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Introduction

The Governments of Canada and the United States, together with their many Great Lakes Water Quality Agreement (GLWQA) partners, have established a set of nine science-based indicators of ecosystem health directly linked to the Objectives of the GLWQA. These nine indicators are supported by sub-indicators that are collectively used to assess the status of the Great Lakes ecosystem and track progress towards achieving the GLWQA General Objectives. This *State of the Great Lakes 2022 Technical Report* contains the 40 sub-indicator reports used to develop the *State of the Great Lakes 2022 Report*.

More than 110 government and non-government Great Lakes scientists and other experts analyzed available data (for most sub-indicators this includes data up to 2019 or 2020) to determine an assessment of condition for each indicator in relation to both ecosystem status and trend over time. Status is described in terms of Good, Fair or Poor conditions. Trends are described as Improving, Unchanging, or Deteriorating and are most often assessed over a 10-year period. Some sub-indicators are not assessed every reporting cycle due to differences in monitoring frequency. In those cases, the assessment from the previous report is presented.

A Summary of Great Lakes 2022 Assessments

Table 1. Summary of assessments for each indicator (assessed at the Great Lakes basin scale) and each sub-indicator (assessed at the Lake scale).

INDICATOR	SUB-INDICATOR	LAKE SUPERIOR	LAKE MICHIGAN	LAKE HURON	LAKE ERIE	LAKE ONTARIO	OVERALL ASSESSMENT
Drinking Water	Treated Drinking Water	Good & Undetermined	Good & Undetermined	Good & Undetermined	Good & Undetermined	Good & Undetermined	**Good & Unchanging**
Beaches	Beach Advisories	Good & Unchanging	Good & Unchanging	Good & Unchanging to Improving	Fair & Unchanging	Good & Improving	**Good & Unchanging to Improving**
Fish Consumption	Contaminants in Edible Fish	Fair & Unchanging	Fair & Improving	Good & Unchanging to Improving	Fair & Improving	Fair & Improving	**Fair & Improving**
Toxic Chemicals	Toxic Chemicals in Sediment	Good & Unchanging	Fair & Unchanging	Good & Unchanging	Fair & Improving	Fair & Improving	**Fair & Unchanging to Improving**
	Toxic Chemicals in Water	Fair & Improving	Fair & Undetermined	Good & Unchanging	Fair & Unchanging	Fair & Unchanging	
	Toxic Chemicals in Whole Fish	Fair & Unchanging	Fair & Unchanging	Fair & Unchanging	Fair & Unchanging	Fair & Unchanging	
	Toxic Chemicals in Herring Gull Eggs	Good & Improving	Good & Improving	Good & Improving	Good & Unchanging	Good & Improving	
	Toxic Chemicals in the Atmosphere	Fair & Improving (No lake-by-lake assessments were determined)					
Habitat & Species	Coastal Wetland Invertebrates	Fair & Undetermined	Fair & Undetermined	Fair & Unchanging	Undetermined	Undetermined	**Fair & Unchanging**
	Coastal Wetland Fish	Fair & Unchanging	Fair & Undetermined	Fair & Unchanging	Poor & Undetermined	Fair & Unchanging	
	Coastal Wetland Amphibians	Fair & Undetermined	Fair & Undetermined	Good & Unchanging	Fair & Unchanging	Fair & Improving	
	Coastal Wetland Birds	Fair & Undetermined	Fair & Undetermined	Good & Unchanging	Fair & Unchanging	Fair & Improving	
	Coastal Wetland Plants	Good & Unchanging	Fair & Unchanging	Fair & Unchanging	Poor & Unchanging	Poor & Unchanging	
	Aquatic Habitat Connectivity	Fair & Improving	Poor & Improving	Fair & Improving	Fair & Improving	Fair & Improving	
	Phytoplankton	Good & Deteriorating	Fair & Deteriorating	Fair & Deteriorating	Poor & Deteriorating	Good & Unchanging	
	Zooplankton	Good & Unchanging	Good & Unchanging	Fair & Unchanging	Good & Improving	Good & Unchanging	
	Benthos	Good & Unchanging	Good & Unchanging	Good & Unchanging	Poor & Unchanging	Fair & Unchanging	
	Diporeia	Good & Unchanging	Poor & Deteriorating	Poor & Deteriorating	Poor & Unchanging	Poor & Unchanging	
	Lake Sturgeon	Poor & Unchanging	Poor & Improving	Poor & Improving	Poor & Improving	Poor & Improving	
	Native Prey Fish Diversity	Good & Unchanging	Fair & Unchanging	Fair & Unchanging	Fair & Deteriorating	Fair & Improving	
	Lake Trout	Good & Improving	Fair & Improving	Fair & Improving	Fair & Improving	Fair & Improving	
	Walleye	Fair & Improving	Good & Unchanging	Good & Unchanging	Good & Improving	Good & Unchanging	
Nutrients and Algae	Nutrients in Lakes	Good & Unchanging	Fair & Unchanging	Fair & Unchanging	Poor & Unchanging	Fair & Unchanging	**Fair & Unchanging**
	Harmful Algal Blooms: nearshore & embayments	Good & Undetermined	Fair & Unchanging	Fair & Unchanging	Poor & Improving	Good & Unchanging	
	Cladophora	Good & Unchanging	Poor & Unchanging	Fair & Undetermined	Poor & Unchanging	Poor & Undetermined	
Invasive Species	Rate of New ANS Establishment ...into GL basin	The overall Great Lakes basin assessment is Good & Unchanging.					**Overall basin-wide assessment for Rate of New ANS: Good & Unchanging**; **Overall assessment for Impacts of Invasive Species: Poor & Unchanging**
	Rate of New ANS Establishment ...into each lake basin	Poor & Improving	Fair & Improving	Poor & Undetermined	Fair & Undetermined	Fair & Unchanging	
	Impacts of Aquatic Invasive Species	Poor & Undetermined	Poor & Undetermined	Poor & Undetermined	Poor & Deteriorating	Poor & Deteriorating	
	Sea Lamprey	Poor & Deteriorating	Good & Improving	Fair & Improving	Good & Improving	Good & Improving	
	Dreissenid Mussels	Good & Unchanging	Poor & Deteriorating	Poor & Deteriorating	Fair & Unchanging	Poor & Deteriorating	
	Terrestrial Invasive Species	The assessment methodology is being updated. This sub-indicator is currently assessed as Undetermined.					
Groundwater	Groundwater Quality	Good & Undetermined	Good & Undetermined	Good & Undetermined	Good & Undetermined	Fair & Undetermined	**Good & Undetermined**
Watershed Impacts	Forest Cover	Good & Improving	Fair & Unchanging	Fair & Unchanging	Poor & Unchanging	Fair & Unchanging	**Fair & Unchanging**
	Land Cover	Good & Unchanging	Fair & Unchanging	Fair & Unchanging	Poor & Deteriorating	Fair & Deteriorating	
	Hardened Shorelines	Good & Undetermined	Good & Deteriorating	Good & Deteriorating	Poor & Deteriorating	Poor & Deteriorating	
	Water Quality in Tributaries	Undetermined	Not Assessed	Fair & Unchanging	Poor & Unchanging	Fair & Unchanging	
	Human Population	Unchanging	Increasing	Increasing	Increasing	Increasing	
Climate Trends	Precipitation Amounts (1950-2020)	Unchanging	Increasing		Increasing	Increasing	**No overall assessment**
	Water Levels (1918-2020)	Unchanging	Unchanging		Increasing	Unchanging	
	Surface Water Temperature (1980-2020)	Increasing	Increasing	Increasing	Increasing	Increasing	
	Ice Cover (1973 - 2020)	Decreasing	Decreasing	Decreasing	Decreasing	Decreasing	

STATUS

Good Fair Poor Undetermined

Light grey cells in the sub-indicator column indicate sub-indicators not used in overall assessments, however full sub-indicator reports are available to provide more detailed information.

The Invasive Species indicator assessment is determined using only the *Rate of New Aquatic Non-indigenous Species Establishment in the Great Lakes* sub-indicator and the *Impacts of Aquatic Invasive Species* sub-indicator. Both of these sub-indicators account for the presence and impact of Sea Lamprey and Dreissenid Mussels in the Great Lakes basin. Sub-indicator assessments of Sea Lamprey and Dreissenid Mussels are also reported to provide additional information on the status and trends of these species. The *Terrestrial Invasive Species* sub-indicator is currently undergoing a review of the assessment methodology, and is not assessed, however the report includes information on invasive terrestrial species that have an impact on the Great Lakes ecosystem.

Table 2. Summary of assessments for each lake and the overall Great Lakes assessment.

	LAKE SUPERIOR	LAKE MICHIGAN	LAKE HURON	LAKE ERIE	LAKE ONTARIO	OVERALL GREAT LAKES
OVERALL ASSESSMENTS	Good & Unchanging	Fair & Unchanging	Good & Unchanging	Poor & Unchanging	Fair & Unchanging to Improving	Fair & Unchanging

STATUS

Good Fair Poor Undetermined

Sub-Indicator: Treated Drinking Water

Overall Assessment

Note: The overall assessment is based on measures of treated water: population served by community water systems (CWSs) that met all health-based standards for the U.S. and treated water sample results in compliance with water quality standards for Ontario. Source water information for Ontario is included in rationale for context only.

Status: Good

Trends:

10-Year Trend: Unchanging

Long-term Trend (Ontario: 2004-2020; U.S.: Undetermined): Unchanging

Rationale:

Ontario: Each year from 2004-2020, 99.74%-99.88% of treated drinking water tests from all of Ontario's municipal residential systems met Ontario drinking water quality standards for microbes, chemicals and radionuclides. Great Lakes source water quality was compared to Ontario drinking water quality standards for select chemicals and tritium using the Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI, or WQI for short). There was insufficient data for comparison for the years 2012-2017 (from 2012-2016 for Lake Erie). For years with enough data to calculate the WQI, the WQI classified Great Lakes source water quality as 'excellent' or 'good'. Data was insufficient to determine a WQI score for some years from 2017-2020, but when the WQI could be calculated, each lake's WQI scores were the same from 2017-2020 as they were from 2007-2011.

U.S.: During the 2020 calendar year, 19.5 million U.S. residents received drinking water from the 810 community water systems (CWSs) that source from the surface waters of the Great Lakes and its connecting river systems (including the international section of the St. Lawrence River). Overall, 98.5 percent of the 810 CWSs met all health-based standards, and 99.1 percent of this population was served by CWSs that met all health-based standards, achieving 'Good' ratings for drinking water quality for the U.S. side of the Great Lakes Basin.

Lake-by-Lake Assessment

Note: Lake-by-Lake Assessments are based on measures of treated water for the U.S. (population served by CWSs that met all health-based standards) and source water quality for Ontario (concentrations of contaminants in source water). See 'Status Assessment Definitions', 'Trend Assessment Definitions', and 'Measures' sections below for more information. There was insufficient data to assess a 10-year trend for the Ontario data, therefore the assessments for all lakes is Undetermined. Lake-by-Lake assessments only assessed 2020 U.S. data for this report, so there is not a 10-year or long-term period to reference. Thus, the assessment is Undetermined.

Lake Superior

Status: Good (U.S. treated drinking water: Good; Ontario source water: Good)

10-Year Trend: Undetermined

Long-term Trend:

Ontario: (2007-2019): Unchanging

U.S.: Undetermined

Rationale:

Ontario: Water quality index (CCME WQI) result for source water is 100.

U.S.: There are 17 CWSs that source from the surface waters of Lake Superior, which serve a population of 174,008 people. The 'Good' rating is based on 94.3 percent of this population was served by CWSs that met all health-based standards, and 15 of the 17 CWSs (88.2%) met all health-based standards, achieving a 'Fair' rating.

