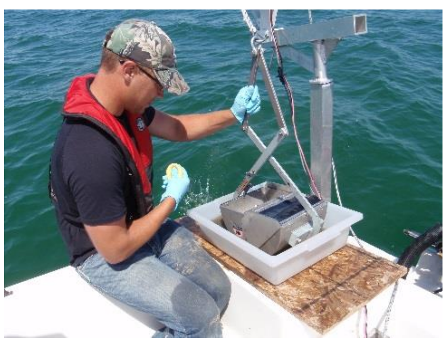
Technical Memorandum

2010 National Coastal Condition Assessment

Great Lakes

(April 2016)



This document was prepared for the Great Lakes Community to highlight the Great Lakes condition assessment using text, figures and data from the 2010 National Coastal Condition Assessment Report. Some tables, figures and content were replicated from the National Coastal Condition Assessment Report to complete this report.

Cover Photos: Top Left (MDEQ collecting sediment using a ponar), Top Right (WDNR collecting water profile data with sonde), Bottom Left (IEPA collecting water clarity information using a PAR meter), Bottom Right (MDEQ filtering chlorophyll A samples).

Acknowledgments:
This memorandum is based on and uses material from the National Coastal Condition Assessment report that had many writers, reviewers, contributors and collaborators. Additionally, we would like to thank Dave Bolgrien, Tom Davenport, Treda Grayson, Ed Hammer, Linda Harwell, Linda Holst, Paul Horvatin, Janice Huang, Jon Kiddon, Chris Korleski, Sarah Lehmann, Julie Lietz, Elizabeth Murphy, Peg Pelletier, Leanne Stahl, Hugh Sullivan, and Glenn Warren for reviewing this document. There was no additional peer review completed for this document because it is based on data and assessment approaches of the 2010 NCCA report.

Attachment Writers:
Jack Kelly, Peder Yurista, Will Bartsch, and Jo Thompson (Embayment and Cyanobacteria); Leanne Stahl (Fish Tissue); Julie Lietz (Underwater Video)

Primary Writers of this Report:
Mari Nord (EPA Region 5), Beth Hinchey (EPA GLNPO), Andrea Bolks and Will Bartsch (ORISE Interns).

Introduction

The purpose of this memorandum is to present a snapshot of the condition of the nearshore waters of the Great Lakes as sampled by the Great Lake States, the U.S. Environmental Protection Agency (EPA) and its partners during the summer of 2010. This technical memorandum focuses on the Great Lakes and complements the 2010 National Coastal Condition Assessment (NCCA) report.

The NCCA is one of five National Aquatic Resource Surveys that is managed by EPA's Office of Water in partnership with States and Tribes. The NCCA is designed to yield unbiased estimates of the condition of the nearshore waters based on a random stratified survey and to assess changes over time. In 2010, the Great Lakes was fully incorporated into the NCCA for the first time.

Design of the Great Lakes NCCA

The focus for the Great Lakes NCCA is the nearshore waters. Nearshore waters are those waters heavily used by humans and most vulnerable to activities within adjacent coastal watersheds. For the Great Lakes, the nearshore area was defined as being within 5 kilometers (3 miles) from shore and less than 30 meters (98 feet) deep. The area covered by the survey totals 17,353 square kilometers (6700 square miles). It does not include the connecting channels of the Great Lakes (e.g., St. Marys River, St. Clair River-Lake St. Clair-Detroit River system, Niagara River) or the St. Lawrence River outlet. This sample frame for the Great Lakes sites was obtained from EPA's Office of Research and Development Mid-Continent Ecology Division (MED). A more detailed explanation of the sample frame and design is available in Appendix A.

Site Selection

A probability-based sample design was used so that statistically-valid estimates of the condition of the nearshore waters of the Great Lakes could be assessed with known confidence. The original design from the Office of Water targeted a sample size of 45 sites in each Great Lake for a total of 225 sites within the United States portion of the Great Lakes. The sites were selected using a Generalized Random Tessellation Stratified (GRTS) survey design. See Appendix A for further discussion on the sample framework and design. In addition to the 225 sites, MED led an effort utilizing the funds from Great Lake Restoration Initiative (GLRI) to add a subset of sites in embayments, and the National Park Service (NPS) utilized GLRI funds to add a subset of sites in coastal waters near National Parks in Lake Michigan and Lake Superior (NPS 2014). The embayment intensification design was added because embayments are generally shallower, sheltered and may have longer water residence time compared to the overall nearshore waters. It was hypothesized that the embayments would show evidence of higher exposure to drainage basin runoff (Kelly et al. 2015). See Appendix A, sample design section, for a definition of embayment areas and Attachment A for discussion on the general findings of the embayments study.

After completion of site selection, there were 405 sites sampled in the Great Lakes during the summer of 2010. During analysis, sites were weighted proportionally to the area they represented. These sites were all used in the national report and in this Great Lakes assessment to reflect assessment of 17,353 square kilometers (6700 square miles) of nearshore area. The embayment subset represented an area of 736 square kilometers (284 square miles).

The NCCA indicators were selected to collect data relevant to the ecological condition of coastal waters and the key stressors affecting them. While most of the indicators included were initially selected when the original National Coastal Assessment (NCA)[1] focused on marine waters, they are also applicable to the Great Lakes embayments and the nearshore waters. Indicators and protocols were also selected so that a minimum of one site could be sampled in a day with a four-person crew.

At each site visit, crews collected water, sediment, fish, and underwater video footage that were sent to labs for analysis. Sediment and benthic samples were collected using a standard ponar. Water samples were collected using a Kemmerer sampler deployed 0.5 meters below the surface. A multiprobe or equivalent device was used at the site to record dissolved oxygen, pH, temperature, and conductivity measurements from the surface to 0.5 m above the bottom. Water clarity was determined using a standardized 20 cm black and white secchi disk and Photosynthetically Active Radiation (PAR) meter. Water samples were filtered with a Whatman GF/F 47 mm 0.7 micron filter for chlorophyll *a* and the resulting filtered water was used to measure dissolved nutrients. Standardized field and laboratory protocols were followed to ensure comparability of results. For further information on field and lab protocols, please see the field operations manual and laboratory operations manual for the 2010 survey (USEPA, 2010a; USEPA, 2010b).

Table 1 shows the number of sites for each lake and includes information on the number of parameters collected for assessment within each lake. The water sample collected is a grab surface sample, the sediment samples collected are ponar grabs for toxicity and chemistry, Ecofish samples collected represents fish caught for whole fish analysis, and the benthic samples collected indicates the number of successful ponar grabs of benthic invertebrate samples. Due to the endpoint used for the Great Lakes benthic indicator (Oligochaete Trophic Index (OTI)), analysts could not use benthos samples that did not have the necessary classified oligochaetes; the last column of Table 1 shows the number of samples that analysts were able to use in calculating the OTI. While there was allowance to sample benthos, sediment, and fish within a 500m radius of the designated probabilistic site, not all samples could be obtained at all sites. To clarify the number of sites and samples where crews found it challenging to collect samples we identified sites where they had not intended to collect samples but were included in the total site list. There were three sites that did not have any samples collected and nine sites that were in NPS waters where fish, sediment or benthos were not collected intentionally. Still, sediment and benthos were collected at 80% of the sites, with Lake Ontario being the most difficult lake in which to collect a substrate sample due to rocky substrates and dreissenid mussel populations. Due to the sampling season and likelihood of the targeted fish migrating towards open waters, it was challenging to collect fish in Lake Michigan, Lake Huron, and Lake Superior.

[1]From 1990-2006 the program was led by EPA Office of Research and Development and called the National Coastal Assessment.

Table 1. Sample design and sampling success.

	Number of Sites Targeted	Number (%) Water samples Collected	Number (%) Sediment samples Collected	Number (%) Ecofish samples Collected	Number (%) Benthic samples Collected	Number (%) Benthic samples Assessed
Lake Michigan	60	60 (100%)	41 (68%)	27 (45%)	45 (75%)	23 (38%)
Lake Erie	45	45 (100%)	34 (76%)	39 (87%)	34 (76%)	30 (67%)
Lake Huron	45	45 (100%)	33 (73%)	20 (44%)	33 (73%)	26 (58%)
Lake Ontario	45	45 (100%)	21 (47%)	39 (87%)	19 (42%)	13 (29%)
Lake Superior	56	56 (100%)	40 (71%)	30 (54%)	40 (71%)	19 (34%)
Embayments	154	151 (98%)	145 (94%)	127 (82%)	146 (95%)	114 (74%)
Totals	405	402 (99%)	314 (78%)	282 (70%)	317 (78%)	225 (56%)

Indices Used To Assess the Great Lakes Condition

Four primary indices were used to assess the condition of the Great Lakes: a benthic index, water quality index, sediment quality index, and ecological fish tissue contaminant index (Table 2). These indices have been used in the marine National Coastal Assessment program since 1990 and for continuity and consistency purposes were selected for the Great Lakes.

Table 2. Indicators used to assess the condition of the Great Lakes.

Benthic Indicator	Water Quality	Sediment Quality	Ecological Fish Tissue
• Benthic macroinvertebrates	• Total Phosphorus • Bottom Dissolved Oxygen • Water Clarity • Chlorophyll *a*	• Sediment Contaminants (PAHs, Metals, Pesticides, PCBs) • Sediment Toxicity	• Whole Fish Tissue Contaminants (Metals, Pesticides, PCBs)

Benthic Index

The benthic community is assessed using the same approach as used by the State of the Lakes Ecosystem Conference (SOLEC) Indicator #104 (SOLEC, 2007; EC and USEPA, 2013). The OTI based on Howmiller & Scott's (1977) index with subsequent modifications by Milbrink (1983) and Lauritsen (1985) was used. The OTI is based on the classification of oligochaete species by their known tolerance to organic enrichment (EC and USEPA, 2014). The OTI ranges from 0 to 3, where scores less than 0.6 indicate oligotrophic conditions, scores between 0.6 and 1.0 indicate mesotrophic conditions, and scores > 1.0 indicate eutrophic conditions. Table 3 presents the thresholds that are used for the condition assessment in this report. See Appendix A for further discussion on how the benthic values were calculated.

Table 3. Thresholds used to assess biological condition using benthic index.

Region	Good	Fair	Poor
Benthic Index	Oligochaete trophic index score is < 0.6	Oligochaete trophic index score is between 0.6 and 1.0	Oligochaete trophic index score is > 1.0

Water Quality Index

The water quality index includes total phosphorus, chlorophyll *a*, near-bottom dissolved oxygen, and water clarity as indicators. Total nitrogen, dissolved inorganic nitrogen, and dissolved inorganic phosphorus data were collected but were not used in this assessment. Nitrogen was excluded because it is not considered to be a limiting nutrient in the Great Lakes and there are currently no published thresholds for nitrogen impairment. Additional work may be undertaken to incorporate nitrogen results in the future. See Appendix A for further clarification.

Water quality is assessed by comparing results to available thresholds. The International Joint Commission (IJC) Phosphorus Management Strategies Task Force (PMSTF) developed thresholds for open water for different Great Lakes basins based on trophic status using total phosphorus, chlorophyll *a,* and secchi depth. The different basins are shown in Figure 1. While the thresholds were developed for open waters, data used to generate the thresholds overlap with some of the waters of the NCCA 2010 design frame. NCCA analysts and partners from the states and the Great Lakes National Program Office (GLNPO) determined that it was appropriate to apply them to the NCCA nearshore and embayment sites for this assessment. The PMSTF report (PMSTF 1980) only identified a single threshold, therefore, the lower threshold (i.e., fair to poor) was derived for the NCCA report as the value indicative of crossing into the next, more nutrient-enriched trophic status. The NCCA analysts and partners used information from a 1979 IJC Nearshore report (IJC 1979) to identify trophic status thresholds for select basins (i.e., Saginaw Bay in Lake Huron and Western, Central, and Eastern basins of Lake Erie), that were not available in the 1980 open water report (PMSTF 1980).

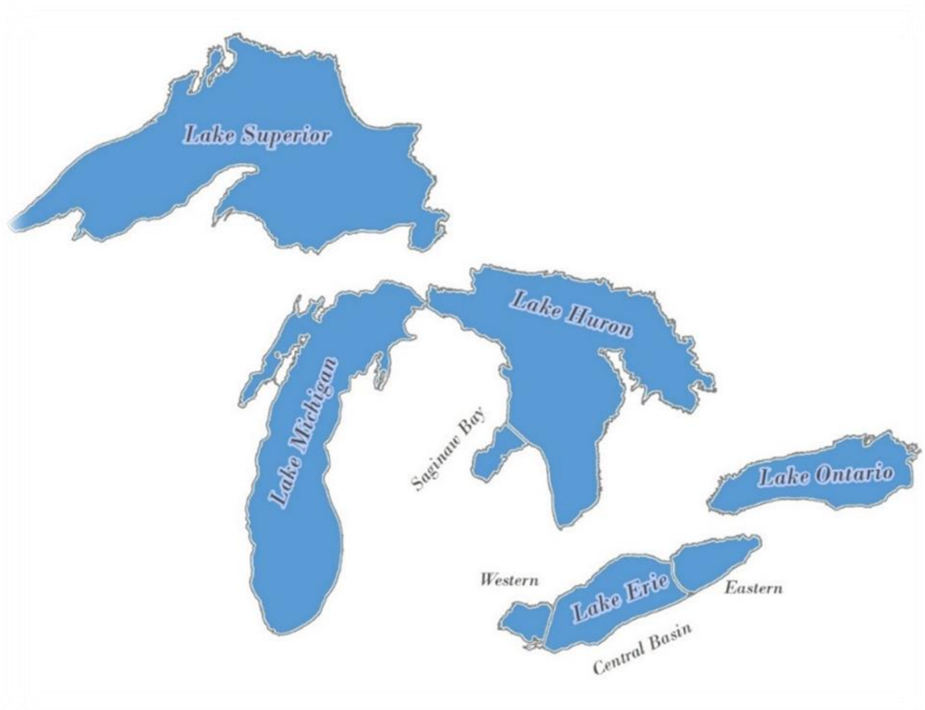


Figure 1. Great Lakes basins.

The bottom dissolved oxygen thresholds are consistent with marine water quality thresholds. Studies in the Great Lakes also support 2 mg/L to define a hypoxic condition and that value was selected as the fair-to-poor threshold (Costantini et al., 2011; Krieger and Bur, 2009). See Table 4 for a summary of the thresholds as they relate to the trophic condition of each basin.

Table 4. Water quality indicator thresholds and basin trophic condition.

Lake/Basin	**Chlorophyll *a* (ug/L)**		**Total Phosphorus (ug/L)**		**Dissolved Oxygen (mg/L)**		**Secchi Depth (m)**		**Trophic Condition**
	Good/Fair	Fair/Poor	Good /Fair	Fair/ Poor	Good /Fair	Fair/ Poor	Good/ Fair	Fair/ Poor	
Superior	1.3	2.6	5	10	5	2	8	5.3	Oligotrophic
Michigan	1.8	2.6	7	10	5	2	6.7	5.3	Oligotrophic
Huron	1.3	2.6	5	10	5	2	8	5.3	Oligotrophic
Saginaw Bay	3.6	6	15	32	5	2	3.9	2.1	Mesotrophic
Western Erie	3.6	6	15	32	5	2	3.9	2.1	Mesotrophic
Central Erie	2.6	3.6	10	15	5	2	5.3	3.9	Oligomesotrophic
Eastern Erie	2.6	3.6	10	15	5	2	5.3	3.9	Oligomesotrophic
Ontario	2.6	3.6	10	15	5	2	5.3	3.9	Oligomesotrophic

The Great Lakes assessment was conducted following the NCCA rules as listed in Table 5. If no component indicators were poor and only one was fair, then the overall index condition was assessed as good for that lake. If one component indicator was poor and two or more were fair then the overall index condition was assessed as fair for that lake. If two or more components were rated poor then the lake was given a poor condition assessment for that indicator.

Table 5. Guidelines to assess condition using water quality index.

Rank	Good	Fair	Poor
Water Quality Index	No component indicators are rated poor, and a maximum of one is rated fair.	One component indicator is rated poor, or two or more component indicators are rated fair.	Two or more component indicators are rated poor.

Sediment Quality Index

The NCCA collected surficial (i.e., top 2 cm) sediment samples and measured them for concentrations of chemical constituents, total organic carbon (TOC), and grain size. Sediment toxicity was assessed by measuring the survival of the freshwater amphipod, *Hyalella azteca* after a 10-day exposure to the sediments under laboratory conditions (USEPA, 2000; USEPA, 2010c). The results of these evaluations are used in assessing sediment condition.

For the Great Lakes, sediment contaminants were assessed using the mean Probable Effect Concentration Quotient (mPEC-Q) (Ingersoll et al., 2001; USEPA, 2002). The mPEC-Q distills data from a mixture of contaminants into one unitless index which can be compared to incidences of sediment toxicity. The mPEC-Q is the average of three PEC-Qs using only those contaminants with reliable PECs: 1) mean PEC-Q for metals; 2) PEC-Q for total PAHs; 3) PEC-Q for total PCBs. See Appendix A for PEC values and specifics on how the mPEC-Q was calculated.

The thresholds for sediment toxicity follow published values found in the National Sediment Inventory (USEPA, 2004) and the PEC quotient thresholds also follow published values. (MacDonald et. al., 2000; Crane and Hennes, 2007).

Sediment quality index condition is assessed as poor (i.e., high potential for exposure effects on biota) at a site if one of the component indicators is categorized as poor; assessed as fair if either indicator is rated fair; and assessed as good if both component indicators are at levels that would be unlikely to result in adverse biological effects due to sediment quality (summarized in Table 6). TOC and grain size are ancillary data that are available but were not used in the Great Lakes sediment condition assessment.

Table 6. Sediment quality indicator thresholds and condition assessment definitions.

Rank	Good	Fair	Poor
Sediment Quality Index	Both indicators are rated good	At least one indicator is rated fair and none are rated poor	At least one indicator is rated poor
Sediment Chemistry	mPEC-Q ≤ 0.1	0.1< mPEC-Q < 0.6	mPEC-Q ≥ 0.6
Sediment Toxicity	≥ 90% control-adjusted survival	≥ 75% control-adjusted survival	< 75% control-adjusted survival

Ecological Fish Tissue Contaminant Index

Fish were collected by crews using various methods. Crews collected fish that measured between 100 and 400 mm from a target list. If fish from the target list were not available, crews were allowed to select and submit alternative fish species. Whole-body fish tissues were assessed based on EPA's *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA 1997). This approach evaluates whether environmental concentrations of contaminants (i.e., soil, sediment, water, and tissue) pose a potential risk to fish and wildlife (receptors of concern) that consume fish. Although the guidelines were developed based on laboratory-controlled test conditions, it was selected as an approach that could still be used to evaluate contaminant-focused in-situ fish tissue quality. Threshold values were calculated using established toxicity reference values (TRVs) for predatory fish, piscivorous fish, and mammals (receptors). See Appendix A to see the TRVs associated with receptors and the no observed adverse effect level (NOAEL) and lowest observed adverse effect level (LOAEL) calculations. The fish quality assessment follows the same approach for both the region and for each lake. Table 7 lists how the region and each lake is rated into good, fair, or poor for potential risk of contaminant exposure to piscivorous fish and wildlife.

Table 7. Biotic exposure risk determination of site and regional conditions for the fish contaminant index.

	Good	Fair	Poor
Ecological Fish Contaminant Index	None of the measured contaminant concentrations exceed the lowest observed adverse effects level (LOAEL) for any receptor group.	At least one measured concentration exceeds lowest observed adverse effects level (LOAEL) for one receptor group.	At least one measured concentration exceeds lowest observed adverse effects level (LOAEL) for two or more receptor groups.

Great Lake Conditions

The Great Lakes basin ecosystem covers 765,990 kilometers^2 (295,710 miles^2), with nearly 17,702 km (11,000 mi) of shoreline, and holds 22,684 km^3 (5,442 mi^3) of water. The Great Lakes are the largest system of fresh surface water on earth, containing roughly 18 percent of the world supply.

Although part of a single system, each lake is different. Lake Superior is the largest by volume, and its basin is mostly forested and sparsely populated. The temperate, southern basin of Lake Michigan, the second largest Great Lake by volume, is among the most urbanized in the system and is home to Milwaukee and Chicago. Lake Huron is the third largest lake by volume and includes the Georgian Bay and Saginaw Bay. Lake Ontario is the fourth largest Great Lake by volume. Lake Erie, the smallest Great Lake by volume is also the shallowest, warmest and most biologically productive Great Lake. Figure 2 is the condition assessment of the Great Lakes as presented in the 2010 National Coastal Condition Report. The bars show the percentage of area within a condition class for a given index and the error bars represent 95% confidence levels.

