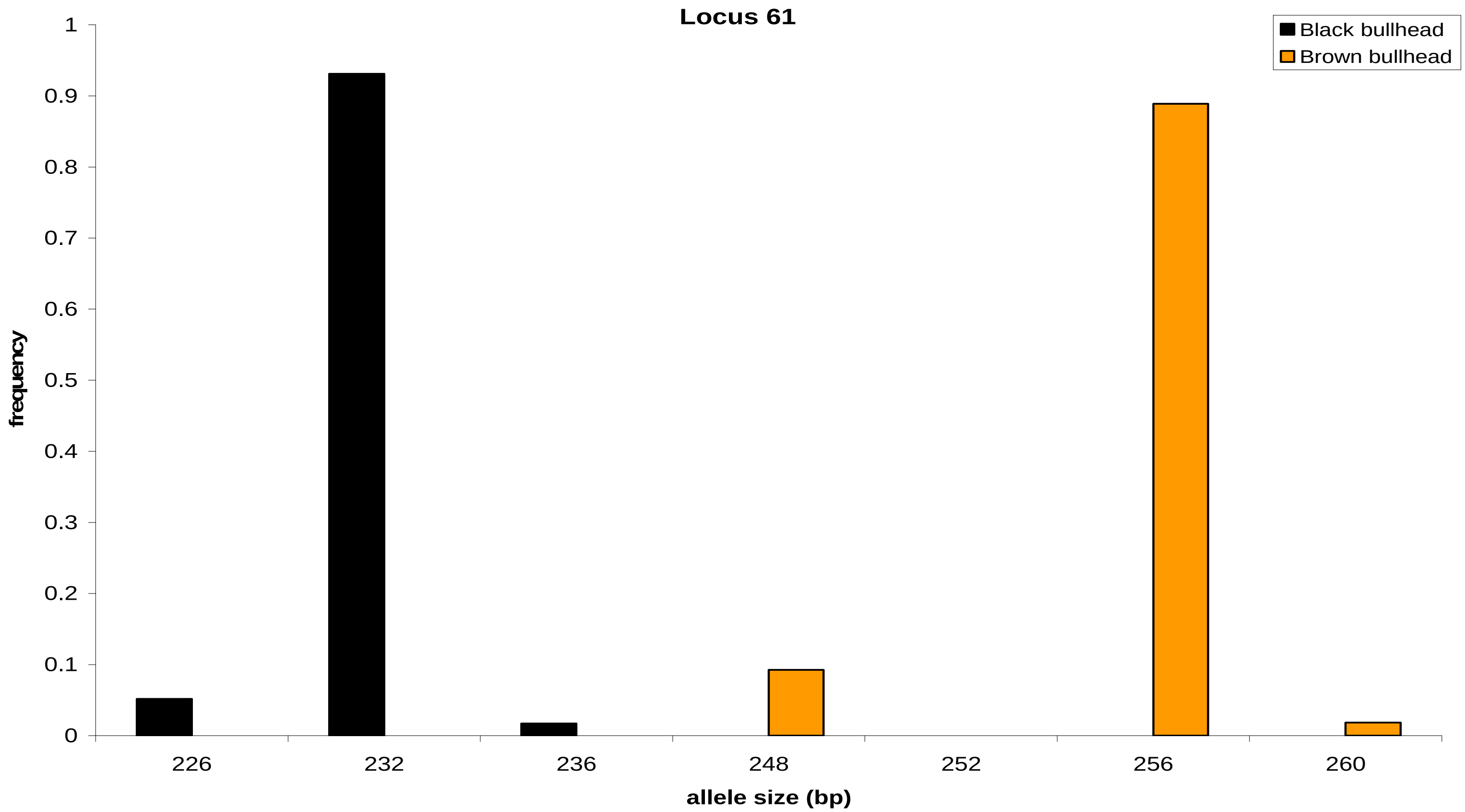

Figure 20. Allele frequencies at microsatellite locus 61 from samples of the reference specimens

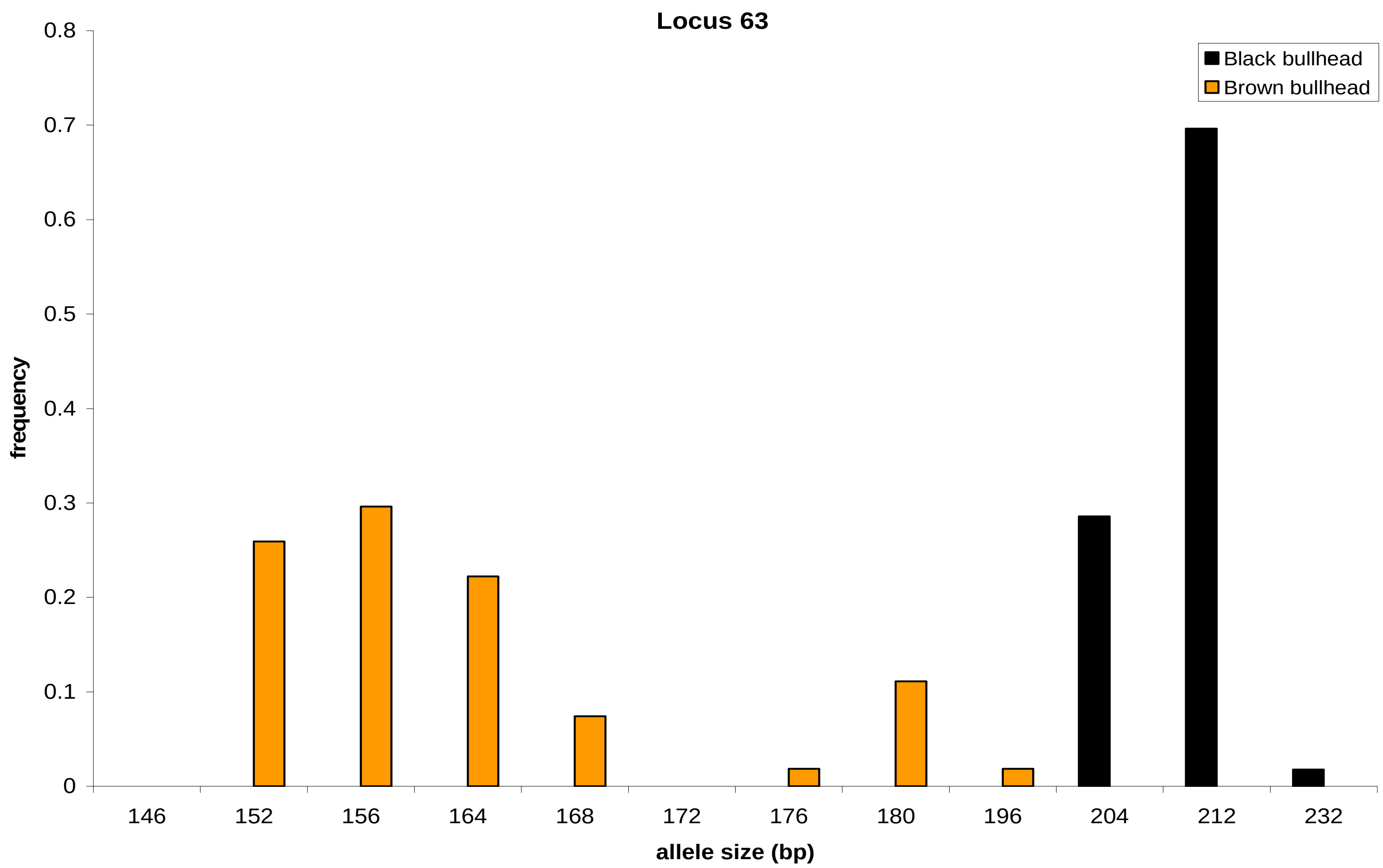

Figure 21. Allele frequencies at microsatellite locus 63 from samples of the reference specimens

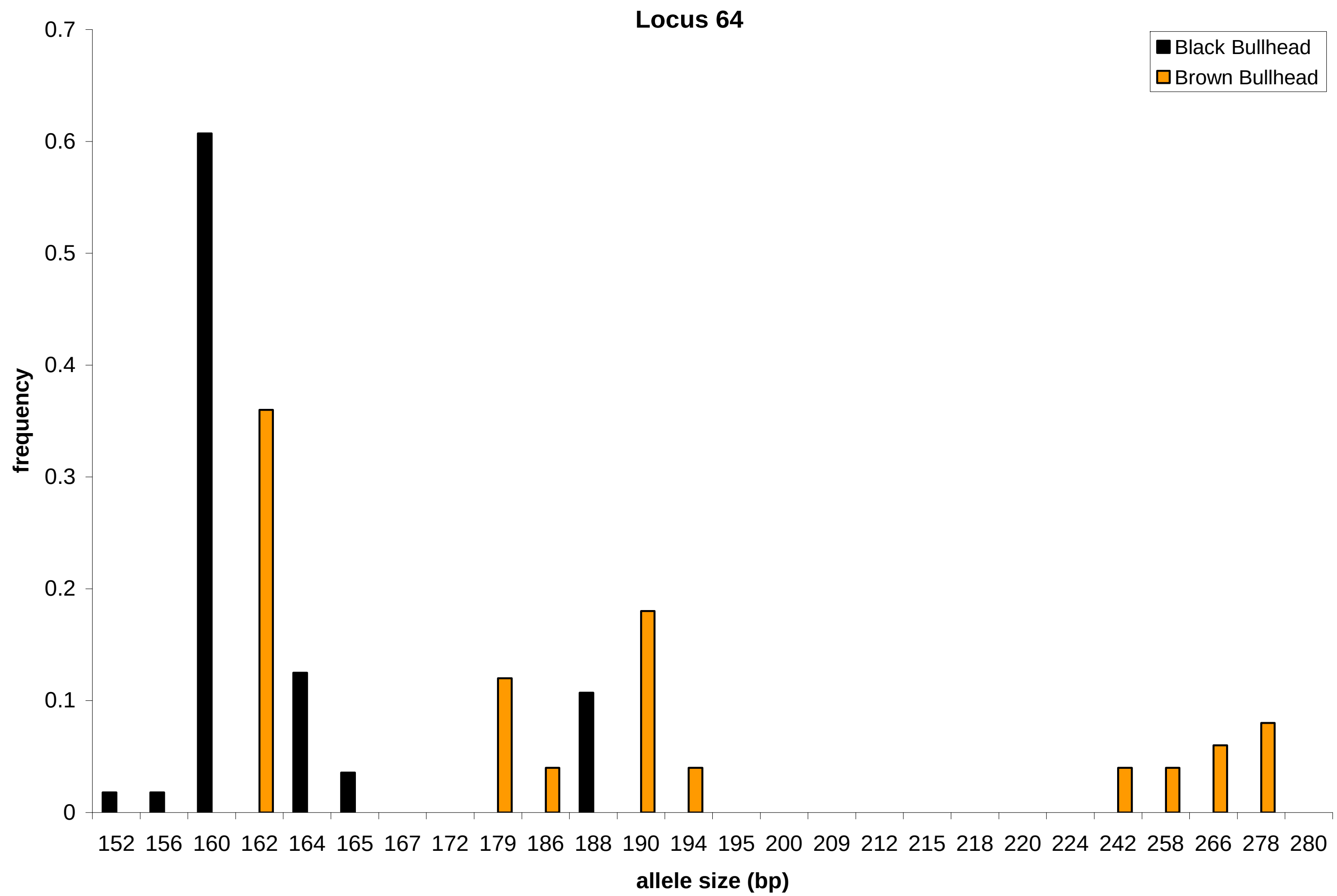

Figure 22. Allele frequencies at microsatellite locus 64 from samples of the Reference specimens.

Table 5. Multilocus variation in the reference *Ameiurus* populations. Numbers of specimens (N), number of alleles per locus (A), size of the allele, observed heterozygosity (Ho), heterozygosity as expected under Hardy-Weinberg equilibrium (He), an unbiased estimate of the P-value of the probability test for Hardy-Weinberg, as described by Raymond and Rousset (1995), and percentage of null alleles per locus (pNull).

A. nebulosus							
Locus	**N**	**A**	**size**	**Ho**	**He**	**p-value**	**pNull**
Aneb16	27	12	242-232	0.778	0.88	0.0771	0.0449
Aneb37	27	8	236-285	0.778	0.803	1	0.0368
Aneb61	27	3	248-260	0.222	0.204	0.004	0
Aneb63	25	7	152-196	0.56	0.866	0	0.1396
Aneb64	26	8	162-278	0.077	0.79	0.0132	0.4572
A. melas							
Locus	**N**	**A**	**size**	**Ho**	**He**	**p-value**	**pNull**
Aneb16	29	18	310-408	0.793	0.9304	0.0006	0.0484
Aneb37	29	15	273-394	0.69	0.839	1	0.0758
Aneb61	28	3	226-236	0.143	0.137	0.3017	0
Aneb63	27	3	204-232	0.518	0.457	0	0
Aneb64	27	6	152-196	0.333	0.0123	0.0082	0.2118

Table 6. Multilocus variation in the Presque Isle Bay *Ameiurus* populations. Numbers of specimens (N), number of alleles per locus (A), size of the allele, observed heterozygosity (Ho), heterozygosity as expected under Hardy-Weinberg equilibrium (He), and an unbiased estimate of the P-value of the probability test for Hardy-Weinberg, as described by Raymond and Rousset (1995).

Presque Isle Bay - Sara's Cove

Locus	N	A	size	Ho	He	p-value
Aneb16	27	19	242-381	1	0.9426	0.9991
Aneb37	27	13	185-283	0.814	0.919	0.0203
Aneb61	27	5	215-256	0.074	0.21	0.0012
Aneb63	27	4	152-168	0.185	0.395	0.0028
Aneb64	27	4	150-164	0.074	0.21	0.0006

Presque Isle Bay - Lagoons

Locus	N	A	size	Ho	He	p-value
Aneb16	27	18	242-230	0.963	1	0.1613
Aneb37	27	13	185-283	0.814	0.971	0.0567
Aneb61	27	2	215-256	0.111	0.177	0.0364
Aneb63	25	4	152-180	0.44	0.385	1
Aneb64	26	8	160-248	0.115	0.633	0

Presque Isle Bay - Thompson's bay

Locus	N	A	size	Ho	He	p-value
Aneb16	30	21	202-344	1	1	0
Aneb37	30	12	236-281	0.833	0.9241	0.0845
Aneb61	29	3	232-256	0.103	0.134	0.6233
Aneb63	28	6	152-180	0.357	0.359	0
Aneb64	30	9	160-224	0.2	0.7256	0.0408

Table 7. Multilocus variation in the Lake Erie *Ameiurus* populations. Numbers of specimens (N), number of alleles per locus (A), size of the allele, observed heterozygosity (Ho), heterozygosity as expected under Hardy-Weinberg equilibrium (He), and an unbiased estimate of the P-value of the probability test for Hardy-Weinberg, as described by Raymond and Rousset (1995).

Old Woman's Creek, Ohio

Locus	N	A	size	Ho	He	p-value
Aneb16	28	17	242-374	0.8925	0.935	0.6488
Aneb37	28	10	228-281	0.714	0.8471	0.1341
Aneb61	28	2	252-256	0.286	0.249	1
Aneb63	28	5	146-172	0.286	0.468	0.0025
Aneb64	27	3	162-167	0.222	0.326	0.1341

Long Point, Ontatio, Canada

Locus	N	A	size	Ho	He	p-value
Aneb16	26	17	250-344	0.923	0.923	0.3862
Aneb37	28	9	241-285	0.785	0.785	0.696
Aneb61	28	2	252-256	0.143	0.232	0.0039
Aneb63	28	4	152-172	0.429	0.544	0.442
Aneb64	26	6	162-224	0.115	0.379	0

Dunkirk Harbor, New York

Locus	N	A	size	Ho	He	p-value
Aneb16	21	17	242-356	0.715	0.928	0.0005
Aneb37	21	11	233-281	0.81	0.897	0.2603
Aneb61	21	2	252-256	0.048	0.048	
Aneb63	16	3	152-164	0.687	0.59	0.6165
Aneb64	21	8	160-280	0.048	0.605	0

Table 8. Multilocus variation in the Tamarack Lake and Wisconsin *Ameiurus* populations. Numbers of specimens (N), number of alleles per locus (A), size of the allele, observed heterozygosity (Ho), heterozygosity as expected under Hardy-Weinberg equilibrium (He), and an unbiased estimate of the P-value of the probability test for Hardy-Weinberg, as described by Raymond and Rousset (1995).

Tamarack Lake - *A. nebulosus*

Locus	**N**	**A**	**size**	**Ho**	**He**	**p-value**
Aneb16	27	13	202-316	0.8333	0.8986	0.0006
Aneb37	27	9	241-281	0.6667	0.6495	0.2268
Aneb61	27	1	256	0	0	na
Aneb63	27	4	152-176	0.1111	0.1338	0.0181
Aneb64	27	6	162-280	0.1111	0.4797	0

Wisconsin - *A. melas*

Locus	**N**	**A**	**size**	**Ho**	**He**	**p-value**
Aneb16	12	12	296-392	0.909	0.935	0.6939
Aneb37	12	8	277-315	1	0.8398	0.3383
Aneb61	12	2	232-234	0.1818	0.4502	0.0096
Aneb63	12	2	204-212	0.1818	0.3117	0.2767
Aneb64	12	3	160-196	0.4545	0.3796	1

Table 9. Average F_{IS} and F_{ST} values and the number of migrants per generation (N_M) for Presque Isle Bay specimens.

	Sara's Cove	Thompson's Bay	Lagoons
Sara's Cove	$F_{IS} = 0.2668$	$N_M = 7.59$	$N_M = 15.\ 78$
Thompson's Bay	$F_{ST} = 0.0319$	$F_{IS} = 0.2198$	$N_M = 38.81$
Lagoons	$F_{ST} = 0.0156$	$F_{ST} = 0.0064$	$F_{IS} = 0.2576$

Table 10. Average F_{IS} and F_{ST} values and the number of migrants per generation (N_M) for Lake Erie and combined Presque Isle Bay specimens.

	Old Woman's Creek, Ohio	Long Point Bay, Ontario	Dunkirk Harbor, NY	Presque Isle Bay
Old Woman's Creek, Ohio	$F_{IS} = 0.155$	$N_M = 124.75$	$N_M = 12.31$	$N_M = 17.60$
Long Point Bay, Ontario	$F_{ST} = 0.0020$	$F_{IS} = 0.2606$	$N_M = 14.03$	$N_M = 13.34$
Dunkirk Harbor, NY	$F_{ST} = 0.0199$	$F_{ST} = 0.0175$	$F_{IS} = 0.2723$	$N_M = 73.28$
Presque Isle Bay	$F_{ST} = 0.0154$	$F_{ST} = 0.0184$	$F_{ST} = 0.0034$	$F_{IS} = 0.303$

Figure 23. Percentage of bullheads possessing all *A. nebulosus* alleles (yellow) and percentage of bullheads with some *A. melas* alleles (black) for Presque Isle Bay; 1. Sara's Cove- 52% have all Brown bullhead alleles, 2. lagoons– 73% have all Brown Bullhead alleles, and 3. Thompson's Bay- 59% have all Brown Bullhead alleles.

Figure 24. Percentage of bullheads possessing all *A. nebulosus* alleles (orange) and percentage of bullheads with some *A. melas* alleles (grey) for Presque Isle Bay adjusted for the suggested presence of null alleles; 1. Sara's Cove- 57% have all Brown bullhead alleles, 2. lagoons– 90% have all Brown Bullhead alleles, and 3. Thompson's Bay- 78% have all Brown Bullhead alleles.

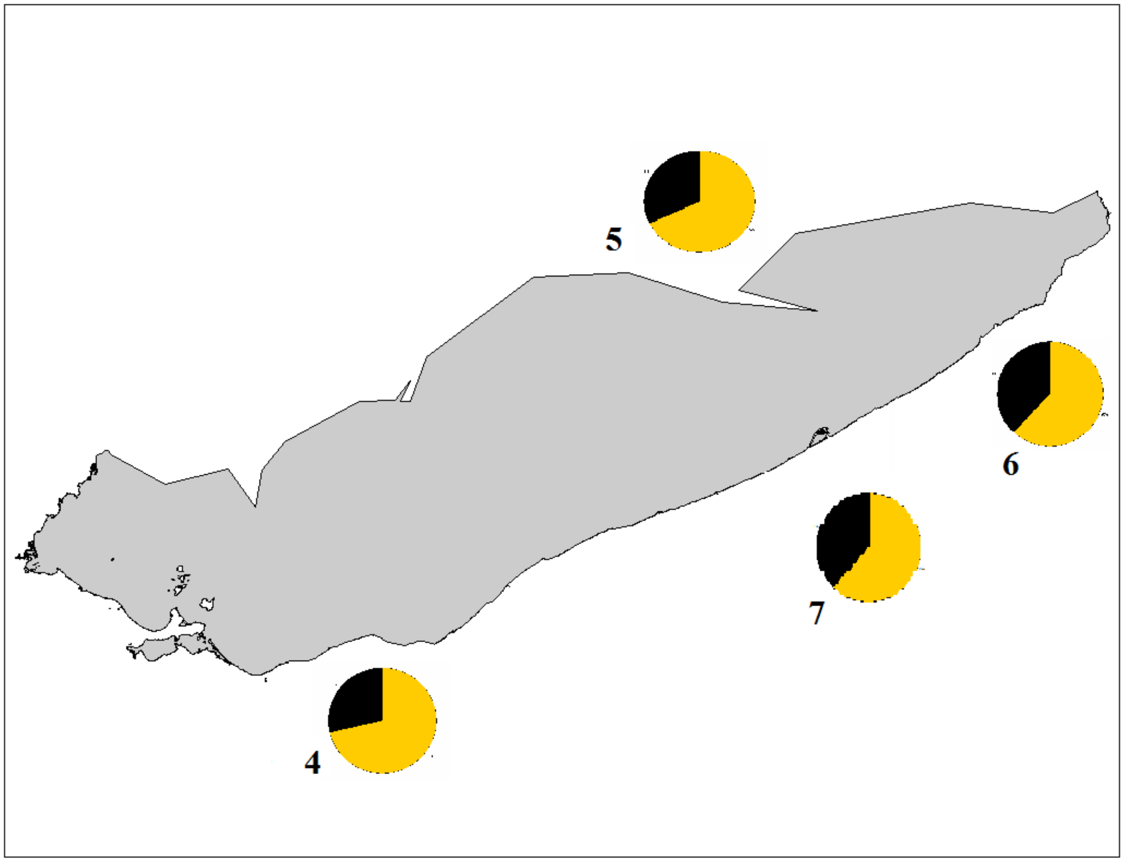

Figure 25: percentage of bullheads possessing all *A. nebulosus* alleles (yellow) and percentage of bullheads with some *A. melas* alleles (black) for the Lake Erie collections; 4. Old Woman Creek Ohio- 71% have all Brown Bullhead alleles, 5. Long Point Bay, Ontario - 68% have all Brown Bullhead alleles, 6. Dunkirk Harbor, New York- 62% has all Brown Bullhead alleles, and 7. Presque Isle Bay, Pennsylvania- 62% has all Brown Bullhead alleles.

Figure 26: percentage of bullheads possessing all *A. nebulosus* alleles (orange) and percentage of bullheads with some *A. melas* alleles (grey) for the Lake Erie collections after adjustment for the suggestion of null alleles; 4. Old Woman Creek Ohio- 75% have all Brown Bullhead alleles, 5. Long Point Bay, Ontario - 71% have all Brown Bullhead alleles, 6. Dunkirk Harbor, New York- 71% has all Brown Bullhead alleles, and 7. Presque Isle Bay, Pennsylvania- 75% has all Brown Bullhead alleles.

Chapter 5

Discussion

Hybridization among freshwater fishes is a common occurrence (Hubbs 1955), but is only known in a few Ictaluridae including *Noturus gyrinus* x *Noturus miurus* hybrid madtoms (Menzel and Raney 1973) and *Ameiurus nebulosus* x *Ameiurus melas* hybrid bullheads (Trautman 1981). The term hybridization is often difficult to define. Hybridization is usually employed in a broad sense to include crosses between genetically differentiated forms regardless of their current taxonomic status. Introgression refers to gene movement between species mediated by hybridization and backcrossing (Avise 2004). The occurrence and frequency of hybridization is related to environmental settings and reproductive ways of the parental species (Jenkins and Burkhead 1994). Hybridization under natural conditions is often associated with crowding of spawning fishes, and tends be facilitated when one species is rare and the other abundant (Stauffer et al. 1997). This is the likely situation for the endangered Black Bullhead and abundant Brown Bullhead in Presque Isle Bay and eastern portions of Lake Erie. Hybridization is also associated with habitat disturbances whereby two or more species may be forced into atypically close proximity during breeding thus increasing the chances of mismating. Turbidity may reduce the ability of a fish to visually discriminate other species form conspecific mates, and hybridization can occur with the introduction of non-native fishes (Jenkins and Burkhead 1994). Hybridization can also occur where there are environmental stresses. If the PCB loads in Presque Isle Bay initiated hybridization and hybrids were less vulnerable for tumor formation, this could result in positive feedback system. Hybrid vigor may allow for the hybrids to reach maturity faster than the putative parents and breed. Those offspring again having traits superior to the putative species should then reach maturity faster and breed. And as this cycle continues, would be more hybrid individuals maturing faster without the presence of tumors.

5.1 Principal Component Analysis

Historically, identification of naturally occurring hybrids has depended on intermediacy of character states between the putative hybrid and the two parental forms (Stauffer et al. 1996). Principal component analysis was used to delimit naturally occurring hybrids and not the use of hybrid-description techniques that require *a priori* identification of the hybrid. Principal component analysis allows us to consider multivariate variability, since the components are composed of all of the initial characters and are in the directions of greatest variance within the data matrix. Hubbs (1955) reported that the vast majority of hybrids possess character indices intermediate of their parental forms. Characters of the hybrids may also lie outside the parental forms. The identification of F1 hybrids is likely using PCA, but the identification of F2 or backcrosses does not appear possible (Neff and Smith 1978), especially when backcrossing can produce an infinite combination of morphological traits.

The morphological graphs from this study do not form distinct intermediate clusters for hybrids relative to the putative species, but still identified a few intermediate hybrid specimens in the Lagoons and Thompson's Bay collections in Presque Isle Bay, and also in the Old Woman Creek, Ohio and Dunkirk Harbor, New York samples. The morphological graphs also suggest heterosis in the Sara's Cove collection. Heterosis, also called hybrid vigor, is the increase in such characteristics as size, growth rate, fertility, and yield of a hybrid organism over those of its parents. It may also occur that a hybrid inherits such different traits from their parents that make them unfit for survival or quite possible, more susceptible to tumors and external abnormalities.

Brown Bullheads and Black Bullheads are phenotypically similar, and can be quite challenging to delimit in sympatric populations without having both representative species present. The importance of the gill raker counts in distinguishing between the two species of *Ameiurus* suggests an ecological separation of the two species in their mode of feeding or in some other particle-size-related aspect of their adaptation to their environment. As emphasized by the loadings of the principal components distinguishing the species, both of the gill raker counts are involved, as would be expected if there were a functional distinction involving the entire branchial basket. This example also suggests that principal components analysis is of use in identifying or confirming functional complexes of characters through the patterns of character loadings on the components.

5.2 Microsatellites

Molecular markers can be of great utility in diagnosing closely related species, even where morphological or other traditional markers fail or are ambiguous. (Avise 2004).

Microsatellites data indicates that gene flow has occurred in Presque Isle Bay and throughout Lake Erie in similar trends. All the intermediate specimens were identified as being backcrossed to the Brown Bullhead. Interspecific hybridization can be costly to the participants, typically yielding progeny with diminished fitness and resulting in hybrid zones that act as genetic sinks. Sometimes fitness of hybrid organisms surpasses those of their putative parents (Avise 2004). Some hybrid populations might also be the sources of adaptive evolution and lineage diversification by possessing novel recombinant genotypes (Avise 2004).

There seems to be small level of genetic differentiation between the sampled populations at each locus, but high levels within populations across Lake Erie. Inbreeding within a subpopulation is caused by the nonrandom mating of the members of that subpopulation, in that mating occurs more often than by chance alone, between closely related individuals. As closely related individuals will contain a large proportion of the same alleles due to common descent, their offspring will have a higher level of homozygosity, and conversely, a lower level of heterozygosity then expected. Positive F_{IS} values indicate heterozygote deficiency (inbreeding) compared with Hardy-Weinberg equilibrium expectations. A consequence known as Wahlunds' effect shows that as allele frequencies in two subpopulations deviate, the average observed heterozygosity in those populations will always be less than that expected from the pooled allele frequencies. Deviations from Hardy-Weinberg equilibrium may also be caused by the presence of null alleles. One method to detect such deviations is to compare the expected levels of heterozygosity to the observed levels of heterozygosity of alleles at a locus within a population. Although null alleles lead to underestimated heterozygosity within samples, it is a minor source of error in estimating heterozygosity excess (Dakin and Avise 2004).

The high levels of F_{IS} (proportion of variation within a population) combined with the fact that backcrossed hybrids are present in all syntopic populations and significantly higher observed heterozygosities than expected from Hardy Weinberg Equilibrium found in all Lake Erie populations are all results of extensive hybridization between the two species.

Chapter 6

Conclusion

The question addressed in this study was to determine if hybridization has occurred between *Ameiurus nebulosus* and *Ameiurus melas* in Presque Isle Bay, Lake Erie. Results of the morphological and meristic analysis using principal component analysis indicate the majority of Brown Bullheads from Presque Isle Bay group with the reference Brown Bullhead population morphologically and not with the reference Black Bullheads. Collections from the Lagoons and Thompson's Bay each include an individual which maybe a hybrid, but what is likely being collected as a Brown Bullhead for the tumor studies in Presque Isle Bay, is morphologically a Brown Bullhead.

Genetically, over half of the Bullheads sampled were identified as having all *Ameiurus nebulosus* alleles, but multi-locus nuclear genotypes suggest the presence of extensive backcrossing between *Ameiurus nebulosus* and *Ameiurus melas* in Presque Isle Bay. The hybrid bullheads have also been reported from the western portion of Lake Erie prior to 1950 (Trautmann 1981) and were present in Presque Isle Bay in 2003 (Hunnicutt et al. 2005).

Presque Isle Bay has been under intensive environmental study due to high rates of liver and skin tumors in Brown Bullhead residing in this bay. It is not possible to state if *Ameiurus nebulosus* x *Ameiurus melas* hybrids are more susceptible to external abnormalities or more resistant to external abnormalities from this study. While hybrid specimens have higher external abnormalities (Figure 27 and 28), there is not a difference between the pure Brown Bullhead and hybrid specimen collected in Presque Isle Bay regarding tumor and deformities rates (p-value = 0.663). The presence of tumors and deformities are related to age of the fish and contaminants in the sediments of the lakes (Pyron et al. 2001). High incidents of external abnormalities on Brown Bullheads and specimens backcrossed to Brown Bullheads indicate their sensitivity to contaminated sediment exposure. This sensitivity may be attributable to lack of scales and exposed skin, metabolic differences that result in formation of carcinogenic PAH metabolites, or extensive contact with contaminated sediments because of habitat requirements (Smith et al. 1994). Brown Bullheads are tolerant of very low dissolved oxygen concentrations and are able to feed on items correlated with these conditions. Brown Bullheads are also known to become sluggish and cease feeding in the late fall and bury themselves in soft, silt, mud and leafy material along the shore (Becker 1983). Long exposure to contaminated sediments may best explain the high incident of external abnormalities on Brown Bullheads from Presque Isle Bay, but further investigation into the role of hybridization and external abnormalities should be considered.

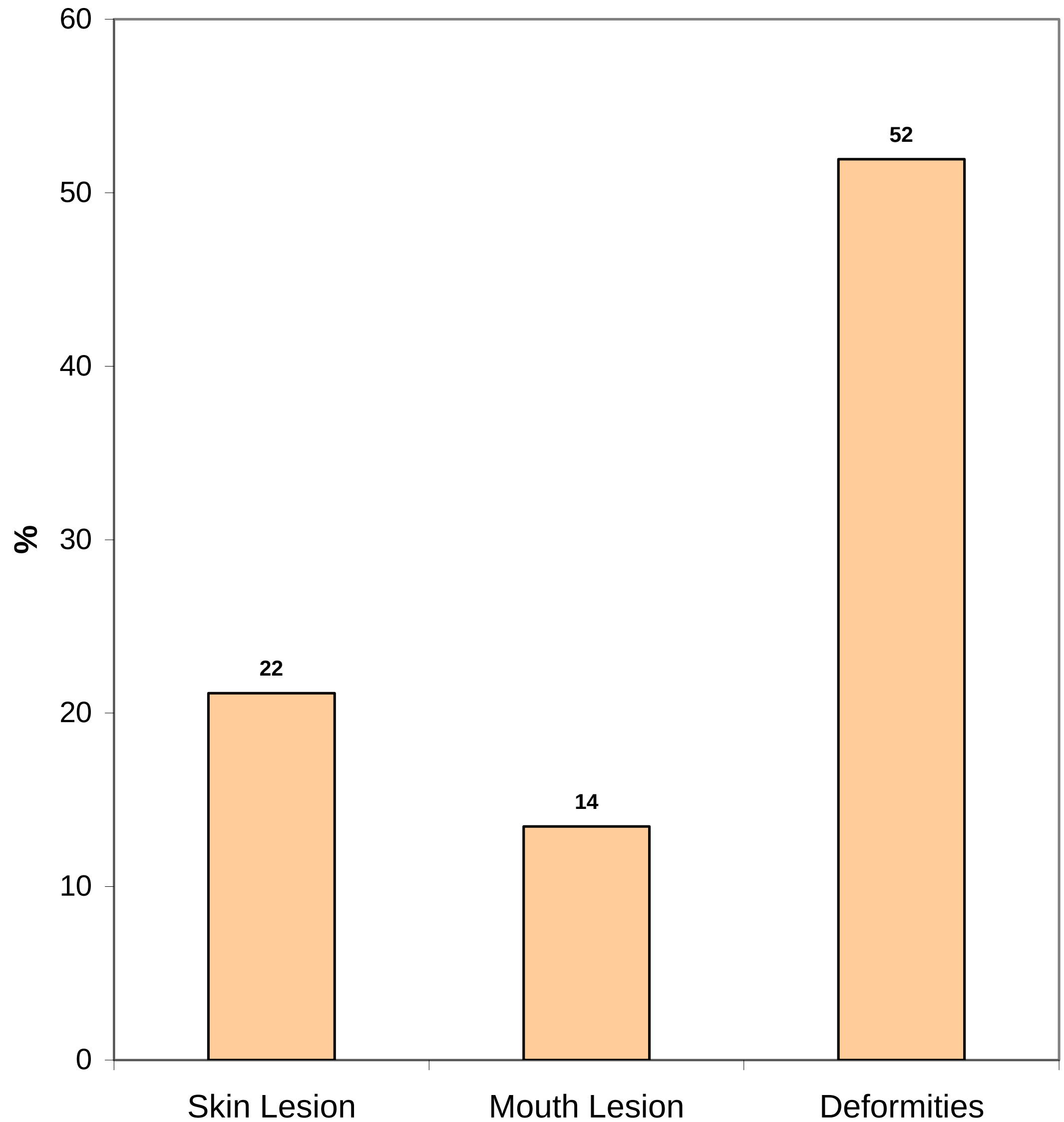

Figure 27. Tumor and deformity rates for individuals collected from Presque Isle Bay and identified as having all brown bullhead alleles, n = 52.

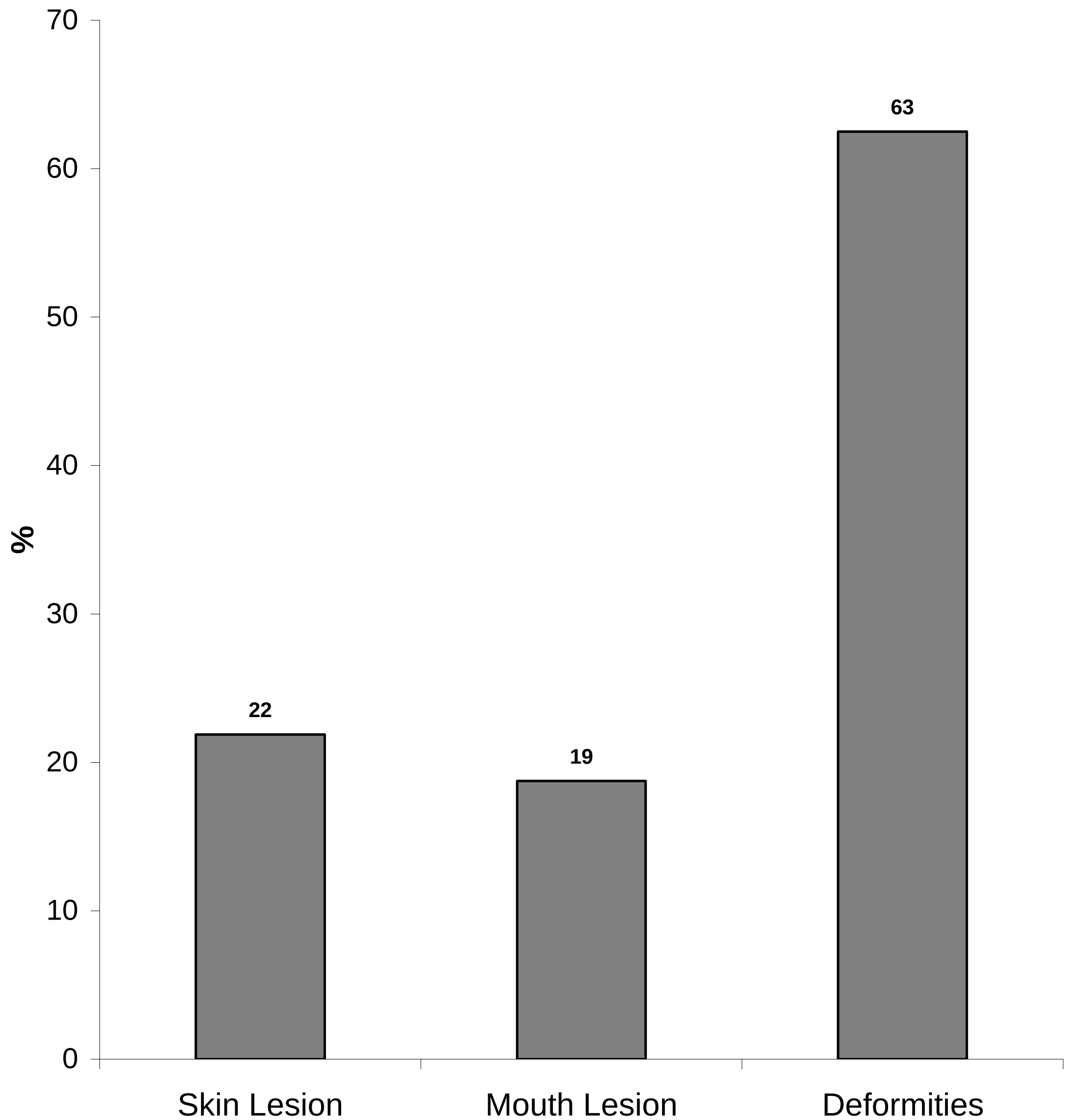

Figure 28. Tumor and deformity rates for individuals collected form Presque Isle Bay and identified as having some black bullhead alleles, n = 32.

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Appendix A

Descriptive Statistics

In tables A1- A31 the following abbreviations are used:

N = number of individuals

StDev = standard deviation

Table A1. Standard length (SL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	225.42	21.68	121.42	245.57
Petersburg, Pennsylvania	28	213.80	30.30	113.35	297.22
Sara's Cove, Presque Isle Bay	28	247.85	25.80	146.16	278.19
Lagoons, Presque Isle Bay	30	263.82	32.49	194.86	322.37
Thompson's Bay, Presque Isle Bay	28	269.06	24.28	205.11	313.67
Old Womans Creek, Ohio	28	242.56	22.91	197.79	283.47
Dunkirk Harbor, New York	22	222.96	38.54	164.83	303.11
Long Point Bay, Canada	28	212.72	23.85	182.98	275.89
Tamarack Lake, Pennsylvania	28	253.76	20.78	152.91	278.15

Table A2. Mean of corrected Head length (HL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.31	0.32	0.31	0.32
Petersburg, Pennsylvania	28	0.31	0.29	0.35	0.29
Sara's Cove, Presque Isle Bay	28	0.30	0.34	0.28	0.31
Lagoons, Presque Isle Bay	30	0.32	1.42	0.28	0.99
Thompson's Bay, Presque Isle Bay	28	0.29	0.31	0.29	0.29
Old Womans Creek, Ohio	28	0.29	0.34	0.29	0.29
Dunkirk Harbor, New York	22	0.29	0.30	0.28	0.29
Long Point Bay, Canada	28	0.29	0.30	0.29	0.30
Tamarack Lake, Pennsylvania	28	0.29	0.47	0.24	0.30

Table A3. Mean of corrected Head Width (HW) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.72	0.84	0.66	0.75
Petersburg, Pennsylvania	28	0.72	0.71	0.64	0.69
Sara's Cove, Presque Isle Bay	28	0.71	0.79	0.67	0.75
Lagoons, Presque Isle Bay	30	0.66	0.21	0.73	0.24
Thompson's Bay, Presque Isle Bay	28	0.72	1.00	0.68	0.79
Old Womans Creek, Ohio	28	0.71	0.89	0.69	0.74
Dunkirk Harbor, New York	22	0.68	0.89	0.60	0.78
Long Point Bay, Canada	28	0.69	0.87	0.67	0.74
Tamarack Lake, Pennsylvania	28	0.69	0.49	0.81	0.69

Table A4. Mean of corrected Post Orbital Head Length (POHL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.52	0.56	0.50	0.53
Petersburg, Pennsylvania	28	0.51	0.55	0.48	0.52
Sara's Cove, Presque Isle Bay	28	0.50	0.49	0.50	0.49
Lagoons, Presque Isle Bay	30	0.46	0.12	0.49	0.15
Thompson's Bay, Presque Isle Bay	28	0.51	0.56	0.49	0.52
Old Womans Creek, Ohio	28	0.53	1.10	0.50	0.89
Dunkirk Harbor, New York	22	0.51	0.54	0.49	0.52
Long Point Bay, Canada	28	0.51	0.52	0.49	0.51
Tamarack Lake, Pennsylvania	28	0.51	0.34	0.61	0.51

Table A5. Mean of corrected Horizontal Eye Depth (HED) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.12	0.19	0.13	0.14
Petersburg, Pennsylvania	28	0.12	0.08	0.15	0.11
Sara's Cove, Presque Isle Bay	28	0.12	0.13	0.15	0.12
Lagoons, Presque Isle Bay	30	0.11	0.03	0.12	0.03
Thompson's Bay, Presque Isle Bay	28	0.11	0.15	0.11	0.12
Old Womans Creek, Ohio	28	0.13	0.77	0.12	0.47
Dunkirk Harbor, New York	22	0.11	0.11	0.12	0.12
Long Point Bay, Canada	28	0.12	0.12	0.12	0.12
Tamarack Lake, Pennsylvania	28	0.11	0.11	0.14	0.13

Table A6. Mean of corrected Vertical Eye Depth (VED) of Ameiurus species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.11	0.15	0.15	0.13
Petersburg, Pennsylvania	28	0.12	0.08	0.14	0.11
Sara's Cove, Presque Isle Bay	28	0.11	0.12	0.14	0.11
Lagoons, Presque Isle Bay	30	0.10	0.02	0.12	0.03
Thompson's Bay, Presque Isle Bay	28	0.11	0.11	0.13	0.11
Old Womans Creek, Ohio	28	0.10	0.11	0.11	0.12
Dunkirk Harbor, New York	22	0.11	0.10	0.11	0.10
Long Point Bay, Canada	28	0.12	0.09	0.11	0.10
Tamarack Lake, Pennsylvania	28	0.11	0.10	0.12	0.11

Table A7. Mean of corrected Pre-Orbital Lenght (PRE) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.41	0.47	0.38	0.42
Petersburg, Pennsylvania	28	0.41	0.50	0.16	0.41
Sara's Cove, Presque Isle Bay	28	0.52	0.54	0.53	0.52
Lagoons, Presque Isle Bay	30	0.46	0.12	0.52	0.15
Thompson's Bay, Presque Isle Bay	28	0.52	0.58	0.53	0.53
Old Womans Creek, Ohio	28	0.49	0.86	0.12	0.51
Dunkirk Harbor, New York	22	0.42	0.50	0.38	0.48
Long Point Bay, Canada	28	0.52	0.51	0.50	0.51
Tamarack Lake, Pennsylvania	28	0.43	0.33	0.48	0.43

Table A8. Mean of corrected Cheek Depth (CD) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.23	0.33	0.19	0.24
Petersburg, Pennsylvania	28	0.22	0.28	0.17	0.22
Sara's Cove, Presque Isle Bay	28	0.23	0.33	0.19	0.26
Lagoons, Presque Isle Bay	30	0.22	0.07	0.21	0.08
Thompson's Bay, Presque Isle Bay	28	0.23	0.34	0.20	0.24
Old Womans Creek, Ohio	28	0.23	0.34	0.20	0.25
Dunkirk Harbor, New York	22	0.22	0.29	0.20	0.24
Long Point Bay, Canada	28	0.22	0.36	0.20	0.25
Tamarack Lake, Pennsylvania	28	0.20	0.20	0.23	0.22

Table A9. Mean of corrected Lower Jaw Lenght (LJL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.40	0.51	0.40	0.43
Petersburg, Pennsylvania	28	0.40	0.43	0.28	0.36
Sara's Cove, Presque Isle Bay	28	0.41	0.58	0.34	0.44
Lagoons, Presque Isle Bay	30	0.35	0.12	0.37	0.12
Thompson's Bay, Presque Isle Bay	28	0.38	0.57	0.38	0.43
Old Womans Creek, Ohio	28	0.37	0.60	0.34	0.43
Dunkirk Harbor, New York	22	0.37	0.59	0.31	0.44
Long Point Bay, Canada	28	0.36	0.49	0.34	0.39
Tamarack Lake, Pennsylvania	28	0.37	0.38	0.40	0.42

Table A10. Mean of corrected Head Depth (HD) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.49	0.55	0.46	0.49
Petersburg, Pennsylvania	28	0.49	0.59	0.46	0.55
Sara's Cove, Presque Isle Bay	28	0.48	0.62	0.40	0.51
Lagoons, Presque Isle Bay	30	0.43	0.13	0.43	0.14
B	28	0.48	0.56	0.46	0.50
Old Womans Creek, Ohio	28	0.44	0.51	0.41	0.48
Dunkirk Harbor, New York	22	0.45	0.61	0.42	0.54
Long Point Bay, Canada	28	0.45	0.59	0.43	0.49
Tamarack Lake, Pennsylvania	28	0.47	0.36	0.57	0.48

Table A11. Mean of corrected Body Depth (BD) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.21	0.24	0.19	0.23
Petersburg, Pennsylvania	28	0.21	0.19	0.26	0.21
Sara's Cove, Presque Isle Bay	28	0.25	0.34	0.23	0.27
Lagoons, Presque Isle Bay	30	0.24	0.29	0.21	0.24
Thompson's Bay, Presque Isle Bay	28	0.23	0.30	0.24	0.25
Old Womans Creek, Ohio	28	0.20	0.29	0.19	0.23
Dunkirk Harbor, New York	22	0.19	0.25	0.17	0.21
Long Point Bay, Canada	28	0.22	0.28	0.20	0.23
Tamarack Lake, Pennsylvania	28	0.21	0.25	0.21	0.22

Table A12. Mean of corrected distance from Snout to Dorsal Fin Insertion (SNDOR) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.42	0.44	0.42	0.43
Petersburg, Pennsylvania	28	0.42	0.40	0.46	0.40
Sara's Cove, Presque Isle Bay	28	0.40	0.44	0.38	0.41
Lagoons, Presque Isle Bay	30	0.40	0.44	0.39	0.39
Thompson's Bay, Presque Isle Bay	28	0.40	0.45	0.39	0.40
Old Womans Creek, Ohio	28	0.40	0.46	0.41	0.41
Dunkirk Harbor, New York	22	0.38	0.41	0.38	0.40
Long Point Bay, Canada	28	0.40	0.42	0.41	0.41
Tamarack Lake, Pennsylvania	28	0.39	0.43	0.37	0.40

Table A13. Mean of corrected distance from Snout to Pelvic Fin Insertion (SNPEL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.32	0.36	0.31	0.34
Petersburg, Pennsylvania	28	0.33	0.52	0.52	0.50
Sara's Cove, Presque Isle Bay	28	0.51	0.59	0.49	0.54
Lagoons, Presque Isle Bay	30	0.50	0.55	0.48	0.49
Thompson's Bay, Presque Isle Bay	28	0.49	0.54	0.49	0.50
Old Womans Creek, Ohio	28	0.48	0.53	0.48	0.50
Dunkirk Harbor, New York	22	0.48	0.53	0.45	0.51
Long Point Bay, Canada	28	0.49	0.51	0.47	0.48
Tamarack Lake, Pennsylvania	28	0.48	0.58	0.45	0.51

Table A14. Mean of corrected distance from Dorsal Fin Base Length (DFBL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.08	0.10	0.10	0.10
Petersburg, Pennsylvania	28	0.09	0.07	0.09	0.07
Sara's Cove, Presque Isle Bay	28	0.08	0.10	0.09	0.09
Lagoons, Presque Isle Bay	30	0.08	0.11	0.08	0.09
Thompson's Bay, Presque Isle Bay	28	0.09	0.07	0.10	0.09
Old Womans Creek, Ohio	28	0.08	0.09	0.08	0.08
Dunkirk Harbor, New York	22	0.08	0.07	0.08	0.07
Long Point Bay, Canada	28	0.08	0.10	0.07	0.08
Tamarack Lake, Pennsylvania	28	0.10	0.12	0.09	0.10

Table A15. Mean of corrected distance from Anterior Dorsal Fin Insertion to Anterior Anal Fin Insertion (ADAA) of *Ameiurus* species for each site

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.33	0.39	0.34	0.34
Petersburg, Pennsylvania	28	0.33	0.34	0.37	0.34
Sara's Cove, Presque Isle Bay	28	0.35	0.40	0.32	0.37
Lagoons, Presque Isle Bay	30	0.35	0.43	0.33	0.37
Thompson's Bay, Presque Isle Bay	28	0.35	0.32	0.36	0.36
Old Womans Creek, Ohio	28	0.34	0.38	0.34	0.37
Dunkirk Harbor, New York	22	0.34	0.40	0.33	0.36
Long Point Bay, Canada	28	0.34	0.39	0.32	0.36
Tamarack Lake, Pennsylvania	28	0.35	0.42	0.34	0.36

Table A16. Mean of corrected distance from Anterior Dorsal Fin Insertion to Posterior Anal Fin Insertion (ADPA) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.49	0.50	0.51	0.50
Petersburg, Pennsylvania	28	0.49	0.51	0.48	0.50
Sara's Cove, Presque Isle Bay	28	0.52	0.53	0.51	0.52
Lagoons, Presque Isle Bay	30	0.52	0.66	0.37	0.53
Thompson's Bay, Presque Isle Bay	28	0.53	0.51	0.54	0.53
Old Womans Creek, Ohio	28	0.50	0.61	0.39	0.49
Dunkirk Harbor, New York	22	0.52	0.56	0.51	0.53
Long Point Bay, Canada	28	0.51	0.53	0.50	0.52
Tamarack Lake, Pennsylvania	28	0.53	0.57	0.53	0.53

Table A17. Mean of corrected distance from Posterior Dorsal Fin Insertion to Anterior Anal Fin Insertion (PDAA) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.26	0.28	0.29	0.27
Petersburg, Pennsylvania	28	0.26	0.26	0.29	0.26
Sara's Cove, Presque Isle Bay	28	0.28	0.32	0.26	0.30
Lagoons, Presque Isle Bay	30	0.29	0.34	0.27	0.29
Thompson's Bay, Presque Isle Bay	28	0.28	0.30	0.28	0.28
Old Womans Creek, Ohio	28	0.27	0.30	0.28	0.28
Dunkirk Harbor, New York	22	0.28	0.35	0.25	0.29
Long Point Bay, Canada	28	0.28	0.34	0.27	0.29
Tamarack Lake, Pennsylvania	28	0.28	0.35	0.26	0.29

Table A18. Mean of corrected distance from Posterior Dorsal Fin Insertion to Posterior Anal Fin Insertion (PDPA) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.41	0.39	0.43	0.41
Petersburg, Pennsylvania	28	0.41	0.44	0.41	0.43
Sara's Cove, Presque Isle Bay	28	0.43	0.44	0.42	0.43
Lagoons, Presque Isle Bay	30	0.44	0.56	0.31	0.45
Thompson's Bay, Presque Isle Bay	28	0.44	0.45	0.44	0.45
Old Womans Creek, Ohio	28	0.42	0.54	0.31	0.42
Dunkirk Harbor, New York	22	0.44	0.50	0.42	0.47
Long Point Bay, Canada	28	0.43	0.44	0.43	0.44
Tamarack Lake, Pennsylvania	28	0.44	0.48	0.44	0.44

Table A19. Mean of corrected distance from Posterior Dorsal Fin Insertion to ventral point of least caudal peduncle (PDVC) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.53	0.52	0.56	0.55
Petersburg, Pennsylvania	28	0.54	0.52	0.58	0.54
Sara's Cove, Presque Isle Bay	28	0.55	0.83	0.25	0.56
Lagoons, Presque Isle Bay	30	0.56	0.57	0.56	0.56
Thompson's Bay, Presque Isle Bay	28	0.57	0.56	0.56	0.58
Old Womans Creek, Ohio	28	0.55	0.53	0.54	0.54
Dunkirk Harbor, New York	22	0.56	0.61	0.53	0.57
Long Point Bay, Canada	28	0.56	0.58	0.55	0.56
Tamarack Lake, Pennsylvania	28	0.56	0.56	0.57	0.55

Table A20. Mean of corrected distance from Posterior Anal Fin Insertion to Dorsal Point of Least Caudal Peduncle (PADC) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.19	0.23	0.20	0.21
Petersburg, Pennsylvania	28	0.19	0.17	0.24	0.19
Sara's Cove, Presque Isle Bay	28	0.20	0.17	0.28	0.22
Lagoons, Presque Isle Bay	30	0.19	0.22	0.18	0.20
Thompson's Bay, Presque Isle Bay	28	0.19	0.24	0.17	0.19
Old Womans Creek, Ohio	28	0.18	0.20	0.17	0.18
Dunkirk Harbor, New York	22	0.19	0.20	0.19	0.19
Long Point Bay, Canada	28	0.19	0.19	0.19	0.20
Tamarack Lake, Pennsylvania	28	0.18	0.18	0.20	0.19

Table A21. Mean of corrected distance from Anterior Dorsal Fin Insertion to Pelvic Fin (ADP2) of Ameiurus species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.23	0.29	0.25	0.26
Petersburg, Pennsylvania	28	0.23	0.20	0.29	0.22
Sara's Cove, Presque Isle Bay	28	0.27	0.32	0.30	0.30
Lagoons, Presque Isle Bay	29	0.26	0.29	0.24	0.27
Thompson's Bay, Presque Isle Bay	30	0.26	0.31	0.27	0.26
Old Womans Creek, Ohio	28	0.22	0.26	0.21	0.25
Dunkirk Harbor, New York	22	0.23	0.27	0.21	0.23
Long Point Bay, Canada	28	0.24	0.30	0.22	0.25
Tamarack Lake, Pennsylvania	28	0.23	0.28	0.24	0.26

Table A22. Mean of corrected distance from Posterior Dorsal Fin Insertion to Pelvic Fin (PDP2) . of Ameiurus species for each site

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.19	0.27	0.20	0.22
Petersburg, Pennsylvania	28	0.19	0.15	0.25	0.17
Sara's Cove, Presque Isle Bay	28	0.23	0.38	0.19	0.26
Lagoons, Presque Isle Bay	30	0.23	0.28	0.20	0.24
Thompson's Bay, Presque Isle Bay	28	0.23	0.30	0.21	0.23
Old Womans Creek, Ohio	28	0.18	0.24	0.17	0.21
Dunkirk Harbor, New York	22	0.19	0.24	0.17	0.20
Long Point Bay, Canada	28	0.21	0.28	0.19	0.22
Tamarack Lake, Pennsylvania	28	0.20	0.25	0.20	0.21

Table A23. Mean of corrected Caudal Peduncle Length (CPL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.15	0.20	0.16	0.17
Petersburg, Pennsylvania	28	0.15	0.14	0.17	0.15
Sara's Cove, Presque Isle Bay	28	0.15	0.16	0.16	0.15
Lagoons, Presque Isle Bay	30	0.14	0.19	0.14	0.15
Thompson's Bay, Presque Isle Bay	28	0.15	0.19	0.14	0.15
Old Womans Creek, Ohio	28	0.14	0.19	0.11	0.15
Dunkirk Harbor, New York	22	0.15	0.14	0.16	0.14
Long Point Bay, Canada	28	0.14	0.16	0.13	0.14
Tamarack Lake, Pennsylvania	28	0.15	0.17	0.16	0.15

Table A24. Mean of corrected Least Caudal Peduncle Length (LCPD) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.10	0.12	0.09	0.11
Petersburg, Pennsylvania	28	0.10	0.09	0.13	0.10
Sara's Cove, Presque Isle Bay	28	0.11	0.11	0.12	0.12
Lagoons, Presque Isle Bay	30	0.11	0.12	0.11	0.11
Thompson's Bay, Presque Isle Bay	28	0.11	0.13	0.10	0.11
Old Womans Creek, Ohio	28	0.11	0.11	0.11	0.11
Dunkirk Harbor, New York	22	0.10	0.12	0.10	0.11
Long Point Bay, Canada	28	0.11	0.13	0.10	0.12
Tamarack Lake, Pennsylvania	28	0.11	0.12	0.12	0.11

Table A25. Mean of corrected Anal Fin Base Length (AFBL) of *Ameiurus* species for each site.

Site	N	Mean	StDev	Minimum	Maximum
Clear Lake, Iowa	28	0.21	0.25	0.22	0.23
Petersburg, Pennsylvania	28	0.22	0.21	0.23	0.21
Sara's Cove, Presque Isle Bay	28	0.23	0.27	0.28	0.25
Lagoons, Presque Isle Bay	30	0.23	0.23	0.23	0.22
Thompson's Bay, Presque Isle Bay	28	0.23	0.38	0.12	0.24
Old Womans Creek, Ohio	28	0.21	0.23	0.21	0.22
Dunkirk Harbor, New York	22	0.22	0.28	0.10	0.22
Long Point Bay, Canada	28	0.22	0.21	0.22	0.22
Tamarack Lake, Pennsylvania	28	0.23	0.26	0.24	0.24

Table A26. Frequency distribution of the number of Dorsal fin rays (Dray) for *Ameiurus* species for each site.

Site	N	4	5	6	7	8	Mean	StDev
Clear Lake, Iowa	28		1	27	2		6.032258	0.314523
Petersburg, Pennsylvania	28	1	1	26			5.849896	0.423659
Sara's Cove, Presque Isle Bay	28		2	28			5.928571	0.262265
Lagoons, Presque Isle Bay	30		2	27			5.931034	0.257881
Thompson's Bay, Presque Isle Bay	28		1	26			5.962963	0.19245
Old Womans Creek, Ohio	28		1	25		1	6.037037	0.436902
Dunkirk Harbor, New York	22			19	3		6.136364	0.35125
Long Point Bay, Canada	28		3	25	1		5.931034	0.371391
Tamarack Lake, Pennsylvania	28			27	2		6.068966	0.257881

Table A27. Frequency distribution of the number of Anal fin rays (Aray) for *Ameiurus* species for each site.

Site	N	17	18	19	20	21	22	Mean	StDev
Clear Lake, Iowa	28	8	6	9	7			18.48	1.12
Petersburg, Pennsylvania	28			1	17	10		20.33	0.55
Sara's Cove, Presque Isle Bay	28	1		4	15	4	4	20.18	1.09
Lagoons, Presque Isle Bay	30	1	3	5	10	6	4	20.00	1.31
Thompson's Bay, Presque Isle Bay	28		1	5	13	8		20.11	0.89
Old Womans Creek, Ohio	28		6	7	10	3	1	19.48	1.34
Dunkirk Harbor, New York	22		2	3	7	7	2	20.23	1.11
Long Point Bay, Canada	28		1	7	9	9	3	20.21	1.05
Tamarack Lake, Pennsylvania	28		1	7	10	10	1	20.10	0.94

Table A28. Frequency distribution of the number of Pectoral fin rays (P1rays) for *Ameiurus* species for each site.

Site	N	6	7	8	9	Mean	StDev
Clear Lake, Iowa	28	1	2	27		7.87	0.43
Petersburg, Pennsylvania	28	1	5	22		7.74	0.53
Sara's Cove, Presque Isle Bay	28		2	28		7.93	0.26
Lagoons, Presque Isle Bay	30		4	24	1	7.90	0.41
Thompson's Bay, Presque Isle Bay	28		1	22	4	8.11	0.42
Old Womans Creek, Ohio	28		8	18	1	7.74	0.53
Dunkirk Harbor, New York	22	2	3	17		7.68	0.65
Long Point Bay, Canada	28		3	26		7.90	0.31
Tamarack Lake, Pennsylvania	28			29		8.00	0.00

Table A29. Frequency distribution of the number of Pelvic fin rays (P2rays) for *Ameiurus* species for each site.

Site	N	6	7	8	9	Mean	StDev
Clear Lake, Iowa	28		2	28		7.94	0.25
Petersburg, Pennsylvania	28			28		8.00	0.00
Sara's Cove, Presque Isle Bay	28		1	29		7.96	0.19
Lagoons, Presque Isle Bay	30		1	27	1	7.96	0.19
Thompson's Bay, Presque Isle Bay	28		1	24	2	8.04	0.34
Old Womans Creek, Ohio	28			26	1	8.04	0.19
Dunkirk Harbor, New York	22		1	19	2	8.05	0.38
Long Point Bay, Canada	28		1	27	1	8.00	0.27
Tamarack Lake, Pennsylvania	28		1	26	2	8.03	0.33

Table A30. Frequency distribution of the number of Epibranchial gill raker (EGR) for *Ameiurus* species for each site.

Site	N	3	4	5	6	7	Mean	StDev
Clear Lake, Iowa	28			25	2	3	5.23	0.56
Petersburg, Pennsylvania	28	2	26				3.93	0.27
Sara's Cove, Presque Isle Bay	28	13	17				3.54	0.51
Lagoons, Presque Isle Bay	30	5	19	5			4.00	0.60
Thompson's Bay, Presque Isle Bay	28	6	21				3.78	0.42
Old Womans Creek, Ohio	28	5	20	2			3.89	0.51
Dunkirk Harbor, New York	22	6	15	1			3.77	0.53
Long Point Bay, Canada	28	11	18				3.62	0.49
Tamarack Lake, Pennsylvania	28	5	24				3.83	0.38

Table A31. Frequency distribution of the number of Ceratobranchial gill raker (CGR) for *Ameiurus* species for each site.

Site	N	7	8	9	10	11	12	13+	Mean	StDev
Clear Lake, Iowa	28				5	12	8	6	11.61	1.26
Petersburg, Pennsylvania	28		10	18					8.63	0.49
Sara's Cove, Presque Isle Bay	28	8	18	3	1				7.89	0.69
Lagoons, Presque Isle Bay	30	1	12	16					8.48	0.69
Thompson's Bay, Presque Isle Bay	28	7	12	6	2				8.22	0.85
Old Womans Creek, Ohio	28		11	16					8.59	0.50
Dunkirk Harbor, New York	22		12	10					8.45	0.51
Long Point Bay, Canada	8	4	18	7					8.14	0.69
Tamarack Lake, Pennsylvania	28	2	14	13					8.38	0.62

Appendix B

Hybrid Index

A hybrid index was created for two individuals from Presque Isle Bay following Stauffer, Hocutt, and Denoncourt (1978);

$$H = [(X_H - u_1) / (u_2 - u_1)] \times 100$$

Where H = hybrid index, X_H = hybrid value, u_1 = value for *Ameiurus melas* and u_2 = value for *Ameiurus nebulosus*. An index value of 50 denotes exact intermediacy; over 50 indicates that the particular character is closer to *A. melas* and less than 50 indicates a closer resemblance with *A. nebulosus*.

Table B1. Comparison of the intergeneric hybrid *Ameiurus Nebulosus* x *Ameiurus melas* from Presque Isle Bay with pure species.

	Ameiurus melas				Hybrid		*Ameiurus nebulosus*			*Hybrid*
		n=30			n=2			n=27		*index*
Character	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	
SL (mm)	121.42	245.57	225.4197	281.08	283.9	282.49	113.35	297.22	158.4089	…..
HL	0.294946	0.324763	0.311124	0.306411	0.312402	0.309406	0.276444	0.370534	0.293939	*-106*
HW	0.202739	0.253354	0.225346	0.68318	0.689619	0.6864	0.179035	0.260962	0.198207	*
POHL	0.469379	0.557332	0.515097	0.522036	0.526957	0.524497	0.459524	0.531649	0.497811	*
HED	0.094401	0.147715	0.121778	0.123562	0.124727	0.124145	0.114374	0.176813	0.14819	*-81.9*
VED	0.088865	0.152767	0.114607	0.105455	0.106449	0.105952	0.106	0.155448	0.138276	**
PRE	0.384697	0.452797	0.413275	0.515089	0.519945	0.517517	0.14607	0.444696	0.397787	*
CD	0.165665	0.272309	0.225556	0.221387	0.223474	0.222431	0.152316	0.243441	0.190495	*-119*
LJL	0.347347	0.479312	0.398883	0.391869	0.395563	0.393716	0.261429	0.380746	0.330793	*-121*
HD	0.135718	0.167346	0.152571	0.150088	0.151594	0.150841	0.133735	0.183591	0.150105	*-102*
BD	0.187145	0.723193	0.224904	0.25044	0.252953	0.251697	0.208431	0.323599	0.232838	*
SNDOR	0.397606	0.450968	0.424508	0.426418	0.430696	0.428557	0.371751	0.455492	0.393786	*
SNPEL	0.302524	0.345205	0.323892	0.473089	0.477835	0.475462	0.456696	0.524956	0.486259	*-51.5*
DFBL	0.072444	0.103101	0.084875	0.08429	0.085136	0.084713	0.068272	0.103426	0.092571	**
ADAA	0.225621	0.362638	0.329439	0.316414	0.319589	0.318001	0.315834	0.402294	0.335949	**
ADPA	0.397332	0.517614	0.48892	0.522332	0.527572	0.524952	0.484076	0.549292	0.512253	*
PDAA	0.21923	0.296497	0.262288	0.247587	0.250071	0.248829	0.249028	0.361006	0.270331	**
PDPA	0.386428	0.437307	0.407916	0.439451	0.443859	0.441655	0.397735	0.46669	0.428841	*
PDVC	0.509024	0.563864	0.533495	0.54727	0.552761	0.550015	0.529031	0.589322	0.557947	*-94*
PADC	0.16247	0.214905	0.187275	0.165586	0.167248	0.166417	0.173217	0.255404	0.200617	**
ADP2	0.185048	0.274863	0.231756	0.253188	0.255728	0.254458	0.222933	0.339921	0.257648	*-87.7*
PDP2	0.142752	0.240294	0.192276	0.272561	0.275295	0.273928	0.172768	0.323776	0.219179	*
CPL	0.133258	0.174171	0.154175	0.152025	0.153551	0.152788	0.13763	0.172469	0.156848	**
LCPD	0.08401	0.119178	0.099408	0.116273	0.11744	0.116857	0.09898	0.170534	0.111999	*
AFBL	0.190879	0.240377	0.214306	0.243572	0.246015	0.244794	0.214824	0.265549	0.238155	*
DRay	5	7	6.032258	6	6	6	4	6	5.888889	*-106*
Aray	17	20	18.48387	18	18	18	19	21	20.33333	**
P2Rays	7	8	7.935484	8	8	8	8	8	8	*-92.7*
P1RAYS	6	8	7.870968	8	8	8	6	8	7.740741	*
TGR	5	7	5.225806	5	5	5	3	4	3.925926	*-156*
BGR	10	15	11.6129	8	9	8.5	8	9	8.62963	**

*Hybrid value greater than the mean for either parent.

** Hybrid value lower than the mean for either parent.

Appendix C

Table C1. Fluorescently labeled microsatellite primer sets for *Ameiurus nebulosus.*

Primer	Sequence	Microsatellite	length
Aneb16F	5' ATA TGA TAC TGA AAA CAG GTT GCC 3'	(GATA)24	~287
Aneb16R	5' GCT CCA AAT GTG TGC AAT TAG TAG 3'		~300
Aneb37F	5' CTT CCG AAC ATG CTG GGG TAT G 3'	(CTAT)12(CTGT)10	~267
Aneb37R	5' GAC TGC GGT TGC TGA TAT GGC 3'		
Aneb39F	5' AGC TTA GCT GCT GTC CTG CTA TCA CAC 3'	(GTAT)16	~240
Aneb39R	5' GCT GTC GCT TAC GGC CAT ATT 3'		
Aneb42F	5' AGC AAA CAC TTC TAT CCC AAA C 3'	(GTTT)10	~217
Aneb42R	5' CTA AAG ACC CAC CTC CTA CG 3'		
Aneb51F	5' GCT TAT AGA GAC CCA CAG TTA T 3'	(GATA)8(GAT)(GATA)15	~238
Aneb51R	5' TTT GAG CTA CTA GGA TCC C 3'		
Aneb61R	5' GTG TGC CTG AAC AAG CTC 3'	(CAGA)8	
Aneb61F	5' TGG GTT GAA AAT GAT GTA ATT C 3'		~255
Aneb63F	5' CTA ACT AAC TAG CCA ACA AAC C 3'	(CTAT)7	~167
Aneb63R	5' CGC ATG TTT TAT TTT CTC AA 3'		
Aneb64F	5' GCT GCA GCT GCC ACT ACT GCT GTG ACC 3'	(GTAT)8	~161
Aneb64R	5' TCC AAT CTT CAC CAA ATC TCG C 3'		
Aneb86F	5' CCA GCA GAG GAA CTG ATT AG 3'	(CTAT)13	~264
Aneb86R	5' ATT TCC TAC TGA CAG ACG GAT A 3'		

Appendix D

Allele Frequencies at microsatellite Locus Aneb16, Aneb37, Aneb61, Aneb63, and Aneb64 for the Presque Isle Bay collections; Lake Erie collections; and Tamarack Lake and Wisconsin collections.

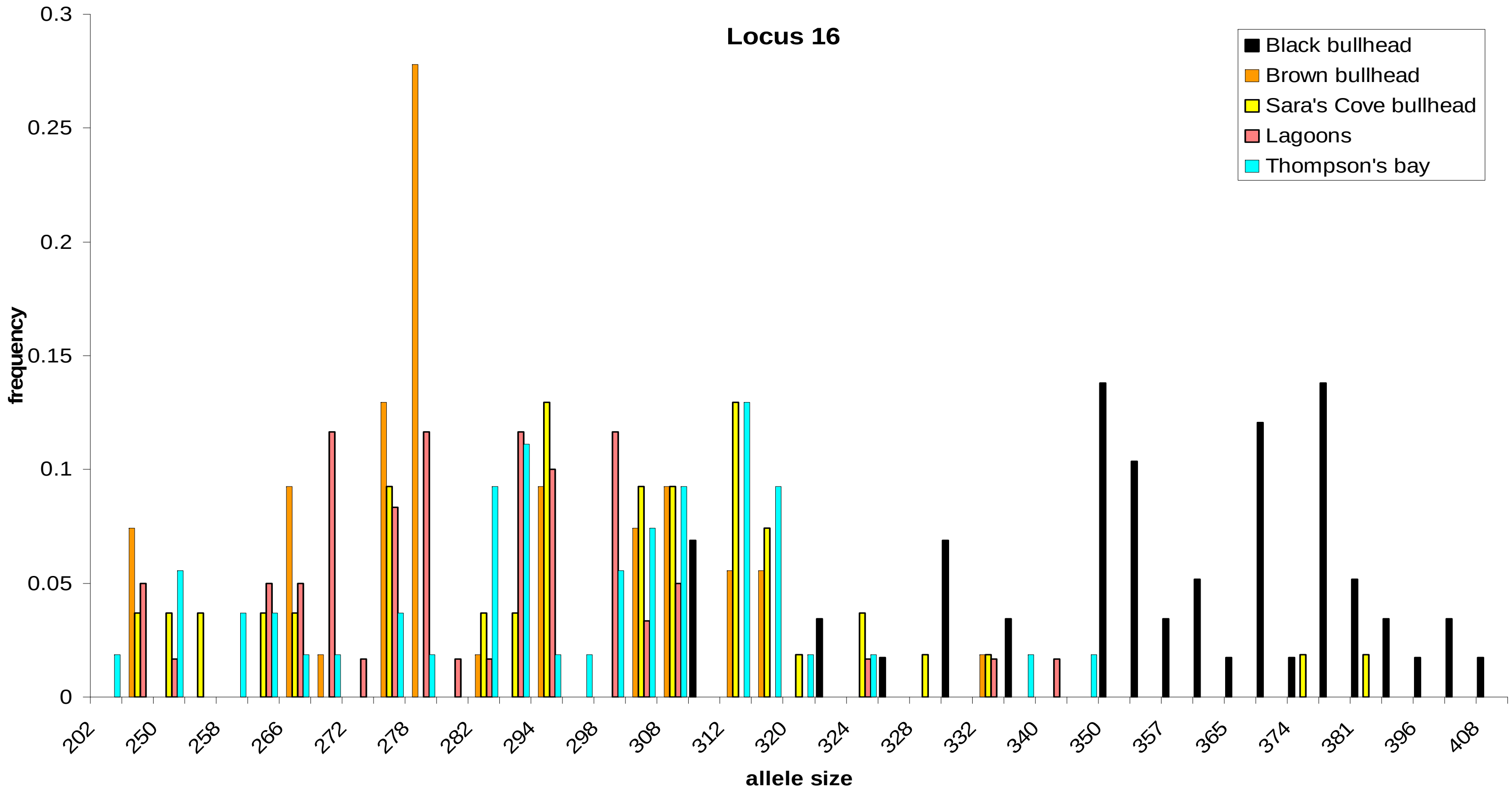

Figure D1. Allele frequencies at microsatellite locus 16 from samples of Presque Isle Bay populations.