Lake Michigan

Status: Good

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: Lake Michigan is wholly within the United States. There are 356 CWSs that source from the surface waters of Lake Michigan, which serve a population of 9,552,028 people. The 'Good' ratings are based on 99.4 percent of these 356 CWSs that met all health-based standards, and 99.1 percent of this population was served by CWSs that met all health-based standards.

Lake Huron (including St. Marys River)

Status: Good (U.S. treated drinking water: Good; Ontario source water: Good)

10-Year Trend: Undetermined

Long-term Trend:

Ontario (2007-2019): Unchanging

U.S.: Undetermined

Rationale:

Ontario: Water quality index (CCME WQI) result of 100.

U.S.: There are 133 CWSs that source from the surface waters of Lake Huron (n=132) and the St. Marys River (n=1), which serve a population of 1,678,942 people. The 'Good' ratings are based on 99.2 percent of these 133 CWSs that met all health-based standards, and 99.1 percent of this population was served by CWSs that met all health-based standards.

Specifically assessing Lake Huron, of the 132 CWSs that source from the lake, 100 percent met all health-based standards, and 100 percent of the population was served by CWSs that met all health-based standards, both measures achieving a 'Good' rating.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Good (U.S. treated drinking water: Good; Ontario source water: Good)

10-Year Trend: Undetermined

Long-term Trend:

Ontario (2007-2019): Unchanging

U.S.: Undetermined

Rationale:

Ontario: Water quality index (CCME WQI) results of 100.

U.S.: There are 205 CWSs that source from the surface waters of Lake Erie (n=117) and the St. Clair-Detroit River (n=88), which serve a population of 6,705,016 people. The 'Good' ratings are based on 96.6 percent of these 205 CWSs that met all health-based standards, and 99.0 percent of this population was served by CWSs that met all health-based standards.

Specifically assessing Lake Erie, 95.7 percent of the 117 CWSs that source from the lake met all health-based standards, and 99.1 percent of the population was served by CWSs that met all health-based standards, both measures achieving a 'Good' rating.

Specifically assessing the St. Clair-Detroit River, 97.7 percent of the 88 CWSs that source from the St. Clair-Detroit River met all health-based standards, and 98.8 percent of the population was served by CWSs that met all health-based standards, both measures achieving a 'Good' rating.

Lake Ontario (including Niagara River, and International section of the St. Lawrence River)

Status: Good (U.S. treated drinking water: Good; Ontario source water: Good)

10-Year Trend: Undetermined

Long-term Trend:

Ontario (2007-2020): Unchanging

U.S.: Undetermined

Rationale:

Ontario*: Water quality index (CCME WQI) results between 94 and 95.

U.S.: There are 99 CWSs that source from the surface waters of the Niagara River (n=30)**, Lake Ontario (n=51) and the international section of the St. Lawrence River (n=18), which serve a population of 1,443,789 people. The 'Good' ratings are based on 100 percent of the 99 CWSs that met all health-based standards, and 100 percent of the population was served by CWSs that met all health-based standards.

*****Note: The Ontario analysis includes three drinking water systems (Port Colbourne, Welland and St. Catherines) which source their water from the Welland Canal.

******Note: The CWSs in the U.S. that rely on the Niagara River for source water all obtain water from the Upper Niagara River (i.e., upstream of Niagara Falls). No CWSs in the U.S. draw water from the Lower Niagara River (i.e., downstream of Niagara Falls).

Status Assessment Definitions

Ontario

Treated water quality: Based on the percentage of treated drinking water tests meeting the Ontario drinking water quality standards as follows:

Good: At least 99%

Fair: 97-98.9%

Poor: Less than 97%

Undetermined: Data are not available or are insufficient to determine the percent of treated drinking water tests meeting standards

Source water quality: Based on values calculated by the Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI).

For State of the Great Lakes reporting, CCME WQI values are grouped as follows:

Good: 80-100

Fair: 65-79

Poor: 0-64

Undetermined: Data are not available or are insufficient to determine the CCME WQI value

United States

Good: At least 90 percent of CWSs met all health-based standards and at least 90 percent the population was served by CWSs that met all health-based standards.

Fair: 80 percent to 90 percent of CWSs met all health-based standards and 80 to 90 percent of the population was served by CWSs that met all health-based standards.

Poor: Less than 80 percent of CWSs met all health-based standards and less than 80 percent of the population was served by CWSs that met all health-based standards.

Undetermined: Data were not available or were insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Ontario

Treated water quality:

Improving: Increase in percent of treated drinking water test results meeting standards.

Unchanging: No change in percent of treated drinking water tests meeting standards.

Deteriorating: Decrease in percent of treated drinking water tests meeting standards.

Undetermined: Data are not available or are insufficient to determine the trend in percent of treated drinking water test results meeting standards.

Source water quality:

Improving: Increase in CCME WQI value

Unchanging: No change in CCME WQI value

Deteriorating: Decrease in CCME WQI value

Undetermined: Data are not available or are insufficient to determine the CCME WQI value

United States

Improving: Increase in the proportion of the population served treated drinking water that met all health-based standards.

Unchanging: No change in the proportion of the population served treated drinking water that met all health-based standards.

Deteriorating: Decrease in the proportion of the population served treated drinking water that met all health-based standards.

Undetermined: Trend in health-based violations of treated drinking water was not determined or data were not available.

Endpoints and/or Targets

Levels of disease-causing organisms, concentrations of hazardous or toxic chemicals, and radioactive substances should not exceed established U.S. federal, state and provincial human health objectives, drinking water standards or guidelines.

Sub-Indicator Purpose

The purposes of this sub-indicator are to:

- Assess chemical, microbial and radiological contaminant levels in drinking water;
- Evaluate the potential for human exposure to drinking water contaminants; and,
- Evaluate the efficacy of policies and technologies to ensure safe drinking water.

Ecosystem Objective

As vital sources of drinking water, the objective is for the Great Lakes ecosystem to be protected from harmful microbial, chemical, and radiological contaminants that require advanced treatments to meet all drinking water standards.

This sub-indicator best supports work towards General Objective #1 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should “be a source of safe, high-quality drinking water.”

Measure

Due to substantial differences in the drinking water regulatory structures between the two nations, this indicator was assessed independently by both nations with specific measures and analyses for each.

Ontario

Source water quality was compared to Ontario drinking water quality standards, using the Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI). Since 2001, the CCME WQI has been used extensively in Canada and throughout the world for reporting on the state of water quality. Values calculated by the CCME WQI are based on a combination of three factors:

- The number of parameters that don't meet standards (scope)
- The frequency with which the standards aren't met (frequency)
- The amount by which the standards aren't met (amplitude)

Ontario drinking water quality standards apply to treated water only; however, comparing source water quality to the standards provides useful insights into the state of the Great Lakes.

The treated drinking water assessment criterion is the percentage of tests meeting Ontario drinking water quality standards.

United States

For the U.S.-portion of the Great Lakes Basin, drinking water quality was assessed by calculating the proportion of the population served by community water systems (CWSs) that provide drinking water sourced from the surface waters of the Great Lakes that meets all applicable health-based water standards, as reported in the Safe Drinking Water Information System (SDWIS) Federal Reporting Services system. Health-based violations (HBVs) may occur in the finished (i.e., treated) drinking water of a public water system (includes drinking water treatment plants and their distribution systems). HBVs may be issued due to a public water system's failure to comply with requirements for a treatment technique or sampling results with elevated levels of contaminants, such as chemicals (e.g., arsenic, atrazine, nitrate, nitrite, radionuclides, heavy metals), disinfection byproducts (e.g., haloacetic acids, trihalomethanes), or microorganisms (e.g., *E. coli*).[1]

To rate U.S. drinking water quality, the following indicators were used:

- Proportion of CWSs that met all health-based standards
- Proportion of population served by CWSs that met all health-based standards[2]

Ecological Condition

The quality of source water and its intended use applications determine the level of treatment required (i.e., fit-for-purpose specifications). Drinking water has stringent standards and treatment requirements to protect public

[1] See *Drinking Water Regulations* for more information about U.S. EPA's health-based standards.

[2] Because the drinking water sub-indicator is part of the human health assessment, the proportion of population served by CWSs that met all health-based standards was the measure that determined the overall and the lake-by-lake statuses.

health, whereas different standards and treatment can be appropriate for non-potable purposes, such as irrigation, fire suppression, toilet flushing, and industrial processes. For all use applications, protecting source water quality is an important strategy to reduce treatment needs and expenses. Governments have enacted laws and support programs requiring monitoring and reporting, treatment for contaminant removal, and source water protection, which are described below.

Ontario

Even good quality source water requires treatment to make it safe to drink. To lower the risk of source water contamination reaching consumers' taps, and to keep drinking water treatment costs low, continual efforts should be made to decrease microbial, chemical and radiological contamination of source water.

The Ontario Ministry of the Environment, Conservation and Parks provided the data for the Canadian component of this report, as follows:

- Source water data for 2007-2017 are from the Drinking Water Information Management System (DWIMS). Source water data for 2018-2020 are from DWIMS and the Drinking Water Information System (DWIS). DWIMS is the database of the Drinking Water Surveillance Program, a scientific program run in partnership with volunteer municipalities and First Nations. DWIS is the database of the Drinking Water and Environmental Compliance Division which is the Ministry's regulatory compliance division.
- Treated water data for all years (2004-2020) is from DWIS.

Source Water

Eleven parameters were chosen for the CCME WQI. They represent a wide spectrum of threats to drinking water quality. Some have been found at high levels in source water and a few have been found at high levels in treated drinking water. The eleven parameters are:

- Glyphosate - accounts for about 54% of all agricultural pesticide use in Ontario.
- Atrazine + dealkylated metabolites - the most frequently detected pesticide in Ontario's treated drinking water.
- 2,4-D has been detected in Ontario's treated drinking water.
- Microcystin-LR - one of the most toxic and commonly occurring algal bloom toxins in Ontario.
- Nitrilotriacetic acid - a widely used replacement for phosphate in detergents and other cleaning products. It was detected at high levels in source water at two Ontario water treatment plants, including one on Lake Huron.
- Toluene - of the gasoline components BTEX (benzene, toluene, ethylbenzene and xylene), it is the most commonly detected in Drinking Water Surveillance program (DWSP) samples. Also used as a solvent and as an intermediate in chemical manufacturing.
- Nitrate - a commonly occurring contaminant from fertilizers, sewage plants and livestock waste.
- Nitrite - a commonly occurring contaminant from fertilizers, sewage plants and livestock waste.
- Arsenic - was detected at a high level in a source water sample from an Ontario water treatment plant.
- Sodium - one of the main components of road salt.
- Tritium - a radioactive compound released in spills from nuclear power plants.

Source water data were from 79 Great Lakes water treatment plants as follows:

- Lake Ontario – 36 plants
- Lake Erie – 18 plants
- Lake Huron – 21 plants
- Lake Superior – 4 plants

Figure 1 shows the location of the plants.

Typically, source water results for any parameter came from a subset of the water treatment plants on each lake because data for any given parameter wasn't available from all plants. As a percent of plants on each lake, subsets were smallest on Lake Huron where data for any given parameter in any given year was often from 4-5 plants.

WQI values were calculated for each lake rather than each water treatment plant since there was insufficient data at many water treatment plants to calculate a WQI value.

Table 1 presents the annual CCME WQI values for each lake's source water. Where there wasn't enough data to calculate annual WQI values, there was sometimes enough data over two- or three-year periods to calculate the WQI value for those periods. However, even with the option to group years, there were still periods with insufficient data.

The CCME WQI classifies water quality as follows:

Excellent: WQI value 95-100
Good: WQI value 80-94
Fair: WQI value 65-79
Marginal: WQI value 45-64
Poor: WQI value 0-44

The best possible WQI value is 100, which indicates that none of the source water measurements were higher than treated water standards for the parameters included in the WQI calculation. The State of the Great Lakes reporting schema classifies water quality as Good, Fair, Poor or undetermined. It doesn't include a category of 'excellent', so for the purposes of this report, source waters with WQI results of 'excellent' or 'good' were classified as having good water quality.