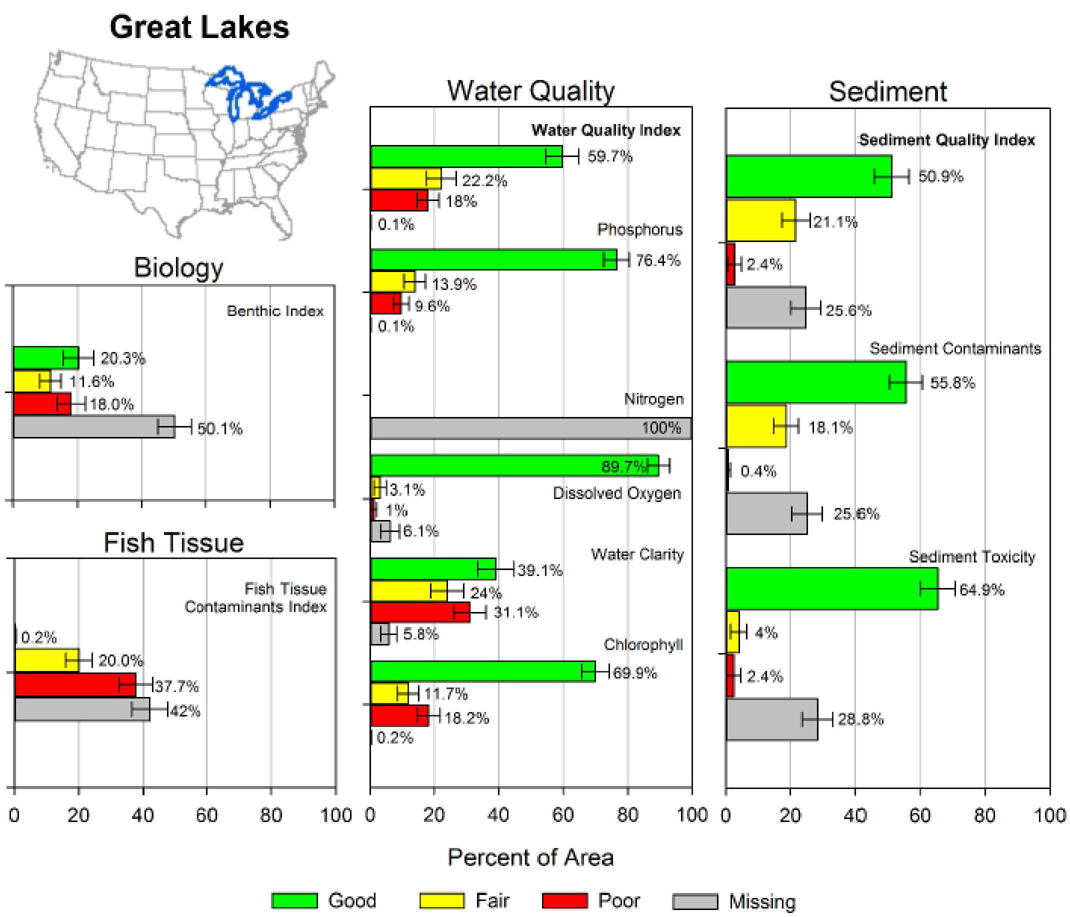


Figure 2. Great Lakes NCCA condition assessment (taken from 2010 NCCA report).

The results of the NCCA assessment of the Great Lakes nearshore and embayment area show that:

Missing Benthic Data: A noticeable impact on the assessment is the missing results that represents a fairly large spatial area of the overall assessment of the Great Lakes. In general, in the areas where sediment could not be collected there was also no benthic sample. Still, there were additional sites where there are no benthic assessment results because the appropriate oligochaete species was not present in the sample to be used in the OTI calculation.

- 60% of the Great Lakes nearshore area is in good condition based on the water quality index, 22% is fair and 18% is in poor condition.
- 51% of the nearshore sediments are in good condition based on the sediment quality index, 21% are fair and 2% are poor condition.
- 20% of the nearshore area is in good condition based on the benthic index, 12% is in fair and 18% is in poor condition. However, more than 50% of the nearshore area could not be assessed for this indicator.
- The ecological fish contaminant index shows that <1% of the nearshore is in good condition, 38% is in poor condition, 20% is in fair condition, and 42% is classified as missing due to no sample collected.

  For the ecological fish contaminant index, the contaminants that most often exceeded the LOAELs (poor) were selenium and mercury, and to lesser degrees total PCBs, total DDTs and hexachlorobenzene. It is important to note that the values used for ecological fish tissue assessment for PCBs and mercury are much higher than the human health cancer and non-cancer values used in the Great Lakes Human Health special study (see Attachment A). For example, the ecological fish tissue value for PCBs for the avian group is 1.29 ppm (or 1290 ppb) while the human health cancer value is 0.012 ppm (or 12 ppb). As such, a lower percentage of nearshore waters exceed the values established for ecological assessments; a higher percentage of waters would exceed such values if we were applying human health criteria. Additionally, the values used to assess ecological fish tissue for this study are higher than the values used by other entities in the Great Lakes region for assessment. Both the governments of Canada and the United States use the 2012 Great Lakes Water Quality Agreement (GLWQA, 2012) General Objective 9: "The waters of the Great Lakes should be free from pollutants in quantities or concentration that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain" to assess the chemicals present in whole fish tissue, essentially the NOAEL. For instance, the NCCA uses the LOAEL (1.29 ppm) for PCBs in whole fish while the Great Lakes region uses criteria similar to the NOAEL (0.013 ppm) for PCBs in whole fish. Because these targets differ, the interpretation of status differs. It is also important to note that the Great Lakes Region assesses chemicals in whole body fish mainly from the offshore of each lake, resulting in differing results and targets from the NCCA assessment.

In addition to reporting on the Great Lakes nearshore waters as a whole, the NCCA design allows for each lakes nearshore area to be assessed separately. Figure 3 shows the water quality indicators for each of the Great Lakes nearshore waters. The 95% confidence interval is included in the bar graphs and in the descriptions below, the values are provided as point estimates with the 95% confidence interval range included.

Figure 3. Nearshore waters condition assessment for each lake by water quality indicator.

Total Phosphorus: Of the five Great Lakes, Lake Erie had the highest percentage, 31 ± 8%, of nearshore area in poor condition based on phosphorus levels followed by Lake Michigan with 9 ± 4%. Lake Michigan also had the highest percentage of waters in good condition for total phosphorus at 86 ± 6% while Lake Erie had the least at 47 ± 9%.

Chlorophyll *a*: Lakes Superior, Michigan, Huron, and Ontario all had high proportions of nearshore area classified as good for chlorophyll *a*, ranging from 57 ± 10% (Lake Ontario) to 87 ± 8% (Lake Superior). Lake Erie had 17 ± 8% in good condition. Conversely, Lake Erie also had the largest percent of nearshore area in poor condition at 60 ± 11%, followed by Lake Ontario at 19 ± 9% and Lake Michigan at 15 ± 7%.

Dissolved Oxygen: All of the lakes showed predominantly good conditions based on near-bottom dissolved oxygen levels. Only Lake Erie and Lake Ontario had a proportion of nearshore area assessed in the poor range (6 ± 6% and > 1% respectively). Lake Erie also had the largest percentage of nearshore area in fair condition of the five lakes at 11 ± 7% while Lake Michigan showed 3 ± 3% in fair condition. Based on these results, it appears that low levels of dissolved oxygen are not a widespread problem in the nearshore waters of Great Lakes but should continue to be monitored.

Water Clarity: As shown in Figure 3, poor water clarity was found in all of the lakes. Again Lake Erie, the shallowest of the lakes, had the largest percent in poor condition (67 ± 10%), followed by Lake Michigan (29 ± 10%), Lake Superior (27 ± 9%) and Lake Ontario (25 ± 8%) in poor condition.

In the national report, nitrogen data are noted as missing for 100% of the Great Lakes nearshore area (see Figure 2 of this report). However, labs did analyze samples for nitrogen but the data was not assessed for the NCCA because a threshold was not available or developed in time to be used. To begin to fill that gap, this report includes information on nitrogen concentration levels in the Great Lakes nearshore. Figure 4 shows total nitrogen as percentiles at each site. The concentrations ranged from 0.17 to 4.42 mg N/L. The highest concentrations are found in western Lake Erie basin and the lowest in Lake Michigan and Lake Huron. Additional information is available in Appendix A.

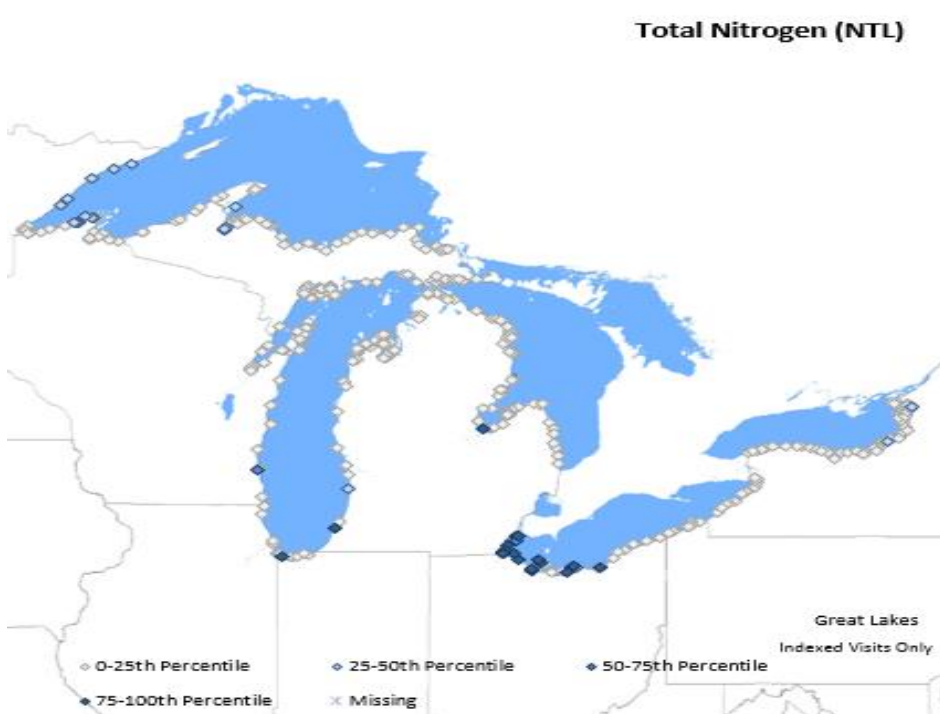


Figure 4. Total Nitrogen.

Lake Conditions

The following section describes each lake and summarizes the condition assessment of each lake using the 2010 NCCA data. The information is represented by point estimates of the area and uses data collected in both the nearshore and embayments. This section is not available in the national report and is the primary purpose of this report. For a discussion on the findings in embayments as they relate to the watershed influences, refer to the Attachment section.

Lake Superior

Lake Superior, the most northern and largest of the Great Lakes has a surface area of approximately 82,100 km^2 (31,700 mi^2) and a water volume of 12,000 km^3 (2900 mi^3). The average depth of Lake Superior is 147 m (483 ft) with a maximum depth of 406 m (1332 ft). It is at the highest elevation of the Great Lakes at 183 meters (600 ft) and its shoreline touches Canada, Minnesota, Wisconsin, and Michigan as well as several tribal nations. Major cities on Lake Superior include Duluth in Minnesota, Marquette in Michigan, and Superior in Wisconsin. Thunder Bay and Sault Saint Marie, along the Canadian border, are also areas of development along the shoreline of Lake Superior.

Lake Superior's condition assessment reflects 3056 km^2 (1180 mi^2) of nearshore area. A total of 93 sites were sampled of which 56 were base sites and 37 were embayment sites. The embayment sites represent a subset of the nearshore area, 178.7 km^2 (69 mi^2). See Figure 5 for map of Lake Superior and sample sites.

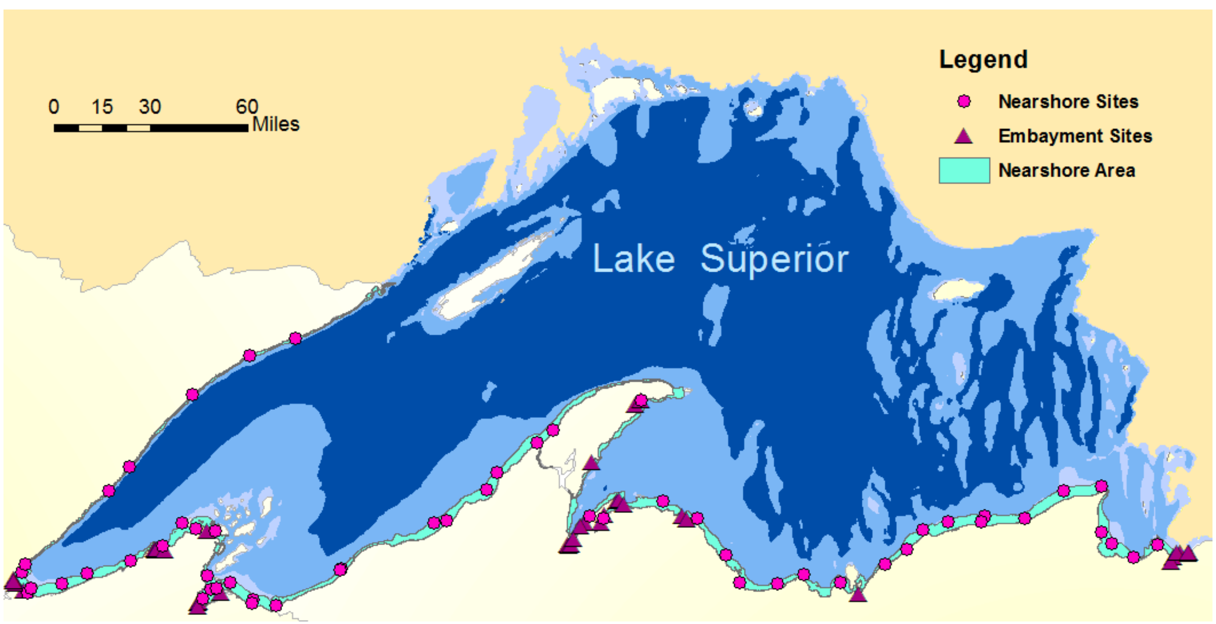


Figure 5. Lake Superior sample locations.

Figure 6 includes pie charts showing the results of the condition assessment for each index with the water quality index shown in three pie charts broken down to reflect all areas assessed, nearshore only areas, and embayment areas. For all nearshore and embayment areas of Lake Superior, water quality condition is good at over 60% and sediment quality is good at over 50% of the nearshore area. Benthos quality has missing data but of the available data, this indicator is primarily assessed as good. Sediment contaminants were the driving indicator in determining the area of fair as opposed to sediment toxicity when calculating the sediment quality index in Lake Superior. There was only one site that was evaluated as poor for both sediment toxicity and sediment contaminants. The site was near the mouth of the Portage River in Michigan. Overall, Lake Superior has the least amount of area assessed in poor condition for all indicators.

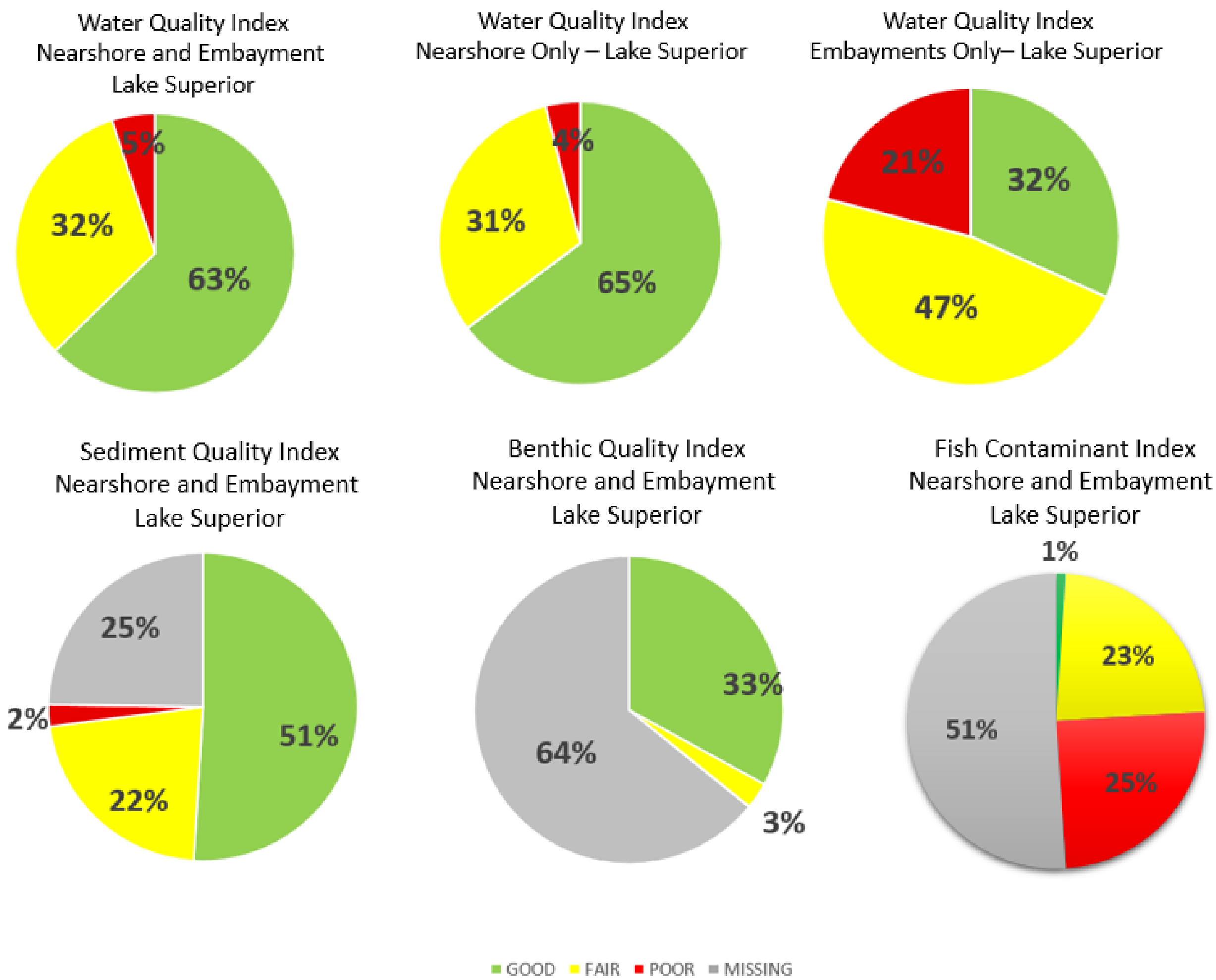


Figure 6. Pie charts of the condition assessment of each index in Lake Superior.

Lake Michigan

Lake Michigan is the only Great Lake located wholly in the United States. It has a surface area of 57,800 km^2 (22,300 mi^2) and contains 4,920 km^3 (1180 mi^3) of water volume. The average depth is 85m (279 ft) with a maximum depth of 282 m (925 ft) and an elevation just 7 m (22 ft) lower than Lake Superior. Michigan, Wisconsin, Illinois, and Indiana all share Lake Michigan waters and major cities along the shores include Chicago in Illinois, Milwaukee and Green Bay in Wisconsin, Gary in Indiana, and Benton Harbor and Traverse City in Michigan.

126 sites were sampled to assess 6951 km^2 (2684 mi^2) of Lake Michigan's nearshore area of which 60 were base sites and 66 were embayment sites. The embayment sites represents a subset of the area, 331.5 km^2 (128 mi^2). See Figure 7 for a map of Lake Michigan and sample sites.

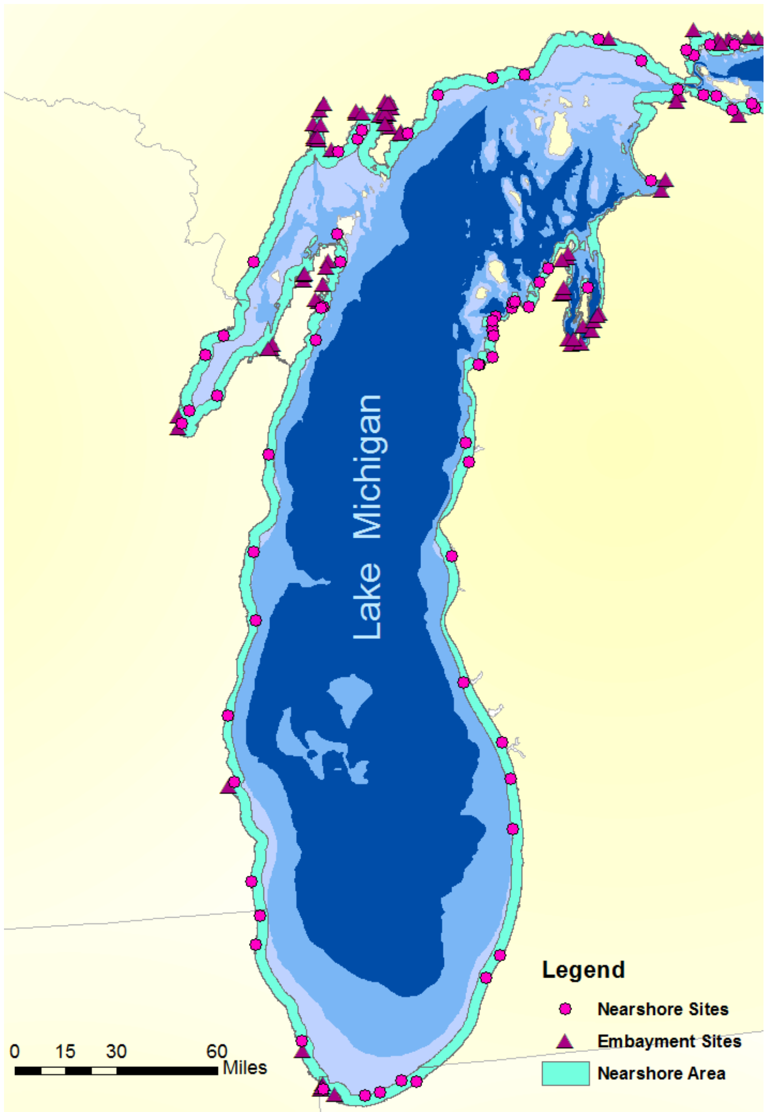


Figure 7. Lake Michigan sample locations.