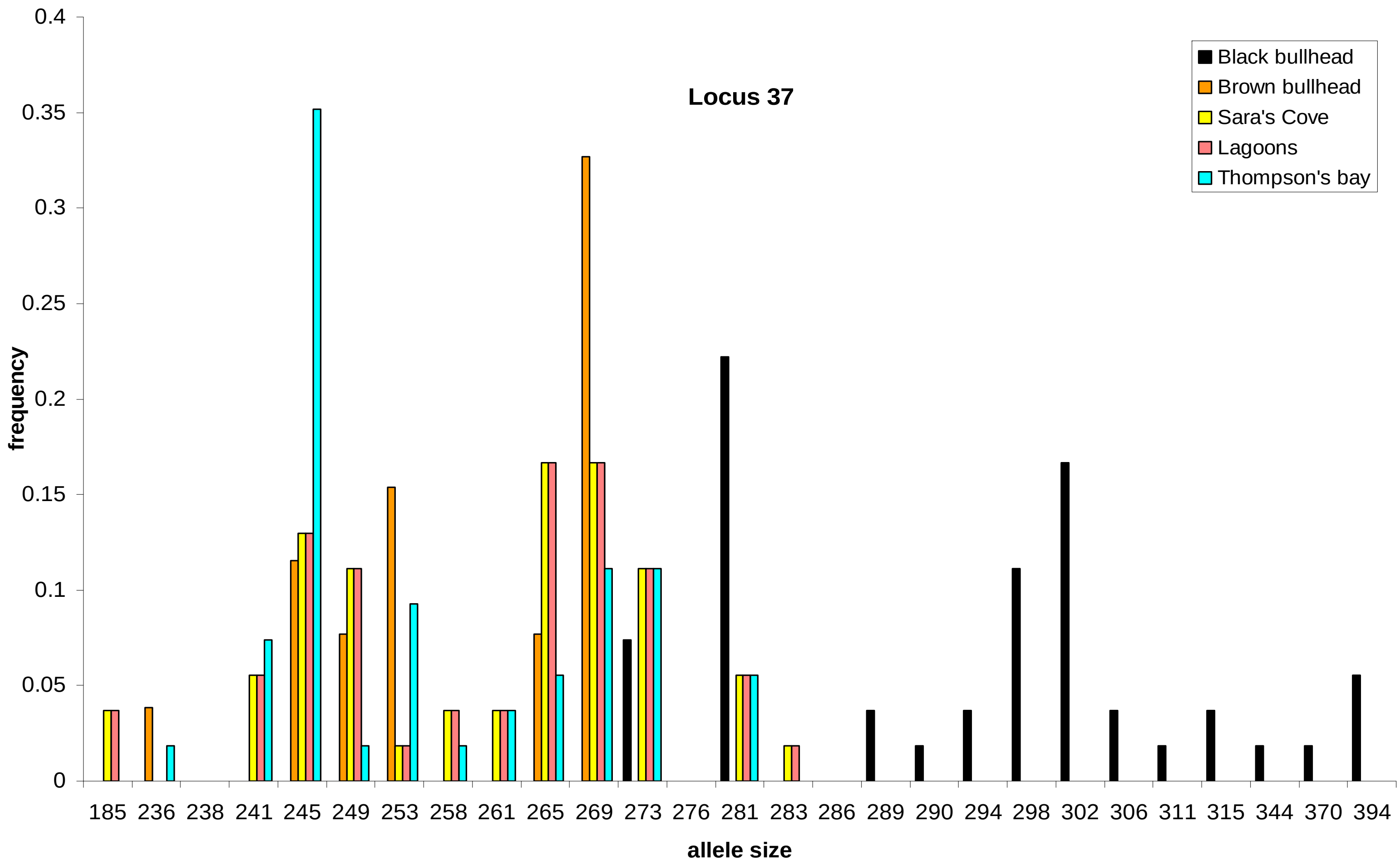

Figure D2. Allele frequencies at microsatellite locus 37 from samples of Presque Isle Bay populations.

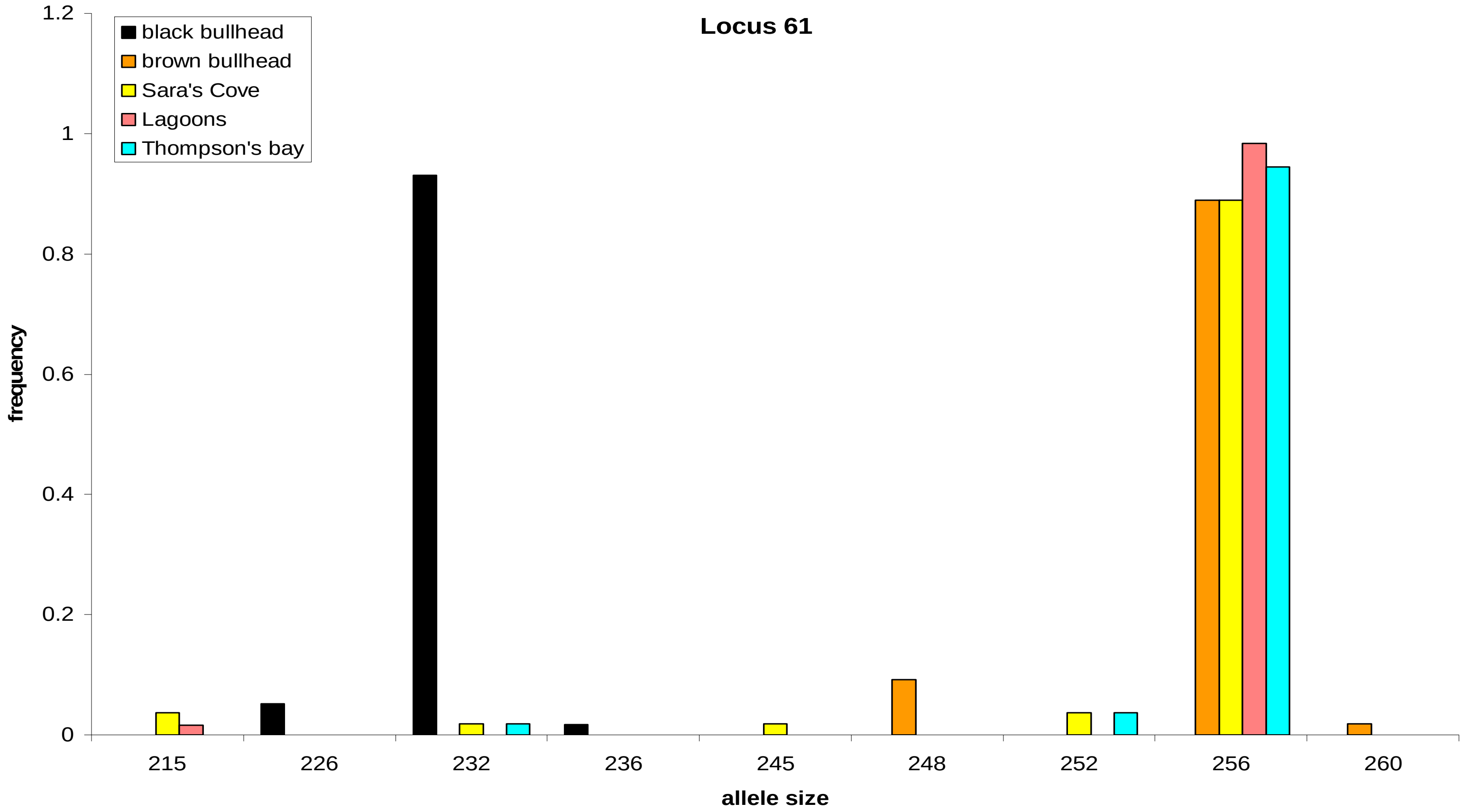

Figure D3. Allele frequencies at microsatellite locus 61 from samples of Presque Isle Bay populations.

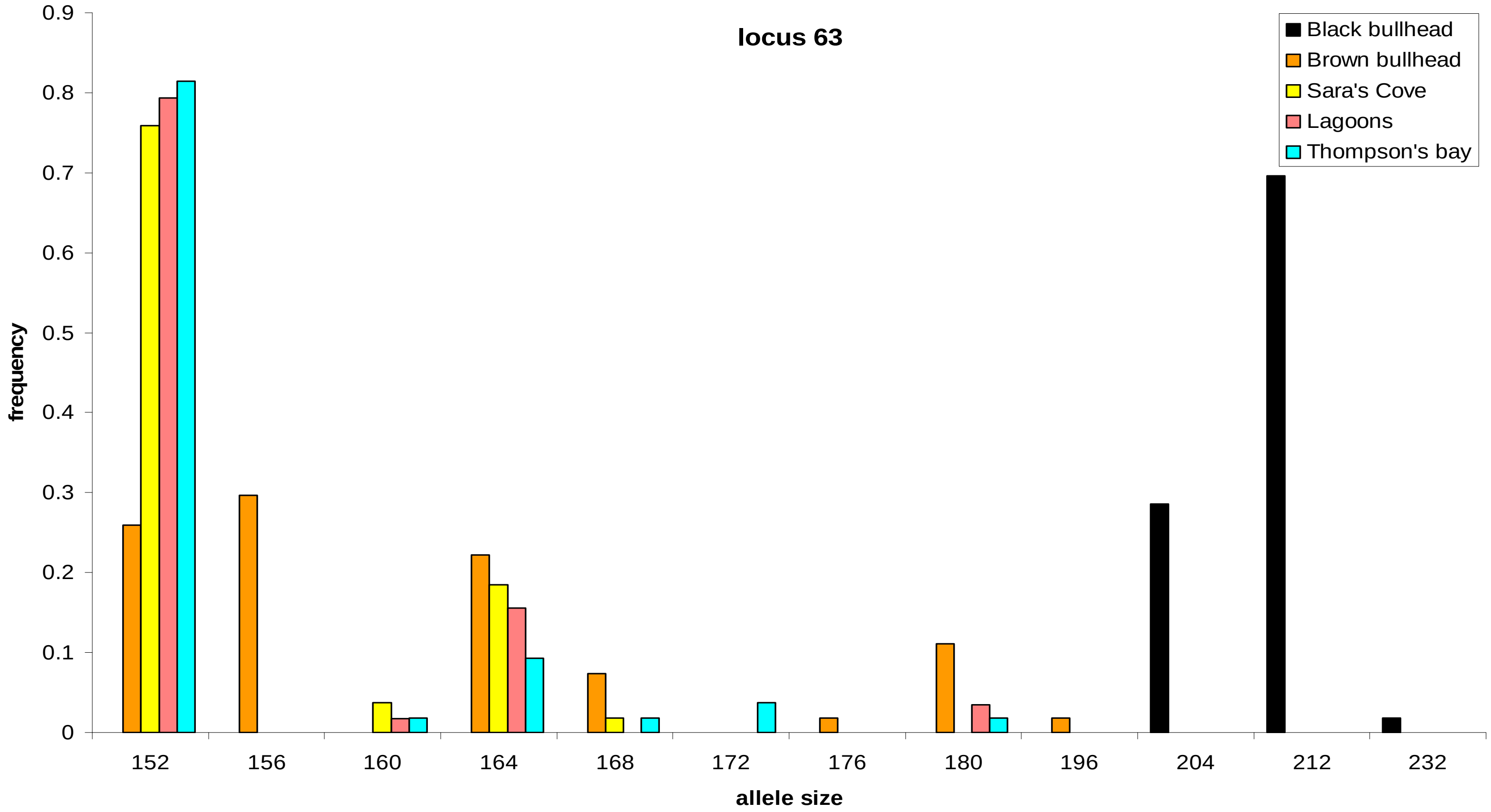

Figure D4. Allele frequencies at microsatellite locus 63 from samples of Presque Isle Bay populations.

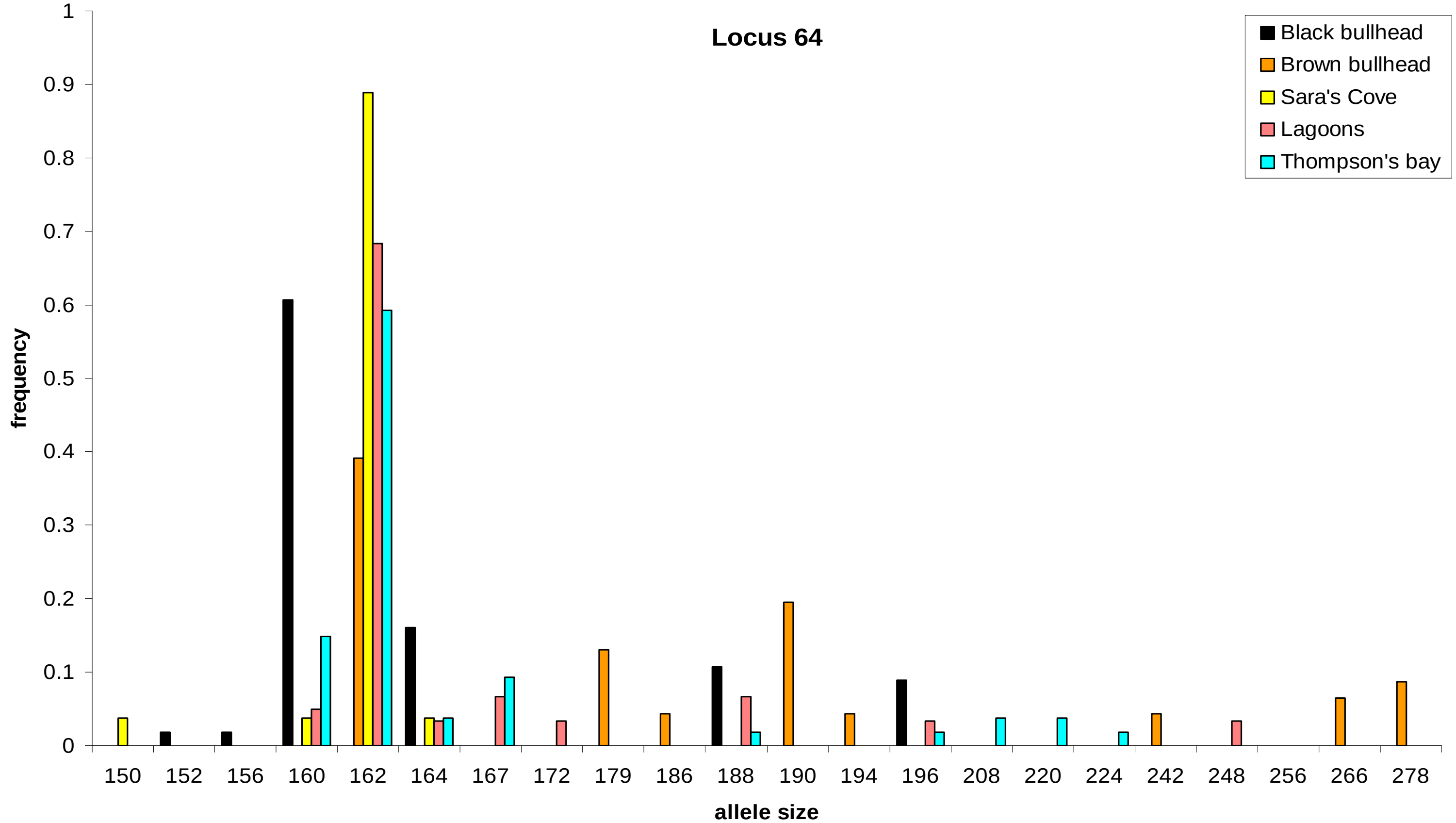

Figure D5. Allele frequencies at microsatellite locus 64 from samples of Presque Isle Bay populations.

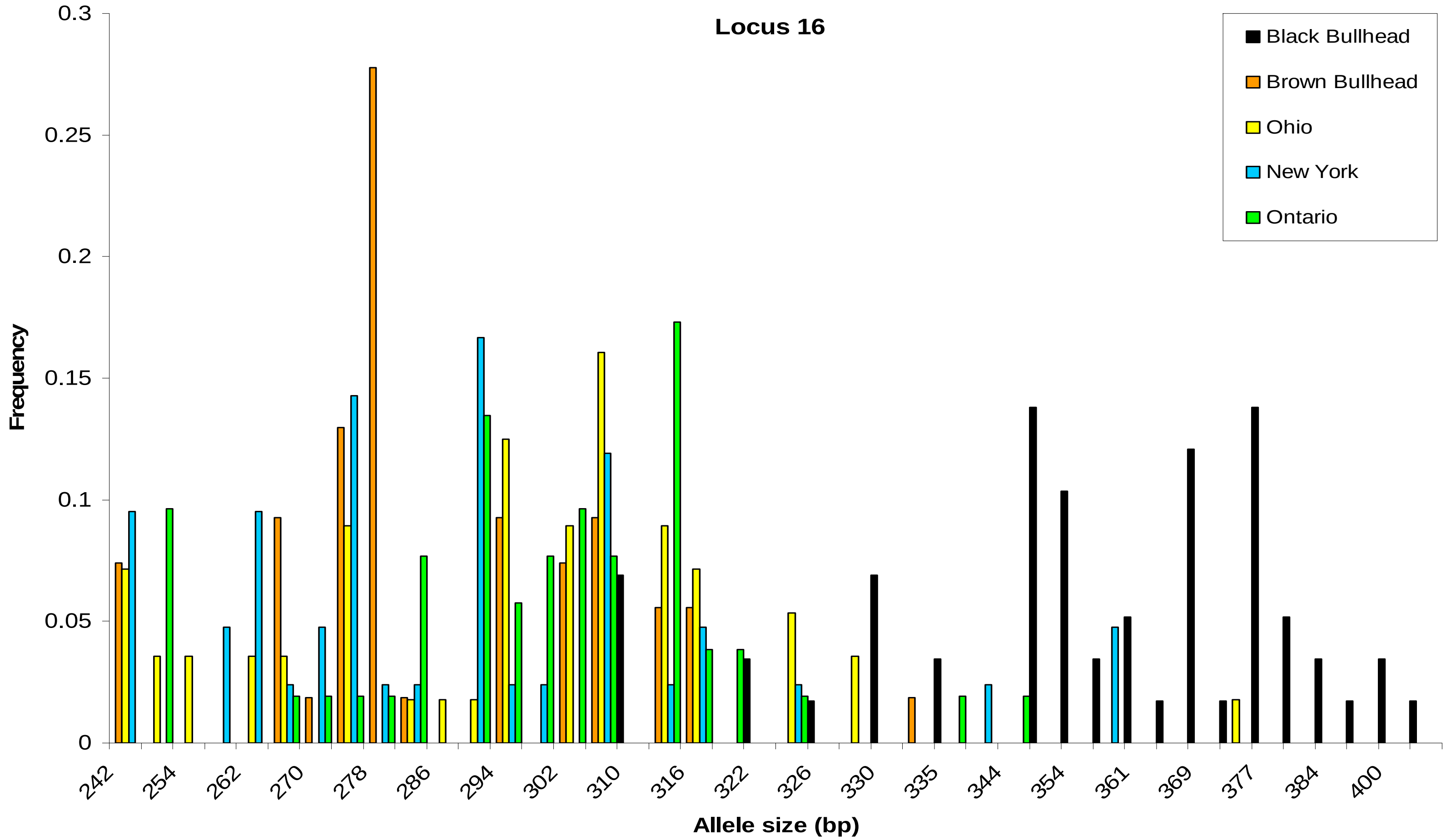

Figure D6. Allele frequencies at microsatellite locus 16 from samples of Lake Erie populations.

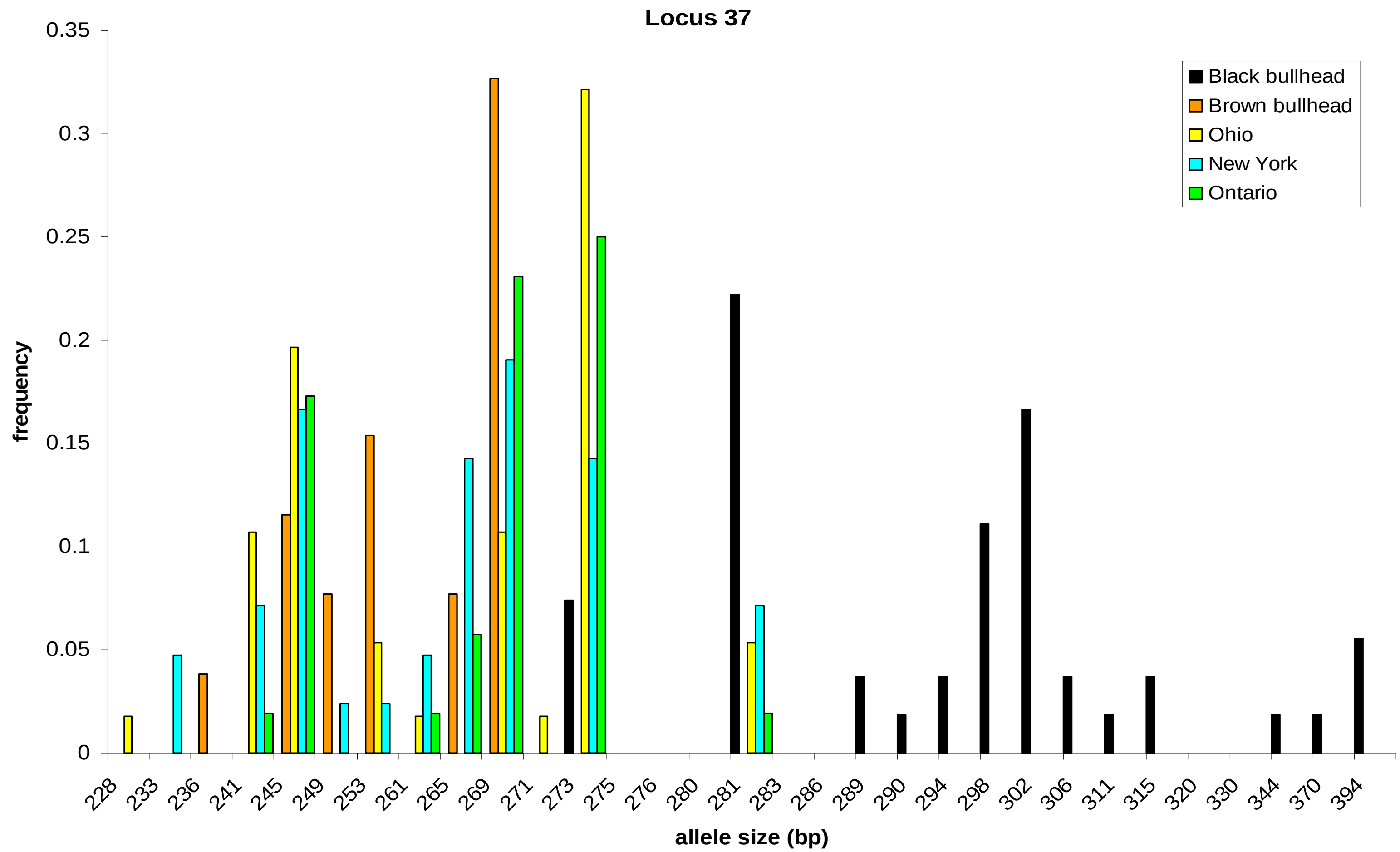

Figure D7. Allele frequencies at microsatellite locus 37 from samples of Lake Erie populations.

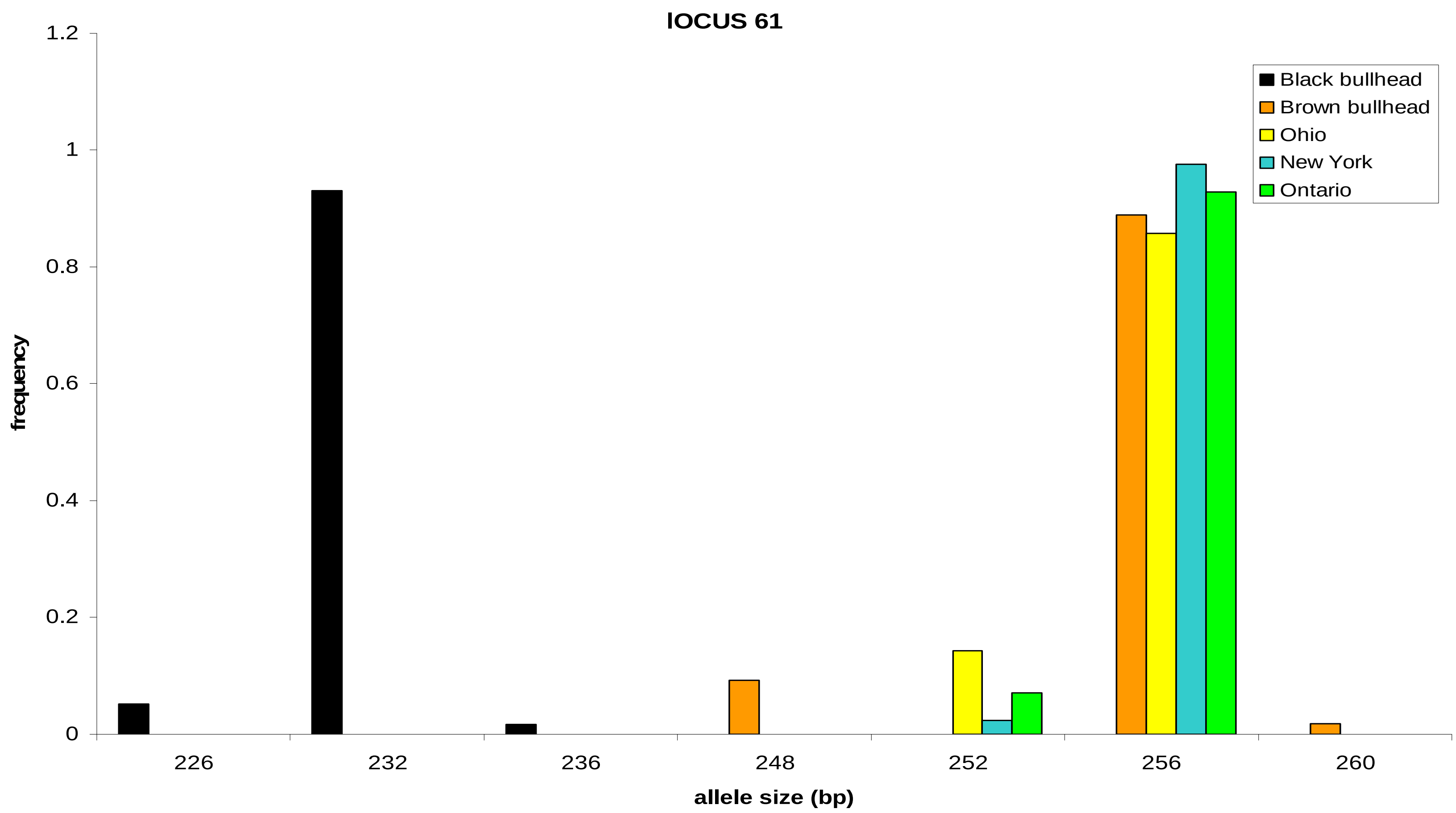

Figure D8. Allele frequencies at microsatellite locus 61 from samples of Lake Erie populations.

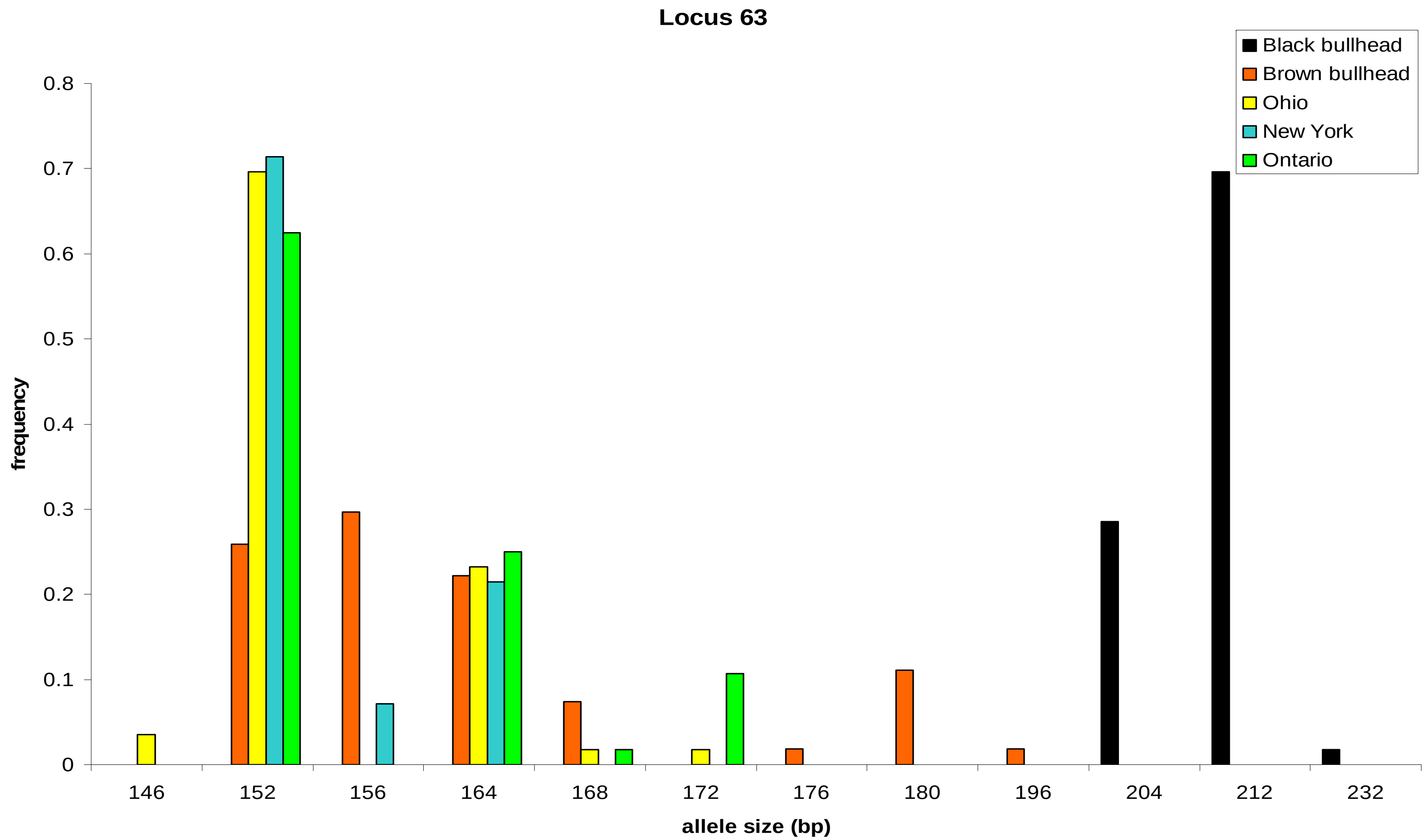

Figure D9. Allele frequencies at microsatellite locus 63 from samples of Lake Erie populations.

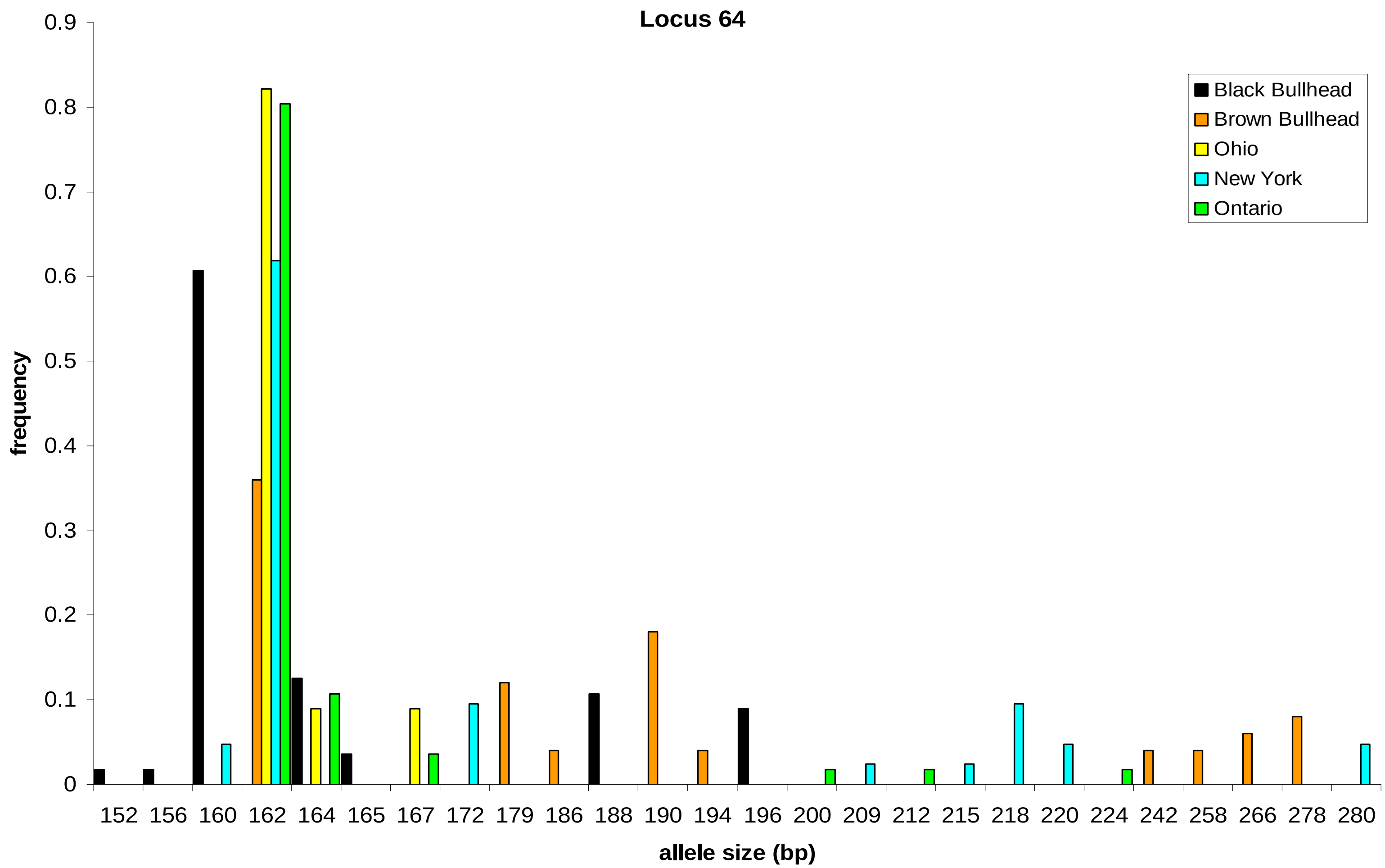

Figure D10. Allele frequencies at microsatellite locus 64 from samples of Lake Erie populations.

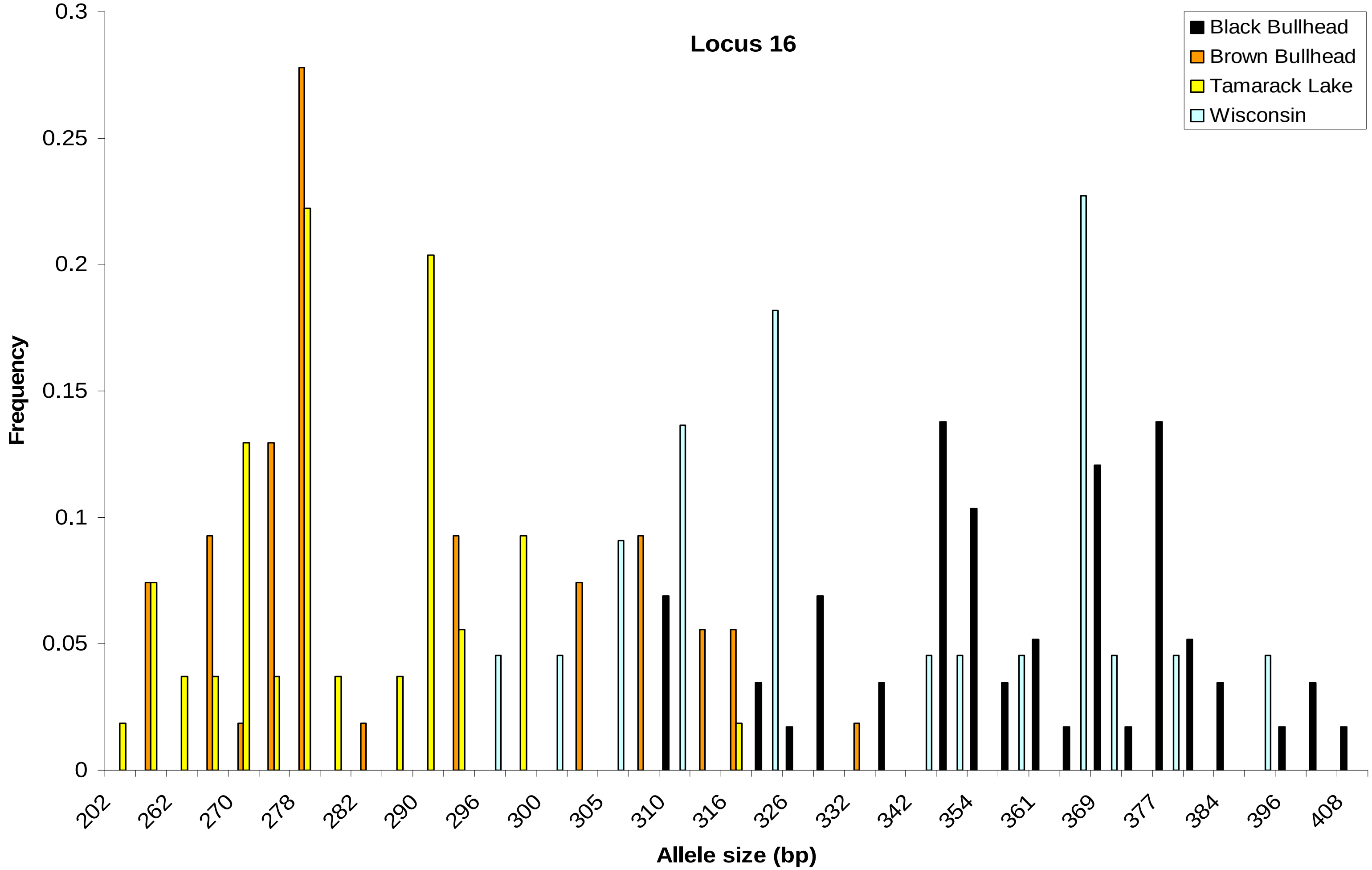

Figure D11. Allele frequencies at microsatellite locus 16 from samples of Tamarack Lake and Wisconsin populations.

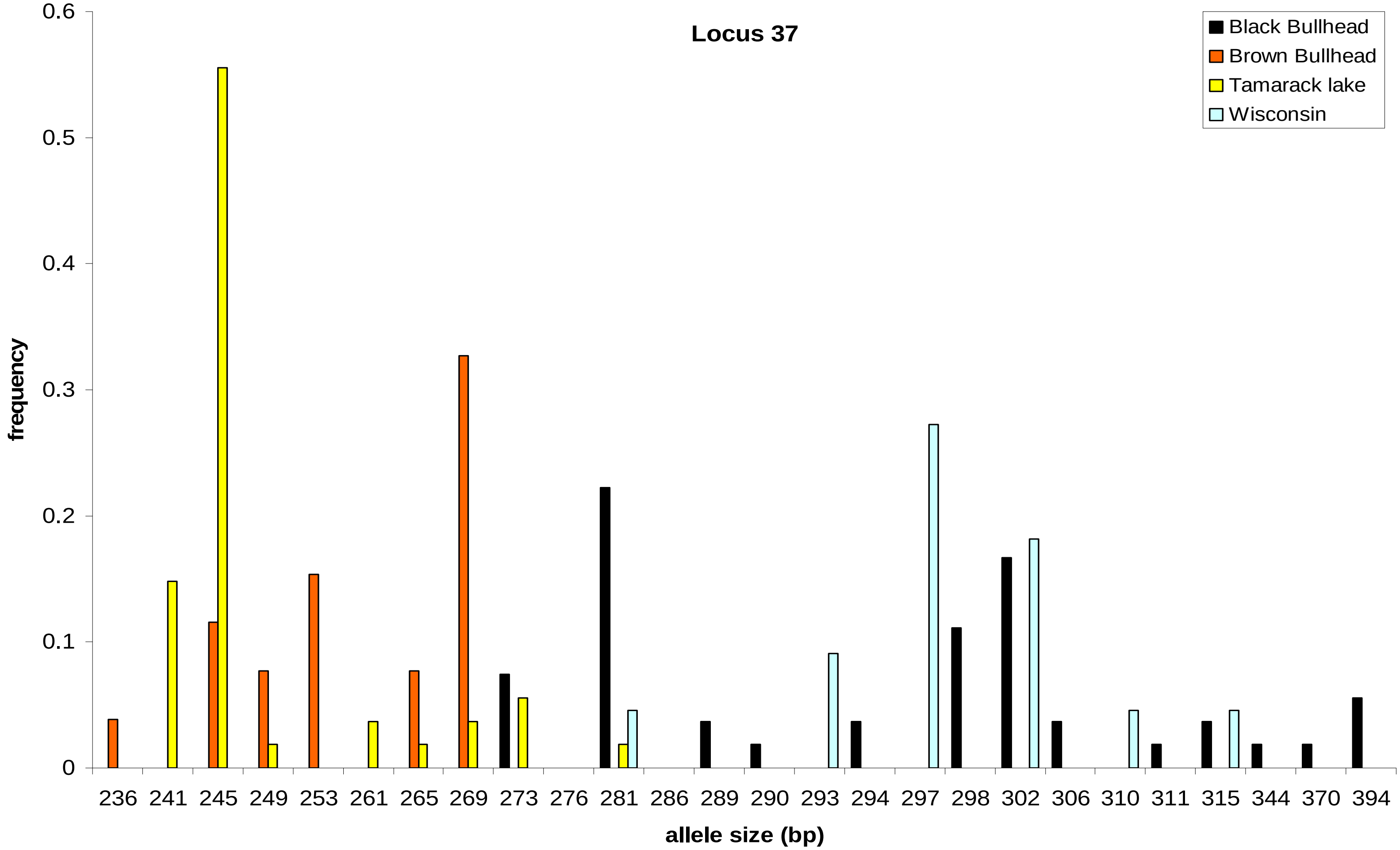

Figure D12. Allele frequencies at microsatellite locus 37 from samples of Tamarack Lake and Wisconsin populations.

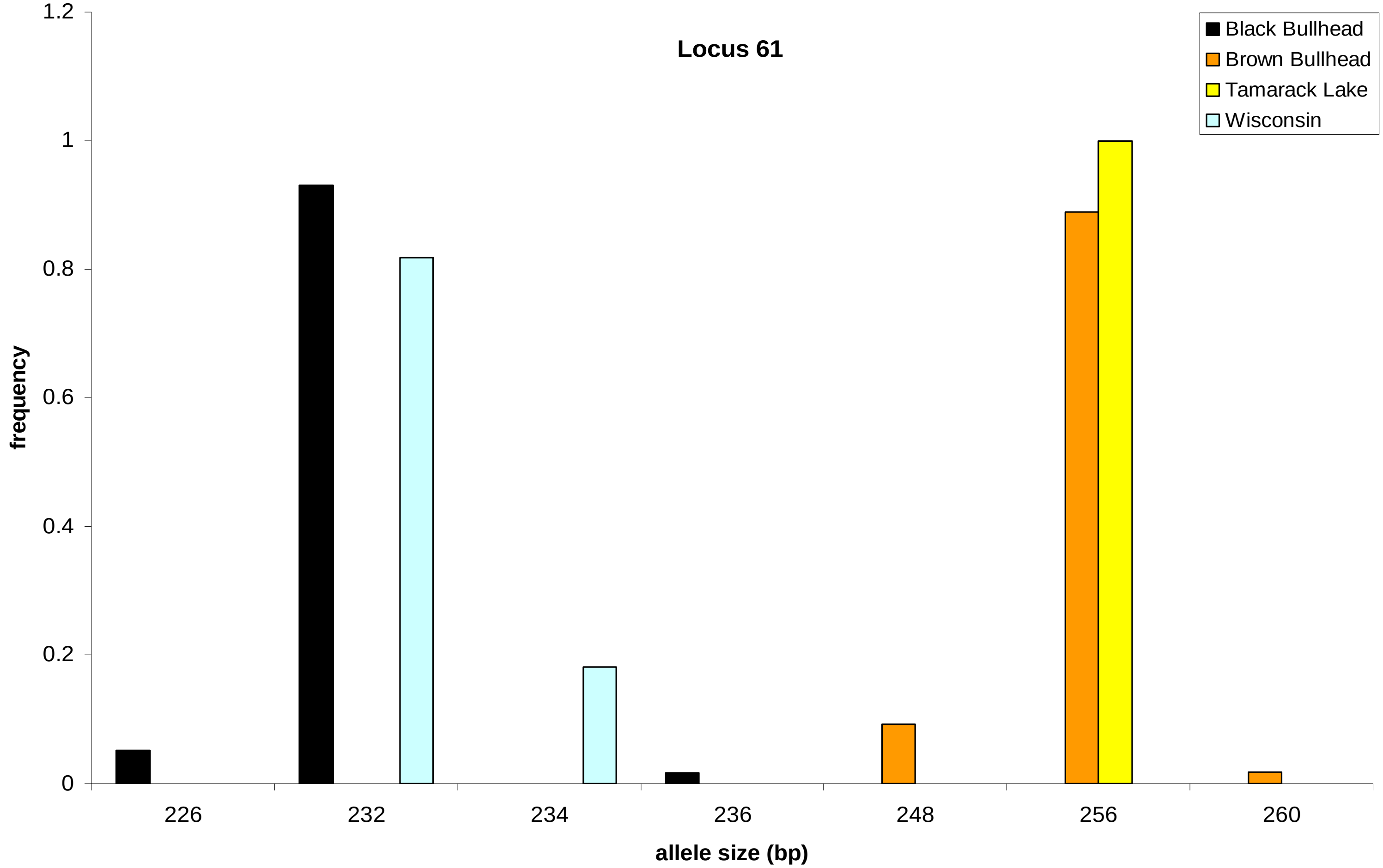

Figure D13. Allele frequencies at microsatellite locus 61 from samples of Tamarack Lake and Wisconsin populations.

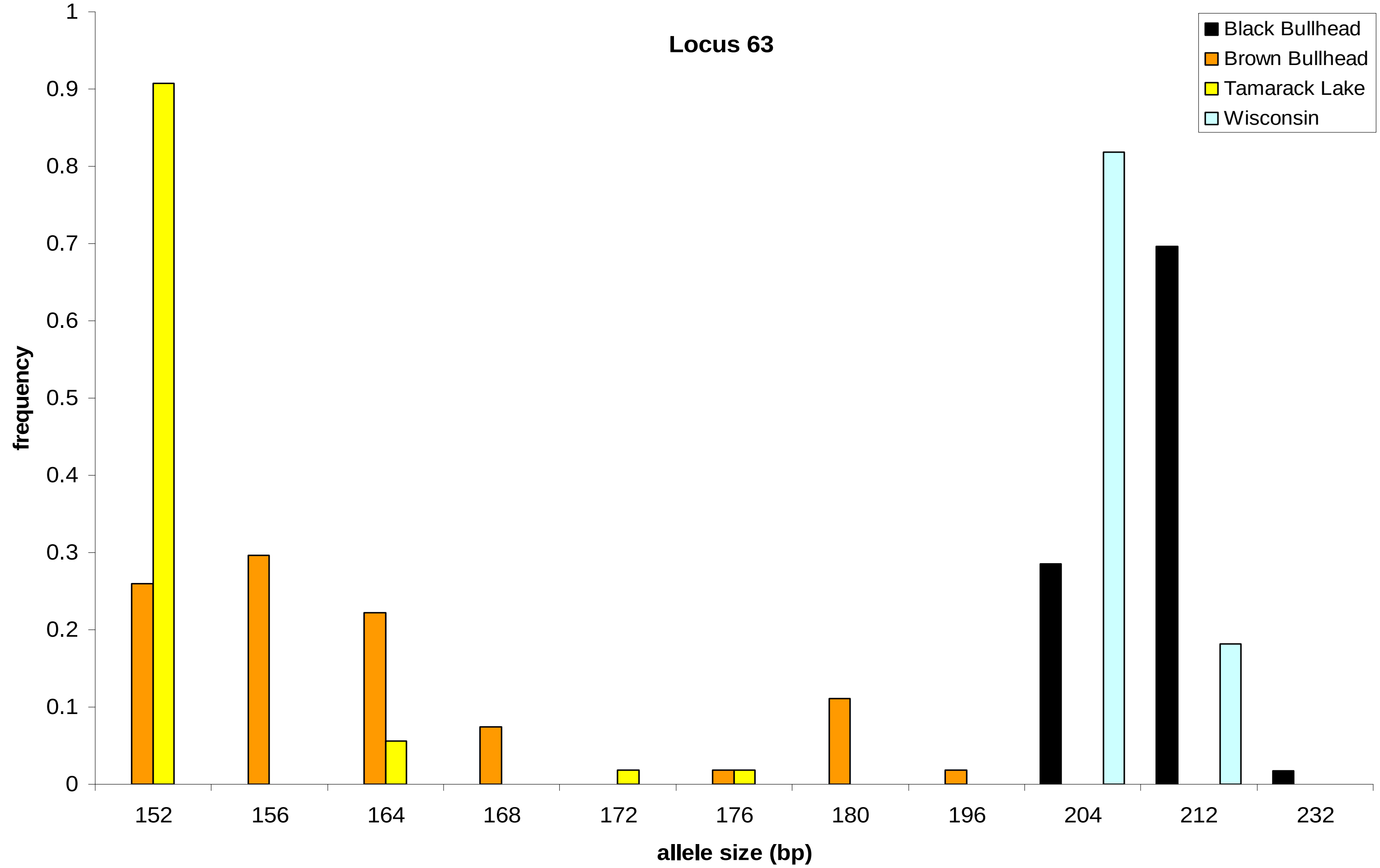

Figure D14. Allele frequencies at microsatellite locus 63 from samples of Tamarack Lake and Wisconsin populations.

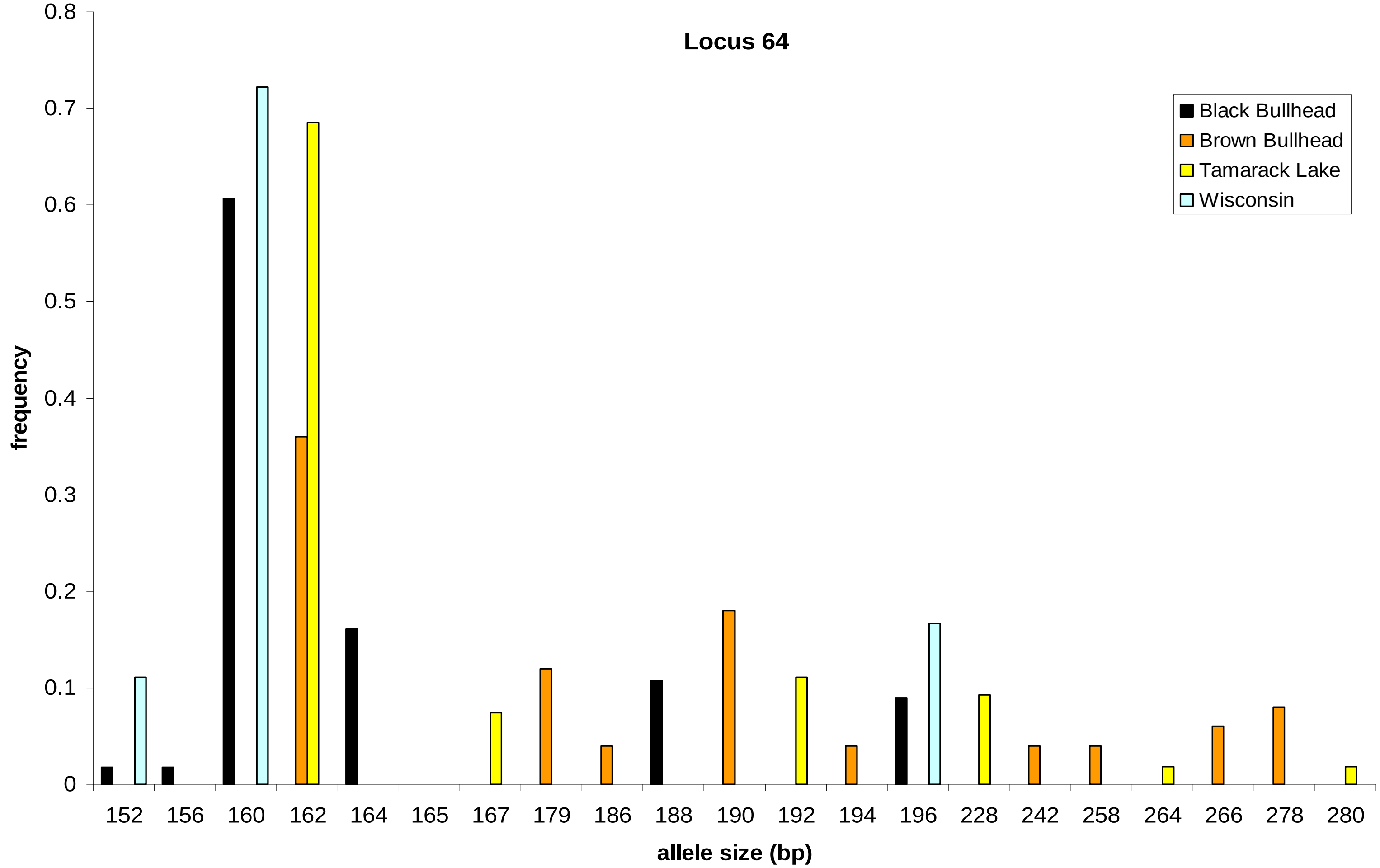

Figure D15. Allele frequencies at microsatellite locus 64 from samples of Tamarack Lake and Wisconsin populations

Appendix B

Investigating the Possible Association of Virus with Skin Papillomas in Brown Bullhead

2012

Investigating the Possible Association of Virus with Skin Papillomas in Brown Bullhead

Submitted to Sean Rafferty, Pennsylvania Sea Grant

Submitted by Iwanowicz, L.R, Iwanowicz, D.D. and Hahn C.M
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Introduction

The observation of tumors in brown bullheads is currently used as a 'Beneficial Use Impairment (BUI)' in Great Lakes Areas of Concern. An increased incidence of hepatic neoplasms is associated with PAH and other contaminant exposure in this species (Harshbarger and Clark 1990; Baumann and Harshbarger 1995; Blazer et al 2009b). Likewise skin tumors, including papillomas and squamous cell carcinomas have been used as indicators of chemical exposure in bullhead and other species (Grizzle et al. 1981; Smith et al. 1989; Black and Baumann 1991; Pinkney et al. 2001; Blazer et al 2009a). Although no cause-and-effect correlations have been made in wild populations, papillomas have been experimentally induced in brown bullhead by repeated dosing of the skin with sediment extracts with high levels of PAHs (Black et al.1985). Based on the syntax of this BUI definition there is no distinction between liver tumors and skin tumors. This BUI designation is based on the assumption that contaminant exposure is the sole inducer of tumors in this species. Interestingly, there is a weak correlation between the prevalence of liver and skin tumors (Pinkney et al. 2001; Pinkney et al. unpublished).

While there is considerable evidence that contaminants are associated with bullhead tumors, biotic factors including viruses are known to induce tumors in mammals and lower vertebrates. Viruses of the Herpesviridae and Retroviridae are associated with skin tumors in fish (reviewed in Getchell et al. 1998). In fact, Edwards and Samsonoff (1977) reported the presence of intracytoplasmic, virus-like particles via electron microscopic examination of a bullhead papilloma. This observation has not been reported since that original investigation however. Of note, many viruses that induce skin tumors are uncultureable by standard tissue culture methods. Thus, it is possible that previous investigations specific to a putative viral etiology for bullhead skin tumors may have been hindered by this characteristic. Additionally, exploration for viruses via electron microscopy is akin to searching for a needle in a haystack especially if virus titre is low. Nucleic acid-based techniques are now available that allow sequence independent amplification and screening of transcriptomes and genomes making discovery of novel genomes (viral, prokaryotic and eukaryotic) possible. Techniques developed for the specific purpose of novel virus identification are also available. (Hanson et al. 2006, Biagini et al. 2007, Nanda et al 2008).

Methods:

Sample Collection - Brown bullheads were collected from Sarah's Cove and Lagoons during late May 2010. Fish were transported to the National Fish Health Laboratory, Kearneysville, WV. Upon arrival all fish were either moribund or dead. Live fish were euthanized with a lethal dose of MS-222 and skin tumors were be excised using a sharp scalpel. Tissues were then transferred to RNA Later and stored for molecular analyses.

In addition to tumors from bullheads collected in Pennsylvania, orocutaneous neoplasms and barbel lesions were procured from an ongoing study in the Chesapeake Bay watershed (Pinkney et al. unpublished). Similarly, historic samples of walleye skin tumors infected with walleye dermal sarcoma virus (WDSV-1) were kindly provided by Dr. Paul Bowser (Cornell University) to serve as a positive control.

Sample processing – DNA and RNA were extracted from normal tissue and overt tumors using the DNeasy Tissue Kit (Qiagen) or Total RNA Kit I (Omega Bio Tek) nucleic acid purification kits respectively. RNA samples from bullheads and walleye were reverse transcribed into cDNA for the subsequent primer-extension-enrichment-reaction (PEER) method (Biagani et al 2007). Total RNA was analyzed for quality/ degradation using an Agilent 2100 Bioanalyzer. These methods have been modified in our lab, and have facilitated the successful identification of a fish Herpesvirus (channel catfish virus) and amphibian Irodovirus (frog virus-3) grown in tissue culture and in host tissue (Figure 1). Additional RNA was shipped to CoFactor Genomics for 150 bp paired end read, high throughput Illumina sequencing. Ribosomal RNA was depleted from the sample and messenger RNA was normalized using proprietary methods.

Quality Control/ Verification of WDSV-1 infection

In order to verify that control walleye skin was negative for WDSV-1 and that intact viral nucleic acid was present in the tumor tissue, a simple RT-PCR was conducted. Genomic DNA was used as template from the skin as this represents the available nucleic acids in the driver for the PEERmod. Template from the tumor was cDNA. Both templates were amplified using the primer pair WDSV-1 P3 and WDSV-1 P4 which target a 343 bp amplicon of the WDSV-1 genome (Table 1; Poulet et al. 1996). Amplification conditions were 95 °C for 5 minute for initial denaturation followed by 95 °C for 1 minute, 45 °C for 30 sec and 72 °C for 30 sec for 30 cycles. Extension of the amplicons was completed at 72 °C for 5 minutes, and the final products were chilled to 10 °C. Products were resolved in agarose.

PEER - Extracted DNA (18ul) from above was mixed with 2.5 µL 10x buffer, 1.0 µL 10mM dNTP solution, and 2.0 µL of each forward and reverse primer at 10 pM. One *driver* (DNA from control tissue) and two *testers* (cDNA and DNA each from tumors) were then processed using 10 pmol of different primers (Table 1). Samples were incubated at 94°C for 2 min and on ice for 2 min. Then2.5 units of 3'-5' exo- Klenow DNA polymerase (New England Biolabs, Ipswich, MA) was added and the samples were incubated at 37 °C for 1 h. This denaturation-annealing-elongation cycle was repeated and then enzymes were heat inactivated at 75 °C for 10 min. The protocol was then continued as written by Biagini et al. 2007.

PEERmod – A modified version of the protocol was tested using only genomic DNA as a driver. This approach in theory would allow for the detection of unique genomic DNA or expressed transcripts between the samples.

PEER products were then cloned into the pCR-XL-TOPO vector (Invitrogen) and transfected into Mach1-T1 chemically competent E. coli. Colonies were expanded in S.O.C medium and plated onto LB agar containing 50ug/ ml of kanamycin. The resulting clones were PCR amplified using M13 primers provided with the TOPO-XL cloning kit to verify the presence of inserts.

PEERmod products were cloned using a pGEM-T Easy Vector kit (Promega). Postive clones were amplified with SP6 and T7 primers. Clones were then heat lysed and plasmid DNA was harvested for direct sequencing. Sequences were identified via nucleotide blast (Blastn) or Blastx searches to the NCBI database (http://blast.ncbi.nlm.nih.gov/Blast.cgi).

Large DNA virus PCR – A number of tumorogenic viruses contain a large DNA genome. Hanson et al. (2006) developed a method that allows for the amplification of such genomes using degenerate primer sets that target the highly conserved DNA polymerase gene. Extracted DNA from pooled brown bullhead skin tumors were amplified with the primer pairs Cons Lower and Adeno, or Cons Lower and HV. Primer sequences are listed in Table 1. PCR was conducted in a Rotor-Gene Q using GoTaq qPCR mastermix. PCR used the appropriate forward primer and the consensus reverse primer (Table 1). The reaction conditions were: 93°C, 1 min for one cycle followed by 93°C, 30 sec; 45°C, 2 min; 72°C 3 min for 35 cycles followed by a single cycle at 72°C for 4 min. Product was evaluated by electrophoresis on 1.5% agarose gels. Products were cloned into the using the PGEM cloning kit as above.

Results and Discussion

Quality Control

Analysis of extracted RNA from bullhead and walleye tumors revealed low RNA quality. This was apparent by the absence of clear 18S and 28S rRNA peaks and an accumulation of small fragments in the Bioanalyzer trace (Figure 2). Given that these were the only available samples, however, we continued with the virus discovery analysis as planned. Amplification of WDSV-1 was successful from the cDNA synthesized from the tumor samples. A weak bank of the appropriate size was amplified from control DNA however (Figure 3). This indicates that WDSV-1 had integrated into the host DNA of the control fish, or that some viral replication was in progress in the absence of a tumor.

PEER

- The PEER generated less than 20 clones for bullhead samples. None of the inserts yielded sequence with homology to viruses in the NCBI database.
- Approximately 100 clones resulted from the PEER for the walleye samples. Similar to the results for the bullhead, none of the sequenced inserts shared homology to viruses in the NCBI database.

PEERmod

- The PEERmod generated approximately 100 clones for bullhead samples. None of the inserts yielded sequence with homology to viruses in the NCBI database.
- Approximately 300 clones resulted from the PEER for the walleye samples. Similar to the results for the bullhead, none of the sequenced inserts shared homology to viruses in the NCBI database.

Large DNA virus PCR

- Amplification of DNA extracted from brown bullhead skin tumors with the degenerate DNA polymerase resulted in numerous amplicons (Figure 4).
- None of these amplicons had similar homology to viruses in the NCBI database. Interestingly three brown bullhead genes were identified and included 1) toll-like receptor 3, Immunoglobulin heavy chain and the NOD1 receptor.

Illumina deep sequencing

Sequencing of brown bullhead RNA which included neoplastic tissue yielded:

Total Contigs: 13756672

Total Length: 695585772

Mean Contig Length: 50.563520886447

Median Contig Length: 28

Max Contig Length: 2246

N50 Contig Length: 74

This expression library was generated for another on-going project at the Leetown Science Center; however, we screened for viral transcripts to augment the current investigation. None of the transcripts had similar homology to viruses in the NCBI data base.

At the present time no viral sequences have been identified in brown bullhead skin tumors. Failure to identify a microbial agent does not confirm that these skin tumors are not virally induced; however, there is no evidence to support that viruses are the causative agent. Here, a number of sample quality problems likely contributed to the lack of successfully identifying a virus in these samples if, in fact, one was present. Many of the processed samples were from a study conducted in the Chesapeake Bay watershed as fish collected in Presque Isle did not survive transportation. Thus these findings, or lack thereof, do not represent bullhead skin tumors of bullhead from the Great Lakes. Also, fresh, high quality samples would likely increase the chances of identifying a microbial agent if one is present. Additionally, virally induced tumors are typically seasonal in occurrence (Getchell et al. 1998). Sampling bullhead skin tumors from fish collected during multiple seasons may increase the likelihood of capturing the window of peak viral replication. Without accounting for such possible temporal differences, the confidence that a virus is not associated is lessened.

While the methods applied here have been successful at identifying viruses in tissue culture, they are less often used in mixed host-pathogen preparations. Recently high throughput next-generation sequencing techniques and metagenomics analyses have provided an unbiased approach to pathogen discovery (Cox-Foster et al. 2007; Williamson et al. 2008; Nakamura et al. 2009). Advances in this technology have made this a viable approach to simultaneous pathogen discovery and host transciptome analysis for reasonable costs. Such approaches may be necessary to elucidate the cause(s) of bullhead skin tumors. Application of these techniques would also allow transcriptome profiling that may insight tumor associated host genes.

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Table 1. Primer sequence.

Primer Name	**Sequence**	
Driver		
PDSMART	AAGCAGTGGTAACAACGCAGAGTACGCGGG	
PDON7	AAGCAGTGGTAACAACGCAGAGTAIIIIIII	
Tester		
PT1G	ACACTCGAGGAGGTCTGGAGGGG	
PT1N7	ACACTCGAGGAGGTCTGGAGIIIIIII	
Tester II		
PT8G	AAGCAGAGGCAGCATTGGAGGG	
PT7N7	GAGCTGTGGTGAGTTGGTTGGAAIIIIIII	
Universal		
SP6	TATTTAGGTGACACTATAG	
T7	TAATACGACTCACTATAGGG	
WDSV-1 P3	TGAAGCAGGAATACCTACCT	
WDSV-1 P4	CTGTAAGTCCGTTCTCTTGT	
Cons Lower	cccgaattcagatcTCNGTRTCNCCRTA	
Adeno	gggaattctaGAYATHTGYGGNATGTAYGC	
HV	cggaattctaGAYTTYGCNWSNYTNTAYCC	
I = Inosine		

Figure 1. PEER amplification profile obtained either directly from tissue or BF2 cell culture supernatant inoculated with cell free extract from infected tissues. Final PCR results using A4/PT7 or A4/PT8 primers are shown. M: molecular weight markers. T1: negative control. T2: tumor sample taken from a frozen brown bullhead barbel. T3: tumor sample taken from a frozen brown bullhead barbel. T4: tumor tissue sample taken from a brown bullhead barbel preserved in RNAlater. T5: Unknown sample taken from the coelomic tissue of Leopard frogs (*Rana pipiens*). T6: Known Ranavirus isolate grown on BF2 cells initially inoculated with cell free extract from the American toad (*Bufo americanus*). T7: Known channel catfish virus isolate grown on BF2 cells initially inoculated with cell free extract from Brown bullhead.

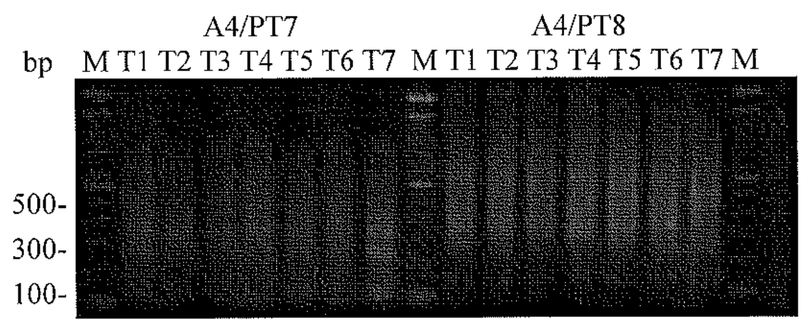


Figure 2. Bioanalyzer trace of total RNA from brown bullhead tissues. Trace indicates significant RNA degradation denoted by the lack of distinct 18S and 28S peaks and an accumulation of low molecular weight fragments.

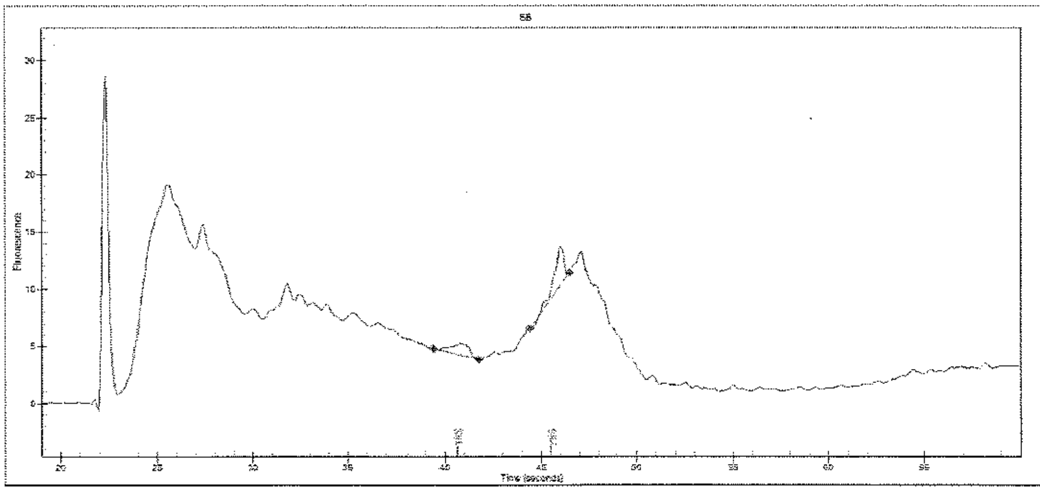


Figure 3. Amplification of the WDSV-1. DNA from control walleye skin (1) and cDNA from a WDSV-1 infected skin tumor.

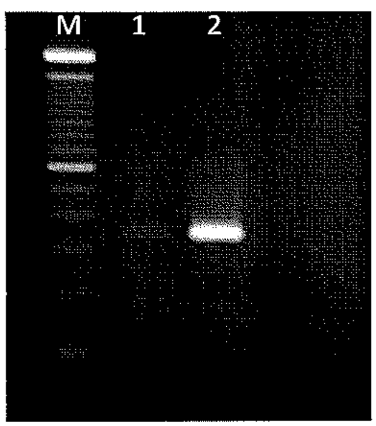
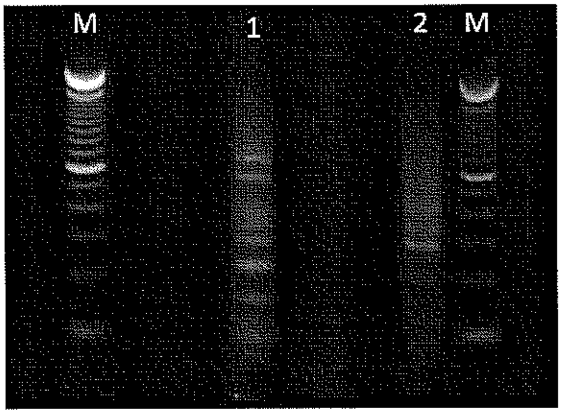


Figure 4. Amplification of DNA isolated from brown bullhead skin tumors with degenerate primers for Herpesvirus and Adenovirus DNA polymerase. Numerous amplicons generated with the HV primer (1) and Adeno primer (2).



Appendix C

Whole-sediment exposure of brown bullhead (*Ameiurus nebulsos)* to industrially contaminated sediment

July 2012

Brown bullhead *(Ameiurus nebulosus)* whole-sediment exposure experiment

Pennsylvania Department of Environmental Protection
Office of the Great Lakes

Introduction

Brown bullhead catfish (*Ameiurus nebulosus*) are a widely-used indicator of exposure to carcinogens in the environment. This is particularly true in the Great Lakes, where an elevated rate of tumors and other lesions in this species is the primary reason for the "Fish Tumors or Other Deformities" Beneficial Use Impairment in affected US Areas of Concern (IJC, 1989). The phenomenon of tumors in brown bullhead is most often attributed to exposure to polycyclic aromatic hydrocarbon (PAH) contaminants in the environment (Baumann *et al.* 1987, 1991; Baumann and Harshbarger 1995, 1998; Bunton, 2000; Harshbarger *et al.* 1984; Leadley *et al.* 1998; Pinkney *et al.* 2001, 2004a, 2004b; Smith *et al.* 1994). However, the putative relationship between sediment PAH exposure and elevated incidence rates of tumors in brown bullhead is based primarily on correlative research with this species and from inferences drawn from work with other species. Limited experimental investigation into the causal relationship between contaminated sediment exposure and fish neoplasia includes the work of Black (1983), Black et al. (1985), Grady et al. (1992b), and Stoker et al. (1985).

In contrast to the evidence suggesting an association between PAH-contaminated sediment exposure and neoplasia in brown bullhead, a number of investigators have reported elevated incidences of tumors and other lesions in this species from uncontaminated sites. Poulet et al. (1994) examined 17 waterbodies in New York State and failed to find a consistent relationship between the presence of chemical carcinogens and orocutaneous neoplasms in bullhead. These authors noted that orocutaneous neoplasms appear to be the leading neoplastic pathologic condition of wild brown bullhead populations. Spitzbergen and Wolfe (1995) similarly failed to find a relationship between hepatic neoplasia and environmental contamination in nine New York State waterbodies, speculating that liver neoplasms in these fish may instead be caused by naturally occurring carcinogens such as nitrosamines or radon gas. Pinkney and Harshbarger (2005) found highly elevated incidences of skin neoplasms (53%) and liver neoplasms (20%) in bullhead in the South River—a tributary of the Chesapeake Bay near Annapolis Maryland. However, total PAH levels in this river (2.2 ppm) are similar to the Tuckahoe River (1.8 ppm)—a reference site used by Pinkney for comparison to PAH-contaminated tributaries throughout the Chesapeake Bay (Pinkney *et al.* 2001, 2004a, 2004b).

The purpose of this study was to conduct an experimental investigation into the relationship between exposure to PAH-contaminated sediment and neoplasia in brown bullhead (*Ameiurus nebulosus*). A laboratory microcosm approach was used to expose brown bullhead to contaminated whole sediment.

Methods

Experimental Specimens

Juvenile brown bullhead (N=66; 63-143 mm total length) were obtained from established research stock from the United States Geological Survey's Leetown Science Center in Kearneysville, West Virginia on 4 December 2008 and transferred to a 400 gallon circular tank with recirculating dechlorinated water at the Tom Ridge Environmental Center (TREC). Specimens were acclimated for 68 d prior to random assignment to experimental condition. As a scaleless species, baseline age determination was not possible without injuring the specimens. However, the sizes of the specimens were consistent with bullhead in the 0-1 year age range. Otoliths were obtained from all specimens for aging at the termination of the experiment.

Sediment

Sediment was collected from Presque Isle Bay, Erie County, PA (treatment condition) and Canadohta Lake, Crawford County, PA (sediment control condition) using a stainless steel van Veen grab sampler. Past sediment sampling (e.g., PADEP 2006) had revealed Presque Isle Bay sediment to be contaminated by moderate levels of various heavy metals and organic compounds including PAHs. No prior data were available on the sediment quality of Canadohta Lake. However, past bullhead samplings had consistently revealed a healthy population with low levels of neoplasms and other deformities. Given the putative causal relationship between sediment contamination and bullhead neoplasia, Canadohta Lake sediment was deemed to be appropriate for experimental control purposes.

Past sampling results were consulted in an effort to obtain the most highly contaminated sediment possible from Presque Isle Bay. Equal volumes of sediment were collected from one "shallow" site (mouth of Mill Creek; approximately 1.2 m) and one "deep" site (near center of bay; approximately 7.6 m) known from past samplings to contain elevated levels of PAHs, homogenized into a single sample, and placed in washed polyethylene storage containers. Matching "shallow" and "deep" sites (in terms of depth) were selected from Canadohta Lake and similarly homogenized into a single sample. A subsample of sediment from each site was placed in a 500 ml amber glass container and sent to the Department of Environmental Protection's laboratory for analysis of metals and a suite of semi-volatile organic compounds. The remainder of the sediment was stored at 4 °C prior to use in the exposure study.

Equal volumes of sediment were added to 14, 20-gallon aquaria (24" x 12" x 16") to a depth of approximately 10 cm and the aquaria were filled to within 3 cm of the top with dechlorinated water. Seven aquaria were filled with sediment from Presque Isle Bay and seven with sediment from Canadohta Lake. Two additional aquaria were filled with dechlorinated water only and served as water-only controls. Sponge filters with aerators (for nitrification and maintenance of dissolved oxygen) were added to all aquaria and were powered using two Super Luft air pumps. Aquaria were randomly assigned numbers ranging from 1-16 and placed in sequential order on shelving in the TREC Aquatics Lab, resulting in a randomized block design. No identifying markings were placed on the tanks other than the assigned number in order to keep research assistants "blind" with respect to treatment condition.