Lakes Erie and Superior had good source water quality with WQI values of 100 for every year or multi-year period that the WQI could be calculated.

Lake Huron had good source water quality with WQI values of 100 for most years they could be calculated, and a value of 94.2 in 2008. In 2008, a nitrilotriacetic acid result from one Lake Huron source water sample was above the Ontario drinking water quality standard.

Lake Ontario had good source water quality. WQI values were between 94 and 95 for every year WQI values could be calculated, except for 2008 when the WQI value was 100. WQI values between 94 and 95 were due to elevated sodium levels in some source water samples, likely caused by road salting in the watershed. For the eight years 2007-2011 and 2018-2020, the annual percentage of samples from Lake Ontario with sodium results above 20 mg/L ranged from 0.0%-6.8%, with a median of 3.6%. Sodium is not toxic and there is no Ontario drinking water quality standard for sodium; however, persons suffering from hypertension or congestive heart disease may require a sodium-restricted diet, in which case the intake of sodium from drinking water could become significant. The local

Medical Officer of Health should be notified when the sodium concentration exceeds 20 mg/L, so that this information may be passed on to local physicians.

The 10-year trend in source water quality could not be determined for any of the lakes because the WQI could not be calculated for the years 2012-2016 and sometimes 2017; however, the long-term trend can be estimated. For each lake, WQI results from 2007-2011 were the same as those from 2017/18-2019/20. This suggests that the long-term trend in each lake is unchanging.

Treated Water

Figure 2 presents treated water test results for the period 2004-2020. The results include microbial, chemical, and radiological tests. They are from all of Ontario's municipal residential drinking water systems. In 2004, there were 729 municipal residential drinking water systems in Ontario. That number decreased to 657 by 2020 due mainly to amalgamation of systems. The majority of these systems do not use Great Lakes water. They use groundwater or water from rivers and inland lakes; however, the Great Lakes are among the best quality source waters in Ontario and therefore the percentages of treated water tests from Great Lakes systems meeting standards would likely be similar to or higher than those shown in Figure 2.

Since 2004, the percentage of tests meeting Ontario drinking water quality standards has remained in the range 99.74% - 99.88%. From 2004-2020, Ontario's treated drinking water quality was consistently good. A small percentage of tests did not meet standards. When a test result does not meet a standard, it does not necessarily mean that the drinking water is unsafe but does mean that the drinking water system owner and operator need to investigate what may have caused the adverse test result and take all steps necessary to resolve it.

United States

In the United States, the principal law governing pollution of the nation's surface waters is the Federal Water Pollution Control Act, better known as the Clean Water Act. Originally enacted in 1948, it was totally revised by amendments in 1972 that gave the act its current dimensions. The 1972 legislation spelled out ambitious programs for water quality improvement that have since been expanded and are still being implemented by industries and municipalities (CRS, 2016).

The Safe Drinking Water Act (SDWA), Title XIV of the Public Health Service Act, is the key federal law for protecting public drinking water supplies from harmful contaminants. First enacted in 1974 and substantially amended in 1986, 1996, and 2018, the act is administered through programs that establish standards and treatment requirements for public drinking water supplies, finance drinking water infrastructure projects, promote water system compliance, and control the underground injection of fluids to protect underground sources of drinking water. The 1974 law established the current federal-state arrangement in which states may be delegated primary implementation and enforcement authority (i.e., primacy) for the drinking water program. The state-administered Public Water Supply Supervision Program remains the basic program for regulating the nation's public water systems (PWSs), and 49 states and the Navajo Nation have assumed this primacy authority (CRS, 2017). As of December 2020, the U.S. Environmental Protection Agency has set legally enforceable standards for 94 contaminants per the National Primary Drinking Water Regulations (NPDWRs) (CRS, 2021).

The SDWA Amendments of 1996 (P.L. 104-182) require all community water systems (CWSs) to provide drinking water quality information annually to their consumers. To satisfy this obligation, CWSs produce an annual

Consumer Confidence Report (CCR).[3] These CCRs provide information including the specific drinking water sources, the availability of source water assessments, and a summary of:

- Potential sources of contamination
- Water treatment processes
- Contaminants detected in finished drinking water
- Violations that occurred
- Other relevant information

Depending on the applicable rules, monitoring for contaminants may be required in the source water (prior to treatment), in the drinking water treatment plant at the point-of-entry to the distribution system (after treatment), and/or in the distribution system. For microbial contaminants, while *Legionella*, *Giardia*, and viruses are regulated under the surface water treatment rules (SWTR), no monitoring is required (treatment technique requirements). *Cryptosporidium* monitoring is required for 24 months for new PWSs under the LT2 rule.[4] Notably, required sampling results are not reported directly to the U.S. EPA except in limited situations, such as direct implementation, rule violations, SDWA six-year review cycles,[5] and unregulated contaminant monitoring rule compliance.[6] Sampling results are recorded in databases maintained by the primacy agencies (e.g., states' departments of environmental protection), and violations of the rules are issued by the primacy agencies to the PWSs and are reported to the U.S. EPA.

For the purposes of this sub-indicator report, the available information assessed by the U.S. EPA focused on finished (i.e., treated) drinking water quality as indicated by health-based violations of CWSs with the primary source of surface water from the Great Lakes or its connecting rivers. Thus, only a small subset of all PWSs was included in our analyses.[7] Excluded were CWSs with the primary source of surface water from inland waterbodies or groundwater, non-community water systems, and systems that serve less than 25 people.

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[3] For more information and to find your local CCR, visit https://www.epa.gov/ccr/ccr-information-consumers.

[4] The Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) went into effect in January 2006 and required surface water systems to conduct 24 months of source water monitoring, calculate an average *Cryptosporidium* concentration, and use those results to determine if their source was vulnerable to contamination, which may have required further treatment. Most PWSs have already fulfilled the monitoring requirements of this rule and are no longer required to report surface water monitoring results to their primacy agency or the U.S. EPA. New surface water systems are subject to this rule. See Surface Water Treatment Rules for more information.

[5] For six-year review contaminant occurrence data, visit https://www.epa.gov/dwsixyearreview/contaminant-occurrence-and-related-data-six-year-review-drinking-water-standards.

[6] For unregulated contaminant monitoring rule data, visit https://www.epa.gov/monitoring-unregulated-drinking-water-contaminants/occurrence-data-unregulated-contaminant.

[7] U.S. EPA has established three broad categories of public water systems (PWSs). A community water system (CWS) serves the same population year-round. A non-transient non-community water system (NTNCWS) regularly supplies water to at least 25 of the same people at least six months per year but not year-round (e.g., schools, factories, office buildings, and hospitals that have their own wells). Transient non-community water systems (TNCWS) provide water in places where people do not remain for long periods of time (e.g., gas stations, restaurants, and campgrounds). Because individuals typically obtain drinking water from multiple PWSs (e.g., home, work, school, community building, restaurant) throughout the day, there may be double

Reducing the number of health-based violations (HBVs) is a component of the U.S. EPA's FY2018-2022 Strategic Plan, which calls for a 25 percent reduction in the number of CWSs that are out of compliance with health-based standards (USEPA, 2021). Data of the number and type of HBVs are recorded and publicly available in the nationwide U.S. EPA Safe Drinking Water Information System (SDWIS) Federal Reporting Services system. HBVs in the U.S. include violations of:

- Maximum Contaminant Level (MCL) - i.e., the highest level of a contaminant that is allowed in drinking water
- Maximum Residual Disinfectant Level (MRDL) - i.e., the highest level of a disinfectant allowed in drinking water
- Treatment Technique (TT) - i.e., a required process intended to control contaminants in drinking water

HBVs are expected to return to compliance as soon as possible but long-term changes may be needed to do so, such as securing additional technical, managerial, and financial support or infrastructure upgrades (e.g., additional advanced treatment processes, changing source water with a new well or intake or a purchase agreement).

Health-Based Violations Assessment

For context, during the 2020 calendar year there were nearly 144,000 active PWSs across the country that served over 330 million people. Across the eight Great Lakes States, there were nearly 58,000 active PWSs that served over 80 million people, of which 11,358 were CWSs that served 73 million people (Table 2).[8]

Categorizing by primary water source, of the 11,358 CWSs across the eight Great Lakes States, only 2,371 (20.9%) were CWSs with the primary source of surface water, which served nearly 52 million people (70.7%) (Table 3). Additionally, there were 8,987 (79.1%) CWSs with the primary source of groundwater, which served almost 21.5 million people (29.3%). Due to this report's focus on assessing drinking water quality sourced from the Great Lakes, the CWSs with the primary source of groundwater were excluded from our analyses.

As shown in Table 4, of the 2,371 CWSs across the eight Great Lakes States with the primary source of surface water, only 810 were CWSs with the primary source of surface water from the Great Lakes or its connecting rivers (Superior-to-Huron ecosystem: St. Marys River; Huron-to-Erie ecosystem: St. Clair River, Lake St. Clair and Detroit River; Erie-to-Ontario ecosystem: Niagara River; and the international section of the St. Lawrence River),[9] which were only 7.0 percent of all active CWSs across the eight Great Lakes States. Those 810 CWSs served a population of 19,554,093, which was 37.7 percent of the population served by all active CWSs across the Great Lakes States.

To assess Great Lakes drinking water quality, the 810 CWSs with the primary source of surface water from the Great Lakes and its connecting rivers were included in our analyses for this sub-indicator report. These CWSs were

counting of population served across the three categories. See EPA draft guidance manual for clarification of definitions and methods (WSG 61A).

[8] The number of water systems and population served includes Tribal water systems. EPA Region 2 and EPA Region 5 conduct direct implementation of NPDWRs for the Tribal systems in the Great Lakes region. See Tables 2, 3 and 4 for a breakdown of water systems by primacy agency.

[9] The international section of the St. Lawrence River is the section of the river approximately between Cape Vincent, New York and St. Regis, New York. See the Great Lakes Water Quality Agreement Article 1 for the formal definition.

identified in response to a request to the primacy agencies and aided using the Drinking Water Mapping Application to Protect Source Waters (DWMAPS) application.[10]

For results, during the 2020 calendar year there were 810 CWSs that sourced surface water from the Great Lakes or its connecting rivers, which served a population of 19,553,763 (Table 4). Lake Michigan was the top surface water source by both the number of systems with 356 (44.0%) CWSs and by population served with 9,552,028 (48.9%) people served. Figure 3 is a map of the CWSs that source surface waters from the Great Lakes by county. Figure 4 is a map of the population served by the CWSs that source surface waters from the Great Lakes by county.

Table 5 shows the number of HBVs by surface water source. During the 2020 calendar year, there were 16 HBVs at 12 of the 810 CWSs. Therefore, 798 (98.5%) of the CWSs met all health-based standards, achieving a 'Good' rating (>90%) overall. By population served, there were 181,856 (0.9%) served by a CWS with one or more HBVs. Therefore, 99.1 percent of the population was served by CWSs that met all health-based standards, achieving a 'Good' rating overall.

By surface water source, all but one of the surface water sources achieved 'Good' ratings for both measures: by proportion of the CWSs that met all health-based standards and by proportion of population served by a CWS that met all health-based standards.[11] In fact, all the CWSs with the primary source of surface water from Lake Huron (132 CWSs that served 1,664,253 people), Lake Ontario (51 CWSs that served 1,053,451 people), the Niagara River (30 CWSs that served 341,678 people), and the international section of the St. Lawrence River (18 CWSs that served 48,660 people) met all the health-based standards during the 2020 calendar year.