Figure 8 shows the results of the condition assessment for each index with the water quality index shown in three pie charts broken down to reflect all sites, nearshore sites, and embayment sites. For all sites, water quality condition is good in nearly 70% of the nearshore area while sediment quality is good in nearly 60% of the nearshore area. Benthic index has a large amount of missing data, but of the nearshore areas that were assessed, they are equally distributed between good, fair, and poor. The majority of sites that were assessed as poor were in the embayments. Sediment contaminants were the driving indicator in determining the area of fair as opposed to sediment toxicity. When calculating the sediment quality index in Lake Michigan, there was only one site that was evaluated as poor for both sediment toxicity and sediment contaminants and it was in Indiana Harbor. A site that was assessed as poor for benthic and sediment toxicity was found near the shores of Calumet Park in Indiana.

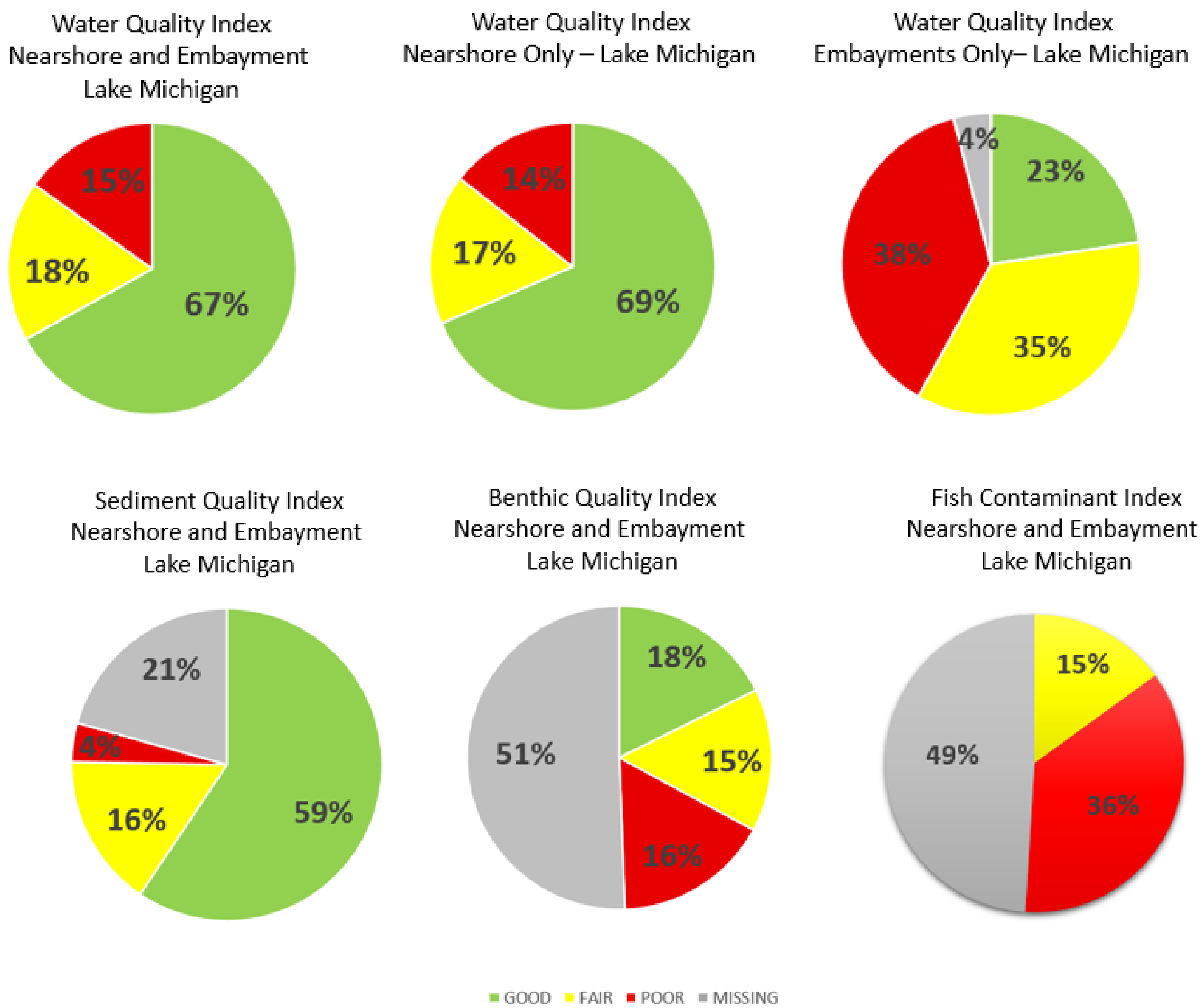


Figure 8. Pie charts of the condition assessment of each index in Lake Michigan.

Lake Huron

Lake Huron has a surface area of 59,600 km^2 (23,000 mi^2) and contains 3,540 km^3 (850 mi^3) of water. The average depth is 59 m (195 ft) and maximum depth of 229 m (750 ft). It has the same elevation as Lake Michigan. Major cities along the shoreline of Lake Huron include Bay City, Alpena, and Port Huron in Michigan and Owen Sound and Sarnia on the Canadian shoreline.

A total of 67 sites were sampled to assess 3248 km^2 (1254 mi^2) of Lake Huron's nearshore (45 base sites and 22 embayment sites). The embayment sites represents a subset of the area, 80.3 km^2 (31 mi^2). See Figure 9 for a map of Lake Huron and sample sites.

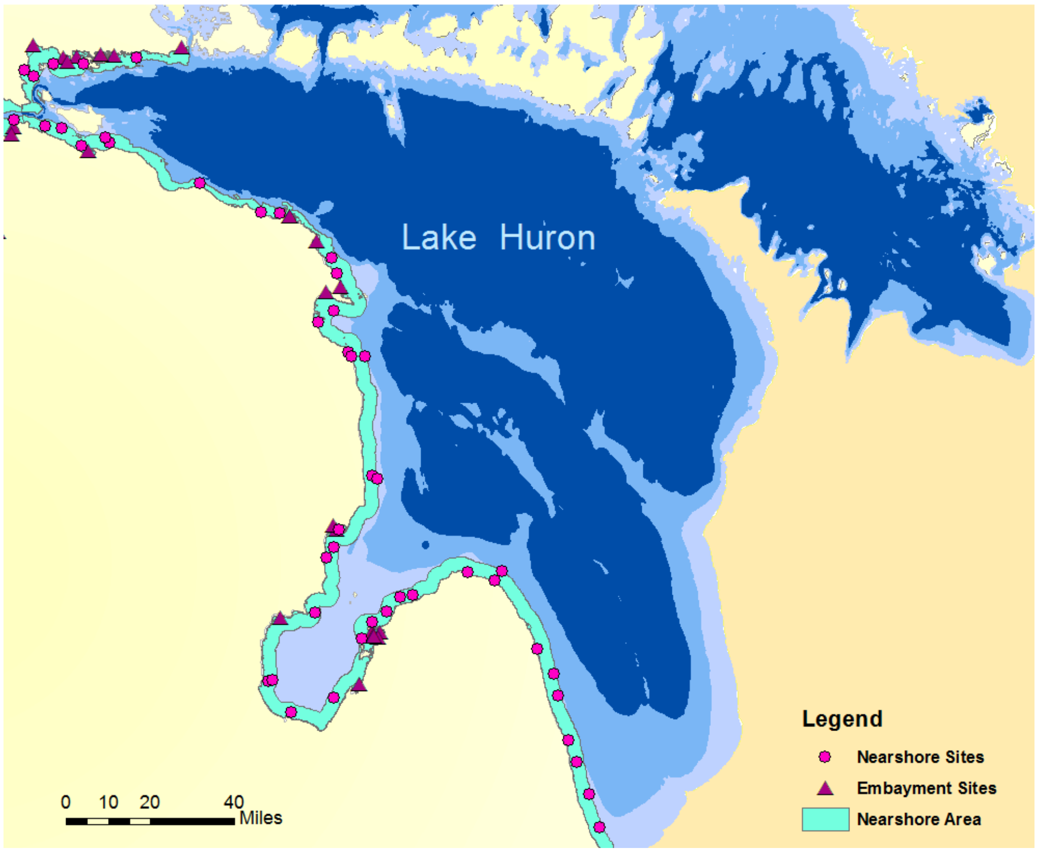


Figure 9. Lake Huron sample locations.

Figure 10 includes pie charts showing the results of the condition assessment for each index with water quality index shown in three pie charts broken down to reflect all sites, nearshore sites and embayment sites. Water quality condition is good at nearly 80% of the nearshore area while sediment quality is good at nearly 70% of the nearshore area and 0% of the area was assessed as poor. Benthic index has a large amount of missing data but of the remaining results, the areas generally assessed are equally distributed between good, fair, and poor. There was a site assessed as poor for toxicity in Lake Huron near Cheboygan otherwise the sediment quality is in fair or good condition for Lake Huron.

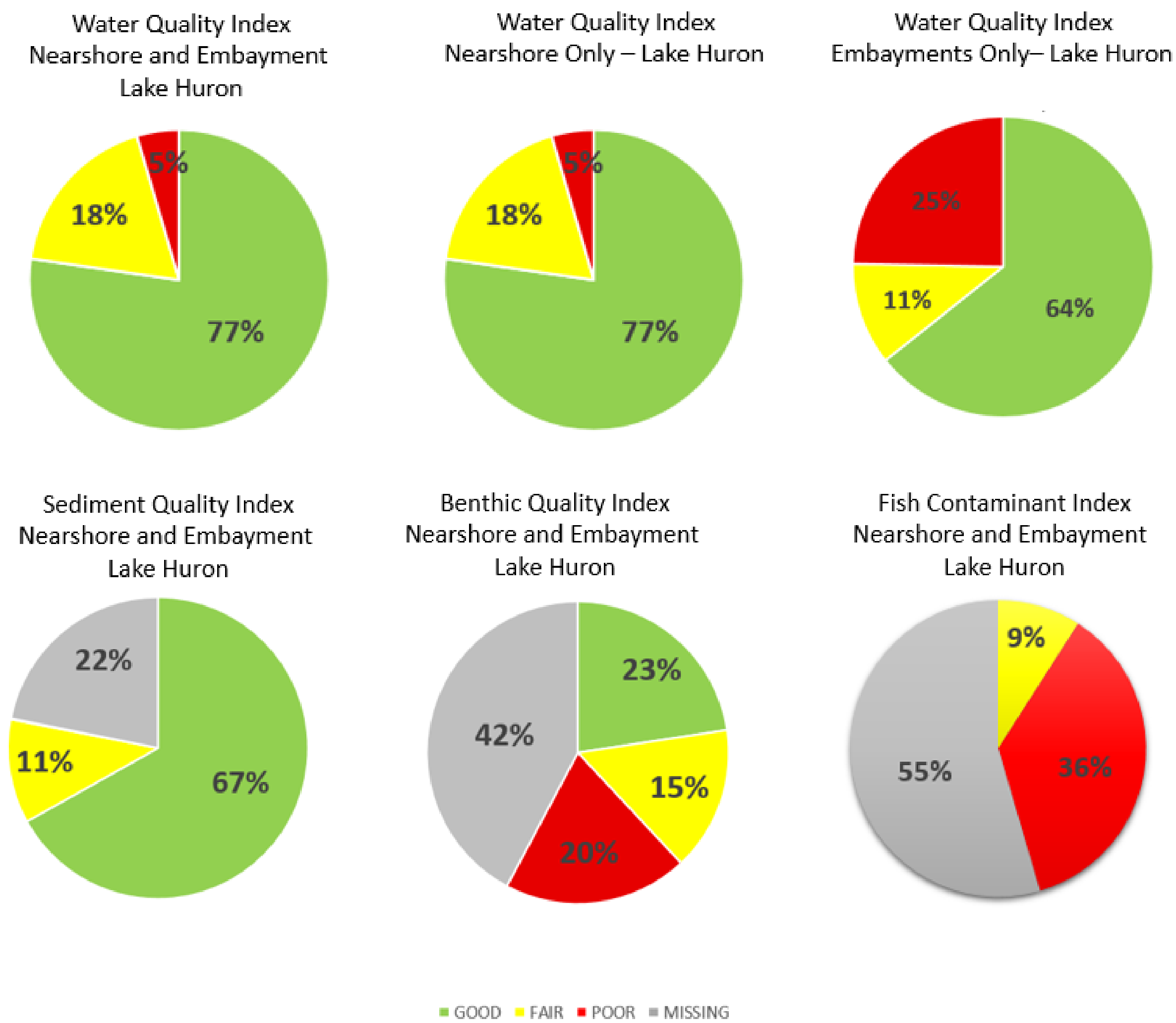


Figure 10. Pie charts of the condition assessment of each index in Lake Huron.

Lake Erie

Lake Erie has a surface area of 25,700 km^2 (9,910 mi^2) and a water volume of 484 km^3 (116 mi^3). The average depth is 19 m (62 ft) and is the shallowest of the Great Lakes with a maximum depth of 64 m (210 ft). There are major cities along the shoreline including Buffalo, New York; Cleveland, Ohio; Toledo, Ohio; and Erie, Pennsylvania.

Figure 11 is a map of the 57 sites sampled of which 45 were base sites and 12 were embayments. These sites represent an area of 2670 km^2 (1031 mi^2) of the US nearshore coastal area. The embayment sites represents a subset of the area, 57.5 km^2 (22.2 mi^2). Additionally, 12 sites were sampled using the NCCA sample design in Canadian waters by the Department of Biological Sciences from the University of Windsor. Canadian data are included in one of the water quality condition pie charts in Figure 12. It should be noted when looking at the pie chart that the 12 Canadian sites were weighted to represent the entire Lake Erie Canadian nearshore waters and therefore are heavily weighted. The results may differ if more sites had been sampled.

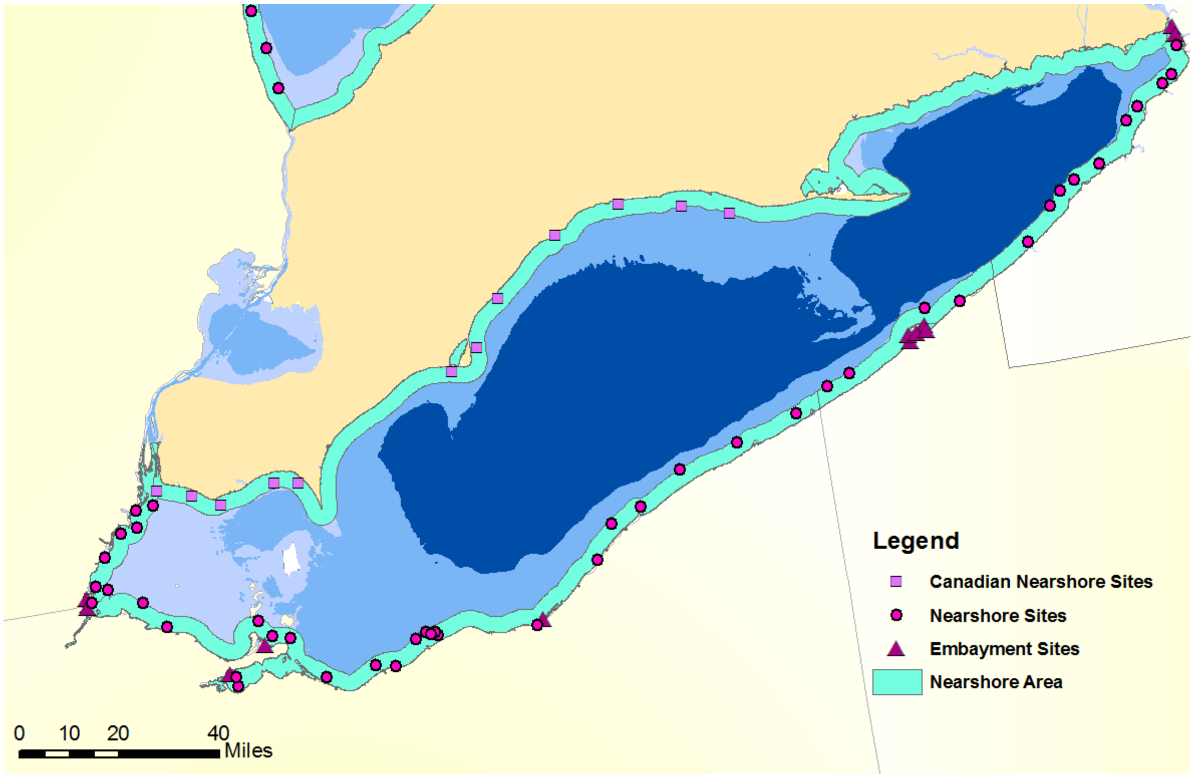


Figure 11. Lake Erie sample locations for both US locations and Canadian locations.

Figure 12 contains pie charts showing the results of the condition assessment for each index with water quality index shown in four pie charts broken down to reflect all sites including Canadian sites, all US sites, US embayment sites, and US nearshore sites. All US sites water quality condition is assessed as good for 20% of the nearshore area and poor at just over 50% of the nearshore area. Sediment quality is good at nearly 30% of the nearshore area and fair at over 40% of the nearshore area. For the Benthic index, 30% of the area is unassessed due to missing samples, 46% of the area is poor, and 11% is good. One site was assessed as poor for sediment quality and benthic quality near Cleveland, Ohio.

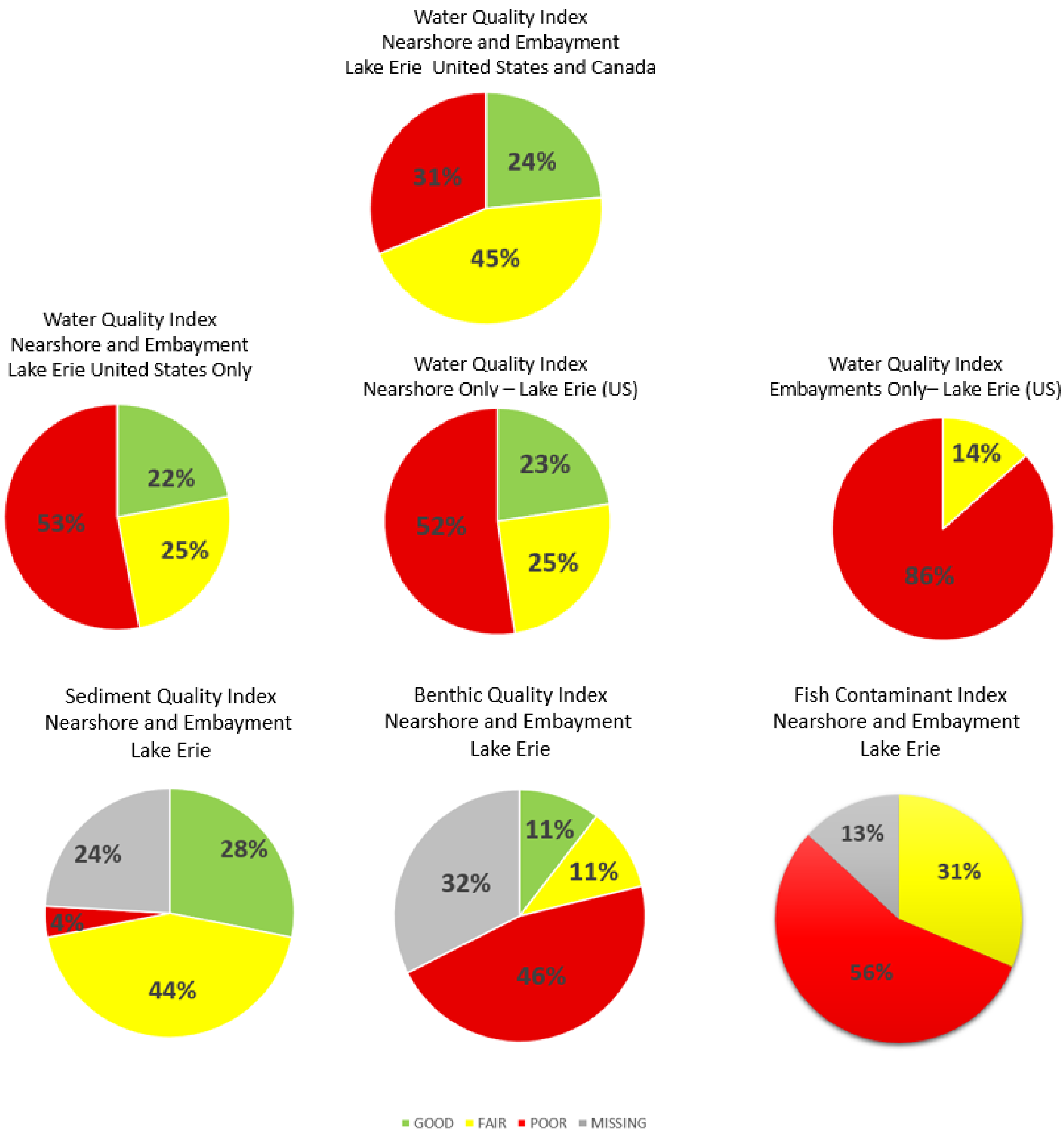


Figure 12. Pie charts of the condition assessment of each index in Lake Erie including a Water Quality assessment with Canadian site results.