Experimental Design

Bullhead (N=56) were randomly assigned to either the Presque Isle Bay sediment or Canadohta Lake sediment conditions on 10 February 2009. Specimen health, total length, and weight were assessed prior to assignment following the methods in Rafferty and Grazio (2007). In order to evenly match the biomass of bullhead assigned each sediment condition, specimens were sequentially assigned to each aquarium beginning with the smallest specimens (based on total length) and ending with the largest specimens. The remaining largest bullhead (N=10) were randomly assigned to the two water-only control conditions. Student's t test analysis confirmed that there were no significant differences between specimens assigned to the Presque Isle Bay and Canadohta Lake sediment conditions in terms of mean total length (100.86 mm v. 99.36 mm, respectively; $p>.05$) or mean weight (11.55 g in each condition; $p>.05$). Bullhead assigned to the water-only control conditions were both longer ($\overline{x}$ =127 mm, $p<.05$) and heavier ($\overline{x}$ =23.07 g, $p<.05$) than the specimens in the sediment-containing conditions.

Bullhead were fed sinking shrimp and plankton pellets (Aquadine Nutritional System) at a rate of 1% of body weight per day. Water quality parameters were regularly monitored using a YSI 556 multi-parameter meter. Aquaria were cleaned on a weekly basis by scraping the walls with a squeegee. The squeegee was rinsed in dechlorinated water between tanks to avoid cross contamination. Dechlorinated water was added to the aquaria on a weekly basis in order to maintain the original volume of water. In order to minimize sediment loss, sediment was gently brushed away from the sponge filters and allowed to settle back into the tank. Airstones were replaced as indicated by assessment of dissolved oxygen levels and observation of airstone performance. Research assistants were kept blind with respect to sediment treatment condition.

Bullhead were collected from the aquaria on 28 May 2009, 13 August 2009, 10 March 2010, and 20 August 2010, weighed, measured, and grossly examined according to Rafferty and Grazio (2007). On 13 August 2009 and 10 March 2010, the most massive bullhead from each tank was prepared for necropsy by severing the spinal chord. Livers were excised along with attached gall bladders and grossly examined for the presence of lesions and parasites. Gall bladders were then excised from the liver, discarded, and the hepatic tissue was weighed. Hepatic tissue was then dissected for subsequent histopathological analysis and analysis of DNA adducts. Histopathology samples were preserved in 10% neutral buffered formalin and archived for future analysis. DNA adduct samples were placed in cryovials, flash frozen in liquid nitrogen, and then transferred to a -80C freezer for future analysis. Otoliths and pectoral spines were collected and stored in labeled coin envelopes for future specimen aging. All bullhead not subjected to necropsy were returned to their respective aquaria unharmed.

The experiment was terminated on 20 August 2010 at which point all remaining specimens were necropsied as above. In order to determine changes in sediment chemistry over the course of the experiment, equal volumes of sediment were collected from each of the Presque Isle Bay and Canadohta Lake tanks following the termination of the experiment, composited, homogenized, and submitted for chemical analysis as described above.

Biomarker Analyses

Environmental carcinogensis in fish is conceptualized as a multi-stage process that takes years from the exposure to the initiating carcinogen to the progression to diagnosable neoplasms. In order to detect the earliest possible onset of carcinogenesis, DNA adduct assays were employed to detect the binding if the initiating carcinogen with hepatocellular DNA and histopathological analyses were conducted to detect neoplastic and/or pre-neoplastic lesions that would not yet be visible to the naked eye.

<u>DNA Adduct analyses</u>

DNA adduct analyses were conducted at the Texas A&M University's Center for Cancer and Stem Cell Biology using the nuclease P1-enhanced bisphosphate version of the 32P-postlabeling method (Philips and Arlt, 2007; Reddy MV and Randerath, 1986). Briefly, DNA (10 µg) was enzymatically degraded to normal (Np) and adducted (Xp) deoxyribonucleoside 3'-monophosphates with micrococcal nuclease and spleen phosphodiesterase at pH 6.0 and was incubated at 37°C for 3.5 h. After treatment of the mixture with nuclease P1 to convert normal nucleotides to nucleosides, adducted nucleotides (Xp) were converted to 5'-32P-labeled deoxyribonucleoside 3',5'-bisphosphates (pXp) by incubation with carrier-free [γ-32P]ATP and polynucleotide kinase. Radioactively labeled modified nucleotides were mapped by multidirectional anion-exchange thin-layer chromatography (TLC) on polyethyleneimine (PEI)-cellulose sheets (Mabon et al., 1996). Radioactively labeled products were purified and partially resolved by one-dimentional development overnight with solvent 2.3 M sodium phosphate, pH 5.7 (D1). Bulky labeled DNA adducts retained in the lower (2.8 x 1.0 cm) part of the D1 chromatogram were, after brief autoradiography on Kodak X-OMAT LS X-ray film, contact-transferred to fresh thin-layer sheets and resolved by two-dimensional TLC. The bulky DNA adducts were separated with with 3.82 M lithium formate, 6.75 M urea, pH 3.35 and 0.72 M sodium phosphate, 0.45 M Tris–HCl, 7.65 M urea, pH 8.2 in the first (D3) and second (D4) dimensions, respectively. 32P-labeled I-compounds were visualized by screen-enhanced autoradiography at -80°C using Kodak BioMax XAR film or with the aid of a Packard Instruments InstantImager (Zhou et al., 1999).

Quantitative analyses were conducted by first determining radioactivities of TLC fractions from individual animals with the aid of the Instant Imager. The extent of covalent DNA adducts was estimated by calculating Relative Adduct Labeling (RAL) values from sample count rates, the amount of DNA assayed (expressed as pmol DNA monomer units or DNA-P), and the specific activity of [γ-32P]ATP according to Reddy MV and Randerath (1986).

$$RAL = \frac{\text{DNA adduct(s) [cpm]}}{\text{DNA-P [pmol]} \times \text{Spec.act.}_{ATP}\ \text{[cpm/pmol]}}$$

Histopathology Methods

Preserved liver tissues were analyzed following the methods of Blazer et al., 2006. Briefly, tissues are cut under a chemical fume hood as to pieces no bigger than 4 cm and placed in standard cassettes. Cassettes are rinsed in 70% ethanol or tap water for 1-2 hours and placed in the Autotechnicon for automatic processing (dehydration and infiltration with paraffin). Parrafin blocks are cut on either a standard or automatic rotary microtome equipped with disposable blades into 5 μm thick sections. Sections are examined for quality of cut placed on a properly labeled microscope slide. All slides are routinely stained with hematoxylin and eosin (H&E), covered with a cover slip, and examined for pathologic lesions by a minimum of two trained pathologists.

Aging Methods

[Reserved, in progress but not available as of printing date]

Results

Sediment Analyses

Presque Isle Bay and Canadohta Lake sediment differed in terms of solids and moisture content, with the former being comprised by a greater proportion of solids and the latter having greater moisture content (Table 1). Particle size analysis was not conducted. However, Presque Isle Bay sediment had visibly higher levels of sand as a result of contributions from the shallow Mill Creek site while the shallow Canadohta Lake sample had a higher proportion of woody detritus. The deeper sediment samples from both waterbodies were dominated by organic muck.

Presque Isle Bay sediment had higher levels of 10 of 13 detectable chemical compounds at the inception of the experiment. Most notably, the PAHs chrysene, benzo(b)fluoranthene, pyrene, benz(a)anthracene, phenanthrene, benzo(g,h,i)perylene, benzo(a)pyrene, and fluoranthene were all present in Presque Isle bay sediment. The total concentration of detected PAHs was 19.41 mg/kg and individual PAH compounds exceeded Probable Effects Concentrations (PECs)--the level above which harmful effects are likely to be observed (MacDonald et al., 2000). The only PAH detected in the Canadohta Lake sediment was benzo(a)pyrene. This chemical was present at low levels comparable to those found in Presque Isle Bay (1.54 v. 1.49 mg/kg, respectively). Arsenic levels were over three times higher in the Canadohta Lake sediment, however (31.0 mg/kg v. 9.55 mg/kg, respectively). All detected chemical compounds from both sites are listed in Table 1.

Sediment PAH levels were again analyzed after the termination of the experiment in order to better understand losses of or changes in these compounds over the course of the experiment. In general, sediment PAH composition in both treatment conditions remained remarkably consistent over time. With respect to Presque Isle Bay sediment, 18.14 mg/kg of detected PAHs remained at the termination of the experiment. Benzo(g,h,i)perylene was present at low levels (1.17 mg/kg) in the initial sample but was not detected in the post-exposure sample. Benzo(k)fluoranthene was not detected in the initial sample but was found at a level of 2.18 mg/kg in the post-exposure sample. However, this PAH is difficult to resolve from benzo(b)fluoranthene in chromatograms and may have been reported as all benzo(b)fluoranthene in the original analysis (J. Black, personal communication). With respect to Canadohta Lake sediment, benzo(a)pyrene was not detected in the post-exposure samples. Other chemical changes were *de minimis*.

Bullhead Assessment

Behavioral Observations

Turbid conditions in the aquaria made ethological observations challenging in the sediment-containing tanks. In general, bullhead spent the majority of time in close contact with or buried in the tank sediment. After several days of acclimation to experimental condition, bullhead exhibited a vigorous feeding response. Bullhead actively routed in the tank sediment for sunken food pellets and quickly ingested daily rations. Bullhead appeared to become conditioned to daily feeding regimes over time and actively patrolled their aquaria near the front glass in the presence of researchers. Eight specimens died prior to the termination of the experiment as described below. No specimens displayed obvious signs of morbidity prior to death.

Growth and Condition Factor

Bullhead in all conditions appeared to thrive based on mean increases in total length and body weight (Figures 1 and 2). Overall mean body mass increased nearly four-fold from 13.30 (± 0.73) g to 49.59 (± 3.06) g. Overall mean total length increased from 104.18 (± 1.95) mm to 157.15 (± 3.35) mm. There were no significant differences in length, weight, or condition factor between bullhead exposed to Presque Isle Bay sediment compared to those exposed to Canadohta Lake sediment over the course of the experiment $\alpha = 0.05$ level. By design, bullhead in both sediment conditions were significantly shorter and less massive than bullhead in the water-only control condition at the inception of the experiment, but these differences were no longer significant beginning with the 13 August 2009 observations (ANOVA, $p > 0.05$). Mean condition factor (K) remained > 1.0 in all conditions throughout the experiment, although K of the bullhead in the water-only control condition was significantly lower than the sediment conditions on 28 May 2009 (Student's t, $p=0.002$; Figure 3). Condition factor subsequently improved in the water-only controls and was not significantly different (Student's t, $p>0.05$) than the sediment conditions on other observation dates. After 556 days of exposure, the mean total lengths of bullhead in the Presque Isle Bay and Canadohta Lake conditions were 156.58 ± 6.53 mm and 152.0 ± 9.56 mm, respectively. The mean weights of bullhead in these conditions were 46.84 ± 7.06 g and 44.88 ±8.08 g. The three remaining water-only control specimens had a mean total length of 178.33 ± 13.87 mm and a mean weight of 77.83 ± 19.58 g at the end of the experiment.

Necropsy and Hepatosomatic Index

Of the 58 bullhead surviving to the date of assigned necropsy, 19 were male, 38 were female, and the gender of one specimen was unable to be determined. Necropsy revealed that all of these fish were sexually immature based on visual observation of gonads. Bullhead in all experimental conditions had a heavy burden of trematode parasites in their liver tissue. White nodules were also noted on the liver tissue of four necropsied specimens in each of the sediment conditions and in both necropsied specimens form the water-only control tanks (Figure 4). The mean HSI values for the Presque Isle Bay, Canadohta Lake Sediment Control, and Water Control fish were 0.040 (±0.002), 0.037 (±0.004), and 0.035 (±0.004), respectively. These differences were not significant among all three conditions (ANOVA, $p=0.68$) or between the sediment conditions (Student's t-test, $p=0.63$). The liver in one fish from the Presque Isle Bay sediment conditions was notably pale and soft, tearing upon excision. The conditions of other internal organs were unremarkable.

Histopathology

No raised orocutaneous lesions (putative neoplasms) developed on any of the experimental fish. Therefore, none were available for histopathological analysis. Liver tissues from 53 fish were examined for pathological conditions following the methods of Blazer et al. (2009). A single liver neoplasm (a hepatic adenoma) was diagnosed in a specimen in the Canadohta Lake sediment control condition. Two specimens from both

Canadohta Lake and Presque Isle Bay sediment conditions developed altered foci (a potentially pre-neoplastic lesion. Livers from all specimens were found to be heavily parasitized by trematodes. However, there was no difference among conditions with respect to parasite levels, cellular inflammation, cellular necrosis, or fibrosis ($F(2,55)$, $P>0.05$).

DNA Adducts

Typical radiograms are provided in Figure 5. No DNA-PAH adducts were detected and no significant differences in Relative Adduct Labeling (RAL) values were found among conditions. RALs in all conditions were found to increase with increasing exposure duration and there was a significant increase in polar adduct formation in all treatment conditions (including the water-only control groups) by the end of the experiment ($F(2,55)$, $p<0.0001$)

Discussion

Sediment analyses revealed that Presque Isle Bay sediment had higher concentrations of seven of eight detected PAHs than did sediment from Canadohta Lake. The exception was benzo(a)pyrene, a known carcinogen (Grady et al., 1992b), where both sites had levels in the 1.5 mg/kg range. It is noteworthy that all PAH compounds, when detected, exceeded PECs (MacDonald et al., 2000). None of the metal results exceeded PECs, although Canadohta Lake arsenic levels (31.0 mg/kg) approached the 33.0 mg/kg threshold. In general, sediment chemistry analyses confirmed that Presque Isle Bay sediment is contaminated with moderate levels of PAHs and that Canadohta Lake sediment was appropriate for experimental control.

The 556-d exposure to Presque Isle Bay sediment had no grossly visible adverse effects on brown bullhead, a result consistent with the 28-d exposure study by Grady et al. (1992a) (despite the author's assertion that gill inflammation represents a severe and profound lesion). From a regulatory standpoint, the critical dependent variable in this study is the development of neoplasms. None of the bullhead developed raised external lesions suggestive of hyperplasms or neoplasms. Livers of specimens in all conditions had a heavy parasite burden, but this level did not vary among conditions.

Biometrics of growth and condition factor were similarly unremarkable. Bullhead in all experimental conditions appeared to thrive throughout the first 184 days of the experiment based on increases in both total length and body weight. The general condition factor (K) indicated that fish were robust and able to add adequate biomass under all experimental conditions. The relative decrease in K in bullhead in the water-only controls on 28 May 2009 was unexpected, but may have been related to stress related to the inability to burrow into sediment for cover. Still, no differences were found among experimental conditions in terms of total length or body weight. Hepatosomatic index (HSI) values in the present study are somewhat higher than those reported by other investigators. For example, Baumann *et al.* (1991) reported a mean HSI for brown bullheads ranging from 0.020 – 0.022 in sites where no bullhead liver tumors were detected (Menominee River and Fox River, respectively) to 0.026 from two sites with elevated liver tumor rates (Cuyahoga River and Munuscong Lake). In general, surveys have found the incidence of tumors to be positively correlated with HSI values (Pinkney et al., 2004b). The higher HSI values reported in this study may in fact reflect the presence of visible nodules (potential neoplasms) in hepatic tissue. Alternately, the relatively large HSI values may reflect the notable trematode biomass parasitizing the bullhead livers. The presence of trematode parasites in the water-only controls suggests that the fish were infected with parasites prior to the initiation of the experiment. In any case, there were no significant differences in HSI values among conditions.

Biomarkers of early-stage carcinogenesis (histopathology and DNA adducts) were negative. A single neoplastic lesion (a hepatic adenoma) was found on a single specimen assigned to the Canadohta Lake control sediment condition. The white liver nodules noted during necropsy were determined to be encysted trematode parasites upon histopathological examination analysis and there was no difference in this parasite burden among treatment conditions. No PAH-DNA adducts formed in any experimental condition, providing further evidence that the PAH carcinogens present in Presque Isle Sediment are not bioavailable to bullhead. Low levels of polar DNA adducts were detected in fish upon termination of the experiment but, again, there was no difference among conditions.

It is important to acknowledge the limitations of this research and temper inferences accordingly. While this study represents the longest reported experimental exposure of bullhead to contaminated whole sediment, the lack of external neoplasms could simply reflect insufficient exposure duration or missing a critical window for the exposure of juvenile bullhead (bullhead aging is not yet complete). Carcinogenesis in fishes, as in mammals, in widely conceptualized as a multi-stage process involving initiation, promotion, and progression (Baumann and Okihiro 2000; Bunton 2000; Groff 2005). Therefore, it is possible that additional time is needed for the manifestation of neoplasms. In addition, it is possible that promoting chemicals are present in the Presque Isle Bay water column that were not present in the dechlorinated water used in the experiment. Finally, by design, the contaminant biomagnification pathway was not formally assessed. Still, a rich oligochaete infauna was noted in all sediment-containing tanks at the termination of the experiments and it is probable that bullhead were ingesting these organisms in addition to the food rations provided by the researcher.

The above considerations not withstanding, this work strongly suggests that simple exposure to Presque Isle Bay sediment is not the cause of the tumors and other deformities seen in the brown bullhead population.

Tables

Analyte	Probable Effects Concentration (PEC)	Presque Isle Bay t_0	Presque Isle Bay t_1	Canadohta Lake t_0	Canadohta Lake t_1
MOISTURE	N/A	55.30 %	-	84.01 %	-
SOLIDS	N/A	44.70 %	-	15.99 %	-
Co, dry wt	N/A	9.20	-	ND	-
Cd, dry wt	4.98	2.56	-	ND	-
As, dry wt	33	9.55	-	31.0	-
Ni, dry wt	48.6	N/A	-	26.8	-
Pb, dry wt	128	66.8	-	56.2	-
Chrysene	1.29	2.15	1.75	ND	ND
Benzo(b)fluoranthene	N/A	4.24	2.90	ND	ND
Benzo(k)fluoranthene	N/A	ND	2.18	ND	ND
Pyrene	1.52	3.15	3.04	ND	ND
Benz(a)anthracene	1.05	1.86	1.45	ND	ND
Phenanthrene	1.17	1.52	1.70	ND	ND
Benzo(g,h,i)perylene	N/A	1.17	ND	ND	ND
Benzo(a)pyrene	1.45	1.49	1.51	1.54	ND
Fluoranthene	2.23	3.83	3.61	ND	ND
Sum of detected PAHs		19.41	18.14	1.54	0

Table 1. Sediment chemistry at initiation (**t_0**) and termination (**t_1**) of experiment. PEC based on MacDonald et al. 2000. Results reported as mg/kg unless otherwise noted. N/A= Not Available. ND= Not detected during analysis. The "-" symbol means the test was not conducted.

Type of Abnormality	Presque Isle Bay (N=26)	Canadohta Lake (N=25)	Water Control (N=7)
Missing nasal barbel	1	3	
Short nasal barbel	1	2	2
Forked nasal barbel	4		
Dark pigmentation on barbel	5	1	1
Dark pigmentation on body	1		3
White cyst	1	2	
Raised red lesion		2	1
Broken pectoral spine			
Spinal cord deformity	1		
Stunted	2	3	
Hemorrhaging	4	6	
Ulcer	3	2	

Table 2. Summary of bullhead abnormalities.

FIGURES

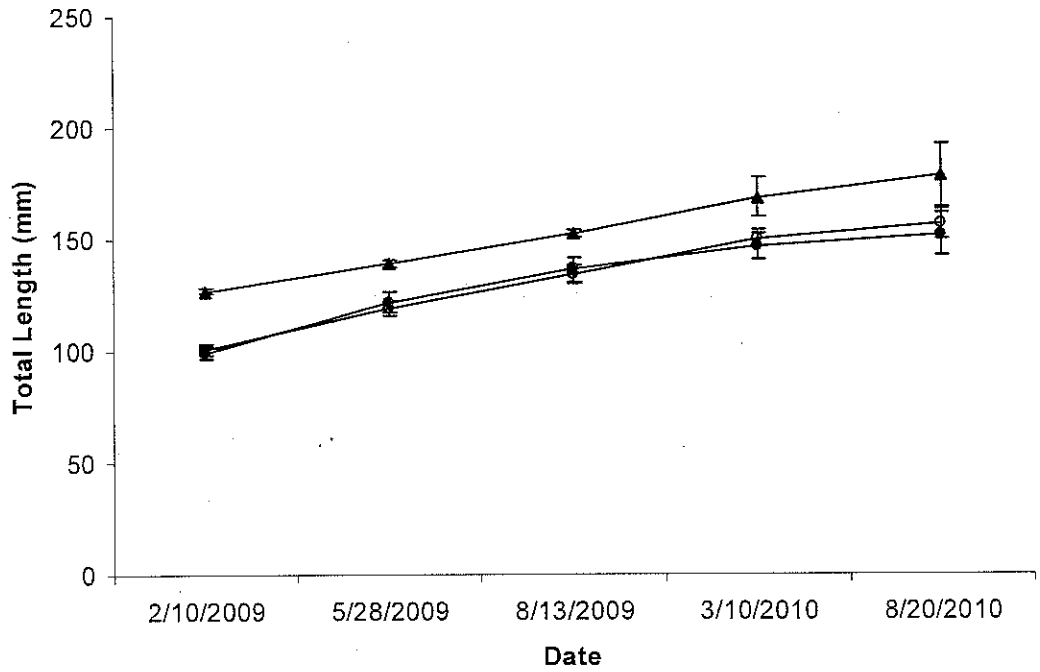


Figure 1. Mean total body length (mm) of bullhead over time in Presque Isle Bay (open circles), Canadohta Lake Control Sediment (closed circles) and Water Control (closed triangles) conditions. Bars indicate ± SEM.

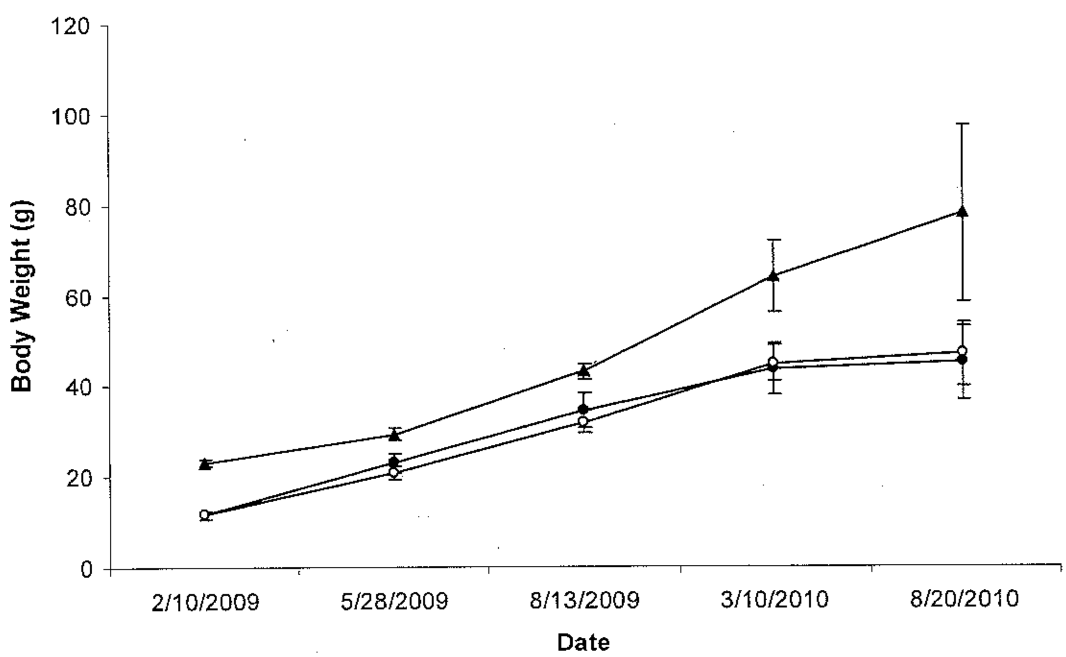


Figure 2. Mean body weight (g) of bullhead over time in Presque Isle Bay (open circles), Canadohta Lake Control Sediment (closed circles) and Water Control (closed triangles) conditions. Bars indicate ± SEM.

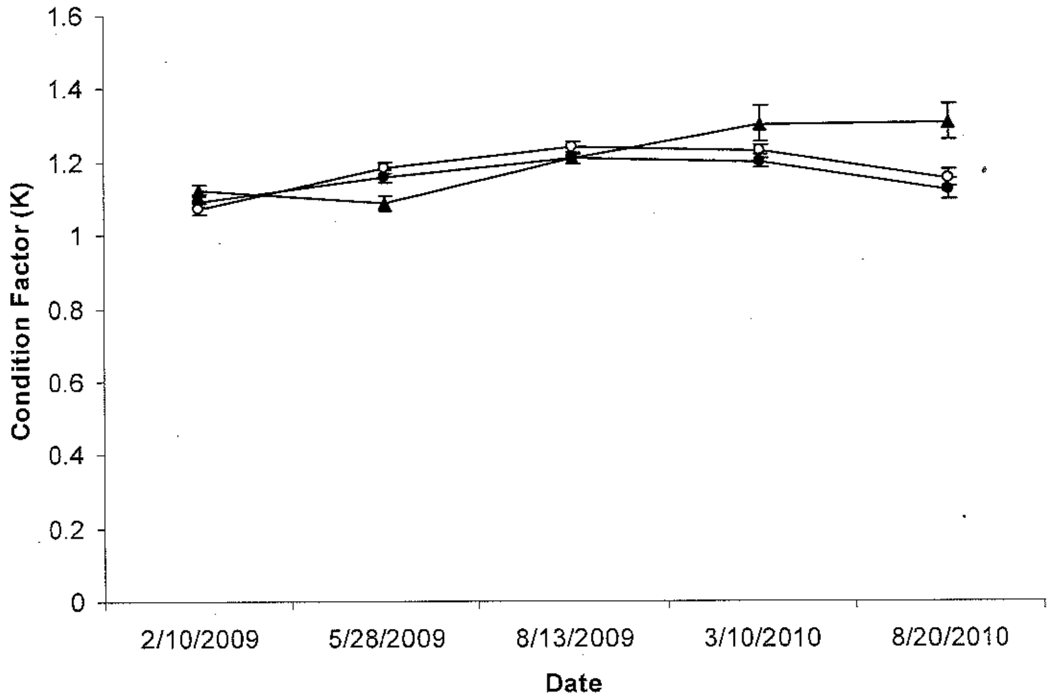


Figure 3. Condition Factor (K) of bullhead over time in Presque Isle Bay (open circles), Canadohta Lake Control Sediment (closed circles) and Water Control (closed triangles) conditions. Bars indicate ± SEM.

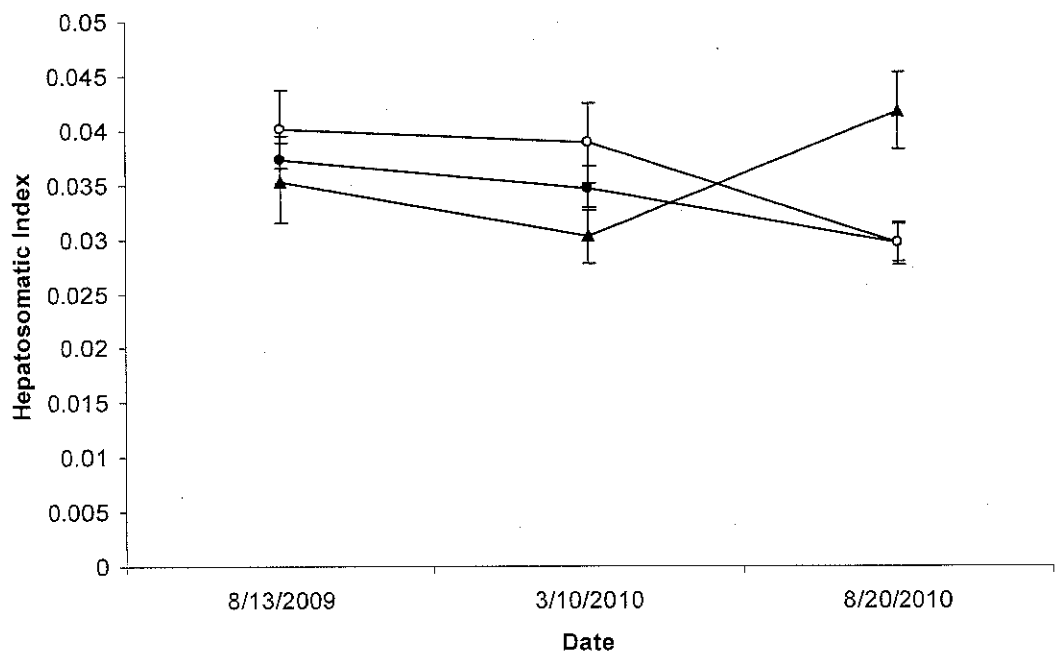


Figure 5. Hepatosomatic Index over time in Presque Isle Bay (open circles), Canadohta Lake Control Sediment (closed circles) and Water Control (closed triangles) conditions. Bars indicate ± SEM .

Figure 4. Brown bullhead liver with grossly visible white nodule and trematode parasites.

8/13/2009 3/10/2010 8/20/2010

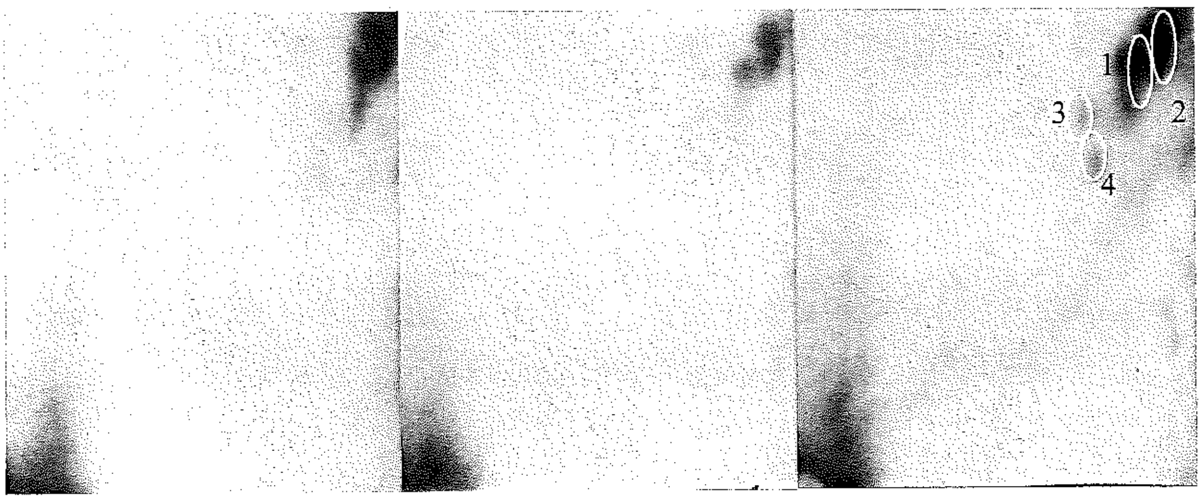


Figure 5. Typical profiles of DNA adducts in liver of Brown Bullhead Catfish. Adduct Types are denoted in Figure 5(c).

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Presque Isle Bay Area of Concern

Screening-Level

Ecological Risk Assessment

Erie, Pennsylvania

Prepared for:
Pennsylvania Department of Environmental Protection
Jim Grazio and Lori Boughton

7-9-2012

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LIST OF ABBREVIATIONS

AOC	Area of Concern
BUI	Beneficial Use Impairment
COPCs	chemicals of potential concern
CSM	conceptual site model
CSOs	combined sewer overflows
ERA	ecological risk assessment
ESB-TUs	equilibrium sediment benchmark- toxic unit
GLNPO	Great Lakes National Program Office
GLWQA	Great Lakes Water Quality Agreement
HQ	hazard quotient
IJC	International Joint Commission
LOAEL	lowest observable adverse effects level
mg/kg/d	milligram chemical per kilogram body weight per day
ND	Not Detected
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observable adverse effects level
PA DEP	Pennsylvania Department of Environmental Protection
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
PEC	probable effects concentration
PEC-Q	probable effects concentration quotient
PIB	Presque Isle Bay
SEM AVS	simultaneously extracted metals/ acid volatile sulfides
SEM-AVS/foc	simultaneously extracted metals/ acid volatile sulfides/ fraction of organic carbon content of sediment
SLERA	screening-level ecological risk assessment
SSOs	Sanitary sewer overflows
SQV	sediment quality value
TECs	toxic effects concentration
TOC	total organic carbon
TRV	toxicity reference value
USEPA	U.S. Environmental Protection Agency

EXECUTIVE SUMMARY

In 1991, Presque Isle Bay (PIB) became the 43rd and final Area of Concern (AOC) listed under the Great Lakes Water Quality Agreement. The listing was primarily driven by observations of external fish tumors on bullhead collected within the Bay. Unfortunately, addressing the fish tumors or other deformities BUI directly within the PIB AOC, as well as AOCs across the Great Lakes has proved to be challenging. The scientific understanding of the cause and effect relationships for fish tumors is complicated and confounding, and there is a lack of specific assignments of control sites, and lack of clear definitions of tumor types and background rates. As such, the data collected on fish tumors have created more questions than answers for assessing fish tumor conditions. This lack of understanding the cause-effect relationship between legacy contaminants and fish tumors has complicated the ability of AOC partners and researchers to define attainable targets for this BUI. Thus, PA DEP and its partners have opted to use the ecological risk assessment approach to evaluate contaminant risks to the PIB ecosystem. The presence and frequency of tumor occurrence is one line of evidence in the assessment of risks to fish. The ERA was designed to address the following question, originally posed by Diz (2002):

Do legacy contaminants (contaminants of potential concern) continue to pose a risk to ecosystem receptors within Presque Isle Bay?

A screening-level ecological risk assessment (SLERA) was prepared following the using the ecological risk assessment (ERA) approaches for Presque Isle Bay (PIB). The SLERA used existing collected data and combined the findings of previous studies with SLERA evaluations to understand the potential risks that concentrations of the contaminants of potential concern (COPCs) pose to the ecosystem. For the purpose of the PIB SLERA, evaluations focused on ecological components most likely affected by sediments containing COPCs. The ecological evaluations within PIB were represented by the stakeholder-developed ecosystem objectives, supporting questions and attainment targets (PA DEP 2006):

- Maintain and protect the benthic invertebrate community
- Maintain a quality fishery
- Protect and improve the near-shore habitat (to support aquatic-dependent wildlife)

The evaluation of the target objectives conducted for this SLERA was conducted using the available data to establish a weight of evidence examining the risk to ecosystem receptors. The weight of evidence concluded:

1) Surface sediment COPCs appear to be the primary chemical stressor in this system, although habitat (substrate) and invasive species may be additional stressors on the ecological community that may be challenging to tease apart.

2) The potential risk of COPC exposure benthic invertebrates across PIB are generally low based on whole sediment toxicity tests. Isolated areas may pose a moderate to high risk of exposure.

3) Benthic invertebrate exposure risk has decreased through time and are generally meeting toxicity targets.

4) The probable effect concentration (PEC) targets are generally met across PIB for most COPCs. Exceedences do occur for metals like barium and cadmium and for some PAHs. Studies focused on high concentration areas tend to exceed PEC in most cases but skew the baywide results.

5) Metals bioavailability across the PIB appears to be decreasing through time, with recent samples meeting low toxicity thresholds.

6) The quality fishery objective within PIB are supported by good water quality, a low risk of prey base (benthic invertebrates) exposure to COPCs, and fish tissue concentration of monitored compounds that are similar to background levels.

7) Water quality conditions are based on qualitative evaluations and fish tissue concentrations for monitored contaminants (e.g., mercury and PCBs) and are similar to or better than other Lake Erie levels.

8) Near-shore sediment habitats suggest that ingestion exposure risks to wildlife are moderate to low, and the elevated surface sediment concentrations of PAHs and metals (dry weight) in PIB tend to be in the vicinity of the docks and shipping channel.

Overall, it appears that the sediment targets supporting the PIB ecosystem are being met. Gaps in data to definitively describe all targets and metrics exist, but the current weight of evidence suggests that the COPC risk to ecosystem receptors within PIB is improving through time currently rates low to moderate risk.

1. INTRODUCTION AND BACKGROUND

This report presents a screening-level ecological risk assessment for Presque Isle Bay (PIB), located in Erie, PA. PIB was listed as a Great Lakes Area of Concern (AOC) in 1991as a result of two Beneficial Use Impairments (BUIs) that were identified related to contaminants in sediments: 1) restrictions on dredging; and 2) presence of fish tumors. Since 1991, several investigations and studies have been conducted by the Pennsylvania Department of Environmental Protection (PA DEP), federal agencies, and academic researchers to characterize contaminants in sediments and their potential effects on benthic fauna and fish. The studies indicated that the historical contaminant sources to PIB were largely addressed, and that concentrations of contaminants in surficial sediment and the incidence of tumors in fish were declining over time. As a result, in 2002 the AOC was designated as being “In Recovery,” and Monitored Natural Recovery was determined to be the most cost-effective remedial alternative to address residual contamination. In 2006, the restriction on dredging BUI was removed, leaving the BUI of tumors in fish as the only remaining identified impairment.

The USEPA Great Lakes National Program Office (GLNPO) and PA DEP are currently assessing whether PIB has sufficiently recovered to remove the remaining BUI and delist the AOC. To support that assessment, GLNPO contracted LimnoTech to review the site data and perform an screening-level ecological risk assessment of PIB. Gannon University was concurrently contracted to perform a human health risk assessment of PIB. This report presents the results of the ecological risk assessment (ERA). This ERA is considered to be a screening-level ERA, largely because the historical studies were designed to address specific objectives of each individual study and not designed to support a comprehensive ERA. As a result, data and information regarding some ecological exposure pathways and endpoints are not available. The assessment presented herein, however, provides a complete summary of the existing data and evaluations of the implications for ecological risks, and will help inform risk management decisions by EPA and PA DEP.

1.1 PROJECT AREA DESCRIPTION

Presque Isle Bay (PIB) is located adjacent to Erie, PA, in northwestern Pennsylvania on the southern shore of Lake Erie (Figure 1). Presque Isle Bay is 7.3 km long and 2.4 km across at its widest point, and has an average depth of approximately 4 meters. Its drainage basin includes much of the City of Erie, as well as parts of Mill Creek, Summit, Greene, and Harbor Creek Townships. The PIB watershed consists of the Bay itself, Mill Creek watershed (including Garrison Run), Cascade Creek watershed, Scott Run watershed, and the aquatic habitats (including ponds) within Presque Isle State Park.

The Bay is formed by Presque Isle, 11.3 km long sand spit. The eastern end of the Bay connects to Lake Erie through a narrow channel. This channel is dredged to allow commercial shipping traffic and recreational boaters to enter the PIB from the lake. Presque Isle State Park borders the northern edge of the Bay. Presque Isle comprises primarily sand and glacial sediments, with a series of ponds and lagoons representing the principal aquatic habitats. Presque Isle supports a diversity of wildlife, with over 320 bird species, 47 mammal species, and 30 amphibian and reptile species. Many of these species are included on Pennsylvania's list of Species of Special Concern.

Figure 1.1 Presque Isle Bay, risk assessment project area.

1.2 HISTORY

The waterfront of Erie, PA, has historically been dominated by heavy industry and commercial developments. For many years, discharges from industry and commercial developments were released directly into PIB or were directed to the City of Erie's wastewater treatment, collection, and conveyance system. During periods of elevated runoff, untreated industrial, commercial, and residential wastewater escaping from combined sewer overflows (CSOs) were discharged to the Bay. While recent efforts to control contaminant sources have been effective in reducing discharges, historical releases resulted in substantial loadings of sediment-bound contaminants. Some of the pollutants released to PIB have decayed through natural biodegradation processes; however, substances like heavy metals and more persistent organic contaminants remain in the sediment (PA DEP 2002).

Several studies have been conducted over the past 20 years to evaluate sediment quality conditions in and across PIB. The results of these investigations show that Bay sediments contain measurable concentrations of a variety of chemicals of potential concern (COPCs), including polycyclic aromatic hydrocarbons (PAHs), heavy metals, polychlorinated biphenyls (PCBs), organochlorine pesticides (e.g., chlordane, DDTs), and several other substances. No impairments to the water column were indicated, but the presence of such chemicals in aquatic sediments represents a potential environmental concern (PA DEP 2002) for reasons including:

- Bed sediments provide essential and productive habitats for communities of sediment-dwelling organisms, including epibenthic and infaunal organisms;
- Sediment-dwelling organisms are important elements of freshwater ecosystems, representing important sources of food for many fish and other wildlife species;
- The presence of sediment-associated contaminants in freshwater ecosystems can be harmful to sediment-dwelling organisms, fish, and aquatic-dependant wildlife species; and,
- Certain sediment-associated contaminants can accumulate to high concentrations in the tissues of aquatic organisms and, as a result, pose a potential hazard to those species that consume aquatic organisms, including wildlife and humans.

In 1991, Presque Isle Bay became the 43rd and final Area of Concern (AOC) listed under the Great Lakes Water Quality Agreement (GLWQA). The listing was primarily driven by observations of external fish tumors on bullhead collected within the Bay, and reported to U.S. Fish and Wildlife Service at that time (PA DEP 2006).

Based upon a limited analysis of the existing data, sediment contamination and tumors in brown bullheads were the biggest AOC concerns. Regarding pollutants of concern, work on both sediments and brown bullheads indicated that PAHs could be of greater concern than the heavy metals. The main source for the contaminants appeared to be in-place sediments, as no correlation was found between water and sediment contaminant concentrations (PA DEP 1992).

From these assessments, PA DEP believed that two of the 14 beneficial uses were potentially present in the Bay: (1) fish tumors and other deformities, and (2) restrictions on dredging activities. Following an impaired uses evaluation, the only pollutants of concern identified were sediment-bound contaminants. No water column impairments were indicated. Fish impairments, if environmentally caused, were believed related to the sediment contamination; however, no correlation was made between sediment contamination and tumor rates (PA DEP 1992).

Between 1991 and 2006, the extensive efforts of PA DEP and its partners culminated with the removal of the dredging BUI, as documented in the removal petition and detailed rational described in the 2006 PA DEP report, *Delisting Restrictions on Dredging Activities Beneficial Use Impairment in the Presque Isle Bay Area of Concern* (PA DEP 2006).

A number of factors were taken into consideration when evaluating removing the dredging beneficial use impairment for Presque Isle Bay. Contaminants were detected in the sediment at concentrations greater than sediment quality guidelines associated with increased toxicity to benthic organisms; however, when the overall contamination was considered, none of the whole-sediment samples exceeded levels linked with reduced survival or growth of benthic organisms. Also, it was found that levels of measured contaminants in sediments were not sufficient to adversely affect fish and aquatic-dependent wildlife in the AOC. For bioaccumulative compounds, fish tissue data indicate that PIB sediments are not a significant source—concentrations of mercury and PCBs in tissue from Presque Isle Bay fish were similar to those found in Lake Erie fish, indicating a lake-wide rather than AOC-specific problem.

The evaluation of sediment quality in the Bay indicated that factors other than the contaminants in the sediment might be contributing to the limited toxicity to benthic organisms that was observed. Analysis of the data shows that metals and PAHs, while present, did not or rarely occurred in the AOC or study area sediments at concentrations sufficient to adversely affect benthic organisms, fish, or aquatic-dependent wildlife. Ecosystem health targets were being met in the AOC, and there was no evidence that the moderate level of contamination found during sediment studies was responsible for degrading the ecosystem.

Finally, given that the only "restriction" on dredging activities was regulatory, and that sediment from any location within the AOC met those requirements, it was recommended that the dredging beneficial impairment be removed (PA DEP 2006).

The extensive and combined efforts described above resulted in the delisting of the restrictions on dredging activities BUI, leaving one remaining BUI within the PIB AOC—fish tumors or other deformities.

1.3 ERA PROBLEM STATEMENT

The presence of fish tumors is considered a beneficial use impairment when, "the incidence rate of fish tumors or other deformities exceeds rates at unimpacted or control sites, or when survey data confirm the presence of neoplastic or pre-neoplastic liver tumors in bullhead or suckers" (IJC 1991). Unfortunately, addressing the fish tumors or other deformities BUI directly within the PIB AOC, as well as AOCs across the Great Lakes, has proved to be challenging (Rafferty et al. 2009). The scientific understanding of the cause and effect relationships for fish tumors is complicated and confounding, and there is a lack of specific assignments of control sites, and lack of clear definitions of tumor types and background rates. As such, the data collected on fish tumors have created more questions than answers for assessing fish tumor conditions (Rafferty et al. 2009). Section 3.4.4 provides a summary of the state of the science, citing recent publications, and existing challenges that remain in addressing the IJC (1991) definition of impairment, with respect to understanding causes or incidences of tumors on sentinel indicator species such as brown bullhead (*Ameiurus nebulosus*). Further, there is no information to indicate that the presence of tumors on fish adversely impacts their survival, growth, or reproduction, or poses a threat to ecological predators of those species.

The state of the science and the lack of understanding of cause-effect relationships have complicated the ability of AOC partners and researchers to define attainable targets for this BUI (Rafferty et al. 2009). Thus, PA DEP and its partners have opted to use the ERA approach, using existing information. The presence and frequency of tumor occurrence is one line of evidence in the assessment of risks to fish. The ERA was designed to address the following question, originally posed by Diz (2002):

Do legacy contaminants (contaminants of potential concern) continue to pose a risk to ecosystem receptors within Presque Isle Bay (Diz 2002)?

1.4 REPORT SCOPE

The PIB SLERA included an extensive report evaluation and data review of documents provided by PA DEP and its partners. The data and information are synthesized and summarized to support the development of a detailed conceptual site model (CSM) for identified contaminants of potential concern (COPCs) within PIB. The model uses existing information to identify pathways of exposure and ecosystem receptors at potential risk. Where available and appropriate, existing data have been examined, assessed and summarized to support risk assessments for ecosystem receptors including benthic invertebrates, fish, and wildlife. The evaluation culminates with a qualitative weight-of-evidence evaluation that assesses the likelihood that COPCs pose unacceptable risks to ecosystem receptors. A summary of findings, uncertainties, and conclusions is also included in this report.

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2. PROJECT APPROACH

2.1 OVERVIEW

In response to the challenges of defining clear targets that lead to delisting for the fish tumor and other deformity BUI in PIB, PA DEP and its AOC partners have opted to pursue a screening-level ecological risk assessment approach (SLERA). The SLERA process is commonly used to systematically evaluate how likely it is that adverse ecological effects might occur as a result of exposure to stressors (USEPA 1998). Ecological risk assessments can be prospective and used as a prediction of the likelihood of future effects or current as an evaluation of the likelihood that observed effects are associated with current exposure to stressors (Gala et al. 2009). The PIB SLERA process evaluates and assesses risk of chemicals of potential concern (COPCs) within the Bay and whether their concentrations pose a significant risk to receptors in the ecosystem.

2.2 RISK ASSESSMENT FORMAT

The ecological risk assessment for Presque Isle Bay generally follows EPA guidance for conducting baseline ecological risk assessments under the Comprehensive Environmental Response, Compensation, and Liability Act (EPA 1998). In addition, the framework provided by Cura et al. (2001) was followed as it was specifically developed for PAH-contaminated sediments, such as those found in PIB. However, comprehensive ERAs performed under the EPA guidance are typically iterative investigations specifically designed to meet ERA data quality objectives established prior to conducting the investigations to assess all potential significant and complete exposure routes and receptors potentially at risk from exposure to COPCs at a site. This ERA relies on data readily available with no new data collection to fill data gaps. The supporting data were collected as part of several independent historical investigations that were not specifically designed to support a formal and comprehensive ERA. While the ERA presented herein is more detailed than and goes beyond the objectives of a typical screening level ERA, it is considered to be a SLERA. This assessment uses a mix of previously developed assessments (published and unpublished), conclusions, and recommendations combined with components of analysis of the best available datasets and estimation methods to develop a weight-of-evidence evaluation of risk to ecosystem receptors within the AOC, for those COPCs, pathways and receptors where data are available.

The PIB ERA addresses the four primary components used in the assessment of ecosystem risk (EPA 1998; Cura et al. 2001). A fifth component is included that summarizes the findings, identifies uncertainties, and conclusions based on the ERA. The ERA components include the following:

> *Problem Formulation.*
> The Problem Formulation includes defining the objectives, developing the conceptual site model (CSM), identifying COPCs, selecting and characterizing receptors, and identifying the endpoints of the assessment

(USEPA 1992). The key components of problem formulation are detailed in Section 2.1.1, based on existing documents and evaluations.

Based upon the objectives and the CSM, three ecosystem receptor groups were identified for assessing risks, including benthic invertebrates, fish, and wildlife. Risk assessments were conducted for each of these receptor groups, and consisted of the following components:

> *Exposure Assessment.*
> The Exposure Assessment estimates the magnitude of actual and/or potential ecological exposure to a contaminant of concern, the frequency and duration of exposure, and the pathways of exposure. For the PIB SLERA, COPC concentrations measured in sediments served as the primary basis for the quantitative exposure assessments. Data for other exposure media (e.g., water column, pore water, food web) were not available and so were either estimated through exposure models or qualitatively characterized.
>
> *Effects Assessment.*
> The Effects Assessment summarizes and weighs available evidence regarding the potential for contaminants to cause adverse effects in exposed organisms, and estimates the relationship between the extent of exposure to a contaminant and the increased likelihood and/or severity of adverse effects. The effects assessments for the PIB SLERA relies primarily on published toxicity reference values (TRVs), sediment quality values (SQVs), and whole-sediment toxicity tests.
>
> *Risk Characterization*.
> The Risk Characterization summarizes and integrates the Exposure Assessment and Effects Assessment into a quantitative and qualitative expression of risk, supporting the weight-of-evidence conclusions of ecosystem effect. The benthic invertebrate risk assessment relied primarily on comparison of COPC concentrations in sediments with consensus based SQVs and site specific sediment toxicity tests. The fish risk assessment relied primarily on general conclusions of prior investigations of water quality, and available measured tissue concentrations of bioaccumulative chemicals. The wildlife risk assessment relied on estimated exposure and uptake model results compared with TRVs.

Uncertainties and Conclusions.
Finally, an overall summary of the risk assessments for the three receptor groups is provided in Section 4 and includes a characterization of uncertainties and presents conclusions.

2.3 PROBLEM FORMULATION

The problem formulation step provides background to conduct the screening-level risk assessment to determine if chemicals of potential concern within the Bay pose a significant risk to receptors in the ecosystem. Many of the problem formulation components required for this phase of the ERA were captured and defined by the extensive efforts of the AOC partners and summarized in the delisting documentation for removing the restriction on dredging BUI (PA DEP 2006). That is, much of the effort included in, *The Delisting of the Restrictions on Dredging Activities Beneficial Use Impairment in the Presque Isle Bay Area of Concern* (PA DEP 2006), focused on defining ecological receptors and ecosystem components, and thus present a robust framework for defining and directing the assessment of ecosystem risk within the PIB AOC.

2.3.1 Risk Assessment Objectives for PIB

The PIB ERA risk assessment objectives were developed to be consistent with the PIB ecosystem objectives developed by PA DEP (2006) for sediment COPCs. The ecosystem objectives in PIB (PA DEP 2006) include the following:

1. Protect and preserve recreational uses;
2. Maintain and protect the benthic invertebrate community;
3. Maintain a quality fishery;
4. Protect and improve the near-shore habitat;
5. Maintain the aesthetic qualities (e.g., prevent algal blooms, unpleasant odors, visual impairments, etc.);
6. Maintain and improve water quality conditions; and
7. Eliminate the restrictions on dredging.

For the purpose of the PIB screening-level ecological risk assessment, evaluations will focus on ecological components of concern in the system most likely affected by sediments containing COPCs. Thus, the ecological evaluation within PIB is best represented by the stakeholder-developed ecosystem objectives (Objectives 2-4 from above), supporting questions and attainment targets (PA DEP 2006):

- <u>Maintain and protect the benthic invertebrate community</u>
  1) Are the levels of contaminants in whole sediments from Presque Isle Bay greater than benchmarks for the survival or growth of benthic organisms?
  2) Is the survival or growth of benthic organisms exposed to whole sediments from Presque Isle Bay significantly lower than that in control or reference sediments?

- <u>Maintain a quality fishery</u>
  1) Are the levels of contaminants in water and whole sediments from Presque Isle Bay greater than benchmarks for the health of fish?
  2) Are the levels of contaminants in fish tissues from Presque Isle Bay greater than the levels of contaminants in fish from elsewhere in Lake Erie?

- Protect and improve the near-shore habitat (to support aquatic-dependent wildlife)
  1) Are the levels of contaminants in whole sediments from the Presque Isle Bay near-shore environments greater than benchmarks for the health of aquatic-dependent wildlife?

2.3.2 Conceptual Site Model

The purpose of the CSM is to describe the sources of COPCs, routes of transport, media, routes of exposures, and ecological receptors (Suter 1996). The model framework for PIB includes sources, routes of transport from contaminated media (sediment), routes of exposure of receptors to media, and endpoint receptors initiated by PA DEP (2006). Following Suter (1996), the CSM for the PIB ecosystem as depicted in MacDonald (2008) has been expanded to identify specific sources, COPC transfer paths, sediment processes that may contribute to COPC transfer, and specific receptors identified in PIB and other supporting documents (Figure 2.1). The important components of the CSM are described below.

Chemicals of Potential Concern - Identification of COPCs represents an essential element of the overall sediment quality assessment process (USEPA 1998). The COPC list and associated sources stem from several evaluations specifically conducted to assess PIB AOC conditions (MacDonald 2008). For the PIB model, only the toxic COPCs that partition into sediments were considered, and COPCs that usually (90% or more) or always occurred at levels below analytical detection limits were eliminated from further consideration (MacDonald 2008). Thus, the COPCs evaluated in PIB ERA were selected because of their frequency of exceeding toxicity thresholds (probable effect concentration (PEC)) in surficial sediment samples, as identified by MacDonald (2008). PECs are sediment quality guidelines established as concentrations of individual chemicals above which adverse effects in sediments are expected to frequently occur (EPA 2000). Adverse effect documentation is complex and includes uncharacterized chemicals or stressors, localized conditions of bioavailability, movement of organisms, responses of organisms, and representation of unsampled areas and errors in chemical and biological responses (Simpson et al. 2005).

Table 2.1 describes the COPCs (MacDonald 2008) included in the PIB model and SLERA.

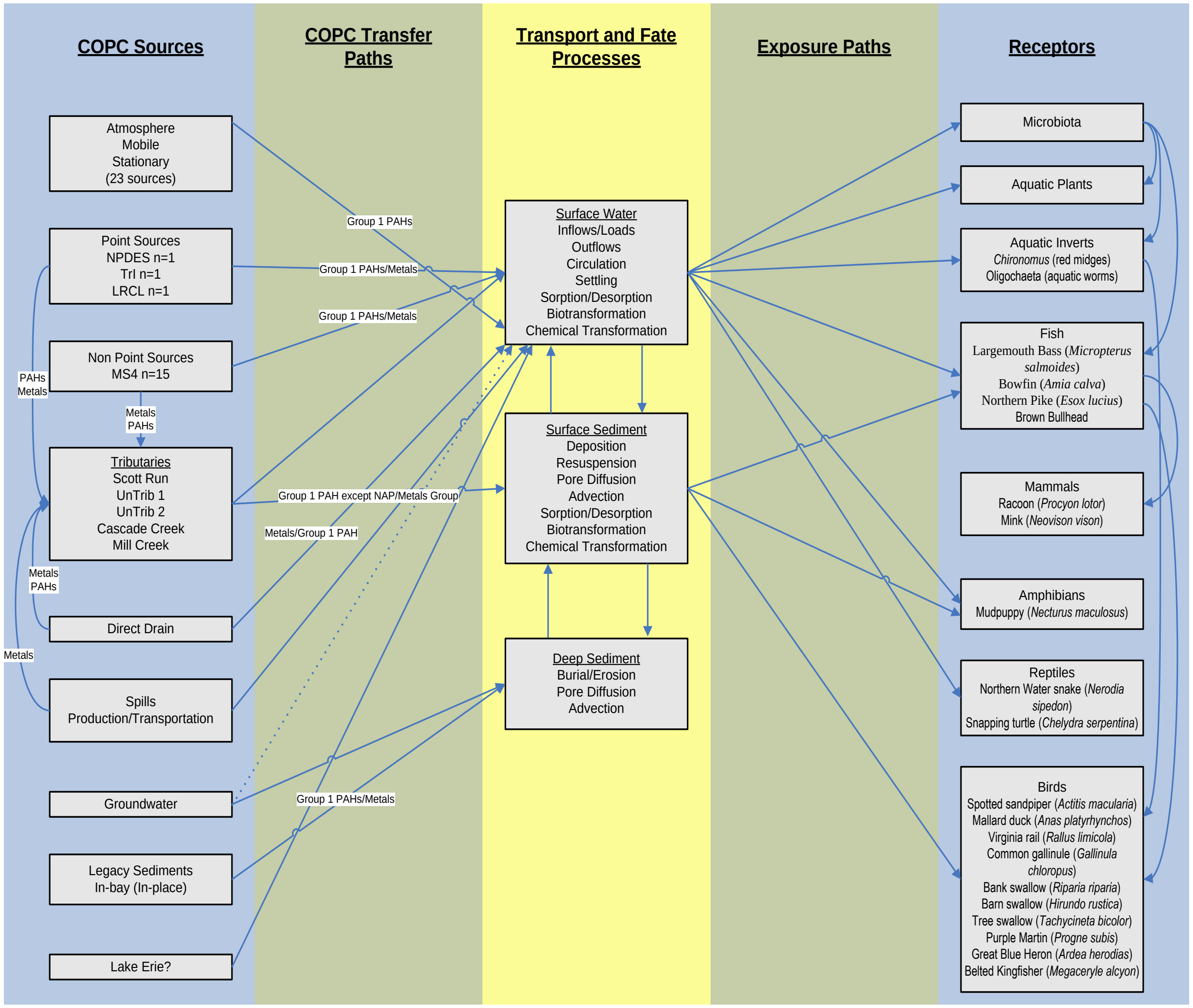

Figure 2.1 Conceptual site model (CSM) for Presque Isle Bay sediment processes across the Bay. The CSM includes the COPC sources, sediment processes and potential receptors within the Bay

Table 2.1. COPCs within PIB with more than 10% exceedance of selected toxicity levels (MacDonald 2008).

Chemical of Potential Concernt (COPC)	Selected Toxicity Threshold (PEC)	Exceedance in PIB study area	Bioaccumulative[1]	Source
Metals (mg/kg DW)				
Antimony	25	43% (6 of 14)		MacDonald (2008)
Arsenic	33	22% (29 of 131)		MacDonald (2008)
Barium	60	66% (80 of 121)		MacDonald (2008)
Cadmium	4.98	45% (56 of 125)	Y	MacDonald (2008)
Lead	128	29% (41 of 141)	Y	MacDonald (2008)
Nickel	48.6	34% (47 of 140)		MacDonald (2008)
Zinc	459	11% (16 of 141)		MacDonald (2008)
Polycyclic Aromatic Hydrocarbons (PAHs; µg/kg DW)				
Acenaphthene	88.9	45% (15 of 33)		MacDonald (2008)
Acenaphthylene	128	31% (12 of 39)		MacDonald (2008)
Benz(a)anthracene	1050	25% (16 of 63)		MacDonald (2008)
Benzo(a)pyrene	1450	17% (11 of 63)		MacDonald (2008)
Chrysene	1290	25% (16 of 63)		MacDonald (2008)
Dibenz(a,h)anthracene	135	51% (20 of 39)		MacDonald (2008)
Fluoranthene	2230	26% (17 of 65)		MacDonald (2008)
Phenanthrene	1170	19% (12 of 63)		MacDonald (2008)
Pyrene	1520	59% (37 of 63)		MacDonald (2008)

[1] accumulation of chemicals in the tissue of organisms through any route, including respiration, ingestion, or direct contact with contaminated water, sediment, and pore water in the sediment." – US EPA 2000

<u>Sediment Processes</u> – Diz (2002) described the surficial sediment samples as dominated by fine sediments such as sand, silt, and clay. After discarding zebra mussel shells found in sediment samples at some locations, the average composition of the sediments sampled in 2000 consisted of 16.5% sand, 42.8% silt, and 40.8% clay. Although the samples were spatially varied across the Bay, they are assumed to represent the general sediment composition of PIB surficial sediment conditions.

Analysis of chemical concentrations in whole-sediment samples serve as a simple and common method for estimating risk of exposure from COPCs. However, physical and chemical properties of sediments vary from site to site and affect the bioavailability of the chemicals to exposed receptors. As such, measured COPC concentrations in one sample may have a very different effect than the same concentrations measured in a different sample. Chemicals of concern in PIB such as PAHs are hydrophobic and tend to immediately adsorb (bind) to fine sediments and organic carbon in the sediment matrix (Fuchsman et al. 2001). In addition, different types of organic carbon have different adsorption properties; for example, "black" carbon adsorbs hydrophobic chemicals more highly and irreversibly than carbon from detritus. Metal

COPCs' bioavailability is also controlled by processes such as: 1) speciation (e.g., metal binding with particulate sulfide, organic carbon, and iron hydroxide phases); 2) sediment–water partitioning relationships; 3) organism physiology (e.g., COPC uptake rates from surface and pore water particles); and 4) organism behavior (e.g., feeding on organisms exposed to COPCs and other sediment disturbing behaviors) (Simpson and Batley 2006; Fuchsman et al. 2001).

Measures of sediment concentrations for COPCs can provide an indication of relatively long-term environmental exposures. The risk of exposure to organisms in waterbodies with hydrophobic COPCs depends on the factors described above. Organic carbon is a critical factor controlling the availability of PAHs and metals in sediment and effect on aquatic organisms (USEPA 2000; Simpson and Batley 2006). Physiochemical processes like temperature increase the solubility and organism uptake potential of hydrophobic COPCs, while increases in salinity decrease the solubility of such compounds. Finally, the behavior of species within COPC waterbodies can affect the potential for exposure. For example, the behavior of some bottom-dwelling fishes can result in the resuspension of sediment-bound chemicals, thus increasing the risk of exposure.

However, because fish metabolize chemicals like PAHs, it remains a challenge to scientists to establish adverse sediment exposure and injury to these organisms, and thus the organisms that feed on fishes themselves (Fuchsman et al. 2001).Upon release into aquatic ecosystems, COPCs partition into the water and sediment, depending on their physical and chemical properties and the characteristics of the receiving waterbody (PA DEP 2006). Aquatic organisms may be exposed to the COPCs in the water or sediment, so the CSM attempts to represent sediment transport processes operating in the ecosystem (Suter 1996; ITRC 2011). That is, the model depictions identify pathways and sources of bioavailability operating within the system (ITRC 2011). The exposure pathways for COPCs were developed using PIB documents such as PA DEP (2006) and MacDonald (2008), as well as general guidance documents such as ITRC (2011).

The PIB AOC is currently listed as "In Recovery." The natural capping of contaminated Bay floor areas with inputs of "cleaner than in the past" sediments supplied from watershed is considered a likely solution to the Bay's contaminated sediments issue (PA DEP 2006). Foyle and Norton (2006) suggests that the complex nature of the sediment transport conditions in PIB includes a mix of resuspension of legacy COPCs in shallow zones and deposition in deeper areas, and that inputs from outside sources have highly variable deposition rates, where deposition processes dominate. The variability in the deposition rates across the Bay may require several decades to reduce the physical availability of COPCs in the system (Foyle and Norton 2006). Foyle and Norton's (2006) evaluation of the sediment processes in PIB for COPCs helped refine the ecological receptor list below (Table 2.2).

<u>Receptors</u> – The purpose of including ecological receptors in the model is to depict how exposure from COPCs may occur to organisms of concern (Suter 1996). There are a wide variety of ecological receptors that could be exposed to contaminated

sediment in Presque Isle Bay. The aquatic organisms that occur in the Bay are numerous and include microbiota (e.g., bacteria, fungi and protozoa), aquatic plants, aquatic invertebrates, fish, amphibians, reptiles, bird and mammals. A specific list of key indicator organisms was developed in collaboration with PA DEP (personal communications, Jim Grazio 2011) to refine the risk evaluation (Table 2.2).

Table 2.2. Presque Isle Bay list of primary ecological receptors of concern (PA DEP 2011)

Group	Species
Invertebrates	
	Chironomus (red midges)
	Oligochaeta (aquatic worms)
Fish	
	Largemouth Bass (*Micropterus salmoides*)
	Bowfin (*Amia calva*)
	Northern Pike (*Esox lucius*)
Birds	
Sediment-Probing	Spotted sandpiper (*Actitis macularia*)
	Mallard duck (*Anas platyrhynchos*)
	Virginia rail (*Rallus limicola*)
	Common gallinule (*Gallinula chloropus*)
Insectivorous	Bank swallow (*Riparia riparia*)
	Barn swallow (*Hirundo rustica*)
	Tree swallow (*Tachycineta bicolor*)
	Purple Martin (*Progne subis*)
Carnivorous Wading	Great Blue Heron (*Ardea herodias*)
Mammals	
	Racoon (*Procyon lotor*)
	Mink (*Neovison vison*)
Reptile	
	Northern Water snake (*Nerodia sipedon*)
	Snapping turtle (*Chelydra serpentina*)
Amphibian	
	Mudpuppy (*Necturus maculosus*)

Using existing studies and data, the PIB CSM can be used to identify the source, pathway, and receptors that are best and least understood within the PIB AOC.

For the PIB SLERA, the list of ecological receptor groups was refined to three groups (benthic invertebrates, fish, and wildlife (mammals and birds)) based upon the availability of data and published TRVs. Exposure routes and effects are different for each of these groups, so separate risk assessments were performed for each. Representative species within each group were used in each risk assessment to estimate exposure and effects and to characterize the risks to each group.

2.3.3 Assessment Endpoints

EPA (1992) defines assessment endpoints as explicit statements of the ecological systems that are to be protected. General considerations for selecting assessment and measurement endpoints include ecological relevance, policy goals and societal values, and susceptibility to chemical stressors (EPA 1992; 1996). The ecosystem objectives and endpoints developed from the extensive efforts of PA DEP and its partners in, *The Delisting of the Restrictions on Dredging Activities Beneficial Use Impairment in the Presque Isle Bay Area of Concern* (PA DEP 2006), are appropriate as a foundation for the PIB SLERA.

Consistent with the CSM presented above and the PA DEP endpoints, the assessment endpoints for the purposes of this SLERA include survival, growth, and reproduction of: 1) benthic invertebrates, 2) fish, and 3) wildlife (mammals and birds). Representative receptors, measurement endpoints and target metrics, and lines of evidence are presented below for each of these ecosystem receptor groups. Multiple lines of evidence and targets were evaluated where data were available and then compiled and collectively assessed under a qualitative weight-of-evidence assessment approach.

Weight-of-evidence is a process by which multiple lines of evidence, often expressed as measureable endpoints (targets), are related to assessment endpoints (objectives) to evaluate whether significant risk is posed to the environment (Menzie et al. 1996). Because the PIB SLERA is relying on existing studies and findings supported by limited independent data evaluation, the weight-of-evidence approach relies heavily on the data and findings of previously conducted studies. The PIB SLERA approach uses endpoints interpreted from the PIB studies combined with data assessments described in Section 3 to evaluate targets that support SLERA objectives. The results of the combined sets of evaluations are used as the qualitative weight-of-evidence assessment to describe objective attainment. The qualitative approach is applied because the SLERA is using a mix of studies whose approach, targets, and purposes were not always directly comparable. The assessment endpoints for each of the three ecosystem receptor groups for the SLERA are discussed below.

2.3.3.a Benthic Invertebrate Assessment Endpoints, Objectives and Targets

As described in Section 2.3., an ecosystem objective for PIB is to maintain and protect the benthic invertebrate community. Target metrics to assess the growth survival and reproduction of benthic invertebrates for the SLERA were developed using published sediment quality guidelines as a relatively simple, conservative calculation of toxicity threshold and are consistent with targets presented in PA DEP (2006). However, as discussed above, this simple comparison of sediment chemistry SQVs may not adequately account for reduced site-specific bioavailability. As such, the weight-of evidence sediment quality triad approach was used. The approach integrates sediment chemistry, sediment toxicity testing and macroinvertebrate community analysis (ITRC 2011). Unfortunately, within PIB, no studies evaluated all components of the sediment quality triad simultaneously. Further, there are no studies that quantitatively assess the existing benthic community structure, abundance and diversity in comparison with non-impacted reference areas. However, key sediment quality components (e.g., site-specific sediment toxicity tests) and data trends can be used to help assess PIB ecosystem objectives to maintain and protect the benthic invertebrate community.

The specific targets and metrics used in the PIB SLERA for the assessment of benthic invertebrate risks include the following:

Target 1- at least 90% of the sediment samples from Presque Isle Bay have the conditions necessary to support healthy benthic invertebrate communities, as indicated by metrics in Table 2.3.

Table 2.3. Benthic invertebrate community target metrics.

Metric	Target Value
Bulk Sediment Quality Benchmarks (Median PEC-Q)	< 1.0 (ratio) and <6 PEC exceedances[1] per station (OEPA 2010)
Metals Mixtures (SEM-AVS)	< 0.0
Metals Mixtures with Organic Carbon (SEM-AVS/foc)	< 130
PAH Mixtures ESB-TUs	< 1.0
Sediment toxicity to amphipods and midges for survival and growth[2]	- Control-adjusted survival of amphipods > 75% - Control-adjusted growth of amphipods >90% - Control-adjusted survival of midges > 75% - Control-adjusted growth of midges >70%
Notes - SEM (simultaneously extracted metals); AVS (acid-volatile sulfide); foc (fraction of organic carbon); ESB (equilibrium partitioning sediment benchmarks); TU (toxicity units)	

[1] MacDonald et al. 2000. [2] Control-adjusted survival of midges >75% means that the test results must be more than 25% different from the control result to be considered toxic.

These metrics are consistent with the triad approach in that the metrics use target values to assess benthic community conditions, and rely on comparisons of sediment chemistry with SQVs as well as site-specific sediment toxicity tests. Sediment chemistry targets and toxicity tests are the main triad components available in PIB to support the weight-of-evidence determinations.

Benthic community descriptions are under-represented within PIB. Describing such targets for benthic communities in lentic environments is challenging because the structure and composition of these communities are dependent on many factors, such as physical sediment characteristics and are highly variable both spatially and temporally (Reice and Wohlenberg 1993), requiring intensive and expansive sampling efforts across the micro and macro environments within PIB, as well as among seasons and years. Grab samples of PIB sediments were described as black or brown and dominated by fine sediments, based on particle size analysis with categories of sand, silt and clay best describing the dominant substrates found in PIB samples (Diz 2002). The macroinvertebrate evaluations within PIB (Diz 2002) found samples dominated by zebra mussels (*Dreissena)*, two pollution-tolerant macroinvertebrates; segmented worms (*Oligochaetae*), and midges (*Chironimidae*), as well as moderately tolerant gastropods and amphipods.

The limited number of macroinvertebrates samples support the generally held view that accumulations in surface fine sediments lead to changes (generally reductions) in macroinvertebrate community diversity (Harrison et al. 2007) and dominated by taxa such as *Chironomidae, Oligochaeta* and *Sphaeridae* (frequently associated with fine sediments because they are able to burrow into sediments). Therefore, fine sediment covered substrates such as those within PIB, contain less diverse macroinvertebrate communities that are primarily habitat limited and dominated by taxa that are tolerant to fine sediments (Waters 1995).

PEC-Q

Sediment chemistry metrics include a number of target values. PEC-Q is the ratio of the concentration of a COPC to its probable effect concentration (PEC). The PEC-Q approach provides a direct way for determining if the concentration of COPCs impedes biological resources (MacDonald 2008). This determination can be made by comparing the measured concentrations of COPCs to acute or chronic toxicity thresholds. For this study, consensus-based PECs were used to identify the substances at concentrations high enough to affect benthic invertebrates.

To calculate toxicity of sediment, the average of the PEC-Qs in the sediment is calculated. The mean PEC-Q allows for the mixture of chemicals in the sediment to be quantified. This quantification makes it a desirable metric to report full-sediment toxicity (MacDonald 2008). Although the Mean PEC-Q is the value typically calculated using procedures established by USEPA (2000). The median (Median PEC-Q) was used here because the high standard deviations identified within PIB PEC-Q may limit the value of the arithmetic mean as an accurate estimate of central tendency, particularly when multiple areas (including random and targeted), targeted studies (targeted at COPC concentration), sample methods (multiple gear types) and processing approaches (differing QA/QCs) are being evaluated.

SEM-AVS

Simultaneously extracted metals (SEM) - acid volatile sulfides (AVS) & (SEM-AVS)/f_{OC} models were applied to PIB samples, as developed by USEPA (2005) to evaluate the toxicity of metals to sediment-dwelling organisms. The application of these models is dependent on the collection of SEM and AVS data in whole-sediment samples. The models assume that specific metals can only cause or contribute to sediment toxicity when the sum of their concentrations of copper, lead, nickel, and zinc exceed the concentration of AVS. The presence and quantity of AVS and organic carbon in sediments affects the likelihood that COPCs will affect sediment-dwelling organisms (ITRC 2011). That is, the EPA-adopted equations (EPA 2000) assume that greater concentrations of sulfides and organics in sediments results in binding of COPCs to these particulates, reducing the bioavailablity of contaminants to the ecosystem (ITRC 2011).

(SEM-AVS)/f_{OC}

Further, since metals can bind to organic carbon in sediment, the model has been updated by incorporating the fraction of total organic carbon (f_{OC}) into the model (OEPA 2010). Like AVS, the presence and quantity of organic carbon in sediments affects the likelihood that COPCs will affect sediment-dwelling organisms (ITRC 2011). It is recognized that the organic carbon content of the sediment is the component most responsible for controlling bioavailablity of organic COPCs (Adams and Rowland 2003; Burgess 2009). Thus, it is believed that the (SEM-AVS)/f_{OC} model represents a more reliable representation of the toxicity of COPCs to sediment-dwelling organisms from whole-sediment samples (Adams and Rowland 2003).

ESB-TU

Finally, equilibrium partitioning sediment benchmarks (ESB) approach was included (USEPA; 2000) because this approach predicts chemical interactions among sediments, interstitial water and COPCs. The ESB estimates direct toxicity to benthic organism and offers several advantages over other effects–based benchmarks because the calculations are contaminant-specific, address causal relationships between COPCs and their potential for toxicity, and encompass site-specific conditions that affect bioavailability (ITRC 2011). However, it should be cautioned that care should be used in interpreting ESBs in dynamic systems such as PIB. In highly erosional or depositional environments (e.g., wind, seiche, navigation), partitioning may only reach a state of near-equilibrium (EPA 2003).

PAHs tend to occur in the environment in mixtures, so assessing the toxicity of PAH mixture effects uses concept of toxic units (US EPA 2003). Toxic units (TUs) are described as the ratio of the concentration of the PAH mixtures relative to the toxic effect of the concentration. The ESB-TU method was initially developed for sediments where 34 PAHs were analyzed. However 13 or 23 PAHs are the more commonly measured combination of PAHs, so to characterize the uncertainty in the ESB-TU calculations, uncertainty factors were applied to ESB-TU values calculated within PIB as suggested by EPA (2003)

In principle, the uncertainty factor serves as a multiplier to convert TUs when less than 34 PAHs are evaluated. However, uncertainty factors are site-specific because the variability of PAHs in contaminated sediments is uniquely distributed at each site (Burgess 2009) based on the processes controlling the sediment distribution (e.g., wind, seiche, navigation, dredging) and the methods used to collect samples (e.g., within and across PIB). So uncertainty factors should only be used as a very general estimate of TU (Burgess 2009).