There were a few CWSs that rely on Lake Michigan and Lake Erie for source water that were issued HBVs but both lakes received "Good" scores on the two measures because over 90 percent of the population was served by CWSs that met all health-based standards and over 90 percent of the CWSs met all health-based standards.

Lake Superior received both a 'Fair' and a 'Good' rating (Table 5). The 'Fair' rating resulted from the three HBVs at two of the 17 CWSs (88.2% met all health-based standards) that source from Lake Superior. Despite these two CWSs with HBVs, 94.3 percent of the population was served by CWSs that met all health-based standards, resulting in a 'Good' rating overall. Notably, Lake Superior has the fewest CWSs (n=17) among the five Great Lakes making this proportion-based rating especially sensitive to even one CWS with a HBV (1 of 17 is a 5.9%-change in score). The other four Great Lakes each have 99 or more CWSs (1 of 99 is a 1.0%-change in score).

Figure 5 shows the total population served compared to population served by CWSs with HBVs by surface water source.

Figure 6 shows the type of HBVs by rule and surface water source. There were seven Lead and Copper Rule treatment technique violations (failure to submit an optimal corrosion control study, failure to maintain optimal water quality parameters, failure to complete all public education requirements, and lead action level exceedance), six Disinfectants and Disinfection Byproducts Rules violations (systems whose locational running annual average exceeded the MCL for total trihalomethanes or haloacetic acids and a system that was not operated by a state-approved operator), two surface water treatment rule treatment technique violations (a system that did not meet the minimum residual disinfectant concentration and one post-filtration turbidity sample that exceeded 1 NTU), and one revised total coliform rule violation (one positive *E. coli* sample result in the distribution system).

[10] To request the list of water systems assessed for this report, please contact the authors. See Acknowledgements section for the list of authors and contact information.

[11] The St. Marys River was not rated due to an insufficient CWS sample size (n=1).

Linkages

Following is a brief discussion of other Great Lakes sub-indicators that can influence drinking water quality. In general, the quality of treated drinking water can be linked with other sub-indicators and may be negatively impacted by the demands of an increasing Great Lakes human population.

- Groundwater Quality – important because many municipalities obtain their drinking water from groundwater.
- Water Quality in Tributaries – important because some municipalities use tributaries as their drinking water source and because tributaries are the main route by which contaminants reach the Great Lakes.
- Precipitation Amounts in the Great Lakes Basin, Watershed Stressors, Forest Cover, Land Cover and Tributary Flashiness are linked to the Treated Drinking Water sub-indicator as they can influence the potential for contaminants to wash into tributaries and to reach drinking water intakes within the Great Lakes basin.
- Harmful Algal Blooms – can cause algal toxin contamination of drinking water sources. By extension the related sub-indicators of Nutrients in Lakes and Surface Water Temperature are important to drinking water quality.
- Toxic Chemicals in the Atmosphere and Toxic Chemicals in Water can influence toxics concentrations at drinking water intakes.

This sub-indicator also links directly to the other human health related sub-indicators including Beach Advisories and Contaminants in Edible Fish.

Traditional Ecological Knowledge (TEK), Citizen Science and other Bodies of Knowledge

The lifeways of the Tribes and First Nations of the Great Lakes Basin are inexorably linked to the Great Lakes water resources, who have retained their rights to hunt, fish and gather within the Great Lakes Basin in areas ceded to the United States or Canada in various treaties (GLRC, 2005; HETF, 2005). As an example of this commitment, in 2004 the Tribes and First Nations of the Great Lakes sent an accord to the federal, state and provincial governments of the Great Lakes Region to demand their rights to protect and preserve the Waters of the Great Lakes Basin, to secure a healthy future for the Great Lakes, and to be included to fully participate in the negotiations of the Great Lakes Compact.

In a recent poll of Great Lakes Basin residents, more respondents that identified as Indigenous or Métis felt it was important to protect the Great Lakes water quality than non-Indigenous respondents (GLWQB, 2018). In fact, Indigenous respondents were substantially more likely to contact public officials, attend public meetings, and engage online to protect the Great Lakes water quality than non-Indigenous respondents (Guo et al., 2020).

In a series of interviews of Anishinaabe and non-Indigenous residents across the Great Lakes Basin, both groups of interviewees expressed that water characterizes “the way of life” in the region as well as a greater concern about water quality than water quantity, but Anishinaabe respondents expressed greater water-related values than non-Indigenous residents (Kozich et al., 2018).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada			X	
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: U.S.: Sources are described and linked in the text. Contact authors to request files of the CWSs analyzed. Canada: DWSP data can be found here: https://data.ontario.ca/dataset/drinking-water-surveillance-program		

Data Limitations

The purpose of this sub-indicator is to assess the waters of the Great Lakes as a source of safe, high-quality drinking water. There are numerous data limitations that prevented a harmonized binational assessment of a consistent suite of indicators in source water and treated drinking water.

The measures employed by the U.S. EPA in this report were of CWSs that met all health-based standards for treated drinking water, which are more reflective of compliance with the existing SDWA regulations and primacy agencies' oversight than source water quality. An assessment of a harmonized suite of water quality indicators comparing of source water quality at the intakes versus finished water quality would be ideal, but limited source water sampling requirements and data accessibility are barriers. Access to source water monitoring data would require increased sampling and collaboration with the CWSs or other agencies. For example, the National Oceanic and Atmospheric Administration's Great Lakes Environmental Research Laboratory, the Great Lakes Observing System, and citizen science organizations (e.g., Waterkeeper Alliance organizations, watershed councils) conduct seasonal source water monitoring and modelling, which could be included in future analyses. Finished water quality monitoring data are maintained by the primacy agencies (e.g., states' departments of environmental protection), which would require additional efforts to coordinate access to the compliance sampling data from the 145 CWSs in the U.S. with intakes in the surface waters of the Great Lakes and connecting rivers.[12] Additional monitoring may be needed, such as through the EPA's National Aquatic Resource Surveys.

Additional Information

First incorporated in the 1987 amendments of the GLWQA was a procedure for data assessment and identification of Areas of Concern (AOC) and development of corresponding remedial action plans (RAPs) based on a suite of 14 beneficial use impairments (BUI).[13] Relevant to drinking water is BUI#9: "restrictions on drinking water consumption, or taste and odor problems." Of the 43 locations formally identified as AOCs by Canada and the United States, eight locations were deemed to have a BUI affecting drinking water (BUI#9). Since 1987, six of these eight locations have completed remedial actions to delist the drinking water BUI. The two locations with the drinking water BUI still assigned are Lower Green Bay/Fox River AOC in Wisconsin[14] and the binational St. Clair River AOC.[15] There are no CWSs that use the Fox River or Lower Green Bay as its source water. In Michigan, there are 12 CWSs serving nearly 100,000 residents that rely on the St. Clair River as their source water and all 12 CWSs met all health-based standards in 2020.

United States

We improved the methods of our assessments from previous SOGL reports. Notably, the set of U.S. public water systems analyzed in this sub-indicator report is more specific than previous SOGL reports. Previous SOGL reports assessed all active CWSs (including those with the primary source of groundwater) in the counties that border the Great Lakes shores in the U.S. As shown in Figure 2 and Figure 3, not all Great Lakes bordering counties have CWSs that source surface water from the Great Lakes, and there are 23 non-bordering (i.e., inland) counties (e.g., DuPage and Will counties, Illinois; Waukesha county, Wisconsin; Genesee, Kent, Lapeer, Midland, Oakland, Saginaw, and Washtenaw counties, Michigan; Ashland, Fulton, Geauga, Huron, Medina, Portage, and Wood counties, Ohio; and Cattaraugus, Genesee, Livingston, Onondaga, Ontario, and Wyoming counties, New York) that have 166 CWSs (20.5% of CWSs assessed) that sourced surface water from the Great Lakes, which served a population of 3,433,077 (17.6% of population assessed). Our current assessment also included an analysis by the surface water source (i.e., by Great Lake and connecting river) individually whereas previous reports provided assessments by the state where the CWSs were located and not by surface water source.

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[12] As assessed for this report, there are 810 CWSs that rely on the surface waters of the Great Lakes and connecting rivers as their source of water. Of these 810 CWSs, 145 are surface water systems with intakes in these waterbodies. The other 665 CWSs purchase water (i.e., surface water purchase systems; consecutive systems) from the 145 surface water systems. A list of these systems is available upon request.

[13] For more information about the Areas of Concern program, see https://binational.net/annexes/a1/, https://www.ijc.org/en/what/glwq-aoc, https://www.epa.gov/great-lakes-aocs, and https://www.canada.ca/en/environment-climate-change/services/great-lakes-protection/areas-concern.html.

[14] For more information, see https://www.epa.gov/great-lakes-aocs/lower-green-bayfox-river-aoc.

[15] For more information, see https://www.epa.gov/great-lakes-aocs/st-clair-river-aoc and https://www.canada.ca/en/environment-climate-change/services/great-lakes-protection/areas-concern/st-clair-river.html.

Additionally, the population served drinking water sourced from the Great Lakes and its connecting rivers reported in this sub-indicator report is different than the resident population of the Great Lakes Basin. This is due to many factors, including millions of residents outside the Great Lakes Basin who receive drinking water sourced from the Great Lakes (particularly in the Chicago and Milwaukee metropolitan areas where the basin boundary is near the surface waters of Lake Michigan and numerous groundwater sources have elevated levels of contaminants requiring expensive treatment to achieve compliance)[16] and millions of Great Lakes Basin residents who are provided drinking water from other sources that were excluded from our analyses (e.g., groundwater, inland surface water bodies, systems that serve less than 25 people).[17] For information about trends of human population within the Great Lakes Basin, see the human population sub-indicator section of this report.

Acknowledgments

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[16] As an example, the City of Waukesha, Wisconsin, currently has a deep aquifer groundwater supply that's contaminated by high radium concentrations which requires expensive treatment to comply with SDWA National Primary Drinking Water Regulations. The city is under an amended court order to comply with radium standards by September 1, 2023. Because of the groundwater contamination and expensive treatment requirements, Waukesha applied to the Wisconsin Department of Natural Resources (WDNR) for a diversion of Lake Michigan water under the Great Lakes Compact and Agreement as a "community in a straddling county" in a revised application in 2013. The WDNR forwarded their review of the application to the Compact Council in 2016. The Compact Council approved Waukesha's diversion application in June 2016 with conditions. The WDNR approved the city's diversion and water supply service area plan in June 2021, and with this approval, the city has obtained all necessary federal, state and local permits to implement the diversion. For more information about this approved diversion, see https://dnr.wisconsin.gov/topic/wateruse/waukesha.html.

[17] As shown in Table 3, nearly 21.5 million U.S. residents across the Great Lakes Region are served by CWSs with the primary source of groundwater. In Michigan, nearly 4.5 million (45%) residents are served treated drinking water from groundwater sources, and Michigan has 1.25 million private household wells (the most of any state) serving nearly 2.6 million residents (MDEQ, 2018). Based on this report's focus on assessing drinking water quality from surface waters of the Great Lakes, groundwater sources were excluded from our analyses.

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Table 1. CCME WQI Source Water Results by Great Lake, 2007-2020

Source: Pat McInnis, Ontario Ministry of the Environment, Conservation and Parks, Toronto, Ontario, 2021.

	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Lake Ontario	94.7	94.7	94.2	100	94.7	---	---	---	---	---	---	94.7	94.1	94.2

	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Lake Erie	100	100	100	100	100	---	---	---	---	---	100	100	100	---

	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018-2019	2020
Lake Huron	100	94.2	100	100	100	---	---	---	---	---	---	100	---

	2007-2009	2010	2011	2012	2013	2014	2015	2016	2017	2018-2019	2020
Lake Superior	100	100	100	---	---	---	---	---	---	100	---

"---" = insufficient data to determine the CCME WQI value

Table 2. U.S. Public Water Systems in the Great Lakes Region

Notes: Data are from CY2020 (2021Q1 SDWIS/Fed data freeze) and include all active PWSs (both surface water and groundwater sources).