Lake Ontario

Lake Ontario has the smallest surface area of the Great Lakes at 18,960 km^2 (7,340 mi^2) but holds more water than Lake Erie with a water volume of 1,640 km^3 (393 mi^3). The average depth is 86 m (283 ft) with a maximum depth of 244 m (802 ft). Lake Ontario is at the lowest elevation of the Great Lakes. It is approximately 100 m (330 ft) lower than Lake Erie. Major cities along Lake Ontario include Rochester, New York on the US side and Toronto on the Canadian side.

Lake Ontario had 62 total sites that were sampled to assess 1373 km^2 (530 mi^2) of nearshore area. There were 45 NCCA base sites and 17 embayment sites. The embayment sites represents a subset of the area, 88.1 km^2 (34 mi^2).

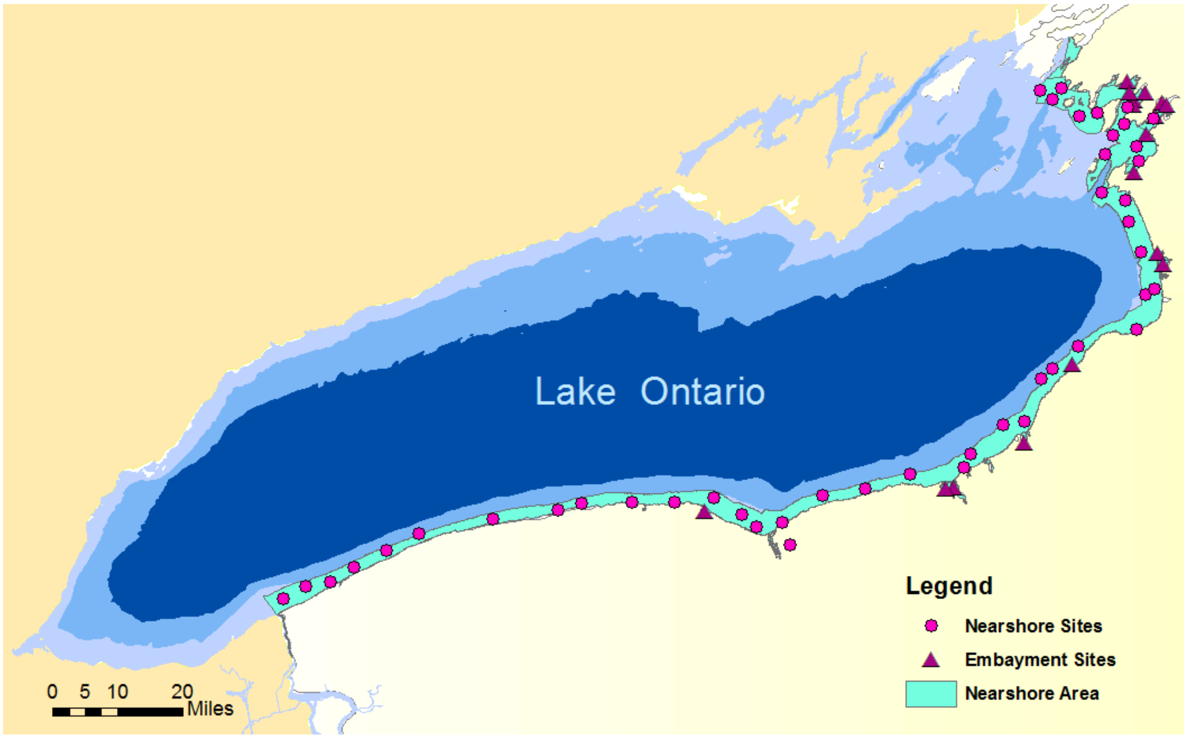


Figure 13. **Lake Ontario sample locations.**

Figure 14 contains pie charts showing the results of the condition assessment for each index with water quality index shown in three pie charts broken down to reflect all sites, embayment sites, and nearshore sites. Water quality condition is good in over 60% of the nearshore area. Over half of the nearshore area was not assessed for sediment quality due to missing samples. Of the remaining area, the sediment was assessed as good at 16% and fair at 33%. Benthic condition was not assessed at nearly 70% of the nearshore area and of what was assessed, 19% fell into the good category, 5% was fair, and 7% was poor. There were no sites assessed as poor for sediment quality in Lake Ontario. Eight sites were assessed as poor for benthic of which five were in embayments.

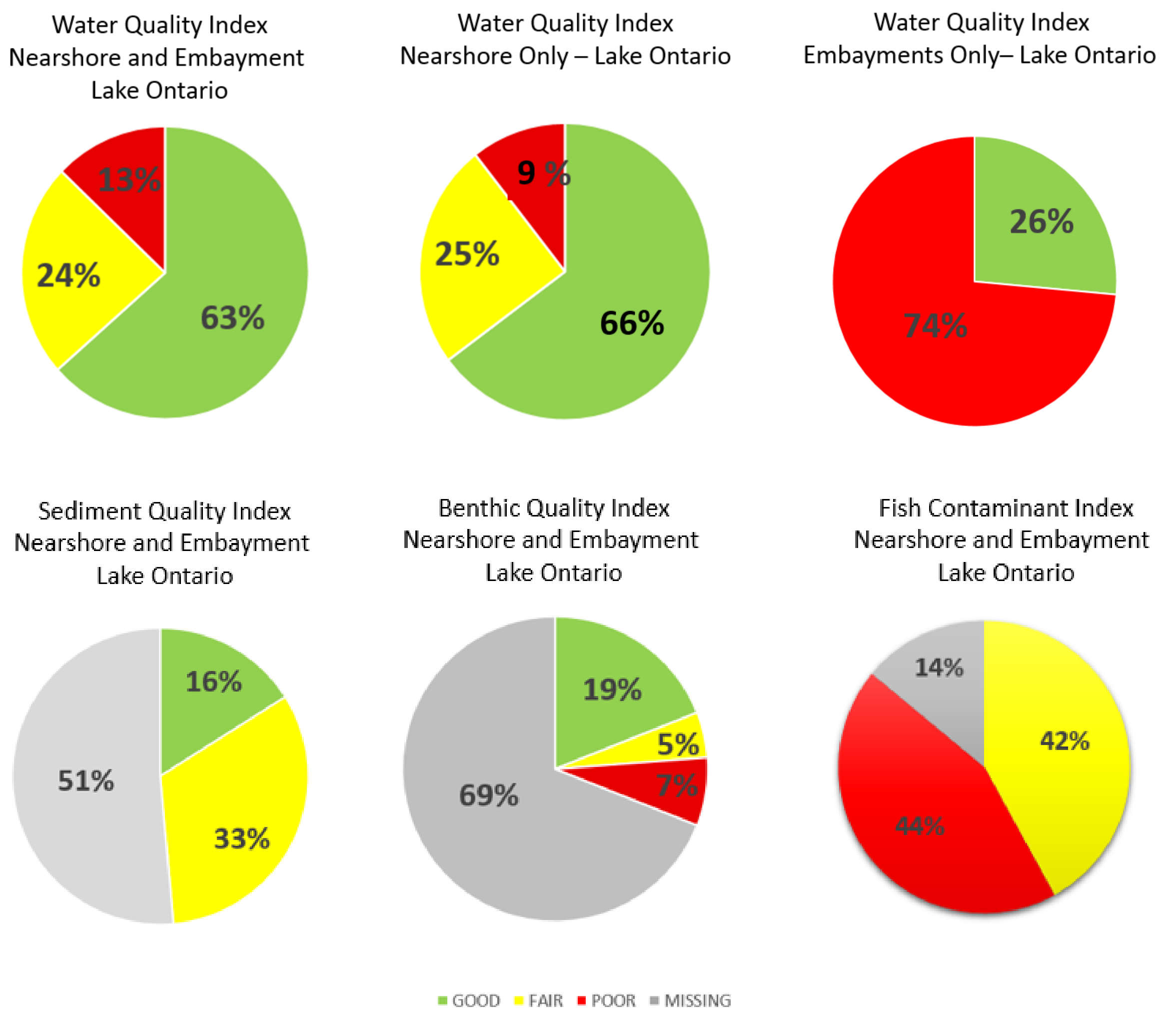


Figure 14. Pie charts of the condition assessment of each index in Lake Ontario.

Summary

The probabilistic design and comparable methods of data collection make this survey a useful tool to assess the general conditions of the Great Lakes nearshore waters. To get a sense of how the lakes compare to each other, the following is a summary of the lakes ranked for overall condition by the different indexes.

The water quality index is ranked for both all nearshore waters and for embayments. The water quality index for all sites showed that Lake Erie had the largest percentage of area assessed as poor and Lake Huron had the largest area assessed as good. Lake Michigan and Lake Superior were fairly comparable. In ranking the lakes for the water quality index, Lake Huron ranks highest in condition followed in rank order by Lake Superior, Lake Michigan, Lake Ontario, and Lake Erie. Regarding the water quality index in embayments, Lake Erie and Lake Ontario had the highest area assessed as being in poor condition followed by Lake Michigan. Lake Superior has a larger area assessed as fair but slightly less of an area assessed as poor compared to Lake Huron.

Rankings for sediment, benthic and fish are difficult due to the amount of area not assessed. There was approximately 20-25% of each lake that was not assessed for sediment quality and Lake Ontario had over 50% of the lake area not assessed for sediment quality. Lake Huron and Lake Ontario did not have any areas assessed as poor for sediment quality and Lake Erie had the lowest amount of nearshore area assessed in good condition for sediment quality. Consequently, for the sediment quality index the Lakes were ranked from highest to lowest as follows: Lake Huron, Lake Michigan, Lake Superior, and Lake Erie. Lake Ontario was omitted from the ranking because it had a much larger area that was not assessed relative to the other lakes. The benthic indicator results were also difficult to rank by lake because of the amount of area that could not be assessed. But upon visual inspection of the benthic indicator results, Lake Erie did not have any area assessed as good and Lake Huron had very little area assessed as good, as opposed to Lake Superior which had very little area assessed as poor. As a result, for the benthic indicator Lake Superior was ranked highest followed by Lake Michigan and Lake Ontario then Lake Huron and Lake Erie. The fish indicator was not ranked as fish were difficult to collect in Lake Michigan, Lake Superior, and Lake Huron and the results from the sites that could be sampled are very similar among the lakes.

Additional Studies

There were several enhancements conducted in the Great Lakes during the 2010 NCCA. Several of these enhancements were funded through the Great Lakes Restoration Initiative and also resulted from the States interest in furthering their nearshore monitoring programs.

U.S. EPA ORD MED, with the support of the Great Lakes National Program Office (GLNPO) and GLRI funding, enhanced the Great Lakes component of the NCCA by adding an embayment study (Kelly et al., 2015). The results from the embayment study were incorporated into the national report as well as this report. In addition to the embayment work, MED enhanced the project by collecting data using underwater video cameras (Lietz et al., 2015; https://gispub.epa.gov/NCCA/) and collecting an additional water sample at each site that was analyzed for phytoplankton. See the Attachments section for more information on these studies.

Among the States, Wisconsin Department of Natural Resources (WDNR) added several sites and incorporated the results into their annual nearshore water quality monitoring effort and report. Illinois Environmental Protection Agency (IEPA) requested that the sites for the state's probabilistic annual sample design be incorporated into the NCCA design. This allowed IEPA to continue with their annual sampling plan and incorporate the national sites into their existing fieldwork. All states also volunteered to collect additional underwater footage and an additional water sample to be analyzed for phytoplankton at the request of MED (as described in the Attachments section).

Additionally, the EPA Office of Science and Technology (OST) used GLRI funds to have a subset of sites sampled to the support the human health fish tissue study. See the Attachments section for a summary of the study. Additionally the National Park Service (NPS) used GLRI funds to sample NPS sites in Lake Superior and Lake Michigan (see National Park Service 2014 for more information.

These studies were made possible due to the collaboration of different offices, agencies, and states and maximized the amount of information collected while in the field, thereby minimizing collective costs. These projects were deemed successful and have been included in the 2015 NCCA sampling effort.

Next Steps

As the analysis of the 2010 NCCA samples was being conducted, some gaps and methodological shortcomings were discovered. Many of these gaps were addressed in the 2015 NCCA field year by the collection of additional parameters. For example, Microcystin and other algal toxins and fish tissue plugs for mercury were be added to the suite of parameters collected. The algal toxin list includes analysis of anatoxin-A, cylindrospermopsin, microcystin-LA, microcystin-LF, microcystin-LR, microcystin-LW, microcystin-LY, microcystin-RR, microcystin-YR, microcystin-WR, microcystin-HtYR, and microcystin-HiLR.

In 2010, the Great Lakes connecting channels were not included in the national design framework. To attempt to fill this gap, GLNPO funded a pilot effort using GLRI funds to sample the Huron-Erie Corridor (HEC) that includes the St. Clair River, Lake St. Clair, and the Detroit River in 2014. Due to the success of this pilot effort, the HEC was sampled again in 2015 and the St. Mary's River, which connects Lake Superior and Lake Huron, was sampled in 2015. The connecting channel work will be included as a highlight within the 2015 National Coastal Condition Report.

Another area of interest within the Great Lakes has been Lake Erie. In 2015, at the request of USEPA Water Division, 34 additional sites were added to the Lake Erie design frame to support analyses by the Nutrients Annex of the Great Lakes Water Quality Agreement. The additional sites allow for the assessment of the different Lake Erie sub basins with respect to water quality and phytoplankton metrics.

Because of the prevalence of missing benthic information that resulted from absence of oligochaetes and/or challenging substrates for ponar sampling, it was determined that review of alternate benthic condition assessment tools would be useful for the NCCA Great Lakes samples. As a result, a work group has been formed to review and possibly develop a different approach to assessing benthic data that would incorporate benthic species in addition to oligochaetes, and include the presence of invasive benthic organisms. GLNPO is leading this workgroup.

In an attempt to increase the likelihood of fish collection in 2015, the area where fish can be collected was extended out to 1000 m (3300 ft) from the original 500 m.

In 2015, the NCCA enhancements (embayment work, video footage, phytoplankton, and human health fish tissue work) were again added to the NCCA and funded through GLRI. MED is the lead on the embayment, video footage, and phytoplankton enhancements that will be highlighted in the 2015 National Coastal Condition Report. Additionally, OST will manage the data analysis and assessment of the human health samples and will include a highlight within the 2015 National Coastal Condition Report.

References

Costantini, M., Kolesar, S., Ludsin, S. A., Mason, D. M., Rae, C. M., & Zhang, H. (2011). Does hypoxia reduce habitat quality for Lake Erie walleye? A bioenergetics perspective. *Canadian Journal of Fisheries and Aquatic Sciences*, 68(5), 857-879.

Crane, J., & Hennes, S. (2007). Guidance for the use and application of sediment quality targets for the protection of sediment–dwelling organisms in Minnesota. Minnesota Pollution Control Agency. MPCA Document: tdr-gl-04.

Environment Canada and U.S. Environmental Protection Agency (EC and USEPA). (2013). *State of the Great Lakes 2011.* Cat No. En161-3/1-2011E-PDF. EPA 950-R-13-002. Available at http://binational.net

Great Lakes Water Quality Agreement (GLWQA). (2012). Protocol amending the agreement between Canada and the United States of America on Great Lakes water quality, 1978, as amended on October 16, 1983 and on November 18, 1987. Available at http://www.ijc.org/en_/great_lakes_water_quality#sthash.ryne4kfb.dpuf

Howmiller, R. P., & Scott, M. A. (1977). An environmental index based on relative abundance of oligochaete species. *Journal of Water Pollution Control Federation*, 49, 809-815.

Ingersoll, C. G., MacDonald, D. D., Wang, N., Crane, J. L., Field, L. J., Haverland, P. S., & Smorong, D. E. (2001). Predictions of sediment toxicity using consensus-based freshwater sediment quality guidelines. *Archives of Environmental Contamination and Toxicology*, *41*(1), 8-21.

International Joint Commission (IJC). (1979). *Trophic characterization of the US and Canadian nearshore zones of the Great Lakes*. Available at www.ijc.org/files/publications/ID530.pdf

Kelly, J. R., Yurista, P. M., Starry, M., Scharold, J., Bartsch, W., & Cotter, A. (2015). Exploration of spatial variability in nearshore water quality using the first Great Lakes National Coastal Condition Assessment survey. *Journal of Great Lakes Research*, 41(4), 1060-1074.

Krieger, K. A., & Bur, M. T. (2009). *Nearshore hypoxia as a new Lake Erie metric*. Lake Erie Protection Fund Project SG334-07. Available at http://lakeerie.ohio.gov/Portals/0/Closed%20Grants/small%20grants/SG%20334-07%20Final%20Report.pdf

Lauritsen, D. D., Mozley, S. C., & White, D. S. (1985). Distribution of oligochaetes in Lake Michigan and comments on their use as indices of pollution. *Journal of Great Lakes Research*, *11*(1), 67-76.

Lietz, J. E, Kelly, J. R., Scharold, J. V., Yurista, P. M. (2015). Can a rapid underwater video approach enhance the benthic assessment capability of the National Coastal Condition Assessment in the Great Lakes? *Journal of Environmental Management,* 55(1), 1446-1456.

MacDonald, D. D., Ingersoll, C. G., & Berger, T. A. (2000). Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology*, *39*(1), 20-31.

Milbrink, G. (1983). An improved environmental index based on the relative abundance of oligochaete species. *Hydrobiologia,* 102(2), 89-97.

National Park Service (2014) http://www.nature.nps.gov/water/crystalclear/assets/docs/GRLA_Crystal_Clear_Brief.pdf

Phosphorus Management Strategies Task Force (PMSTF). (1980, July). *Phosphorus management for the Great Lakes: Final report of the Phosphorus Management Strategies Task Force to the International Joint Commission's Great Lakes Water Quality Board and Great Lakes Science Advisory Board*. Available at http://agrienvarchive.ca/download/P-management_G_lakes.pdf

State of the Lakes Ecosystem Conference (SOLEC). (2007). *State of the Great Lakes Draft, Indicator #104*.

U.S. Environmental Protection Agency, Environmental Response Team (USEPA). (1997). *Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments – Interim final* (EPA/540/R-97/006). Edison, NJ: Author.

U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology (USEPA) (2000). *Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates* (2nd ed., EPA 600-R-99-064). Washington, DC: Author.

U.S. Environmental Protection Agency, Great Lakes National Program Office (USEPA). (2002). Interpretation of the results of sediment quality investigations. In *A Guidance Manual to Support the Assessment of Contaminated Sediments in Freshwater Ecosystems* (Vol. III, EPA-905-B02-001-C). Chicago, IL: Author.

U.S. Environmental Protection Agency, Office of Science and Technology, Standards and Health Protection Division (USEPA). (2004). *The incidence and severity of sediment contamination in surface waters of the United States, National Sediment Quality Survey* (2nd ed., EPA-823-R-04-007). Washington, DC: Author.

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010a). *National Coastal Condition Assessment: Field operations manual* (EPA-841-R-09-003). Washington, DC: Author.

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010b). *National Coastal Condition Assessment: Laboratory methods manual* (EPA 841-R-09-002). Washington, DC

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010c). *National Coastal Condition Assessment: Quality assurance project plan 2008-2012* (EPA/841-R-09-004). Washington, DC: Author.

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010d). *National Coastal Condition Assessment: Site evaluation guidelines*. U.S. Environmental Protection Agency, Washington, DC.