Sediment Toxicity Tests

Toxicity tests provide an important complement to ESB assessments in determining overall risk from COPCs (EPA 2003). Like other procedures for detecting adverse affect, toxicity tests provide value as an independent parameter of effect, but include limitations that should be considered from the results. Toxicity tests are capable of detecting any toxic chemical and are useful for detecting the combined effect of chemical mixtures, if those effects are not considered in the formulation of the applicable chemical-specific benchmark (EPA 2003). However, they only provide information on the toxicity to the species being tested. Typically, species used for toxicity tests reflect more sensitive and less tolerant benthic species (EPA 2003). Toxicity tests conducted with PIB sediments are included to provide a valuable and complementing component for interpreting the assessment of adverse affect to the biota.

Diz (2002) evaluated PIB sediment toxicity. The sediment toxicity tests, the survival of *C. tentans* was slightly lower in the PIB sediments than in the control. The growth of the organisms was both greater and less than the control for various PIB sites, but not significantly different from the control. The survival and growth of *H. azteca* in PIB sediment was not significantly different from the control. The survival of *D. magna* was more sensitive to PIB sediments with survival rates generally lower than the control and reproduction was more affected by PIB sediments when compared to the control. Finally, mouthpart deformities of chironimids was another indication of sediment toxicity tested and out of the 90 individuals tested, only one exhibited mouthpart deformities, indicating low toxicity to chirominids.

The diversity and distribution of the PIB benthic community may be limited by the dominance of fine sediments, as measured by the PIB grab samples. Systems dominated by fine sediments exert physical limits on the potential of benthic communities by reducing the density and distribution of food sources, oxygen for respiration and interstitial spaces available that support diverse habitat types (Harrison et al. 2007). Although the metrics for the health of the benthic community target are chemistry and toxicity based, the physical limits affecting the benthic community might be considered in future evaluations of benthic community health as well.

2.3.3.b Fishery Risk Assessment Endpoints, Objectives and Targets

A second ecosystem objective for Presque Isle Bay is to maintain a quality fishery. Several targets and lines of evidence were evaluated for the assessment of the conditions conducive to the survival, growth and reproduction of fish in PIB, as described below:

Target 1 - Water Quality Standards protective of Aquatic Life are met**.** EPA and Pennsylvania water quality standards and criteria for chemicals are based upon toxicity tests and are developed to be protective of aquatic life. Comparison of water quality data for the COPCs to their respective criteria would provide an assessment of potential risks or lack thereof posed by chemicals in PIB. While water samples have historically been collected and analyzed for PIB, the data from these studies were not readily available in published reports. However, the previous studies where samples have been collected concluded that the quality of the water column in PIB was good and that there was no correlation between sediment COPC concentrations and the overlying water column (PA DEP 1992).

Target 2 - At least 90% of the sediment samples from Presque Isle Bay should have benthic conditions necessary to support healthy benthic invertebrate communities to support fish communities. This is the same metric that was evaluated in the benthic invertebrate risk assessment as described above.

Target 3 - The concentrations of bioaccumulative COPCs in the tissue of fish from Presque Isle Bay are not significantly higher than the levels in fish tissues from the same species in Lake Erie.

> Analyses of COPC in fish are not available within PIB for comparison to Lake Erie species, but most of the COPCs are metabolized by fish and not bioaccumulated. PCB and Mercury are regularly assessed contaminants within PIB and the Great Lakes (including Lake Erie) and are the predominant chemicals of concern for bioaccumulation and resulting effects in Great Lakes AOCs. While these chemicals have not been identified as COPCs for PIB, PCB and mercury were used as surrogates within PIB as an indicator of PIB ecosystem exposure to bioaccumulative compounds.

Target 4 – The presence of lesions and tumors in individuals has not diminished the survival, growth and reproduction of the PIB black bullhead population:

- The Bullhead population within PIB represents a single population with little interaction outside of the bay (Millard et al. 2009) so the health of the population is likely responding primarily to internal dynamics including contaminant stressors. Pyron et al. (2001) noted that the overall health of the brown bullhead population in Presque Isle Bay has improved dramatically since 1992. Skin and liver tumor rates have decreased to background levels, the population is reproducing, and the brown bullhead population estimate appears to be stable.
- Kuehn et al. (1995) attempted to establish a correlation between PAH contaminated sediments, instances of liver pathology (although not definitively cancerous) and species diversity and densities of fishes. Kuehn et al. (1995) found that some differences among bullhead species and diversity appeared to exist, although the differences were not significant. Within PIB, no evidence to suggest that the presence of tumors are currently impacting the health, growth, survival, reproduction of fish Pyron et al. (2001).

So that in light of all risk assessment information, PIB appears to provide conditions that support the survival, growth and reproduction for fish as well as other ecosystem receptors.

2.3.3.c Wildlife Assessment Endpoints, Objectives and Targets

A third objective identified as part of protecting and improving the near-shore habitat would be to ensure that COPC concentrations do not pose unacceptable risks to wildlife, particularly birds and mammals. The following targets have been established for this assessment:

Target 1 – Exposure concentrations of COPCs in sediments and benthic fauna that serve as food sources should not pose unacceptable risks to mammals or birds.

COPC data for benthic fauna are not available within PIB. However, exposure of potential wildlife receptors to COPCs in PIB can be estimated using sediment data and ingestion exposure models. Sediment-probing birds (Table 2.2) consume mostly sediment-associated invertebrates and may incidentally ingest more sediment than birds in other feeding guilds. Accordingly, exposure of sediment-probing birds to sediment contamination is expected to be higher than exposure of other groups, such as herbivorous birds and ducks in shallow areas containing such sediments. Further, piscivorous birds and mammals have a high exposure potential to contaminants through the consumption of secondary aquatic consumers, such as invertivorous fish. Several aquatic-dependent bird and mammal species use habitats within the PIB (Table 2.2).

Target 2- The concentrations of bioaccumulative COPCs in the tissue of fish from Presque Isle Bay are not significantly higher than the levels in fish tissues from the same species in Lake Erie. Fish serve as a food source for birds and mammals in Presque Isle Bay. This is the same target as Target 3 for the fish risk assessment.

Target 3 - At least 90% of the near-shore sediment samples from Presque Isle Bay have the conditions necessary to support healthy benthic invertebrate communities to support wildlife that consume benthic invertebrates as a food source.

The evaluations for the Target 3 objectives within near-shore habitats are the same as COPC evaluations conducted to ***Maintain and Protect the Benthic Invertebrate Community***, with a focus on samples collected shallower than 2 meters deep (the finest depth resolution available in GIS within PIB).

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3. RISK ASSESSMENT RESULTS

3.1 SUMMARY OF PREVIOUS INVESTIGATIONS

The PIB SLERA relies heavily on the extensive documentation and efforts of PA DEP, its partners, and other researchers in evaluating the PIB AOC conditions. The dataset and ecological exposure assessments described below have been used to identify the existing lines of evidence that support an interpretation of the COPCs' effect on ecosystem receptors. Table 3.1 summarizes the previous investigations conducted within PIB and the primary components supporting the CSM processes. By linking PIB studies to the CSM, components can be identified that depict the known and unknown pathways to support a weight-of-evidence assessment of COPC impacts and trends relative to ecosystem receptors.

The focus of most of the investigations within PIB has been on the distribution and potential impact of COPCs in sediments. Earlier studies within PIB focused on legacy contaminants in sediments as the potential source of toxicity causing fish tumors (Obert 1993), as opposed to the overlying water column. For example, Obert (1993) sampled water quality above sediments within the Bay and found no clear correlation between elevated levels of sediment chemicals observed and water column chemicals. Diz (2002) describes the water quality in PIB to be satisfactory, and MacDonald (2008) describes legacy contaminants in PIB as the most important routes of COPC exposure. Thus, the bulk of the evaluations of toxicity in PIB have been focused on sediments (Table 3.1 and Attachment 2).

Whole sediment toxicity tests were conducted on PIB sediment samples in 2005 (Kemble et al. 2006) and are summarized in Attachment 1 and used as a line of evidence in the SLERA. Diz (2002) evaluated macroinvertebrate community structure within PIB, but no reference areas were evaluated as part of the study for comparison.

Attachment 1 provides summaries of each of the investigations identified in Table 3.1. The agency and stakeholder efforts examining the PIB AOC conditions are extensive, and the supporting investigations and rationale in the delisting of the dredging BUI in 2006 (PA DEP 2006) provide a comprehensive but not entirely updated source of information for the ERA. Supplemental analyses were conducted to expand and update existing PIB data sets using data gathered from PA Sea Grant and PA DEP (described later in this section).

Table 3.1. PIB AOC investigations evaluating sources, processes and ecological receptors potentially affected by COPCs.

Study (abbreviated title)/Data. (Expanded summaries provided in Attachment 2.)	Conceptual Site Model Components Described		
	COPC Source	**Sediment Processes (Transport and Fate)**	**Receptors**
PIB RAP (1992)	(PAHs primarily) Legacy (in-bay) sediments, SSOs and CSOs (primary), groundwater, nonpoint and atmospheric (secondary)	Suspended sediment inputs and deposition	Fish and Wildlife - No link to fish tumors confirmed, no indication of wildlife impairment
PIB RAP Update (2002)	Legacy (in-bay) sediments	Deposition	Bullheads
Diz (2002) Sediment Quality in PIB	ND	Deposition and burial	Macroinvertebrates, *Dreissena*, gastropods and amphipods
Diz (2005)	SSOs and Tributaries	Deposition and burial	ND
Kemble et al. (2006) PIB toxicity evaluation	Legacy (in-bay) sediments	Deposition	Amphipod and midge
Foyle and Norton (2006) Sediment Loading in PIB	Tributaries, Lake Erie	Erosion, resuspension, deposition (accretion), loading	ND
Gannon University (2007) Atmospheric PAHs in PIB	(PAHs only) Atmospheric	Deposition	ND
MacDonald (2008)	SSOs. CSOs and Tributaries	Deposition	Sediment dwelling organisms (all COPCs), benthic invertivors (Cd, Pb)
Rafferty et al. (2009) Historical review of BUI	PAHs only	Deposition	Bullheads
Blazer et al. (2009) Assessment of BUI on bullheads- Liver neoplasia	PAHs only	Deposition	Bullheads
Blazer et al. (2009) Assessment of BUI on bullheads- Orocutaneous tumors	PAHs only	Deposition	Bullheads
NOAA (2011) Musselwatch Data for Lake Erie (unpublished data)	PAHs and some metals	Deposition	Mussels

PA DEP and its USGS partners compiled a sediment chemistry database containing data from most of the sampling efforts conducted within PIB. The following describes the follow-up evaluations using the PA DEP and USGS data to build upon the evidence of COPCs' effects from sediments within PIB. The sample datasets were evaluated many different ways in an effort to understand and identify sample patterns and trends within and across the Bay. A geodatabase was developed to depict spatial patterns of samples because some samples within the dataset were poorly described by their spatial coordinates and appeared to be located outside the AOC. These samples were eliminated from further analysis. Other samples eliminated from analysis within the AOC analysis included samples located within areas dredged for navigation and mooring. Finally, samples collected by the USACE in the 1980s were deemed questionable for analysis because of a lack of QA/QC procedures for sampling and processing, as well as the poorly described sample locations (Diz, personal communication 2011).

Two primary sediment databases were combined (PA DEP and USGS), and one minor fish tissue dataset (PA DEP) was used to further evaluate sediment concentrations and exposure within PIB. The following describes the datasets, some of which included data from overlapping investigations.

> MacDonald (2008) compiled datasets of sediment chemistry from investigations focused on PIB. Studies dated from 1982, 1986, 1991, 1993, 1994 (two studies), and 2002 (two studies) data collections. Sediment quality conditions were evaluated from each study, and information on the chemical composition of whole sediments was compiled for both surficial and subsurface sediment samples. Samples were divided into three areas of interest: Presque Isle Bay AOC, Presque Isle ponds (outside AOC), and the near-shore areas of Lake Erie. Sediment samples included 212 surficial samples: 157 within the AOC described spatial descriptions of the samples (Diz, personnel communication 2011). The data were structured such that the evaluations were limited to the COPCs described by MacDonald (2008; Table 3.2 above), 38 within the ponds, and 17 near-shore in Lake Erie. Datasets were further evaluated by pre-AOC listing (1982-1991) and post-AOC listing (1992-2001) periods.
>
> PA DEP (2006) – Sediment samples were collected during 2003 and 2005 to support the evaluation of ecosystem health. In 2003, 11 historically sampled locations were resampled using a ponar grab sample within the PIB AOC boundary. In 2005, core samples were collected to attempt to assess temporal trends at four locations. The cores were cut into sections for analysis, and each section was mixed and analyzed. Analysis sections included surface samples at 0-5 cm, 5-10 cm, 10-30 cm, 30-50 cm, and 50-80 cm. Each section was dated using Pb210 and Cs135 isotopes. Additionally, to assess compliance with ecosystem health targets, surficial sediment samples were collected from 32 locations, with 12 samples collected from directed point sampling stations based on historical sampling locations and 20 samples from randomly selected locations. The top 4 inches of sediment was collected using a Van Veen grab sampler.
>
> Fish tissue data were provided by PA DEP. Although the tissue data was collected for the purpose of supporting the fish consumption advisory program (http://www.portal.state.pa.us/portal/server.pt/community/fish_consumption/), the data may offer some insight into the relative level of fish exposure to local

contaminants. The fish tissue dataset included samples collected between 1989 and 2003 within and near PIB. Eighteen species were represented, although a smaller set (N = 9) included species collected within and outside the Bay. The COPCs for PIB (metals and PAHs) are generally not bioaccumulative, so fish tissue data are not typically collected for these parameters and no data exist for PIB fish. However, PCB and mercury fish tissue data were included in the tissue analysis, so these data are assessed in the SLERA for relative exposure comparisons.

3.2 SLERA DATA SUMMARY AND ANALYSIS

The data from the historical investigations were compiled in a geodatabase as part of the SLERA to easily link samples to locations within PIB. Thematic layers of dredged areas and bathymetry were included. Samples located within the dredged layers were eliminated. The bathymetric data were available at 2-meter intervals. This layer was included to identify shallow, near-shore areas across the Bay, recognizing that the 2-meter interval depth exceeds the typical depth of contact for many wading birds and wildlife.

A total of 12 studies looking at sediment chemistry were conducted between 1990 and 2009 and are included in the SLERA database. The data from these studies had differing degrees of spatial coverage and spatial focus. Some studies attempted to sample the same locations or areas as previous studies, while others focused on areas of particular interest (high concentrations of COPCs) for that study. The spatial coverage of surface sampling locations is presented in Figure 3.1. Investigations included surficial sediment and core sediment sampling. For the purpose of this document, the sediment composited over a depth of 0-15 cm was considered surficial sediment. Core data were used, if the resolution of the intervals was deemed to be sufficient, to provide an estimate of sediment quality temporal changes given sedimentation rates in PIB. The spatial location of core data available for analysis is given in Figure 3.2; only the two 2005 core stations were used for analysis. A summary of sediment chemistry data used in this document is provided in Table 3.2.

Table 3.2. Studies included in the SLERA database for PIB.

Study Name	Sample Year	Analytes Present		
		Metals	PAHs	TOC
USFWS 1990	1990	X	X	X
Gannett Fleming, Inc. 1993	1992	X	X	X
PA DEP 1993	1993	X	X	
Battelle 1994a	1994	X	X	X
USACE 1997	1997	X	X	
ECDH 1998	1998	X	X	
USGS 1999	1999	X		
Diz 2002	2000	X	X	X
ECDH 2002	2002	X		X
PA DEP 2003	2003	X	X	X
PA DEP 2005	2005	X	X	X
PA DEP 2009	2009	X	X	X

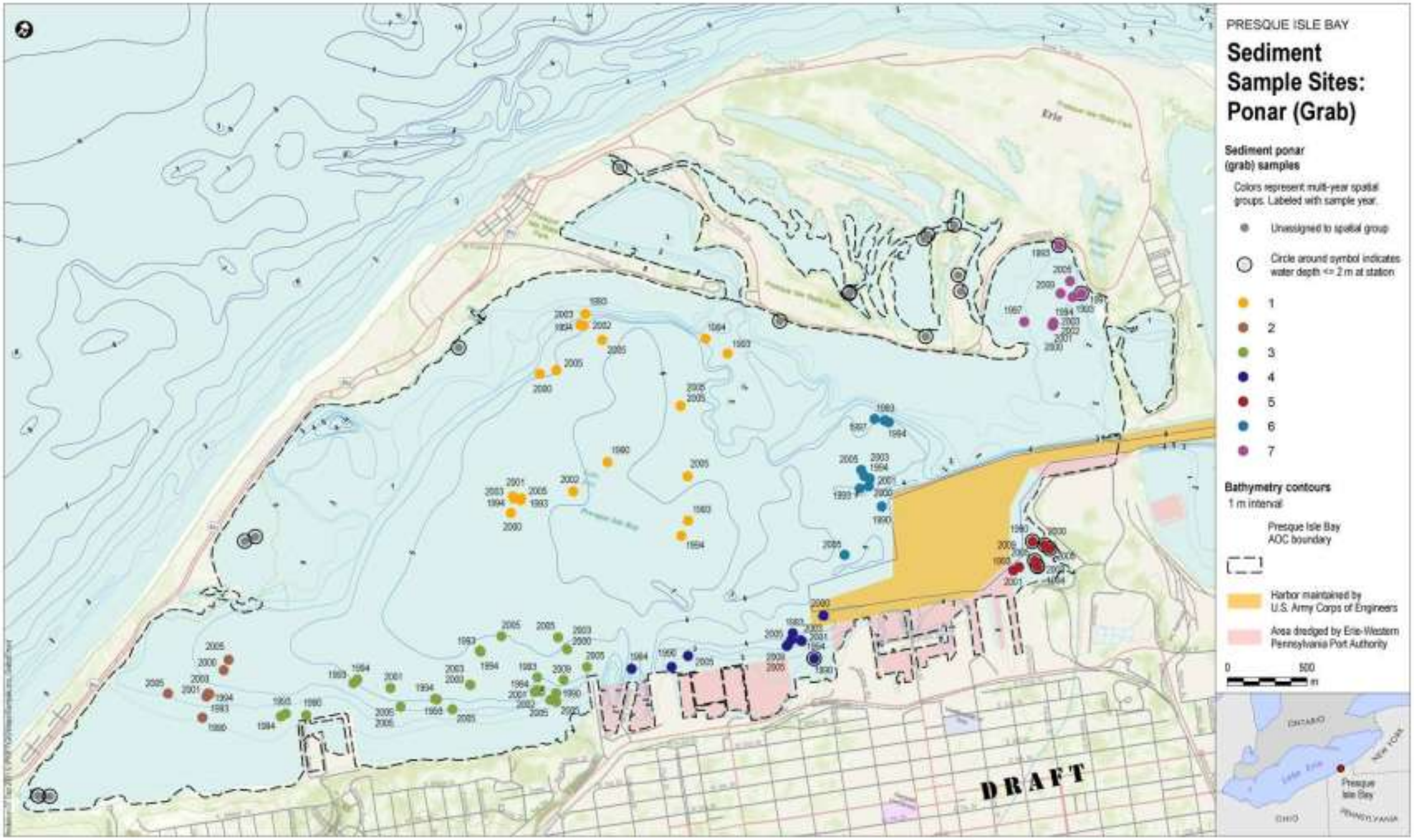

Figure 3.1. Surficial sediment sampling locations within PIB used in study.

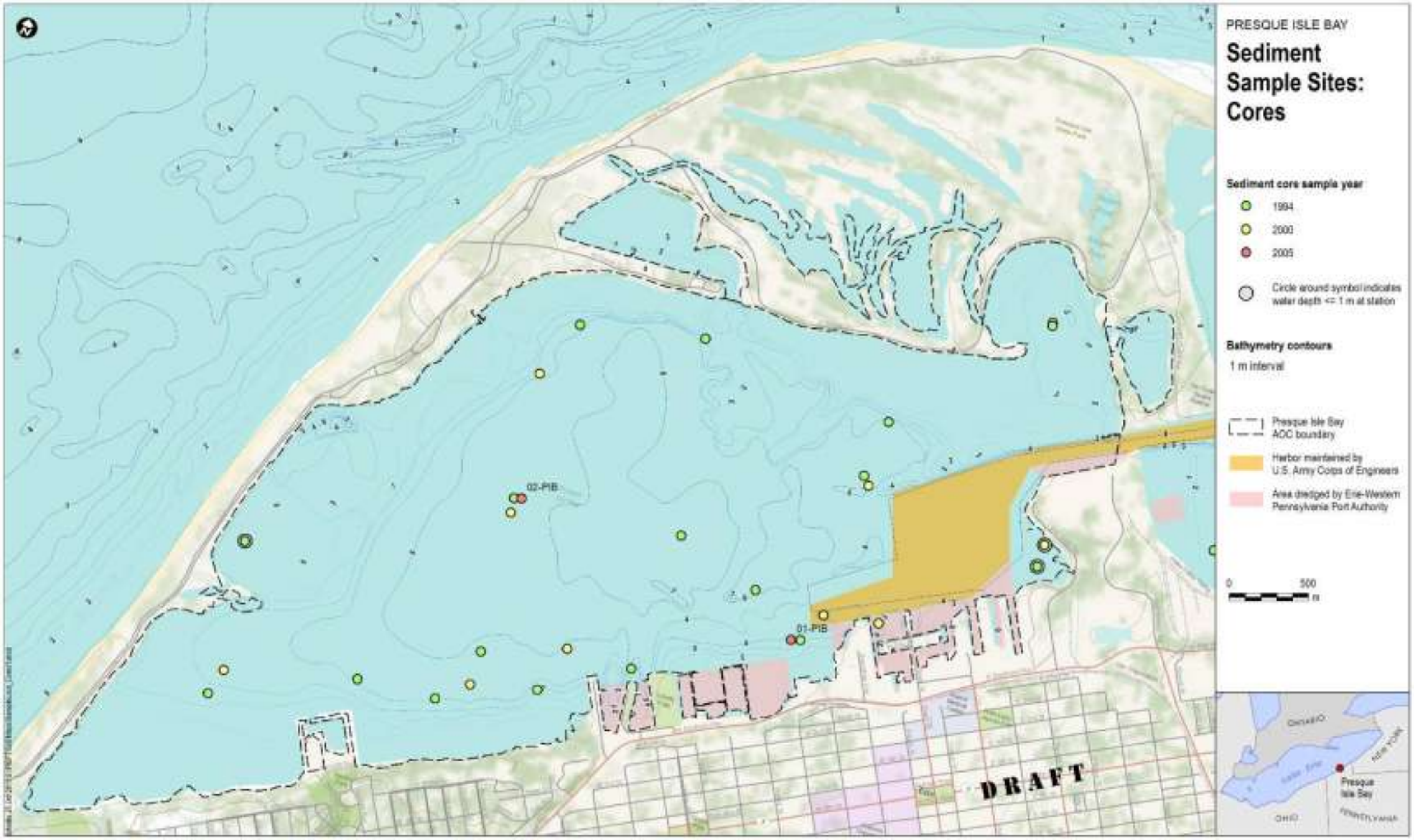

Figure 3.2 Core sediment sampling locations performed in PIB

Data for other exposure media such as the water column, sediment pore water, and food web components are not available. Water column data have been collected historically, but only the conclusions of the studies were reported. For example, Obert (1993) sampled water quality above sediments within the Bay and found no clear correlation between elevated levels of sediment chemicals observed and water column chemicals. Diz (2002) describes the water quality in PIB to be satisfactory. No data have been collected for sediment pore water or food web components. As such, the sediment data serve as the primary basis for the SLERA.

3.3 BENTHIC INVERTEBRATE RISK ASSESSMENT

A number of studies have assessed the sediment COPC data in comparison to freshwater sediment quality guidelines applicable to PIB. The sediment quality guidelines from these studies are presented in Table 3.3. Comparison of sediment COPC data to these guidelines comprise the exposure and effects assessment portion of the SLERA for the benthic invertebrates. These guidelines are not site-specific, are considered conservative, and do not indicate that an effect will be witnessed if the guideline is exceeded (Long et al. 1998). Much of the toxicity data used to develop such guidelines are based on whether effects were observed in bioassays from field-collected samples.

Table 3.3. PIB sediment quality guideline sources used in the ERA.

Description	Study	Magnitude of Effect and Criteria	
		Low	High
Bulk Sediment Quality Benchmarks (Average*PEC-Q)	MacDonald (2008)	<1 and <6 PEC exceedances per sample	>=1 or >=6 PEC exceedances per sample
	Diz (2005)	NA	NA
	PA DEP (2006)	<1 and <6 PEC exceedances per sample	>=1 or >= 6 PEC exceedances per sample
Metals Mixtures (SEM-AVS)	MacDonald (2008)	(SEM-AVS) < 0	(SEM-AVS) > 0
	Diz (2005)	(SEM/AVS) <1	(SEM/AVS) >1
	PA DEP (2006)	(SEM-AVS) < 0	(SEM-AVS) > 0
Metals Mixtures with Organic Carbon (SEM-AVS)/f_{OC}	MacDonald (2008)	<130	>130
	Diz (2005)	NA	NA
	PA DEP (2006)	<3000	>3000
PAH Mixtures (ESB-TU)	MacDonald (2008)	<1	>1
	Diz (2005)	NA	NA
	PA DEP (2006)	<1	>1
*Median is used to reflect central tendency of COPCs			

For this SLERA, the median PEC-Q was the parameter used for analysis of bulk sediment quality. The PEC-Q is the ratio of the concentration of the contaminant to its PEC value. For each sample location, the median PEC-Q was chosen because the data are log-normally distributed and contained many outliers that would bias the arithmetic mean PEC-Q value. The median is a better representation of central tendency (average) of the data and average exposure, so this criterion was used for this assessment (Table 3.3).

3.3.1 Sediment COPC Data

Surficial and core sediments were evaluated for sediment quality in PIB. Most of the studies performed in PIB focused on surface sediment, but the sediment cores collected in 2005 were evaluated to attempt to observe a trend of the chemical concentrations over time.

3.3.1.a Core sediments

Sediment cores were collected in 1994, 2000, and 2005 from the locations depicted in Figure 3.2. Of these, only the two of sediment cores collected and analyzed in 2005 by PA DEP were vertically segmented at relatively fine depth intervals, sub-sampled and analyzed for COPCs. The 1994 and 2000 cores were subsampled at relatively coarse vertical intervals so vertical profiles are not discernible from the data. Plots of the concentrations of COPCs in the two 2005 sediment cores are shown in Figures 3.3 to 3.6. Figure 3.3 and 3.4 show the results of metals and PAHs, respectively from the finely segmented core collected in the near shore location in PIB and Figures 3.5 and 3.6 show the results for the central Bay core. Lines depicting the PECs and TECs are shown on the plots for comparative purposes. Generally, the concentrations of metals in both cores were at a maximum in the 10-30 cm core interval, and have shown a decreasing trend in the surface (< 10 cm) sediments. These profiles suggest that loadings of metals to the Bay peaked years ago and have since declined, and the historically deposited sediment has been buried by more recently deposited sediment with lower metals concentrations. Concentrations of arsenic, cadmium and lead in the surface samples from both cores have declined and are below PECs. Concentrations of nickel and zinc have declined over time as well, but exceed PECs at these two locations. The concentration of PAHs in the near-shore core (Figure 3.4) has a peak in the shallow sediments, indicating a more recent source and/or resuspension and redeposition of surface sediments. The PAH concentrations in the central Bay core (02-PIB) show a slight decrease in shallow sediment concentrations from the maximum sediment concentrations at deeper intervals, consistent with the profile of metals and suggest. Concentrations of PAHs are higher in the near-shore core than at the central Bay location. PECs for PAH constituents are exceeded in the surface sediment at both locations, but the total PAH concentrations are below the total PAH PEC.

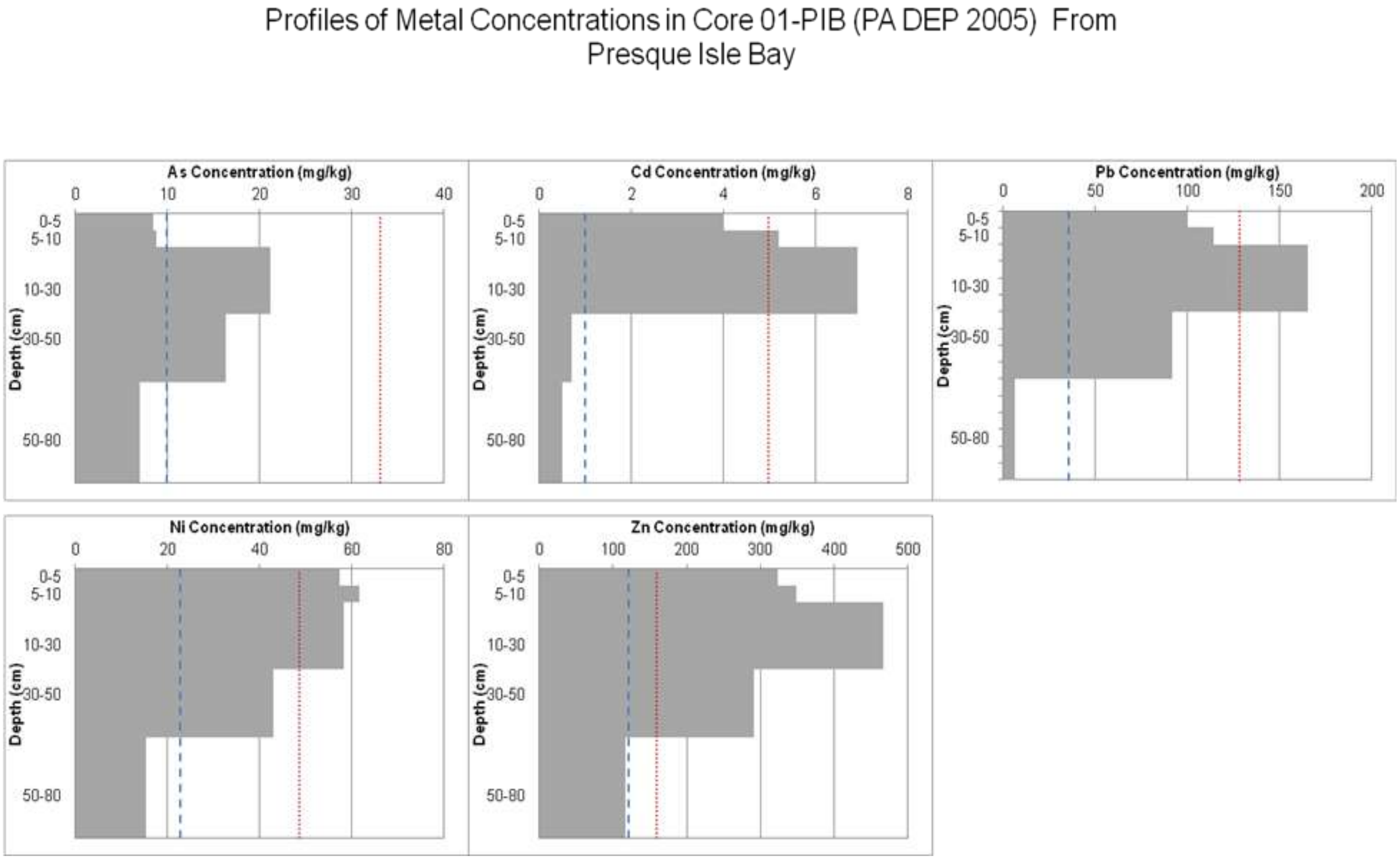

Figure 3.3 Sediment core profiles for metals in 2005 near-shore sampling location.

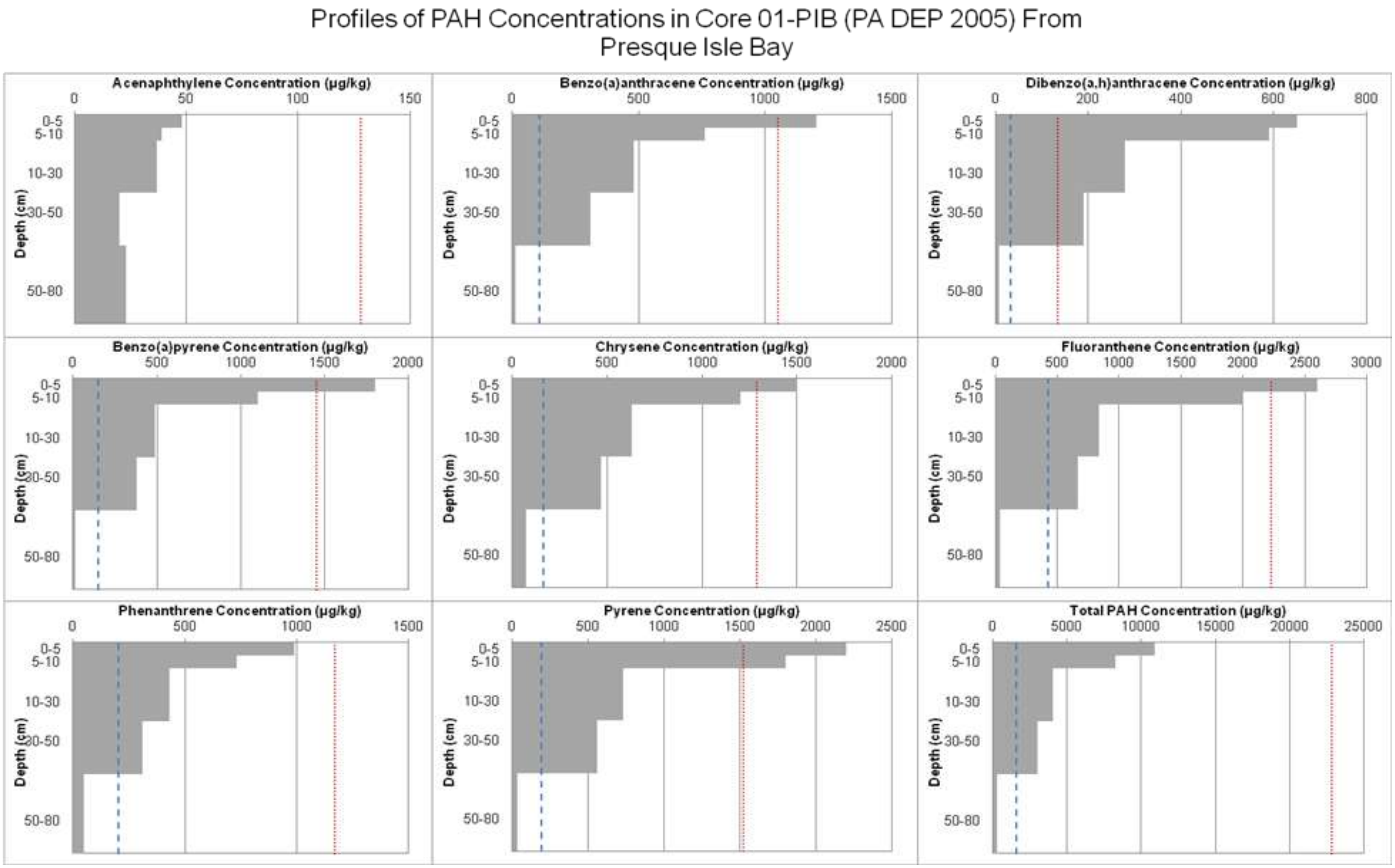

Figure 3.4. Sediment core profiles for PAHs in 2005 in the near shore sampling location.

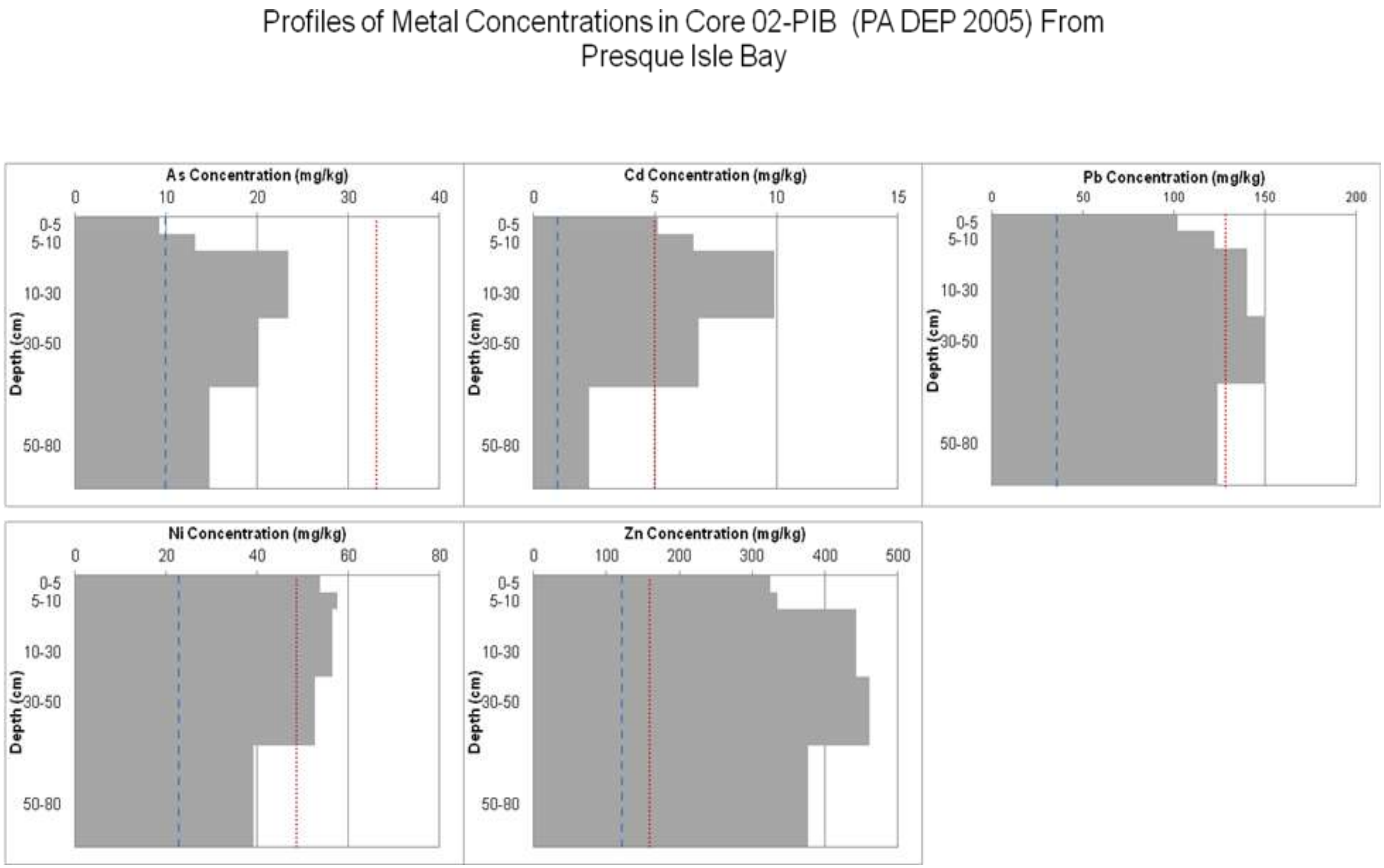

Figure 3.5. Sediment core profiles for metals in 2005 from the central Bay sampling location.

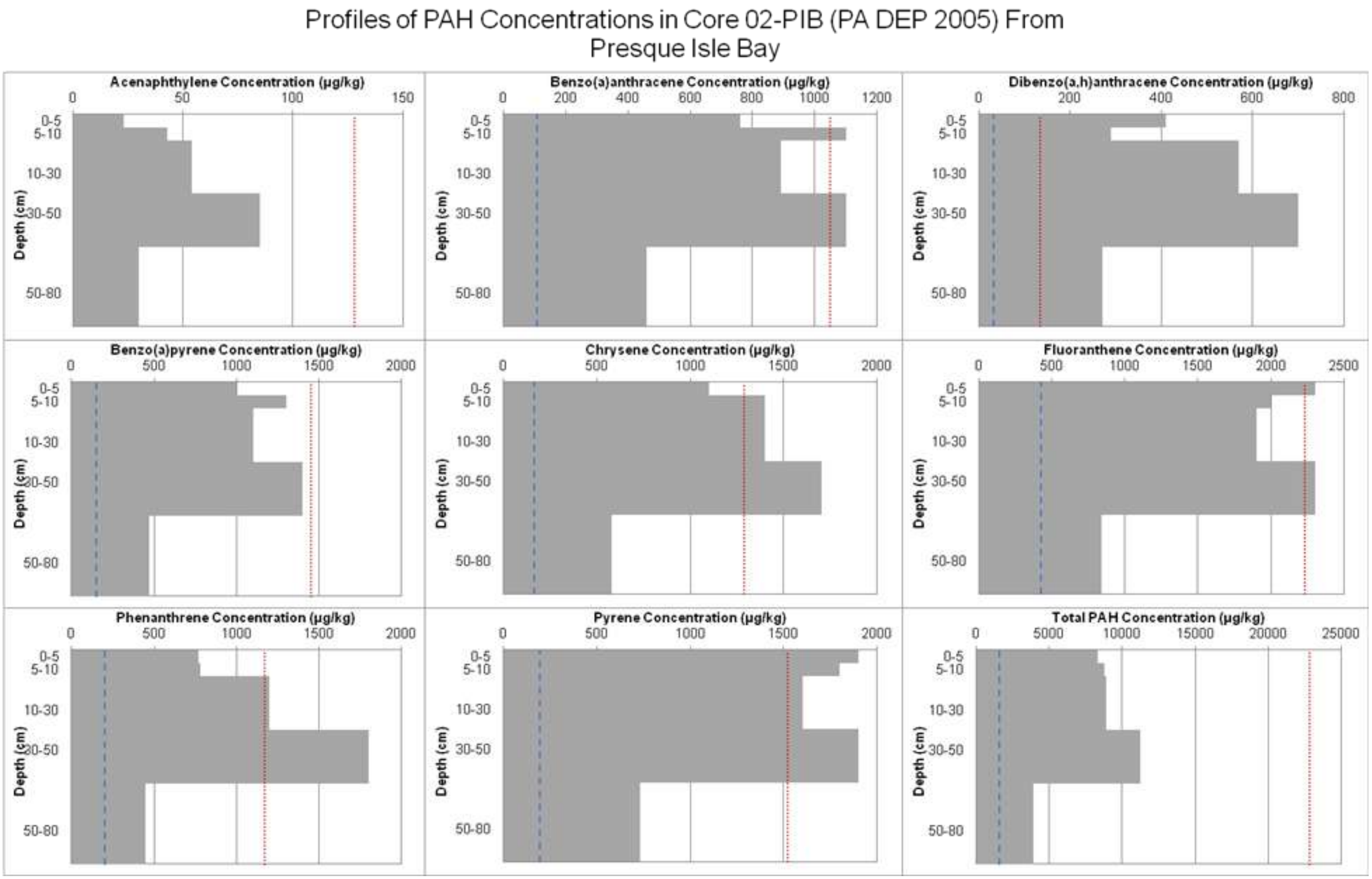

Figure 3.6. Sediment core profiles for PAHs in 2005 central Bay sampling location.

3.3.1.b Surface Sediments

The data available for surface sediment were temporally and spatially varied in PIB. Sampling locations of the surface sediment samples are shown above in Figure 3.1. The COPC concentrations were evaluated in comparison with the metrics presented in Table 3.3, as discussed below for each metric.

PEC-Q

The distribution of concentrations of COPCs in surface sediments are plotted in Figures 3.7 and 3.8, for metals and PAHs, respectively. The “box and whiskers” plots show the range and quartiles of the data for each COPC analyzed as well as the TECs and PECs for each COPC. As is evident in Figure 3.7, with the exception of barium, the majority of metals concentrations in the surface samples collected since 2000 are below PECs. Similarly, the plots presented in Figure 3.8 show that PAH concentrations in the majority of surface sediment samples collected since 2000 are below PECs. Median PEC-Qs were calculated for surface sediment samples collected over time from seven spatial zones (stations) and the combined near-shore areas of PIB shown in Figure 3.1. The median PEC-Q represents the average PEC-Q of all COPCs in a given sample. The results of the spatial and temporal analysis are shown in Figure 3.9. As evident from Figure 3.9, the vast majority of median PEC-Qs were less than 1 for all individual spatial areas and sampling years. No discernible temporal trend was observed, likely reflecting the high variability in surface sediment concentrations and varying sampling objectives and methods from year to year.

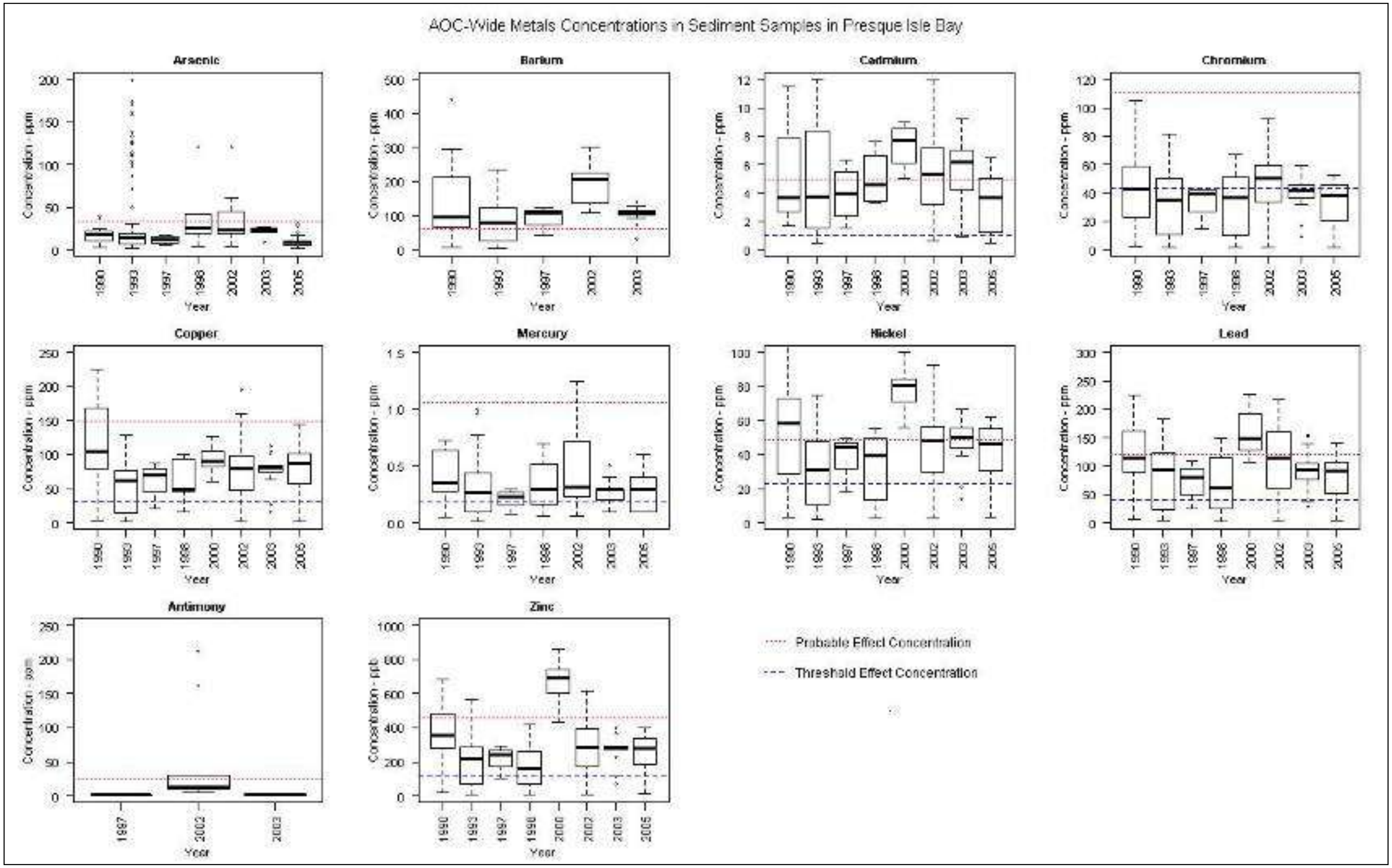

Figure 3.7. Metals concentrations of Bay-wide sediments in PIB.

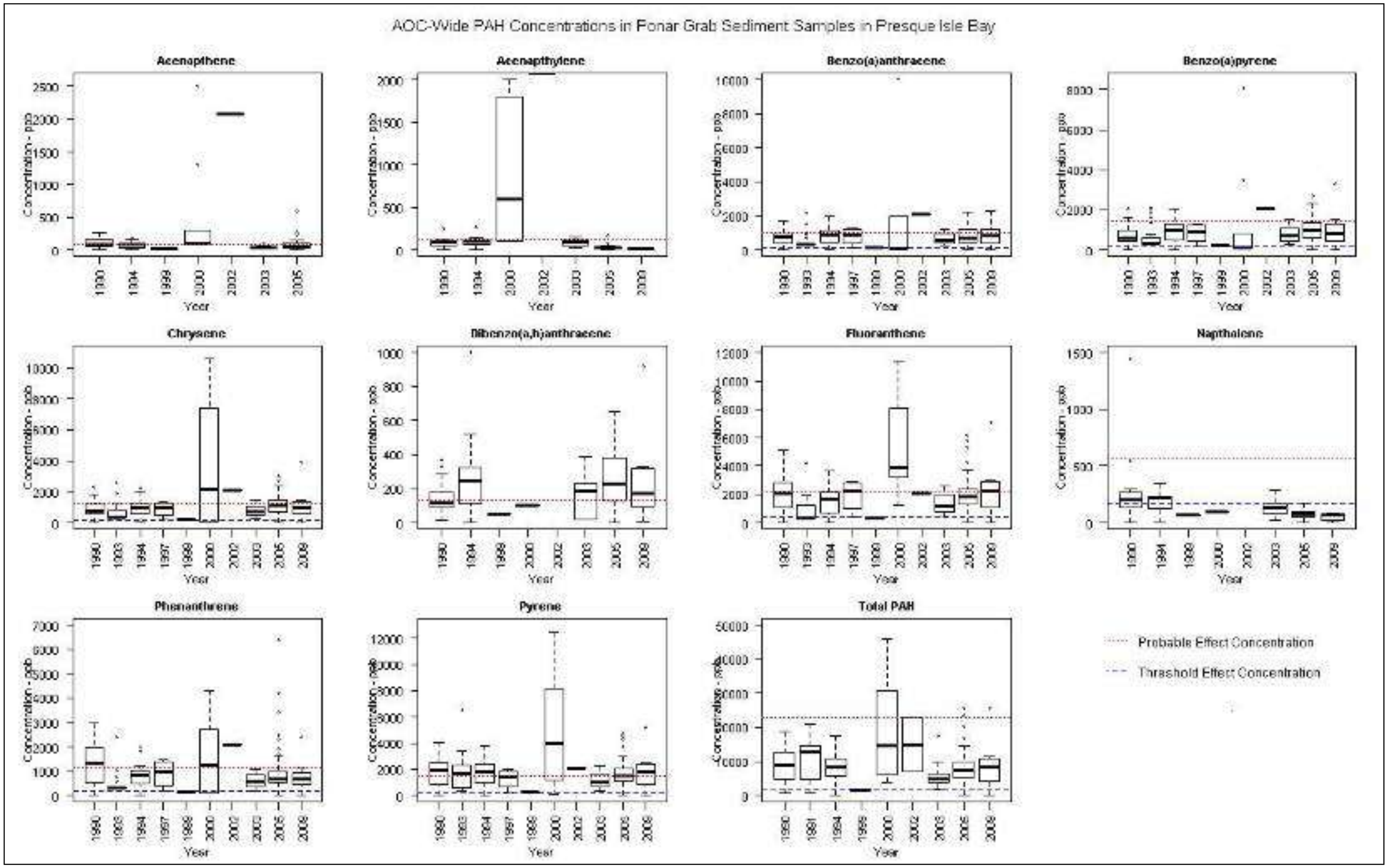

Figure 3.8. PAH concentrations of Bay-wide sediments in PIB.

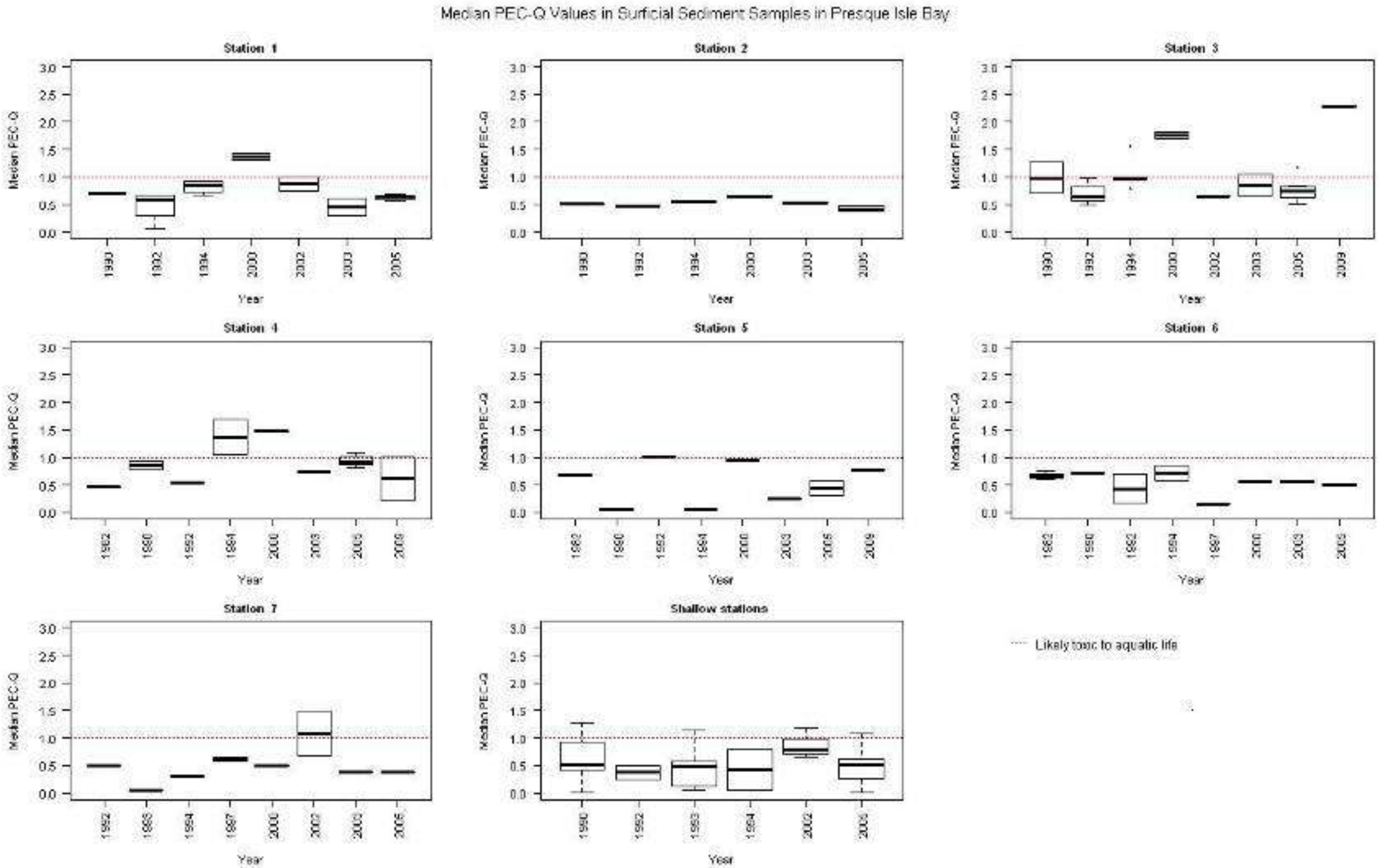

Figure 3.9. Median PEC-Q values for surface sediments in PIB, by station number represented in Figure 3.1, and shallow areas (< 2m).

Table 3.4 shows that the median PEC-Q values across all sampling locations were below the threshold of 1.0, except for the samples collected in 2000. The majority of samples (57%- 100%) had median PEC-Qs that were less than one and the majority of samples had fewer than six PEC exceedances, except for samples collected in 2000. However, the target to achieve median PEC-Qs and fewer than six PEC exceedances for at least 90% of the sampling was not consistently met and was not met in the most recent sampling events. The results for sampling conducted in 2000 were significantly different than results for other years most likely because the 2000 investigation targeted "…sediments from locations identified in previous studies as having high concentrations of contaminants, or having exhibited toxicity in previous testing" (Diz 2002). The focus on potentially highly contaminated sites in 2000 study helps to explain why the samples collected in 2000 had higher concentrations than samples collected from random locations in other years.

Table 3.4. Studies meeting criteria for bulk sediment quality targets

Sample Year	Percent of stations meeting Median PEC-Q criteria	Median PEC-Q Value for All Samples	Stations with >=6 PEC Exceedances	Sample Count	Dataset
1990	55%	0.71	5	11	USFWS 1990
1992	72%	0.54	5	18	Gannett Fleming, Inc. 1993
1993	68%	0.48	0	88	PA DEP 1993
1997	100%	0.58	0	3	USACE 1997
1998	100%	0.23	0	2	ECDH 1998
2000	0%	1.31	7	9	Diz 2002
2002	50%	0.76	2	10	ECDH 2002
2003	78%	0.56	2	9	PA DEP 2003
2005	66%	0.59	10	29	PA DEP 2005
2009	67%	0.60	1	6	PA DEP 2009

(SEM-AVS) and (SEM-AVS)/f_{oc}

To analyze the potential toxicity of metals, the values of (SEM-AVS) and (SEM-AVS)/f_{oc} were calculated for the two studies, Diz (2002) and PA DEP (2005), where AVS data were available. The methods from the Ohio EPA (2010) were used to calculate (SEM-AVS) and (SEM-AVS)/f_{oc} at each station. The target criteria of (SEM-AVS) < 0 from PA DEP (2006) and (SEM-AVS)f_{oc}< 130 from Ohio EPA

(2010) were used. The percentage of samples meeting these criteria can be seen in Table 3.5; the spatial distributions and results of the samples are given in Figure 3.10 for (SEM-AVS) and Figure 3.11 for (SEM-AVS)/f_{oc}.The table and figure show that, generally, the samples meet the target of 90% of samples meeting their respective criteria, even when the targeted high concentration sediments are included. This indicates that the metals concentrations in PIB are meeting acceptable levels.

Table 3.5. Results of analysis for (SEM-AVS) and (SEM-AVS)/f_{OC}

Sample Year	Sample Count	Samples Meeting Criteria (SEM-AVS)	Samples Meeting Criteria (SEM-AVS)/fOC	Dataset
2000	9	67%	100%	Diz 2002
2005	27	93%	93%	PA DEP 2005

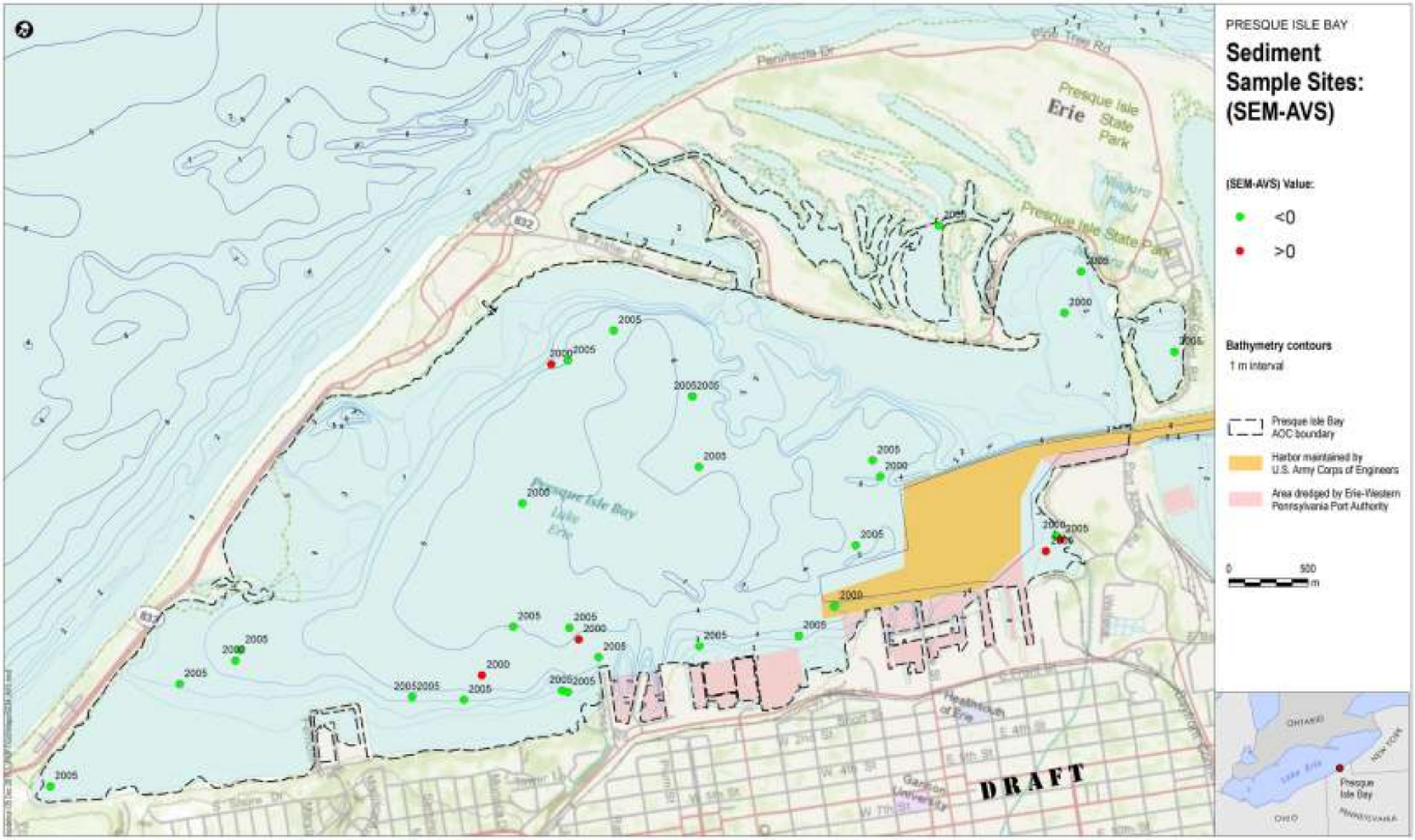

Figure 3.10. Sampling Locations and Results for (SEM-AVS) Analyses in PIB Surface Samples.

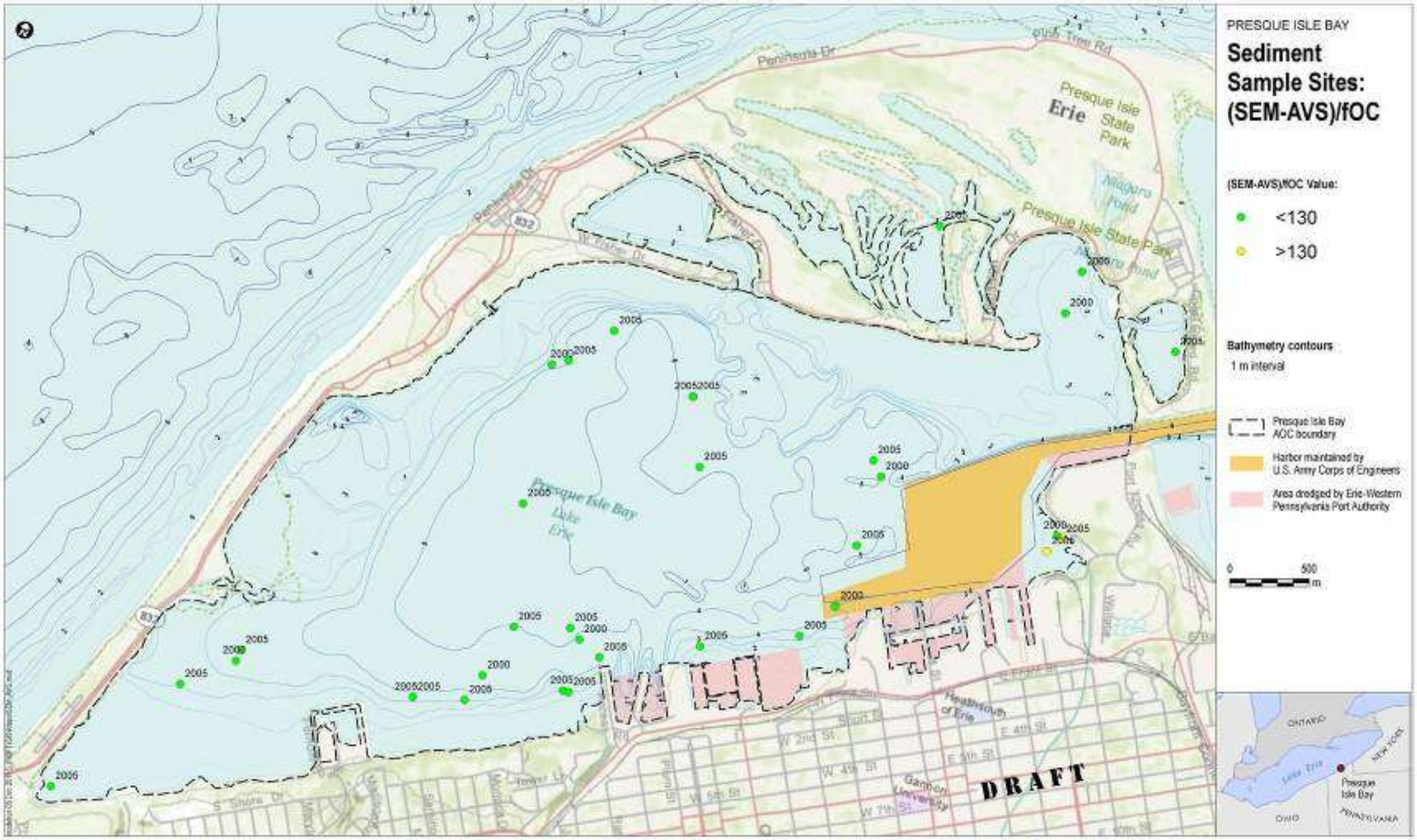

Figure 3.11. Sampling Locations and Results for (SEM-AVS)/f_{OC} Analyses of Surface Samples in PIB.

ESB-TU

As an additional line of evidence to characterize the toxicity of total PAHs and attempt to for sample specific bioavailability, ESB-TUs were calculated for datasets where PAH and sediment TOC data were available. The methods from the Ohio EPA (2010) were used to calculate ESB-TUs at each station. The target criteria of ESB-TUs less than 1.0 from PA DEP (2006) were used. The results of the analysis are shown in Table 3.6 and depicted in Figure 3.12, and indicate that the criteria are essentially met for all sample years with the exception of samples collected in 2000.

Table 3.6. Results of ESB-TU Analysis

Sample Year	Sample Count	Samples Meeting Criteria	Dataset
1990	11	100%	USFWS 1990
1992	18	89%	Gannet Fleming, Inc. 1993
1994	19	95%	Battelle 1994a
2000	9	67%	Diz 2002
2003	9	100%	PA DEP 2003
2005	36	94%	PA DEP 2005
2009	5	80%	PA DEP 2009

However, there is uncertainty in the ESB-TU calculations because only a subset of PAH constituents were typically analyzed for the PIB sediments. The ESB-TU calculation method is based on the analysis of 34 PAHs, and analytical data for only 13 PAHs were consistently available for PIB samples. To characterize the uncertainty, the method specifies uncertainty factors to be applied for different levels of confidence when analysis data for < 34 PAHs are available. However, these uncertainty factors should be locally derived because of the unique distribution of PAHs in contaminant data resulting from their source(s) (Burgess 2009). Establishing locally appropriate levels of uncertainty were outside the scope of this SLERA. Rather, Table 3.7 and Figure 3.13 show the results of the ESB-TU analysis at a 90% confidence level using previously developed, Ohio EPA (2010) data. While the ESB-TU criteria are met for the majority of surface samples without the inclusion of uncertainty factors, they are not met for the majority of samples if the uncertainty factors are included, and therefore attainment of these criteria in PIB is inconclusive.

Table 3.7. ESB-TU Analysis with Inclusion of Uncertainty Factor for 90% Level of Confidence

Sample Year	Sample Count	Samples Meeting Criteria
1990	11	45%
1992	18	22%
1994	19	47%
2000	9	56%
2003	9	26%
2005	36	11%
2009	5	0%

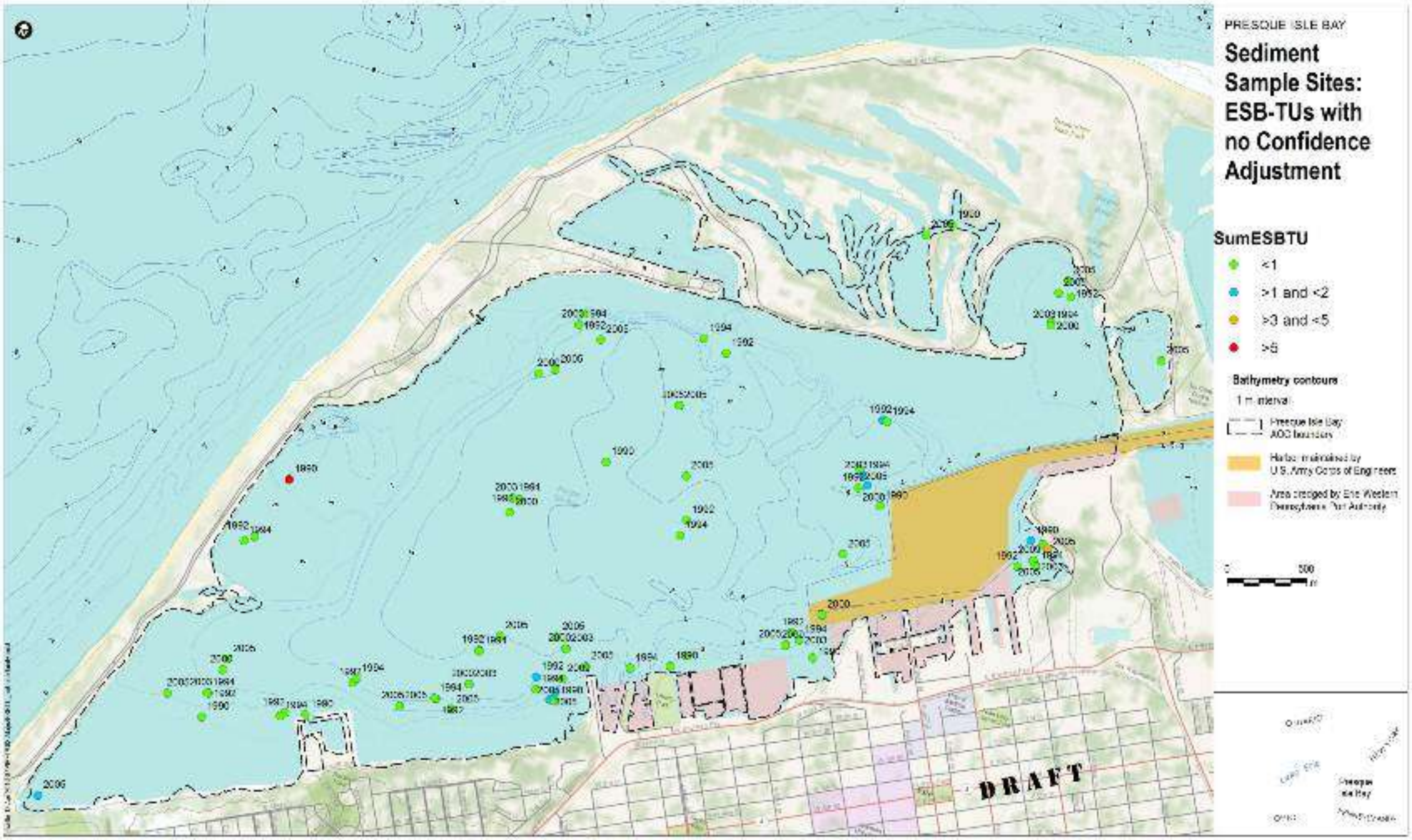

Figure 3.12. Location and Results for ESB-TU Analyses for PIB Sediment

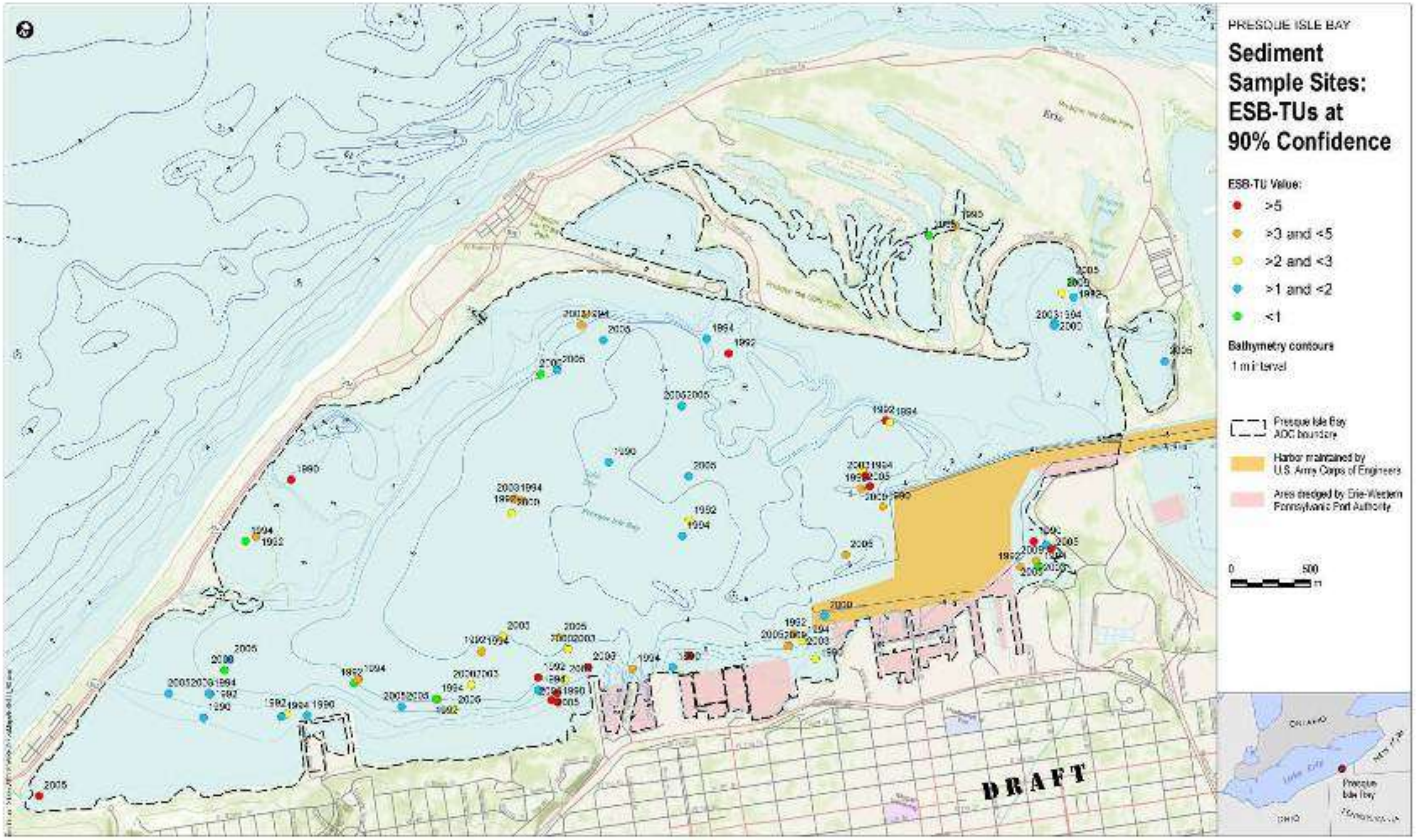

Figure 3.13 – Sample Locations and Results for ESB-TU Analyses with Inclusion of Uncertainty Factors

3.3.2 Whole Sediment Toxicity Test Results

Whole-sediment toxicity was evaluated using the results of 10-d toxicity tests with the midge, *Chironomus dilutus,* and 10- and 28-d toxicity tests with the amphipod, *Hyalella azteca* (Endpoints: survival or growth for both tests) at 21 stations in 2005 (Kemble et al. 2006). The results of the study are presented in Table 3.8.