Rows display primacy agencies, which exercise primary enforcement responsibility of the National Primary Drinking Water Regulations (NPDWRs). The eight Great Lakes States each have primacy to implement the NPDWRs. EPA Region 2 conducts direct implementation for the NPDWRs for Tribal water systems in New York. EPA Region 5 conducts direct implementation of the NPDWRs for Tribal water systems in Indiana, Michigan, Minnesota, and Wisconsin. There are not any federally recognized Tribes in Illinois, Ohio or Pennsylvania (86 FR 7554).

U.S. EPA has established three broad categories of public water systems (PWSs). A community water system (CWS) serves the same population year-round. A non-transient non-community water system (NTNCWS) regularly supplies water to at least 25 of the same people at least six months per year but not year-round (e.g., schools, factories, office buildings, and hospitals that have their own wells). Transient non-community water systems (TNCWS) provide water in places where people do not remain for long periods of time (e.g., gas stations, restaurants, and campgrounds). Because individuals typically obtain drinking water from multiple PWSs (e.g., home, work, school, community building, restaurant) throughout the day, there may be double counting of population served across the three categories. See the EPA draft guidance manual for clarification of definitions and methods (WSG 61A).

Sources: Safe Drinking Water Information System (SDWIS) Federal Reports Advanced Search, Report Filters: Water System Summary by PWS Type, Submission Year is 2021 and Quarter is 1 and EPA Region in (05), EPA Region 2 and New York (within EPA Region 2) and Pennsylvania (within EPA Region 3), and Activity Status is Active.

	Community water system		Non-Transient non-community system		Transient non-community system		Overall	
Primacy Agency	# Systems	Population Served	# Systems	Population Served	# Systems	Population Served	# Systems	Population Served
EPA Region 2	7	12,190	1	175	1	200	9	12,565
EPA Region 5	74	121,782	22	20,026	14	2,171	110	143,979
Illinois	1,759	12,029,152	422	164,275	3,097	298,756	5,278	12,492,183
Indiana	774	4,984,430	580	210,224	2,648	352,104	4,002	5,546,758
Michigan	1,377	7,377,605	1,277	306,032	7,569	1,080,609	10,223	8,764,246
Minnesota	964	4,475,035	475	70,272	5,209	507,387	6,648	5,052,694
New York	2,292	18,243,235	725	228,813	5,271	796,278	8,288	19,268,326
Ohio	1,161	10,485,619	610	264,666	2,591	374,727	4,362	11,125,012
Pennsylvania	1,911	11,423,803	1,176	526,450	4,981	694,829	8,068	12,645,082
Wisconsin	1,039	4,087,264	901	197,308	8,970	688,852	10,910	4,973,424
Column Total	11,358	73,240,115	6,189	1,988,241	40,351	4,795,913	**57,898**	**80,024,269**
% Overall Total	19.6%	91.5%	10.7%	2.5%	69.7%	6.0%	**100%**	**100%**

Table 3. U.S. Great Lakes Region Community Water Systems by Primary Source

Notes: Data are from CY2020 (2021Q1 SDWIS/Fed data freeze) and only include CWSs (non-community water systems were excluded).

'% CWSs' and '% Population Served' are proportions of their respective overall totals.

Rows display primacy agencies, which exercise primary enforcement responsibility of the National Primary Drinking Water Regulations (NPDWRs). The eight Great Lakes States have primacy to implement the NPDWRs. EPA Region 2 conducts direct implementation for the NPDWRs for Tribal water systems in New York. EPA Region 5 conducts direct implementation of the NPDWRs for Tribal water systems in Indiana, Michigan, Minnesota, and Wisconsin. There are not any federally recognized Tribes in Illinois, Ohio or Pennsylvania (86 FR 7554).

Acronyms: Community Water Systems (CWSs); Calendar Year (CY)

Sources: Safe Drinking Water Information System (SDWIS) Federal Reports Advanced Search, Report Filters: Water System Summary by Primary Source, Submission Year is 2021 and Quarter is 1 and EPA Region in (05), EPA Region 2 and New York (within EPA Region 2) and Pennsylvania (within EPA Region 3), and Activity Status is Active.

	Surface Water				Ground Water				Overall			
Primacy Agency	**# CWSs**	**% CWSs**	**Population Served**	**% Population Served**	**# CWSs**	**% CWSs**	**Population Served**	**% Population Served**	**# CWSs**	**% CWSs**	**Population Served**	**% Population Served**
EPA Region 2	2	0.02%	10,500	0.01%	5	0.04%	1,690	0.002%	7	0.1%	12,190	0.02%
EPA Region 5	3	0.03%	5,823	0.01%	71	0.6%	115,959	0.2%	74	0.7%	121,782	0.2%
Illinois	611	5.4%	8,827,995	12.1%	1,148	10.1%	3,201,157	4.4%	1,759	15.5%	12,029,152	16.4%
Indiana	103	0.9%	2,382,580	3.3%	671	5.9%	2,601,850	3.6%	774	6.8%	4,984,430	6.8%
Michigan	304	2.7%	5,598,159	7.6%	1,073	9.4%	1,779,446	2.4%	1,377	12.1%	7,377,605	10.1%
Minnesota	42	0.4%	1,434,304	2.0%	922	8.1%	3,040,731	4.2%	964	8.5%	4,475,035	6.1%
New York	568	5.0%	14,073,101	19.2%	1,724	15.2%	4,170,134	5.7%	2,292	20.2%	18,243,235	24.9%
Ohio	265	2.3%	7,680,417	10.5%	896	7.9%	2,805,202	3.8%	1,161	10.2%	10,485,619	14.3%
Pennsylvania	416	3.7%	9,917,478	13.5%	1,495	13.2%	1,506,325	2.1%	1,911	16.8%	11,423,803	15.6%
Wisconsin	57	0.5%	1,820,620	2.5%	982	8.6%	2,266,644	3.1%	1,039	9.1%	4,087,264	5.6%
Column Total	**2,371**	**20.9%**	**51,750,977**	**70.7%**	**8,987**	**79.1%**	**21,489,138**	**29.3%**	**11,358**	**100%**	**73,240,115**	**100%**

Table 4. U.S. Population Served by CWSs that source Surface Water from the Great Lakes and Connecting Rivers

Notes: Data are from CY2020 (2021Q1 SDWIS/Fed data freeze) and only include CWSs with the primary source of surface water from the Great Lakes or its connecting rivers.

Primacy agencies exercise primary enforcement responsibility of the National Primary Drinking Water Regulations (NPDWRs). The eight Great Lakes States have primacy to implement the NPDWRs. EPA Region 2 conducts direct implementation for the NPDWRs for Tribal water systems in New York. EPA Region 5 conducts direct implementation of the NPDWRs for Tribal water systems in Indiana, Michigan, Minnesota, and Wisconsin. There are not any federally recognized Tribes in Illinois, Ohio or Pennsylvania (86 FR 7554).

See Figure 3 and Figure 4 for maps of the CWSs by county and population served by county.

Acronyms: Community Water Systems (CWSs); Calendar Year (CY)

Sources: Safe Drinking Water Information System (SDWIS) Federal Reports Advanced Search for selected systems. Report Filters: Health Based is Yes.

		Surface Water Source									
Primacy Agency	**Statistic**	**Lake Superior**	**Lake Michigan**	**St. Marys River**	**Lake Huron**	**St. Clair-Detroit River**	**Lake Erie**	**Niagara River**	**Lake Ontario**	**St. Lawrence River**	**Row Total**
EPA Region 2	# CWS	-	-	-	-	-	1	-	-	1	2
	Population Served	-	-	-	-	-	5,000	-	-	5,500	10,500
EPA Region 5	# CWS	2	-	-	1	-	-	-	-	-	3
	Population Served	1,170	-	-	4,653	-	-	-	-	-	5,823
Illinois	# CWS	-	231	-	-	-	-	-	-	-	231
	Population Served	-	6,612,558	-	-	-	-	-	-	-	6,612,558
Indiana	# CWS	-	16	-	-	-	-	-	-	-	16
	Population Served	-	485,520	-	-	-	-	-	-	-	485,520
Michigan	# CWS	5	62	1	131	88	5	-	-	-	292
	Population Served	30,746	888,760	14,689	1,659,600	2,690,668	104,326	-	-	-	5,388,789
Minnesota	# CWS	8	-	-	-	-	-	-	-	-	8
	Population Served	103,406	-	-	-	-	-	-	-	-	103,406
New York	# CWS	-	-	-	-	-	30	30	51	17	128
	Population Served	-	-	-	-	-	818,445	341,678	1,053,451	43,160	2,256,734
Ohio	# CWS	-	-	-	-	-	73	-	-	-	73
	Population Served	-	-	-	-	-	2,840,982	-	-	-	2,840,982
Pennsylvania	# CWS	-	-	-	-	-	8	-	-	-	8
	Population Served	-	-	-	-	-	245,575	-	-	-	245,575
Wisconsin	# CWS	2	47	-	-	-	-	-	-	-	49
	Population Served	38,686	1,565,190	-	-	-	-	-	-	-	1,603,876
Column Total	# CWS	17	356	1	132	88	117	30	51	18	**810**
	Population Served	174,008	9,552,028	14,689	1,664,253	2,690,668	4,014,328	341,678	1,053,451	48,660	**19,553,763**

Table 5. U.S. Great Lakes Community Water Systems: Health-Based Violations and Population Served

Notes: Data are from CY2020 (2021Q1 SDWIS/Fed data freeze) and only include CWSs with the primary source of surface water from the Great Lakes or its connecting river ecosystems. "St. Lawrence River" includes only the CWSs that source from the international section of the river (between Cape Vincent, NY and St. Regis, NY).

Rating key: Good is >90% met all health-based standards; Fair is 80-90% met all health-based standards; Poor is <80% met all health-based standards. The St. Marys River was not rated due to an insufficient CWS sample size (n=1).

See Figure 5 U.S. Great Lakes Community Water Systems: Total Population Served vs. Population Served by CWSs with Health-Based Violations

See Figure 6. U.S. Great Lakes Community Water Systems: Health-Based Violations by Rule and Surface Water Source

Acronyms: Community Water Systems (CWSs); Health-Based Violation (HBV); Calendar Year (CY)

Sources: Safe Drinking Water Information System (SDWIS) Federal Reports Advanced Search for selected systems. Report Filters: Health Based is Yes.