World Health Organization. 2003. Guidelines for safe recreational water environments Volume 1: Coastal and fresh waters. http://www.who.int/water_sanitation_health/bathing/srwe1-chap8.pdf

ATTACHMENTS

Watershed Influence on Open Nearshore Waters and Embayments of the U.S. Great Lakes Coastal Zone

In the NCCA 2010 survey design for the Great Lakes, two aquatic resource classes are defined. The first is a nearshore population of waters extending from the shoreline to an outer boundary (as far as 3.1 miles from shore or as deep as the 98 foot depth contour, whichever is reached first). A second resource class is small embayments — semi-enclosed areas formed by the configuration of the shoreline, tucked in along the shore, and perhaps more vulnerable to land drainage. Embayment areas are a small portion of the nearshore zone totaling 284 square miles, compared to 6700 square miles of U.S. nearshore coast.

NCCA sampling for the Great Lakes Region as a whole was conducted at 251 open nearshore sites and an additional 154 sites in embayment areas. Embayments were expected to show evidence of higher exposure to land drainage because of more sheltered conditions, shallower waters, perhaps longer residence times, and in general less dilution than the more "open" nearshore. Statistical analyses demonstrated that embayments had higher phosphorus concentrations, lower bottom-water dissolved oxygen concentrations, shallower measured secchi depth and a faster light extinction than the open nearshore. There was no difference between embayment and nearshore chlorophyll *a* and nitrogen levels. It may be that more turbidity (from suspended solids loading and/or wind-driven sediment resuspension in shallower waters in embayments) inhibits plankton growth slightly, in spite of a nearly doubled phosphorus concentration on average.

To assess the potential influence of watershed disturbance upon observed coastal conditions, analysts compared 2010 NCCA data and watershed disturbance data at appropriate lake-basin scales for each of the five Great Lakes (Figure A-1). This analysis provides evidence of a strong relationship between phosphorus and agricultural intensity in the watershed. The pattern for embayments reflects generally higher phosphorus concentrations at each level of watershed agricultural intensity, compared to an equivalent nearshore. The pattern peaks sharply towards the highest levels of agricultural development in Lake Erie, especially its western basin. Water quality changes and associated increasing plankton blooms have been seen in the recent past decade in this basin.

Further analyses of these NCCA data will study the watershed drivers at play, since the effects of agricultural development, human population growth and urban coastal development, forested area declines, and other factors are not fully independent of each other. Nonetheless, the results suggest that embayments are indeed more vulnerable to landscape-derived stressors than the more open nearshore zone and may be more sensitive water quality sentinels.

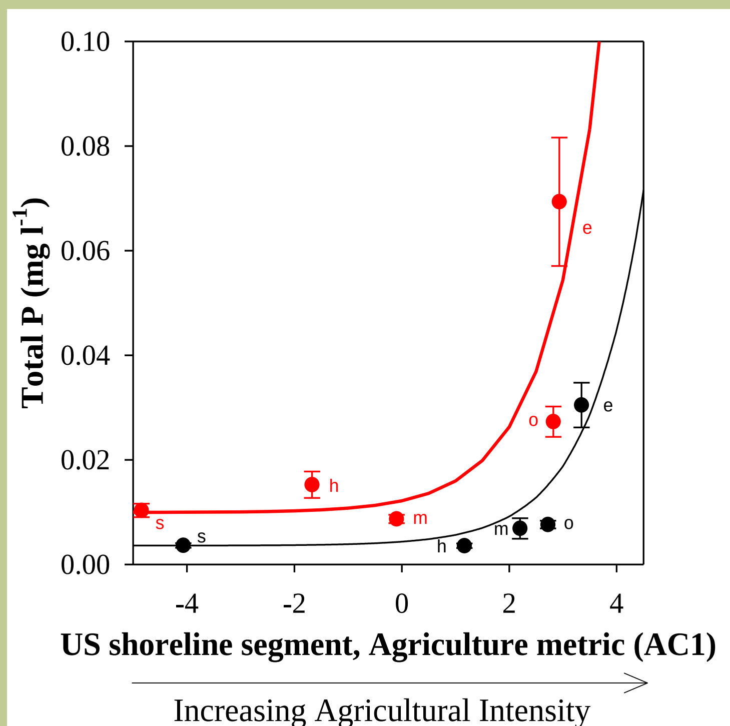


Figure A-1. Total Phosphorus as a function of increasing agricultural intensity shows that embayments have higher levels of TP than nearshore areas. Mean and 95% confidence Intervals are shown for NCCA 2010 results by lake (identified by first letter of lake name), for nearshore (in black) and embayment populations (in red). From Kelly et al. 2015.

An Underwater View

The Potential Utility of Video Sampling in Assessing Coastal Condition

In 2010, researchers added underwater video to the NCCA sampling protocol in the Great Lakes to evaluate whether video can supplement traditional benthic sampling. Video sampling is simple and can provide some rapid visual feedback, an historical archive, and sometimes a different perspective on local conditions. More specifically, researchers expected that video sampling would accurately show the presence or absence of invasive species such as dreissenid mussels (zebra and quagga mussels) and round gobies (bottom dwelling fish). These invasives can cause ecological changes that affect coastal condition.

Traditionally, deep water benthic sampling is conducted using a grab sampler such as a PONAR or Ekman dredge lowered from a boat to the bottom of the waterbody. Processing benthic grab samples takes time and expertise. Grab samplers are also limited to sampling soft sediments such as sand, silt, clay, or mud. For Great Lakes video sampling, a SeaViewer Sea-Drop color camera 950 with Unified Sea-Light™ LED light and Sea-DVR: Mini Digital Video Recorder were used. Once a clear image of the station bottom was observed on the Sea-DVR screen, researchers held the camera as still as possible and began recording. Recording duration was at least one minute, and 309 videos were collected.

Some of the findings of this video sampling pilot in the Great Lakes include the following:

- Video sampling can be more effective than ponar in detecting mussels on rocky substrate that is not amenable to ponar or dredge grab sampling (Figure A-2).

- When mussel or vegetation distribution is variable or dispersed, video sampling can provide a better estimate of the presence or absence of mussels and vegetation than a single ponar grab at that site (Figures A-3 and A-4).

- Video sampling detected dreissenid mussels at 8 sites where ponar sampling did not, and at 17 sites were ponar sampling was unsuccessful. At 32 sites, video sampling did not detect dreissenids although ponar data showed they were present. Video was better able to detect mussels as their abundance increased.

- 45% of videos were rated as marginal or poor in quality, either because of controllable reasons such as the view not being close enough to the bottom, or because of uncontrollable reasons such as poor visibility due to high suspended solids or chlorophyll *a* concentrations.

- Since video sampling provides a broad site view, occasionally researchers were treated to glimpses of passing native fish (Figure A-5).

- To get a broader understanding of a sample site's dreissenid and vegetation density and distribution, taking a video in addition to a ponar sample provides more comprehensive data.

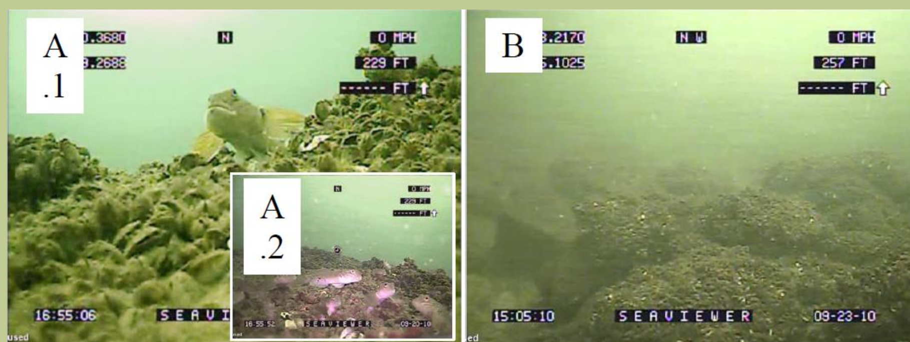
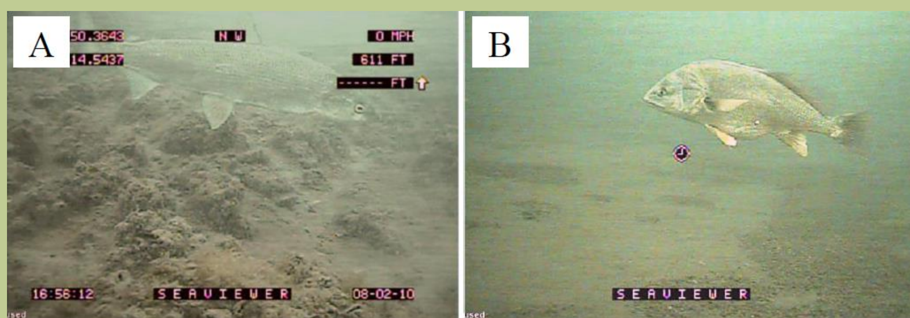


Figure A-2. Video screen shots from sites in Lake Ontario where ponar sampling was not successful. Site A was colonized by dreissenid mussels (A.1) and round gobies (A.2). Site B shows large rocks encrusted with dreissenid mussels, a substrate not effectively sampled by PONAR.

Figure A-3. A video screen shot in Lake Huron where ponar sampling collected no dreissenid mussels. Dreissenid mussels are visible on the left-hand side of the image.

Figure A-4. Video screen shots from sites in Lake Huron (A) and Lake Michigan (B & C) where PONAR sampling collected no vegetation. Sites A & C had patchy vegetation on sediment, while Site B had vegetation only on rock.

Figure A-5. Video screen shots of a lake whitefish in Lake Huron (A) and a freshwater drum in Lake Ontario (B).



Conclusion

It is evident that ponar and video sampling have unique strengths and weaknesses as sampling techniques; when paired together, they are complementary and appear to provide a more complete benthic dataset that can be used for purposes beyond the NCCA analysis.The "landscape" perspective provided by video sampling creates a useful visual archive. In fact, having images from the same area over time could be a new way for researchers to document change in coastal resources (Leitz et al. 2015).

The Great Lakes Human Health Fish Tissue Study

Studying Chemicals in Great Lakes Fish to Protect Human Health

As part of the NCCA 2010, EPA conducted the first human health-related study to provide statistically based data on toxic chemicals in Great Lakes fish. For this Great Lakes Human Health Fish Tissue Study, EPA collected samples of fish commonly consumed by humans at 157 of the statistically representative 225 Great Lakes nearshore sampling locations (about 30 fish samples per lake) and analyzed the skin on fillet (muscle) tissue for toxic chemicals. EPA analyzed the tissue samples for total mercury, all 209 congeners of polychlorinated biphenyls (PCBs), 52 polybrominated diphenyl ether (PBDE) congeners, 13 perfluorinated compounds (PFCs) and 5 omega 3 fatty acids. The results identify which chemicals pose greater risks to people who eat Great Lakes fish. The following section briefly describes the contaminants examined and associated human health risk concerns.

About the Targeted Contaminants

PCBs

PCBs bioaccumulate (build up in the tissues) in aquatic organisms, and people can be exposed to potentially harmful levels of PCBs through fish consumption. Animal studies have established that PCBs cause cancer. Based on those findings and additional evidence from human studies, EPA classified PCBs as probable human carcinogens. Other potential health effects include suppression of the immune system, reproductive effects (such as lower birth weight and reduced periods for fetus development), thyroid function impacts, and effects on nervous system development related to short-term memory and learning.

Mercury

People are exposed to methylmercury primarily by eating fish and shellfish. Monitoring mercury levels in fish is critical because about 80% of all fish consumption advisories in the U.S. involve mercury. Fetuses and young children can be exposed to harmful amounts of methylmercury when pregnant women and nursing mothers eat fish with elevated mercury concentrations. These exposures can lead to impairments in neurological development that may impact cognitive thinking, memory, attention, language, and fine motor skills. Exposure to unsafe levels of methylmercury can also affect adult health, leading to cardiovascular disease, loss of coordination, muscle weakness, and impairment of speech and hearing.

PBDEs

A number of studies conducted since 2000 confirm that PBDEs, often referred to as brominated flame retardants, biomagnify (increase in concentration from one level in a food chain to another) in aquatic environments and accumulate in fish. In 2003, EPA began testing fish for the presence of PBDEs because they are persistent, bioaccumulative and toxic chemicals with widespread distribution in the environment. Potential human health impacts include endocrine disruption (such as thyroid function effects) and neurodevelopmental toxicity.

PFCs

PFCs are a large group of synthetic chemicals used in the manufacture of a wide variety of commercial products, including non-stick cookware, food packaging, waterproof clothing, and stain-resistant carpeting. They have emerged as contaminants of concern due to their toxicity, global distribution, and persistence in the environment. Studies have shown that a majority of people living in industrialized nations have detectable concentrations of a number of PFCs in their blood serum. Higher concentrations of PFCs in human blood have been linked to potential health effects, such as decreased sperm count, low birth weight, and thyroid disease. Recent studies estimate that PFC contamination in food may account for more than 90% of human exposure to perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA) and that fish from contaminated waters may be the primary source of exposure to PFOS.

Results

Results from this Great Lakes study showed that PCBs and one or more compounds in the other contaminant groups (mercury, PBDEs, and PFCs) were detected in all 157 fish fillet samples. PCBs and mercury occurred most frequently in these samples at levels exceeding human health thresholds for fish consumption. Tables A-6 and A-7 present a summary of the analytical and statistical results.

Of note, nearly 99% of the Great Lakes nearshore area sampled (or 4,227 square miles) exceeded the 12 ppb human health screening value for PCBs (Table A-6). There are currently fish consumption advisories in all of the Great Lakes because of the presence of toxic contaminants in fish. States, tribes, and the province of Ontario have extensive fish contaminant monitoring programs and issue advice to their residents about how much fish and which fish are safe to eat.

Table A-6. Summary of detections and contaminant concentrations in 157 Great Lakes fish fillet samples (Source: EPA GLHHFT Study).

Chemical	Number of Detections	Minimum Concentration[a] (ppb)	Median Concentration[b] (ppb)	Maximum Concentration[a] (ppb)
PCBs (Total)	157	6	179	2,379
Mercury (Total)	157	23	139	956
PBDEs (Summed)	157	<1	13	227
PFOS	157	2	15	80

[a]Observed data (minimum and maximum concentrations) measured in 157 Great lakes fish fillet samples.
[b]Statistical estimates of the median fish fillet concentrations for the nearshore Great lakes sampled population of 4,282 square miles.

Table A-7. Human health screening value exceedances for contaminants in Great Lakes fish (Source: EPA GLHHFT Study).

Chemical	Human Health Screening Value (SV)	Total Sampled Population	Percentage of Sampled Population Exceeding the SV	Nearshore Area of Sampled Population Exceeding the SV
PCBs (Total)	12 ppb cancer health threshold	4,282 mi^2	98.7%	4,227 mi^2
	60 ppb Great Lakes Sport Fish Advisory Task Force non-cancer threshold	4,282 mi^2	81.7%	3,499 mi^2
Mercury (Total)	300 ppb EPA tissue-based water quality criterion for methylmercury	4,282 mi^2	10.9%	467 mi^2
	110 ppb Great Lakes Sport Fish Advisory Task Force non-cancer threshold	4,282 mi^2	59.5%	2,548 mi^2
PBDEs (Summed)	210 ppb California Environmental Protection Agency non-cancer threshold	4,282 mi^2	<1%	28 mi^2
PFOS	40 ppb Minnesota Department of Health Fish Consumption Advisory Program non-cancer threshold	4,282 mi^2	9.0%	385 mi^2

Cyanobacteria in Nearshore Waters of the Great Lakes

For the Great Lakes portion of the NCCA 2010, analyses of phytoplankton (free-floating algae) were conducted to examine potential risks to human health through recreational exposure. Some phytoplankton are known as blue-green algae, or cyanobacteria, because they have characteristics of both algae and bacteria. Some cyanobacteria species produce toxins that can affect the health of animals and humans. Cyanobacteria are generally found at low cell counts, but occasionally conditions are right for populations to "bloom" to high cell concentrations, and under extreme conditions they can form visible green scums coating the surface of the water. High nutrient levels, other water quality measures, and certain weather conditions can trigger phytoplankton (and, potentially, cyanobacteria) blooms.

The World Health Organization published guidelines for potential human health risks based in part on cyanobacteria cell counts (Table A-8). The guidelines are intended as general alert levels.

Table A-8. Cyanobacteria guidelines for safe practice in managing fresh waters for recreation use (Modified from WHO 2003, Table 8.3, p. 150).

Guidance Level or Situation	Health Risks
Relatively low probability of adverse health effects	
20,000 cyanobacteria cells/ml or 10 µg chlorophyll *a* /L with dominance of cyanobacteria	• Short-term adverse health outcomes, e.g., skin irritations, gastrointestinal illness
Moderate probability of adverse health effects	
100,000 cyanobacterial cells/ml or 50 µg chlorophyll *a* /L with dominance of cyanobacteria	• Potential for long-term illness with some cyanobacterial species • Short-term adverse health outcomes, e.g., skin irritations, gastrointestinal illness
High probability of adverse health effects	
Cyanobacterial scum formation in areas where whole body contact and/or risk of ingestion/aspiration occur	• Potential for acute poisoning • Potential for long-term illness with some cyanobacterial species • Short-term adverse health outcomes, e.g., skin irritations, gastrointestinal illness

The 2010 NCCA survey data indicate that a relatively small percentage of Great Lakes nearshore areas have cyanobacteria levels warranting alert actions. The U.S. nearshore area occupies 6700 square miles. Of this nearshore area, 12% ± 3% exceeded the threshold for a low WHO guideline risk and <2.4% (± 1.8%) exceeded the moderate WHO guideline risk category.

Figure A-9 displays results from 371 sites where phytoplankton samples were collected and analyzed. Alert levels were concentrated in small clusters of sites, particularly within the lower lakes (lower Lake Huron, western Lake Erie, and Lake Ontario) as well as Green Bay, Lake Michigan. Sites were sampled once from May through September; estimates of risk may have been higher if all sampling had occurred in late summer when peak chlorophyll/bloom conditions often occur.

Both embayment and nearshore sites were affected; however, embayments on average had slightly higher cyanobacterial counts than the open nearshore sites. In general, the pattern confirmed a number of historically-known problem areas where phytoplankton blooms have occurred, including those dominated by cyanobacteria (western Lake Erie/Maumee Bay, Saginaw Bay, Green Bay, and south-eastern Lake Ontario). Some of these areas receive regular remote sensing and intensive field sampling for harmful algal bloom species, including by NOAA's Watch Program.

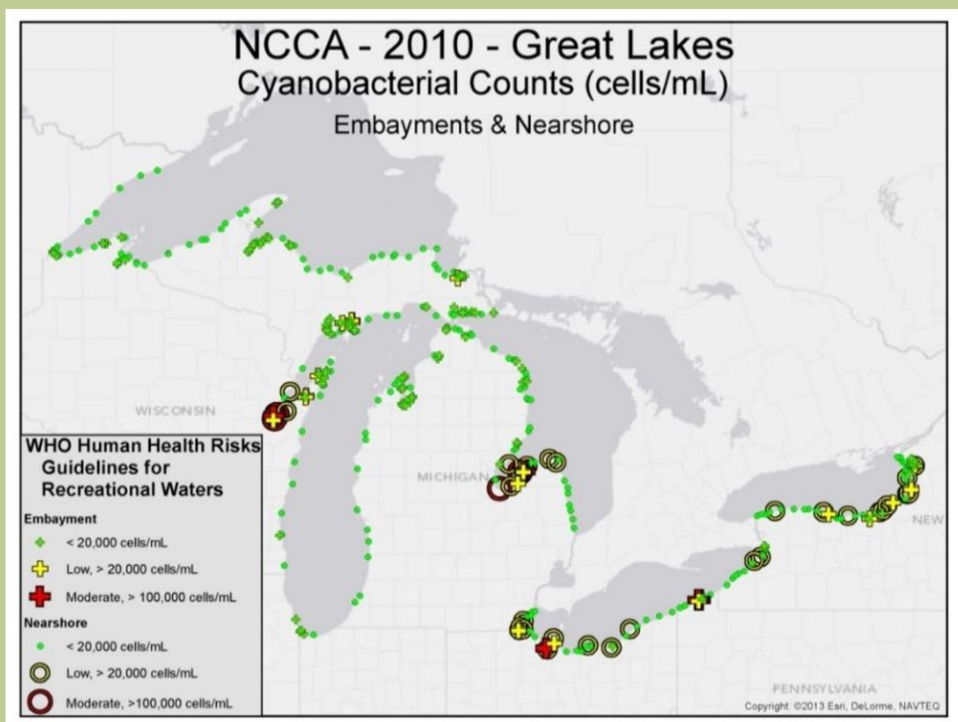


Figure A-9. Sampled sites categorized by WHO alert levels, according to cyanobacteria cell counts.