Table 3.8. Summary of MacDonald (2008) findings of risk of exposure to benthic invertebrates by COPCs within PIB.

Year	Number of Samples	Potential Risk (Percent (n) of samples by risk category) Whole Sediment Toxicity		
		Low	Moderate	High
2005	21	67% (14)	5% (1)	29% (6)

Whole sediment toxicity risks were low at the majority (67%) of sampling locations throughout PIB and that evaluated samples posing a high risk (6 of 21) were located in shallow portions of the Bay (n= 5) and at the confluence of Mill Creek (n=1). MacDonald (2008) reviewed the whole-sediment toxicity tests along with whole-sediment COPC data compared with TRVs and concluded that overall, the potential risks to benthic invertebrates associated with exposure to COPC contaminated sediments were frequently low across PIB. Therefore, potential risks to benthic invertebrates are considered to be low, however isolated locations within PIB may pose a moderate risk to benthic invertebrates

3.3.3 Summary of Risk Characterization for Benthic Invertebrates

The results of the comparisons of PIB sediment data to the various target metrics discussed above is summarized in Figure 3.14. As is apparent from the plots, the majority of sediment COPC data meet the criteria for the various metrics. While all the targets for benthic invertebrates have not been consistently met for 90% of the historical surface sediment samples, all targets have been consistently met for the majority of samples collected over the past decade and the risks to benthic invertebrates from COPCs are low in most areas. As a result of the source controls that have been implemented historically, conditions are expected to continue to improve.

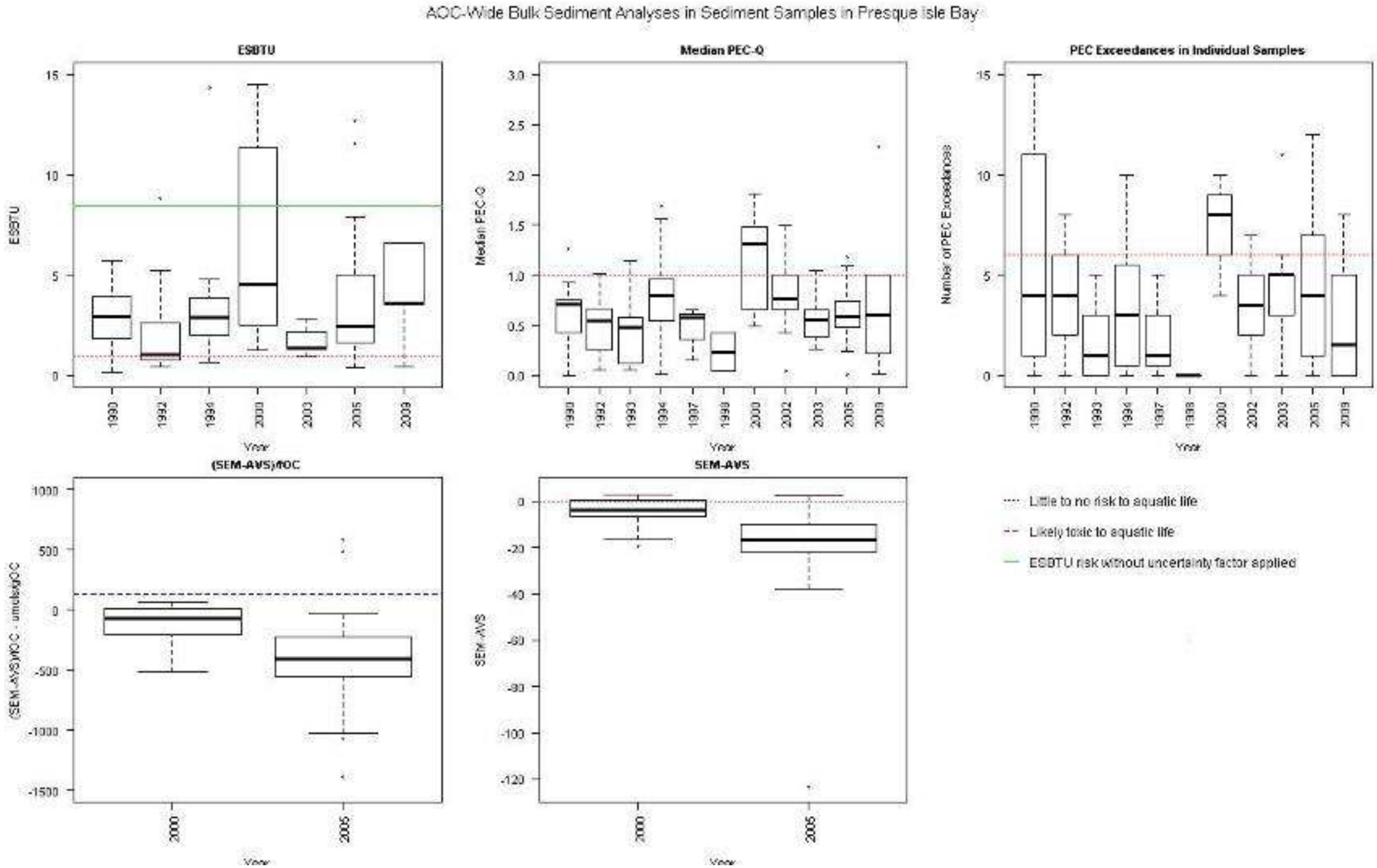

Figure 3.14. Bulk sediment analysis of Bay-wide PIB sediments

3.4 FISHERY RISK ASSESSMENT

Four lines of evidence were evaluated to assess the risks potentially posed by sediment COPCs to the survival, growth and reproduction of fish in PIB, consistent with the ecosystem objectives and targets discussed in Section 2.3.3b., including 1) water quality; 2) benthic fauna health; 3) fish tissue concentrations of COPCs; and 4) prevalence of lesions and tumors. The evaluation of each of these lines of evidence is presented below.

Water Quality

The first line of evidence to assess potential risks to fish in PIB would be an evaluation of the water quality in PIB to determine whether the concentrations of COPCs in water meet water quality criteria that are protective of aquatic species. No quantitative water quality data for the Bay were available for the SLERA, but historical investigation where water samples were reportedly collected and analyzed for PIB, concluded that the quality of the water column in PIB was good and that there was no correlation between sediment COPC concentrations and the overlying water column (PA DEP 1992).

Benthic Fauna

The second target for the protection of the survival, growth and reproduction of fish is to maintain conditions to support healthy benthic invertebrate communities to support fish communities. This is the same metric as was evaluated in the benthic invertebrate risk assessment as described above in the benthic invertebrate risk assessment. As discussed above, while there are localized areas where COPC concentrations in sediment may adversely impact the abundance and diversity of benthic invertebrates, risks from COPCs in sediments are low in most areas. Available information on TRVs for fish in PIB was insufficient to quantitatively evaluate fish exposure and resulting risks to fish posed by COPCs in benthic fauna that serve as a food source.

COPCs in Fish Tissue

Tissue sample, concentration data for fish were evaluated from fish consumption advisory data provided by PA DEP (http://www.portal.state.pa.us/portal/server.pt/community/fish_consumption/). There are no data for concentrations of COPCs in fish tissue, largely because most of the COPCs are not bio-accumulative and are typically not analyzed in fish tissue samples. Only PCB and Mercury were available for comparison. While these compounds (PCB and Mercury) were not identified as COPCs in historical PIB investigations, they are the primary bioaccumulative chemicals of concern for most of the Great Lakes AOCs, as well as contaminated sediment sites throughout the nation. PCB and Mercury are often the compounds of greatest concern to fish, wildlife and human health at contaminated sediment sites. As such, the evaluation of risks from PCB and Mercury in PIB provides a useful surrogate for assessing contaminant risk (or lack thereof) to fish and wildlife in PIB and serves as an indicator of relative risks to other contaminated sediment sites.

While only a handful of sample results for PCB concentrations in fish tissue collected from PIB were between 1998 and 2004 were available for the SLERA, the average PCB concentrations for the available five samples is 0.075 mg/kg with a maximum concentration of 0.28 mg/kg detected in a carp sample collected in 2000. These PCB concentrations are very low relative to fish tissue results for other areas of the Great Lakes and particularly for Lake Erie which are approximately an order of magnitude higher. In contrast to the sparse PCB data, much more data for Mercury concentrations in fish tissue collected from PIB were available. Mercury data for seven fish species were available for comparison of PIB levels with Lake Erie levels and are summarized in Table 3.9. The sample results indicate that there are no significant departures between the concentrations of Mercury measured in PIB fish and Lake Erie fish (Table 3.9) and that the concentrations of Mercury are relatively low.

Table 3.9. Fish tissue samples evaluated for Mercury from PIB and Lake Erie collections.

Range of Sample Dates		No. of Fish	Species	Location	Avg. HG	Max. HG	Min. Hg
11/14/2001	11/14/2001	1	Brown Trout	Lake Erie	0.07	0.07	0.07
9/11/1996	9/11/1996	1		PIB	0.17	0.17	0.17
10/17/1990	9/28/1999	4	Carp	Lake Erie	0.11	0.13	0.07
6/5/1995	8/10/2000	2		PIB	0.09	0.14	0.05
8/6/1996	8/13/2003	4	Freshwater Drum	Lake Erie	0.14	0.28	0.04
6/20/1995	6/20/1995	1		PIB	0.19	0.19	0.19
10/15/1993	10/13/2004	8	Smallmouth Bass	Lake Erie	0.19	0.29	0.08
6/5/1995	6/5/1995	1		PIB	0.14	0.14	0.14
10/24/1989	9/17/2002	7	Walleye	Lake Erie	0.27	0.44	0.07
6/20/1995	6/20/1995	1		PIB	0.15	0.15	0.15
8/20/2003	8/20/2003	1	White Perch	Lake Erie	0.09	0.09	0.09
6/20/1995	6/20/1995	1		PIB	0.16	0.16	0.16
8/1/1989	2/14/2010	16	Yellow Perch	Lake Erie	0.08	0.14	0.02
10/17/1990	10/25/1996	4		PIB	0.06	0.11	0.03

Fish Tumor and Lesion Prevalence and Population Level Effects

Although several studies have aimed to link causal effects of sediment PAH exposure to lesion prevalence, few have attempted to link lesion and tumor prevalence to adverse effects at the population or higher trophic level. Brown bullhead studies conducted on the Black River in Ohio reported liver histopathology data that suggested a link between sediment PAH concentrations, liver lesions, and population age structure (Baumann 2000). For example, Baumann (2000) noted a truncated age structure in the Black River population examined during the contaminated study period, whereby few individuals in the population survived beyond 4 years of age. Following site remediation (e.g., PAH removal), the cancer prevalence decreased along with the associated reference populations absent of PAH contamination.

In a study of English sole, Johnson and Landahl (1994) examined the relationship between lesion prevalence and population-level effects by comparing estimated annual mortality rates at both highly contaminated (e.g., Eagle Harbor) and uncontaminated sites throughout Puget Sound. English sole mortality rates from contaminated sites associated with high liver lesion prevalence were not found to be significantly greater than mortality rates for English sole from Puget Sound as a whole. The investigators also examined the population structure and found no evidence of increased age-related mortality in fish with lesions or in populations associated with areas of high concentrations of PAHs and PCBs. The authors concluded that fish populations that have high incidence of lesions do not necessarily have increased mortality. Other factors that affect English sole populations, such as fishing pressure, predation, and fluctuations in food supply, may mask population-level effects associated with chemical contamination and lesion incidence. Thus, Johnson and Landahl (1994) did not identify a link between lesion prevalence and population structure in areas with widely varying ranges of PAH concentrations in sediment, so the relationship remains uncertain.

Alternatively, a recent study by Breckles and Neff (2010) suggested that the historically contaminated (including PAHs) sites in the Detroit River have resulted in populations (such as bullhead) and an ecosystem that has adapted to and is tolerant of the legacy contaminant conditions, suggesting an evolved ecosystem response. Breckles and Neff (2010) also noted that more focused assessments at the community level are warranted, but their observations are worth noting nonetheless.

While these describe tumor incidences of benthic fish exposed to PAH-contaminated sediment, they are not conclusive with respect to population or higher-level effects due to this exposure. The incidence of abnormalities in fish remains a challenge to attribute to a single factor and is likely to result from confounding factors, including species, age, disease, organic matter, temperature, nutrition, season, and geographic location in addition to contaminants and catch methods (Adams et al. 1996). Because of the highly qualitative nature of the field health observations and the uncertainties associated with their interpretation, a conclusive link between the field observations of tumor prevalence and the affect on the population and community levels is lacking.

3.5 WILDLIFE RISK ASSESSMENT

For the wildlife risk assessment, three lines of evidence were evaluated to assess the potential risks in PIB and determine whether the targets discussed in Section 2.3.3c are met. These include: 1) risks from ingestion exposure to COPCs in sediments and benthic fauna that serve as food sources; 2) the effects of COPCs on benthic community health; and 3) the risks posed by COPCs that bio-accumulate in fish that serve as a food source for PIB wildlife.

The second and third lines of evidence were evaluated above as part of the benthic invertebrate and fish risk assessments. This section presents the evaluation of the first line of evidence, risks from exposure to COPCs to wildlife that feed on benthic fauna.

COPC data for benthic fauna are not available within PIB. However, exposure of potential wildlife receptors to COPCs in PIB can be estimated using sediment data and exposure models. The exposure assessment, effects assessment and risk characterization for wildlife is presented below.

3.5.1 Exposure Assessment

Exposure of avian and mammalian receptors to chemicals in Presque Isle Bay were estimated using sediment data from near-shore areas with overlying water depths of less than 2 meters. Chemical ingestion exposure of these organisms was expected to occur through food consumption and incidental sediment ingestion, because the chemical accumulation of their prey is expected to be through sediment exposure. Water data were not available for this analysis, but water exposure was expected to be insignificant when compared to exposure from food and sediment ingestion. Exposure was assessed for the following representative species.

Piscivorious Mammals - The mink is the species most represented by piscivorious mammals in PIB. Mink will feed on both fish and aquatic invertebrates, though it is assumed for calculations that the diet of mink is completely of fish. The process described by Battelle (2002) was used to calculate the ingestion rate of piscivorious mammals.

Insectivorious Waterfowl - The mallard duck and spotted sandpiper are the species most represented by insectivorious waterfowl in PIB. The process described by Battelle (2002) was used to calculate the ingestion rate of insectivorious waterfowl. The fraction of diet of insectivorious waterfowl composed of invertebrates was considered to be 75% for calculation of ingestion rate.

Piscivorious bird – The Great Blue Heron represents the wading, piscivorious avian species in PIB. The process described by Battelle (2002) was used to calculate the ingestion rate of the Great Blue Heron.

Chemical exposures of avian and mammalian wildlife receptors were evaluated by estimating daily oral doses. These doses were expressed as milligram chemical per kilogram body weight per day (mg/kg/d). Accordingly, estimates of receptor ingestion rates and body weights were required so cconservative ingestion rates and body weight assumptions required.

To calculate the ingestion rates for wildlife, COPC concentrations needed to be calculated in benthic invertebrates and fish based on the sediment concentrations. The chemical concentrations in fish were calculated based on the methods used by Battelle (2002), which also required the calculation of estimated concentrations in benthic invertebrates.

The estimated concentration of each COPC in benthic invertebrates was calculated using the following equation:

$$C_b=(C_s / f_{OC}) \times BSAF \times fL$$

Where:

C_b =	Concentrations of COPC in benthic invertebrates (mg/kg –wet weight)
C_s =	Concentration of COPC in sediment (mg/kg dry weight)
f_{OC} =	Fraction of organic carbon content of sediment
$BSAF$ =	Biota Sediment Accumulation Factor (mg/kg-OC/mg/kg lipid) (Metals assumed value of 1)
fL =	Conversion factor to convert lipid-normalized body burden to a wet-weight concentration (mg/kg-lipid/mg/kg-wet-weight) (assumed to equal 0.01)

The estimated concentration of each COPC in fish was calculated using the following equation, assuming fraction of diet of fish composed of benthic invertebrates is one.

$$C_{fs} = (C_b \times IR \times AF)/(GR+ER)$$

Where:

C_{fs}=	Estimated COPC concentration in fish from the ingestion of benthic invertibrates (mg/kg-wet-weight)
C_b =	Estimated concentration of COPC in benthic invertebrates (mg/kg-wet-weight)
IR =	Ingestion rate of fish (kg/kg-day) (Assumed 0.05)
AF =	Absorption factor of COPC (Metals assumed value of 1)
GR =	Growth rate (equivalent to $0.01 \times (BW)^{-0.2}$
ER =	Excretion rate (equivalent to 0.25×IR

Using the measured and estimated concentrations for COPCs in sediment, benthic invertebrates, and fish, the estimated daily intake of each COPC for the Great Blue Heron was calculated using the following equation:

$$DI = [(C_{fs} \times IR_f)+(C_b+IR_b)+(C_s \times IR_s)]/BW$$

Where:

DI =	Daily intake (mg/kg-d)
C_{fs}=	Estimated concentration of COPC in fish (mg/kg-wet-weight)
IR_f =	Ingestion rate of fish by end species (kg/kg-day) (Using EPA 1993)
C_s =	Measured concentration of COPC in sediment (mg/kg-dry-weight)
IR_b =	Ingestion rate of benthic invertebrates by end species (kg/kg-day) (Using EPA 1993)
C_b =	Measured concentration of COPC in benthic invertebrates (mg/kg-dry-weight)
IR_s =	Sediment ingestion rate (based on EPA 1993)
BW =	Body weight (kg)

3.5.2 Effects Assessment

To study the health of wildlife in the near-shore area of the AOC, the Hazard Quotients for piscivorious mammals (mink), insectivorious waterfowl (mallard duck), probing birds (spotted sandpiper), and piscivorious birds (great blue heron) were analyzed using the process described by Battelle (2002). The hazard quotient is the ratio of the COPCs ingested to the "no observed adverse effect level" (NOAEL) and "lowest observed adverse effect level" (LOAEL), provided by EPA (EPA 2008). Hazard quotients (HQs) were calculated for near-shore sample sites that had an overlying water depth of less than two meters. The spread sheet calculations for the HQs for the COPCs for both the NOAELs and LOAELs for the three representative receptors (mink, mallard duck, and Great Blue Heron) are presented in Attachment 2.

3.5.3 Risk Characterization

The following risk criteria from Battelle (2002) were adopted for the purposes of characterizing risk to wildlife posed by contaminated sediments:

- Low: Samples where HQs for all COPCs were less than 1.
- Medium: Samples where HQs for no more than 3 COPCs were greater than 1 and all HQs were less than 10.
- High: Samples where HQs for more than 3 COPCs were greater than 1 or at least 1 HQ was greater than 10.

Since the HQ is the ratio of the contaminant ingested to an effect level, low risk is desired because it suggests that the amount of contaminant ingested is less than the adverse effect level. These categories were selected on approaches used by Battelle (2002) using a much more robust dataset. The Battelle (2002) approach and target for low risk HQ is assumed useful for assessing relative risks of COPC ingestion to PIB wildlife. It is important to note that, lacking locally collected data on COPC concentrations of potentially ingested fish and benthic invertebrates, the calculated estimates are assumptions, included for relative comparison purposes, and may not reflect local or regional levels of risk. HQ values should only be use for relative risk comparison by species among sample periods and not among species.

The percentages of stations with hazard quotients that meet the criteria for each study are given in Table 3.10. Table 3.10 evaluations include samples from 1990 (n=5), 1992 (n=2), 2002 (n-1), and 2005 (n=6). Figures 3.15 through 3.20 depict the spatial distribution of sediment samples and respective estimated NOAEL and LOAEL risk levels for mink, mallard duck, and Great Blue Heron. A plot of hazard quotients for each study and effect level is given in Figure 3.21.

Table 3.10 shows that, for the two periods of larger samples (1990 (n=5) and 2005 (n=6)), in 1990, the risk criteria for all endpoint species is Low or Medium for both the LOAEL and NOAEL criteria. The risk criteria for 2005 samples for all endpoint species is Low or Medium for both the LOAEL and NOAEL criteria. The relative difference between the two samples finds the 2005 samples with a greater percentage of low risk categorized samples among all wildlife species for both LOAEL and NOAEL than those of 1990. Although the overall sample sizes are relatively small, the calculations do suggest a slight decrease in risk from medium to low between the 1990 and 2005 sample periods.

Table 3.10. Percentage of samples at risk criteria by endpoint species and risk level

Sample Year	Number of Samples	Risk Criteria	Great Blue Heron		Mink		Mallard	
			LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL
1990	5	Low	20%	20%	20%	20%	100%	60%
		Med	80%	80%	80%	80%	0%	40%
		High	0%	0%	0%	0%	0%	0%
1992	2	Low	0%	0%	0%	0%	100%	100%
		Med	100%	50%	100%	100%	0%	0%
		High	0%	50%	0%	0%	0%	0%
2002	1	Low	0%	0%	0%	0%	100%	0%
		Med	100%	0%	100%	0%	0%	100%
		High	0%	100%	0%	100%	0%	0%
2005	6	Low	50%	33%	67%	33%	100%	83%
		Med	50%	67%	33%	67%	0%	17%
		High	0	0	0%	0%	0%	0%

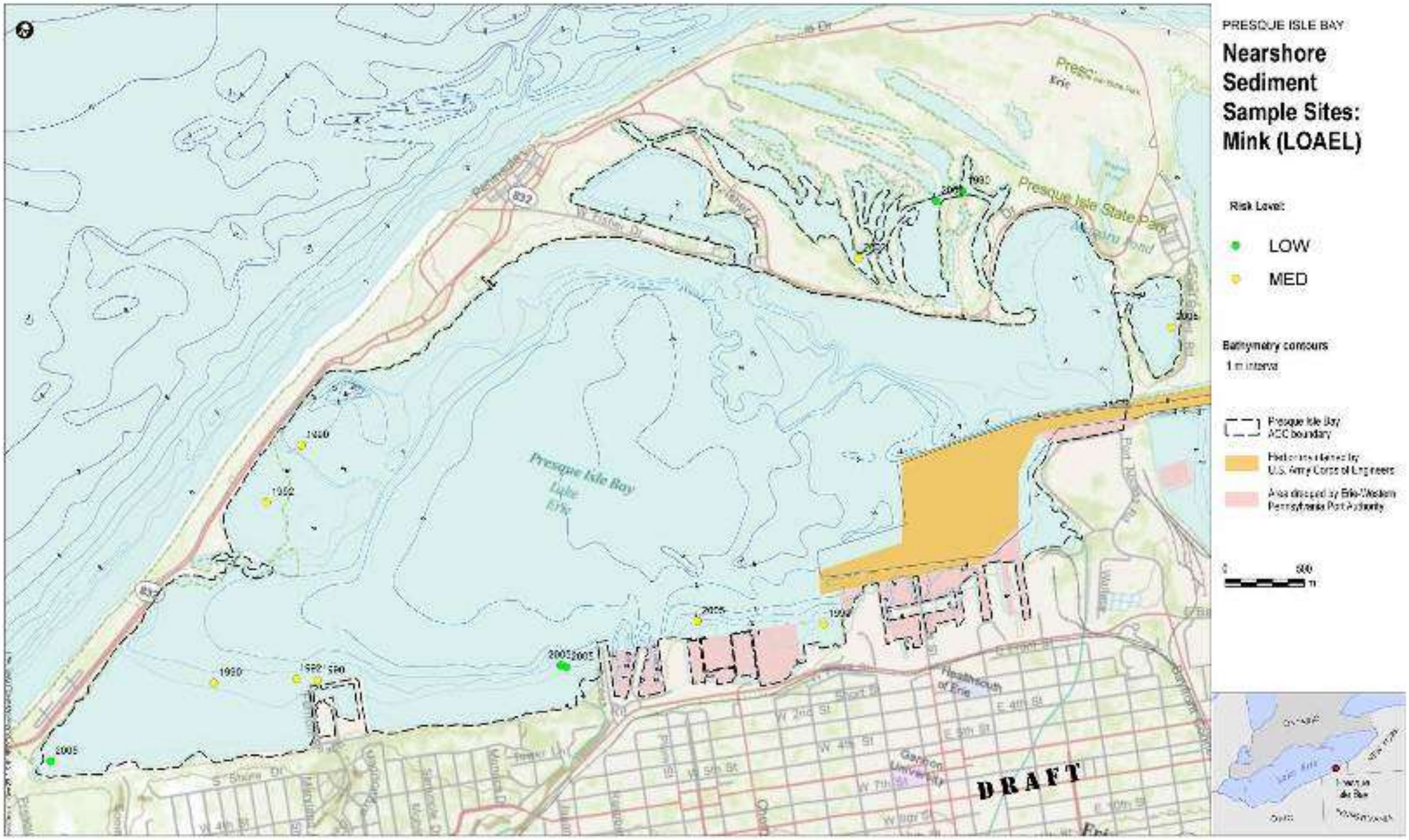

Figure 3.15. Risk levels for Mink at LOAEL toxicity reference value.

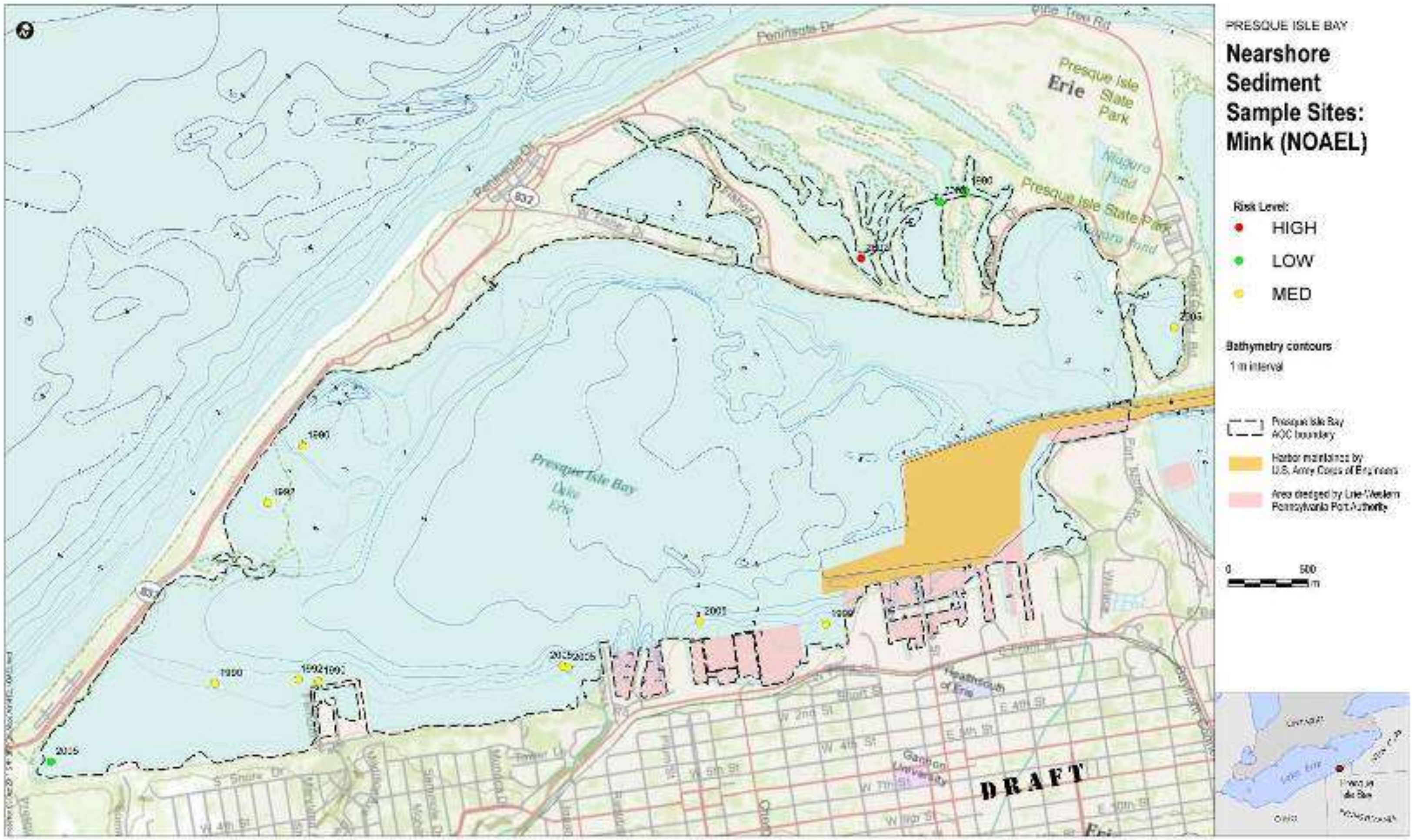

Figure 3.16. Risk levels for Mink at NOAEL toxicity reference value.

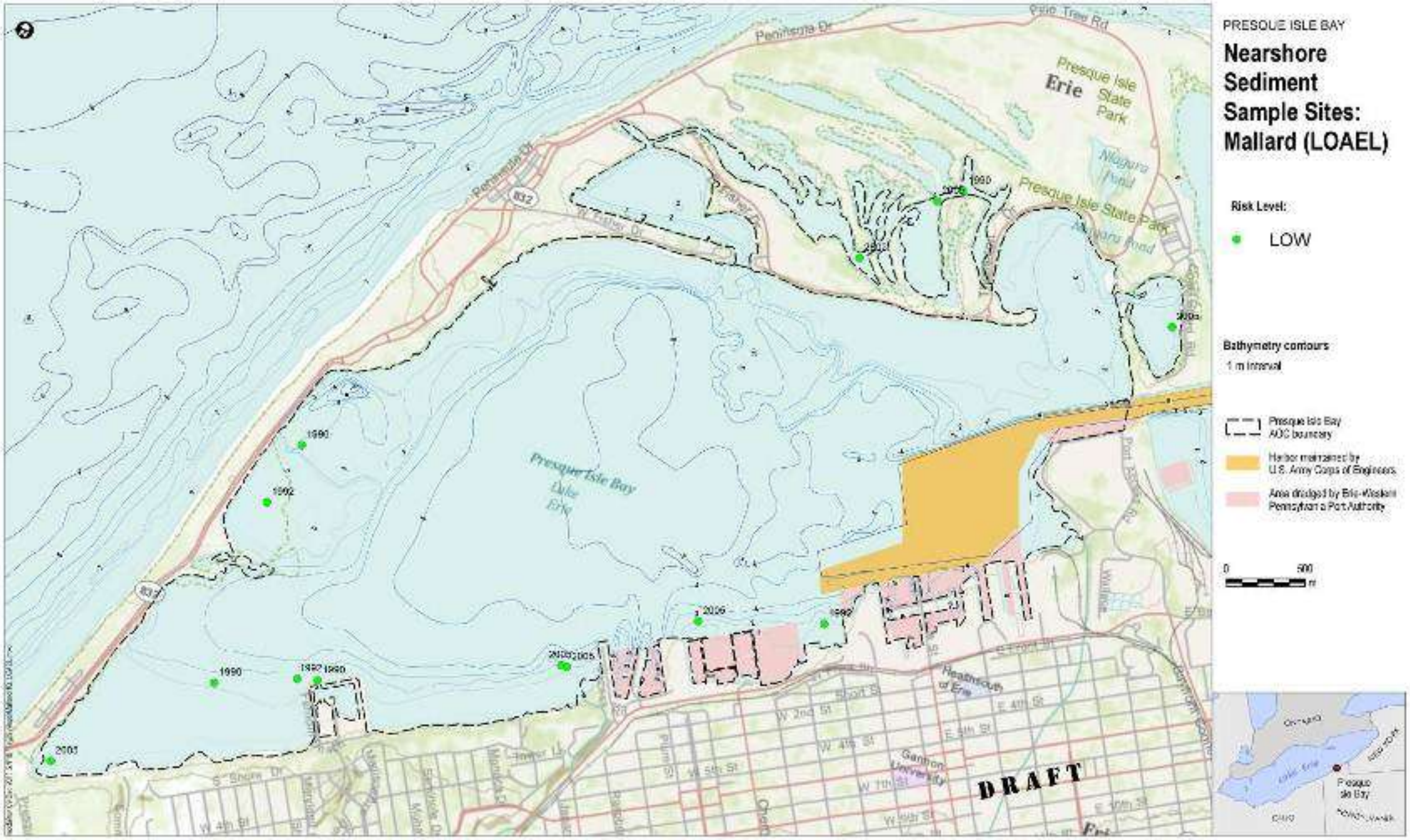

Figure 3.17. Risk levels for Mallard at LOAEL toxicity reference value.

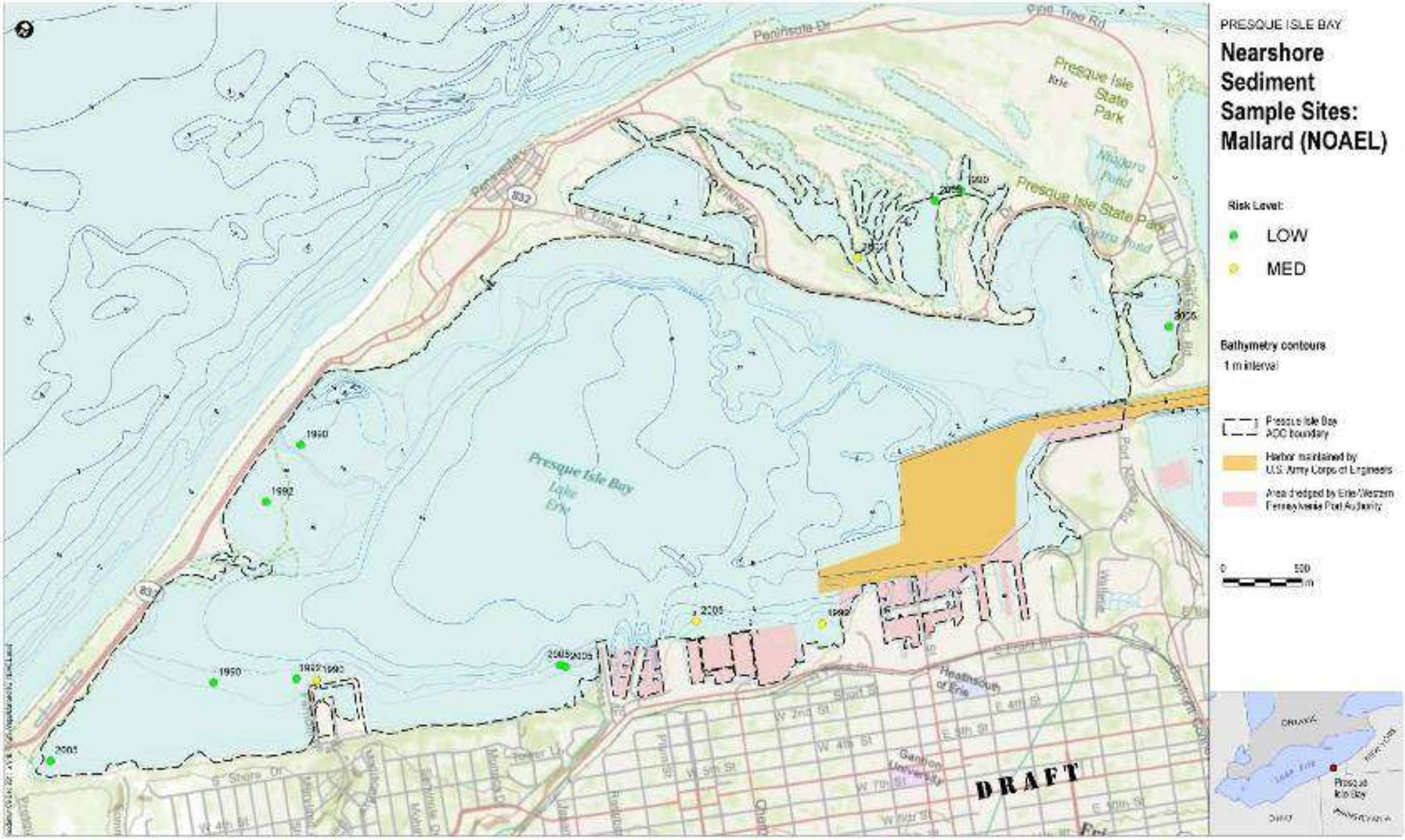

Figure 3.18. Risk levels for Mallard at NOAEL toxicity reference value.

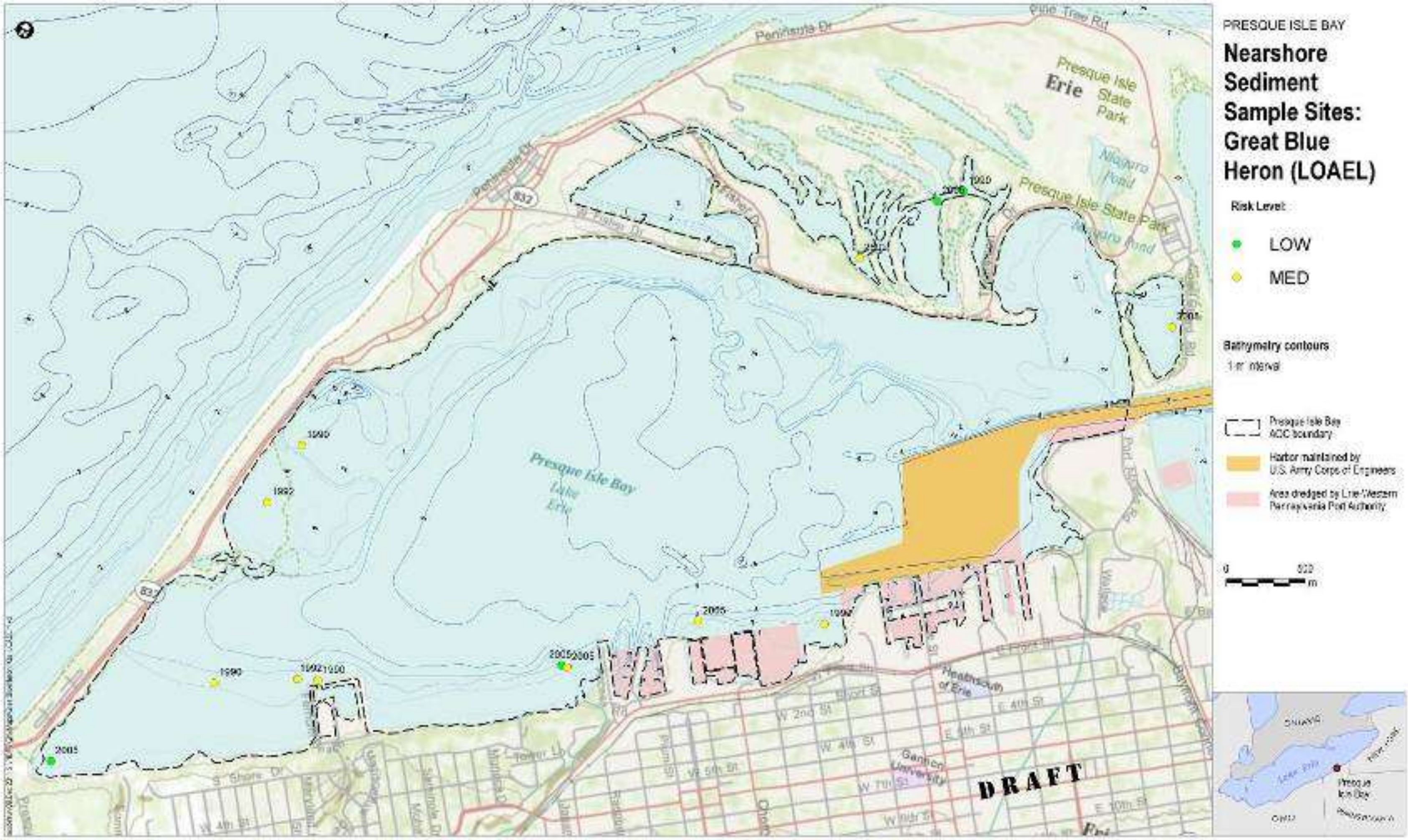

Figure 3.19. Risk levels for Great Blue Heron at LOAEL toxicity reference value.

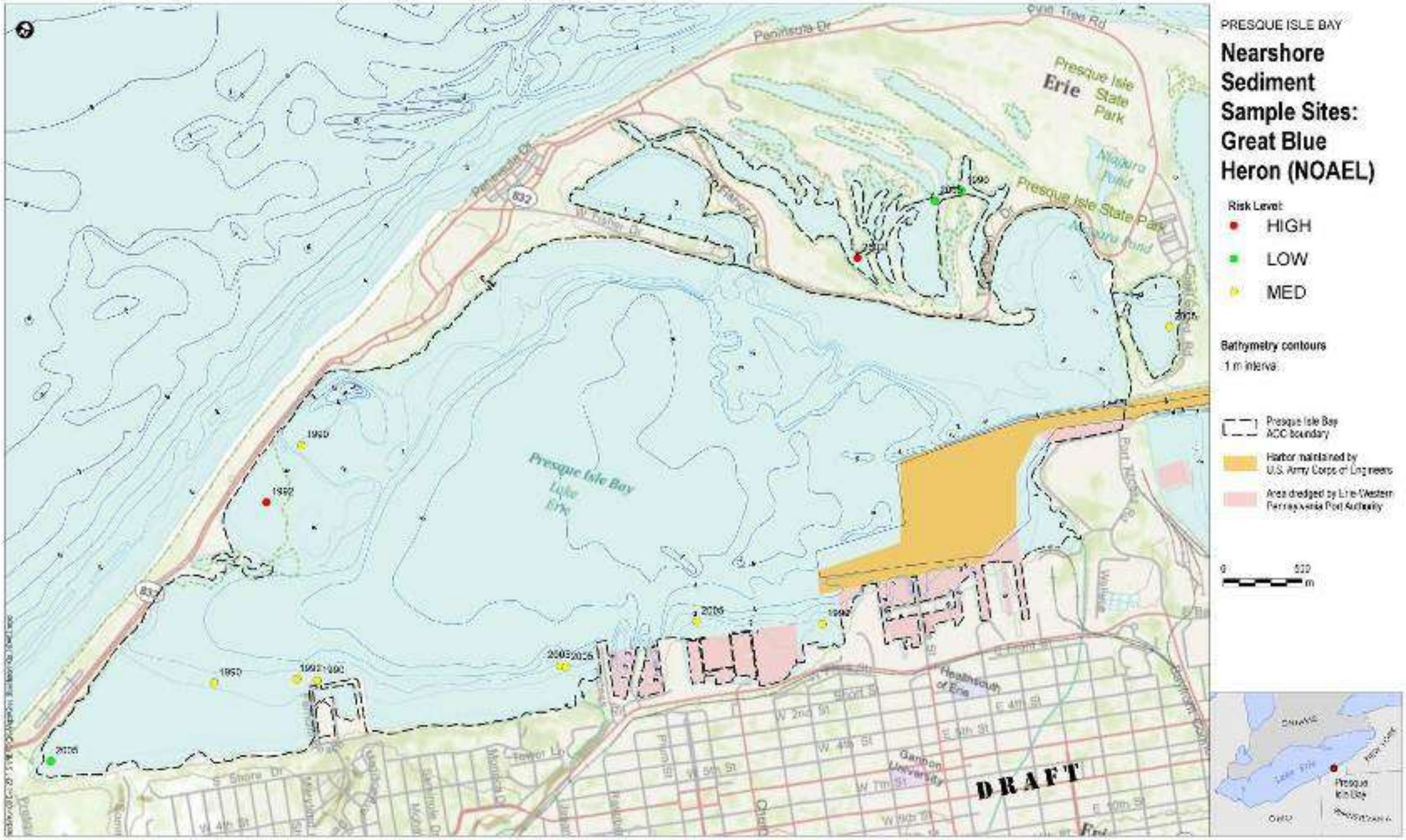

Figure 3.20. Risk levels for Great Blue Heron at NOAEL toxicity reference value.

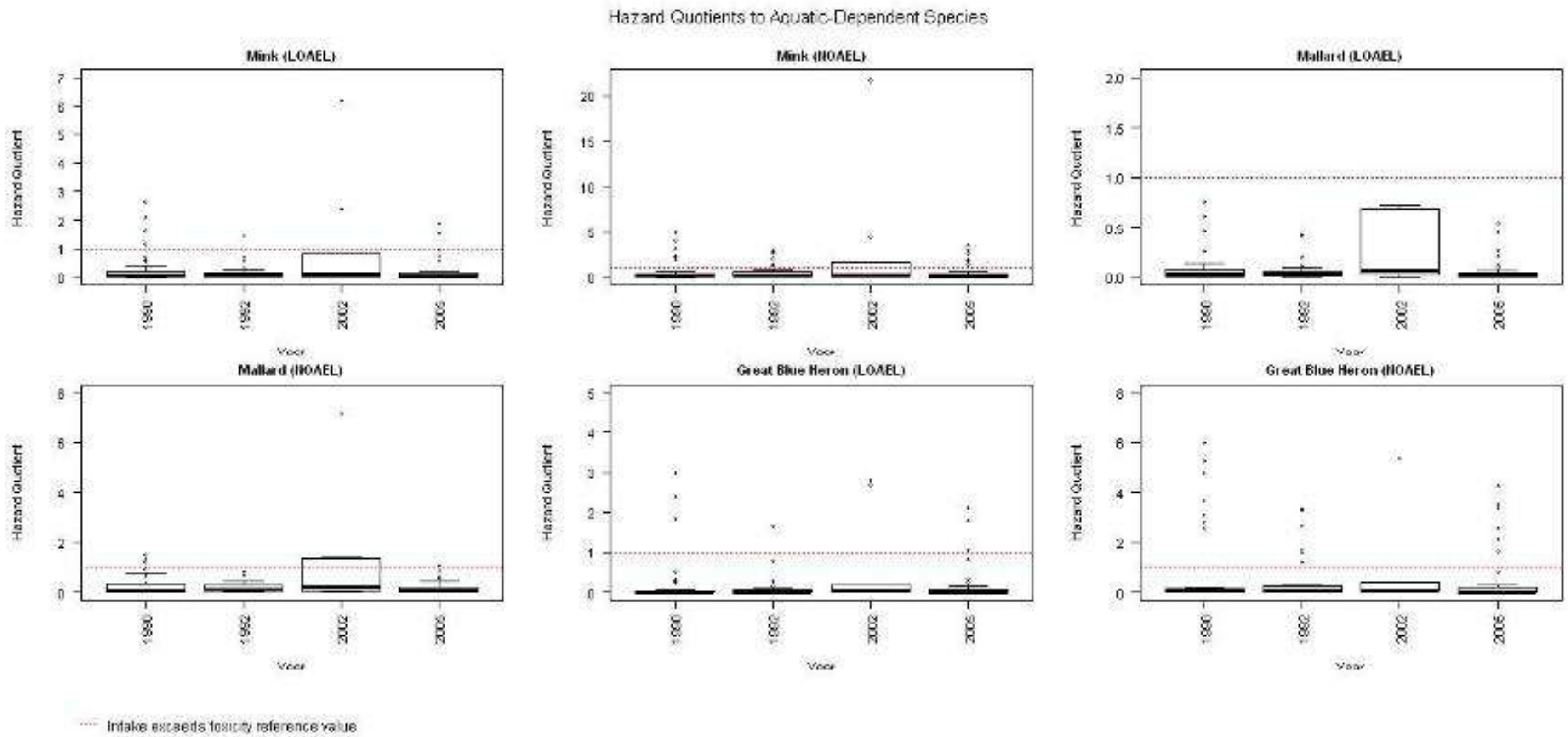

Figure 3.21. Hazard quotients and exceedances of hazard quotients for different receptors from PIB sediments

3.6 RISK SUMMARY

The risk characterization integrates the exposure and effects characterizations to assess whether chemical concentrations (COPCs) are sufficiently high to pose unacceptable risks to ecological receptors. Other authors provide varying levels of direct or indirect evaluations of risk on receptors within PIB (Attachment 2). It should be emphasized that this screening-level ecological risk assessment, where possible, incorporated conservative estimates where uncertainties were apparent, which is typical for a screening analysis (i.e., risks are likely to be overestimated rather than underestimated). The chemicals identified as chemicals of potential concern (i.e., COPCs) may be evaluated further in site-specific assessments to further characterize the risks they pose. The following sections present the risk characterizations within the PIB ecosystem from both previous investigations (Section 2) and primary evaluations of available data (below).

The evaluation of the target objectives conducted for this SLERA has been compiled to establish a weight of evidence supporting the previously posed question of:

Do legacy contaminants (COPCs) continue to pose a risk to ecosystem receptors within Presque Isle Bay?

Below (Table 3.11) is a summary of how the various findings supported the evaluation of the PIB ecosystem objectives.

Objectives	**Benthic Invertebrate Community**					**Quality Fishery**				**Near-shore Habitats (Wildlife)**		
Target	**90% of samples meeting criterion**					**Water Quality**	**Benthic Invertebrate Health (Prey Base)**	**Bioaccuumulation (Tissue Samples)**	**Tumor Lesion Effect (Brown Bullead)**	**COPC Exposure Risk (Exposure Models)**	**Bioaccumulation (Fish Tissue Samples)**	**Benthic Invertebrate Health (Prey Base)**
Metrics	**PEC-Q**	**SEM-AVS**	**SEM_AVS foc**	**ESB-TUs**	**Sed. Tox. And Surv.**							
Studies												
Diz (2002)		Y			N							
Diz (2005)		Y										
Kemble et al. (2006)					Y							
MacDonald (2008)	N	Y	Y	Y	Y		Y					Y
Pyron et al. (2001)									Y			
SLERA	N	Y	Y	U	N	Y	Y	Y	U	Y	Y	Y

Y = Supports Target Metric; N = Does Not Support Target Metric; U = Inconclusive Consistency

Table 3.11. PIB Ecosystem Objectives, targets and metrics evaluated by previous and current investigations.

(White boxes depict PIB studies using primary source data to evaluate targets and/or metrics, Gray boxes depict areas that are not applicable)

3.6.1 Weight of Evidence

1) Surface sediment COPCs appear to be the primary chemical stressor in this system, although habitat (substrate) and invasive species may be additional stressors on the ecological community that may be challenging to tease apart.

2) The potential risk of COPC exposure benthic invertebrates across PIB are generally low based on whole sediment toxicity tests. Isolated areas may pose a moderate to high risk of exposure.

3) Benthic invertebrate exposure risk has decreased through time and are generally meeting toxicity targets.

4) The probable effect concentration (PEC) targets are generally met across PIB for most COPCs. Exceedences do occur for metals like barium and cadmium and for some PAHs. Studies focused on high concentration areas tend to exceed PEC in most cases but skew the baywide results.

5) Metals bioavailability across the PIB appears to be decreasing through time, with recent samples meeting low toxicity thresholds.

6) The quality fishery objective within PIB are supported by good water quality, a low risk of prey base (benthic invertebrates) exposure to COPCs, and fish tissue concentration of monitored compounds that are similar to background levels.

7) Water quality conditions are based on qualitative evaluations and fish tissue concentrations for monitored contaminants (e.g., mercury and PCBs) and are similar to or better than other Lake Erie levels.

8) Near-shore sediment habitats suggest that ingestion exposure risks to wildlife are moderate to low, and the elevated surface sediment concentrations of PAHs and metals (dry weight) in PIB tend to be in the vicinity of the docks and shipping channel.

Overall, it appears that the targets supporting the PIB ecosystem are being met. Gaps in data to definitively describe all targets and metrics exist, but the current weight of evidence suggests that the risk to ecosystem receptors within PIB is improving through time currently rates low to moderate risk.

4. UNCERTAINTIES AND CONCLUSIONS

4.1 UNCERTAINTIES

A discussion of uncertainties is important in any risk assessment and can be critical in making risk management decisions. A consideration of uncertainties is also imperative in using the lines of evidence approach discussed above. For example, the lines of evidence need to be balanced by considering the amount of uncertainty associated with each (U.S. EPA 1998). This screening level assessment relied entirely on previously conducted investigations and data collected by other organizations and agencies, so it is assumed that standard QA/QC protocols of data design, collection, processing, and analysis were maintained.

The CSM is intended to define the linkages between stressors, potential exposure, and predicted effects on ecological receptors. Potential uncertainties arise from lack of knowledge regarding ecosystem functions, failure to adequately address spatial and temporal variability in the evaluations of sources, fate and effects, omission of stressors, and overlooking secondary effects (USEPA 1998).

Of the CSM components, the identification of exposure pathways probably represents the primary source of uncertainty in the conceptual model. In this assessment, supported by MacDonald (2008), it was assumed that exposure to whole sediments represents the most important pathway for exposing benthic invertebrate communities and macrofauna that have a benthic component in their food web to COPCs (i.e., as the benthic invertebrates associated with benthic habitats likely play key ecological functions, and contaminant concentrations are likely to be highest in this medium). However, receptor communities may also be exposed to COPCs in the water column, but this pathway was not examined and data supporting an examination of this pathway are lacking. As result, this potential pathway has not been considered in this risk analysis to the ecosystem, and may be underestimated if this represents a COPC route.

The exposure assessment is intended to describe the actual or potential co-occurrence of stressors with receptors. As such, the exposure assessment identifies the exposure pathways and the intensity and extent of contact with stressors for each receptor or group of receptors at risk. There are a number of potential sources of uncertainty in the exposure assessment, including measurement errors, extrapolation errors, and data gaps (MacDonald 2008).

The range of investigations, their scales, methods and results increase the uncertainty of results as direct comparisons among investigations. Most of the included investigations were not designed to support an ecological risk assessment, thus a screening level assessment has been applied. The range, focus and site selection strategies of investigations conducted in PIB complicate the ability to make “apples to apples” comparisons among years to quantify trend and spatial variability.

PIB is a dynamic system. The system contains areas of active erosion, deposition and resuspension influenced by relatively small contributions of watershed-level sediment inputs, dredging for navigation and recreation boating, and seiche effects that complicate any analysis of legacy inputs. The erosive and resuspension dynamics as sources of exposure remain to be understood in the system.

Wherever possible, conservative assumptions were used in estimating receptor exposures to chemicals and in identifying toxicity thresholds. The largest sources of data for the screening-level assessment were the chemistry data for sediment. These data were used to estimate whether individual chemicals, and in some cases classes of chemicals, were present at sufficiently high concentrations to pose a potential risk to ecological receptors. This approach uses site-specific chemistry data, but assumptions are required in estimating the magnitude of exposure to biota.

Limited fish tissue samples were available for this study, and for those samples available, the constituent data were for mercury, and none listed the COPCs within Presque Isle Bay. The levels of COPCs used in this screening-level assessment within fish tissue remain unknown.

Fish tumor science is still evolving, but the evidence thus far suggests that external tumors and the frequency of external tumor rates are less strongly linked to legacy contaminants than liver tumors and liver tumor frequencies. External and internal tumor frequencies do not necessarily support one another. It may be years before scientists fully understand the causes or relationships of COPCs to external tumors and frequency.

There is uncertainty associated with the calculation of a hazard quotient (HQ) and its strength of association with toxicity to an endpoint. One level of uncertainty is associated with the feeding areas associated with each endpoint. The calculations were conservative and assumed that a specific species feeds only in Presque Isle Bay throughout the whole year. This may be a reasonable assumption for the mink, but may not be for more mobile species like the mallard duck or great blue heron. The calculations also did not take into account the complexities of the diets of the endpoint species. In reality, the species evaluated most likely obtain their food from a variety of sources. Though their diets may be focused on benthic invertebrates or fish, the types of invertebrates or fish being ingested will have varying levels of contamination for each prey species.

4.2 CONCLUSION

The purpose of this study was to explore the potential to remove the fish tumor BUI for PIB on the basis of an examination of the effects of fish tumor suspected stressors (essentially surface sediment COPC concentrations) on other components of the ecosystem. To make that assessment, a screening-level ecological risk analysis based on a weight-of-evidence of existing data for PIB has been conducted as a surrogate for a formal risk assessment of the exposure pathways that are leading to the occurrence of external fish tumors. The weight-of-evidence for sediment COPC effects on receptors conducted in this study, suggest that exposure to surface sediment

COPCs are not posing a significant adverse impact on the overall PIB ecosystem. These results should be used in conjunction with the incidence of tumor rates within PIB and an overall assessment of ecosystem effects from COPCs. At present, the combination of data suggests that the incidence of internal fish tumors in PIB is not significantly different from reference sites and the combined information may provide sufficient justification for removing the fish tumor BUI from PIB. Of course, moderately elevated external skin lesions remain in PIB benthic fish, and additional research is needed to establish the stressor or stressors (e.g., sediment physical properties, exposure to viruses) that are causing this result.

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ATTACHMENTS

ATTACHMENT 1 SUMMARIES OF PIB SUPPORTING PAPERS

ATTACHMENT 2 - SPREADSHEET CALCULATIONS OF HAZARD QUOTIENTS FOR WILDLIFE RISK ASSESSMENT

An Evaluation of Human Health Risks from Contaminants in Presque Isle Bay

Erie, Pennsylvania

Prepared for:
Pennsylvania Department of Environmental Protection
Meadville, PA

Prepared by:
Michelle Homan, Ph.D.
Gannon University
July 19, 2012

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List of Acronyms

ABS	Absorption Efficiency Factor
ALM	Adult Lead Model
AOC	Area of Concern
ATSDR	Agency for Toxic Substances and Disease Registry
BCF	Bioconcentration Factor
BHC	Hexachlorocyclohexane
Cal EPA	California Environmental Protection Agency
CDC	Centers for Disease Control
COPC	Chemicals of Potential Concern
CSF	Cancer Slope Factor
CSM	Conceptual Site Model
CTE	Central Tendency Exposure
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
ELCR	Excess Lifetime Cancer Risk
EPC	Exposure Point Concentration
FDA	Food and Drug Administration
HEAST	Health Effects Assessment Summary Tables
HHRA	Human Health Risk Assessment
HI	Hazard Indices
HQ	Hazard Quotient
IEUBK	Integrated Exposure Uptake Biokinetic
IRIS	Integrated Risk Information System
NCEA	National Center for Exposure Assessment
PADEP	Pennsylvania Department of Environmental Protection
PAHs	Polycyclic Aromatic Hydrocarbons
PbB	Blood Lead Concentration
PCBs	Polychlorinated Biphenyls
PIB	Presque Isle Bay
PIBPAC	Presque Isle Bay Public Advisory Committee
PPRTV	Provisional Peer Reviewed Toxicity Values
RfD	Reference Dose
RME	Reasonable Maximum Exposure
RSL	Recommended Screening Level
95% UCL	95^{th} Percent Upper Confidence Limit
USEPA	United State Environmental Protection Agency
VOCs	Volatile Organic Compounds

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EXECUTIVE SUMMARY

ES.1 Overview

This report summarizes the results of a human health risk assessment (HHRA) for Presque Isle Bay in Erie, Pennsylvania. This is a companion document to the Ecological Risk Assessment of Presque Isle Bay prepared by Limnotech, Incorporated. (Limnotech 2011). The purpose of this HHRA was to develop estimates of current human health risks due to contact with contaminated sediments and from fish consumption utilizing existing datasets (i.e., sediment sampling database and fish species collected for fish advisory program). Both noncarcinogenic (i.e., liver, developmental and kidney toxicity) and carcinogenic (probability of developing cancer over a lifetime) risks were then compared to guidelines developed for the U.S. Environmental Protection Agency (EPA) Superfund program. These risk estimates were developed to support the policy and decision-making process.

The overall objectives of this risk assessment include to:

- compare the levels of contaminants in sediment and fish tissue to screening levels established by EPA in order to determine which contaminants should be included in the risk estimate process;
- estimate the current (or baseline) human health risks associated direct contact with Presque Isle Bay sediments and consumption of fish; and
- determine which exposure pathways and contaminants contribute most to human health risks.

The datasets utilized for this HHRA included sampling data collected between 2004 and 2010 for selected metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and pesticides in sediment and fish tissue. Fish tissue data utilized in this risk assessment included species sampled from Presque Isle Bay as well as Lake Erie. It should be noted that both the sediment and fish tissue sampling data were collected for purposes other than this human health risk assessment which may increase the uncertainty of the risk calculations.

Only a limited subset of the chemical constituents were retained and used to develop chemical-specific risk estimates. As recommended by EPA, chemical constituents were screened from inclusion in the risk assessment using established risk-based screening levels (USEPA 2011a, 2011b). For the sediment data, this resulted in a total of nine contaminants being included within the risk assessment. For fish tissue sampling results, the number of chemicals retained varied between zero for pumpkinseed and bluegill species to 16 for lake trout.

To minimize the likelihood of underestimating risks, conservative, health-protective assumptions were incorporated into the identification of exposure scenarios, the estimates of exposure, and the use of toxicity values. These are reflected in the reasonable maximum exposure (RME) scenarios presented in the report which represent high-end exposures that are likely to occur. This risk assessment also includes central tendency exposure (CTE) estimates which correspond to average exposures experienced by affected populations. Table 1 summarizes the exposure groups and pathways that were evaluated in this HHRA based on the most likely and significant exposures and data availability.

Table 1: Summary of Exposure Groups and Pathways Evaluated in this HHRA

	In-water Sediment		**Fish Tissue**
	Dermal contact	**Incidental ingestion**	**Ingestion**
Adult recreational water user	●	●	
Child recreational water user	●	●	
Adult recreational angler			●
Adult urban/subsistence angler			●
Children of recreational angler			●
Children of urban/subsistence angler			●

Potential cancer and noncancer risks were calculated for each chemical retained in the risk assessment for the above exposure scenarios. Noncancer effects were evaluated by calculating the hazard quotient (HQ) which represents the estimated exposure level divided by the reference dose (RfD). An HQ less than 1 indicates that exposures are not likely to be associated with adverse noncarcinogenic health effects while values above 1 may be of concern. HQs were summed across exposure pathways and chemicals to develop summary hazard indices (HIs). These are interpreted in a similar manner to the HQs.

For cancer risks, the endpoint is the ELCR or excess lifetime cancer risk, representing the probability of developing cancer over a lifetime due to exposure to a carcinogen. These values are calculated as the product of the lifetime exposure level to a chemical and its established cancer slope factor (CSF). Carcinogenic effects were summed across exposure pathways and across multiple chemicals. Estimated total cancer risks (summed across all chemicals) were compared to a 10^{-4} to 10^{-6} (1 in 10,000 to 1 in 1 million) risk range representing the target range required by EPA as part of the Superfund program (USEPA 1991a). Cancer risks in the 10^{-5} (1 in 100,000) range and higher are generally considered to be of concern

ES. 2 Summary of Results

1. Overall, these results show that the main exposure route for contaminants in Presque Isle Bay is through fish consumption. These risks were several orders of magnitude greater than those associated with direct contact with contaminated sediments.
2. The cancer and noncancer risk estimates generated from consumption of fish tissue were highly dependent on the fish species and location (refer to Table 11). Based on the dataset utilized in this HHRA, several species from Lake Erie contributed to higher risks compared to species from Presque Isle Bay. These findings include:
   - Lake trout and smallmouth bass represented the fish species with the highest cancer and noncancer risk estimates. These species are likely to have a higher residence time and thus represent exposures to chemical constituents that occurred mainly from open water areas of the lake.
     - The summative noncancer risk for lake trout was approximately 3 for the typical or CTE estimate and 184 for the high-end or RME estimate (target level = 1.0). This latter value indicates that the estimated exposure to this chemical from consuming fish is 184 times greater than the level recommended by the EPA.
     - The summative cancer risk for lake trout was 5×10^{-5} (5 in 100,000) and 5×10^{-3} (5 in 1,000) for the CTE and RME estimates respectively (target level < 1 in 100,000).
     - The contaminant with the highest contribution to the noncancer and cancer risk estimates for lake trout and smallmouth bass was Arochlor 1254 or 1260.

       (It should be noted that the cancer and noncancer risk estimates include the assumption of a single species diet and that all fish consumed originates from Lake Erie. These assumptions are conservative in nature and likely to overestimate the cancer and noncancer risks from consumption of fish. It should also be considered that these risk estimates are based on a limited sampling of fish tissue.)
3. Values for certain fish species from Presque Isle Bay were also greater than the applicable cancer and noncancer risk thresholds. These results include:
   - Common carp and largemouth bass were the species with the highest associated risks.
     - The summative noncancer risk for common carp was 6 for the typical or CTE estimate and 48 for the high-end or RME estimate.

  - The summative cancer risk for lake trout was 3 X 10^{-5} (3 in 100,000) and 8 X 10^{-4} (8 in 10,000) for the CTE and RME estimates respectively (target level = 1 in 100,000).
  - The contaminant with the highest contribution to the noncancer and cancer risk estimates for common carp was Arochlor 1254.
- The concentration of chemical constituents measured in panfish, including bluegill and pumpkinseed species, were all below the applicable fish tissue screening levels. Therefore, risk estimates were not calculated for these species.

(It should be noted that the cancer and noncancer risk estimates include the assumption of a single species diet and that all fish consumed originates from Lake Erie. These assumptions are conservative in nature and likely to overestimate the cancer and noncancer risks from consumption of fish. It should also be considered that these risk estimates are based on a limited sampling of fish tissue.)

4. The cancer and noncancer risk estimates for direct contact with contaminated sediments from Presque Isle Bay were generally below the target risk levels for all exposure groups evaluated in this HHRA. All chemical-specific and cumulative excess lifetime cancer risk estimates were below 1 X 10^{-5} (1 in 100,000) and all chemical-specific and cumulative hazard indices were below 1.0. The exception to this was the RME cancer risk estimate for child recreational water users which was 4 X 10^{-5} (4 in 100,000). This value is mainly driven by dermal exposure from total PCBs. It should be noted that these risk estimates are conservative in nature and likely to overestimate the risk (the uncertainties associated with these estimates are discussed in section 6 of this report).

5. The uncertainties associated with this risk assessment should be considered in utilizing the results for risk management decisions. A summary of the uncertainties inherent to this HHRA are discussed in section 6 of this report. The major uncertainties noted include the:

- small dataset from which the risk estimates were drawn (i.e., data for certain fish species included one composite sample of five individual fish);
- lack of specific data for the environmental media to which exposure groups are more likely to contact (i.e., for children beach sediment is a more likely exposure media compared to in-water sediment on which the risk estimates are based); and
- lack of site-specific information on fish consumption patterns within the study area.

1. INTRODUCTION

1.1. Overview of Risk Assessment

This HHRA has been prepared in support of the investigation to address potential human health risks associated with contaminated environmental media and fish consumption in Presque Isle Bay. This HHRA presents the potential for current cancer risks and noncancer health hazards to people who may be exposed to contaminants. The overall goals of this assessment are to:

- compare the levels of chemicals of potential concern (COPCs) in sediment and fish tissue to EPA screening levels in order to determine which constituents should be carried through the full risk assessment process;
- compare the estimated human health risks from consuming fish from Presque Isle Bay with those from Lake Erie;
- determine which exposure pathways lead to the highest human health risks; and
- quantify the current (or baseline) human health risk associated with the COPCs using existing dataset.

Potential human health risks were characterized based on COPC concentrations detected in sediment samples collected in 2005 and fish tissue samples from various species collected between 2004 and 2010. The sampling and analytical details are summarized in this report and presented in detail elsewhere (PADEP 2006). Both CTE and RME estimates were included in order to represent both typical exposures (representative of the average exposures that are likely to occur) and conservative exposures (representative of the maximal exposure that is reasonably likely to occur).

The procedures and guidelines followed in this HHRA are consistent with those outlined by EPA. This methodology includes a four-stage process: hazard identification, exposure assessment, toxicity assessment, and risk characterization (see Figure 1). This HHRA was conducted in a manner consistent with the following documents:

- *Risk Assessment Guidance for Superfund: Human Health Evaluation Manual, Part A* (USEPA 1989);
- *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors* (USEPA 1991b);
- *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children. Office of Solid Waste and Emergency* (USEPA 1994);
- *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories Volume 1 Fish Sampling and Analysis*, 3rd ed. (USEPA 2000);
- *Human Health Toxicity Values in Superfund Risk Assessments* (USEPA 2003a);
- *Recommendations of the Technical Review Workgroup for Lead for an Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil* (USEPA 2003b);
- *Risk Assessment Guidance for Superfund: Human Health Evaluation Manual, Part E, Supplemental Guidance for Dermal Risk Assessment* (USEPA 2004).

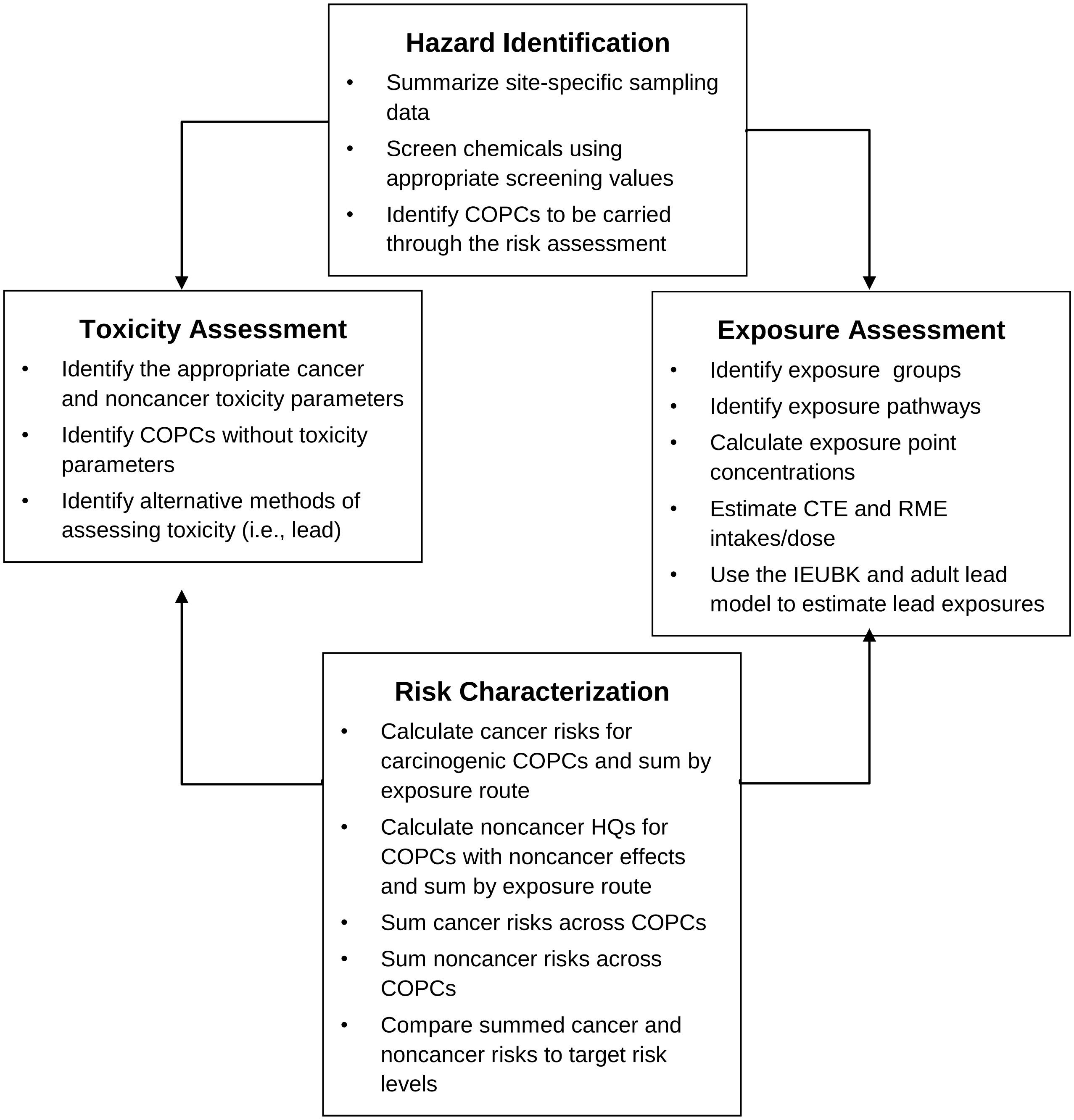

Figure 1: Overview of the Four-Stage Risk Assessment Process Followed for this HHRA

1.2. Site Location and History

Presque Isle Bay is located in northwestern Pennsylvania at the southeastern end of Lake Erie (refer to Figure 2). The bay is approximately 4.5 miles long and 1.5 miles wide across at its widest point with an average depth of 13 feet. Access to Lake Erie occurs through a narrow dredged channel at the southeastern end of the bay. The bay is bordered by the City of Erie on the southern shore, Presque Isle State Park on the northern shore and Millcreek township on the western side (PADEP 2002).

Figure 2: Aerial photo of Presque Isle Bay with Area of Concern Boundary

The drainage basin for PIB is approximately 25 square miles consisting mainly of urban and industrial land uses within the City of Erie and the townships of Millcreek, Summit, Greene and Harborcreek. Approximately two-thirds of the water flowing into the bay originates from two main tributaries: Mill Creek and Cascade Creek. Approximately 80 percent of this watershed is comprised of urban land usage (Foyle 2006).

Mud-dominated sediments comprise much of the bay and are known to be contaminated with polycyclic aromatic hydrocarbons, metals and hydrocarbons (Batelle 1994, 1997; Diz 2002; PADEP 2002, 2006) originating from historical sources along the bay and within the watershed (Foyle 2006). Prior to the City of Erie's changes in its wastewater treatment, conveyance and collection system untreated wastewater from industrial, commercial and residential sources was able to reach the bay through combined sewer overflows. Additionally, stormwater runoff from sources within the urbanized watershed has also contributed to pollutant loading of the bay. Many of these contaminants have decayed over the years through natural biodegradation processes, however, substances such as heavy metals and persistent organics still remain in the sediment.

In 1991, PIB was designated as the 43rd Area of Concern (AOC) due to two beneficial use impairments including: restrictions on dredging (due to contaminant concentrations in sediments) and fish tumors and other deformities (PADEP 2002, 2006). Sediment sampling studies have been conducted since the 1980s by the Pennsylvania Department of Environmental Protection (PADEP) in collaboration with community partners and other governmental agencies. While differences exist across these studies, similar conclusions were reached including that bay sediments were found to contain widespread but low levels of polycyclic aromatic hydrocarbons (PAHs) and several heavy metals (i.e., nickel, lead and cadmium). Sediment dredged from the navigation channel and turning basin within the bay by the U.S. Army Corp of Engineers has consistently met the requirements for open lake disposal in Lake Erie (PADEP 2002 and 2006).

The major concern with regard to fish began in the 1980s, when the United States Fish and Wildlife Service began receiving reports of "tumorous" growths on brown bullhead catfish caught within the bay. Since these fish are non-migratory and bottom-dwelling these are in direct and prolonged contact with contaminated sediments (Blazer, et al. 2009a). A number of studies have been conducted on brown bullheads in the bay to examine the rates of both internal and external tumors, their migration habits, and a potential causal relationship between these tumors and sediment contaminants. While the risk factors for the tumors in bay bullheads have yet to be elucidated, these studies taken together show a trend of decreasing tumors in brown bullheads since 1990 (PADEP 2002). It should be noted, however, that the rate of tumor incidence still appears to be higher in PIB compared to non-AOC reference locations (Blazer et al. 2009a and 2009b).