Great Lake (including connecting river)	Surface Water Source	HBVs	CWSs with HBV	Total CWSs	% of CWSs that Met all Health-Based Standards	Rating	Population Served by CWSs with HBV	Total Population Served	% of Population Served by CWSs that met all Health-Based Standards	Rating
Lake Superior	**Lake Superior**	3	2	17	88.2%	Fair	9,935	174,008	94.3%	Good
Lake Michigan	**Lake Michigan**	3	2	356	99.4%	Good	88,354	9,552,028	99.1%	Good
Lake Huron	**St. Marys River**	1	1	1	0.0%	-	14,689	14,689	0.0%	-
	Lake Huron	0	0	132	100%	Good	0	1,664,253	100%	Good
	River & Lake combined	1	1	133	99.2%	Good	14,689	1,678,942	99.1%	Good
Lake Erie	**St. Clair - Detroit River**	2	2	88	97.7%	Good	32,820	2,690,688	98.8%	Good
	Lake Erie	7	5	117	95.7%	Good	36,058	4,014,328	99.1%	Good
	River & Lake combined	9	7	205	96.6%	Good	68,878	6,705,016	99.0%	Good
Lake Ontario	**Niagara River**	0	0	30	100%	Good	0	341,678	100%	Good
	Lake Ontario	0	0	51	100%	Good	0	1,053,451	100%	Good
	St. Lawrence River	0	0	18	100%	Good	0	48,660	100%	Good
	Rivers & Lake combined	0	0	99	100%	Good	0	1,443,789	100%	Good
Overall		**16**	**12**	**810**	**98.5%**	**Good**	**181,856**	**19,553,763**	**99.1%**	**Good**

Figure 1. Locations of Ontario Drinking Water Systems that Contributed Source Water Data

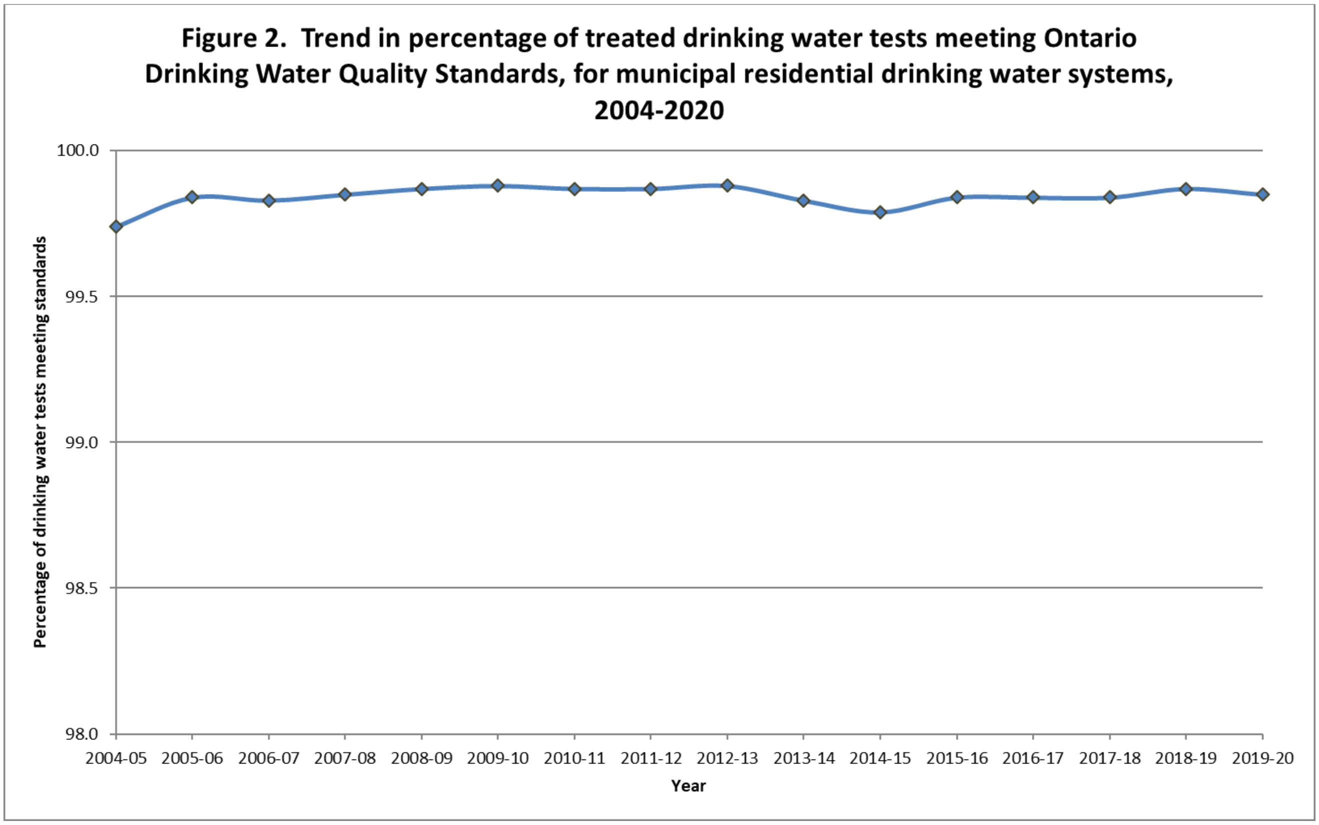


Figure 2. Trend in percentage of treated drinking water tests meeting Ontario Drinking Water Quality Standards, for municipal residential drinking water systems, 2004-2020. Source: Ontario Ministry of the Environment, Conservation and Parks, *Chief Drinking Water Inspector Annual Report*, 2019 – 2020, https://www.ontario.ca/page/2019-2020-chief-drinking-water-inspector-annual-report

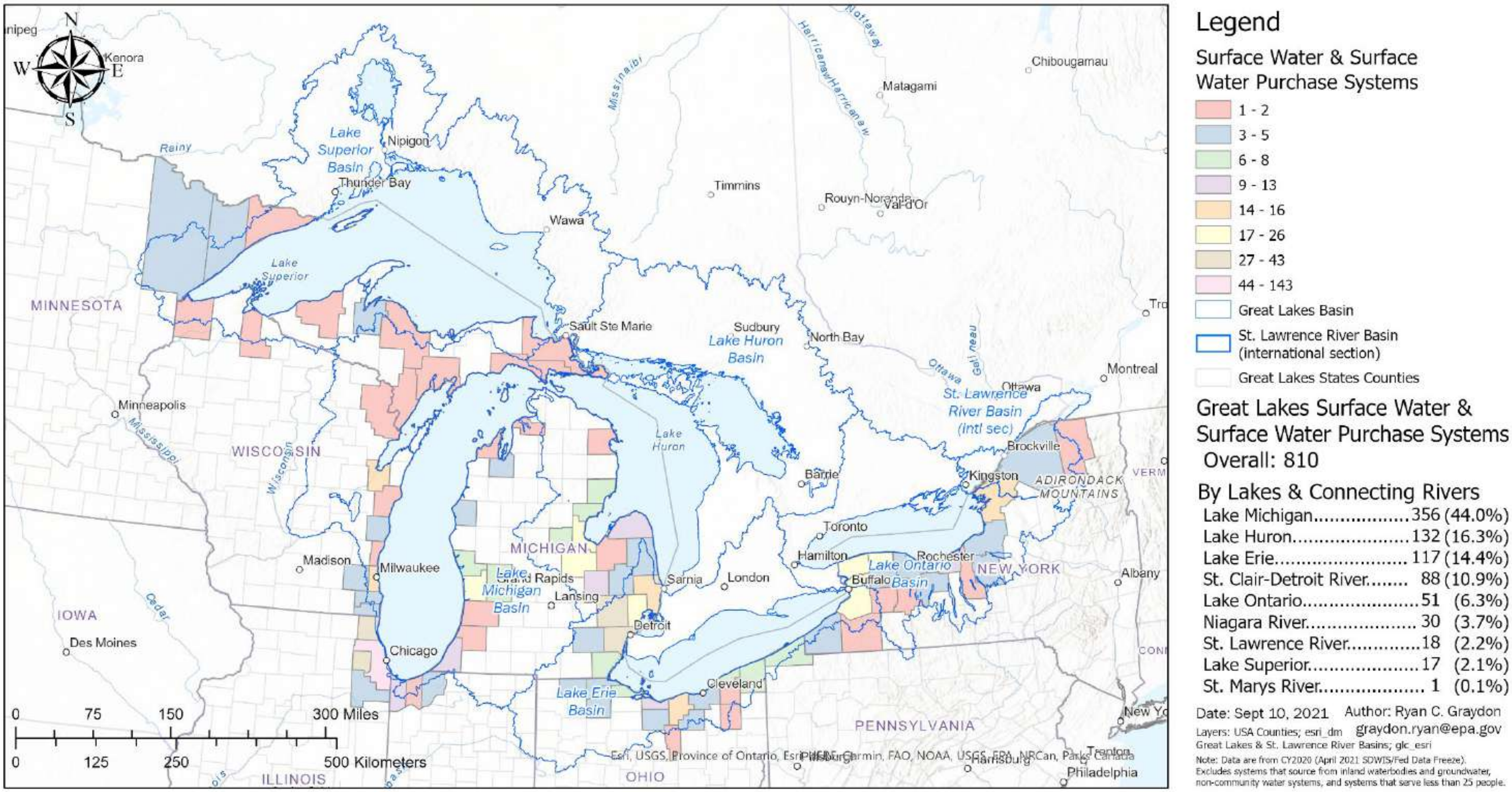


Figure 3. U.S. Community Water Systems that Source from Great Lakes Surface Waters

Note: The symbology displays the number of CWSs that source from the surface waters of the Great Lakes and connecting rivers (surface water and surface water purchase systems) by county.