One of the benefits of the NCCA survey style of sampling illustrated by Figure A-9 is a broad-scale picture of where certain problems are likely. The NCCA survey thus complements more intensive and/or frequent, local surveillance of cyanobacteria, yet offers additional information helpful to view local trends in broader context. This view may assist in setting priorities among different environmental protection and restoration issues, arising from a comprehensive basin-wide perspective.

APPENDIX A - Technical notes on indicators

Sample Design Framework (NCCA 2010)

The sample area for the NCCA is the nearshore coastal waters. The nearshore waters are considered to be heavily used by humans and most vulnerable to activities within adjacent coastal watersheds.

Sample Framework

The Great Lakes nearshore waters sample framework was determined from existing standard GIS medium vector shoreline coverage from NOAA. That coverage was then modified slightly to include an extension 500 m upstream in river mouths and include a few embayment areas which were noticeably missing from the existing shoreline coverage. For the Great Lakes, the nearshore area was defined as being within 5 kilometers (3 miles) from shore and 30 meters (98 feet) or less in depth. This sample frame for the Great Lakes sites was obtained from EPA ORD Mid-Continent Ecology Division (MED) (Olsen, 2009). See Figure A-1 for example. This uniquely "coastal" land-water interface zone includes: river mouths, open and semi-enclosed bays, embayments, and the more open waters adjacent to shorelines (Olsen, 2010). It does not include the connecting channels of the Great Lakes (between lakes and the St. Lawrence River outlet). The sample design targeted a sample size of 45 sites in the shallow nearshore zones of each Great Lake for a total of 225 sites within the United States portion of the Great Lakes. Sample sizes were allocated proportional to shoreline length by state within each Great Lake. In addition, sample sizes for major basins were set to restrict the number of sites occurring within a basin so that sites would not be disproportionally assigned to "narrow" shorelines.

Within the Great Lakes, a number of sites were added and included in this assessment as enhancements. One set of sites that were added are the embayment sites that are areas nested within the shallow nearshore (main GL NCCA target population). Embayments are semi-enclosed by shoreline features, making them less hydrologically-open to open lake wind and waves, see Figure A-2 for examples. Embayments include harbors with man-made shoreline structures (e.g., breakwalls) which make them semi-enclosed. Embayments come in a variety of geomorphologic forms and thus vary in the degree of physical restrictions to water movement between the embayment and more open nearshore waters. In general though, embayments represent more protected waters that are proximal to, and vulnerable, to watershed activities. They may or may not have tributaries from land, but have a continuously-open water connection/channel to the adjacent Great Lake. The target population was limited to distinct, semi-enclosed open water areas no smaller than 1 km^2 and no larger than 100 km^2.

Figure A-1: Example of nearshore sample framework in Lake Erie.

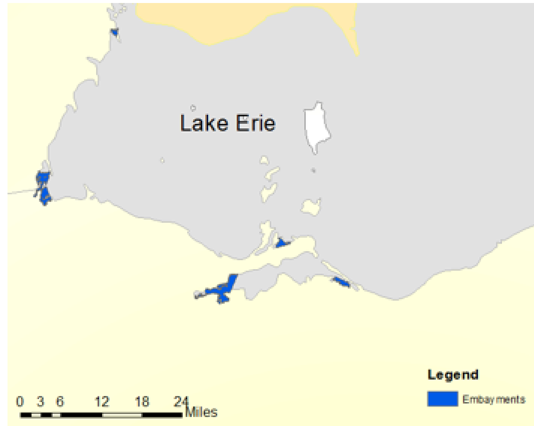


Figure A-2: Example of embayment areas in Lake Erie

Site Selection

A Generalized Random Tessellation Stratified (GRTS) survey design was used to select sample sites. This probability-based approach was selected because it can be used to provide statistically-valid estimates of the condition of coastal waters within the area with known confidence (Diaz et. al, 1996; Stevens, 1997; Stevens et. al., 1999; Stevens et al., 2003; Stevens et al., 2004). Sites that cannot be sampled are replaced. Over sample sites were generated using the above criteria sub-sequentially to the initial base site draw. Sites that could not be sampled were replaced with sites from the oversample list within each stratum. As a result, when the original sites are replaced the original survey design weights must be adjusted. The weight adjustment incorporates information as to why a site is replaced. For instance, if a site was on land or was in an artillery testing area and the site was replaced by a site from the oversample list of the same stratum, it is documented. When estimates of characteristics for the entire target population are computed, the statistical analysis accounts for any stratification or unequal probability selection in the design. Procedures for doing this are available from the Aquatic Resource Monitoring (ARS) web page, http://www.epa.gov/nheerl/arm. A statistical analysis library of functions is also available from the ARS web page to perform common population estimates in the statistical software environment R.

In some incidences, sites were relocated and not replaced. This occurred when the X-site (site location as defined by a latitude and longitude) is not sampleable, but a nearby location is sampleable. A relocation can occur if the site is within a 0.02 nautical miles (37 m or 120 ft) radius of the intended location. A site is not rejected if parameters such as sediment cannot be collected and successful deployment of fish collection gear was not used as a determining factor for rendering a site “unsampleable”. But to increase the chances of a crew collecting sediment, benthic and fish samples, crews were allowed to sample within a 500 m (1650 ft) radius of the designated site location.

Missing data

There was a total of 405 Great Lake sites (i.e., 251 nearshore and 154 embayment sites). Data were categorized missing if there were no samples collected or if they did not meet the quality assurance requirements. Sites are weighted and the condition assessments represent percentage of area and not percent of samples. This is why there is a difference between percentage of samples missing and percent area categorized as missing in the report.

The 2010 NCCA field year was the first time that the probabilistic sample design and NCCA sampling methods was applied in the Great Lakes. Table 1 from the above report shows a summary of the missing data. Of the 405 Great Lake sites, three sites were not sampled and nine sites in National Park Service waters were not intended to have fish, sediment or benthics collected but these 12 sites appear as missing in this assessment. Sediment and benthics were collected at 80% of the sites, with Lake Ontario being the most difficult lake in which to collect a substrate sample due to rocky substrates and dreissenid mussel populations. Lake Michigan, Lake Huron, and Lake Superior proved to be the most challenging lakes for collecting fish samples. In an effort to improve the number of fish samples collected in the NCCCA 2015 sampling, the area of fish collection will be extended out to 1000m. And although benthic samples were collected at 78% of the sites in the Great Lakes, only 55% of the samples could be used in the assessment due to the fact that specific oligochaete species required for calculation of the Oligochaete Trophic Index were missing from 92 of the 317 benthic samples collected.

BENTHIC

Of the 317 samples collected for benthic analysis, 225 samples were included in the condition assessment. The Oligochaete trophic index requires a sample to have oligochaetes that are listed in Table B-1. There were 92 sites that either did not contain any oligochaete species or did not contain the oligochaetes listed in Table B1 and therefore were not included in the analysis. The remaining 225 samples were assessed using the OTI, as described below.

The abundance of oligochaete species in each group is calculated for each site as:

$$\text{OTI} = c\,\frac{\frac{1}{2}\sum n_0 + \sum n_1 + 2\sum n_2 + 3\sum n_3}{\sum n_0 + \sum n_1 + \sum n_2 + \sum n_3} \qquad \text{eqn. B-1}$$

Where:

n_0, n_1, n_2, n_3 refer to the total abundance of species in Group 0, 1, 2, 3 and c adjusts the ratio to the total abundance of tubificid and lumbriculid oligochaetes (n = number per m^2)

c=1 when n ≥ 3600
c=0.75 when 1200 ≤ n < 3600
c=0.5 when 400 ≤ n < 1200
c=0.25 when 130 ≤ n < 400
c=0 when n < 130

Table B-1. Benthic Oligochaete Trophic Index categories.

Group 0	Group 1	Group 2	Group 3	Unassigned[4]
Limnodrilus profundicola *Rhyacodrilus coccineus* ***Rhyacodrilus montana*** ***Rhyacodrilus* sp.** ***Spirosperma nikolskyi*** *Stylodrilus heringianus* *Lumbriculidae*[3] *Trasserkidrilus superiorensis* ***Trasserkidrilus americanus*** *Tubifex tubifex***	*Arcteonais lomondi*[2] *Aulodrilus americanus* *Aulodrilus limnobius* *Aulodrilus pigueti* *Dero digitata*[2] *Ilyodrilus templetoni* ***Isochaetides freyi*** *Slavina appendiculata*[2] *Spirosperma ferox* *Uncinais uncinata*[2]	*Aulodrilus pluriseta* *Limnodrilus angustipenis* *Limnodrilus cervix* *Limnodrilus claparedianus* ***Limnodrilus maumeensis*** *Limnodrilus udekemianus* *Potamothrix bedoti* *Potamothrix moldaviensis* *Potamothrix vejdovskyi* *Quistadrilus multisetosus*	*Limnodrilus hoffmeisteri* *Tubifex tubifex***	*Branchiura sowerbyi (2)* *Chaetogaster diaphanus (2)* *Dero* sp. (2) *Ilyodrilus frantzi* *Naidinae* *Nais* sp. *Nais bretscheri* *Ophidonais serpentina (2)* *Paranais grandis* *Paranais litoralis* *Piguetiella* sp. *Piguetiella blanci (2)* *Specaria* *Stylaria lacustris (2)* *Tubificinae* *Varichaetadrilus* *Vejdovskyella intermedia (1)*
† Species in bold above were not reported from NCCA 2010 Great Lakes samples				
**Tubifex tubifex* is assigned to Group 0 or Group 3 according to the following rules: - if $n_0 : n_3$ < 0.75 then Group 0; - if $n_0 : n_3$ > 1.25 then Group 3; - if $n_0 : n_3$ = 0.75 – 1.25 then Group 0 if c < 0.5 or Group 3 if c ≥ 0.5; - if n_3 = 0 then Group 0 if n_0 is relatively high and/or c is low; otherwise Group 3				
[1] from State of the Great Lakes 2011 –Benthic Diversity and Abundance Table 1. [Classifications are from Howmiller and Scott (1977), Milbrink (1983), Kreiger (1984), and Lauritsen et al (1985)]. Only species in the families, Naididae (formerly Tubificidae) and *Lumbriculidae* were included.				
[2] These species were not included in State of the Great Lakes 2011 list presumably because they were thought to be in the family Naididae, not Tubificidae, although they were included in group 2 in earlier publications. However, recent taxonomy changes have reclassified Tubificidae to Naididae which has several subfamilies including Naidinae and Tubificinae, so they were included in Group 1.				
[3] SOLEC classified all immature Lumbriculidae as *Stylodrilus heringianus*. Therefore taxa in NCCA 2010 GL samples that were identified as Lumbriculidae are assigned Group = 0.				
[4] Taxa with numbers are group assignments recommended by Kurt Schmude, Univ. of Wisconsin - Superior				

WATER QUALITY

The water quality index assessment approach was based on information found in existing International Joint Commission reports (IJC, 1979; PMSTF, 1980). Although the 1980 PMSTF guidelines were intended for open water, some components of the design framework overlapped with the design framework of the 2010 NCCA. Additionally some of the 1979 IJC thresholds focused on nearshore waters which also overlapped with the 2010 NCCA design frame. In the process of determining water quality thresholds for this report, it was determined that because the United States and Canada under Annex 4 of the Great Lakes Water Quality Agreement (GLWQA, 2012) are currently reviewing and negotiating new water quality guidelines, it would not be appropriate to introduce interim NCCA assessment methods or thresholds at this time.

Thresholds for total phosphorus, chlorophyll *a*, and secchi depth are as specified in the current International Joint Commission guidelines (IJC, 1979; PMSTF, 1980). Note that thresholds vary by lake and basins within a lake. Dissolved oxygen thresholds were the same as those used in estuaries (Diaz and Rosenberg, 1995; USEPA, 2011) for the upper threshold and were based on onset of hypoxic conditions for the lower threshold (Costantini et al., 2011; Krieger and Bur, 2009).

Discrete surface total phosphorus and chlorophyll a samples were collected from 0.5 meters below the water surface, and analyzed by multiple laboratories that all followed methods as outlined in the EPA laboratory manual (USEPA, 2010a) and quality assurance project plan (USEPA, 2010b). Water clarity was characterized in the Great Lakes primarily by secchi depth, and secondarily by secchi depth estimated from Photosynthetically Active Radiation (PAR) attenuation at sites lacking Secchi results such as sites where it was clear to the bottom. Secchi depth was estimated from PAR attenuations by using normalized PAR intensity data as recorded from a sensor that was lowered through the water column, and an attenuation coefficient K_d was calculated from a regression of PAR vs. depth (equations WQ-1 through WQ-3). To begin, PAR attenuation was measured using two PAR sensors. One sensor was lowered through the water column, measuring PAR intensity (Iz) at depths z. A second sensor in air reported varying incident PAR intensity (Io) arising, for instance, from changing cloud cover. The normalized PAR attenuation (Iz/Io) is assumed to follow Beer's law, i.e., light intensity decreasing exponentially with distance. Where K_d is the PAR attenuation coefficient; larger K_d magnitudes indicate greater attenuation, i.e., poorer water clarity.

$$Iz/Io = \exp(-K_d*z) \qquad \text{eqn WQ-1}$$
$$\ln(Iz/Io) = - K_d *z \qquad \text{eqn WQ-2}$$

Operationally, K_d is calculated as the negative slope of a regression of ln(Iz/Io) vs depth. PAR intensities and depth measurements are reported in a "hydrolab" data file available at the NARS website (http://water.epa.gov/type/watersheds/monitoring/aquaticsurvey_index.cfm). An Excel spreadsheet was devised to review regression plots for every site in order to identify, flag, and remove errant data values used in the regression calculation—a necessary step, as errant values were common. Once reliable K_d values are obtained, % transmittance at one meter (i.e., Iz/Io at one meter) was calculated from eqn WQ-1 as:

$$\%\ Trans\ @\ 1m = \exp(-K_d)*100 \qquad \text{eqn WQ-3}$$

A best-fit relationship was then determined between secchi depths and K_d from sites where both measurements were available (eqn WQ-4). For the 2010 survey, this relationship was:

$$\text{Secchi depth (estimated)} = 1.31* K_d^{-0.91} \qquad \text{eqn WQ-4}$$

This relationship was then used to estimate secchi depths at sites where only K_d values were available. If neither K_d or Secchi depth was available, the condition at the site was set to "missing".

Condition assessment of nitrogen-based nutrients was not included in this report although samples were collected and measured. Figure WQ-1 and Table WQ-2 below summarize total nitrogen concentrations measured in Great Lakes basin in the 2010 survey. These data provide an indication of nitrogen condition in the Great Lakes and will be reviewed when deciding whether total nitrogen concentration will be included as a metric in future surveys.

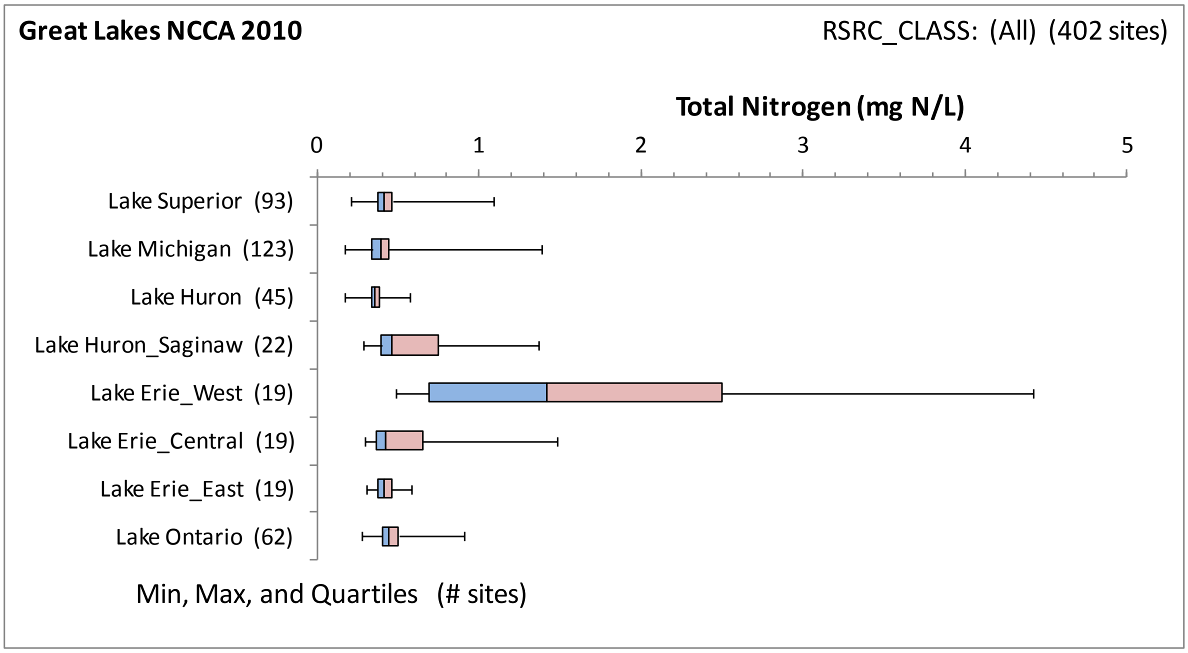


Figure WQ-1. Comparison of total nitrogen concentrations in Great Lakes basins in 2010. The plot presents the minimum, maximum, and unweighted quartile values for all measured sites.

Table WQ-2. Total nitrogen concentration values plotted in Figure WQ1.

NCCA 2010 Great Lakes
Total Nitrogen (mg N/L)
RSRC_CLASS: (All) (402 sites)

	Lake Superior	Lake Michigan	Lake Huron	Lake Huron_Saginaw	Lake Erie_West	Lake Erie_Central	Lake Erie_East	Lake Ontario
count	93	123	45	22	19	19	19	62
avg	0.45	0.43	0.36	0.58	1.69	0.54	0.43	0.46
stdev	0.15	0.19	0.06	0.27	1.26	0.29	0.08	0.11
min	0.21	0.17	0.17	0.28	0.49	0.30	0.30	0.28
25 %ile	0.37	0.34	0.34	0.39	0.69	0.36	0.37	0.40
50 %ile	0.41	0.40	0.36	0.47	1.42	0.42	0.41	0.44
75 %ile	0.46	0.44	0.38	0.74	2.49	0.65	0.46	0.50
max	1.09	1.38	0.57	1.37	4.42	1.48	0.58	0.90

SEDIMENT

The freshwater consensus-based Probable Effect Concentration (PEC) values were derived from an aggregation of several different sediment quality guidelines having similar narrative intent based on an empirical approach. In order to calculate a PEC value for a given contaminant, there needed to be at least three published values that met the narrative description of predicting sediment toxicity (Table S-1). The detection limit also had to be below or at the Threshold Effect Concentration (TEC). For the purpose of data interpretation of sediment chemistry in the Great Lakes, the recommendation is to calculate the mean PEC quotient (PEC-Q). The mean PEC-Q is a calculation that combines the effects of multiple contaminants in the sediment to further assist in the interpretation of the results in predicting the likelihood of the sediment having a toxic effect. The mean PEC quotient is the contaminant class average of the ratio of sample results to reliable PEC values (i.e., metals, total PAHs, and total PCBs). Total PAHs were used instead of individual PAHs to eliminate redundancy. The following is how the value is calculated:

Individual PEC-Q = chemical concentration (dry wt.)/ corresponding PEC value eqn S-1

(Use ½ the detection limit or exclude if detection limit is greater than PEC value. If more than 50% of the chemicals in a chemical class are nondetects, then they should not be included in the quotient).

Mean PEC-$_{\text{Qmetals}}$ = (∑individual metal PEC-Qs)/n, where n is the number of metals with reliable PECs (i.e., arsenic, cadmium, chromium, copper, lead, nickel, and zinc) as shown by eqn S-2.