In 2002, PIB was the first AOC in the United States to be designated as in the Recovery Stage. This was based on the determination by the PA DEP in conjunction with the PIB Public Action Committee (PAC) that natural attenuation, rather than active remediation within the AOC, would provide the most practical and cost-effective method for removing the restrictions on dredging activities. This determination, along with the downward trend in fish tumors during the 1990s, contributed to the re-designation of PIB as an AOC in the Recovery Stage (PADEP, 2006).

Evaluation of Presque Isle Bay sediment and fish populations continued after the re-designation in 2002 (PADEP 2006). This HHRA along with an evaluation of the ecological health (Limnotech 2011) of the bay is an additional dataset that adds to the body of data and research that exists for Presque Isle Bay to assist in the policy and decision-making process.

1.3. Recreational Uses of the Bay

Presque Isle Bay has many recreational uses including fishing, boating, sailing, and other water-related activities. Fishing on the bay is prevalent and occurs through access from piers, docks, boats, ice (in winter) and from the shoreline at many locations, as shown in Figure 3. There are numerous public and private marinas providing boat access to both the bay and to Lake Erie. While there are no designated swimming areas or beaches along the shores of the bay, swimming access from the shoreline or boats is likely to occur. Additionally, water skiing and the use of personal watercraft (jet skis) are common within the bay.

The bay is particularly attractive to anglers throughout much of the year. Depending on the season, anglers will commonly pursue opportunities to catch panfish, perch, bass, muskellunge, walleye, northern pike, crappie, or steelhead salmon (PADCNR 2011). Popular shore fishing locations include the Waterworks and Ferry Dock ponds, East and West piers, Perry Monument, North Pier, lagoons and all boat landings. Fishing along the north shore of the bay within Presque Isle State Park is permitted throughout almost the entire length of the park. Many areas along the bay’s shore in the park are suitable for wading due to the shallow depths of near-shore areas.

On the western and southern shores of the bay, fishing occurs on-shore and at numerous public docks and piers at various access points located to both west and east of the mouth of Cascade Creek. In addition to these popular areas, anglers also attain fishing access at the public piers located at Dobbins Landing, Liberty Street Dock, Bay Harbor Marina, and the South Pier.

Ice fishing occurs on the bay when there is sufficient ice which usually occurs during the months of December or January. The most popular location for ice fishing includes the head (western end) of the bay, Misery Bay and Horseshoe Pond since these tend to be the first areas to develop a thick enough layer of ice.

Figure 3: Map of Presque Isle Bay showing boat launch and marina locations.

1.4. Fish Consumption Advisories

The 2012 fish advisory for Presque Isle Bay recommends limiting the number of meals of specific sport fish in order to reduce the exposure to mercury and polychlorinated biphenyls (PCBs) (PA Fish and Boat 2011). Table 2 shows the 2012 fish consumption advisory for areas within the Lake Erie basin. In order to limit PCB exposure it is recommended that the following fish be consumed at a rate of one meal per month: smallmouth bass, northern pike, white perch, freshwater drum, bowfin, carp, Coho salmon and steelhead (Rainbow Trout).

Pennsylvania has issued a general, statewide health advisory for recreationally caught sport fish. This advisory recommends no more than one meal (one-half pound) per week of sport fish caught in the state's waterways. This general advice was issued to protect against eating large amounts of fish that have not been tested or that may contain unidentified contaminants (PA Fish and Boat 2011).

Other aquatic species from the bay with the potential to be consumed include clams, mussels and turtles. In the state of Pennsylvania the harvesting of live mussels and clams is prohibited. Currently there are no restrictions on the consumption of turtles caught within the Lake Erie basin. However, the advisory does warn consumers that small amounts of PCBs have been found in snapping turtles and that these tend to accumulate in fat and internal organs. The advisory therefore recommends that consumers remove fat and internal organs before consuming turtle meat (PA Fish and Boat 2011).

Table 2: Fish Consumption Advisories for Areas within the Lake Erie Basin[1]

Advisory Area	Species	Meal frequency	Contaminant
Lake Erie - Open Waters	Walleye, Coho salmon[2], Steelhead[2] (Rainbow trout), Smallmouth bass, White perch, White bass, Lake whitefish, Carp under 20", Freshwater drum, Lake trout and Channel catfish	1 meal/month	PCBs
	Carp over 20 inches	Do not eat	PCBs
Lake Erie – Presque Isle Bay	Smallmouth bass, Northern pike, White perch, Freshwater drum, Bowfin, Carp, Coho salmon[2] and Steelhead[2] (rainbow trout)	1 meal/month	PCBs
Conneaut Creek (Erie County) SR 0215 bridge to PA/OH border	Smallmouth bass	2 meals/month	Mercury

Notes:

(1) Fish and Boat Commission. 2012 Fish Consumption Advisory

(2) Salmon and trout are migratory. They may be found seasonally in Presque Isle Bay or Lake Erie tributary streams. Trout, salmon and other fish, whether caught in the lake or elsewhere, should be treated as Lake Erie fish.

2. CHEMICALS OF POTENTIAL CONCERN

The purpose of this section is to identify the chemicals detected at the site that will be included in the overall HHRA. The COPCs were selected by comparing the maximum detected concentrations to the appropriate screening criteria. Chemical concentrations that exceeded the screening criteria were retained and included in the overall risk characterization while those chemicals below the criteria were excluded from further evaluation. Tables 2-1 through 2-16 summarize the results of this evaluation for sampling data for both sediment and fish tissue data.

2.1. Sampling Data

The dataset used in this HHRA included only those matrices relevant for direct human health exposure pathways: surface sediment (0 to 10 centimeter (cm) in depth) and fish tissue. The most recent set of sediment sampling data, collected in 2005, was utilized in this HHRA (PADEP 2006). It should be noted that this data was collected for purposes other than a human health risk assessment and thus the best available data was used whenever possible. For example, ideally on-shore sediment sampling data would be most appropriate to determine a young child's exposure to on-shore sediment through ingestion and direct contact. However, since this data was not available, in-water sediment sampling data was used as a surrogate. It is likely that this provides a more conservative (i.e., higher) estimate of human health exposures.

Contaminant sampling within pore water or the water column was not conducted and thus, this potential exposure pathway could not be evaluated in this HHRA. It is likely that this would be a minor or insignificant exposure pathway for contaminants of concern.

2.1.1. Sediment Sampling Summary

Table 2-1 (Appendix) summarizes the sediment sampling data that was utilized in this HHRA. The dataset used in this HHRA was collected from September 12 through September 15 in 2005 from a comprehensive sediment survey (PADEP 2006). Partners in the survey included PADEP, PIBPAC, Pennsylvania Sea Grant, Gannon University, the Regional Science Consortium and the Erie County Department of Health. Funding for the study was provided by the Great Lakes National Program Office and directed by MacDonald Environmental Services Ltd.

In this survey, a total of 32 surficial samples and four core sediment samples were collected (PADEP 2006). The surficial samples were collected from the top 10 centimeters of sediment using a Van Veen grab sampler. Twelve of the samples were collected based on historical locations while twenty samples were collected from randomly selected locations (refer to Figure 2-1 in Appendix). Two of the four cores were cut into 5 cm sections to a depth of 80 cm and subsequently analyzed. The remaining two core samples were archived.

Only data relevant to the exposure scenarios were included in the risk assessment. Since contact with sediments is only likely to occur during wading and swimming, only near-shore sampling sites were included in the analysis. For the purposes of this HHRA near-shore was considered to be those samples collected from areas with a depth of 10 feet or less. Sampling sites from the center of the bay and within the dredging zone were excluded from the analysis. Additionally, only surficial samples collected from the top 10 cm of sediment were included. The two core samples were not included in this analysis. This resulted in a total of 14 sample sites being included in the risk assessment as shown in Figure 2-1.

The sediment contaminants included in this HHRA include those summarized in Table 3. Additional compounds or parameters were quantified in sediment samples but not included in this risk assessment (refer to Table 2-2 in Appendix). These constituents or quality parameters were excluded due to their lack of correlation with human health risks or, in the case of alkyl-PAHs, due to lack of information that would allow human health risks to be quantified.

Table 3: Inorganic and Organic Analytes Measured in Sediments

Metals	PAHs	PCBs	Pesticides
Arsenic Cadmium Chromium (total) Copper Lead Mercury Nickel Zinc	Acenaphthene Acenaphthylene Anthracene Benzo(a)anthracene Dibenzo(a,h)anthracene Benzo(a)pyrene Benzo(b)fluoranthene Benzo(e)pyrene Benzo(g,h,i)perylene Benzo(k)fluoranthene Chrysene Fluoranthene Fluorene Indeno(1,2,3-c,d)pyrene Naphthalene Perylene Phenanthrene Pyrene	Total PCBS PCB008 PCB018 PCB028 PCB044 PCB052 PCB066 PCB087 PCB101 PCB105 PCB118 PCB128 PCB138 PCB153 PCB170 PCB180 PCB187 PCB195 PCB206 PCB209	Aldrin Chlordane, technical grade Dieldrin o,p'-DDD p,p'-DDD o,p'-DDE p,p'-DDE o,p'-DDT p,p'-DDT Endosulfan-alpha Endosulfan-beta Endrin Heptachlor Heptachlor epoxide Hexachlorobenzene Hexachlorobutadiene Hexachlorocyclohexane-γ Hexachlorocyclopentadiene Methoxychlor Mirex Nonachlor, trans- (chlordane)

2.1.2. Fish Tissue Sampling Summary

Fish tissue data that was included in this HHRA was originally collected expressly for the purpose of and in accordance with DEP's Fish Consumption Advisory Program (PADEP 2010). Tables 2-3 through 2-16 summarize the results of these sampling surveys. Fish species were collected during various time periods from February 8, 2004 through

November 10, 2010 from three approximate sampling locations in Presque Isle Bay and Lake Erie (refer to Figure 2-2). Samples collected in Presque Isle Bay occurred primarily via electrofishing while sampling in Lake Erie occurred via gillnets, trot lines and/or angling until the required number of fish of the target species were caught. Table 4 summarizes the fish species (common name), date of sampling and area of sampling. The last two columns of this table indicate the assumption of where the fish was likely to reside for the majority of its life. Table 4 summarizes the fish tissue species included in this risk assessment, the location of sampling and the assumption of where each species is likely to spend most of its life (Presque Isle Bay or Lake Erie).

One fish tissue sample represents ten scaled, skin-on fillets from a composite of five individuals of the fish species being targeted. Channel catfish and burbot samples consisted of ten skinless fillets. As per PA DEP guidelines, all fish in the composite were of the same species and approximately the same size, (i.e., lengths of all fish in the composite were within 75 percent of the length of the largest fish) (PADEP 2010).

Table 4: Summary of Fish Species, Sampling Information and Residence Time

Category	Common Name	Year	Area caught	Assumption of "residence time" of fish	
				PIB	LAKE
Predator/Game/ Other Species	Bluegill	2004	PIB	X	
	Lake trout	2004 2006 2007 2008 2010	LEW LEW LEW LEW LEW		X
	Largemouth bass	2005 2006	PIB PIB	X	
	Northern Pike	2010	PIB	X	
	Pumpkinseed sunfish	2004	PIB	X	
	Smallmouth bass	2004 2005 2006 2007 2008 2010	LEE LEE LEE LEE LEW LEW		X

Table 4: Summary of Fish Species, Sampling Information and Residence Time (cont.)

Category	Common Name	Year	Area caught	Assumption of "residence time" of fish	
				PIB	LAKE
	Walleye	2007 2008 2010	LEW LEW LEE		X
	White bass	2004	LEW		X
	Yellow Perch	2004 2005 2006 2007 2008 2010	LEW LEW LEW/PIB LEW LEW LEW		X
Bottom dwelling species	Brown bullhead	2005	PIB	X	
	Burbot	2007 2008	LEE LEE		X
	Channel Catfish	2004 2005 2010	LEE LEE LEW		X
	Common carp	2010	PIB	X	
	White sucker	2007	LEE		X

The fish tissue samples were analyzed by validated methods and included the chemical constituents summarized in Table 6. The chemical constituent concentration was determined as the mass of chemical per wet weight of fish tissue except for those chemical constituents as noted in the table. In addition to these constituents, channel catfish were analyzed for a total of 22 radioactive isotopes in 2010 (refer to Table 2-17 within the Appendix). The analytical results showed no levels of these radioactive isotopes within any of the fish tissue samples.

Table 5: Inorganic and Organic Analytes Measured in Fish Tissue[1]

Metals	Arochlors	Pesticides
Barium[2] Cadmium Chromium (total) Copper Lead Mercury Selenium Strontium[2]	Arochlor 1221 Arochlor 1232 Arochlor 1242 Arochlor 1248 Arochlor 1254 Arochlor 1260	Aldrin alpha-BHC alpha-Chlordane gamma-Chlordane Chlordene 4,4'-DDD 4,4'-DDE 4,4'-DDT O,P-DDD O,P-DDE O,P-DDT Methoxychlor Mirex[3] cis-Nonachlor trans-Nonachlor Oxychlordane Endrin Heptachlor Heptachlor epoxide gamma-GHC (Lindane) Dieldrin

Notes:

(1) Chemical concentrations measured as mass of chemical per wet weight of fish tissue.
(2) Barium and strontium analyzed only in tissue of Channel Catfish.
(3) Mirex was measured in all species except bluegill and pumpkinseed.

2.2. Selection of COPCs

Inclusion or exclusion of chemical constituents in the subsequent risk assessment was based on the guidelines established by EPA (USEPA 1989). This guidance recommends utilizing screening criteria to limit the number of chemicals that are carried through the quantitative risk assessment while ensuring that all chemicals that may contribute to the overall risk are still included (USEPA 1989).

In order to achieve this objective the results of the sediment analyses were screened against the EPA Region 3 Risk-based Screening Levels (RSLs) to determine whether the constituents should be included in the next stage of the risk assessment (USEPA 2011a, 2011b). These screening values are likely to be conservative and protective of human health since these are based on residential exposures and assume that the exposure frequency is 365 days per year and the exposure duration is 30 years.

2.2.1. COPCs in Sediment

The residential soil RSLs were chosen utilizing the following selection criteria:

1. If available, 1/10 of the value of the non-carcinogenic RSL was obtained from the residential table for soil (HQ = 0.1);
2. If available, the carcinogenic RSL was obtained from the non-residential table for soil (target risk = 1 x 10^{-6});
3. the screening level was selected by choosing the lower (more stringent) value of the two values identified in steps 1 and 2.

Table 2-1 (Appendix) shows the results of comparing the maximum measured value of each constituent in sediment to the appropriate RSL. As a result of this evaluation, a total of nine constituents (two metals, six PAHs and total PCB congeners) were found to have a maximum concentration greater than the applicable RSL and were subsequently carried forward in the risk assessment for the direct contact with sediment exposure pathway. Table 6 summarizes the screened COPCs, the maximum detected value, the location of the maximum value and the number of values detected above the RSL.

The chromium concentration in sediment was measured and reported as total chromium. There are no RSLs or toxicity values available for total chromium. Instead the RSL for trivalent chromium was used for screening purposes. Studies have demonstrated that hexavalent chromium tends to reduce to trivalent chromium in anaerobic conditions and in the presence of reducing agents such as S^{-2} and Fe.$^{+2}$ A study by Graham, et al. found that the Cr(VI)-reducing capacity of sediments was strongly correlated to the acid volatiles content of the sediments (Graham 2009) and thus trivalent chromium is more prevalent in the environment (ATSDR 2008). In risk assessments, it is often assumed that the ratio of Cr VI to Cr III is 1:6 (reference). The RSLs for hexavalent and trivalent chromium in residential soil are 0.29 mg/kg (cancer effects) and 12,000 mg/kg (for noncancer effects) respectively. The uncertainty associated with using the toxicity parameters for trivalent chromium is further discussed in the Uncertainty Section 7.3.1 (“Use of Trivalent Chromium Toxicity Parameters for Total Chromium.”).

Only two (PCB 105 and PCB 118) of the 19 polychlorinated biphenyls (PCBs) that were evaluated had applicable RSLs. Therefore, the RSL for total high risk PCBs was used to evaluate this class of compounds. This value represents the sum of the concentration of the 19 PCB congeners at each location (refer to Table 2-18 in Appendix). Two of the PCBs measured are considered to be dioxin-like PCBs and were included in the total PCB concentration. These two congeners were measured in concentrations well below the applicable RSL.

Total chlordanes and total DDT and its derivatives were summed and compared to the screening levels for chlordane and DDT respectively. Total chlordanes included the sum of the concentrations of chlordane, heptachlor, heptachlor epoxide, and nonachlor at each sampling site. The summed concentration did not exceed the RSL for chlordane. Similarly, the sum of DDT and its derivatives included the summed concentrations of six derivatives as shown in Table 2-18. Likewise, the total concentration of all derivatives did not exceed the RSL for DDT.

For other chemical constituents without RSLs, structural analogy was utilized in that the RSL for a chemical with a similar structure was substituted. These were based on the surrogates for toxicity values available from the PADEP toxicity database (PADEP 2011). These substitutions included: acenaphthene for acenaphthalene and benzo(g,h,i)perylene; pyrene for benzo(e)pyrene and perylene; and anthracene for phenanthrene.

Table 6: COPCs with Maximum Values Exceeding the Residential Soil RSLs

Chemical	EPA Region 3 RSL[(1)] (mg/kg)	Maximum sediment concentration (mg/kg)	Location of sample with maximum value	Number of samples above the RSL
Arsenic	0.39	30.1	47-PIP	14/14
Benzo(a)anthracene	0.15	2.2	15-PIB/27-MC	13/14
Benzo(a)pyrene	0.015	2.7	15-PIB	14/14
Benzo(b)fluoranthene	0.15	2.7	15-PIB	14/14
Benzo(k)fluoranthene	1.5	2.9	15-PIB	3/14
Dibenz(a,h)anthracene	0.015	0.44	39-PIB	14/14
Indeno(1,2,3-c,d)pyrene	0.15	3.1	15-PIB	14/14
Lead	40	127	18-PIB	11/14
Total PCBs	0.22	0.37	35-PIB	1/14

Notes:
(1) USEPA Region 3 Risk-Based Screening Levels for Residential Soil (USEPA, 2011a)

2.2.2. COPCs in Fish Tissue

Chemical concentrations in fish tissue were screened against either the EPA Region 3 RSLs or other fish consumption advisory levels that are utilized by the PADEP to develop advisories for Pennsylvania lakes and tributaries (Anderson 1993, PADEP 2010, FDA 2011). If a COPC had multiple screening levels, the lowest value of the RSL or fish consumption advisory level was utilized. The fish tissue screening RSLs (USEPA 2011b) were chosen utilizing the following selection criteria:

1. If available, 1/10 of the value of the non-carcinogenic RSL was obtained from the table for fish tissue (HQ = 0.1) (USEPA 2011b);
2. If available, the carcinogenic RSL was obtained from the table for fish tissue (target risk = 1×10^{-6}) (USEPA 2011b);
3. If available, the fish consumption advisory level was selected (Anderson 1993, USEPA 1997, FDA 2011);
4. the screening level was selected by choosing the lower (more stringent) of the values identified in steps 1 through 3.

Tables 2-3 through 2-16 (Appendix) show the results of comparing the maximum measured value of each constituent in fish tissue to the Region 3 RSL values or fish consumption advisory levels as detailed above (USEPA, 2011b). Based on this review, a total of 20 constituents were found to have a maximum concentration greater than the applicable RSL or fish consumption advisory level in at least one fish species and were subsequently carried forward in the risk assessment for the fish consumption exposure pathway. Table 7 summarizes the contaminants that were identified as COPCs by fish species. Lake trout and smallmouth bass were the species with the highest number of maximum values greater than the screening levels at 16 and 14 respectively. Bluegill and pumpkinseed (panfish) did not have chemical concentrations that exceeded the screening levels. Those chemicals identified as COPCs were included in the risk estimates.

As with the sediment samples, the chromium concentrations in fish tissue were reported as total chromium. Since there are no screening levels or toxicity values for total chromium, the screening level for trivalent chromium was used as a surrogate since the majority of chromium in the environment is likely to be in the trivalent form as previously discussed (ATSDR 2008, Graham 2009). This approach is further supported by a study which found the bioconcentration factor (BCF) for Cr(VI) in fish muscle to be less than l.0 which suggests hexavalent chromium is not likely to bioaccumulate in fish tissue (USEPA 1998).

It should be noted that there were numerous chemicals for which the method detection limit was greater than that of the screening value. These chemical constituents were not included in the risk assessment since the concentration in fish tissue could not be ascertained. This is further discussed in the Uncertainty Analysis section 6.1.3 entitled "Detection Limits Greater Than the RSLs."

Table 7: COPC Summary of Contaminants with Maximum Values Greater than the Applicable Screening Level.

	Bluegill	Brown Bullhead	Burbot	Channel catfish	Common carp	Lake Trout	LM bass	North Pike	Pumpki nseed	SM Bass	Walleye	White Bass	White sucker	Yellow Perch
Aldrin										●				
Arochlor 1254			●							●	●			
Arochlor 1260				●	●	●	●			●	●	●	●	
α-BHC						●				●	●			
α -Chlordane				●		●								
γ-chlordane						●								
4,4’-DDD				●		●				●		●		
4,4’-DDE				●		●	●	●		●	●		●	
4,4’-DDT				●		●				●		●	●	
O,P-DDT						●				●				
Dieldrin				●		●		●		●	●	●		
Heptachlor				●		●				●				
Heptachlor Epoxide						●				●				
Mercury			●	●		●	●			●	●		●	●
Mirex											●			
Cis-Nonachlor						●								
Trans-Nonachlor						●				●				
Oxychlordane						●								
Selenium		●	●	●	●	●	●	●		●	●		●	●
Strontium				●				●						

3. EXPOSURE ASSESSMENT

The exposure assessment stage involves the estimation of the magnitude, frequency and duration of current and future human exposures for each complete exposure pathway.

3.1. Conceptual Site Model

The conceptual site model (CSM) for Presque Isle Bay is shown in Figure 3-1. The purpose of the conceptual site model is to identify complete and incomplete exposure pathways. A detailed account of the rationale for including or excluding exposure pathways and receptors is provided in the next two sections (3.2 and 3.3) and summarized in Table 8 below.

3.2. Exposure Pathways

Exposure pathways are defined as the means by which a person comes into contact with a chemical within environmental media. In order for an exposure pathway to be complete the following four elements must be present (USEPA 1989):

- a source of contamination;
- a mechanism for transport of a substance from the source to the air, surface water, groundwater and/or soil;
- a point where people come in contact with contaminated air, surface water, groundwater or soil; and
- a route of entry into the body.

If all four of these elements are met, the pathway is considered complete and potentially included in the next stages (toxicity assessment, exposure assessment, risk characterization) of the risk assessment. If any elements are missing, the pathway is considered incomplete and would not be included in the next stages of the risk assessment. The rationale for including and excluding pathways is provided below. Table 8 summarizes the exposure pathways and the rationale for including or excluding each within this risk assessment.

Table 8: Potential Exposure Pathways for Presque Isle Bay

Media	Exposure Pathway	Pathway	Pathway Retained	Rationale
Sediment	Dermal contact	Dermal contact with contaminated sediments during water-related activities	Yes	Dermal contact with sediment is a potentially complete exposure pathway.
	Ingestion	Incidental ingestion of contaminated sediments during water-related activities	Yes	Incidental ingestion of sediment is a potentially complete exposure pathway.
	Inhalation	Inhalation of re-entrained sediment into air	No	This is likely to be a minor or insignificant exposure pathway for all exposure groups
Fish	Ingestion	Ingestion of contaminated fish tissue by anglers, their families and other fish consumers	Yes	Consumption of fish is likely to be a significant exposure pathway.
Waterfowl	Ingestion	Ingestion of contaminated tissue from waterfowl and other aquatic organisms	No	No comprehensive data available. Many duck/goose species in PA are migratory making it difficult to isolate PIB as a contaminant source.
Clams and Mussels	Ingestion	Ingestion of clams and mussels.	No	No comprehensive data available to evaluate Exposure pathway is likely to be incomplete (see advisory notice in section 1.4)
Turtles	Ingestion	Ingestion of contaminated turtle meat.	No	No comprehensive data available to evaluate
Surface Water	Dermal contact	Dermal contact with chemicals in water while swimming, wading, etc.	No	These are likely to be minor or insignificant exposure pathways. Most organic chemicals have minor to negligible solubility in water. No comprehensive data on chemical concentrations in surface water were available.
	Ingestion	Incidental ingestion of surface water while swimming/wading		
	Inhalation	Inhalation of vapors of VOCs/semi-VOCs from surface water		

3.2.1. Potentially Complete and Significant Pathways

Consumption of contaminated fish was considered to be a complete and potentially significant pathway. Presque Isle Bay and Lake Erie anglers were considered to be a group that is likely to have exposure to chemical contaminants. This is likely to include their families as well.

Anglers may also come into contact with chemical constituents through direct contact with sediments, direct contact with water and inhalation of vapors from surface water. These, however, are likely to be relatively minor sources of exposure. The most significant exposure in this group includes the consumption of contaminated fish. This may also hold true for the family members of anglers who also consume fish from the bay. Children of adult anglers were considered a separate exposure group in this HHRA.

3.2.2. Potentially Complete and Negligible Pathways

Inhalation of vapors or dust from contaminated sediment by exposure groups is considered to be a negligible pathway. Particulate and vapor concentration in the ambient air is likely to be low due to dilution and mixing within the area. There is no comprehensive data for the bay which specifically looks at the flux of various semi-volatile organic compounds (semi-VOCs) from water to air. The PA DEP conducts regular air monitoring for hazardous air pollutants at a site located in Presque Isle State Park. These concentrations, however, represent air concentrations from all sources and are not exclusively representative of volatilization from surface water.

Indirect exposure due to vapor intrusion (movement of vapors from soil/sediment to indoor structures) was not considered a complete exposure pathway for residential receptors since residential receptors are located greater than 100 feet horizontally from the source of soil/sediment contamination (PADEP 2002). While there are some individuals that reside in houseboat structures at various marinas within the bay, it is unlikely that significant amounts of vapors would accumulate in these structures from movement of chemicals from surface water to inside the houseboat structure.

3.2.3. Incomplete Pathways

Water from Presque Isle Bay is not used as a source of drinking water and, therefore, ingestion of contaminated drinking water was considered to be an incomplete pathway. The City of Erie Water Authority supplies potable water to properties located within the City limits. Additionally, the City of Erie Codified Ordinances, Part Nine - Streets, Utilities and Public Services Code, Title Five - Sewers and Water, Article 947 Non-Used Aquifers regulations indicate that “no well or spring located on a property shall be used as a source for drinking water or agricultural purposes.” The regulations of the ordinance state that no owner, lessee or other person shall use any groundwater source for drinking water or agricultural purposes.

3.2.4. Exposure Pathways Not Evaluated

Comprehensive data on chemical concentrations within the water column were not available to evaluate exposures in this risk assessment. Therefore, the exposure pathway of incidental ingestion of water could not be assessed. It is likely, however, that this would be a negligible pathway of exposure due to the hydrophobicity of many of the COPCs; the small amounts of chemicals that would be ingested from the water column; and the potential low probability of incidental ingestion of bay water.

The exposure pathway of consuming contaminated waterfowl was not evaluated. Duck and goose hunting is allowed at limited times and locations in-season within areas of Presque Isle Bay. However, due to the migratory nature of waterfowl and lack of adequate contaminant concentrations in waterfowl tissue, this potential pathway could not be assessed within this risk assessment.

The exposure pathway of consuming contaminated clams or mussels was not evaluated. It was assumed that this pathway would be a nonexistent or rare exposure since the state of Pennsylvania prohibits the harvesting of live mussels and clams.

3.3. Potential Exposure Groups

The goal of this risk assessment is to identify and characterize the predominant and most significant receptor groups rather than identifying every possible group that may exposed no matter how insignificant. Based on the current and most common usages of Presque Isle Bay the primary receptor groups include recreational water users and Presque Isle Bay anglers. While additional receptor groups could have been developed, it is likely that the receptor groups focused on in this HHRA include the dominant and most likely exposure pathways (i.e., groups with the highest potential exposures).

3.3.1. Adult Recreational Water Users

Recreational water users may be exposed to contaminanted sediments while swimming, wading, boating, fishing, and other activities. While there are no public beaches on the bay, it is likely that swimming does occur at various locations. Potential exposures associated with recreational water use include: dermal contact with contaminated sediments, incidental ingestion of contaminated sediments, dermal contact with water, incidental ingestion of water and inhalation of chemicals from surface water.

The exposure assumptions included in the dose and intake calculations were based either on default values (USEPA 1991b, 2004) or best professional judgement using site-specific assumptions. The assumption parameters used to calculate the intakes for adult recreational water users are summarized in Tables 3-1 and 3-2 of the Appendix.

In order to calculate an exposure estimate for sediment ingestion, default soil ingestion values from the EPA Exposure Factor Handbook were included (USEPA 2011c). These default values include 100 mg/day (95th percentile value for RME calculation) and 50 mg/day (mean value for CTE calculation) for adults. The fraction of contaminated soil or sediment ingested was conservatively assumed to be 0.5 for the RME estimate and 0.3 for the CTE estimate. This is based on the assumption that recreational water users would likely have exposures from other areas such as work or home due to dividing their time between various locations.

The average adult recreational water user was assumed to typically wear a short-sleeved shirt, shorts and no shoes. Thus, the exposed skin surface area (5,700 cm^2) was the sum of the average of the 50th percentile surface area for adult males and females for the hands, forearms, calves and feet. This value is the recommended exposed surface area for both CTE and RME estimates (USEPA 2004).

The soil-to-skin adherence factor was assumed to be 0.07 milligrams per square meter (mg/cm^2) and 0.3 mg/cm^2 for the CTE and RME estimates respectively (USEPA 2004, 2011c). These values correspond to the recommended soil adherence factors, or mass of soil that adheres per surface area of skin, for an adult resident. EPA recommends a similar approach for sediments as for soils due to a lack of detailed studies concerning dermal exposures to sediments (USEPA 2004).

The exposure duration assumptions for the CTE and RME estimates included EPA default values (USEPA 1991b). A value of 9 years was used for the CTE calculation which represents the median length of time an individual stays at one residence in the U.S. (USEPA 1991b). For RME estimates, a value of 30 years was included representing the 90th percentile value for the length of time an adult lives at one residence in the U.S.

The values included for exposure frequency for adults were based on best professional judgment. The exposure frequency assumed for the CTE estimate was based on adults that would come into contact with bay sediments an average of 38 days across a year. This value is derived from an individual conducting water-related activities 2 times per week for 13 weeks during the summer and 12 times during the spring and fall months. For the higher-end or RME estimate it was concluded that individuals would come into contact with bay sediments an average of 81 days per year. This value is based on a frequency rate of 5 days per week across 13 weeks (65 days) during the summer and 1 day per week for 16 weeks (16 days) for the spring and fall.

3.3.2. Child Recreational Water Users

Tables 3-3 and 3-4 summarize the assumptions included in the exposure calculations for exposure to sediment for child recreational water users (6 months to 6 years old).

In order to calculate an exposure estimate for sediment ingestion, default soil ingestion values from the EPA Exposure Factor Handbook were utilized (USEPA 2011c). These values include 200 mg/day (RME value) and 100 mg/day (CTE value). The fraction of contaminated soil or sediment ingested from Presque Isle Bay was conservatively assumed to be 0.5 for the RME estimate and 0.3 for the CTE estimate. This is based on the assumption that recreational water users would likely have exposures from other areas such as work or home due to dividing their time between various locations.

In order to calculate an exposure estimate for sediment ingestion, default soil ingestion values from the EPA Exposure Factor Handbook were included (USEPA 2011c). These default values include 200 mg/day (RME value) and 100 mg/day (CTE value) which represent the mean and 95th percentile value for soil ingestion for this age group. The fraction of contaminated soil or sediment ingested was conservatively assumed to be 0.5 for the RME estimate and 0.3 for the CTE estimate. This is based on the assumption that recreational water users would likely have exposures from other areas such as work or home due to dividing their time between various locations.

The child water user was assumed to typically wear a short-sleeved shirt, shorts, and no shoes. Thus, the exposed skin surface area (2,800 cm^2) was the average of the 50th percentile surface area for the forearms, hands, legs, and feet for males and females for children aged 6 months to 6 years (EPA 2004, 2011c).

The soil-to-skin adherence factor was assumed to be 0.2 milligrams per square meter (mg/cm^2)-event and 3.3 mg/cm^2-event for the CTE and RME estimates, respectively (USEPA 2004, 2010). These values correspond to the mean and 95th percentile recommended soil adherence factors for a child resident. EPA currently recommends the identical approach for sediments as for soils since there is a lack of data concerning dermal exposures to sediments (USEPA 2004).

The values for exposure duration, body weight and averaging time included EPA default values (USEPA 1991b). The exposure duration was 6 years which is the default value recommended by EPA for children aged 0 through 7 years (USEPA 1989, 1991b). The average body weight included for both CTE and RME estimates was 15 kg, the average body weight of children under 7 in the United States (USEPA 1991b). For cancer estimates, 70 years, the default average lifetime value for the U.S. population was used (USEPA 1991b).

The same assumptions used to estimate the values for exposure frequencies in adults were also used for children. These assumptions are discussed in the previous section.

3.3.3. Presque Isle Bay Anglers and Their Families

Presque Isle Bay anglers were considered to be a group that is likely to have exposure to chemical constituents within the bay. This may also hold true for their family members. Anglers may come into contact with chemical constituents while fishing through direct contact with sediments, direct contact with water and inhalation of vapors from surface water. The assumptions included in these exposure calculations are identical to those outlined in the previous section entitled “recreational water users.” These, however, are likely to be relatively minor sources of exposure.

The most significant exposure in this group includes the consumption of contaminated fish. This also holds true for family members of anglers who consume fish. The exposure group that may receive the highest exposure includes children which was an exposure group considered in this risk assessment. Based on the location of sampling and habits of each species, the fish species that were considered to reside primarily in Presque Isle Bay included:

- Blue gill
- Largemouth bass
- Northern pike
- Pumpkinseed sunfish
- Brown bullhead
- Common carp

For the purposes of this risk assessment bay anglers were considered to be recreational anglers and urban/subsistence anglers. The CTE calculation was considered to represent the recreational angler while the RME estimate would include a higher end urban/subsistence angler. The exposure assumptions used to calculate the RME and CTE estimates are consistent with the EPA document entitled *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories* (USEPA 2000) and the *Estimated Per Capita Fish Consumption in the United States* report (USEPA 2002). These rates were estimated from a national dietary study and may not be representative site-specific consumption patterns. Additional uncertainties associated with these ingestion rates are discussed in Section 7.2.2.1. “Fish Consumption Rates.”

3.3.3.1. CTE Calculation – Adult Recreational Anglers

Recreational anglers include those who fish in Presque Isle Bay for both sport and non-sport fish. While anglers may not exclusively limit their fishing to the bay to include Lake Erie and its tributaries, individuals were assumed to fish primarily from the bay for the purposes of this risk assessment. Anglers are likely to fish through a variety of means including from boats, the shoreline and various public piers and docks located along the bay.

The ingestion rate for fish used in this calculation was a value of 17.5 grams/day. This value corresponds to the average ingestion rate for uncooked freshwater and estuarine finfish for adults (age 18 and older) within the United States (USEPA 2000 and 2002). This assumption represents an average of 2.3 fish meals per month (28 meals per year) and includes a serving size of 227 grams (8 ounces) per meal for an average 70 kg. adult for every month of the year. A single species diet was assumed in this calculation and all consumed fish originated from the study area. No reduction in chemical concentration was considered for the cooking and cleaning of fish.

The assumptions used for both dermal contact and incidental ingestion of sediment for this group are the same assumptions used for the CTE calculation for adult recreational water users as outlined in Section 3.3.1 and Tables 3-1 and 3-2 within the Appendix. The exposure duration used in this calculation was 9 years which represents the default value used by EPA to represent the average time a U.S. resident resides at their current residence (USEPA 1991b). An average body weight of 70 kg was used in the exposure estimates which correspond to the value for an average adult residing in the U.S.

3.3.3.2. CTE Calculation – Children of Adult Recreational Anglers

This exposure group represents the children of adult recreational anglers who consume fish caught from the study area. The fish consumption rate used for children was assumed to be proportional by body weight to that of the adult angler resulting in an intake rate of 3.75 grams per day (15 kg/70 kg X 17.5 grams/day = 3.75 grams/day). The exposure frequency included is 365 days per year to correspond to the use of an annual average consumption rate. The additional parameters used in these calculations correspond to the default values for body weight, exposure duration and averaging time as discussed in the section describing the assumptions for child recreational water users.

3.3.3.3. RME Calculation – Adult Urban/Subsistence Anglers

The RME calculation for fish consumption is for the high end fish consumer. There is no comprehensive survey data regarding fish consumption of anglers within Presque Isle Bay. It is possible that there is a population that consumes fish close to that of a subsistence angler. A recent focus group survey of anglers within the Great Lakes by Lauber, et al., suggested that "urban sites have significant subpopulations of anglers from different ethnic and cultural backgrounds with different fish consumption norms (Lauber, et al. 2011)." In the same survey urban anglers were more likely to consume large amounts of fish if they could not easily obtain food through other means. The exposure pathways for the urban angler are identical to that of the recreational angler but include higher end values for the exposure parameters.

To account for this possibility the RME calculation uses the default EPA fish consumption rate of a subsistence angler (USEPA 2002) of 142.4 g/day. This value corresponds to the 99th percentile ingestion rate for uncooked freshwater and estuarine finfish for adults (age 18 and older) within the United States (USEPA 2000, 2002). This assumption represents an average of 19 fish meals per month (228 meals per year) and includes a serving size of 227 grams (8 ounces) per meal for an average 70 kg adult.

In calculating the estimated intakes of chemical constituents from ingestion of fish two additional assumptions included a single species diet and that all fish consumed were caught from the Presque Isle Bay area. No reduction in chemical concentration was considered for the cooking and cleaning of fish. The exposure duration used in this calculation was 30 years which represents the default values used by EPA to represent the 90th percentile (high-end) estimate of time a U.S. resident lives at their current residence (USEPA 1991b).

3.3.3.4. RME Calculation – Children of Adult Urban/Subsistence Anglers

This exposure group represents the children of adult urban/subsistence anglers who consume fish caught from the study area. Limited information is available about fish consumption for children 6 months to 7 years of age. The national dietary study, on which the adult fish consumption rates are based, does not include consumption information for young children. Therefore, the fish consumption rate used for children was assumed to be proportional by body weight to that of the adult angler resulting in an intake rate of 30.5 grams per day (15 kg/70 kg × 142.4 grams/day = 30.5 grams/day). The exposure frequency included is 365 days per year to correspond to the use of an annual average consumption rate. The additional parameters used in these calculations correspond to the default values for body weight, exposure duration and averaging time as discussed in the section describing the assumptions for child recreational water users.

3.3.4. Lake Erie Anglers and Their Families

The exposure assumptions used to calculate cancer and non-cancer risks for Lake Erie Anglers and their families (adult and children) are identical to those used for Presque Isle Bay anglers. There is no specific data available to include site-specific parameters about percentage of fish caught in the lake versus the bay. Therefore, for the purposes of this HHRA, risk estimates are based on the assumption that the person consumes one species which are all taken from Lake Erie.

The fish species included in these estimates included those that are assumed to spend a majority of their lives within Lake Erie. For the purposes of this risk assessment, these species included:

- Burbot
- Channel catfish
- Lake trout
- Smallmouth bass
- Walleye
- White bass
- Yellow perch
- White sucker

3.3.5. Exposure Groups not Included in this Analyses

Other groups may be exposed to bay contaminants through work-related activities such as individuals performing dredging activities in the bay and researchers collecting sediment for research purposes. These groups were not considered in this particular analysis. Contact with sediment contaminants among these groups is likely to be of a limited and short-term nature (i.e., less than one year) and lower than other exposure groups included in this analysis. This assumption is based not only on the limited nature of the work but also includes the assumption that workers would limit their exposure to contaminated sediments through various protective measures.

Residents living near or on Presque Isle Bay were not considered as a separate exposure group since their exposure is likely to be low unless they participate in activities that put them in contact with contaminated media. Such individuals would be included in the risk assessment due to their specific activities such as participating in water-related activities and consuming fish.

3.4. Calculation of Exposure Point Concentrations

3.4.1. Exposure Point Concentrations for Sediment

Table 9 summarizes the EPCs used in estimating risk for direct contact exposure to sediment for the designated exposure groups according to EPA guidelines (EPA 1992). In calculating EPCs, the exposure area concept was utilized which includes the assumption that over a long period of time a receptor would contact all parts of the exposure area. For RME intake and dose estimates, the 95% upper confidence limit (UCL) of the mean was utilized as the EPC. The 95% UCL of the mean provides a conservative estimate of the average concentration of a chemical across an exposure area. For central tendency exposure (CTE) estimates of intakes and dose, the arithmetic mean value for each constituent was included as the EPC.

UCLs were calculated using the most current version of the ProUCL software (version 4.1) (USEPA, 2010a). The software evaluates the data distribution (i.e., normal, lognormal, or gamma or nonparametric) using various goodness-of-fit tests in order to calculate the appropriate 95% UCL of the mean (USEPA 2010a). ProUCL requires at least seven values in order to calculate an appropriate UCL. For sediment data that contained non-detect values, one-half the detection limit was substituted for the non-detect value. This maintains a conservative risk assessment approach since this method is likely to overestimate the EPC.

Table 9: Exposure Point Concentrations for COPCs in Sediment

Chemical	CTE Arithmetic Mean (mg/kg)	RME 95% UCL of the Mean (mg/kg)	Maximum Sediment Concentration (mg/kg)
Arsenic	8.9	13.2$^{(1)}$	30.1
Benzo(a)anthracene	1.0	1.3$^{(2)}$	2.2
Benzo(a)pyrene	1.2	1.5$^{(2)}$	2.7
Benzo(b)fluoranthene	1.3	1.7$^{(2)}$	2.7
Benzo(k)fluoranthene	1.2	1.6$^{(2)}$	2.9
Dibenz(a,h)anthracene,	0.23	0.29$^{(2)}$	0.44
Indeno(1,2,3-c,d)pyrene	1.3	1.6$^{(2)}$	3.1
Lead	69.3	85.8$^{(2)}$	127
Total PCBs	0.07	0.11$^{(2)}$	0.37

Notes:
(1) Data distribution as determined by ProUCL
(2) 95% approximate gamma distribution
(3) 95% Student's t-UCL

3.4.2. Exposure Point Concentrations for Fish Tissue

Table 3-7 (Appendix) summarizes the EPCs included for chemical constituents in fish tissue. Due to the limited number of samples (sample size between 1 and 5 composites with each composite representing five individual fish of the same species), the EPC used for the RME estimates included the maximum value for each constituent within each of the 12 species regardless of year. EPCs for CTE estimates included the arithmetic mean or maximum value (if only 1 composite was evaluated) of each constituent within fish species regardless of year.

3.5. Quantification of Exposure

The basic equations used to calculate cancer risk and noncancer hazard estimates for the identified exposure scenarios are taken from various guidance documents (USEPA 1989, 1991b, 2004, 2009 and 2010b).

3.5.1. Dermal Contact with Sediment

The dermal absorbed dose is estimated as the dose that crosses the skin and is systemically absorbed. A dermal absorption factor (ABS) is included in this equation to account for the proportion of the chemical that is likely to be absorbed across the skin surface. This dose was estimated from the following equation (USEPA 2004):

$$Intake = \frac{CS \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT}$$

where:

Intake	= dermal absorbed dose of COPC (mg/kg-day, calculated)
CS	= concentration of COPC in sediment (mg/kg)
SA	= surface area of the skin exposed to sediment (cm^2)
AF	= soil/sediment adherence factor
ABS	= dermal absorption coefficient – COPC-specific (unitless)
EF	= exposure frequency (days/year)
ED	= exposure duration (years)
CF	= conversion factor (1E-06 kg/mg)
BW	= body weight (kg)
AT	= averaging time (days)

The specific assumptions and values included in these calculations are summarized in Tables 3-1 and 3-3 in the Appendix.

3.5.2. Incidental Ingestion of Sediment

The ingested intake of COPCs in sediment is estimated by the following equation:

$$\text{Intake} = \frac{CS \times IR \times FI \times EF \times ED \times CF}{BW \times AT}$$

where:

Intake	= ingested daily intake of COPCs in sediment (mg/kg-day, calculated)
CS	= concentration of COPC in sediment (mg/kg)
IR	= ingestion rate of sediment (mg/day)
FI	= fraction of exposure attributed to site sediment (unitless)
EF	= exposure frequency (days/year)
ED	= exposure duration (years)
CF	= conversion factor (1E-06 kg/mg)
BW	= body weight (kg)
AT	= averaging time (days)

The specific assumptions and values included in these calculations are summarized in Tables 3-2 and 3-4 in the Appendix.

3.5.3. Ingestion of Fish Tissue

The ingested intake of COPCs from fish is estimated by the following equation:

$$Intake = \frac{C_{Fish} \times IR \times EF \times ED \times CF}{BW \times AT}$$

where:

Intake	= ingested daily intake of COPCs from fish (mg/kg-day, calculated)
C_{Fish}	= concentration of COPC in fish (mg/kg)
IR	= ingestion rate of fish (g/day)
EF	= exposure frequency (days/year)
ED	= exposure duration (years)
CF	= conversion factor (kg/g)
BW	= body weight (kg)
AT	= averaging time (days)

The specific assumptions and values included in these calculations are summarized in Tables 3-5 and 3-6 in the Appendix.

4. TOXICITY ASSESSMENT

Toxicity values used in HHRAs quantify the dose-response relationship for a chemical. These values include cancer slope factors (CSFs) and noncancer reference doses (RfDs), both of which are specific to the route of exposure (USEPA 2003a). Tables 4-1 through 4-3 summarize the toxicity values, EPA weight-of-evidence for cancer classification, target organ and health effects and other pertinent information for selected COPCs. The source for these toxicity values was chosen based on the hierarchy as recommended by EPA and includes:

1. Integrated Risk Information System;
2. EPA's Provisional Peer-Reviewed Toxicity Values (PPRTVs); and
3. Other peer-reviewed toxicity values which may include California Environmental Protection Agency (CALEPA), Agency for Toxic Substances Disease Registry (ATSDR) Minimal Risk Levels, and Health Effects Assessment Summary Tables (HEAST) Toxicity Values.

4.1. Oral and Dermal CSFs

EPA has developed CSFs specific to the oral route of exposure. In accordance with EPA guidance (1989), this risk assessment uses route-to-route extrapolation to estimate dermal CSFs from oral CSF values in order to estimate the risk associated with dermal contact with contaminated soil. This extrapolation is done by dividing the oral CSF by a constituent-specific oral absorption factor. To calculate a dermal CSF for a particular chemical, the oral CSF is divided by the oral absorption efficiency value (GIABS) (USEPA 2004, 2010). The adjusted CSFs for dermal exposure are summarized in Table 4-2.

4.2. Oral and Dermal RfDs for Non-carcinogenic Effects

Oral reference doses are expressed in units of daily dose (mg/kg-day) and incorporate uncertainty factors to account for limitations in the quality or quantity of available data. The EPA defines the RfD or RfC as an estimate of the daily maximum level of exposure to human populations (including sensitive sub-populations) that is likely to be without an appreciable risk of adverse effects across a lifetime (USEPA 1989). The oral RfD provides a benchmark against which human intakes (via ingestion) are compared.

In this risk assessment, dermal RfDs were extrapolated from the oral RfD values using the appropriate oral absorption factors. In order to calculate a dermal RfD for a specific chemical, the oral RfD is multiplied by the oral absorption efficiency value expressed in decimal form (USEPA 2004). The absorption efficiencies and the adjusted RfDs used are presented in Table 4-2.

4.3. Evaluation of Potential Exposure to Lead

4.3.1. Lead Exposure in Adults Using the Adult Lead Model

The EPA Adult Lead Model (ALM) was utilized to determine if nonresidential lead exposures at the site pose a significant risk (USEPA 1994b, 2003b, 2007, 2009). This method focuses on estimating blood lead concentrations (PbB) in fetuses carried by women exposed to average concentrations of lead measured in environmental media (adult exposure to soil; ultimate receptor is fetus). Unlike the IEUBK model, the ALM does not consider contributions from other environmental media but it does account for a non-zero baseline blood-lead level. The default baseline blood lead levels were assumed.

This method is based on a probability model for PbB in adult women exposed to lead in environmental media coupled with an estimated constant of proportionality between fetal and maternal PbBs, a geometric mean fetal PbB concentration and an empirically determined geometric standard deviation. The statistical terms used in the method allow the user to estimate an average adult PbB such that a fetus has not more than a five percent probability of PbB exceeding 10 μg/dL. Soil lead levels with no more than a five percent chance that the blood lead level in a fetus will exceed 10 μg/dL are considered to be below the risk threshold (USEPA 2003b).

Table 4-4 summarizes the parameters selected for use in the ALM for assessing lead exposures for adults. As above, intake rates and exposure frequencies for contact with site media are the same as the exposure parameters utilized for non-lead chemicals (Tables 3-1 and 3-2), and the biokinetic modeling parameters are the recommended defaults (USEPA 2003b).

4.3.2. Lead Exposure in Children Using the IEUBK Model

Toxicity values are not available to evaluate the noncancerous health risks associated with lead so it must be evaluated using a separate methodology. EPA considers the development of a reference dose (RfD) to be inappropriate because no threshold has been established for the most sensitive noncancer effects of lead in infants and young children (USEPA 1994). The Integrated Exposure Uptake Biokinetic (IEUBK) Model was developed to predict the probable blood lead level for children between 6 months and 7 years of age who have been exposed to lead through environmental media (air, water, soil, dust, and diet) (USEPA 2010b). This model utilizes separate components for exposure, absorption and the biokinetic transfer of lead to all tissues of the body and calculates age-specific blood lead concentrations for children. According to EPA recommendations, model results protective of human health include those for which the probability of a blood level >10 µg/dL is less than 5 percent in the selected exposure group (USEPA 1994). The 10 µg/dL blood lead level was selected based on studies indicating that exposures resulting in blood lead levels at or above this concentration may present an increased health risk to children (CDC 1991, 2002).

For the current evaluation, input values selected for the parameters in the IEUBK model are summarized in Table 4-5. The model was run with a combination of EPA default parameters and site-specific information for lead in sediment and fish as noted in the table. Upper level values were included in the model run to represent an RME scenario. This model considers additional sources of lead exposure such as outdoor air (1 µg/m^3), drinking water (4 µg/L) and maternal blood lead level at birth (1 µg/dL).

EPA has a goal of limiting exposure to lead in soil such that “a typical (or hypothetical) child or group of similarly exposed children would have an estimated risk of no more than 5 percent of exceeding a 10 µg/dL blood lead level” (USEPA 1994). The 10 µg/dL blood lead level was selected based on studies indicating that exposures resulting in blood lead levels at or above this concentration may present an increased health risk to children (CDC 1991, 2002). The results of the model run are included in Table 4-5 and discussed in the next section.

5. RISK CHARACTERIZATION

5.1. Direct Contact with Contaminated Sediment

5.1.1. Characterization of Cancer Risks

Quantification of cancer risks involves the calculation of ELCRs or excess lifetime cancer risks. These values represent the probability of an individual developing cancer over a 70-year lifetime associated with exposure to a cancer-causing chemical. An ELCR of 1×10^{-6} indicates that an exposed individual has a one in a million increased risk of developing cancer as a result of exposure to the specified chemical.

ELCRs for evaluation of dermal contact and ingestion pathways were calculated for each COPC using the following formula:

$$ELCR = \sum_{i=1}^{n} (LADI_i \ or \ LADD_i) \ x \ CSF_i$$

The CSFi is expressed in units of mg/kg-day)$^{-1}$ for each compound and the lifetime average daily intake (LADI) and lifetime average daily dose (LADD) are expressed in units of mg/kg-day for each compound. The resultant product, or ELCR, is dimensionless since the units cancel out.

Table 10 summarizes the estimated CTE and RME summative cancer risk estimates for contact with sediments by exposure group. The majority of risk estimates across exposure groups are below the risk level of 1×10^{-5}. However, the cancer risk estimate for the high-end child exposure group was higher than this level at 3.7×10^{-5}. This summative risk was driven by the dermal exposure pathway.

The cancer risks by COPC are summarized in Tables 5-1 through 5-4 in the Appendix. For the dermal contact pathway, benzo(a)pyrene contributed the most to the summative cancer risk followed by arsenic. For the incidental ingestion pathway, the converse was true with arsenic having the highest contribution to the risk estimates followed by benzo(a)pyrene. This was true for all exposure groups.

Table 10: Summary of Noncancer and Cancer Risks for Direct Contact with Sediments for Presque Isle Bay Recreational Water Users

	Noncancer risks			Cancer risks		
	HQ_{dermal}	HQ_{oral}	HI	$ELCR_{dermal}$	$ELCR_{oral}$	$ELCR_{sum}$
Adult RME	1.9E-02	9.4E-03	2.9E-02	6.5E-06	2.5E-06	9.0E-06
Adult CTE	1.4E-03	8.9E-04	2.2E-03	1.6E-07	7.8E-08	2.4E-07
Child RME	4.8E-01	8.8E-02	5.7E-01	3.3E-05	4.7E-06	3.7E-05
Child CTE	8.9E-03	8.3E-03	1.7E-02	7.1E-07	4.9E-07	1.2E-06
Adult Lead Model(3) RME	**Probability that fetal PbB > PbB_t = <0.6%**					
Adult Lead Model(3) CTE	**Probability that fetal PbB > PbB_t = <0.4%**					
Child IEUBK Model(4)	**Probability that child PbB > 10 µg/dL < 0.3%**					

Notes:

(1) No available RfD or RfC
(2) No cancer slope factor available for oral exposures
(3) The Adult Lead Model (EPA, 2009) was used to assess the noncancer risks from lead exposure.
(4) The Child IEUBK Model was used to assess the noncancer risk from lead exposure

Acronyms

HQ = hazard quotient
HI = Hazard Index (sum of HQs across exposure pathways)
ELCR = excess lifetime cancer risk

5.1.2. Characterization of Noncancer Risks

Risk characterization of noncancer effects of a chemical involves comparing the ratio of the Average Daily Intake (ADI) or the Average Daily Dose (ADD) to the RfD for the ingestion or dermal contact routes. This ratio is referred to as the Hazard Quotient (HQ) and is calculated as follows:

$$HQ_i = \frac{ADI_i \text{ or } ADD_i}{RfD_i}$$

HQs for the same chemical but from different exposure pathways were calculated by summing across all HQs. To calculate the cumulative HI, which represents the adverse effects associated with simultaneous exposure to all detected chemicals, all the calculated chemical-specific HQs were summed to derive a hazard index for each chemical. The HIs were summed across exposure pathways for all COPCs since there is considerable overlap between the systemic effects.

The noncancer risk estimates for recreational water users are summarized in Table 10. The HIs across all exposure pathways for each COPC were less than 1 indicating that noncarcinogenic effects from contact with contaminated sediments are not likely to occur. Arsenic and total PCBs were the only COPCs which had established RfDs allowing for noncancer risks to be calculated. Total PCBs contributed the most to noncancer risks for deremal contact while arsenic had the higher contribution to incidental ingestion of sediments.

5.1.3. Adult Lead Exposures (Noncancer Effects)

Results of the adult lead model for adult recreational water users are summarized in Table 10 and in Table 4-4 (Appendix). The modeling results included the updated adult female PbB estimates from the 2000-2004 NHANES III Study (USEPA 2009b). The modeling results estimated the probability that fetal PbB would exceed 10 µg/dL on-site to be less than one percent for both the CTE and RME calculation for adult recreational water users. This result suggests that females exposed to lead through direct contact with sediment within the study area have a low probability of developing blood lead levels that would cause harm to the fetus (USEPA 2003b).

5.1.4. Lead Exposures in Children (Noncancer Effects)

The results of the IEUBK model show a low probability of risks from lead exposure among children exposed to sediments and fish tissue within the study area (refer to Table 10 and Table 4-5 in the Appendix). The model was run including both EPA default values and site specific assumptions. The results include exposures from contact with contaminated sediments and ingestion of contaminated fish. It also includes exposures to lead from other sources such as outdoor air, drinking water and from maternal exposures. The results of the IEUBK model suggest that the probability of a child developing blood lead levels of 10 µg/dL and above is less than 0.3 percent considering site-specific and other exposures. This is well below the EPA target level of five percent.

5.2. Ingestion of Contaminants in Fish Tissue

5.2.1. Characterization of Cancer Risks

The procedure for calculating cancer risks for contaminants in fish tissue was identical to that utilized for the exposure pathways for sediment. The summative cancer risks by exposure group are summarized in Table 11. The summative excess cancer risks for the high end exposure scenarios (urban/subsistance anglers) were very significant varying from 1.0 × 10^{-4} to 4.9 × 10^{-3} for adult anglers. The fish species showing the highest summative cancer risk was lake trout at 4.9 × 10^{-3}. The main COPC driving these risk estimates was either Arochlor 1254 or 1260 or both (refer to Tables 5-5 through 5-8 in the Appendix). Summative CTE estimates for cancer risk in adult anglers were in the range of 1 × 10^{-6} to × $10^{-5.}$ These values are still within the range of concern as delineated by EPA.

Table 11: Summary of Noncancer and Cancer Risks for Ingestion of Fish

Fish species	Noncancer risks (summative HIs)				Cancer risks (summative ELCRs)			
	Adult		Child		Adult		Child	
	RME	CTE	RME	CTE	RME	CTE	RME	CTE
Presque Isle Bay Anglers and Their Children								
Bluegill	No COPCs							
Brown bullhead[(1)]	0.41	0.05	0.47	0.05	n/a	n/a	n/a	n/a
Common carp	48.2	6.0	48.3	6.0	8.0E-04	3.0E-05	1.6E-04	2.0E-05
Largemouth bass	11.4	1.6	11.4	1.5	1.0E-04	2.8E-06	2.6E-05	1.9E-06
Northern pike	0.67	0.08	0.67	0.08	1.0E-04	3.6E-06	1.9E-05	2.4E-06
Pumpkinseed	No COPCs							
Lake Erie Anglers and Their Children								
Burbot	10.8	0.95	10.8	0.91	1.3E-04	2.7E-06	2.6E-05	1.8E-06
Channel catfish	100	8.5	99.8	8.4	2.1E-03	5.0E-05	4.0E-04	3.3E-05
Lake trout	183.7	3.2	183.8	3.2	4.9E-03	5.0E-05	1.0E-03	3.1E-05
Smallmouth bass	132.8	12.5	150.1	12.5	3.1E-03	7.0E-05	6.1E-04	4.7E-05
Walleye	39.4	3.2	39.3	3.2	7.4E-04	2.0E-05	1.4E-04	1.2E-05
White bass	26.4	3.1	26.4	3.1	5.9E-04	2.0E-05	1.2E-04	1.4E-05
White sucker	30.5	3.5	30.5	3.5	4.6E-04	2.0E-05	9.4E-05	1.1E-05
Yellow perch[(2)]	3.1	0.27	3.1	0.27	n/a	n/a	n/a	n/a

Notes:
(1) only selenium found over the RSL – no cancer toxicity value is available
(2) only selenium and mercury found over the RSL – no cancer toxicity values available

5.2.2. Characterization of Noncancer Risks

The procedure for calculating noncancer risks for contaminants in fish tissue was identical to that utilized for the exposure pathways for sediment. The summative noncancer risks, or hazard indices, by exposure group are summarized in Table 11. Lake trout was the species with a consistently higher hazard index across exposure groups. The summative hazard index for RME estimates was approximately 184 for both adult anglers and children of adult anglers. These values are much higher than the target level established by EPA which is 1. As with cancer risk estimates, Arochlor 1254, 1260 or both was the COPC with the highest contribution to the noncancer risk estimates.

6. UNCERTAINTY ANALYSIS

Uncertainties are inherent in any human health risk assessment due to the use of environmental sampling results, modeling approaches, assumptions regarding exposure, and the toxicity of particular constituents. This risk assessment has incorporated site-specific information, where feasible, in order to reduce the uncertainty associated with those assumptions. However, in many instances, there was little quantitative information to include in terms of site-specific assumptions for Presque Isle Bay and/or Lake Erie.

Analysis of the critical areas of uncertainty in risk assessment provides context for better understanding the assessment conclusions by identifying the uncertainties expected to most significantly affect the results. In this risk assessment, where assumptions were made, the uncertainty errs on the conservative side in order to protect human health (i.e., overestimate human health risks). Table 6-1 within the Appendix summarizes the major sources of uncertainty in this risk assessment and provides a qualitative judgment on the magnitude of each source in terms of its likelihood to under- or over-estimate human health risks.

6.1. Hazard Identification

6.1.1. Data Accuracy and Site Characterization

A major concern of any risk assessment is the accuracy and completeness of COPC identification, both in terms of ensuring that all contaminants have been correctly identified as COPCs, and ensuring that concentrations are adequately quantified. In order to maintain precision and accuracy of sampling and analytical procedures, EPA-approved sampling and analytical procedures were followed in order to characterize the site. All samples were collected and analyzed following appropriate quality assurance/quality control procedures.

The accuracy of COPC identification is directly related to the quality of COPC characterization data, including information on contaminant identification, location, and concentrations. The sampling data included in this analysis was collected for purposes other than an HHRA. As a result, samples were not necessarily collected in areas specific to exposure activities (i.e., swimming, wading, fishing) such as those characterized in this study. Based on best professional judgment, the sampling data were grouped in an attempt to best reflect exposure areas. It is possible, however, that sample locations could have been included for an exposure that may overestimate potential exposure for some populations while underestimating potential exposure for others.

A limitation of the fish tissue data is that in many instances, the chemical concentration was based on one composite sample of five individual fish. This adds uncertainty to the risk assessment in that it is difficult to ascertain how representative this composite is for the species as a whole in Presque Isle Bay. Less uncertainty exists for those species which included more than one composite in the analysis.

Sediment and fish tissue samples were collected from the site over a limited number of days. Although these data were collected during the spring, summer and fall, they represent a snapshot in time and may not be representative of concentrations present at other times of the year under different conditions.

6.1.2. Screening of COPCs Using RSLs

The screening criteria used at the Site were chosen to represent conservative and reasonable screening criteria as established by EPA Region 3. The screening process was designed to identify those constituents that were site-specific and likely to exceed conservative risk-based criteria for residential use. These criteria therefore, include the assumption that an individual would be exposed to sediments 350 days per year for 30 years. A level of uncertainty exists with chemicals that do not have a specific RSL. In this instance, a surrogate value was used which represents a chemical constituent with a similar structure that is assumed to pose the identical human health risks.

A number of uncertainties exist in the selection of COPCs for inclusion of the risk assessment including those associated with sampling/analytical procedures; the number of samples for use to estimate the COPC and the selection of the appropriate screening criteria.

6.1.3. Detection Limits Greater than the RSLs

Uncertainty exists in the screening and evaluation of chemicals that had method detection limits exceeding the EPA Region 3 RSLs. Site-specific RSLs for some chemicals are exceptionally low, and in some instances, may not be attainable with currently available laboratory methods. For fish tissue samples, detection limits exceeded the screening levels for many of the COPCs (see Table 2-3 through 2-16). Chemicals that were not detected were not carried through the entire risk assessment evaluation. If chemicals were present at concentrations above the screening levels but below the detection limits, it is possible that these chemicals could contribute to unacceptable risks.

6.1.4. Use of Trivalent Chromium Toxicity Parameters for Total Chromium

Chromium was analyzed as total chromium in all media. However, screening values only exist for the hexavalent and trivalent species. For the purposes of this risk assessment it was assumed that the majority of the total chromium measured in sediment and fish tissue was in the trivalent form. Therefore, screening levels for trivalent chromium were used to determine whether chromium would be carried forward in the risk assessment. While there is uncertainty in this approach it is likelihood of underestimating the risk from hexavalent chromium is minimal. As previously discussed, the majority of chromium within a reducing environment is the trivalent form and hexavalent chromium has a low BCF in fish tissue indicating that it is unlikely to bioaccumulate (ATSDR 2008, USEPA 1998).

6.1.5. Use of Structural Analogy to Determine Surrogate Screening Levels

Some chemical constituents in sediment and fish tissue did not have an associated screening level. In this instance structural analogy was used to screen this particular chemical. This is generally due to the lack of toxicity information available for this specific chemical constituent. For chemical constituents in sediment the following substitutions were made: pyrene for benzo(e)pyrene and perylene; anthracene for phenanthrene; and acenaphthene for benzo(g,h,i)perylene and acenaphthalene. The use of surrogate screening levels may under- or overestimate the risk associated with a particular chemical.

6.2. Exposure Assessment

In order to estimate the amount of a COPC for a particular receptor a number of assumptions must be made about the duration and frequency of exposure and characteristics inherent to a particular receptor (i.e., body weight, skin surface exposed to soil). Although effort has been taken to apply site-specific and receptor-specific exposure factors, for those with limited data, EPA defaults were used in a number of cases. These recommended defaults are also based on limited data and are chosen to represent conservative estimates. It is likely that the actual exposure factors are much lower than the default values suggested by the EPA resulting in an overestimation of the human health risks.

6.2.1. Lack of Data to Evaluate Surface Water Exposure

There was no data available to evaluate exposure from contact with chemicals in surface water. Therefore, exposure pathways such as inhalation, dermal contact and incidental ingestion of contaminated water could not be evaluated. This is likely to underestimate the risk, however, the impact is likely to be negligible. Many of the chemicals of concern are lipophilic in nature and not likely to be present in appreciable concentrations within the water column.

6.2.2. Exposure Assessment from Fish Consumption

6.2.2.1. Fish Consumption Rates

There was little quantitative information on fish consumption rates in the Presque Isle Bay area therefore, fish consumption rates were based on the national per capita consumption of estuarine and freshwater fish (USEPA 2002). The 90th and 99th percentile ingestion rates for children and adults were selected to evaluate potential risks over a range of possible ingestion rates. The extent to which these assumptions correspond to consumption patterns in the study area is unknown.

6.2.2.2. Use of Single Fish Species Consumption Pattern

Risk estimates were based on the consumption of individual fish species and tissue types. However, it is very likely that an individual’s diet would include multiple fish species. A mixed-diet scenario was not evaluated for this risk assessment because of the lack of species-specific consumption data for the study area.

6.2.2.3. Use of Fillets to Represent All Fish Consumption Patterns

Bioaccumulation of chemicals in fish tissue will differ depending on the chemical. Organic compounds, especially lipophilic chemicals, tend to accumulate in fatty tissues while metals tend to accumulate in muscle and other tissues (PA Fish and Boat 2011, Gutenmann 1992). The chemicals with the greatest contribution to the cumulative cancer risk and with the highest noncancer HQ are the Arochlors, which are organic compounds that accumulate preferentially in fatty tissue.

Diets consisting of different fish parts result in varying levels of risk to the consumer. Using only whole body or fillet tissue with skin to evaluate risk from all types of fish tissue diets is likely to overestimate chemical exposure from consumption of contaminated fish. Since PCBs contribute to the vast majority of risks from tissue consumption, this uncertainty could have a significant impact on the conclusions of this HHRA. Alternatively, chemicals such as methyl mercury preferentially accumulate in muscle tissue, which means concentrations of mercury in fillet tissue would likely be higher than concentrations of mercury in whole body fish tissue.

6.2.2.4. Assumption of Residence Time of Fish Species

For the purposes of this HHRA the fish species were assumed to be denizens of either Presque Isle Bay or Lake Erie based on sampling location and habits of each species. Species that were considered to reside mostly in Presque Isle Bay included: bluegill, largemouth bass, northern pike, pumpkinseed sunfish, brown bullhead and common carp. Fish species assumed to reside in Lake Erie included: burbot, channel catfish, lake trout, smallmouth bass, walleye, white bass and white sucker. A summary of these assumptions is included in Table 4 (page 14 and 15 of this report).

Since many fish are migratory in nature (i.e., burbot and smallmouth bass during spawning) (Grazio, 2012) and migrate between the lake and bay on a seasonal basis, their exposure may represent chemicals from time spent in both the Bay and Lake waters.

6.2.2.5. Sample Size and Length of Fish Collected for Study

The sample size of the fish analyzed in this study represents another source of uncertainty in the risk estimates. Limited numbers of fish of an individual species were collected between 2004 and 2010. Each composite represents ten fillets that were collected at a given period of time. This small sample size and sampling period may not adequately represent the concentration of contaminants in fish species within the general study area.

Fish were collected such that composite samples included fish species of similar lengths and therefore, age (PADEP 2010). The length, and in essence, the age of fish, is positively correlated with the contaminant body burden concentration within fish tissue (Gutenmann et al. 1992; Young, et al. 1994). The risk estimates in this HHRA are based on various fish species of a given length. Individuals that consume fish of a smaller or longer length than those included in this study may have risks that lower or higher than estimated in this report. Fish species with size limit regulations are more likely to be better represented in the data since the sampled fish are close to the allowable size limits.

6.2.3. Exposure Point Concentration

The sampling data included in this analysis was collected for purposes other than an HHRA and, therefore, may have limitations in terms of adequately characterizing all the human exposure scenarios evaluated in this report. In order to account for these uncertainties upper bound estimates or maximum values were included in this evaluation so as not to understate any potential risk.

Uncertainties associated with the exposure assessment include calculation of exposure point concentrations and selection of exposure parameters. The RME approach was utilized in this HHRA to characterize risks. The use of high-end values as exposure parameters, including the 95% UCLs, prevents an underestimation of the health risks. In addition, the maximum value of the COPCs were utilized for the RME estimates which is likely to overestimate the intake and dose calculations from exposure to sediment and thus overestimate the health risks.

For some chemicals in sediment samples, the calculation of average exposure point concentrations relied upon sample data where the concentration was reported as below the detection limit. These chemicals were assumed to be present at a concentration equal to one-half the detection limit in order to calculate an EPC. This practice increases the uncertainty of the resulting exposure point concentrations because the actual sample concentration may range from zero to the full detection limit.

6.2.4. Extrapolation of Chemical Concentrations Over Time

Another source of uncertainty in this risk assessment involves the use of the average chemical concentrations for fish and sediment collected over a short period of time to estimate human exposure durations of 9, 30 and 70 years. If average chemical concentrations in these media have changed over time, or are likely to change in the future, the risk estimates presented in this report may either underestimate or overestimate the risk to individuals. The existing historical data on sediment contamination in Presque Isle Bay suggests that many of the chemical concentrations are decreasing over time (PADEP 2005). If this trend continues, the extrapolation of current chemical concentrations into the future is likely to overestimate the human health risks from exposure to sediment and fish tissue.

6.2.5. Exposure Duration and Frequency

Exposure duration is defined as the time period over which an individual is exposed to one or more contaminants. Two defaults were used for the risk assessment: 70 years, which represents the average lifetime exposure duration; 30 years, which represents the 90th percentile length of time that an individual resides at one residence; and 9 years which represents the median amount of time an individual resides at a given residence These parameters are conservative default values obtained from EPA guidance documents and are typically used to estimate CTEs and RME. These values may overestimate the risks for actual receptors.

The frequency of residents' exposure to bay sediments was determined by using best professional judgment with consideration of the weather conditions in western Pennsylvania. This value is higher than the default value indicated in the exposure factor handbook of 12 days per year. The exposure frequency utilized in the exposure estimates may overestimate the exposure contact with sediments for some but underestimate it for individuals frequently participating in water activities,

6.2.6. Use of Dermal Absorption Factors for Soil

The bioavailability of COPCs in sediment was considered by using the dermal absorption factors for soil in the dose calculations (USEPA 2011a). Unlike soil, sediments are consistently water-covered, more likely to wash off, and consequently tend to have a shorter contact time on skin than soil. As a result, dose calculations may be overestimated. Default absorption factors were included for those chemicals that did not have a specific one available. For example, inorganic chemicals, such as chromium was considered to have a value of 0.1. This assumption is likely to overestimate the actual amount of chromium absorbed through the skin.

Also, the greater the moisture content of the sediment, the greater the difference between wet vs. dry weight contaminant concentration. Because estimates of sediment adherence reflect wet weight (i.e., in situ), and the estimated intakes are based on sediment sample results recorded in dry weight, the resulting risk estimates are over-estimated in direct proportion to the moisture content of the sediment. Conversely, increased moisture content increases the ability of sediment to adhere to skin and may also affect the relative percent absorbed. Therefore, EPA recommends the use of the same dermal absorption fraction for sediments as for soil until more information becomes available (EPA 2004a).

6.3. Toxicity Assessment

6.3.1. Toxicity Values

The toxicity information used in the health risk assessment adds a degree of uncertainty to the risk estimates. The uncertainties specific to the toxicity assessment are associated with: the toxicity studies that form the basis for the toxicity values recommended by EPA and the lack of sufficient toxicity data to develop toxicity values for certain substances. In order to reduce the extent of this uncertainty to the extent possible, the most current toxicity values were utilized in this risk assessment (USEPA 2011a, 2011b). The extrapolation used in developing toxicity values may contribute to uncertainty in the risk estimates. An additional source of uncertainty originates from toxicity values that are chemical-specific and do not take into account interaction with other chemicals.

The toxicity values (i.e., RfDs and CSFs) used in this risk assessment were developed by the EPA for regulatory purposes and are intended to represent upper-bound estimates of toxicity. For example, most of the RfDs incorporate large uncertainty factors which are intended to lie well below the true threshold for toxicity in humans. While this helps ensure the protectiveness of decisions based on the RfD, it should be recognized that a dosage exceeding the RfD (i.e., a HQ > 1.0) does not necessarily indicate the likelihood for toxicity given the level of uncertainty within various elements of the risk assessment process.

Similarly, the CSFs developed by the EPA incorporate a number of conservative choices in risk extrapolation. These include the assumption of a linear, non-threshold dose-response relationship for cancer, interpretation of animal carcinogenicity data, and dose-metrics for extrapolation of results from rodents to humans. As a result, estimates of lifetime cancer risks including these values reflect conservative upper bound estimates of risk associated with specific exposures. They may be extrapolated from high-dose to low-dose models, laboratory animal studies, and/or subchronic studies.

6.3.2. PCB Congeners and Arochlor Mixtures

In this risk assessment, two different classes of PCBs were measured. In sediment samples, 19 PCB congeners were measured. Only two of the congeners had specific toxicity values. The approach used was to sum the concentrations of all PCB congeners to develop a total PCB concentration. This total concentration was then compared to the toxicity values for high risk PCBs. Only two of the congeners (PCB 105 and PCB 118) are considered to be dioxin-like congeners. Therefore, this approach is likely to overestimate the risks associated with PCB congeners in sediment.

In fish tissue, a total of 6 Arochlors were measured including Arochlor 1221, 1232, 1242, 1248, 1254 and 1260. These represent commercial mixtures of PCBs with the last two digits of the Arochlor representing the percentage by weight of chlorine in the mixture. The reference dose for Arochlor 1254 was used as a surrogate for Arochlor 1260 which does not have an associated reference dose. This approach adds uncertainty to the calculation and is likely to overestimate the noncarcinogenic risk associated with exposure to Arochlor mixtures.

6.3.3. DDD, DDE, DDT and its Derivatives

DDT and its derivatives, DDD and DDE, were measured in both sediment and fish tissue samples. For noncarcinogenic risk estimates, a conservative approach was employed which involved the summation of DDT, DDD, and DDE per sample (total DDT) and the use of the RfD associated with DDT to calculate an HQ. Alternatively, only DDT could have been used in the HQ because it alone has an RfD. DDT has been identified as having a hepatic health endpoint as based on the RfD value, and therefore the treatment of DDT and its derivatives will affect the HQ and the HI for hepatic toxicity.