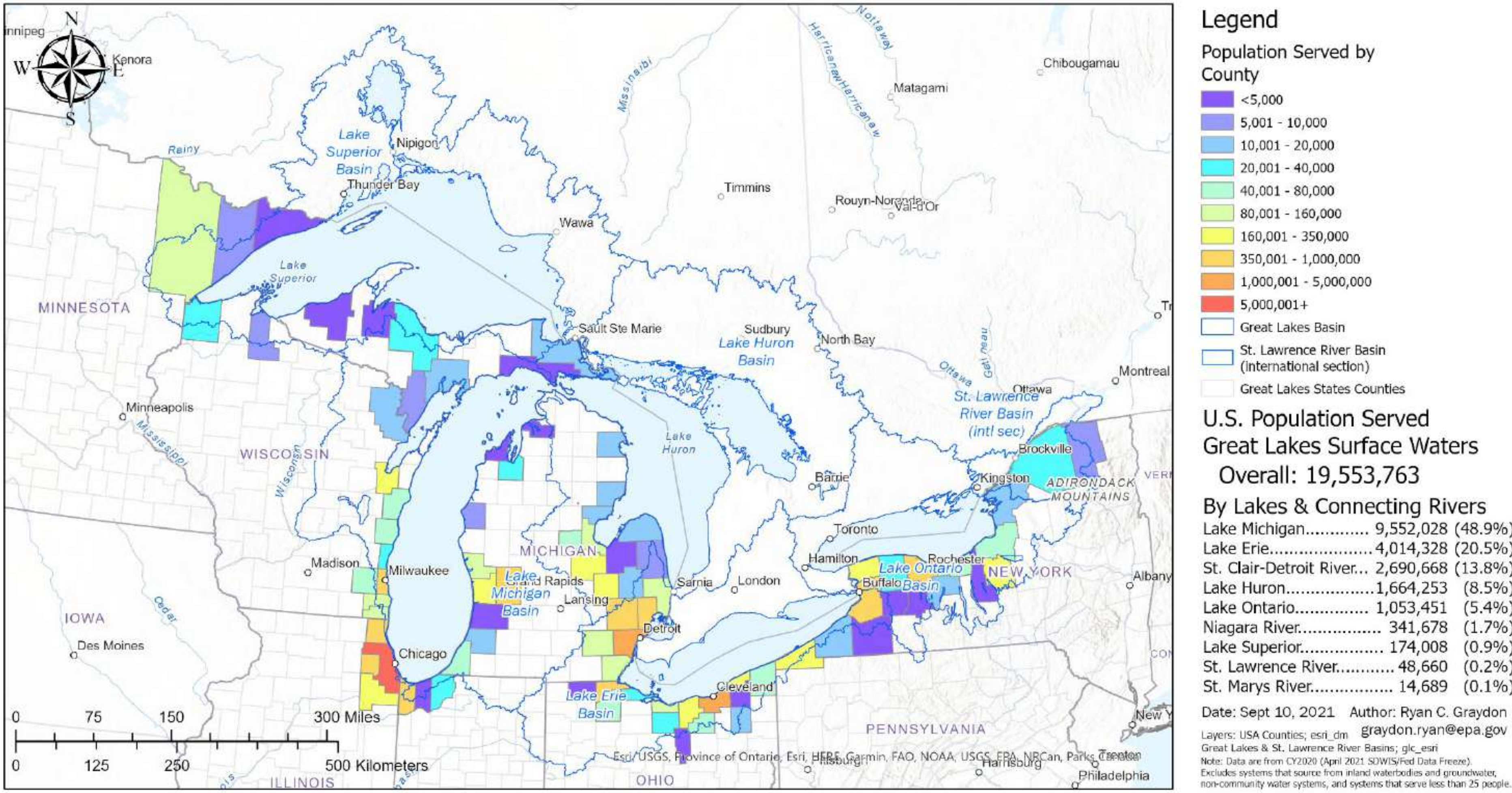
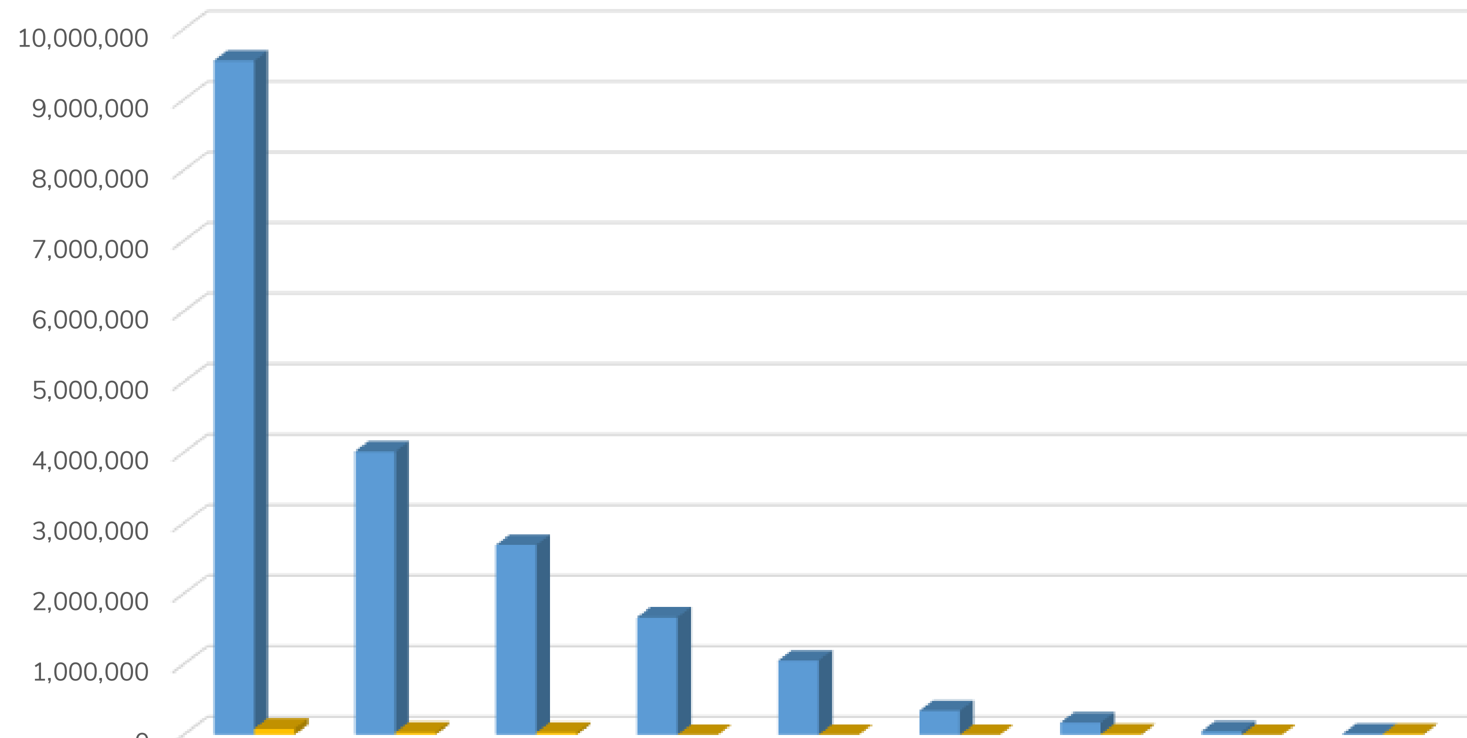


Figure 4. U.S. Population Served by Community Water Systems that Source from Great Lakes Surface Waters

Note: The symbology displays the population served by CWSs that source from the surface waters of the Great Lakes and connecting rivers (surface water and surface water purchase systems) by county.



	Lake Michigan	Lake Erie	St. Clair - Detroit River	Lake Huron	Lake Ontario	Niagara River	Lake Superior	St. Lawrence River	St. Marys River
Total Population Served	9,552,028	4,014,328	2,690,668	1,664,253	1,053,451	341,678	174,008	48,660	14,689
Population Served by CWSs with HBV	88,354	36,058	32,820	0	0	0	9,935	0	14,689

Figure 5. U.S. Great Lakes Community Water Systems: Total Population Served vs. Population Served by CWSs with Health-Based Violations

Note: Data are from CY2020 (2021Q1 SDWIS/Fed data freeze) and only include CWSs with the primary source of surface water from the Great Lakes or its connecting rivers.

Acronyms: Community Water Systems (CWSs); Health-Based Violation (HBV)

Sources: Safe Drinking Water Information System (SDWIS) Federal Reports Advanced Search for selected systems. Report Filters: Health Based is Yes.

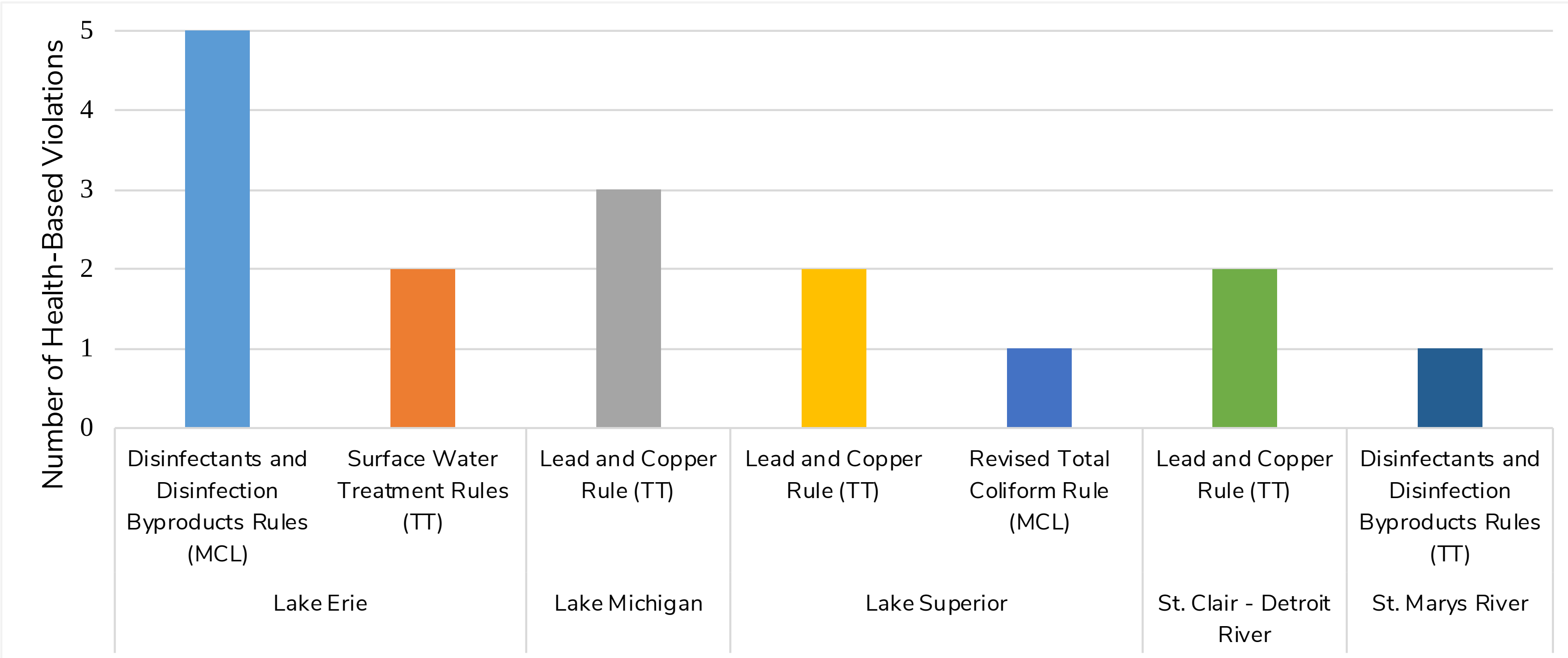


Figure 6. U.S. Great Lakes Community Water Systems: Health-Based Violations by Rule and Surface Water Source

Notes: Data are from CY2020 (2021Q1 SDWIS/Fed data freeze) and only include CWSs with the primary source of surface water from the Great Lakes or its connecting rivers. DBPR violations included systems whose locational running annual average exceeded the MCL for TTHMs or HAA5s, and a system that was not operated by a state-approved operator (TT). LCR violations included failure to submit an optimal corrosion control study, failure to maintain optimal water quality parameters, failure to complete all public education requirements, and a lead action level exceedance. SWTRs violations included a system that did not meet the minimum residual disinfectant concentration and one post-filtration turbidity sample that exceeded 1 NTU. The RTCR violation was a positive *E. coli* sample result in the distribution system.

As shown in Table 5, the community water systems that source surface water from Lake Huron, Niagara River, Lake Ontario, and St. Lawrence River did not have any health-based violations during CY2020.

Acronyms: Calendar Year (CY); Disinfectants and Disinfection Byproducts Rule (DBPR); Haloacetic acids (HAA5s); Lead and Copper Rule (LCR); Maximum Contaminant Level (MCL); Nephelometric Turbidity Unit (NTU); Revised Total Coliform Rule (RTCR); Surface Water Treatment Rules (SWTRs); Total Trihalomethanes (TTHMs); Treatment Technique (TT)

Sources: Safe Drinking Water Information System (SDWIS) Federal Reports Advanced Search for selected systems. Report Filters: Health Based is Yes

Sub-Indicator: Beach Advisories

Overall Assessment

Status: Good

Trends:

10-Year Trend: Unchanging to Improving

Long-term Trend (2007-2019): Unchanging*

Rationale: From 2018 to 2019, monitored Great Lakes beaches were assessed as "Good." Monitored U.S. Great Lakes beaches were open and safe for swimming 94% of the swimming season and monitored Canadian Great Lakes beaches were safe for swimming 90% of the swimming season based on the respective acceptable *Escherichia coli (E. coli)* concentrations. The 10-year trend is considered "Unchanging to Improving" as determined by linear regression analysis, which indicated no change over time in the U.S. data and a slight improving trend in the Canadian data. The long-term trend is considered "Unchanging" because regression analysis of the U.S. data available from 2007 to 2019 indicated no change over time.

*Note: long-term trends are based only on U.S. data for the overall and lake-by-lake assessments.

In order to determine a 10-year trend for 2010-2019 Canadian data using a consistent benchmark, data from 2010-2019 were re-analyzed using the post-2018 Ontario *E. coli* thresholds to determine the percentage of days that beaches were safe to swim.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Unchanging

Long-term Trend (2007-2019): Unchanging*

Rationale: From 2018 to 2019, monitored Lake Superior beaches were assessed as "Good." U.S. Lake Superior beaches were open and safe for swimming for an average of 95% of the swimming season and monitored Canadian Lake Superior beaches were safe for swimming an average of 99% of the swimming season based on the respective acceptable *Escherichia coli (E. coli)* concentrations. The 10-year trend is considered "Unchanging" as determined by linear regression analysis of both U.S. and Canadian data. The long-term trend is considered "Unchanging" because regression analysis of the U.S. data available from 2007 to 2019 indicated no change over time.

*Note: long-term trends are based only on U.S. data for the Lake Superior assessment.