Calculate the mean PEC-Q for the three main classes of chemicals with reliable PECs as follows:

Mean PEC-Q = (mean PEC-Q$_{\text{metals}}$ + PEC-Q$_{\text{T. PAHs}}$ + PEC-Q$_{\text{T. PCBs}}$)/n eqn. S-2

Where:
n = number of classes of chemicals for which sediment chemistry data are available (i.e., 1 to 3)

The proposed cutpoints for the consensus-based mean PEC-Q is based on Ingersoll et al. 2001, Crane et al. 2002, Crane and Hennes 2007, and group consensus. The 0.6 mean PEC-Q reflects the 50% incidence of toxicity for the 28-day *Hyalella azteca* growth and survival test, and the 0.1 mean PEC-Q reflects a 20% incidence of toxicity in both the 10-day *Hyalella azteca* and *Chironomus* survival test and 8% for the 28-day *Hyalella azteca* survival test (Ingersoll et al., 2001

Table S-1. Sediment parameters analyzed and identification of parameters included in calculation of indices. (Metals in µg/g; PAHs, Pesticides and PCBs in ng/g)

Sediment Chemicals analyzed for NCCA 2010	Consensus-Based TEC Values	Consensus-Based PEC Values	Reliable PEC-Q Used (½ detection limit used, exclude if DL>PEC value)
Aluminum			
Antimony			
Arsenic	9.79	33.0	x
Cadmium	.99	4.98	x
Chromium	43.4	111	x
Copper	31.6	149	x
Iron			
Lead	35.8	128	x
Manganese			
Mercury	.18	1.06	
Nickel	22.7	48.6	x
Selenium			
Silver			
Tin			
Zinc	121	459	x
Total PCB congeners*	60	676	x
Aldrin			

Sediment Chemicals analyzed for NCCA 2010	Consensus-Based TEC Values	Consensus-Based PEC Values	Reliable PEC-Q Used (½ detection limit used, exclude if DL>PEC value)
Acenaphthene	6.7	89**	
Acenaphthylene	5.9	130**	
Anthracene	57.2	845	
Benz(a)anthracene	108	1050	
Benzo(b)fluoranthene			
Benzo(e)pyrene			
Benzo(k)fluoranthene			
Benzo(g,h,i)perylene			
Benzo(a)pyrene	150	1450	
Biphenyl			
Chrysene	166	1290	
Dibenz(a,h)anthracene			
Dibenzothiophene			
2,6-dimethylnaphthalene			
Fluoranthene	423	2230	
Fluorene	77.4	536	
Indeno(1,2,3-c,d)pyrene			
1-methylnaphthalene			
2-methylnaphthalene	20.2	200**	

Alpha-Chlordane			
Lindane	2.37	4.99	
2,4'DDD			
4,4'DDD			
2,4'DDE			
4,4'DDE			
2,4'DDT			
4,4'DDT			
Total DDT	5.28	572	
Dieldrin	1.9	61.8	
Endosulfan I			
Endosulfan II			
Endosulfan sulfate			
Endrin	2.2	207	
Heptachlor			
Heptachlor epoxide	2.5	16	
Heptachlorobenzene			
Mirex			
Trans-Nonachlor			

1-methylphenanthrene			
Naphthalene	176	561	
Perylene			
Phenanthrene	204	1170	
Pyrene	195	1520	
2,3,5-trimethylnaphthalene			
LMW PAH			
HMW PAH			
Total PAHs***	1610	22800	x

* Congeners – 8,18,28,44,52,66,77,101,105,110,118,126,128,138,153,170,180,187,195,206,209 – Total PCB calculation is all congeners
**Not consensus-based values but Canadian Environmental Quality Guidelines (CCME 1999) also adopted for use in Minnesota (Crane et al., 2002; Crane and Hennes, 2007)
*** Total PAHs (Sum of LMW PAH (Acenaphthene, Acenaphthylene, Anthracene, Fluorene, 2-methylnaphthalene, Naphthalene, Phenanthrene) and HMW PAH (Benz(a)anthracene, Benzo(a)pyrene, Chrysene, Dibenz(a,h)anthracene, Fluoranthene, Pyrene).

ECOLOGICAL FISH CONTAMINANT INDEX

For this assessment, an ecologically based method is used to assess contaminant burdens in whole fish using a generalized approach based on U.S. EPAs Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (1997). Whole fish tissue samples of predominantly forage-size fish were analyzed for measurable concentrations of multiple contaminants of concern (USEPA 2010b). Crews were provided with a recommended list shown in table F-1 of fish to collect. Analytical results were compared with updated fish tissue screening values that were developed to evaluate risk to upper-trophic level fish and wildlife, including birds and mammals.

Using an ecological risk assessment approach (USEPA, 1997), risk was defined by developing a ratio of exposure concentration compared to a concentration that is known to have toxicological effects. The exposure concentration is developed based on known characteristics of each of the receptors of concern (i.e., fish, birds and mammals) including body weight, food ingestion rate, and home range (i.e., natural range of receptor with respect to foraging, breeding, and other activities). The concentration of contaminant that is known to elicit toxicological effects (i.e., toxicological reference value or TRV), is reported in the literature for certain species for each contaminant. Using an ecological risk assessment framework, a ratio greater than 1.0 indicates that exposure concentration is greater than the toxicological reference value. By using the minimum risk level of 1.0, the fish tissue concentration that would indicate this minimum risk can be calculated.

Table F-1. Recommended Great Lakes target species for whole body fish tissue collection.

Family	Scientific name	Common Carp	Great Lakes				
			Erie	Huron	Michigan	Ontario	Superior
Catostomidae	Pumpkinseed	Lepomis gibbosus				●	
	Rock Bass	Ambloplites rupestris				●	
	Shorthead Redhorse	Moxostoma macrolepidotum				●	
Centrarchidae	Black Crappie	Pomoxis nigromaculatus				●	
	Bluegill	Lepomis macrochirus			●	●	
	Freshwater Drum	Aplodinotus grunniens				●	
	Pumpkinseed	Lepomis gibbosus			●		
	Rock Bass	Ambloplites rupestris			●		
	Smallmouth Bass	Micropterus dolomieu		●		●	
	White Crappie	Pomoxis annularis				●	
Cottidae	Common Carp	Cyprinus carpio				●	
	Mottled Sculpin	Cottus bairdii			●	●	
	Slimy Sculpin	Cottus cognatus		●	●	●	
Cyprinidae	Bluntnose Minnow	Pimephales notatus			●	●	
	Common Carp	Cyprinus carpio	●				
	Lake Chub	Couesius plumbeus			●	●	
Esocidae	Muskellunge	Esox masquinongy				●	
	Northern Pike	Esox lucius				●	
Gasterosteidae	Three-Spined Stickleback	Gasterosteus aculeatus aculeatus				●	
Gobiidae	Round Goby	Neogobius melanostomus	●		●	●	
	Tubenose Goby	Proterorhinus marmoratus			●	●	
Ictaluridae	Brown Bullhead	Ameiurus nebulosus				●	
	Channel Catfish	Ictalurus punctatus	●			●	
	Stonecat	Noturus flavus				●	
Lotidae	Burbot	Lota lota			●	●	
Moronidae	Percidae Logperch	Percina caprodes				●	
	Ruffe	Gymnocephalus cernuus				●	
	Sauger	Sander canadensis				●	
	Walleye	Sander vitreus				●	
	White Bass	Morone chrysops	●			●	
	White Perch	Morone americana	●			●	
	Yellow Perch	Perca flavescens				●	
Osmeridae	American/ Rainbow Smelt	Osmerus mordax					●
	Lake Whitefish	Coregonus clupeaformis					●
Percidae	Logperch	Percina caprodes			●		
	Walleye	Sander vitreus	●	●			
	Yellow Perch	Perca flavescens	●	●	●		
Percopsidae	Trout-Perch	Percopsis omiscomaycus				●	
Salmonidae	Chinook Salmon	Oncorhynchus tshawytscha				●	
	Cisco/ Lake Herring	Coregonus Artedi					●
	Coho Salmon	Oncorhynchus kisutch				●	
	Lake Trout	Salvelinus namaycush			●	●	●
	Lake Whitefish	Coregonus clupeaformis			●	●	
	Pink Salmon	Oncorhynchus gorbuscha				●	
	Rainbow Trout	Oncorhynchus mykiss				●	
Sciaenidae	Freshwater Drum	Aplodinotus grunniens	●				

Methods for Developing Fish Tissue Contaminant Threshold Values

Risk potential was derived by calculating a hazard quotient (HQ), or the ratio of exposure concentration divided by a concentration known to elicit toxicological effects (Low Observed Adverse Effects Level or LOAEL) or known not to elicit toxicological effects (No Observed Adverse Effect Level or NOAEL). Risk can be expressed as:

$$\text{Risk (HQ)} = \text{Exposure Concentration/Toxicity} \qquad \text{eqn F-1}$$

Thus, when the exposure concentration is greater than the concentration known to elicit toxic effects, the HQ is greater than 1.0, and the receptor is at risk.

The derivation of the exposure concentration was specific for each receptor and dependent on known characteristics for each receptor, including body weight, food ingestion rate, exposure area relative to the amount of time the organism spends in the area (or Area Use Factor, AUF), and fish tissue concentration. The exposure concentration can be represented by the formula:

$$Exposure\ Concentration = \frac{FI * [Fish] * AUF}{BW} \qquad \text{eqn F-2}$$

where:

FI = food ingestion (kg/kg bw/d)

[Fish] = concentration in fish tissue (mg/kg)

AUF = area use factor

BW = body weight of receptor (kg-bw)

For added conservativeness the AUF was set to 1.0, indicating all foraging, resting, breeding and other activities are expected to occur within the exposure area of concern. Toxicity was quantified as toxicity reference values (TRVs). Toxicity reference values are established from the available scientific literature. For the 2010 NCCA survey, the NOAEL and LOAEL served as the basis for establishing threshold contaminant values. Toxicity reference values are typically established for each receptor of concern or group of receptors (i.e., avian, freshwater or fish, marine mammals, etc.).

Receptors of Concern

For NCCA, upper trophic level organisms, including birds and mammals, are considered receptors of concern (ROCs). ROCs are typically those animals that are exposed to contaminants through ingestion, dermal contact, and/or inhalation. The exposure of ROCs to contaminants by ingestion is through either incidental media uptake (i.e., eating soil or sediment that is associated with prey items), drinking contaminated surface water, or through the ingestion of prey items which have accumulated contaminants in their tissues. For NCCA, data evaluated were whole-body forage fish tissue concentrations; therefore, the only pathway of exposure evaluated for the assessment focused on the uptake of contaminants that have been accumulated in the tissues of prey items (i.e., fish).

Classes of receptors were created to develop potential exposure-based screening values since data consisted of both freshwater and marine fish tissues. These classes include: freshwater predatory fish, marine predatory fish, piscivorous birds, piscivorous freshwater mammals and piscivorous marine mammals. Receptors were chosen based on their diet (predominantly fish) and the availability of data in the literature. Potential receptors evaluated for NCCA represent those species that are typically included in ecological risk assessments (Table F-2).

Table F-2. Potential receptors of concern often evaluated in ecological risk assessments.

Avian Receptor	Freshwater Mammalian Receptor	Marine Mammalian Receptor	Freshwater Fish Receptor	Marine Fish Receptor
Great Blue Heron	River Otter	Harbor Seal	Largemouth Bass	Bluefin Tuna
Osprey	Mink	Bottlenose Dolphin	Florida Gar	Yellowfin Tuna
Bald Eagle		Walrus	Muskellunge	Shortfin Mako
Herring Gull			Snakehead	Sandbar Shark
Belted Kingfisher			Lake Walleye	Mackerel Tuna
Brown Pelican				Swordfish

The list summarized in Table F-2 may not be representative of potential receptor species at all sampling locations. To account for this limitation, generalized body weights and food ingestion rates for freshwater and marine fish, birds, and mammals were estimated from the receptor species listed. To be most protective, the lowest body weight and highest food ingestion rate where chosen for each receptor category for calculating dosage estimates. Table F-3 summarizes the minimum and maximum receptor factors considered in determining weight and ingestion rate constants applied in the developing the threshold values. Table F-4 describes the “generalized” receptor factors used to derive the new NCCA threshold values.

Table F-3. Minimum and Maximum Body Weights and Derived Food Ingestion Rates for Selected Receptors of Concern.

Group	Receptors	Body Weight (kg)		Ref.	Food Ingestion Rate (kg food/kg BW/d)	
		Min/Ave	Max		Min/Ave BW	Max BW
	Great Blue Heron	1.47	2.99		0.051	0.040
	Osprey	1.22	1.95		0.054	0.046
Avian[1]	Bald Eagle	3.00	4.50	a	0.040	0.034
	Herring Gull	0.83	1.62		0.062	0.049
	Belted Kingfisher	0.13	0.22		0.120	0.100
	Brown Pelican	3.00	3.50	b	0.040	0.038
Freshwater Mammals[1]	River Otter	5.00	15.00		0.052	0.042
	Mink	0.55	2.08	a	0.076	0.060
	Harbor Seal	58.80	124.00		0.033	0.029
Marine Mammals[1]	Bottlenose Dolphin	150.00	490.00	c	0.028	0.023
	Walrus	900.00	1400.00	d	0.020	0.019
	Bluefin Tuna	32.00	219.00	e	0.044	0.016
Marine Fish[2]	Yellowfin Tuna	23.42	52.45	f	0.023	0.010
	Shortfin Mako	63.50		g	0.046	

Group	Receptors	Body Weight (kg)		Ref.	Food Ingestion Rate (kg food/kg BW/d)	
		Min/Ave	Max		Min/Ave BW	Max BW
	Sandbar Shark	34.00		h	0.009	
	Mackerel Tuna	34.55		i	0.022	
	Swordfish	58.00		j	0.016	
Freshwater Fish[2]	Brown Trout	0.91	3.63	k	0.0095	
	Muskellunge	0.34	31.64	l	0.064	
	Largemouth Bass	0.45	4.50	m	0.024	

[1] Avian and mammalian food ingestion rates were calculated using equations derived from Nagy (1987).

[2] Food ingestion rates for fish were calculated based on daily rations. Daily rations were converted from percent body weight/day to kg food/ kg body weight/day in order to estimate food ingestion rates that are comparable to the avian and mammalian values. Data for the shortfin mako, sandbar shark, mackerel tuna, and swordfish are based on average body weight and daily ration as opposed to minimum and maximum body weight.

a – USEPA 1993; b – Schreiber, 1976; c – Kastelein et al., 2002; d – Born et al., 2003
e – Aguado-Gimenez and Garcia-Garcia, 2005; f – Maldeniya, 1996; g – Wood et al., 2009
h – Stillwell and Kohler, 1993; i – Griffiths et al., 2009; j – Stillwell and Kohler, 1985
k – Becker, 1983; l – Carlander, 1969; m – Carlander, 1977

Table F-4. Summary of generalized receptor body weights and food ingestion rates used to calculate screening fish tissue values.

Receptor Group	Body Weight (kg)	Food Ingestion Rate (kg food/kg BW/d)
Birds	0.13	0.1203
Freshwater Mammals	0.55	0.0764
Marine Mammals	58.8	0.0333
Freshwater Fish	0.34	0.0640
Marine Fish	23.42	0.0458

Toxicity Reference Values

Literature-based toxicological data typically used to derive reference values are based on laboratory species. The laboratory-based tests used to develop TRVs may not have resulted in an endpoint that is protective of chronic exposure. A chronic exposure endpoint was extrapolated from the reported endpoint using a conversion factor (CF). CFs have been used for various extrapolations, and their applications reflect policy to provide conservative estimates of risk (Chapman et al., 1998). Table F-5 summarizes conversion factors applied to laboratory-based endpoints to estimate chronic NOAEL or no observable effects concentration (NOEC) (Wentsel et al., 1996).

Table F-5. Conversion factors to estimate chronic NOAELs or NOECs (Wentsel et al., 1996)

Convert From	Convert To	Multiply By
Chronic NOAEL or NOEC	Chronic NOAEL or NOEC	1.0
Chronic LOAEL or LOEC	Chronic NOAEL or NOEC	0.2
Subchronic NOAEL or NOEC	Chronic NOAEL or NOEC	0.1
Subchronic LOAEL or LOEC	Chronic NOAEL or NOEC	0.05
Acute NOAEL or NOEC	Chronic NOAEL or NOEC	0.033
Acute LOAEL or LOEC	Chronic NOAEL or NOEC	0.02
LD50 or LC50	Chronic NOAEL or NOEC	0.01
Durations are defined as follows (USEPA, 1999; Sample et al., 1996): • Acute: < 14 days (fish, birds, mammals) • Subchronic: 14-90 days (fish, birds, mammals) • Chronic: > 90 days or during critical life stage (fish, birds, mammals)		

Generally, reference values were developed from laboratory tests using non-wildlife species (e.g., chickens, quail, duck, rat, mouse, rainbow trout, and Japanese medaka). Using the reported body weights of laboratory test species and wildlife receptors, laboratory based endpoints were normalized to wildlife receptors using formulae developed by Sample and Arenal (1999). TRVs were calculated using the following equation:

$$TRV_{wildlife} = \left(BW_{test}/BW_{wildlife}\right)^{(1-x)} \qquad \text{eqn. F-3}$$

where:

$TRV_{wildlife}$	=	*toxicity reference value for wildlife species*
NOAELtest	=	*no observed adverse effect level for test species*
BWtest	=	*body weight for test species*
BWwildlife	=	*body weight for wildlife species*
X	=	*scaling factor*

Scaling factors presented by Sample and Arenal (1999) indicated that mammalian sensitivity increases with increased body weight, and avian sensitivity increases with decreased body weight. Scaling factors were unavailable for fish receptors but, like avian receptors, an increase in sensitivity with decreased body weight was reported (Buhler and Shanks, 1970). A scaling factor of 0.94 was used for mammalian receptors (Sample and Arenal, 1999) and a scaling factor of 1.2 was used for avian (Sample and Arenal, 1999) and fish receptors

(Buhler and Shanks, 1970). Table F-6 shows calculated TRVs for each NCCA analyte of interest that was used for estimating threshold values.

Table F-6. Calculated toxicity reference values (TRVs) based cited literature and estimation methods.

Constituent	TRV Type	Calculated Wildlife TRVs									
		Avian		Mammal				Fish			
				Freshwater		Marine		Freshwater		Marine	
		TRV	Ref.	TRV	Ref.	TRV	Ref.	TRV	Ref.	TRV	Ref.
Arsenic	NOAEL	3.39	z	0.11	b	0.08	b	0.027	aa	0.06	aa
	LOAEL	8.51		0.53		0.4		0.14		0.3	
Cadmium	NOAEL	0.94	b	0.89	x	0.67	x	76.34	y	168	y
	LOAEL	12.93		4.46		3.37		763.49		1680	
Mercury (methyl)	NOAEL	0.02	v	0.31	b	0.024	b	0.14	w	0.31	w
	LOAEL	0.12		0.16		0.12		0.28		0.62	
Selenium	NOAEL	0.27	b	0.19	b	0.15	b	5.02	u	11.04	u
	LOAEL	0.53		0.32		0.24		6.7		14.75	
Chlordane	NOAEL	0.53	a	3.85	b	2.91	b	NA	NA	NA	NA
	LOAEL	2.66		7.69		5.81		NA		NA	
DDTs	NOAEL	0.15	a	0.78	b	0.59	b	0.28	t	0.62	t
	LOAEL	1.47		3.89		2.94		1.42		3.12	
Dieldrin	NOAEL	0.08	b	0.033	q	0.025	q	0.065	r	0.14	r
	LOAEL	0.39		0.17		0.13		0.33		0.72	
Endosulfan	NOAEL	7.99	b	1.19	o	0.9	o	0.26	p	0.6	p
	LOAEL	39.93		5.95		4.5		0.6		1.31	
Endrin	NOAEL	0.019	b	0.15	b	0.11	b	0.16	n	0.34	n
	LOAEL	0.099		0.77		0.58		0.78		1.72	
Heptachlor epoxide	NOAEL	1.16	l	0.21	b	0.16	b	8.09	m	17.8	m
	LOAEL	5.79		1.037		0.78		16.2		35.6	

Constituent	TRV Type	Calculated Wildlife TRVs									
		Avian		Mammal				Fish			
				Freshwater		Marine		Freshwater		Marine	
		TRV	Ref.	TRV	Ref.	TRV	Ref.	TRV	Ref.	TRV	Ref.
Hexachlorobenzene	NOAEL	0.11	h,j	0.97	j	0.74	j	0.0018	k	0.0039	k
	LOAEL	0.56		1.95		1.47		0.0088		0.019	
Lindane	NOAEL	0.54	b	7.79	b	5.88	b	14.99	g	32.98	g
	LOAEL	2.19		38.93		29.41		74.95		164.91	
Mirex	NOAEL	0.0066	d	0.064	e	0.048	e	0.4	f	0.87	f
	LOAEL	0.66		0.64		0.048		1.98		4.35	
Toxaphene	NOAEL	0.66	a	7.79	b	5.88	b	0.0011	c	0.0024	c
	LOAEL	3.32		38.93		29.41		0.0056		0.012	
PCBs (Arochlor 1254)	NOAEL	0.12	b	0.055	b	0.041	b	0.078	bb	0.17	bb
	LOAEL	1.2		0.55		0.41		0.39		0.86	
High Molecular Weight PAHs	NOAEL	4.35	ii	0.58	jj	0.44	jj	0.55	kk	1.21	kk
	LOAEL	21.77		2.92		2.21		2.76		6.07	
Low Molecular Weight PAHs	NOAEL	15.16	ll	2.97	mm	2.24	mm	NA	NA	NA	NA
	LOAEL	151.6		297		224.4		NA		NA	

a – Wiemeyer, 1996
b – Sample et al., 1996
c – Fahraeus-Van Ree and Payne, 1997
d – Hyde et al., 1973
e – NTP, 1990
F – Skea et al., 1981
g – Cossarini-Dunier et al., 1987

h – Coulston and Kolbye, 1994
i – Terretox, 2002
j – ATSDR, 2002a
k – Woodburn et al., 2008
l – USEPA, 1972
m – Andrews et al., 1996
n – Argyle et al., 1973
o – ATSDR, 2000

p – Lundebye et al., 2010
q – ATSDR, 2002b
r – Argyle et al., 1975
s – USEPA, 1995
t – Macek et al., 1970
u – Ogle and Knight, 1989
v – Heinz and Locke, 1976
w – Berntssen et al., 2003

x – ATSDR 2008,
y – Szczerbik et al., 2006
z – USDI, 1964
aa – Pedlar et al., 2002
bb – Leatherland and Sonstegard, 1980
cc – Giesy et al., 2002
dd – USEPA, 2008
ee – Nakayama, 2004

Calculating Fish Tissue Contaminants Threshold Values

The tissue contaminant concentration threshold values for the suite of NCCA analytes were derived using the following equation:

[Fish] = (TRV*BW)/FI

Where:

[Fish]	=	*threshold concentration in fish tissue (mg/kg) for a specific analyte*
TRV	=	*related estimated toxicity reference value*
BW	=	*generalized body weight of receptor (kg-bw)*

Thus, using the toxicity reference values plus estimated body weights and food ingestion rates, the concentration value of a selected analyte measured in fish tissues that presented a minimum exposure risk (HQ=1.0) was calculated for each group of receptors. The calculated fish tissue concentrations can be used to screen fish tissue data to determine if piscivorous fish and wildlife may be at risk due to the consumption of fish. A fish tissue concentration for each receptor group was calculated and can be used individually to screen for the potential risk to each receptor group. The lowest calculated fish tissue concentration can be used to screen tissue concentration for risk to any receptor group regardless of the source of equation terms. In Table F-7, the results for each group of receptors used for the NCCA Ecological Fish Tissue is summarized.