6.4. Risk Characterization

The summation of HQs and ELCRs across chemicals and pathways are primary uncertainties in the risk characterization. Summation of HQs across different COPCs is most properly applied to compounds that induce the same effects by the same mechanism. However, in the absence of information on the toxicity of specific chemical mixtures, it is assumed that ELCRs and HQs are additive (i.e., cumulative) (EPA 1989). One of the limitations of this approach for noncarcinogens is that the effects of a mixture of chemicals are generally unknown and it is possible that the interactions could be synergistic, antagonistic, rather than additive. Additionally, the estimated values of the RfDs have different accuracy and precision and are not based on the same severity or effect.

7. CONCLUSIONS

As part of this human health risk assessment excess lifetime cancer risks and hazard indices were calculated for direct contact with sediments from Presque Isle Bay and ingestion of fish from Presque Isle Bay and Lake Erie. These risks were compared to the target levels established by EPA of 10^{-5} (cancer risks) and 1.0 (noncancer risks) for the following exposure groups:

- PIB Adult recreational water users (RME and CTE);
- PIB Child recreational water users (RME and CTE);

- PIB and Lake Erie Adult urban/subsistence anglers (RME)
- PIB and Lake Erie Adult recreational anglers (CTE)
- PIB and Lake Erie Children of urban/subsistence anglers (RME)
- PIB and Lake Erie Children of recreational anglers (CTE)

7.1. Risks Associated with Direct Contact with Contaminated Sediments

A quantitative analysis was conducted to evaluate risks to recreational water users for direct contact of contaminants in sediments. All chemical-specific and cumulative excess lifetime cancer risk estimates were below 1×10^{-5} and all chemical-specific and cumulative hazard indices were below 1.0 with the exception of the RME cancer estimate child recreational water users. The estimate for this exposure group was 4×10^{-5} and mainly driven by dermal exposure from total PCBs. It should be noted that these risk estimates are conservative in nature and likely to overestimate the risk (the uncertainties associated with these estimates are discussed in section 6 of this report).The results of the Adult Lead and IEUBK models show that lead concentrations measured in bay sediments pose insignificant noncancer health risks to child and adult female populations.

7.2. Consumption of Contaminated Fish

Cancer and noncancer risks were analyzed for the consumption of 14 separate fish species using sampling data gathered in Presque Isle Bay and Lake Erie between 2004 and 2010. The total number of COPCs varied by fish species from zero to 16. For this particular dataset, lake trout and smallmouth bass were the fish species with the highest and second highest cancer and noncancer risks compared to the other twelve species. Panfish, including both pumpkinseed and bluegill species, had the lowest concentrations and lowest risks of all fish species evaluated. The contaminants with the largest contribution to the summative risk estimates included Arochlor 1254 and 1260. In all cases, these COPCs contributed more than 50 percent of the overall cancer and noncancer risk estimates (data not shown).

Fish tissue sampling data from the current study was compared with data collected at other areas within Lake Erie (refer to Table 7-1 in Appendix) (Carlson et al. 2000, Perez-Fuentetaja et al. 2006 and Sadraddini et al. 2011). There was not enough data from the current study or information in the comparison studies from which to conduct statistical analyses. A qualitative comparison shows that many of the concentrations measured in fish tissue in Presque Isle Bay were comparable or lower than those measured in other studies.

The results of this comparison should be used with caution due to the limited amount of sampling data and differences in study methodology. Other issues that should be noted include differences in sampling time period, sample size, and species evaluated.

7.3. Main Conclusions

1. Overall, these results show that the main exposure route for contaminants in Presque Isle Bay is through fish consumption. These risks were several orders of magnitude greater than those associated with direct contact with contaminated sediments.
2. The cancer and noncancer risk estimates generated from consumption of fish tissue were highly dependent on the fish species and location (refer to Table 11). Based on the dataset utilized in this HHRA, several species from Lake Erie contributed to higher risks compared to species from Presque Isle Bay. These findings include:
   - Lake trout and smallmouth bass represented the fish species with the highest cancer and noncancer risk estimates. These species are likely to have a higher residence time and thus represent exposures to chemical constituents that occurred mainly from open water areas of the lake.
     - The summative noncancer risk for lake trout was approximately 3 for the typical or CTE estimate and 184 for the high-end or RME estimate (target level = 1.0). This latter value indicates that the estimated exposure to this chemical from consuming fish is 184 times greater than the level recommended by the EPA.
     - The summative cancer risk for lake trout was 5×10^{-5} (5 in 100,000) and 5×10^{-3} (5 in 1,000) for the CTE and RME estimates respectively (target level = 1 in 100,000).
     - The contaminant with the highest contribution to the noncancer and cancer risk estimates for lake trout and smallmouth bass was Arochlor 1254 or 1260.

       (It should be noted that the cancer and noncancer risk estimates include the assumption of a single species diet and that all fish consumed originates from Lake Erie. These assumptions are conservative in nature and likely to overestimate the cancer and noncancer risks from consumption of fish. It should also be considered that these risk estimates are based on a limited sampling of fish tissue.)
3. Values for certain fish species from Presque Isle Bay were also greater than the applicable cancer and noncancer risk thresholds. These results include:
   - Common carp and largemouth bass were the species with the highest associated risks.
     - The summative noncancer risk for common carp was 6 for the typical or CTE estimate and 48 for the high-end or RME estimate.

    - The summative cancer risk for lake trout was 3 X 10^{-5} (3 in 100,000) and 8 X 10^{-4} (8 in 10,000) for the CTE and RME estimates respectively (target level = 1 in 100,000).
    - The contaminant with the highest contribution to the noncancer and cancer risk estimates for common carp was Arochlor 1254.
  - The concentration of chemical constituents measured in panfish, including bluegill and pumpkinseed species, were all below the applicable fish tissue screening levels. Therefore, risk estimates were not calculated for these species.

4. The cancer and noncancer risk estimates for direct contact with contaminated sediments from Presque Isle Bay were generally below the target risk levels for all exposure groups evaluated in this HHRA. All chemical-specific and cumulative excess lifetime cancer risk estimates were below 1 X 10^{-5} (1 in 100,000) and all chemical-specific and cumulative hazard indices were below 1.0. The exception to this was the RME cancer risk estimate for child recreational water users which was 4 X 10^{-5} (4 in 100,000). This value is mainly driven by dermal exposure from total PCBs. It should be noted that these risk estimates are conservative in nature and likely to overestimate the risk (the uncertainties associated with these estimates are discussed in section 6 of this report).

5. The uncertainties associated with this risk assessment should be considered in utilizing the results for risk management decisions. A summary of the uncertainties inherent to this HHRA are discussed in section 6 of this report. The major uncertainties noted include the:

- small dataset from which the risk estimates were drawn (i.e., data for certain fish species included one composite sample of five individual fish);
- lack of specific data for the environmental media to which exposure groups are more likely to contact (i.e., for children beach sediment is a more likely exposure media compared to in-water sediment on which the risk estimates are based); and
- lack of site-specific information on fish consumption patterns within the study area.

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APPENDIX for

An Evaluation of Human Health Risks from Contaminants in Presque Isle Bay, Erie, Pennsylvania

July 19, 2012

Table 2-1
Sediment Sampling Data and COPC Selection Summary

COPC	Units	Screening Criteria EPA Region 3 RSLs: Conc. Used for Screening Level	Screening Criteria EPA Region 3 RSLs: Basis for Screening Toxicity Value	Contaminant range and frequency: Minimum Detected Value	Contaminant range and frequency: Maximum detected value	Contaminant range and frequency: No. of detects/ No. of samples	Contaminant range and frequency: Location of maximum value	Selection of COPCs: Constituent selected as COPC	Selection of COPCs: Reason for Inclusion/ Exclusion
METALS									
Arsenic	**mg/kg**	**0.39**	**C**	**1.7**	**30.1**	**14/14**	**47-PIP**	**Y**	**>RSL**
Cadmium	mg/kg	7	NC	<0.5	6.4	12/14	35-PIB	N	<RSL
Chromium	mg/kg	12,000(1)	N	10.8(2)	48.4(2)	14/14	35-PIB	N	<RSL
Copper	mg/kg	310	NC	24.6	103	14/14	18-PIB	N	<RSL
Lead	**mg/kg**	**40**	**NC**	**9**	**127**	**14/14**	**18-PIB**	**Y**	**>RSL**
Mercury	mg/kg	0.78(3)	NC	0.2(4)	0.4(4)	9/14	18-PIB	N	<RSL
Nickel	mg/kg	150	NC	13.5	58.9	14/14	39-PIB	N	<RSL
Zinc	mg/kg	2,300	NC	82.7	385	14/14	47-PIP	N	<RSL
PESTICIDES									
Aldrin	µg/kg	29	C	-	<7.3	1/14	23-PIB	N	<RSL
Chlordane(5)	µg/kg	1,600	C	-	<73	0/14	-	N	<RSL
Dieldrin	µg/kg	30	C	0.31	2.5	6/14	26-CC	N	<RSL
o,p'-DDD	µg/kg	2,000	C	0.32	3.3	4/14	32-PIB	N	<RSL
p,p'-DDD	µg/kg	2,000	C	0.4	6.3	12/14	26-CC	N	<RSL
o,p'-DDE	µg/kg	1,400	C	-	<7.3	0/14	-	N	<RSL
p,p'-DDE	µg/kg	1,400	C	0.23	6.7	13/14	15-PIB	N	<RSL
o,p'-DDT	µg/kg	1,700	C	0.88	0.88	1/14	39-PIB	N	<RSL
p,p'-DDT	µg/kg	1,700	C	0.63	1.5	4/14	18-PIB	N	<RSL
Endosulfan-alpha	µg/kg	37,000	NC	-	<7.3	0/14	-	N	<RSL
Endosulfan-beta	µg/kg	37,000	NC	0.69	3.3	4/14	15-PIB	N	<RSL
Endrin	µg/kg	1,800	NC	0.15	2	7/14	35-PIB	N	<RSL
Heptachlor	µg/kg	110	C	-	<7.3	0/14	-	N	<RSL
Heptachlor epoxide	µg/kg	53	C	-	<7.3	0/14	-	N	<RSL
Hexachlorobenzene	µg/kg	300	C	0.24	0.77	3/14	38-PIB	N	<RSL
Hexachlorobutadiene	µg/kg	6,200	C	1.3	1.3	1/14	38-PIB	N	<RSL
Hexachlorocyclohexane-γ	µg/kg	520	C	0.82	<7.3	0/14	-	N	<RSL
Hexachlorocyclopentadiene	µg/kg	37,000	NC	1.6	<7.3	0/14	-	N	<RSL
Methoxychlor	µg/kg	31,000	NC	0.86	0.86	1/14	38-PIB	N	<RSL
Mirex	µg/kg	27	C	0.82	<7.3	0/14	-	N	<RSL
Nonachlor, trans-	µg/kg	1,600	C	0.5	3.6	7/14	15-PIB	N	<RSL
Total Chlordanes	µg/kg	1,600	C	-	<94.9		28-PIB	N	<RSL
Total DDD, DDE, DDT	µg/kg	2,000	C	<4.89	29.7	-	15-PIB	N	<RSL

Table 2-1 (cont.) Sediment Sampling Data and COPC Selection Summary									
		Screening Criteria EPA Region 3 RSLs		Contaminant range and frequency				Selection of COPCs	
COPC	Units	Conc. Used for Screening Level	Basis for Screening Toxicity Value	Minimum Detected Value	Maximum detected value	No. of positive detects/ No. of samples	Location of maximum value	Constituent selected as COPC	Reason for Inclusion/ Exclusion
POLYCHLORINATED BIPHENYLS (PCBs)									
PCB008 (2,4'-dichlorobiphenyl)	µg/kg	NA	-	0.21	3.5	11/14	18-PIB	N	NSL
PCB018 (2,2',5-trichlorobiphenyl)	µg/kg	NA	-	0.27	12	14/14	39-PIB	N	NSL
PCB028 (2,4,4'-trichlorobiphenyl)	µg/kg	NA	-	0.44	13	14/14	35-PIB	N	NSL
PCB044 (2,2',3,5'-tetrachlorobiphenyl)	µg/kg	NA	-	0.39	10	14/14	35-PIB	N	NSL
PCB052 (2,2',5,5'-tetrachlorobiphenyl)	µg/kg	NA	-	0.54	14	14/14	35-PIB	N	NSL
PCB066 (2,3',4,4'-tetrachlorobiphenyl)	µg/kg	NA	-	0.59	14	14/14	35-PIB	N	NSL
PCB087 (2,2',3,4,5'-pentachlorobiphenyl)	µg/kg	NA	-	0.43	10	14/14	35-PIB	N	NSL
PCB101 (2,2',4,5,5'-pentachlorobiphenyl)	µg/kg	NA	-	0.61	30	14/14	35-PIB	N	NSL
PCB105 (2,3,3',4,4'-pentachlorobiphenyl)	µg/kg	110	C	0.35	9.9	14/14	35-PIB	N	<RSL
PCB118 (2,3',4,4',5-pentachlorobiphenyl)	µg/kg	110	C	0.23	14	14/14	35-PIB	N	<RSL
PCB128 (2,2',3,3',4,4'-hexachlorobiphenyl)	µg/kg	NA	-	0.16	7.2	14/14	35-PIB	N	NSL
PCB138 (2,2',3,4,4',5'-hexachlorobiphenyl)	µg/kg	NA	-	0.76	38	13/14	35-PIB	N	NSL
PCB153 (2,2',4,4',5,5'-hexachlorobiphenyl)	µg/kg	NA	-	0.79	40	14/14	35-PIB	N	NSL
PCB170 (2,2',3,3',4,4',5-heptachlorobiphenyl)	µg/kg	NA	-	0.27	15	12/14	35-PIB	N	NSL
PCB180 (2,2',3,4,4',5,5'-heptachlorobiphenyl)	µg/kg	NA	-	0.49	34	14/14	35-PIB	N	NSL
PCB187 (2,2',3,4',5,5',6-heptachlorobiphenyl)	µg/kg	NA	-	0.18	22	14/14	35-PIB	N	NSL

Table 2-1 (cont.) Sediment Sampling Data and COPC Selection Summary									
		Screening Criteria EPA Region 3 RSLs		Contaminant range and frequency				Selection of COPCs	
COPC	**Units**	**Conc. Used for Screening Level**	**Basis for Screening Toxicity Value**	**Minimum Detected Value**	**Maximum detected value**	**No. of positive detects/ No. of samples**	**Location of maximum value**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
PCB195 (2,2',3,3',4,4',5,6-octachlorobiphenyl)	µg/kg	NA	-	0.092	1.1	11/14	18-PIB	N	NSL
PCB206(2,2',3,3',4,4',5,5',6-nonachlorobiphenyl)	µg/kg	NA	-	0.18	9.1	13/14	35-PIB	N	NSL
PCB209 (decachlorobiphenyl)	µg/kg	NA	-	0.15	5.2	11/14	38-PIB	N	NSL
TOTAL PCBS(6)	**µg/kg**	**220**	**C**	**13.5**	**373**	**-**	**35-PIB**	**Y**	**>RSL**
POLYCYCLIC AROMATIC HYDROCARBONS (PAHs)									
Acenaphthene	µg/kg	340,000	NC	9.8	590	14/14	27-MC	N	<RSL
Acenaphthylene(7)	µg/kg	340,000	NC	3.3	39	14/14	47-PIP	N	<RSL
Anthracene	µg/kg	1,700,000	NC	27	1,800	14/14	27-MC	N	<RSL
Benzo(a)anthracene	**µg/kg**	**150**	**C**	**120**	**2,200**	**14/14**	**15-PIB/27-MC**	**Y**	**>RSL**
Benzo(a)pyrene	**µg/kg**	**15**	**C**	**160**	**2,700**	**14/14**	**15-PIB**	**Y**	**>RSL**
Benzo(b)fluoranthene	**µg/kg**	**150**	**C**	**200**	**2,700**	**14/14**	**15-PIB**	**Y**	**>RSL**
Benzo(e)pyrene(8)	µg/kg	170,000	NC	160	2,100	14/14	15-PIB	N	<RSL
Benzo(g,h,i)perylene(7)	µg/kg	340,000	NC	140	2,200	14/14	15-PIB	N	<RSL
Benzo(k)fluoranthene	**µg/kg**	**1,500**	**C**	**190**	**2,900**	**14/14**	**15-PIB**	**Y**	**>RSL**
Chrysene	µg/kg	15,000	C	190	3,000	14/14	15-PIB	N	<RSL
Dibenzo(a,h)anthracene	**µg/kg**	**15**	**C**	**37**	**440**	**14/14**	**39-PIB**	**Y**	**>RSL**
Fluoranthene	µg/kg	230,000	NC	280	6,200	14/14	27-MC	N	<RSL
Fluorene	µg/kg	230,000	NC	19	700	14/14	27-MC	N	<RSL
Indeno(1,2,3-c,d)pyrene	**µg/kg**	**150**	**C**	**210**	**3,100**	**14/14**	**15-PIB**	**Y**	**>RSL**
Naphthalene	µg/kg	3,600	C	13	130	14/14	18-PIB	N	<RSL
Perylene(8)	µg/kg	170,000	NC	110	870	14/14	15-PIB	N	<RSL
Phenanthrene(9)	µg/kg	1,700,000	NC	90	6,400	14/14	27-MC	N	<RSL
Pyrene	µg/kg	170,000	NC	250	4,700	14/14	27-MC	N	<RSL

Notes:

(1) RSL is for trivalent chromium
(2) The results for chromium are reported as total chromium
(3) RSL is for methyl mercury – no RSL for total mercury is available.
(4) The results for mercury are reported as total mercury
(5) Chlordane – technical grade
(6) RSL is for high risk total PCBs
(7) Toxicity criteria for acenaphthene was used to evaluate this constituent as no RSL exists
(8) Toxicity criteria for pyrene was used to evaluate this constituent as no RSL exists
(9) Toxicity criteria for anthracene was used to evaluate this constituent as no RSL exists

Acronyms:

C = cancer effects
N = noncancer effects

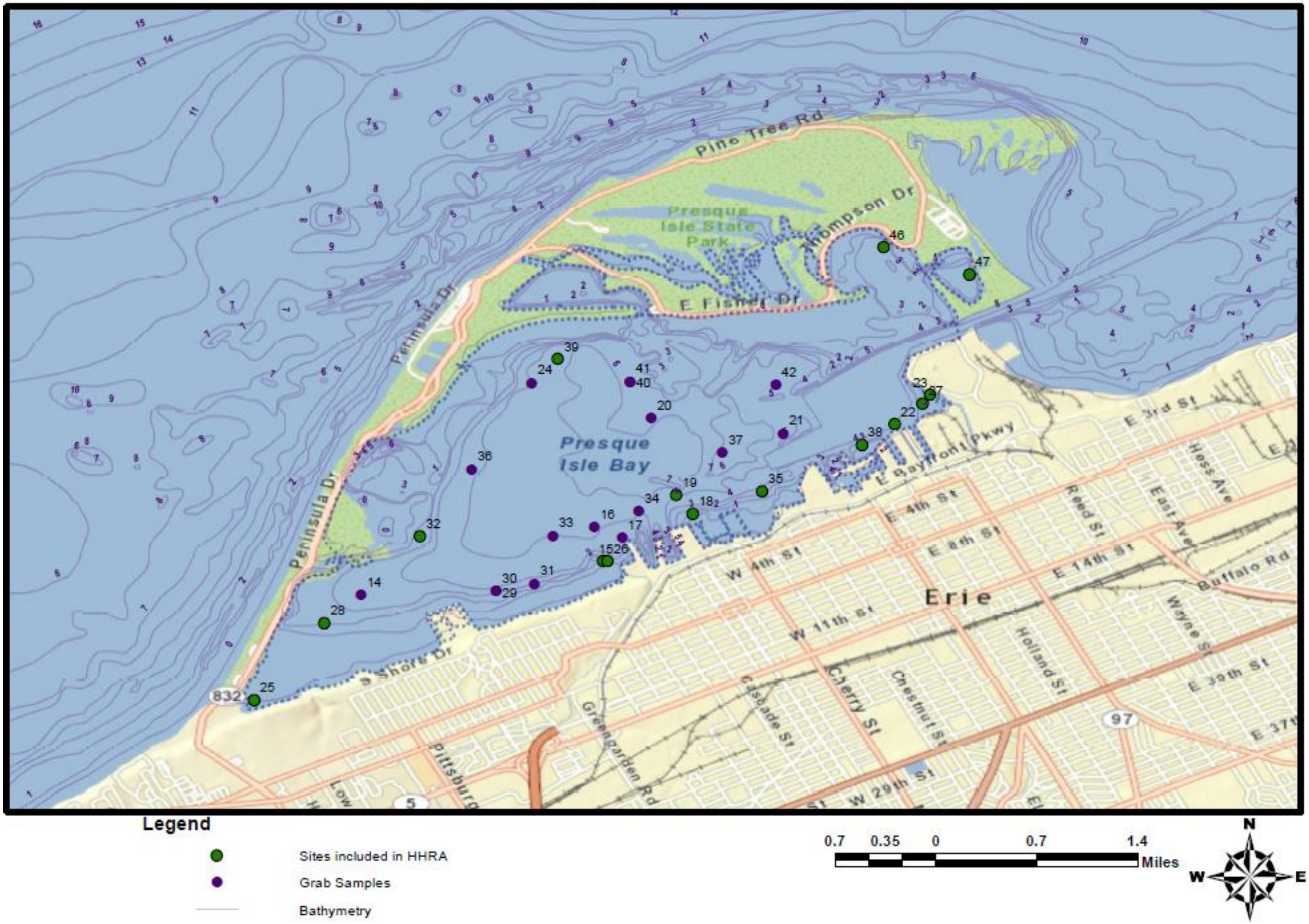


Figure 2-1: Summary of Grab Sediment Sampling Locations in Presque Isle Bay

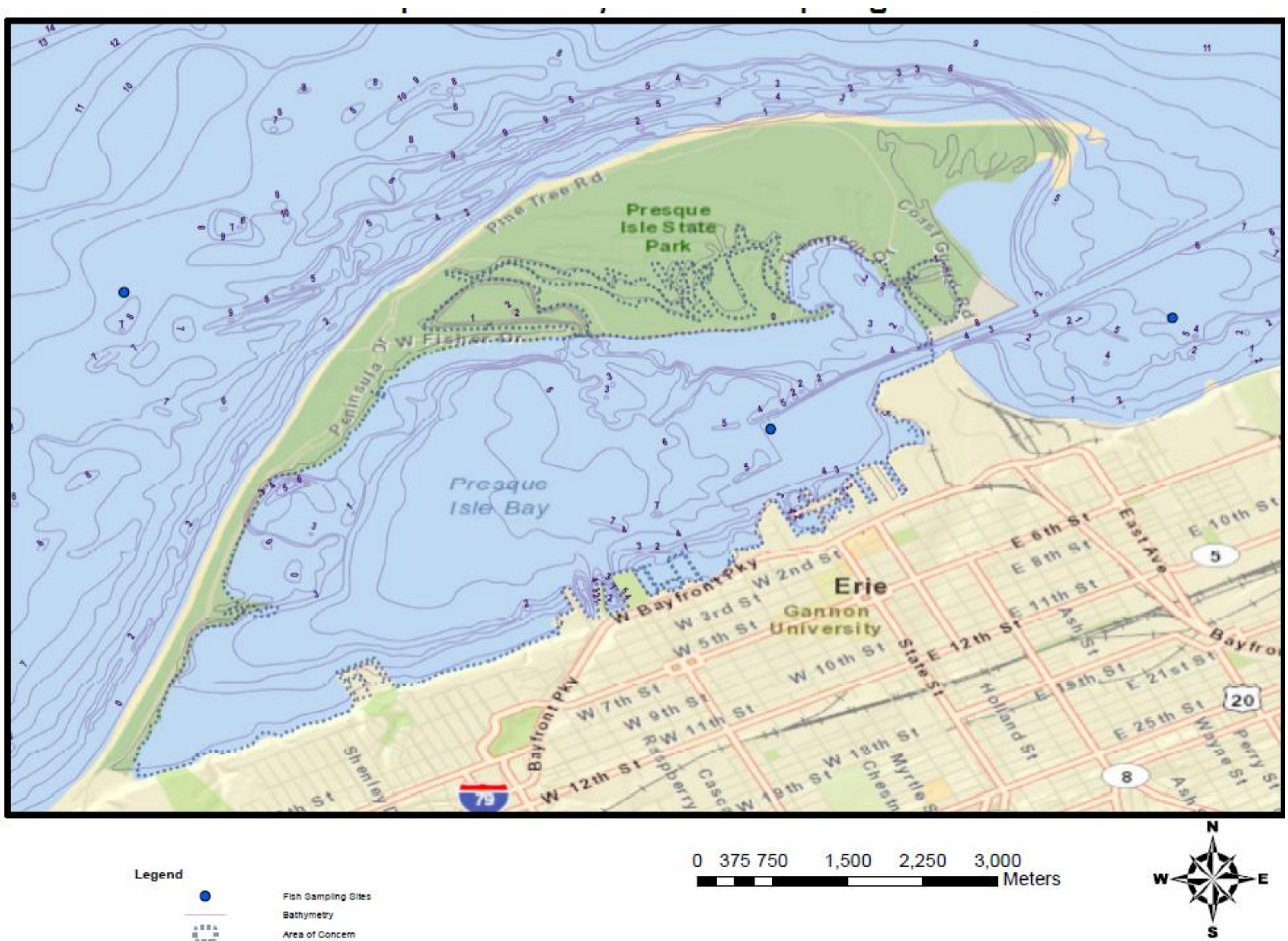


Figure 2-2: Summary of Fish Sampling Locations in Presque Isle Bay and Lake Erie

Table 2-2

Sediment Constituents/Parameters Not Included in Current HHRA

Alkyl PAHs	Other
C1-Chrysenes (methyl)	Acid volatile sulfides
C1-Fluorenes (methyl)	Total organic carbon
C1-Fluoranthenes/pyrenes	Simultaneously extracted metals, total
C1-Naphthalenes (methyl)	Simultaneously extracted cadmium
C1-Phenanthrenes/anthracenes	Simultaneously extracted copper
C2-Chrysenes (ethyl)	Simultaneously extracted lead
C2-Fluorenes (ethyl)	Simultaneously extracted nickel
C2-Naphthalenes (ethyl)	Simultaneously extracted zinc
C2-Phenanthrenes/anthracenes	Solids, percent (measured with total metals)
C3-Chrysenes (propyl)	Solids, percent (measured with PAH)
C3-Fluorenes (propyl)	Solids, percent (measured with SE metals)
C3-Naphthalenes (propyl)	Total organic carbon
C3-Phenanthrenes/anthracenes	
C4-Chrysenes (butyl)	
C4-Naphthalenes (butyl)	
C4-Phenanthrenes/anthracenes	

Table 2-3 Bluegill Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2004	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	290	290	n/a	2004	N	No RSL
Copper	µg/kg	n/a	5,400	N	601	601	n/a	2004	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	47	47	n/a	2004	N	No RSL
Mercury	µg/kg	130	140	N	61	61	n/a	2004	N	< RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 19	< 19	U	2004	N	< RSL
Aroclor 1232	µg/kg	60	1,600	C	< 19	< 19	U	2004	N	< RSL
Aroclor 1242	µg/kg	60	1,600	C	< 19	< 19	U	2004	N	< RSL
Aroclor 1248	µg/kg	60	1,600	C	< 19	< 19	U	2004	N	< RSL
Aroclor 1254	µg/kg	60	1,600	C	< 19	< 19	U	2004	N	< RSL
Aroclor 1260	µg/kg	60	1,600	C	42	42	n/a	2004	N	< RSL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 3.86	< 3.86	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.86	< 3.86	U	2004	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.86	< 3.86	U	2004	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 3.86	< 3.86	U	2004	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 3.86	< 3.86	U	2004	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 3.86	< 3.86	U	2004	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.86	< 3.86	U	2004	N	< MDL
p,p'-DDD	µg/kg	n/a	13	C	< 3.86	< 3.86	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.86	< 3.86	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.86	< 3.86	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.86	< 3.86	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.86	< 3.86	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.86	< 3.86	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.86	< 3.86	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.86	< 3.86	U	2004	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL

Table 2-4 **Brown Bullhead Tissue Sampling Data and COPC Selection Summary**										
			Screening Criteria EPA Region III RSLs		**Contaminant range[2]**				**Selection of COPCs**	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2005	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	191	191	n/a	2005	N	No RSL
Copper	µg/kg	n/a	5,400	N	531	531	n/a	2005	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2005	N	No RSL
Mercury	µg/kg	130	140	N	83	83	n/a	2005	N	< RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2005	Y	>RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4	< 4	U	2005	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4	< 4	U	2005	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4	< 4	U	2005	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4	< 4	U	2005	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4	< 4	U	2005	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4	< 4	U	2005	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 4	< 4	U	2005	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	5.83	5.83	P	2005	N	< RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4	< 4	U	2005	N	< MDL
p,p'-DDD	µg/kg	n/a	13	C	< 4	< 4	U	2005	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4	< 4	U	2005	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4	< 4	U	2005	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4	< 4	U	2005	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4	< 4	U	2005	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4	< 4	U	2005	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4	< 4	U	2005	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4	< 4	U	2005	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4	< 4	U	2005	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4	< 4	U	2005	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4	< 4	U	2005	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4	< 4	U	2005	N	< MDL

Table 2-5 Burbot Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range[(2)]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[(1)]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2007	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2008	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	183	183	n/a	2007	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	161	161	n/a	2008	N	No RSL
Copper	µg/kg	n/a	5,400	N	1,186	1,186	n/a	2007	N	< RSL
Copper	µg/kg	n/a	5,400	N	704	704	n/a	2008	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	53	53	n/a	2007	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	58	58	n/a	2008	N	No RSL
Mercury	µg/kg	130	140	N	143	143	n/a	2007	Y	> RSL
Mercury	µg/kg	130	140	N	142	142	n/a	2008	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2007	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2008	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 20	< 20	U	2008	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 20	< 20	U	2007	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 20	< 20	U	2008	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 20	< 20	U	2008	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 20	< 20	U	2008	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	74	74	P	2008	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	< 20	< 20	U	2008	N	< MDL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.32	< 4.32	U	2007	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.94	< 3.94	U	2008	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.32	< 4.32	U	2007	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.94	< 3.94	U	2008	N	< MDL

Table 2-5 (cont.)

Burbot Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
Chlordane, alpha-	µg/kg	300	9	C	< 4.32	< 4.32	U	2007	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 3.94	< 3.94	U	2008	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.32	< 4.32	U	2007	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.94	< 3.94	U	2008	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.32	< 4.32	U	2007	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.94	< 3.94	U	2008	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.32	< 4.32	U	2007	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 3.94	< 3.94	U	2008	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.32	< 4.32	U	2007	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 3.94	< 3.94	U	2008	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.32	< 4.32	U	2007	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.94	< 3.94	U	2008	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.32	< 4.32	U	2007	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 3.94	< 3.94	U	2008	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.32	< 4.32	U	2007	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.94	< 3.94	U	2008	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.32	< 4.32	U	2007	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.94	< 3.94	U	2008	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.32	< 4.32	U	2007	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.94	< 3.94	U	2008	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.32	< 4.32	U	2007	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.94	< 3.94	U	2008	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.32	< 4.32	U	2007	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.94	< 3.94	U	2008	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.32	< 4.32	U	2007	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.94	< 3.94	U	2008	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.32	< 4.32	U	2007	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.94	< 3.94	U	2008	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.32	< 4.32	U	2007	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.94	< 3.94	U	2008	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.32	< 4.32	U	2007	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.94	< 3.94	U	2008	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.32	< 4.32	U	2007	N	< MDL

Table 2-5 (cont.)
Burbot Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Nonachlor, trans-	µg/kg	300	9	C	< 3.94	< 3.94	U	2008	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.32	< 4.32	U	2007	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.94	< 3.94	U	2008	N	< MDL

Table 2-6
Channel Catfish Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	6	6	n/a	2004	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2005	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	121	121	n/a	2004	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	223	223	n/a	2005	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	288	288	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	574	574	n/a	2004	N	< RSL
Copper	µg/kg	n/a	5,400	N	322	322	n/a	2005	N	< RSL
Copper	µg/kg	n/a	5,400	N	284	284	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	87	87	n/a	2004	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2005	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2010	N	No RSL
Mercury	µg/kg	130	140	N	188	188	n/a	2004	Y	> RSL
Mercury	µg/kg	130	140	N	166	166	n/a	2005	Y	> RSL
Mercury	µg/kg	130	140	N	185	185	n/a	2010	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2005	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2010	Y	> RSL
Strontium	µg/kg	n/a	81,000	N	250,000	250,000	n/a	2010	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 30	< 30	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 30	< 30	U	2010	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 30	< 30	U	2010	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 30	< 30	U	2010	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL

Table 2-6 (cont.)
Channel Catfish Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
Aroclor 1254	µg/kg	60	1,600	C	< 30	< 30	U	2010	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	760	760	n/a	2004	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	200	200	n/a	2005	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	920	920	n/a	2010	Y	> AAL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 3.18	< 3.18	U	2004	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.50	< 3.50	U	2005	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.96	< 3.96	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.18	< 3.18	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.50	< 3.50	U	2005	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.96	< 3.96	U	2010	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	5.73	5.73	n/a	2004	N	< RSL
Chlordane, alpha-	µg/kg	300	9	C	19.24	19.24	n/a	2005	Y	> RSL
Chlordane, alpha-	µg/kg	300	9	C	7.90	7.90	n/a	2010	N	< RSL
Chlordane, gamma-	µg/kg	300	9	C	< 3.18	< 3.18	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.50	< 3.50	U	2005	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.96	< 3.96	U	2010	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.18	< 3.18	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.50	< 3.50	U	2005	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.96	< 3.96	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	22.83	22.83	PQI	2004	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	< 3.50	< 3.50	U	2005	N	< MDL
Dieldrin	µg/kg	300	0.2	C	13.573	13.573	PQ	2010	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	49.38	49.83	n/a	2005	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	18.60	18.60	PQ	2010	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	56.76	56.76	PZ	2004	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	140.04	140.04	n/a	2005	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	44.59	44.59	n/a	2010	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	57.43	57.43	PQX	2004	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.50	< 3.50	U	2005	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	31.33	31.33	P	2010	Y	> RSL
o,p'-DDD	µg/kg	n/a	13	C	< 3.18	< 3.18	U	2004	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 3.50	< 3.50	U	2005	N	< MDL

Table 2-6 (cont.) Channel Catfish Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
o,p'-DDD	µg/kg	n/a	13	C	< 3.96	< 3.96	U	2010	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.18	< 3.18	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.50	< 3.50	U	2005	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.96	< 3.96	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.18	< 3.18	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.50	< 3.50	U	2005	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.96	< 3.96	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.18	< 3.18	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	12.07	12.07	PX	2005	N	< RSL
Endrin	µg/kg	n/a	41	N	< 3.96	< 3.96	U	2010	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.18	< 3.18	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.50	< 3.50	U	2005	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.96	< 3.96	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.18	< 3.18	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	12.82	12.82	n/a	2005	Y	> RSL
Heptachlor	µg/kg	300	0.7	C	< 3.96	< 3.96	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.18	< 3.18	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.50	< 3.50	U	2005	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.96	< 3.96	U	2010	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.18	< 3.18	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.50	< 3.50	U	2005	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.96	< 3.96	U	2010	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.18	< 3.18	U	2004	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.50	< 3.50	U	2005	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.96	< 3.96	U	2010	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.18	< 3.18	U	2004	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.50	< 3.50	U	2005	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	6.60	6.60	P	2010	N	< RSL
Nonachlor, trans-	µg/kg	300	9	C	11.93	11.93	PI	2004	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	39.68	39.68	P	2005	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	13.36	13.36	Q	2010	Y	> RSL
Oxychlordane	µg/kg	300	9	C	5.06	5.06	n/a	2004	N	< RSL
Oxychlordane	µg/kg	300	9	C	< 3.50	< 3.50	U	2005	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.96	< 3.96	U	2010	N	< MDL

Table 2-7
Common Carp Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	167	167	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	556	556	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2010	N	No RSL
Mercury	µg/kg	130	140	N	93	93	n/a	2010	N	< RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2010	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 31	< 31	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 31	< 31	U	2010	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 31	< 31	U	2010	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 31	< 31	U	2010	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 31	< 31	U	2010	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	470	470	n/a	2010	Y	> AAL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.16	< 4.16	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.16	< 4.16	U	2010	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.16	< 4.16	U	2010	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.16	< 4.16	U	2010	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.16	< 4.16	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.16	< 4.16	U	2010	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	4.57	4.57	n/a	2010	N	< RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.16	< 4.16	U	2010	N	< MDL
p,p'-DDD	µg/kg	n/a	13	C	< 4.16	< 4.16	U	2010	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.16	< 4.16	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.16	< 4.16	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.16	< 4.16	U	2010	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.16	< 4.16	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.16	< 4.16	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.16	< 4.16	U	2010	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.16	< 4.16	U	2010	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.16	< 4.16	U	2010	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.16	< 4.16	U	2010	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.16	< 4.16	U	2010	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.16	< 4.16	U	2010	N	< MDL

Table 2-8
Lake Trout Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2006	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2007	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2008	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	100	100	n/a	2006	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	373	373	n/a	2007	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	292	292	n/a	2008	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	177	177	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	493	1,046	n/a	2006	N	< RSL
Copper	µg/kg	n/a	5,400	N	1,445	1,445	n/a	2007	N	< RSL
Copper	µg/kg	n/a	5,400	N	547	547	n/a	2008	N	< RSL
Copper	µg/kg	n/a	5,400	N	775	775	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	46	n/a	2006	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	97	97	n/a	2007	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2008	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	41	41	n/a	2010	N	No RSL
Mercury	µg/kg	130	140	N	123	185	n/a	2006	Y	> RSL
Mercury	µg/kg	130	140	N	146	146	n/a	2007	Y	> RSL
Mercury	µg/kg	130	140	N	181	181	n/a	2008	Y	> RSL
Mercury	µg/kg	130	140	N	171	171	n/a	2010	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2006	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2007	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2008	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2010	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 31	< 31	U	2006	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 37	< 37	U	2007	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 34	< 34	U	2008	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 34	< 34	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 31	< 31	U	2006	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 37	< 37	U	2007	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 34	< 34	U	2008	N	< MDL

Table 2-8
Lake Trout Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Aroclor 1232	µg/kg	60	1,600	C	< 16	< 16	U	2010	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 31	< 31	U	2004	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 37	< 37	U	2006	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 34	< 34	U	2007	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 34	< 34	U	2008	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 16	< 16	U	2010	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 31	< 31	U	2004	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 37	< 37	U	2006	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 34	< 34	U	2007	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 34	< 34	U	2008	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 16	< 16	U	2010	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 16	< 16	U	2004	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 31	< 36	U	2006	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 37	< 37	U	2007	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 37	< 37	U	2008	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 34	< 34	U	2010	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	1,200	1,200	n/a	2004	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	430	1,400	P	2006	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	380	380	n/a	2007	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	460	460	n/a	2008	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	1,700	1,700	n/a	2010	Y	> RSL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 3.13	< 3.13	U	2004	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.16	< 4.76	U	2006	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.96	< 4.96	U	2007	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.93	< 4.93	U	2008	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.59	< 4.59	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.13	< 3.13	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	39.90	45.94	Q	2006	Y	> RSL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.96	< 4.96	U	2007	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.93	< 4.93	U	2008	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.59	< 4.59	U	2010	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	15.23	15.23	n/a	2004	Y	> RSL

Table 2-8 (cont.)
Lake Trout Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
Chlordane, alpha-	µg/kg	300	9	C	34.23	50.20	P	2006	Y	> RSL
Chlordane, alpha-	µg/kg	300	9	C	10.01	10.01	P	2007	Y	> RSL
Chlordane, alpha-	µg/kg	300	9	C	6.49	6.49	P	2008	N	< RSL
Chlordane, alpha-	µg/kg	300	9	C	17.53	17.53	n/a	2010	Y	> RSL
Chlordane, gamma-	µg/kg	300	9	C	< 3.13	< 3.13	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.16	< 4.76	U	2007	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.96	< 4.96	U	2008	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.93	< 4.93	U	2010	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	9.31	9.31	n/a	2010	Y	> RSL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.13	< 3.13	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.16	< 4.76	U	2006	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.96	< 4.96	U	2007	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.93	< 4.93	U	2008	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.59	< 4.59	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	60.87	60.87	P	2004	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	45.96	80.90	PQL	2006	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	26.18	26.18	P	2007	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	15.48	15.48	P	2008	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	48.24	48.24	PQ	2010	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	61.50	124.74	P	2006	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	33.70	33.70	P	2007	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	24.33	24.33	P	2008	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	79.50	79.50	n/a	2010	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	188.60	188.60	PZ	2004	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	153.46	366.92	P	2006	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	53.95	53.95	n/a	2007	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	32.79	32.79	n/a	2008	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	187.83	187.83	n/a	2010	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	117.30	117.30	PQX	2004	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	114.65	114.65	PI	2006	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	47.03	47.03	P	2007	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.49	< 4.49	U	2008	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	81.96	81.96	P	2010	Y	> RSL
o,p'-DDD	µg/kg	n/a	13	C	< 3.13	< 3.13	U	2004	N	< MDL

Table 2-8 (cont.)
Lake Trout Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
o,p'-DDD	µg/kg	n/a	13	C	< 4.16	< 4.16	U	2006	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	19.80	19.80	P	2007	Y	> RSL
o,p'-DDD	µg/kg	n/a	13	C	< 4.49	< 4.49	U	2008	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	51.08	51.08	P	2010	Y	> RSL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.13	< 3.13	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.16	< 4.16	U	2006	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.96	< 4.96	U	2007	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.93	< 4.93	U	2008	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.59	< 4.59	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.13	< 3.13	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.16	< 4.16	U	2006	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.96	< 4.96	U	2007	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.93	< 4.93	U	2008	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.59	< 4.59	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.13	< 3.13	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.16	25.02	PI	2006	N	< RSL
Endrin	µg/kg	n/a	41	N	< 4.96	< 4.96	U	2007	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.93	< 4.93	U	2008	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.59	< 4.59	U	2010	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.13	< 3.13	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.16	25.02	U	2006	N	< RSL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.96	< 4.96	U	2007	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.93	< 4.93	U	2008	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.59	< 4.59	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.13	53.79	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.16	25.02	P	2006	Y	> RSL
Heptachlor	µg/kg	300	0.7	C	< 4.96	< 4.96	U	2007	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.93	< 4.93	U	2008	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.59	< 4.59	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.13	< 3.13	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.16	8.70	P	2006	Y	> RSL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.96	< 4.96	U	2007	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.93	< 4.93	U	2008	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.59	< 4.59	U	2010	N	< MDL

Table 2-8 (cont.)
Lake Trout Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Methoxychlor	µg/kg	n/a	680	N	< 3.13	< 3.13	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.16	< 4.16	U	2006	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.96	< 4.96	U	2007	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.93	< 4.93	U	2008	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.59	< 4.59	U	2010	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.13	< 3.13	U	2004	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.16	< 4.16	U	2006	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.96	< 4.96	U	2007	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.93	< 4.93	U	2008	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.59	< 4.59	U	2010	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.13	< 3.13	U	2004	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.16	< 4.16	U	2006	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.96	< 4.96	U	2007	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.93	< 4.93	U	2008	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	22.09	22.09	P	2010	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	45.10	45.10	PI	2004	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	42.81	90.01	P	2006	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	12.50	12.50	P	2007	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	10.26	10.26	P	2008	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	39.93	39.93	n/a	2010	Y	> RSL
Oxychlordane	µg/kg	300	9	C	11.14	11.14	n/a	2004	Y	> RSL
Oxychlordane	µg/kg	300	9	C	< 4.16	< 4.16	U	2006	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.96	< 4.96	U	2007	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.93	< 4.93	U	2008	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.59	< 4.59	U	2010	N	< MDL

Table 2-9
Largemouth Bass Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2005	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2006	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	278	278	n/a	2005	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	200	200	n/a	2006	N	No RSL
Copper	µg/kg	n/a	5,400	N	249	249	n/a	2005	N	< RSL
Copper	µg/kg	n/a	5,400	N	208	208	n/a	2006	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2005	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2006	N	No RSL
Mercury	µg/kg	130	140	N	195	195	n/a	2005	Y	> RSL
Mercury	µg/kg	130	140	N	179	179	n/a	2006	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2005	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2006	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 23	< 23	U	2006	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 23	< 23	U	2006	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 23	< 23	U	2006	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 23	< 23	U	2006	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 23	< 23	U	2006	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	72	72	n/a	2006	Y	> AAL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.42	< 4.42	U	2005	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.52	< 4.52	U	2006	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.42	< 4.42	U	2005	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.52	< 4.52	U	2006	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.42	< 4.42	U	2005	N	< RSL
Chlordane, alpha-	µg/kg	300	9	C	< 4.52	< 4.52	U	2006	N	< RSL

Table 2-9 (cont.) Largemouth Bass Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Chlordane, gamma-	µg/kg	300	9	C	< 4.42	< 4.42	U	2005	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.52	< 4.52	U	2006	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.42	< 4.42	U	2005	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.52	< 4.52	U	2006	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.42	< 4.42	U	2005	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.52	< 4.52	U	2006	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 4.52	< 4.52	U	2006	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.42	< 4.42	U	2005	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	16.01	16.01	n/a	2006	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.42	< 4.42	U	2005	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.52	< 4.52	U	2006	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.42	< 4.42	U	2005	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.52	< 4.52	U	2006	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.42	< 4.42	U	2005	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.52	< 4.52	U	2006	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.42	< 4.42	U	2005	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.52	< 4.52	U	2006	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.42	< 4.42	U	2005	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.52	< 4.52	U	2006	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.42	< 4.42	U	2005	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.52	< 4.52	U	2006	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.42	< 4.42	U	2005	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.52	< 4.52	U	2006	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.42	< 4.42	U	2005	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.52	< 4.52	U	2006	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.42	< 4.42	U	2005	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.52	< 4.52	U	2006	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.42	< 4.42	U	2005	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.52	< 4.52	U	2006	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.42	< 4.42	U	2005	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.52	< 4.52	U	2006	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.42	< 4.42	U	2005	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.52	< 4.52	U	2006	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.42	< 4.42	U	2005	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.52	< 4.52	U	2006	N	< MDL

Table 2-10 Northern Pike Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	234	234	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	219	219	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2010	N	No RSL
Mercury	µg/kg	130	140	N	124	124	n/a	2010	N	< RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2010	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 21	< 21	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 21	< 21	U	2010	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 21	< 21	U	2010	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 21	< 21	U	2010	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 21	< 21	U	2010	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	< 21	< 21	U	2010	N	< MDL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.26	< 4.26	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.26	< 4.26	U	2010	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.26	< 4.26	U	2010	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.26	< 4.26	U	2010	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.26	< 4.26	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	6.48	6.48	P	2010	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	22.08	22.08	n/a	2010	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.26	< 4.26	U	2010	N	< MDL
p,p'-DDD	µg/kg	n/a	13	C	< 4.26	< 4.26	U	2010	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.26	< 4.26	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.26	< 4.26	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.26	< 4.26	U	2010	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.26	< 4.26	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.26	< 4.26	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.26	< 4.26	U	2010	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.26	< 4.26	U	2010	N	< MDL

Table 2-10
Northern Pike Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Nonachlor, cis-	µg/kg	300	9	C	< 4.26	< 4.26	U	2010	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.26	< 4.26	U	2010	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.26	< 4.26	U	2010	N	< MDL

Table 2-11 Pumpkinseed Fish Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	6	6	n/a	2004	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	345	345	n/a	2004	N	No RSL
Copper	µg/kg	n/a	5,400	N	462	462	n/a	2004	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2004	N	No RSL
Mercury	µg/kg	130	140	N	51	51	n/a	2004	N	< RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 20	< 20	U	2004	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 20	< 20	U	2004	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 20	< 20	U	2004	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 20	< 20	U	2004	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 20	< 20	U	2004	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	< 20	< 20	U	2004	N	< MDL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4	< 4	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4	< 4	U	2004	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4	< 4	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4	< 4	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4	< 4	U	2004	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4	< 4	U	2004	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4	< 4	n/a	2004	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4	< 4	U	2004	N	< MDL
p,p'-DDD	µg/kg	n/a	13	C	< 4	< 4	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4	< 4	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4	< 4	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4	< 4	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4	< 4	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4	< 4	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4	< 4	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	4	4	U	2004	N	< RSL
Nonachlor, cis-	µg/kg	300	9	C	< 4	< 4	U	2004	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4	< 4	U	2004	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4	< 4	U	2004	N	< MDL

Table 2-12
Smallmouth Bass Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2004	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2005	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2006	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2007	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2008	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	277	277	n/a	2004	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	191	191	n/a	2005	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	117	117	n/a	2006	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	312	312	n/a	2007	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	210	210	n/a	2008	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	231	231	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	948	948	n/a	2004	N	< RSL
Copper	µg/kg	n/a	5,400	N	417	417	n/a	2005	N	< RSL
Copper	µg/kg	n/a	5,400	N	584	584	n/a	2006	N	< RSL
Copper	µg/kg	n/a	5,400	N	1,353	1,353	n/a	2007	N	< RSL
Copper	µg/kg	n/a	5,400	N	524	524	n/a	2008	N	< RSL
Copper	µg/kg	n/a	5,400	N	290	290	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	87	87	n/a	2004	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2005	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2006	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	62	62	n/a	2007	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2007	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2007	N	No RSL
Mercury	µg/kg	130	140	N	294	294	n/a	2008	Y	> RSL
Mercury	µg/kg	130	140	N	248	248	n/a	2010	Y	> RSL
Mercury	µg/kg	130	140	N	237	237	n/a	2004	Y	> RSL
Mercury	µg/kg	130	140	N	350	350	n/a	2005	Y	> RSL
Mercury	µg/kg	130	140	N	248	248	n/a	2005	Y	> RSL
Mercury	µg/kg	130	140	N	277	277	n/a	2005	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,148	1,148	n/a	2005	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,086	1,086	n/a	2006	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,585	1,585	n/a	2007	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2008	Y	> RSL

Table 2-12 (Cont.)
Smallmouth Bass Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
PCBs										
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2010	Y	> RSL
Aroclor 1221	µg/kg	60	1,600	C	< 25	< 25	U	2004	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 52	< 52	U	2006	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 26	< 26	U	2007	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 27	< 27	U	2008	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 28	< 28	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 25	< 25	U	2004	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 26	< 26	U	2006	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 26	< 26	U	2007	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 27	< 27	U	2008	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 28	< 28	U	2008	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 25	< 25	U	2004	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 26	< 26	U	2006	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 26	< 26	U	2007	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 27	< 27	U	2008	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 28	< 28	U	2008	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 25	< 25	U	2004	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 26	< 26	U	2006	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 26	< 26	U	2007	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 27	< 27	U	2008	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 28	< 28	U	2008	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 25	< 25	U	2004	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 25	< 25	U	2005	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 26	< 26	U	2006	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	180	180	P	2007	Y	> AAL
Aroclor 1254	µg/kg	60	1,600	C	< 27	< 27	U	2008	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 28	< 28	U	2008	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	150	150	n/a	2004	Y	> AAL

Table 2-12 (Cont.)
Smallmouth Bass Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Aroclor 1260	µg/kg	60	1,600	C	< 25	< 25	U	2005	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	240	240	n/a	2006	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	470	470	n/a	2007	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	500	500	PQ	2008	Y	> AAL
Aroclor 1260	µg/kg	60	1,600	C	1,200	1,200	n/a	2010	Y	> AAL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.86	< 4.86	U	2004	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 2.86	< 2.86	U	2005	N	< MDL
Aldrin	µg/kg	300	0.19	C	8.60	8.60	n/a	2006	Y	> RSL
Aldrin	µg/kg	300	0.19	C	< 3.49	< 3.49	U	2007	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.70	< 3.70	U	2008	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.71	< 3.71	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.86	< 4.86	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	28.10	28.10	n/a	2005	Y	> RSL
BHC, alpha-	µg/kg	n/a	0.5	C	17.72	17.72	PQ	2006	Y	> RSL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.49	< 3.49	U	2007	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.70	< 3.70	U	2008	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 3.71	< 3.71	U	2010	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.86	< 4.86	U	2004	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 2.65	< 2.65	U	2005	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 3.45	< 3.45	U	2006	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 3.49	< 3.49	U	2007	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	7.50	7.50	P	2008	N	< RSL
Chlordane, alpha-	µg/kg	300	9	C	< 3.71	< 3.71	U	2010	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.86	< 4.86	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 2.65	< 2.65	U	2005	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.45	< 3.45	U	2006	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.49	< 3.49	U	2007	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.70	< 3.70	U	2008	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.71	< 3.71	U	2010	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.86	< 4.86	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 2.65	< 2.65	U	2005	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.45	< 3.45	U	2006	N	< MDL

Table 2-12 (cont.) Smallmouth Bass Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
Chlordene	µg/kg	n/a	n/a	n/a	< 3.49	< 3.49	U	2007	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.70	< 3.70	U	2008	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.71	< 3.71	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	9.35	9.35	P	2004	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	< 2.65	< 2.65	U	2005	N	< MDL
Dieldrin	µg/kg	300	0.2	C	15.18	15.18	PQI	2006	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	< 3.50	< 3.50	U	2007	N	< MDL
Dieldrin	µg/kg	300	0.2	C	13.35	13.35	n/a	2008	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	13.72	13.72	n/a	2010	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	23.94	23.94	PQ	2005	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	12.02	12.02	PQ	2006	N	< RSL
4,4'-DDD	µg/kg	n/a	13	C	9.22	9.22	P	2007	N	< RSL
4,4'-DDD	µg/kg	n/a	13	C	24.17	24.17	P	2008	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	13.35	13.35	P	2010	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	31.08	31.08	P	2004	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	68.40	68.40	n/a	2005	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	60.57	60.57	n/a	2006	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	29.02	29.02	P	2007	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	33.81	33.81	PQ	2008	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	58.15	58.15	P	2010	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	34.18	34.18	PI	2004	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	2.65	2.65	n/a	2005	N	< RSL
4,4'-DDT	µg/kg	n/a	9.3	C	53.50	53.50	PI	2006	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	34.12	34.12	P	2007	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.70	< 3.70	U	2008	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.71	< 3.71	U	2010	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.86	< 4.86	U	2004	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 2.65	< 2.65	U	2005	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 3.45	< 3.45	U	2006	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	10.39	10.39	P	2007	N	< RSL
o,p'-DDD	µg/kg	n/a	13	C	13.87	13.87	P	2008	Y	> RSL
o,p'-DDD	µg/kg	n/a	13	C	19.05	19.05	P	2010	Y	> RSL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.86	< 4.86	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 2.65	< 2.65	U	2005	N	< MDL

Table 2-12 (cont.) Smallmouth Bass Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.45	< 3.45	U	2006	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.49	< 3.49	U	2007	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.70	< 3.70	U	2008	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.71	< 3.71	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.86	< 4.86	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 2.65	< 2.65	U	2005	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.45	< 3.45	U	2006	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.49	< 3.49	U	2007	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.70	< 3.70	U	2008	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.71	< 3.71	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.86	< 4.86	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	7.68	7.68	PX	2005	N	< RSL
Endrin	µg/kg	n/a	41	N	< 3.45	< 3.45	U	2006	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.49	< 3.49	U	2007	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.70	< 3.70	U	2008	N	< MDL
Endrin	µg/kg	n/a	41	N	7.38	7.38	n/a	2010	N	< RSL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.86	< 4.86	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 2.65	< 2.65	U	2005	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.45	< 3.45	U	2006	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.49	< 3.49	U	2007	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.70	< 3.70	U	2008	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.71	< 3.71	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.86	< 4.86	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	11.65	11.65	P	2005	Y	> RSL
Heptachlor	µg/kg	300	0.7	C	< 3.45	< 3.45	U	2006	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.49	< 3.49	U	2007	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.70	< 3.70	U	2008	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.71	< 3.71	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.86	< 4.86	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	9.60	9.60	P	2005	Y	> RSL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.45	< 3.45	U	2006	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.49	< 3.49	U	2007	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.70	< 3.70	U	2008	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.71	< 3.71	U	2010	N	< MDL

Table 2-12 (cont.)
Smallmouth Bass Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Methoxychlor	µg/kg	n/a	680	N	< 4.86	< 4.86	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 2.65	< 2.65	U	2005	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.45	< 3.45	U	2006	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.49	< 3.49	U	2007	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.70	< 3.70	U	2008	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.71	< 3.71	U	2010	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.86	< 4.86	U	2004	N	< MDL
Mirex	µg/kg	100	0.18	C	< 2.65	< 2.65	U	2005	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.45	< 3.45	U	2006	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.49	< 3.49	U	2007	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.70	< 3.70	U	2008	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.71	< 3.71	U	2010	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.86	< 4.86	U	2004	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 2.65	< 2.65	U	2005	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.45	< 3.45	U	2006	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.49	< 3.49	U	2007	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.70	< 3.70	U	2008	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	5.22	5.22	n/a	2010	N	< RSL
Nonachlor, trans-	µg/kg	300	9	C	8.63	8.63	PI	2004	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	14.80	14.80	n/a	2005	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	< 3.45	< 3.45	U	2006	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	5.31	5.31	n/a	2007	N	< RSL
Nonachlor, trans-	µg/kg	300	9	C	12.10	12.10	P	2008	Y	> RSL
Nonachlor, trans-	µg/kg	300	9	C	13.20	13.20	P	2010	Y	> RSL
Oxychlordane	µg/kg	300	9	C	< 4.86	< 4.86	U	2004	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 2.65	< 2.65	U	2005	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.45	< 3.45	U	2006	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.49	< 3.49	U	2007	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.70	< 3.70	U	2008	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.71	< 3.71	U	2010	N	< MDL

Table 2-13 Walleye Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2007	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	7	7	n/a	2008	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	220	220	n/a	2007	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	152	152	n/a	2008	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	151	151	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	809	809	n/a	2007	N	< RSL
Copper	µg/kg	n/a	5,400	N	513	513	n/a	2008	N	< RSL
Copper	µg/kg	n/a	5,400	N	383	383	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	40	40	n/a	2007	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	26	26	n/a	2008	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	46	46	n/a	2010	N	No RSL
Mercury	µg/kg	130	140	N	156	156	n/a	2007	Y	> RSL
Mercury	µg/kg	130	140	N	261	261	n/a	2008	Y	> RSL
Mercury	µg/kg	130	140	N	238	238	n/a	2010	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2007	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2008	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 23	< 23	U	2007	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 25	< 25	U	2008	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 26	< 26	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 23	< 23	U	2007	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 25	< 25	U	2008	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 26	< 26	U	2010	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 23	< 23	U	2007	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 25	< 25	U	2008	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 26	< 26	U	2010	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 23	< 23	U	2007	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 25	< 25	U	2008	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 26	< 26	U	2010	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	56	56	P	2007	Y	> RSL
Aroclor 1254	µg/kg	60	1,600	C	< 25	< 25	U	2008	N	< MDL

Table 2-13 (cont.)

Walleye Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[(2)]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[(1)]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
Aroclor 1254	µg/kg	60	1,600	C	< 26	< 26	U	2010	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	97	97	P	2007	Y	> RSL
Aroclor 1260	µg/kg	60	1,600	C	270	270	n/a	2008	Y	> RSL
Aroclor 1260	µg/kg	60	1,600	C	160	160	n/a	2010	Y	> RSL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.62	< 4.62	U	2007	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.37	< 3.37	U	2008	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 3.47	< 3.47	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.62	< 4.62	U	2007	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	6.55	6.55	P	2008	Y	> RSL
BHC, alpha-	µg/kg	n/a	0.5	C	5.80	5.80	P	2010	Y	> RSL
Chlordane, alpha-	µg/kg	300	9	C	< 4.62	< 4.62	U	2007	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 3.37	< 3.37	U	2008	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 3.47	< 3.47	U	2010	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.62	< 4.62	U	2007	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.37	< 3.37	U	2008	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 3.47	< 3.47	U	2010	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.62	< 4.62	U	2007	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.37	< 3.37	U	2008	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 3.47	< 3.47	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	8.13	8.13	n/a	2007	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	9.26	9.26	n/a	2008	Y	> RSL
Dieldrin	µg/kg	300	0.2	C	4.78	4.78	n/a	2010	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	< 3.37	< 3.37	U	2008	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	3.58	3.58	N	2010	N	< RSL
4,4'-DDE	µg/kg	n/a	9.3	C	24.32	24.32	P	2007	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	29.02	29.02	P	2008	Y	> RSL
4,4'-DDE	µg/kg	n/a	9.3	C	15.14	15.14	n/a	2010	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2007	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.37	< 3.37	U	2008	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 3.47	< 3.47	U	2010	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.62	< 4.62	U	2007	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 3.37	< 3.37	U	2008	N	< MDL

Table 2-13 (cont.)
Walleye Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels[1]**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
o,p'-DDD	µg/kg	n/a	13	C	< 3.47	< 3.47	U	2010	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2007	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.37	< 3.37	U	2008	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 3.47	< 3.47	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2007	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.37	< 3.37	U	2008	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 3.47	< 3.47	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.62	< 4.62	U	2007	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.37	< 3.37	U	2008	N	< MDL
Endrin	µg/kg	n/a	41	N	< 3.47	< 3.47	U	2010	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.62	< 4.62	U	2007	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.37	< 3.37	U	2008	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 3.47	< 3.47	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.62	< 4.62	U	2007	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.37	< 3.37	U	2008	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 3.47	< 3.47	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.62	< 4.62	U	2007	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.37	< 3.37	U	2008	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.47	< 3.47	U	2010	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.62	< 4.62	U	2007	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.37	< 3.37	U	2008	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.47	< 3.47	U	2010	N	< MDL
Mirex	µg/kg	100	0.18	C	9.12	9.12	P	2007	Y	> RSL
Mirex	µg/kg	100	0.18	C	< 3.37	< 3.37	U	2008	N	< MDL
Mirex	µg/kg	100	0.18	C	< 3.47	< 3.47	U	2010	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.62	< 4.62	U	2007	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.37	< 3.37	U	2008	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.47	< 3.47	U	2010	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.62	< 4.62	U	2007	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 3.37	< 3.37	U	2008	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 3.47	< 3.47	U	2010	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.62	< 4.62	U	2007	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.37	< 3.37	U	2008	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.47	< 3.47	U	2010	N	< MDL

Table 2-14
White Bass Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	7	7	n/a	2004	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	249	249	n/a	2004	N	No RSL
Copper	µg/kg	n/a	5,400	N	573	573	n/a	2004	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2004	N	No RSL
Mercury	µg/kg	130	140	N	115	115	n/a	2004	N	< AAL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 13	< 13	U	2004	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 13	< 13	U	2004	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 13	< 13	U	2004	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 13	< 13	U	2004	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 13	< 13	U	2004	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	260	260	n/a	2004	Y	> AAL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 2.63	< 2.63	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 2.63	< 2.63	U	2004	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 2.63	< 2.63	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 2.63	< 2.63	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 2.63	< 2.63	U	2004	N	< MDL
Dieldrin	µg/kg	300	0.2	C	9.17	9.17	n/a	2004	Y	> RSL
4,4'-DDD	µg/kg	n/a	13	C	20.86	20.86	n/a	2004	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	15.78	15.78	n/a	2004	Y	> RSL
p,p'-DDD	µg/kg	n/a	13	C	< 2.63	< 2.63	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 2.63	< 2.63	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 2.63	< 2.63	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	< 2.63	< 2.63	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 2.63	< 2.63	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 2.63	< 2.63	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 3.86	< 3.86	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 3.86	< 3.86	U	2004	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 3.86	< 3.86	U	2004	N	< MDL

Table 2-15
White Sucker Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	**Units**	**Advisory/ Action Levels(1)**	**Conc. Used for Screening Level**	**Basis for Screening Level**	**Minimum Detected Value**	**Maximum detected value**	**Qualifier**	**Year Sampled**	**Constituent selected as COPC**	**Reason for Inclusion/ Exclusion**
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	6	6	n/a	2007	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	178	178	n/a	2007	N	No RSL
Copper	µg/kg	n/a	5,400	N	2,031	2,031	n/a	2007	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	76	76	n/a	2007	N	No RSL
Mercury	µg/kg	130	140	N	147	147	n/a	2007	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,152	1,152	n/a	2007	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	260	260	n/a	2007	Y	> AAL
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.42	< 4.42	U	2007	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.42	< 4.42	U	2007	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.42	< 4.42	U	2007	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.42	< 4.42	U	2007	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.42	< 4.42	U	2007	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.42	< 4.42	U	2007	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	8.17	8.17	n/a	2007	N	< RSL
4,4'-DDE	µg/kg	n/a	9.3	C	27.74	27.74	n/a	2007	Y	> RSL
4,4'-DDT	µg/kg	n/a	9.3	C	18.84	18.84	P	2007	Y	> RSL
p,p'-DDD	µg/kg	n/a	13	C	< 4.42	< 4.42	U	2007	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.42	< 4.42	U	2007	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.42	< 4.42	U	2007	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.42	< 4.42	U	2007	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.42	< 4.42	U	2007	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.42	< 4.42	U	2007	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.42	< 4.42	U	2007	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.42	< 4.42	U	2007	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.42	< 4.42	U	2007	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.42	< 4.42	U	2007	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.42	< 4.42	U	2007	N	< MDL

Table 2-16
Yellow Perch Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
METALS										
Cadmium	µg/kg	n/a	n/a	n/a	12	12	n/a	2004	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	9	9	n/a	2005	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	5	5	n/a	2006	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	10	10	n/a	2007	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	8	8	n/a	2008	N	No RSL
Cadmium	µg/kg	n/a	n/a	n/a	10	10	n/a	2010	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	278	278	n/a	2004	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	236	236	n/a	2005	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	22	22	n/a	2006	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	186	186	n/a	2007	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	268	268	n/a	2008	N	No RSL
Chromium (III)	µg/kg	n/a	n/a	n/a	205	205	n/a	2010	N	No RSL
Copper	µg/kg	n/a	5,400	N	540	540	n/a	2004	N	< RSL
Copper	µg/kg	n/a	5,400	N	419	419	n/a	2005	N	< RSL
Copper	µg/kg	n/a	5,400	N	370	370	n/a	2006	N	< RSL
Copper	µg/kg	n/a	5,400	N	367	367	n/a	2007	N	< RSL
Copper	µg/kg	n/a	5,400	N	289	289	n/a	2008	N	< RSL
Copper	µg/kg	n/a	5,400	N	291	291	n/a	2010	N	< RSL
Lead	µg/kg	n/a	n/a	n/a	33	33	n/a	2004	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2005	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2006	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2007	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2008	N	No RSL
Lead	µg/kg	n/a	n/a	n/a	25	25	n/a	2010	N	No RSL
Mercury	µg/kg	130	140	N	132	132	n/a	2004	Y	> AAL
Mercury	µg/kg	130	140	N	55	55	n/a	2005	N	< RSL
Mercury	µg/kg	130	140	N	43	43	n/a	2006	N	< RSL
Mercury	µg/kg	130	140	N	69	69	n/a	2007	N	< RSL
Mercury	µg/kg	130	140	N	84	84	n/a	2008	N	< RSL
Mercury	µg/kg	130	140	N	93	93	n/a	2010	N	< RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2005	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2006	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,032	1,032	n/a	2007	Y	> RSL
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2008	Y	> RSL

Table 2-16 (Cont.)
Yellow Perch Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
Selenium	µg/kg	n/a	680	N	1,000	1,000	n/a	2010	Y	> RSL
PCBs										
Aroclor 1221	µg/kg	60	1,600	C	< 22	< 22	U	2004	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 42	< 48	U	2006	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 22	< 37	U	2007	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 23	< 34	U	2008	N	< MDL
Aroclor 1221	µg/kg	60	1,600	C	< 23	< 34	U	2010	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 22	< 22	U	2004	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 21	< 24	U	2006	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 23	< 23	U	2008	N	< MDL
Aroclor 1232	µg/kg	60	1,600	C	< 23	< 23	U	2010	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 22	< 22	U	2004	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 21	< 24	U	2006	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 23	< 23	U	2008	N	< MDL
Aroclor 1242	µg/kg	60	1,600	C	< 23	< 23	U	2010	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 22	< 22	U	2004	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 21	< 24	U	2006	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 23	< 23	U	2008	N	< MDL
Aroclor 1248	µg/kg	60	1,600	C	< 23	< 23	U	2010	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 22	< 22	U	2004	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 21	< 24	U	2006	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 22	< 22	U	2007	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 23	< 23	U	2008	N	< MDL
Aroclor 1254	µg/kg	60	1,600	C	< 23	< 23	U	2010	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	32	32	n/a	2004	N	< AAL
Aroclor 1260	µg/kg	60	1,600	C	< 21	< 21	P	2006	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	< 22	< 22	n/a	2007	N	< MDL
Aroclor 1260	µg/kg	60	1,600	C	26	26	n/a	2008	N	< AAL
Aroclor 1260	µg/kg	60	1,600	C	25	25	n/a	2010	N	< AAL

Table 2-16 (cont.)
Yellow Perch Tissue Sampling Data and COPC Selection Summary

			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
PESTICIDES										
Aldrin	µg/kg	300	0.19	C	< 4.34	< 4.34	U	2004	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.24	< 4.24	U	2006	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.50	< 4.50	U	2007	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.62	< 4.62	U	2008	N	< MDL
Aldrin	µg/kg	300	0.19	C	< 4.64	< 4.64	U	2010	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.34	< 4.34	U	2004	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.24	< 4.24	U	2006	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.50	< 4.50	U	2007	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.62	< 4.62	U	2008	N	< MDL
BHC, alpha-	µg/kg	n/a	0.5	C	< 4.64	< 4.64	U	2010	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.34	< 4.34	U	2004	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.24	< 4.24	U	2006	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.50	< 4.50	U	2007	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.62	< 4.62	U	2008	N	< MDL
Chlordane, alpha-	µg/kg	300	9	C	< 4.64	< 4.64	U	2010	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.34	< 4.34	U	2004	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.24	< 4.24	U	2006	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.50	< 4.50	U	2007	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.62	< 4.62	U	2008	N	< MDL
Chlordane, gamma-	µg/kg	300	9	C	< 4.64	< 4.64	U	2010	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.34	< 4.34	U	2004	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.24	< 4.24	U	2006	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.50	< 4.50	U	2007	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.62	< 4.62	U	2008	N	< MDL
Chlordene	µg/kg	n/a	n/a	n/a	< 4.64	< 4.64	U	2010	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.34	< 4.34	U	2004	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.24	< 4.24	U	2006	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.50	< 4.50	U	2007	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.62	< 4.62	U	2008	N	< MDL
Dieldrin	µg/kg	300	0.2	C	< 4.64	< 4.64	U	2010	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 4.34	< 4.34	U	2006	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 4.24	< 4.24	U	2007	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 4.50	< 4.50	U	2008	N	< MDL
4,4'-DDD	µg/kg	n/a	13	C	< 4.62	< 4.62	U	2010	N	< MDL

Table 2-16 (cont.) Yellow Perch Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range(2)				Selection of COPCs	
COPC	Units	Advisory/ Action Levels(1)	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.34	< 4.34	U	2004	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.24	< 4.24	U	2006	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.50	< 4.50	U	2007	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2008	N	< MDL
4,4'-DDE	µg/kg	n/a	9.3	C	< 4.64	< 4.64	U	2010	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.34	< 4.34	U	2004	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.24	< 4.24	U	2006	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.50	< 4.50	U	2007	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2008	N	< MDL
4,4'-DDT	µg/kg	n/a	9.3	C	< 4.64	< 4.64	U	2010	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.34	< 4.34	U	2004	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.24	< 4.24	U	2006	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.50	< 4.50	U	2007	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.62	< 4.62	U	2008	N	< MDL
o,p'-DDD	µg/kg	n/a	13	C	< 4.64	< 4.64	U	2010	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.34	< 4.34	U	2004	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.24	< 4.24	U	2006	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.50	< 4.50	U	2007	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2008	N	< MDL
o,p'-DDE	µg/kg	n/a	9.3	C	< 4.64	< 4.64	U	2010	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.34	< 4.34	U	2004	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.24	< 4.24	U	2006	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.50	< 4.50	U	2007	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.62	< 4.62	U	2008	N	< MDL
p,p'-DDT	µg/kg	n/a	9.3	C	< 4.64	< 4.64	U	2010	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.34	< 4.34	U	2004	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.24	< 4.24	U	2006	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.50	< 4.50	U	2007	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.62	< 4.62	U	2008	N	< MDL
Endrin	µg/kg	n/a	41	N	< 4.64	< 4.64	U	2010	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.34	< 4.34	U	2004	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.24	< 4.24	U	2006	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.50	< 4.50	U	2007	N	< MDL
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.62	< 4.62	U	2008	N	< MDL

Table 2-16 (cont.) Yellow Perch Tissue Sampling Data and COPC Selection Summary										
			Screening Criteria EPA Region III RSLs		Contaminant range[2]				Selection of COPCs	
COPC	Units	Advisory/ Action Levels[1]	Conc. Used for Screening Level	Basis for Screening Level	Minimum Detected Value	Maximum detected value	Qualifier	Year Sampled	Constituent selected as COPC	Reason for Inclusion/ Exclusion
GHC, gamma-	µg/kg	n/a	2.9	C	< 4.64	< 4.64	U	2010	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.34	< 4.34	U	2004	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.24	< 4.24	U	2006	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.50	< 4.50	U	2007	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.62	< 4.62	U	2008	N	< MDL
Heptachlor	µg/kg	300	0.7	C	< 4.64	< 4.64	U	2010	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.34	< 4.34	U	2004	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.24	< 4.24	U	2006	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.50	< 4.50	U	2007	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.62	< 4.62	U	2008	N	< MDL
Heptachlor epoxide	µg/kg	300	0.35	C	< 4.64	< 4.64	U	2010	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.34	< 4.34	U	2004	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.24	< 4.24	U	2006	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.50	< 4.50	U	2007	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.62	< 4.62	U	2008	N	< MDL
Methoxychlor	µg/kg	n/a	680	N	< 4.64	< 4.64	U	2010	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.34	< 4.34	U	2004	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.24	< 4.24	U	2006	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.50	< 4.50	U	2007	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.62	< 4.62	U	2008	N	< MDL
Mirex	µg/kg	100	0.18	C	< 4.64	< 4.64	U	2010	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.34	< 4.34	U	2004	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.24	< 4.24	U	2006	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.50	< 4.50	U	2007	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.62	< 4.62	U	2008	N	< MDL
Nonachlor, cis-	µg/kg	300	9	C	< 4.64	< 4.64	U	2010	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.34	< 4.34	U	2004	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.24	< 4.24	U	2006	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.50	< 4.50	U	2007	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.62	< 4.62	U	2008	N	< MDL
Nonachlor, trans-	µg/kg	300	9	C	< 4.64	< 4.64	U	2010	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.34	< 4.34	U	2004	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.24	< 4.24	U	2006	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.50	< 4.50	U	2007	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.62	< 4.62	U	2008	N	< MDL
Oxychlordane	µg/kg	300	9	C	< 4.64	< 4.64	U	2010	N	< MDL

Footnotes for Table 2-3 through 2-16: Fish Tissue Sampling Data:

Qualifier Definitions

U – Indicates compound was analyzed for but not detected. The sample quantitation limit is reported

P – This flag is used with a target analyte when there is greater than a 25% difference between the results obtained from the primary and confirmation columns for dual column analysis methods. (ie, pesticides, triazines, PCB's etc.). The reported value is the average of the two results.

I – Indicates an estimated value, below the quantification limit, but above the method detection limit

X - Non-target analytes co-elute with compound. Identification unable to be confirmed.

Q - This flag identifies the average of multiple results from multiple analysis, or the average of the averages of dual column analysis methods.