Table F-7. Summary of all estimated fish tissue threshold values in mg/kg. Highlighted columns identify the values used for calculating the index.

Ecological Contaminants of Concern	Freshwater Mammal		Marine Mammal		Bird		Freshwater Fish		Marine Fish	
	Fish Tissue		Fish Tissue		Fish Tissue		Fish Tissue		Fish Tissue	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Toxaphene	56.05	280.25	10387.34	51936.72	0.72	3.59	0.0059	0.03	1.25	6.25
Mirex	0.46	4.6	85.24	852.35	0.0072	0.72	2.1	10.5	444.63	2223.15
Lindane	56.05	280.25	10387.34	51936.72	0.59	2.36	79.63	398.15	16865.86	84329.31
Hexachlorobenzene	7.01	14.01	1298.42	2596.84	0.12	0.6	0.0094	0.047	1.99	9.93

Heptachlor epoxide	1.49	7.46	276.57	1382.84	1.25	6.26	42.97	85.95	9102.16	18204.31
Endrin	1.09	5.56	201.68	1030.83	0.02	0.11	0.83	4.15	175.75	878.77
Endosulfan	8.57	42.84	1587.71	7938.54	8.63	43.15	0.0014	0.0032	0.3	0.67
Dieldrin	0.24	1.2	44.46	222.28	0.067	0.33	0.35	1.74	73.69	368.46
DDT	5.61	28.03	1038.73	5193.67	0.16	1.59	1.51	7.54	319.53	1597.65
Chlordane	27.69	55.38	5131.72	10263.45	0.57	2.87	**NA**	**NA**	**NA**	**NA**
Selenium	1.4	2.31	259.68	428.48	0.29	0.57	26.66	35.6	5646.58	7541.15
Mercury	0.22	1.12	41.55	207.75	0.02	0.13	0.76	1.49	160.85	315.99
Cadmium	6.43	32.13	1190.78	5953.9	1.01	13.97	405.6	4056.04	85907.52	859075.2
Arsenic	0.76	3.81	141.18	705.89	3.66	9.2	0.15	0.73	30.81	154.07
PCB	0.39	3.93	72.79	727.86	0.13	1.29	0.41	2.06	87.46	437.31

Sources – Sample Design

Diaz-Ramos, S., Stevens, D. L., Jr, & Olsen, A. R. (1996). EMAP Statistical Methods Manual. EPA/620/R-96/002, U.S. Environmental Protection Agency, Office of Research and Development, NHEERL-Western Ecology Division, Corvallis, Oregon.

Olsen, T. (2010, January). USEPA. National Coastal Assessment 2010 Great Lakes Embayment Survey Design.

Olsen, T. (2009, January). USEPA. National Coastal Assessment 2010 Great Lakes Survey Design.

Stevens, D.L., Jr. (1997). Variable density grid-based sampling designs for continuous spatial populations. *Environmetrics*, 8(3), 167-95.

Stevens, D. L., Jr., & Olsen, A. R. (1999). Spatially restricted surveys over time for aquatic resources. *Journal of Agricultural, Biological, and Environmental Statistics*, 4(4), 415-428.

Stevens, D. L., Jr., & Olsen, A. R. (2003). Variance estimation for spatially balanced samples of environmental resources. *Environmetrics*, *14*(6), 593-610.

Stevens, D. L., Jr., & Olsen, A. R. (2004). Spatially-balanced sampling of natural resources in the presence of frame imperfections. *Journal of the American Statistical Association*, *99*(465), 262-278.

Sources – Benthic Index

Environment Canada and U.S. Environmental Protection Agency (EC and USEPA). (2013). *State of the Great Lakes 2011.* Cat No. En161-3/1-2011E-PDF. EPA 950-R-13-002. Available at http://binational.net

Howmiller, R. P., & Scott, M. A. (1977). An environmental index based on relative abundance of oligochaete species. *Journal of Water Pollution Control Federation*, 49, 809-815.

Lauritsen, D. D., Mozley, S. C., & White, D. S. (1985). Distribution of oligochaetes in Lake Michigan and comments on their use as indices of pollution. *Journal of Great Lakes Research*, *11*(1), 67-76.

Milbrink, G. (1983). An improved environmental index based on the relative abundance of oligochaete species. *Hydrobiologia,* 102(2), 89-97.

Sources – Water Quality Index

Costantini, M., Kolesar, S., Ludsin, S. A., Mason, D. M., Rae, C. M., & Zhang, H. (2011). Does hypoxia reduce habitat quality for Lake Erie walleye? A bioenergetics perspective. *Canadian Journal of Fisheries and Aquatic Sciences*, 68(5), 857-879.

Diaz, R. J., & Rosenberg, R. (1995). Marine benthic hypoxia: A review of its ecological effects and the behavioral responses of benthic macrofauna. *Oceanography and Marine Biology Annual Review*, 33, 245-303.

International Joint Commission (IJC). (1979). *Trophic characterization of the US and Canadian nearshore zones of the Great Lakes*. Available at www.ijc.org/files/publications/ID530.pdf

Krieger, K. A., & Bur, M. T. (2009). *Nearshore hypoxia as a new Lake Erie metric*. Lake Erie Protection Fund Project SG334-07. Available at http://lakeerie.ohio.gov/Portals/0/Closed%20Grants/small%20grants/SG%20334-07%20Final%20Report.pdf

Phosphorus Management Strategies Task Force (PMSTF). (1980, July). *Phosphorus management for the Great Lakes: Final report of the Phosphorus Management Strategies Task Force to the International Joint Commission's Great Lakes Water Quality Board and Great Lakes Science Advisory Board*. Available at http://agrienvarchive.ca/download/P-management_G_lakes.pdf

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010a). *National coastal condition assessment: Field operations manual* (EPA-841-R-09-003). Washington, DC: Author.

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010b). *National coastal condition assessment: Laboratory methods manual* (EPA 841-R-09-002). Washington, DC: Author.

U.S. Environmental Protection Agency, Office of Water (USEPA). (2010c). *National coastal condition assessment: Quality assurance project plan 2008-2012* (EPA/841-R-09-004). Washington, DC: Author.

U.S. Environmental Protection Agency, Office of Research and Development & Office of Water (USEPA). (2011). *National coastal condition report IV* (EPA-842-R-10-003). Washington, DC: Author.

Sources – Sediment Index

Canadian Council of Ministers of the Environment (CCME). (1999). Canadian Protocol for the Derivation of Canadian Sediment Quality Guidelines for the Protection of Aquatic Life Canadian Council of Ministers of the Environment 1995. (CCME EPC-98E). Available at http://ceqg-rcqe.ccme.ca/download/en/226

Crane, J. L., MacDonald, D. D., Ingersoll, C. G., Smorong, D. E., Lindskoog, R. A., Severn, C. G., Berger, T. A., & Field, L. J. (2002). Evaluation of numerical sediment quality targets for the St. Louis River Area of Concern. *Archives of Environmental Contamination and Toxicology*, 43, 1-10.

Crane, J., & Hennes, S. (2007). Guidance for the use and application of sediment quality targets for the protection of sediment–dwelling organisms in Minnesota. Minnesota Pollution Control Agency. MPCA Document: tdr-gl-04.

Ingersoll, C. G., MacDonald, D. D., Wang, N., Crane, J. L., Field, L. J., Haverland, P. S., & Smorong, D. E. (2001). Predictions of sediment toxicity using consensus-based freshwater sediment quality guidelines. *Archives of Environmental Contamination and Toxicology*, *41*(1), 8-21.

MacDonald, D. D., Ingersoll, C. G., & Berger, T. A. (2000). Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology*, *39*(1), 20-31.

U.S. Environmental Protection Agency, Office of Science and Technology, Standards and Health Protection Division (USEPA). (2004). *The incidence and severity of sediment contamination in surface waters of the United States, National Sediment Quality Survey* (2nd ed., EPA-823-R-04-007). Washington, DC: Author.

Sources – Fish Contaminant

Agency for Toxic Substances and Disease Registry (ATSDR). (2000, September). Toxicological profile for endosulfan.

Agency for Toxic Substances and Disease Registry (ATSDR). (2002a, September). Toxicological profile for hexachlorobenzene.

Agency for Toxic Substances and Disease Registry (ATSDR). (2002b, September). Toxicological profile for dieldrin.

Agency for Toxic Substances and Disease Registry (ATSDR). (2008, September). Toxicological profile for cadmium.

Aguado-Giménez, F., & Garcia-Garcia, B. (2005). Growth, food intake and feed conversion rates in captive Atlantic bluefin tuna (*Thunnus thynnus* Linnaeus, 1758) under fattening conditions. *Aquaculture,* 249, 303-309.

Andrews, A. K., Van Valin, C. C., & Stebbings, B. E. (1966). Some effects of heptachlor on bluegills (*Lepomis macrochirus*). *Transactions of the American Fisheries Society*, 95, 297-309.

Argyle, R. L., Williams, G. C., & Dupree, H. K. (1973). Endrin uptake and release by fingerling channel catfish (*Ictalarus punctatus*). *Journal of the Fisheries Research Board of Canada*, 30, 1743-1744.

Argyle, R. L., Williams, G. C., & Daniel, C. B. (1975). Dieldrin in the diet of channel catfish (*Ictalurus punctatus*): uptake and effect on growth. *Journal of the Fisheries Research Board of Canada,* 32, 2197-2204.

Becker, G. C. (1983). *Fishes of Wisconsin*. University of Wisconsin Press.

Berntssen, H. G., Aatland, A., & Handy, R. D. (2003). Chronic dietary mercury exposure causes oxidative stress, brain lesions, and altered behaviour in Atlantic salmon (*Salmo salar*) parr. *Aquatic Toxicology*, 65, 55-72.

Born, E. W., Rysgaard, S., Ehlme´, G., Sejr, M., Acquarone, M., & Levermann, N. (2003). Underwater observations of foraging free-living Atlantic walruses (*Odobenus rosmarus*) and estimates of their food consumption. *Polar Biology*, 26, 348-357.

Buhler, D. R., & Shanks, W. E. (1970). Influence of body weight on chronic oral DDT toxicity in coho salmon. *Journal of the Fisheries Research Board of Canada,* 27, 347-356.

Carlander, K. D. (1969). *Handbook of freshwater fishery biology*. Volume 1. The Iowa State University Press, Ames. Iowa.

Carlander, K. D. (1977). *Handbook of freshwater fishery biology*. Volume 2. The Iowa State University Press, Ames. Iowa.

Chapman, P. M., Fairbrother, A. & Brown, D. (1998). A critical evaluation of safety (uncertainty) factors for ecological risk assessment. *Environmental Toxicology and Chemistry,* 17, 99-108.

Cossarini-Dunier, M., Monod, G., Damael, A., & Lepot, D. (1987). Effect of γ-Hexachlorocyclohexane (Lindane) on carp (Cyprinus carpio). I. Effect of chronic intoxication on humoral immunity in relation to tissue pollutant levels. *Ecotoxicology and Environmental Safety,* 13, 339-345.

Coulston, F., & Kolbye, A. C., Jr. (eds). 1994. Interpretive review of the potential adverse effects of chlorinated organic chemicals on human health and the environment. *Regulatory Toxicology and* Pharmacology. 20, S1-S1056.

Fahraeus-van Ree, G. E., & Payne, J. F. (1997). Effect of toxaphene on reproduction of fish. *Chemosphere*, *34*(4), 855-867.

Giesy, J. P., Jones, P. D., Kannan, K., Newsted, J. L., Tillitt, D. E., & Williams, L. L. (2002). Effects of chronic dietary exposure to environmentally relevant concentrations of 2,3,7,8tetrachlorodibenzo-p-dioxin on survival, growth, reproduction and biochemical responses of female rainbow trout (Oncorhynchus mykiss). *Aquatic Toxicology,* 59, 35-53.

Griffiths, S. P., Kuhnert, P. M., Fry, G. F. & Manson, F. J. (2009). Temporal and size-related variation in the diet, consumption rate, and daily ration of mackerel tuna (*Euthynnus affinis*) in neritic waters of eastern Australia. *Ices Journal of Marine Science*, 66(4), 720-733.

Heinz, G. H., & Locke, L. N. (1976). Brain lesions in mallard ducklings from parents fed methylmercury. *Avian Diseases*, *20*(1), 9-17.

Hyde, K. M., Graves, J. B., Watts, A. B., & Bonner, F. L. (1973). Reproductive success of mallard ducks fed Mirex. *The Journal of Wildlife Management*, 37(4), 479-484.

Kastelein, R. A., Vaughan, N., Walton, S. & Wiepkema, P. R. (2002). Food intake and body measurements of Atlantic bottlenose dolphins (*Tursiops truncates*) in captivity. *Marine Environmental Research*, 53, 199–218.

Leatherland, J. F. & Sonstegard, R. A. (1980). Effect of dietary Mirex and PCB's in combination with food deprivation and testosterone administration on thyroid activity and bioaccumulation of organochlorines in rainbow trout Salmo gairdeneri Richardson. *Journal of Fish Diseases,* 3, 115-124.

Lundebye, E.J. Lock, D. Boyle, K. Ruohonen, M.H. Berntssen. (2010). Tolerance of Atlantic salmon (*Salmo salar*) to dietborne endosulfan assessed by haematology, biochemistry, histology and growth. Aquaculture Nutrition. Vol 16 Issue 5. 549-558.

Macek, K. J., Rodgers, C. R., Stalling, D. L., & Korn, S. (1970). The uptake, distribution and elimination of dietary 14C-DDT and 14C-Dieldrin in rainbow trout. *Transactions of the American Fisheries Society,* 4, 689-695.

Maldeniya, R. (1996). Food consumption of yellowfin tuna, *Thunnus albacares*, in Sri Lankan waters. *Environmental Biology of Fishes,* 47(1), 101-107.

Nagy, K. A. (1987). Field metabolic rate and food requirement scaling in mammals and birds. *Ecological Monographs*, *57*(2) 111-128.

National Park Service (NPS). (2015) Great Lakes Water Quality Projects Updates. Available at http://nature.nps.gov/water/oceancoastal/waterquality_greatlakes_updates.cfm

Nakayama, K., Oshima, Y., Yamaguchi, T., Tsuruda, Y., Kang, I. J., Kobayashi, M., Imada, N., & Honjo, T. (2004). Fertilization success and sexual behavior in male medaka, Oryzias latipes, exposed to tributyltin. *Chemosphere*, 55, 1331–1337.

National Toxicology Program (NTP). (1990). *Toxicology and Carcinogenesis Studies of MIREX (CAS No. 2385-85-5) in F344/N Rats (Feed Studies).* NTP TR 313.

Ogle, S. E., & Knight, A. W. (1989). Effects of elevated foodborne selenium on growth and reproduction of the fathead minnow (Pimephales promelas). *Archives of Environmental Contamination and Toxicology*, 18, 795-803.

Pedlar, R. M., Ptashynski, M. D., Evans, R., & Klaverkamp, J. F. (2002). Toxicological effects of dietary arsenic exposure in lake whitefish (Coregonus clupeaformis). *Aquatic Toxicology*, *57*(3), 167-189.

Sample, B. E., Opresko, D. M., & Suter, G. W. II. (1996). *Toxicological Benchmarks for Wildlife: 1996 revision* (ES/ER/TM-86/R3). Oak Ridge, TN: Health Sciences Research Division Risk Assessment Program.

Sample, B. E. & Arenal, C. A. (1999). Allometric models for interspecies extrapolation of wildlife toxicity data. *Bulletin of Environmental Contamination and Toxicology*, 62, 653-663.

Schreiber, R.W. (1976). Growth and development of nestling brown pelicans. *Bird-Banding*, 47(1), 19-39.

Skea, J.C., Simonin, H.J., Jackling, S., & Symula, J. (1981). Accumulation and retention of Mirex by brook trout fed a contaminated diet. *Bulletin of Environmental Contamination and Toxicology*, 27, 79-83.

Stillwell, C.E., & Kohler, N.E. (1985). Food and feeding ecology of the swordfish *Xiphias gladius* in the western North Atlantic Ocean with estimates of daily ration. *Marine Ecology Progress Series*, 22, 239-247.

Stillwell, C. E., & Kohler, N. E. (1993). Food habits of the sandbar shark *Carcharhinus plumbeus* off the U.S. northeast coast, with estimates of daily ration. *Fishery Bulletin,* 91, 138-150.

Szczerbik, P., Mikołajczyk, T., Sokołowska-Mikołajczyk, M., Socha, M., Chyb, J., & Epler, P. (2006). Influence of long-term exposure to dietary cadmium on growth, maturation and reproduction of goldfish (subspecies: Prussian carp Carassius auratus gibelio B.). *Aquatic Toxicology,* 77, 126-135.

TERRETOX. (2002). On-line terrestrial toxicity database. U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency (USEPA). (1972). *Heptachlor: A review of its uses, chemistry, environmental hazards, and toxicology*. Available at http://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=9100CQF3.txt

U.S. Environmental Protection Agency (USEPA). (1993). *Wildlife Exposure Factors Handbook. Volume I of II.* (EPA/600/R-93/187a).

U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology (USEPA). (1995). *Great Lakes Water Quality Initiative criteria documents for the protection of wildlife: DDT, mercury, 2,3,7,8-TCDD, PCBs* (EPA/820/B-95/008). Washington, DC: Author.

U.S. Environmental Protection Agency, Environmental Response Team (USEPA). (1997). *Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments – Interim final* (EPA/540/R-97/006). Edison, NJ: Author.

U.S. Environmental Protection Agency (USEPA). (1998). *Guidelines for Ecological Risk Assessment* (EPA/630/R-95/002F).

U.S. Environmental Protection Agency (USEPA). (1999). *Issuance of final Guidance: Ecological risk Assessment and Risk Management Principles for Superfund Sites.* OSWER Directive 9285.7-28.

U.S. Department of the Interior, Fish and Wildlife Service (USDI). (1964). Pesticide-wildlife studies, 1963: a review of Fish and Wildlife Service investigations during the calendar year. FWS Circular 199.

Wentsel, R. S., LaPoint, T. W., Simini, M., Checkai, R., Ludwig, D. & Brewer, L. (1996, May). *Tri-Service Procedural Guidelines for Ecological Risk Assessments*. Air Force Center for Environmental Excellence (AFCEE), U.S. Army Environmental Center (USAEC), Naval Facilities Engineering Service Center (NFESC).

Wiemeyer, S.N. (1996). *Other organochlorine pesticides in birds*. Pages 99-115 IN Beyer, W. N., Heinz, G. H., & Redmon-Norwood, A.W. (eds). Environmental contaminants in wildlife: interpreting tissue concentrations. Lewis Publishers, Boca Raton, FL. 494 pp.

Woodburn, K. B., Marino, T. A., McClymont, E. L., & Rick, D. L. (2008). Determination of the dietary absorption efficiency of hexachlorobenzene with the channel catfish (Ictalurus punctatus). *Ecotoxicology and Environmental Safety,* 71, 419-425.

Wood, A. D., Wetherbee, B. M., Juanes, F., Kohler, N. E., & Wilga, C. (2009). Recalculated diet and daily ration of the shortfin mako (*Isurus oxyrinchus*), with a focus on quantifying predation on bluefish (*Pomatomus saltatrix*) in the northwest Atlantic Ocean. *Fishery Bulletin,* 107, 76-88.