Z -

(1) RSL values follow USEPA levels
(2) Pesticide advisory levels follow FDA action levels; mercury advisory levels follow USEPA advisory levels
(3) Detected concentrations were of 5 – 10 composite fish tissue fillets

Table 2-17

Additional Constituents/Parameters Measured in Channel Catfish Tissue Excluded from HHRA

Radionuclides	
Americium 241	Lead 214
Barium 140	Manganese 54
Beryllium 7	Niobium 95
Cesium 134	Radium 226
Cesium 137	Radium 228
Cobalt 58	Ruthenium 103
Cobalt 60	Ruthenium 106
Iodine 131	Uranium 235
Iron 59	Uranium 238
Lanthanum 140	Zinc 65
Lead 212	Zirconium 95

Table 2-18 Summary of chemicals that are additive in Nature		
Chemical Group that are additive	**Sediment**	**Fish Tissue**
Arochlors	Not evaluated	Arochlor 1016 Arochlor 1221 Arochlor 1232 Arochlor 1242 Arochlor 1248 Arochlor 1254 Arochlor 1260
Total chlordane	Chlordane Heptachlor Heptachlor epoxide Nonachlor	Alpha chlordane Cis-Nonachlor Gamma-chlordane trans-Nonachlor Oxychlordane
PCB Congeners	PCB008 (2,4'-dichlorobiphenyl) PCB018 (2,2',5-trichlorobiphenyl) PCB028 (2,4,4'-trichlorobiphenyl) PCB044 (2,2',3,5'-tetrachlorobiphenyl) PCB052 (2,2',5,5'-tetrachlorobiphenyl) PCB066 (2,3',4,4'-tetrachlorobiphenyl) PCB087 (2,2',3,4,5'-pentachlorobiphenyl) PCB101 (2,2',4,5,5'-pentachlorobiphenyl) PCB105 (2,3,3',4,4'-pentachlorobiphenyl) PCB118 (2,3',4,4',5-pentachlorobiphenyl) PCB128 (2,2',3,3',4,4'-hexachlorobiphenyl) PCB138 (2,2',3,4,4',5'-hexachlorobiphenyl) PCB153 (2,2',4,4',5,5'-hexachlorobiphenyl) PCB170 (2,2',3,3',4,4',5-heptachlorobiphenyl) PCB180 (2,2',3,4,4',5,5'-heptachlorobiphenyl) PCB187 (2,2',3,4',5,5',6-heptachlorobiphenyl) PCB195 (2,2',3,3',4,4',5,6-octachlorobiphenyl) PCB206(2,2',3,3',4,4',5,5',6-nonachlorobiphenyl) PCB209 (decachlorobiphenyl)	Not evaluated
DDT and derivatives	o,p'-DDD o,p'-DDE o,p'-DDT p,p'-DDD p,p'-DDE p,p'-DDT	4,4'-DDD 4,4'-DDE 4,4'-DDT O,P-DDD O,P-DDE O,P-DDT

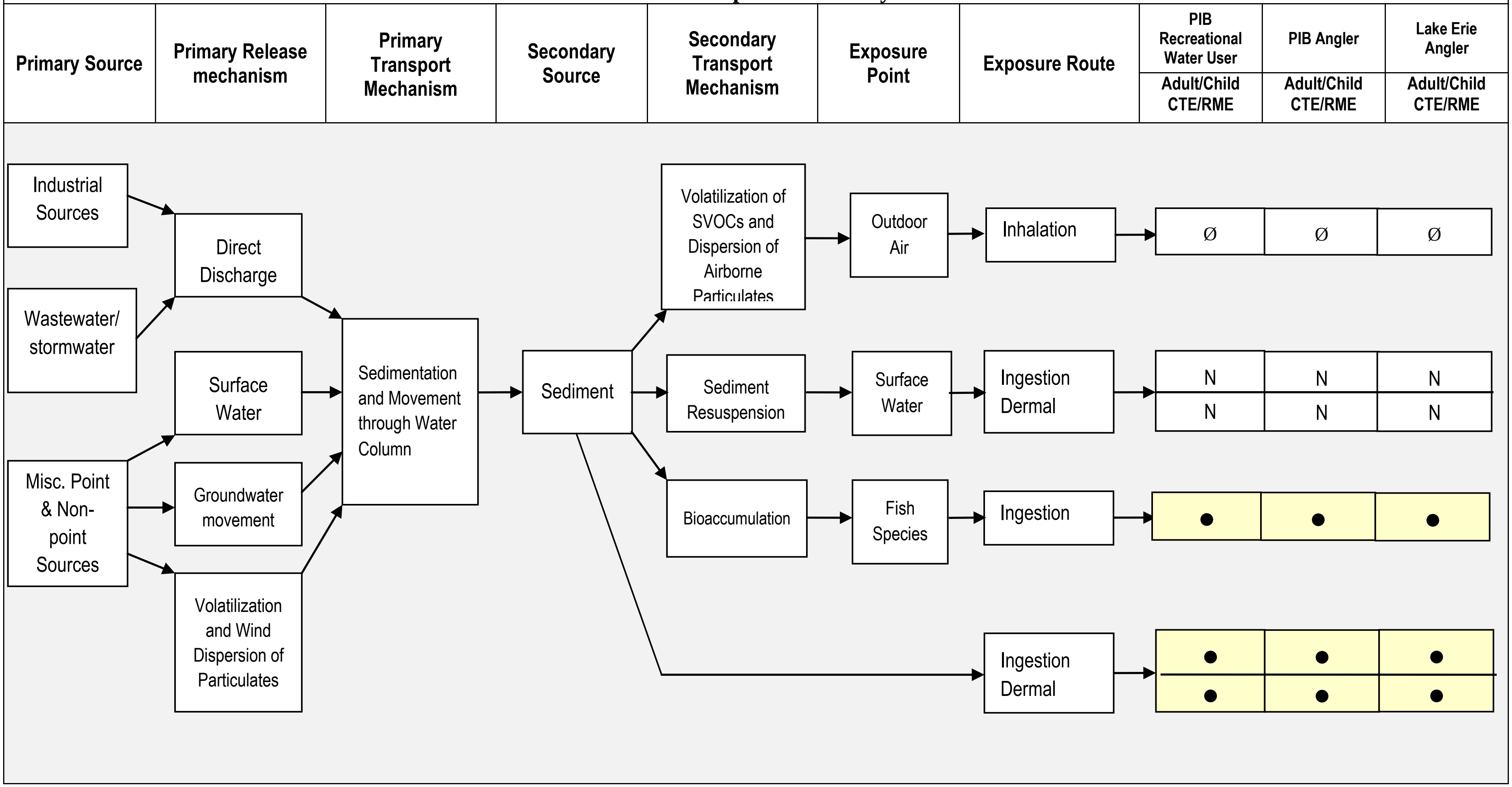
Table 2-19: Summary of Chemical Constituents in Fish Tissue with Federal/State Fish Consumption Advisory Levels

	Fish Consumption Advisory Levels			
Chemical Constituent	**Potential Advisory Level (ppm)**	**Meal advice**	**Source**	**EPA Region 3 RSL[1]**
Mercury	0.13 – 0.25	1/week	USEPA, 1997	0.14[2]
	0.26 – 0.5	2/month		
	0.51 – 1.0	1/month		
	1.1 – 1.9	6/year		
	>1.9	None		
PCBs	0.06 – 0.20	1/week	Great Lakes Protocol (Anderson, 1993)	0.0016[3]
	0.21 – 1.0	1/month		
	1.1 – 1.9	6/year		
	>1.9	None		
Aldrin/Dieldrin (sum)	0.3	None	FDA, 2011[4]	Aldrin: 1.9E-04[2] Dieldrin: 2.0E-04[2]
Chlordane	0.3	None		9.0E-03[2]
Chlordecone	0.3	None		3.2E-04[2]
DDT,DDE & TDE (sum)	5.0	None		DDT: 9.3E-03[2] DDE: 9.3E-03[2] TDE: 1.3E-02[2]
Mirex	0.1	None		1.8E-04[2]
Heptachlor & heptachlor epoxide	0.3	None		Heptachlor: 7.0E-04[2] Heptachlor epoxide: 3.5E-04[2]
Dioxins and furans	25 ppt	2/month		Hexachlorodibenzo-p-dioxin, Mixture: 5.1E-07[2] Dibenzofuran: 1.4[3]
	50 ppt	None		

Notes:
(1) Screening level for risk assessment – these values are not used to determine fish advisory levels
(2) Based on a noncancer hazard quotient=1.0
(3) Based on a target cancer risk of 1 x 10^{-6}
(4) FDA Action Level triggers for a recommendation of Do Not Eat

Figure 3-1
Conceptual Site Model

Potential Exposure Pathways



Notes: ● = complete pathway Ø= complete pathway, negligible exposure N = not able to evaluate (data not available) (Adapted from USFWS 2005)

Table 3-1

Parameters and Values Included in Exposure Calculations

$$DAD = \frac{CS \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT}$$

Dermal Contact with Contaminated Sediment
Adult Recreational Water Users

Parameter	Units	Parameter Value (CTE)	Parameter Value (RME)
DAD = dermal absorbed dose	mg/kg/day	calculated	calculated
CS = COPC concentration in sediment	mg/kg	95%-UCL of mean	95%-UCL of mean
SA = surface area of the skin exposed to sediment	cm^2	5,700 (EPA 2011c)	5,700 (EPA 2011c)
AF = adherence factor (sediment to skin)	mg/cm^2	0.07 (EPA 2004)	0.3 (EPA 2004)
ABS = dermal absorption coefficient	--	Chemical-specific (EPA 2011a)	Chemical-specific (EPA 2011a)
EF = exposure frequency	days/year	38(1) (BPJ)	81(2) (BPJ)
ED = exposure duration	years	9 (EPA 1991b)	30 (EPA 1991b)
CF = conversion factor	kg/mg	1.0E-06	1.0E-06
BW = body weight	kg	70 (EPA 1991b)	70 (EPA 1991b)
AT = averaging time	days	3,285(3) 25,550(4) (EPA 1989)	10,950(3) 25,550(4) (EPA 1989)

Notes:

(1) EF for CTE assumed to be 38 days/year based on 2 days/week for 13 weeks (26) in the summer season
(2) EF for RME exposure assumed to be 81 days/year based on 5 days/week for 13 weeks (36) in the summer season and 1 day/week for 16 weeks (16) in the spring and summer seasons combined
(3) Averaging time for noncancer risk (365 days/yr × 6 years)
(4) Averaging time for cancer risk (365 days/yr × 70 years)

CTE = central tendency exposure
RME = reasonable maximum exposure
COPC = Chemicals of Potential Concern
BPJ=best professional judgment

Table 3-2

Parameters and Values Included in Exposure Calculations

$$ADI = \frac{CS \times IR \times FI \times EF \times ED \times CF}{BW \times AT}$$

Incidental Ingestion of Contaminated Sediment
Adult Recreational Water Users

Parameter	Units	Parameter Value (CTE)	Parameter Value (RME)
ADI = average daily intake	mg/kg/day	calculated	calculated
CS = Chemical Concentration in Sediment	mg/kg	95%-UCL of mean	95%-UCL of mean
IR = Ingestion Rate	mg/day	50 (EPA 2000a)	100 (EPA 2000a)
FI = fraction ingested from site	--	0.3 (BPJ)	0.5 (BPJ)
EF = exposure frequency	days/year	38[(1)] (BPJ)	81[(2)] (BPJ)
ED = exposure duration	years	9 (EPA 1991b)	30 (EPA 1991b)
CF = conversion factor	kg/mg	1.0E-06	1.0E-06
BW = body weight	kg	70 (EPA 1991b)	70 (EPA 1991b)
AT = averaging time	days	3,285[(3)] 25,550[(4)] (EPA 1989)	10,950[(3)] 25,550[(4)] (EPA 1989)

Notes:

(1) EF for CTE assumed to be 38 days/year based on 2 days/week for 13 weeks (26) in the summer season
(2) EF for RME exposure assumed to be 81 days/year based on 5 days/week for 13 weeks (36) in the summer season and 1 day/week for 16 weeks (16) in the spring and summer seasons combined
(3) Averaging time for noncancer risk (365 days/yr × 6 years)
(4) Averaging time for cancer risk (365 days/yr × 70 years)

CTE = central tendency exposure
RME = reasonable maximum exposure
UCL = Upper Confidence Limit
COPC = Chemicals of Potential Concern
BPJ=best professional judgment

Table 3-3
Parameters and Values Included in Exposure Calculations

$$DAD = \frac{CS \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT}$$

Dermal Contact with Contaminated Sediment
Child Recreational Water Users

Parameter	Units	Parameter Value (CTE)	Parameter Value (RME)
DAD = dermal absorbed dose	mg/kg/day	calculated	calculated
CS = chemical concentration in sediment	mg/kg	95%-UCL of mean	95%-UCL of mean
SA = surface area of the skin exposed to sediment	cm^2	2,800 (EPA 2011c)	2,800 (EPA 2011c)
AF = adherence factor (sediment to skin)	mg/cm^2	0.2 (EPA 2004)	3.3 (EPA 2004)
ABS = dermal absorption coefficient	--	Chemical-specific (EPA 2011a)	Chemical-specific (EPA 2011a)
EF = exposure frequency	days/year	38(1) (BPJ)	81(2) (BPJ)
ED = exposure duration	years	6 (EPA 1991b)	6 (EPA 1991b)
CF = conversion factor	kg/mg	1.0E-06	1.0E-06
BW = body weight	kg	15 (EPA 1991b)	15 (EPA 1991b)
AT = averaging time	days	2,190(3) 25,550(4) (EPA 1989)	2,190(3) 25,550(4) (EPA 1989)

Notes:

(1) EF for CTE assumed to be 38 days/year based on 2 days/week for 13 weeks (26) in the summer season
(2) EF for RME exposure assumed to be 81 days/year based on 5 days/week for 13 weeks (36) in the summer season and 1 day/week for 16 weeks (16) in the spring and summer seasons combined
(3) Averaging time for noncancer risk (365 days/yr × 6 years)
(4) Averaging time for cancer risk (365 days/yr × 70 years)

CTE = central tendency exposure
RME = reasonable maximum exposure
COPC = Chemicals of Potential Concern
BPJ=best professional judgment

Table 3-4

Parameters and Values Included in Exposure Calculations

$$ADI = \frac{CS \times IR \times FI \times EF \times ED \times CF}{BW \times AT}$$

Incidental Ingestion of Contaminated Sediment
Child Recreational Water Users

Parameter	Units	Parameter Value (CTE)	Parameter Value (RME)
ADI = average daily intake	mg/kg/day	calculated	calculated
CS = Chemical Concentration in Sediment	mg/kg	95%-UCL of mean	95%-UCL of mean
IR = Ingestion Rate	mg/day	100 (EPA 2000a)	200 (EPA 2000a)
FI = fraction ingested from site	--	0.3 (BPJ)	0.5 (BPJ)
EF = exposure frequency	days/year	38[(1)] (BPJ)	81[(2)] (BPJ)
ED = exposure duration	years	6 (EPA 1991b)	6 (EPA 1991b)
CF = conversion factor	kg/mg	1.0E-06	1.0E-06
BW = body weight	kg	15 (EPA 1991b)	15 (EPA 1991b)
AT = averaging time	days	2,190[(3)] 25,550[(4)] (EPA 1989)	2,190[(3)] 25,550[(4)] (EPA 1989)

Notes:

(1) EF for CTE assumed to be 38 days/year based on 2 days/week for 13 weeks (26) in the summer season
(2) EF for RME exposure assumed to be 81 days/year based on 5 days/week for 13 weeks (36) in the summer season and 1 day/week for 16 weeks (16) in the spring and summer seasons combined
(3) Averaging time for noncancer risk (365 days/yr × 6 years)
(4) Averaging time for cancer risk (365 days/yr × 70 years)

CTE = central tendency exposure
RME = reasonable maximum exposure
UCL = Upper Confidence Limit
COPC = Chemicals of Potential Concern
BPJ=best professional judgment

Table 3-5
Parameters and Values Included in Exposure Calculations

$$Intake = \frac{C_{Fish} \times IR \times EF \times ED \times CF}{BW \times AT}$$

Ingestion of Fish
Adult Anglers

		Recreational Angler	Urban/Subsistence Angler
Parameter	**Units**	**Parameter Value (CTE)**	**Parameter Value (RME)**
Intake = average daily intake or lifetime average daily intake	mg/kg/day	calculated	calculated
C_{Fish} = Chemical Concentration in Fish	mg/kg	Mean/median value	Max value
IR = Ingestion Rate	g/day	17.5 (USEPA 2000 and 2002)	142.4 (USEPA 2000 and 2002)
EF = exposure frequency	days/year	365	365
ED = exposure duration	years	9 (EPA 1991b)	30 (EPA 1991b)
CF = conversion factor	kg/g	1.0E-03	1.0E-03
BW = body weight	kg	70 (EPA 1991b)	70 (EPA 1991b)
AT = averaging time	days	3,285(1) 25,550(2) (EPA 1989)	10,950(1) 25,550(2) (EPA 1989)

Notes:

(1) Averaging time for noncancer risk (365 days/yr × 6 years)
(2) Averaging time for cancer risk (365 days/yr × 70 years)
CTE = central tendency exposure
RME = reasonable maximum exposure

Table 3-6
Parameters and Values Included in Exposure Calculations

$$Intake = \frac{C_{Fish} \times IR \times EF \times ED \times CF}{BW \times AT}$$

Ingestion of Fish
Children of Adult Anglers

		Recreational Angler	**Urban/Subsistence Angler**
Parameter	**Units**	**Parameter Value (CTE)**	**Parameter Value (RME)**
Intake = average daily intake or lifetime average daily intake	mg/kg/day	calculated	calculated
C_{Fish} = Chemical Concentration in Fish	mg/kg	Mean/median value	Max value
IR = Ingestion Rate	g/day	3.75[(1)] (USEPA 2000 and 2002)	30.5[(2)] (USEPA 2000 and 2002)
EF = exposure frequency	days/year	365	365
ED = exposure duration	years	6 (EPA 1991b)	6 (EPA 1991b)
CF = conversion factor	kg/g	1.0E-03	1.0E-03
BW = body weight	kg	15 (EPA 1991b)	15 (EPA 1991b)
AT = averaging time	days	2,190[(3)] 25,550[(4)] (EPA 1989)	2,190[(3)] 25,550[(4)] (EPA 1989)

Notes:

(1) Adult CTE fish consumption value adjusted for body weight (17.5 grams/day × 15 kg/70 kg = 3.75 grams/day)
(2) Adult RME fish consumption value adjusted for body weight (142.4 grams/day × 15 kg/70 kg = 30.5 grams/day)
(3) Averaging time for noncancer risk (365 days/yr × 6 years)
(4) Averaging time for cancer risk (365 days/yr × 70 years)
CTE = central tendency exposure
RME = reasonable maximum exposure

Table 3-7: Exposure Point Concentrations (µg/kg) by Fish Species (maximum value/mean value)

	Bluegill	Brown bullhead	Burbot	Channel catfish	Common carp	Lake trout	LM bass	Northern Pike	Pumpkin-seed	SM bass	Walleye	White bass	White sucker	Yellow perch
Aldrin										**8.6** 0.406				
Arochlor 1254			74 42							**180** 40.1	56 27.2			
Arochlor 1260				**920** 627	470	**1,700** 185	**72** 42.3			**1,200** 887	**270** 176	260	260	
α-BHC						**45.9** 15.6				**28.1** 8.95	**6.55** 4.89			
α -Chlordane				**19.2** 10		**50.2** 22.3								
γ-chlordane						**9.31** 3.6								
4,4'-DDD				**49.4** 30.2		**125** 64.8				**24.2** 20.5		20.9		
4,4'-DDE				**140** 80.5		**367** 164	**16** 9.11	**22.1**		**68.4** 45.2	**29** 28		27.7	
4,4'-DDT				**57.4** 30.2		**239** 100				**53.5** 21.4		**15.8**	**19.8** 19.3	
O,P - DDD						**51.1** 15.4				**19** 8.13				
Dieldrin				**22.8** 12.7		**80** 46.3		**6.48**		**15.2** 9.12	**9.26** 7.39	**9.17**		
Heptachlor				**12.8** 5.46		**53.8** 10.8				**11.7** 3.54				
Heptachlor Epoxide						**8.7** 3.23				**9.6** 3.2				
Mercury			**143**	**188** 180		**181** 171	**195** 187			**350** 276	**261** 218		**174** 161	**132** 88.7
Mirex											**9.12** 4.18			
Cis-Nonachlor						**22.1** 6.15								
Trans-Nonachlor				39.7 21.7		**90** 40.1				**14.8** 9.3				
Oxychlordane						**11** 4.11								
Selenium		**1,000**	**1,000**	**1,000**	**1,000**	**1,000**	**1,000**	**1,000**		**1,590** 1,160	**1,000**		**2,150** 1,650	**1,030** 1,010
Strontium				**250,000**				**250,000**						

Table 4-1
Summary of Cancer-Related Toxicity Values for COPCs in Sediment

	EPA Cancer Class	Oral CSF[1] $(mg/kg\text{-}dy)^{-1}$	Target organ	ABS_{GI}[2]	Dermal CSF $(mg/kg\text{-}dy)^{-1}$	ABS_D[2]
Arsenic	A	1.5E+00	Liver, kidney, lung, and bladder	1.0	1.5 E+00	0.03
Benzo(a)anthracene	B2	7.3E-01[3]	NA	1.0	7.3E-01	0.13
Benzo(a)pyrene	B2	7.3E+00	Stomach and skin	1.0	7.3E+00	0.13
Benzo(b)fluoranthene	B2	7.30E-01[3]	NA	1.0	7.3E-01	0.13
Benzo(k)fluoranthene	B2	7.30E-02[3]	NA	1.0	7.3E-02	0.13
Dibenz(a,h)anthracene	B2	7.3E+00[3]	NA	1.0	7.3E+00 [4]	0.13
Indeno(1,2,3-c,d)pyrene	B2	7.3E-01[3]	NA	1.0	7.3E-01	0.13
Lead	B2	8.5E-03[4]	Kidney	1.0	8.5E-03	--
Total PCBs	B2	2.0E+00	Liver	1.0	2.0E+00	0.14

Notes

(1) Source of toxicity values is EPA IRIS 2011 unless otherwise noted
(2) EPA Region 3 Values (EPA 2011)
(3) EPA Environmental Criteria and Assessment Office
(4) California Environmental Protection Agency

USEPA Cancer Classification

A – Human carcinogen
B1 – Probable human carcinogen
B2 – Possible human carcinogen
C – Possible human carcinogen
D –not classifiable as a human carcinogen

Acronyms

ABS_D – Dermal Absorption Factor
ABS_{GI} – Gastrointestinal absorption Factor
COPCs – Chemical of Potential Concern
CSF – Cancer Slope Factor

Table 4-2 Summary of Noncancer Toxicity Values for COPCs in Sediment					
	Oral RfD[1] (mg/kg-dy)	**Target Organ**	**Confidence**	**ABS_{GI}**	**Dermal RfD (mg/kg-dy)**
Arsenic	3E-04	Skin and CVS	Medium	1.0	3E-04
Benzo(a)anthracene	NA	NA	NA	1.0	NA
Benzo(a)pyrene	NA	NA	NA	1.0	NA
Benzo(b)fluoranthene	NA	NA	NA	1.0	NA
Benzo(k)fluoranthene	NA	NA	NA	1.0	NA
Dibenz(a,h) anthracene	NA	NA	NA	1.0	NA
Indeno(1,2,3-c,d)pyrene	NA	NA	NA	1.0	NA
Lead	NA	CNS, PNS, blood	NA	1.0	NA
Total PCBs	7E-06[2]	Liver	NA	1.0	

Notes

(1) source of toxicity values is EPA IRIS 2011 unless otherwise noted
(2) source = HEAST

USEPA Cancer Classification

A – Human carcinogen
B1 – Probable human carcinogen
B2 – Possible human carcinogen
C – Possible human carcinogen
D –not classifiable as a human carcinogen

Acronyms

COPCs – Chemical of Potential Concern
CSF – Cancer Slope Factor
RfD – Reference Dose
CVS – cardiovascular system
GI – gastrointestinal
CNS – central nervous system
PNS – peripheral nervous system

Table 4-3 Summary of Non-Cancer and Cancer Toxicity Values for COPCs in Fish Tissue						
	Cancer toxicity			Non-cancer toxicity		
	EPA Cancer Class	**Oral CSF(1)** (mg/kg-dy)$^{-1}$	**Target organ**	**Oral RfD(1) (mg/kg-dy)**	**Target Organ**	**Confidence**
Aldrin	B2	1.7E+01	Liver	3.0E-05	Liver	Medium
Arochlor 1254	B2	2.0E+00	Liver	2.0E-05	Immune	Medium
Arochlor 1260	B2	2.0E+00	Liver	2.0E-05(2)	Immune	Medium
Alpha-BHC	B2	6.3E+00	Liver	-	-	-
Alpha-Chlordane	B2	3.50E-01(3)	Liver	5.0E-04(3)	Liver	Medium
Gamma-Chlordane	B2	3.50E-01(3)	Liver	5.0E-04(3)	Liver	Medium
4,4'-DDD	B2	2.4E-01	Lung	-	-	-
4,4'-DDE	B2	2.4E-01	Liver	-	-	-
4,4'-DDT	B2	3.4E-01	Liver	5.0E-04	Liver	Medium
O,P-DDD(5)	B2	2.4E-01	Liver	-	-	-
Dieldrin	B2	1.6E+01	Liver	5.0E-05	Liver	Medium
Heptachlor	B2	4.5E+00	Liver	5.0E-04	Liver	Low
Heptachlor Epoxide	B2	9.1 E+00	Liver	1.3E-05	Liver	Low
Mercury	C	-	-	1.0E-04	CNS	High
Mirex	NA	1.8E+01(4)	-	2.0E-04	Liver	High
Cis-Nonachlor	B2	3.50E-01(3)	Liver	5.0E-04(3)	Liver	Medium
Trans-Nonachlor	B2	3.50E-01(3)	Liver	5.0E-04(3)	Liver	Medium
Oxychlordane	B2	3.50E-01(3)	Liver	5.0E-04(3)	Liver	Medium
Selenium	D	-	-	5.0E-03	Whole body	High
Strontium	NA	-	-	6.0E-1	Bone	Medium

Notes

(1) Source of toxicity values is EPA IRIS 2011 unless otherwise noted
(2) Oral Rfd for Arochlor 1254 used as surrogate
(3) Oral RfD and CSF for Chlordane used as a surrogate
(4) California Environmental Protection Agency
(5) Oral CSF for DDD used as a surrogate

USEPA Cancer Classification

A – Human carcinogen
B1 – Probable human carcinogen
B2 – Possible human carcinogen
C – Possible human carcinogen
D –not classifiable as a human carcinogen
COPCs – Chemical of Potential Concern

Acronyms

CSF – Cancer Slope Factor
RfD – Reference Dose
CVS – cardiovascular system
CNS – central nervous system

Table 4-4
Assumptions and Results of the Adult Lead Model[1]
Evaluation of Blood Lead Concentrations for Adult Recreational Water Users from Contact with Sediment

Exposure Variable	Description of Variable	Units	RME Calculation	CTE Calculation
PbS	Soil lead concentration	µg/g or ppm	85.8	69.3
$R_{fetal/maternal}$	Fetal/maternal PbB ratio	--	0.9	0.9
BKSF	Biokinetic Slope Factor	µg/dL per µg/day	0.4	0.4
GSD_i[2]	Geometric standard deviation PbB	--	2.1	2.1
PbB_0[2]	Baseline PbB	µg/dL	1.5	1.5
IR_S	Soil ingestion rate (including soil-derived indoor dust)	g/day	0.100	0.050
IR_{S+D}	Total ingestion rate of outdoor soil and indoor dust	g/day	--	--
W_S	Weighting factor; fraction of IR_{S+D} ingested as outdoor soil	--	--	--
K_{SD}	Mass fraction of soil in dust	--	--	--
$AF_{S, D}$	Absorption fraction (same for soil and dust)	--	0.30	0.07
$EF_{S, D}$	Exposure frequency (same for soil and dust)	days/yr	81	38
$AT_{S, D}$	Averaging time (same for soil and dust)	days/yr	365	365
PbB_{adult}	**PbB of adult - geometric mean**	**µg/dL**	**1.7**	**1.5**
$PbB_{fetal, 0.95}$	95th percentile PbB among fetuses of adults	**µg/dL**	5.3	4.6
PbB_t	Target PbB level of concern (e.g., 10 ug/dL)	**µg/dL**	**10.0**	**10.0**
$P(PbB_{fetal} > PbB_t)$	**Probability that fetal PbB > PbB_t, assuming lognormal distribution**	%	**0.6%**	**0.4%**

Notes:

(1) USEPA Technical Review Workgroup for Lead, Adult Lead Committee, Version date 6/21/09
(2) GSDi and PbB_o from Analysis of NHANES III (Phases 1 & 2)

Table 4-5: Summary of Assumptions and Results Used for the Child IEUBK Lead Model[1]

Age (years)	Sediment Ingestion Rate (mg/day)[2]	Lead Concentration in Sediment (mg/kg)	Dietary Lead Intake from Fish (µg/day)	Lead Concentration in Fish (mg/kg)	Blood Lead Level µg/dL)	Target Blood Level	Percent above Target Blood Lead Level
0.5 – 1	85	85.8	2.75	97	2.3	10 µg/dL	<0.3%
1 -2	135		3.33		2.6		
2-3	135		3.90		2.5		
3-4	135		3.93		2.4		
4-5	100		3.97		2.1		
5-6	90		4.24		1.8		
6-7	85		4.74		1.7		

Notes:

(1) This model run includes other sources of lead including drinking water (4 µg/L) , outdoor airborne lead concentration (1 µg/m^3) and maternal blood lead level at birth (1 µg/dL) .

(2) Recommended default values for soil intake as recommended by the Technical Review Workgroup for Lead (USEPA 1994b)

Table 5-1
Summary of Cancer and Noncancer Risks by COPC and Exposure Route

Adult Recreational Water Users
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Noncancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Sediment	Dermal absorption	Arsenic	1.32E+01	9.17E-07	1.50E+00	1.38E-06	2.14E-06	3.00E-04	7.13E-03
		Lead	8.58E+01	1.99E-06	8.50E-03	1.69E-08	4.65E-06		
		Total PCBs	1.10E-01	3.58E-08	2.00E+00	7.16E-08	8.35E-08	7.00E-06	1.19E-02
		Benzo(a)anthracene	1.30E+00	3.93E-07	7.30E-01	2.87E-07	9.16E-07		
		Benzo(a)pyrene	1.50E+00	4.53E-07	7.30E+00	3.31E-06	1.06E-06		
		Benzo(b)fluoranthene	1.70E+00	5.13E-07	7.30E-01	3.75E-07	1.20E-06		
		Benzo(k)fluoranthene	1.60E+00	4.83E-07	7.30E-02	3.53E-08	1.13E-06		
		Dibenzo(a,h)anthracene	2.90E-01	8.76E-08	7.30E+00	6.39E-07	2.04E-07		
		Indeno(1,2,3-cd)pyrene	1.60E+00	4.83E-07	7.30E-01	3.53E-07	1.13E-06		
		All Chemicals				**6.46E-06**			**1.91E-02**
Sediment	Incidental ingestion	Arsenic	1.32E+01	8.94E-07	1.50E+00	1.34E-06	2.09E-06	3.00E-04	6.95E-03
		Lead	8.58E+01	5.83E-06	8.50E-03	4.95E-08	1.36E-05		
		Total PCBs	1.10E-01	7.47E-09	2.00E+00	1.49E-08	1.74E-08	7.00E-06	2.49E-03
		Benzo(a)anthracene	1.30E+00	8.83E-08	7.30E-01	6.45E-08	2.06E-07		
		Benzo(a)pyrene	1.50E+00	1.02E-07	7.30E+00	7.44E-07	2.38E-07		
		Benzo(b)fluoranthene	1.70E+00	1.15E-07	7.30E-01	8.43E-08	2.69E-07		
		Benzo(k)fluoranthene	1.60E+00	1.09E-07	7.30E-02	7.93E-09	2.54E-07		
		Dibenzo(a,h)anthracene	2.90E-01	1.97E-08	7.30E+00	1.44E-07	4.60E-08		
		Indeno(1,2,3-cd)pyrene	1.60E+00	1.09E-07	7.30E-01	7.93E-08	2.54E-07		
		All Chemicals				**2.53E-06**			**9.44E-03**
Sediment	**Dermal + Ingestion**	**All Chemicals**				8.99E-06			2.85E-02

Table 5-2
Summary of Cancer and Noncancer Risks by COPC and Exposure Route

Adult Recreational Water Users
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Noncancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)$^{-1}$		(mg/kg-day)	(mg/kg-day)	
Sediment	Dermal absorption	Arsenic	8.90E+00	2.04E-08	1.50E+00	3.06E-08	1.58E-07	3.00E-04	5.28E-04
		Lead	6.93E+01	5.29E-08	8.50E-03	4.49E-10	4.11E-07		
		Total PCBs	7.00E-02	7.48E-10	2.00E+00	1.50E-09	5.82E-09	7.00E-06	8.31E-04
		Benzo(a)anthracene	1.00E+00	9.92E-09	7.30E-01	7.24E-09	7.71E-08		
		Benzo(a)pyrene	1.20E+00	1.19E-08	7.30E+00	8.69E-08	9.26E-08		
		Benzo(b)fluoranthene	1.30E+00	1.29E-08	7.30E-01	9.41E-09	1.00E-07		
		Benzo(k)fluoranthene	1.20E+00	1.19E-08	7.30E-02	8.69E-10	9.26E-08		
		Dibenzo(a,h)anthracene	2.30E-01	2.28E-09	7.30E+00	1.67E-08	1.77E-08		
		Indeno(1,2,3-cd)pyrene	1.30E+00	1.29E-08	7.30E-01	9.41E-09	1.00E-07		
		All chemicals				**1.63E-07**			**1.36E-03**
Sediment	Incidental ingestion	Arsenic	8.90E+00	2.55E-08	1.50E+00	3.83E-08	1.99E-07	3.00E-04	6.62E-04
		Lead	6.93E+01	1.99E-07	8.50E-03	1.69E-09	1.55E-06		
		Total PCBs	7.00E-02	2.01E-10	2.00E+00	4.02E-10	1.56E-09	7.00E-06	2.23E-04
		Benzo(a)anthracene	1.00E+00	2.87E-09	7.30E-01	2.09E-09	2.23E-08		
		Benzo(a)pyrene	1.20E+00	3.44E-09	7.30E+00	2.51E-08	2.68E-08		
		Benzo(b)fluoranthene	1.30E+00	3.73E-09	7.30E-01	2.72E-09	2.90E-08		
		Benzo(k)fluoranthene	1.20E+00	3.44E-09	7.30E-02	2.51E-10	2.68E-08		
		Dibenzo(a,h)anthracene	2.30E-01	6.60E-10	7.30E+00	4.82E-09	5.13E-09		
		Indeno(1,2,3-cd)pyrene	1.30E+00	3.73E-09	7.30E-01	2.72E-09	2.90E-08		
		All chemicals				**7.81E-08**			**8.85E-04**
Sediment	**Dermal + Ingestion**	**All chemicals**				**2.41E-07**			**2.24E-03**

Table 5-3
Summary of Cancer and Noncancer Risks by COPC and Exposure Route

Child Recreational Water Users
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Noncancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)$^{-1}$		(mg/kg-day)	(mg/kg-day)	
Sediment	Dermal absorption	Arsenic	1.32E+01	4.63E-06	1.50E+00	6.94E-06	5.40E-05	3.00E-04	1.80E-01
		Lead	8.58E+01	1.01E-05	8.50E-03	8.55E-08	1.17E-04		
		Total PCBs	1.10E-01	1.80E-07	2.00E+00	3.61E-07	2.11E-06	7.00E-06	3.01E-01
		Benzo(a)anthracene	1.30E+00	1.98E-06	7.30E-01	1.45E-06	2.31E-05		
		Benzo(a)pyrene	1.50E+00	2.28E-06	7.30E+00	1.67E-05	2.67E-05		
		Benzo(b)fluoranthene	1.70E+00	2.59E-06	7.30E-01	1.89E-06	3.02E-05		
		Benzo(k)fluoranthene	1.60E+00	2.44E-06	7.30E-02	1.78E-07	2.84E-05		
		Dibenzo(a,h)anthracene	2.90E-01	4.42E-07	7.30E+00	3.22E-06	5.15E-06		
		Indeno(1,2,3-cd)pyrene	1.60E+00	2.44E-06	7.30E-01	1.78E-06	2.84E-05		
		All chemicals				**3.26E-05**			**4.81E-01**
Sediment	Incidental ingestion	Arsenic	1.32E+01	1.67E-06	1.50E+00	2.50E-06	1.95E-05	3.00E-04	6.49E-02
		Lead	8.58E+01	1.09E-05	8.50E-03	9.25E-08	1.27E-04		
		Total PCBs	1.10E-01	1.39E-08	2.00E+00	2.79E-08	1.63E-07	7.00E-06	2.32E-02
		Benzo(a)anthracene	1.30E+00	1.65E-07	7.30E-01	1.20E-07	1.92E-06		
		Benzo(a)pyrene	1.50E+00	1.90E-07	7.30E+00	1.39E-06	2.22E-06		
		Benzo(b)fluoranthene	1.70E+00	2.16E-07	7.30E-01	1.57E-07	2.52E-06		
		Benzo(k)fluoranthene	1.60E+00	2.03E-07	7.30E-02	1.48E-08	2.37E-06		
		Dibenzo(a,h)anthracene	2.90E-01	3.68E-08	7.30E+00	2.68E-07	4.29E-07		
		Indeno(1,2,3-cd)pyrene	1.60E+00	2.03E-07	7.30E-01	1.48E-07	2.37E-06		
		All chemicals				**4.72E-06**			**8.81E-02**
Sediment	**Dermal + Ingestion**	**All chemicals**				**3.73E-05**			**5.69E-01**

Table 5-4
Summary of Cancer and Noncancer Risks by COPC and Exposure Route

Child Recreational Water Users
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Noncancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Sediment	Dermal absorption	Arsenic	8.90E+00	8.90E-08	1.50E+00	1.33E-07	1.04E-06	3.00E-04	3.46E-03
		Lead	6.93E+01	2.31E-07	8.50E-03	1.96E-09	2.69E-06		
		Total PCBs	7.00E-02	3.26E-09	2.00E+00	6.53E-09	3.81E-08	7.00E-06	5.44E-03
		Benzo(a)anthracene	1.00E+00	4.33E-08	7.30E-01	3.16E-08	5.05E-07		
		Benzo(a)pyrene	1.20E+00	5.20E-08	7.30E+00	3.79E-07	6.06E-07		
		Benzo(b)fluoranthene	1.30E+00	5.63E-08	7.30E-01	4.11E-08	6.57E-07		
		Benzo(k)fluoranthene	1.20E+00	5.20E-08	7.30E-02	3.79E-09	6.06E-07		
		Dibenzo(a,h)anthracene	2.30E-01	9.96E-09	7.30E+00	7.27E-08	1.16E-07		
		Indeno(1,2,3-cd)pyrene	1.30E+00	5.63E-08	7.30E-01	4.11E-08	6.57E-07		
		All chemicals				**7.12E-07**			**8.90E-03**
Sediment	Incidental ingestion	Arsenic	8.90E+00	1.59E-07	1.50E+00	2.38E-07	1.85E-06	3.00E-04	6.18E-03
		Lead	6.93E+01	1.24E-06	8.50E-03	1.05E-08	1.44E-05		
		Total PCBs	7.00E-02	1.25E-09	2.00E+00	2.50E-09	1.46E-08	7.00E-06	2.08E-03
		Benzo(a)anthracene	1.00E+00	1.78E-08	7.30E-01	1.30E-08	2.08E-07		
		Benzo(a)pyrene	1.20E+00	2.14E-08	7.30E+00	1.56E-07	2.50E-07		
		Benzo(b)fluoranthene	1.30E+00	2.32E-08	7.30E-01	1.69E-08	2.71E-07		
		Benzo(k)fluoranthene	1.20E+00	2.14E-08	7.30E-02	1.56E-09	2.50E-07		
		Dibenzo(a,h)anthracene	2.30E-01	4.10E-09	7.30E+00	3.00E-08	4.79E-08		
		Indeno(1,2,3-cd)pyrene	1.30E+00	2.32E-08	7.30E-01	1.69E-08	2.71E-07		
		All chemicals				**4.86E-07**			**8.26E-03**
Sediment	**Dermal + Ingestion**	**All chemicals**				**1.20E-06**			**1.72E-02**

Table 5-5
Summary of Receptor Risks and Hazards for COPCs

Presque Isle Bay Adult Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Bluegill	No COPCs							
	Brown bullhead	Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	0.41
	Burbot	Aroclor 1254	.074	6.45E-05	2	1.30E-04	1.5E-04	2.0E-05	7.5
		Mercury	.143	1.25E-04	n/a	n/a	2.9E-04	1.0E-04	2.9
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	0.41
		All Chemicals				1.3E-04			10.8
	Channel Catfish	Aroclor 1260	0.920	8.02E-04	2	1.6E-03	1.9E-03	2.0E-05	93.6
		Chlordanes	0.040	5.1E-05	3.50E-01	1.8E-05	1.2E-04	5.0E-04	0.20
		DDD, 4,4’ -	0.050	4.3E-05	2.4E+-01	1.0E-05	1.0E-04	n/a	n/a
		DDE, 4,4’ -	0.140	1.2E-04	3.40E-01	4.2E-05	2.9E-04	n/a	n/a
		DDT, 4,4’ -	0.057	5.0E-05	3.40E-01	1.7E-05	1.2E-04	5.0E-04	0.20
		Dieldrin	0.023	2.0E-05	1.6E+01	3.2E-04	4.6E-05	5.0E-05	0.93
		Heptachlor	0.013	1.1E-05	4.5	5.0E-05	2.6E-05	5.0E-04	0.05
		Mercury	0.189	1.6E-04	n/a	n/a	3.8E-04	1.0E-04	3.8
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	.41
		Strontium	0.025	2.2E-01	n/a	n/a	5.1E-01	6.0E-01	0.85
		All Chemicals				2.1E-03			100.0
	Common Carp	Aroclor 1254	0.470	4.1E-04	2	8.2E-04	9.6E-04	2.0E-05	47.8
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	.41
		All Chemicals				8.0E-04			48.2

Table 5-5 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Adult Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Lake Trout	Aroclor 1260	1.70	1.5E-03	2	3.0E-03	3.5E-03	2.0E-05	173
		BHC, alpha -	0.046	4.0E-05	6.3	2.5E-04	9.4E-05	n/a	n/a
		Chlordanes	0.090	1.6E-04	3.50E-01	5.6E-05	3.7E-04	5.0E-04	0.74
		DDDs	0.051	1.5E-04	2.4E+-01	3.7E-05	3.6E-04	n/a	n/a
		DDE, 4,4’	0.367	3.2E-04	3.40E-01	1.1E-04	7.5E-04	n/a	n/a
		DDT, 4,4’	0.239	2.1E-04	3.40E-01	7.1E-05	4.9E-04	5.0E-04	0.97
		Dieldrin	0.081	7.0E-05	1.6E+01	1.1E-03	1.7E-04	5.0E-05	3.29
		Heptachlor	0.054	4.7E-05	4.5	2.1E-04	1.1E-04	5.0E-04	0.22
		Heptachlor Epoxide	0.009	7.6E-06	9.1	6.9E-05	1.8E-05	1.3E-05	1.36
		Mercury	0.181	1.6E-04	n/a	n/a	3.7E-04	1.0E-04	3.7
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	.41
		All Chemicals				4.9E-03			183.7
	Largemouth Bass	Aroclor 1260	0.072	6.3E-05	2	1.3E-04	1.5E-04	2.0E-05	7
		DDE, 4,4’	0.016	1.4E-05	3.40E-01	4.8E-06	3.3E-05	n/a	n/a
		Mercury	0.195	1.7E-04	n/a	n/a	4.0E-04	1.0E-04	4.0
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	.41
		All Chemicals				1.0E-04			11.4
	Northern Pike	DDE, 4,4’ -	0.022	1.9E-05	3.40E-01	6.5E-06	4.5E-05	n/a	n/a
		Dieldrin	.006	5.7E-06	1.6E+01	9.0E-05	1.3E-05	5.0E-05	0.26
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	.41
		All Chemicals				1.0E-04			0.67

Table 5-5 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Adult Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Pumpkinseed Fish	No COPCs							
	Smallmouth Bass	Aldrin	.009	7.5E-06	1.70E+01	1.3E-04	1.8E-05	3.0E-05	0.58
		Aroclors	1.20	1.2E-03	2	2.4E-03	2.8E-03	2.0E-05	122
		BHC, alpha -	.028	2.5E-05	6.3	1.5E-04	5.7E-05	n/a	n/a
		DDDs	0.019	3.8E-05	2.4E+-01	9.0E-06	8.8E-05	n/a	n/a
		DDE, 4,4' -	0.068	6.0E-05	3.40E-01	2.0E-05	1.4E-04	n/a	n/a
		DDT, 4,4' -	0.054	4.7E-05	3.40E-01	1.6E-05	1.1E-04	5.0E-04	0.22
		Dieldrin	0.015	1.3E-05	1.6E+01	2.1E-04	3.1E-05	5.0E-05	0.62
		Heptachlor	0.017	1.0E-05	4.5	4.6E-05	2.4E-05	5.0E-04	0.05
		Heptachlor Epoxide	0.010	8.4E-06	9.1	7.6E-05	2.0E-05	1.3E-05	1.50
		Mercury	0.350	3.1E-04	n/a	n/a	7.1E-04	1.0E-04	7.1
		Nonachlor, Trans -	0.015	1.3E-05	3.50E-01	4.5E-06	3.1E-05	5.0E-04	0.06
		Selenium	1.59	1.4E-03	n/a	n/a	3.2E-03	5.0E-03	0.65
		All Chemicals				3.1E-03			132.8
	Walleye	Aroclors	0.270	2.8E-04	2	5.7E-04	6.6E-04	2.0E-05	33.2
		BHC, alpha -	0.007	5.7E-06	6.3	3.6E-05	1.3E-05	n/a	n/a
		DDE, 4,4' -	0.029	2.5E-05	3.40E-01	8.6E-06	5.9E-05	n/a	n/a
		Dieldrin	0.009	8.1E-06	1.6E+01	1.3E-04	1.9E-05	5.0E-05	0.38
		Mercury	0.261	2.3E-04	n/a	n/a	5.3E-04	1.0E-04	5.3
		Mirex	0.009	8.0E-06	n/a	n/a	1.7E-05	2.0E-04	0.09
		Selenium	1	8.72E-04	n/a	n/a	2.0E-03	5.0E-03	.41

Table 5-5 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Adult Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
	Walleye	**All Chemicals**				7.4E-04			39.4
Fish Tissue	White Bass	Arochlor 1260	0.260	2.3E-04	2	4.5E-04	5.3E-04	2.0E-05	26
		DDD, 4,4'-	0.021	1.8E-05	2.4E+-01	4.4E-06	4.2E-05	n/a	n/a
		DDT, 4,4 -	0.016	1.4E-05	3.40E-01	4.7E-06	3.2E-05	5.0E-04	0.06
		Dieldrin	0.009	8.0E-06	1.6E+01	1.3E-04	1.9E-05	5.0E-05	0.37
		All Chemicals				5.9E-04			26.4
	White Sucker	Aroclor 1260	0.260	2.3E-04	2	4.5E-04	5.3E-04	2.0E-05	26
		DDE, 4,4' -	0.028	2.4E-05	3.40E-01	8.2E-06	5.6E-05	n/a	n/a
		DDT, 4,4' -	0.020	1.7E-05	3.40E-01	5.9E-06	4.0E-05	5.0E-04	0.08
		Mercury	0.174	1.5E-04	n/a	n/a	3.5E-04	1.0E-04	3.5
		Selenium	2.15	1.9E-03	n/a	n/a	4.4E-03	5.0E-03	0.88
		All Chemicals				4.6E-04			30.5
	Yellow Perch	Mercury	0.132	1.2E-04	n/a	n/a	2.7E-04	1.0E-04	2.7
		Selenium	1.03	9.0E-04	n/a	n/a	2.1E-03	5.0E-03	0.42
		All Chemicals				n/a			3.1

Table 5-6
Summary of Receptor Risks and Hazards for COPCs

Adult Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Bluegill	No COPCs							
	Brown bullhead	Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
	Burbot	Aroclor 1254	0.043	1.37E-06	2	2.73E-06	1.06E-05	2.0E-05	0.5
		Mercury	0.143	4.58E-06	n/a	n/a	3.56E-05	1.0E-04	0.4
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
		All Chemicals				2.7E-06			0.95
	Channel Catfish	Aroclor 1260	0.627	2.01E-05	2	4.03E-05	1.57E-04	2.0E-05	7.8
		Chlordanes	0.022	1.0E-06	3.50E-01	6.7E-07	8.1E-06	5.0E-04	0.02
		DDD, 4,4' -	0.030	9.7E-07	2.4E+-01	2.3E-07	7.5E-06	n/a	n/a
		DDE, 4,4' -	0.081	2.6E-06	3.40E-01	8.8E-07	2.0E-05	n/a	n/a
		DDT, 4,4' -	0.030	9.7E-07	3.40E-01	3.3E-07	7.5E-06	5.0E-04	0.02
		Dieldrin	0.013	4.1E-07	1.6E+01	6.5E-06	3.2E-06	5.0E-05	0.06
		Heptachlor	.005	1.8E-07	4.5	7.9E-07	1.4E-06	5.0E-04	0.003
		Mercury	0.180	5.8E-06	n/a	n/a	4.5E-05	1.0E-04	0.45
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
		Strontium	0.025	8.0E-03	n/a	n/a	6.3E-02	6.0E-01	0.10
		All Chemicals				5.0E-05			8.5
	Common Carp	Aroclor 1254	0.470	1.5E-05	2	3.0E-05	1.2E-04	2.0E-05	5.9
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
		All Chemicals				3.0E-05			6.0

Table 5-6 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Adult Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)$^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Lake Trout	Aroclor 1260	0.185	6.0E-06	2	1.2E-05	4.6E-05	2.0E-05	2.3
		BHC, alpha -	0.016	5.0E-07	6.3	3.2E-06	3.9E-06	n/a	n/a
		Chlordanes	0.040	2.5E-06	3.50E-01	8.6E-07	2.5E-06	5.0E-04	0.04
		DDDs	0.051	2.6E-06	2.4E+-01	6.2E-07	2.0E-05	n/a	n/a
		DDE, 4,4’	0.164	5.3E-06	3.40E-01	1.8E-06	4.1E-05	n/a	n/a
		DDT, 4,4’	0.100	3.2E-06	3.40E-01	1.1E-06	2.5E-05	5.0E-04	0.05
		Dieldrin	0.046	1.5E-06	1.6E+01	2.4E-05	1.2E-05	5.0E-05	0.23
		Heptachlor	0.011	3.5E-07	4.5	1.6E-06	2.7E-06	5.0E-04	0.01
		Heptachlor Epoxide	0.003	1.0E-07	9.1	9.4E-07	8.1E-07	1.3E-05	0.06
		Mercury	0.171	5.6E-06	n/a	n/a	4.3E-05	1.0E-04	0.43
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
		All Chemicals				5.0E-05			3.2
	Largemouth Bass	Aroclor 1260	0.0423	1.4E-06	2	2.7E-06	1.1E-05	2.0E-05	1
		DDE, 4,4’	0.009	2.9E-07	3.40E-01	1.0E-07	2.3E-06	n/a	n/a
		Mercury	0.187	6.0E-06	n/a	n/a	4.7E-05	1.0E-04	0.5
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
		All Chemicals				2.8E-06			1.6
	Northern Pike	DDE, 4,4’ -	0.022	7.1E-07	3.40E-01	2.4E-07	5.5E-06	n/a	n/a
		Dieldrin	0.006	2.0E-07	1.6E+01	3.3E-06	1.6E-06	5.0E-05	0.03
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05
		All Chemicals				3.6E-06			0.08

Table 5-6 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Adult Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)$^{-1}$		(mg/kg-day)	(mg/kg-day)	
	Pumpkinseed Fish	No COPCs							
	Smallmouth Bass	Aldrin	0.004	1.3E-07	1.70E+01	2.2E-06	1.0E-06	3.0E-05	0.03
		Aroclors	0.887	3.0E-05	2	6.0E-05	2.3E-04	2.0E-05	11.6
		BHC, alpha -	0.009	2.9E-07	6.3	1.8E-06	2.2E-06	n/a	n/a
		DDDs	0.008	9.2E-07	2.4E+-01	2.2E-07	7.2E-06	n/a	n/a
		DDE, 4,4' -	0.045	1.5E-06	3.40E-01	5.0E-07	1.1E-05	n/a	n/a
		DDT, 4,4' -	0.021	6.9E-07	3.40E-01	2.3E-07	5.3E-06	5.0E-04	0.01
		Dieldrin	.009	2.9E-07	1.6E+01	4.7E-06	2.3E-06	5.0E-05	0.05
		Heptachlor	0.004	1.1E-07	4.5	5.1E-07	8.9E-07	5.0E-04	0.002
		Heptachlor Epoxide	0.032	1.0E-07	9.1	9.4E-07	8.0E-07	1.3E-05	0.06
		Mercury	0.276	8.9E-06	n/a	n/a	6.9E-05	1.0E-04	0.69
		Nonachlor, Trans -	0.009	3.0E-07	3.50E-01	1.1E-07	2.3E-06	5.0E-04	0.005
		Selenium	1.16	3.7E-05	n/a	n/a	2.9E-04	5.0E-03	0.058
		All Chemicals				7.0E-05			12.5
	Walleye	Aroclors	0.176	6.5E-06	2	1.3E-05	5.1E-05	2.0E-05	2.5
		BHC, alpha -	0.005	1.6E-07	6.3	9.9E-07	1.2E-06	n/a	n/a
		DDE, 4,4' -	0.028	7.3E-07	3.40E-01	2.5E-07	5.7E-06	n/a	n/a
		Dieldrin	0.007	2.4E-07	1.6E+01	3.8E-06	1.9E-06	5.0E-05	0.04
		Mercury	0.218	7.0E-06	n/a	n/a	5.5E-05	1.0E-04	0.55
		Mirex	0.004	1.3E-07	n/a	n/a	1.1E-06	2.0E-04	0.01
		Selenium	1	3.21E-05	n/a	n/a	2.50E-04	5.0E-03	0.05

Table 5-6 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Adult Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)$^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Walleye	**All Chemicals**				2.0E-05			3.2
	White Bass	Arochlor 1260	0.260	8.4E-06	2	1.7E-05	6.5E-05	2.0E-05	3
		DDD, 4,4'-	0.021	6.7E-07	2.4E+-01	1.6E-07	5.2E-06	n/a	n/a
		DDT, 4,4 -	0.016	5.1E-07	3.40E-01	1.7E-07	3.9E-06	5.0E-04	0.01
		Dieldrin	0.009	3.0E-07	1.6E+01	4.7E-06	2.3E-06	5.0E-05	0.05
		All Chemicals				2.0E-05			3.1
	White Sucker	Aroclor 1260	0.260	8.4E-06	2	1.7E-05	6.5E-05	2.0E-05	3
		DDE, 4,4' -	0.028	8.9E-07	3.40E-01	3.0E-07	6.9E-06	n/a	n/a
		DDT, 4,4' -	0.019	6.2E-07	3.40E-01	2.1E-07	4.8E-06	5.0E-04	0.01
		Mercury	0.161	5.2E-06	n/a	n/a	4.0E-05	1.0E-04	0.40
		Selenium	1.65	5.3E-05	n/a	n/a	4.1E-04	5.0E-03	0.08
		All Chemicals				2.0E-05			3.5
	Yellow Perch	Mercury	0.089	2.9E-06	n/a	n/a	2.2E-05	1.0E-04	0.22
		Selenium	1.01	3.2E-05	n/a	n/a	2.5E-04	5.0E-03	0.05
		All Chemicals				n/a			0.27

Table 5-7
Summary of Receptor Risks and Hazards for COPCs

Children of Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC (mg/kg)	Cancer Risk Calculations: Intake (mg/kg-day)	CSF (mg/kg-day)$^{-1}$	Cancer Risk	Non-cancer Risk Calculations: Intake (mg/kg-day)	RfD (mg/kg-day)	Hazard Quotient
Fish Tissue	Bluegill	No COPCs							
	Brown bullhead	Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.07E-01
	Burbot	Aroclor 1254	0.074	1.29E-05	2	2.58E-05	1.50E-04	2.0E-05	7.5
		Mercury	0.143	2.49E-05	n/a	n/a	2.91E-04	1.0E-04	2.9
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.01E-01
		All Chemicals				2.6E-05			10.8
	Channel Catfish	Aroclor 1260	0.920	1.60E-04	2	3.21E-04	1.87E-03	2.0E-05	93.5
		Chlordanes	0.040	1.0E-05	3.50E-01	3.6E-06	1.2E-04	5.0E-04	0.02
		DDD, 4,4’ -	0.050	8.6E-06	2.4E+-01	2.1E-06	1.0E-04	n/a	n/a
		DDE, 4,4’ -	0.140	2.4E-05	3.40E-01	8.3E-06	2.9E-04	n/a	n/a
		DDT, 4,4’ -	0.057	1.0E-05	3.40E-01	3.4E-06	1.2E-04	5.0E-04	0.23
		Dieldrin	0.023	4.0E-06	1.6E+01	6.4E-05	4.6E-05	5.0E-05	0.93
		Heptachlor	0.013	2.2E-06	4.5	1.0E-05	2.6E-05	5.0E-04	0.05
		Mercury	0.189	3.3E-05	n/a	n/a	3.8E-04	1.0E-04	3.8
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.01E-01
		Strontium	0.025	4.4E-02	n/a	n/a	5.1E-01	6.0E-01	0.85
		All Chemicals				4.0E-04			99.8
	Common Carp	Aroclor 1254	0.470	8.2E-05	2	1.6E-04	9.6E-04	2.0E-05	47.8
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.07E-01
		All Chemicals				1.6E-04			48.3

Table 5-7 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Children of Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Lake Trout	Aroclor 1260	1.70	3.0E-04	2	6.0E-04	3.5E-03	2.0E-05	173
		BHC, alpha -	0.046	8.0E-06	6.3	5.0E-05	9.3E-05	n/a	n/a
		Chlordanes	0.090	3.2E-05	3.50E-01	1.1E-05	3.7E-04	5.0E-04	0.74
		DDDs	0.015	3.0E-05	2.4E+-01	7.4E-06	3.6E-04	n/a	n/a
		DDE, 4,4’	0.367	6.4E-05	3.40E-01	2.2E-05	7.5E-04	n/a	n/a
		DDT, 4,4’	0.239	4.2E-05	3.40E-01	1.4E-05	4.9E-04	5.0E-04	0.97
		Dieldrin	0.081	1.4E-05	1.6E+01	2.3E-04	1.6E-04	5.0E-05	3.39
		Heptachlor	0.054	9.4E-06	4.5	4.2E-05	1.1E-04	5.0E-04	0.22
		Heptachlor Epoxide	0.009	1.5E-06	9.1	1.4E-05	1.8E-05	1.3E-05	1.36
		Mercury	0.181	3.2E-05	n/a	n/a	3.7E-04	1.0E-04	3.7
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.07E-01
		All Chemicals				1.0E-03			183.8
	Largemouth Bass	Aroclor 1260	0.072	1.3E-05	2	2.5E-05	1.5E-04	2.0E-05	7
		DDE, 4,4’	0.016	2.8E-06	3.40E-01	9.5E-07	3.3E-05	n/a	n/a
		Mercury	0.195	3.4E-05	n/a	n/a	4.0E-04	1.0E-04	4.0
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.07E-01
		All Chemicals				2.6E-05			11.4
	Northern Pike	DDE, 4,4’ -	0.022	3.9E-06	3.40E-01	1.3E-06	5.0E-05	n/a	n/a
		Dieldrin	0.006	1.1E-06	1.6E+01	1.8E-05	1.3E-05	5.0E-05	0.26
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.07E-01
		All Chemicals				1.9E-05			0.67

Table 5-7 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Children of Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
			EPC	Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)-1		(mg/kg-day)	(mg/kg-day)	
	Pumpkinseed Fish	No COPCs							
	Smallmouth Bass	Aldrin	0.009	1.5E-06	1.70E+01	2.6E-05	1.8E-05	3.0E-05	0.58
		Aroclors	1.20	2.4E-04	2	4.8E-04	2.8E-03	2.0E-05	140
		BHC, alpha -	0.028	4.9E-06	6.3	3.1E-05	5.7E-05	n/a	n/a
		DDDs	0.019	7.5E-06	2.4E+-01	1.8E-06	8.8E-05	n/a	n/a
		DDE, 4,4' -	0.068	1.2E-05	3.40E-01	4.0E-06	1.4E-04	n/a	n/a
		DDT, 4,4' -	0.054	9.3E-06	3.40E-01	3.2E-06	1.1E-04	5.0E-04	0.22
		Dieldrin	0.015	2.7E-06	1.6E+01	4.2E-05	3.1E-05	5.0E-05	0.62
		Heptachlor	0.017	2.0E-06	4.5	9.1E-06	2.4E-05	5.0E-04	0.05
		Heptachlor Epoxide	0.010	1.7E-06	9.1	1.5E-05	2.0E-05	1.3E-05	1.50
		Mercury	0.350	6.1E-05	n/a	n/a	7.1E-04	1.0E-04	7.1
		Nonachlor, Trans -	0.015	2.6E-06	3.50E-01	9.0E-07	3.0E-05	5.0E-04	0.060
		Selenium	1.59	2.8E-04	n/a	n/a	3.2E-03	5.0E-03	0.65
		All Chemicals				6.1E-04			150.1
	Walleye	Aroclors	0.270	5.7E-05	2	1.1E-04	6.6E-04	2.0E-05	33.1
		BHC, alpha -	0.007	1.1E-06	6.3	7.2E-06	1.3E-05	n/a	n/a
		DDE, 4,4' -	0.029	5.1E-06	3.40E-01	1.7E-06	5.9E-05	n/a	n/a
		Dieldrin	0.009	1.6E-06	1.6E+01	2.6E-05	1.9E-05	5.0E-05	0.38
		Mercury	0.261	4.6E-05	n/a	n/a	5.3E-04	1.0E-04	5.3
		Mirex	0.009	1.6E-06	n/a	n/a	1.9E-05	2.0E-04	0.09
		Selenium	1	1.74E-04	n/a	n/a	2.03E-03	5.0E-03	4.07E-01
		All Chemicals				1.4E-04			39.3

Table 5-7 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Children of Urban/Subsistence Anglers
Reasonable Maximum Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
	White Bass	Arochlor 1260	0.260	4.5E-05	2	9.1E-05	5.3E-04	2.0E-05	26
		DDD, 4,4'-	0.021	3.6E-06	2.4E+-01	8.7E-07	4.2E-05	n/a	n/a
		DDT, 4,4 -	0.016	2.8E-06	3.40E-01	9.4E-07	3.2E-05	5.0E-04	0.06
		Dieldrin	0.009	1.6E-06	1.6E+01	2.6E-05	1.9E-05	5.0E-05	0.37
		All Chemicals				1.2E-04			26.4
	White Sucker	Aroclor 1260	0.260	4.5E-05	2	9.1E-05	5.3E-04	2.0E-05	26
		DDE, 4,4' -	0.028	4.8E-06	3.40E-01	1.6E-06	5.6E-05	n/a	n/a
		DDT, 4,4' -	0.020	3.5E-06	3.40E-01	1.2E-06	4.0E-05	5.0E-04	0.08
		Mercury	0.174	3.0E-05	n/a	n/a	3.4E-04	1.0E-04	3.5
		Selenium	2.15	3.8E-04	n/a	n/a	4.4E-03	5.0E-03	0.88
		All Chemicals				9.4E-05			30.5
	Yellow Perch	Mercury	0.132	2.3E-05	n/a	n/a	2.7E-04	1.0E-04	2.7
		Selenium	1.03	1.8E-04	n/a	n/a	2.1E-03	5.0E-03	0.42
		All Chemicals				n/a			3.1

Table 5-8
Summary of Receptor Risks and Hazards for COPCs

Children of Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Bluegill	No COPCs							
	Brown bullhead	Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
	Burbot	Aroclor 1254	0.043	9.1E-07	2	1.8E-06	1.1E-05	2.0E-05	0.5
		Mercury	0.143	3.1E-06	n/a	n/a	3.6E-05	1.0E-04	0.36
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				1.8E-06			0.91
	Channel Catfish	Aroclor 1260	0.627	1.3E-05	2	2.7E-05	1.6E-04	2.0E-05	7.8
		Chlordanes	0.022	7.0E-07	3.5E-01	2.5E-07	8.2E-06	5.0E-04	0.02
		DDD, 4,4’ -	0.030	6.5E-07	2.4E+-01	1.6E-07	7.5E-06	n/a	n/a
		DDE, 4,4’ -	0.081	1.7E-06	3.4E-01	5.9E-07	2.0E-05	n/a	n/a
		DDT, 4,4’ -	0.030	6.5E-07	3.4E-01	2.2E-07	7.5E-06	5.0E-04	0.02
		Dieldrin	0.013	2.7E-07	1.6E+01	4.4E-06	3.2E-06	5.0E-05	0.06
		Heptachlor	0.005	1.2E-07	4.5	5.3E-07	1.4E-06	5.0E-04	.003
		Mercury	0.180	3.9E-06	n/a	n/a	4.5E-05	1.0E-04	0.45
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				3.3E-05			8.40
	Common Carp	Aroclor 1254	0.470	1.0E-05	2	2.0E-05	1.2E-04	2.0E-05	5.9
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				2.0E-05			6.0

Table 5-8 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Children of Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
Fish Tissue	Lake Trout	Aroclor 1260	0.185	4.0E-06	2	7.9E-06	4.6E-05	2.0E-05	2.3
		BHC, alpha -	0.016	3.4E-07	6.3	2.1E-06	3.9E-06	n/a	n/a
		Chlordanes	0.040	1.6E-06	3.50E-01	5.7E-07	1.9E-05	5.0E-04	0.04
		DDDs	0.015	1.7E-06	2.4E+-01	6.2E-07	2.0E-05	n/a	n/a
		DDE, 4,4’	0.164	3.5E-06	3.40E-01	1.2E-06	4.1E-05	n/a	n/a
		DDT, 4,4’	0.100	2.2E-06	3.40E-01	7.3E-07	2.5E-05	5.0E-04	0.05
		Dieldrin	0.046	9.9E-07	1.6E+01	1.6E-05	1.2E-05	5.0E-05	0.23
		Heptachlor	0.011	2.3E-07	4.5	1.0E-06	2.7E-06	5.0E-04	0.01
		Heptachlor Epoxide	0.003	6.9E-08	9.1	6.3E-07	8.1E-07	1.3E-05	0.06
		Mercury	0.171	3.7E-06	n/a	n/a	4.3W-05	1.0E-04	0.43
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				3.1E-05			3.2
	Largemouth Bass	Aroclor 1260	0.042	9.1E-07	2	1.8E-06	1.1E-05	2.0E-05	1
		DDE, 4,4’	0.009	2.0E-07	3.40E-01	6.6E-08	2.3E-06	n/a	n/a
		Mercury	0.187	4.0E-06	n/a	n/a	4.7E-05	1.0E-04	0.47
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				1.9E-06			1.5
	Northern Pike	DDE, 4,4’ -	0.022	4.7E-07	3.40E-01	1.6E-07	5.5E-06	n/a	n/a
		Dieldrin	0.006	1.4E-07	1.6E+01	2.2E-06	1.6E-06	5.0E-05	0.03
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				2.4E-06			0.08

Table 5-8 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Children of Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	(mg/kg-day)-1		(mg/kg-day)	(mg/kg-day)	
	Pumpkinseed Fish	No COPCs							
	Smallmouth Bass	Aldrin	0.004	8.7E-08	1.70E+01	1.5E-06	1.0E-06	3.0E-05	0.03
		Aroclors	0.887	2.0E-05	2	4.0E-05	2.3E-04	2.0E-05	11.6
		BHC, alpha -	.009	1.9E-07	6.3	1.2E-06	2.2E-06	n/a	n/a
		DDDs	.008	6.1E-07	2.4E+-01	1.5E-07	7.2E-06	n/a	n/a
		DDE, 4,4' -	0.045	9.7E-07	3.40E-01	3.3E-07	1.1E-05	n/a	n/a
		DDT, 4,4' -	0.021	4.6E-07	3.40E-01	1.6E-07	5.3E-06	5.0E-04	0.01
		Dieldrin	0.009	2.0E-07	1.6E+01	3.1E-06	2.3E-06	5.0E-05	0.05
		Heptachlor	0.004	7.6E-08	4.5	3.4E-07	8.9E-07	5.0E-04	0.002
		Heptachlor Epoxide	0.032	6.9E-08	9.1	6.2E-07	8.0E-07	1.3E-05	0.06
		Mercury	0.276	5.9E-06	n/a	n/a	6.9E-05	1.0E-04	0.69
		Nonachlor, Trans -	0.009	2.0E-07	3.50E-01	7.0E-08	2.3E-06	5.0E-04	0.005
		Selenium	1.16	2.5E-05	n/a	n/a	2.9E-04	5.0E-03	0.06
		All Chemicals				4.7E-05			12.5
	Walleye	Aroclors	0.176	4.4E-06	2	8.7E-06	5.1E-05	2.0E-05	2.5
		BHC, alpha -	0.005	1.1E-07	6.3	6.6E-07	1.2E-06	n/a	n/a
		DDE, 4,4' -	0.028	4.9E-07	3.40E-01	1.7E-07	5.7E-06	n/a	n/a
		Dieldrin	0.007	1.6E-07	1.6E+01	2.5E-06	1.9E-06	5.0E-05	0.04
		Mercury	0.218	4.7E-06	n/a	n/a	5.5E-05	1.0E-04	0.55
		Mirex	0.004	9.0E-08	n/a	n/a	1.1E	2.0E-04	0.01
		Selenium	1	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	5.0E-02
		All Chemicals				1.2E-05			3.2

Table 5-8 (cont.)
Summary of Receptor Risks and Hazards for COPCs

Children of Recreational Anglers
Central Tendency Exposure

Exposure Point	Exposure Route	COPC	EPC	Cancer Risk Calculations			Non-cancer Risk Calculations		
				Intake	CSF	Cancer Risk	Intake	RfD	Hazard Quotient
			(mg/kg)	(mg/kg-day)	$(mg/kg\text{-}day)^{-1}$		(mg/kg-day)	(mg/kg-day)	
	White Bass	Arochlor 1260	0.260	5.6E-06	2	1.1E-05	6.5E-05	2.0E-05	3
		DDD, 4,4'-	0.0291	4.5E-07	2.4E+-01	1.1E-07	5.2E-06	n/a	n/a
		DDT, 4,4 -	0.016	3.4E-07	3.40E-01	1.2E-07	3.9E-06	5.0E-04	0.01
		Dieldrin	0.009	2.0E-07	1.6E+01	3.1E-06	2.3E-06	5.0E-05	0.05
		All Chemicals				1.4E-05			3.1
	White Sucker	Aroclor 1260	0.260	5.6E-06	2	1.1E-05	6.5E-05	2.0E-05	3
		DDE, 4,4' -	0.028	5.9E-07	3.40E-01	2.0E-07	6.9E-06	n/a	n/a
		DDT, 4,4' -	0.019	4.1E-07	3.40E-01	1.4E-07	4.8E-06	5.0E-04	0.01
		Mercury	0.161	3.4E-06	n/a	n/a	4.0E-05	1.0E-04	0.40
		Selenium	1.65	3.5E-05	n/a	n/a	4.1E-04	5.0E-03	8.3E-02
		All Chemicals				1.1E-05			3.5
	Yellow Perch	Mercury	0.089	1.9E-06	n/a	n/a	2.2E-05	1.0E-04	0.22
		Selenium	1.01	2.1E-05	n/a	n/a	2.5E-04	5.0E-03	0.05
		All Chemicals				n/a			0.27

Table 6-1 Summary of Qualitative Uncertainty Analysis	
Risk Assessment Stage/Major Sources of Uncertainty	**Likely Effect of Uncertainty on Risk Estimates(1)**
Hazard Identification	
Sampling data and site characterization (limited sampling data, use of in-water sediments from areas away from shore, limited time period of sampling)	+/-
Analytical error	+/-
Detection limits above the RSLs for COPCs	-
Use of EPA RSL screening values for identifying COPCs	+
Use of Surrogate Values (RSLs of COPCs with structural analogy) to Screen COPCs	+/-
Exposure Assessment	
No chemical concentration for water column – unable to evaluate surface water exposure pathway	-
Use of in-water sediment data from depths greater than most waders/swimmers would encounter	+
Assumption that contaminant burden in fish tissue originates from Presque Isle Bay	+/-
Use of default fish consumption rates to represent those of study population	+/-
Use of maximum sediment concentrations to represent EPCs for RME calculations	+
Assumption of single fish species diet	+
Extrapolation of sediment data collected in 2005 to the present	+/-
Use of dermal absorption factors for soil to represent that of sediment	+
Use of current chemical concentrations to represent future concentrations	+
Use of default values for body weight, exposure duration, etc. to represent those of the study population	+
Use of best professional judgment to determine exposure frequency for sediment contact	+/-
Effect of cooking fish on chemical concentrations not considered	+
Toxicity Assessment	
Extrapolation of toxicity factors from high to low doses	+/-
Use of surrogate toxicity parameters based on structural analogy	+/-
Use of toxicity values with low confidence levels	+
Extrapolation of toxicity factors from animal to humans	+/-
Risk Characterization	
Addition of cancer and noncancer risks by exposure pathway	+
Addition of HIs and ELCRs across COPCs	+/-
Effects of multiple chemical exposures not considered (synergism, antagonism, etc.)	+/-

Notes:

(1) + indicates potential for overestimation of risk - indicates potential for underestimation of risk.

Table 7-1: Comparison of Contaminants in Fish Tissue Samples from Presque Isle Bay to other Sampling Locations within Lake Erie				
Fish Species/Contaminant	**Current Study Range (2004-2010)**	**Study 1 (Carlson and Swachkhamer, 2000)**	**Study 2 (Perez-Fuentetaja and Lupton et. al., 2006)**	**Study 3 (Sadraddini and Ekram et. al.,1977-2007)**
	(µg/kg)	**(µg/kg)**	**(µg/kg)**	**(µg/kg)**
Bass, Smallmouth				
Total PCBs	150 – 1,200			1,043
Hg	237 - 350			173
Bass, White				
Total PCBs	260			399
Hg	115			146
Common Carp				
Total PBDE			1.5 - 100	
Total PCBs	470		16,000	236
Hg	93			154
Drum, Fresh Water				
Total PCBs				247
Hg				187
Northern Pike				
Total PCBs	< MDL			309
Hg	124			237
Salmon, Coho				
Total PCBs				749
Hg				100
Trout, Lake				
Total PCBs	380 – 1,700			463
Hg	123 - 181			118
Trout, Rainbow				
Total PCBs				220
Hg				104
Walleye				
BDE 47		32		
BDE 99		5.9		
BDE 100		7.8		
BDE 153		2.6		
BDE 154		2.4		
c – Chlordane		6.7		
t - Chlordane		5.3		
p,p – DDE	< MDL	67		
p,p – DDT	< MDL			
Dieldrin	4.78 – 9.26	12		
HCB		3.2		
Hg	156 - 261	114		122 - 199
c - Nonachlor	< MDL	6.7		
t - Nonachlor	< MDL	7.8		
OCS		5.8		
Total PCBs	56 - 270	1,241		115 -1,329
Toxaphene		189		