Lake Michigan

Status: Good

10-Year Trend: Unchanging

Long-term Trend (2007-2019): Improving

Rationale: All Lake Michigan beaches are in the United States, so there is no binational assessment for this lake. From 2018 to 2019, monitored Lake Michigan beaches were assessed as "Good" because beaches were open and safe for swimming for an average of 96% of the swimming season based on U.S. acceptable *Escherichia coli (E. coli)* concentrations. The 10-year trend is considered "Unchanging" as determined by linear regression analysis. However, the long-term trend is "Improving" because regression analysis of the U.S. data available from 2007 to 2019 indicated a slight improving trend in the percent of beach days open and safe for swimming.

Lake Huron (including St. Marys River)

Status: Good

10-Year Trend: Unchanging to Improving

Long-term Trend (2007-2019): Unchanging*

Rationale: From 2018 to 2019, monitored Lake Huron beaches were assessed as "Good." Monitored U.S. Lake Huron beaches were open and safe for swimming for an average of 97% percent of the swimming season, and monitored Canadian Lake Huron beaches were safe for swimming for an average of 93% of the swimming season based on the respective acceptable *Escherichia coli (E. coli)* concentrations. The 10-year trend is considered "Unchanging to Improving" as determined by linear regression analysis, which indicated no change over time in the U.S. data and a slight improving trend in the Canadian data. The long-term trend is considered "Unchanging" because regression analysis of the U.S. data available from 2007 to 2019 indicated no change over time.

*Note: long-term trends are based only on U.S. data for the Lake Huron assessment.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (2007-2019): Unchanging*

Rationale: From 2018 to 2019, monitored Lake Erie beaches were assessed as "Fair." Monitored U.S. Lake Erie beaches were open and safe for swimming for 84% of the swimming season, and monitored Canadian Lake Erie beaches were safe for swimming an average of 79.9% of the swimming season based on the respective acceptable *Escherichia coli (E. coli)* concentrations. The 10-year trend is considered "Unchanging" as determined by linear regression analysis of both U.S. and Canadian data. The long-term trend is considered "Unchanging" because regression analysis of the U.S. data available from 2007 to 2019 indicated no change over time.

*Note: long-term trends are based only on U.S. data for the Lake Erie assessment.

Lake Ontario (including Niagara River)

Status: Good

10-Year Trend: Improving

Long-term Trend (2007-2019): Improving*

Rationale: From 2018 to 2019, monitored Lake Ontario beaches were assessed as "Good." Monitored U.S. Lake Ontario beaches were open and safe for swimming for 95% of the swimming season, and monitored Canadian Lake Ontario beaches were safe for swimming an average of 91% of the swimming season based on the respective acceptable *Escherichia coli (E. coli)* concentrations. The 10-year trend is considered "Improving" as determined by linear regression analysis of both U.S. and Canadian data. The long-term trend is considered "Improving" because regression analysis of the U.S. data available from 2007 to 2019 indicated a slightly increasing trend in the percentage of beach days open and safe for swimming over time.

*Note: long-term trends are based only on U.S. data for the Lake Ontario assessment.

Status Assessment Definitions

Good: Greater than 90% of available beach days in the U.S. or greater than 80% of beach days in Canada meet bacterial standards and beaches remain open and safe for swimming.

Fair: 80% to 89.99% of available beach days in the U.S. or 70% to 79.99% of beach days in Canada meet bacterial standards and beaches remain open and safe for swimming.

Poor: Less than 79.9% of available beach days in the U.S or less than 69.99% of beach days in Canada meet bacterial standards and beaches remain open and safe for swimming.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: The percentage of beach days that meet bacterial standards shows a change towards more acceptable conditions based on linear regression analysis.

Unchanging: The percentage of beach days that meet bacterial standards shows no change based on linear regression analysis.

Deteriorating: The percentage of beach days that meet bacterial standards shows a change away from acceptable conditions based on linear regression analysis.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoint and/or Targets

Target: All monitored Great Lakes beaches should achieve a status of Good and a trend of Improving or Unchanging. US Great Lakes beaches should be open and safe for swimming for 90% or more of the swimming season. Canadian Great Lakes beaches should be open and safe for swimming for 80% or more of the swimming season.

Endpoint: All monitored beaches in the Great Lakes should be open and safe for swimming for 100% of the swimming season.

Sub-Indicator Purpose

- To assess the percentage of days that Great Lakes beaches are open and safe for swimming by assessing health-related swimming advisories and closings at recreational beaches.

- To infer potential harm from pathogens to human health through body contact with nearshore recreational waters.

Ecosystem Objective

Waters should be safe for recreational use. Waters used for recreational activities involving bodily contact should be free from pathogens, such as bacteria, parasites, and viruses that may harm human health.

This sub-indicator supports work towards General Objective #2 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "allow for swimming and other recreational use, unrestricted by environmental quality concerns."

Measure

This sub-indicator measures the percentage of days of the beach season that Great Lakes beaches monitored by beach safety programs have acceptable *Escherichia coli* (*E. coli*) concentrations, and are therefore open and safe for swimming.

The swimming season is generally from the Memorial Day/Victoria Day weekend to Labor Day weekend; however, some health units/counties vary so all beach days that are reported by counties and Public Health Units (PHU) will be used in this report. The number of days that each beach was open and safe for swimming was calculated based on this standard to be consistent with the past State of the Great Lakes reports. The trends in the percentage of beach days open for both Canadian and U.S. beaches were determined using linear modeling in R, using an F-test to determine any trends in the percentage of beach days open over time. A p-value ≤ 0.05 was considered significant (R Core Team, 2020).

In the US, the frequency of beach monitoring varies from state to state - from once a week to daily sampling - depending on available funding and staffing making it difficult to compare beach water quality among states. Non-monitored beaches are not included in this sub-indicator. Non-monitored beaches are entered into US databases as open and safe for swimming for 100% of the beach season because the lack of monitoring resulted in no postings. The assumption that non-monitored beaches were always safe for swimming would result in an overstatement of the safety of Great Lakes beaches, and therefore non-monitored beaches are removed from the analysis.

The Canadian data being assessed in this report are exclusively from beaches monitored by PHUs located on the waters of the Great Lakes and connecting channels (excluding the St. Lawrence River). Beaches not located on the waters of the Great Lakes are not included in this assessment, such as beaches on in-land lakes. The frequency of beach monitoring conducted by PHUs ranges from daily to once per month. This report assesses 10 years of data (2010-2019) using the *E. coli* thresholds as outlined in the Ontario Guidelines for Recreational Water Quality, which were implemented in 2018 (Ontario Ministry of Health and Long-term Care, 2018). In order to determine a 10-year trend for 2010-2019 using a consistent benchmark, data from 2010-2019 were re-analyzed using the 2018 *E. coli* thresholds to determine the percentage of days that beaches were safe to swim. Therefore, it is important to note that the data being reported here are not necessarily based on the actual number of days that beaches were closed by PHUs, but rather the number of days that beaches would have been closed, if the 2018 Ontario thresholds were used. Please see The Ecological Condition section below for more information.

Ecological Condition

Background

Recreational activities such as swimming, boating, and beach activities can involve direct contact with water. In some situations, water can be contaminated with pathogens (parasites, bacteria, viruses), which can lead to gastrointestinal illness (including diarrhea, nausea, and vomiting), as well as skin, ear, eye, respiratory and wound infections. *E. coli* in fresh water, and Enterococci in salt water, are the two most commonly monitored bacteria used for indicating water safety for human health. Great Lakes beach monitoring is conducted to primarily detect *E. coli*, which is a type of bacteria which normally lives in intestines in humans and animals. *E. coli* can indicate the presence of harmful pathogens from fecal contamination. Recreational waters may become contaminated with *E. coli*, and other organisms from animal and human feces due to failing septic systems, combined sewer overflows, storm water runoff, waterfowl, boat wastes, and other pollution sources. When monitoring results reveal elevated levels of *E. coli*, the state or local government/health units issue a beach advisory or closure notice until further sampling shows that the water quality is meeting the applicable water quality standards.

In the U.S., the U.S. Environmental Protection Agency (U.S. EPA) suggests the use of a Beach Action Value (BAV) to make beach advisory or closure decisions. Any single sample above the BAV could trigger a beach notification until another sample below the BAV is collected. U.S. EPA's recommended BAVs are outlined in U.S. EPA's Recreational Water Quality Criteria (RWQC) which were revised in December 2012, in accordance with the Beaches Environmental Assessment and Coastal Health (BEACH) Act. The revised criteria reflect the latest scientific knowledge and are designed to protect the public from exposure to harmful levels of pathogens while participating in water-contact activities.

U.S. EPA's revised RWQC correspond to two different illness rates that states must select and apply at their inland and coastal recreation waters. U.S. EPA suggests that a state's chosen criterion illness rate be used to determine the corresponding BAV. Based on an estimated illness rate of 36 per 1,000 primary contact recreators, EPA recommends a BAV of 235 *E. coli* cfu per 100 mL or 70 Enterococci cfu per 100 mL. Based on an estimated illness rate of 32 per 1,000 primary contact recreators, EPA recommends a BAV of 190 *E. coli* cfu per 100 mL or 60 Enterococci cfu per 100 mL (U.S. EPA Recreational Water Quality Criteria 2012). The State of Michigan uses 130 *E. coli* cfu per 100 mL as a 30-day geometric mean, and a maximum of 300 *E. coli* cfu per 100 mL based on the geometric mean of three or more samples taken during the same sampling event at representative locations within a defined sampling area, to make beach notification decisions.

U.S. EPA is authorized by the BEACH Act to award grants to coastal and Great Lakes states, territories and eligible tribes to help local authorities monitor their coastal and Great Lakes beaches and notify the public of water quality conditions that may be unsafe for swimming. Great Lakes beach managers are able to regularly monitor beach water quality and advise bathers of potential risks to human health when water quality standards for bacteria are exceeded. When levels of fecal indicator bacteria exceed a state's BAV, swimming at beaches is prohibited or advisories are issued to inform beachgoers that swimming may be unsafe. The swimming season typically starts Memorial Day weekend and ends on Labor Day. The U.S. EPA provides publicly-accessible data about beach closings and advisories for U.S. coastal beaches at its Beach Advisory and Closing On-line Notification (BEACON) system at:. http://www2.epa.gov/waterdata/beacon-20-beach-advisory-and-closing-online-notification (U.S. EPA BEACON).

In Ontario, recreational public beaches are routinely monitored for E. coli concentrations by the responsible PHU. The PHUs observe the E. coli beach monitoring results to determine whether it is safe to swim in the waters of the beach. Up until 2017, the Ontario thresholds for acceptable E. coli concentrations was a geometric mean of 100 E.

coli coliform forming units (cfu) per 100 mL (Ontario Ministry of Health and Long-term Care, 2014). This threshold was used in previous SOGL reports for Canadian data, however, this report and future SOGL reports will assess this sub-indicator using the E. coli concentration thresholds as outlined in the Ontario Operational Approaches for Recreational Water Quality, which were implemented in 2018 (Ontario Ministry of Long-term Health and Care, 2018). These new thresholds are consistent with those from Health Canada's Guidelines for Canadian Recreational Water Quality (2012). In accordance with these guidelines, in order for beaches to be considered safe to swim, E. coli concentrations in water should not exceed the following:

- 200 *E. coli* cfu per 100 mL (geometric mean concentration, minimum of 5 samples); and,
- 400 *E. coli* per 100 mL, maximum limit for a single water quality sample.

Following these guidelines, recreational beach waters are posted as unsafe for swimming when *E. coli* levels exceed this threshold, and the waters remain posted until further sampling indicates that *E. coli* levels have dropped below the threshold. It is important to note however that not all PHUs follow these exact thresholds. It is up to the discretion of the PHU to make the final decision as to whether or not the beach shall be closed based on the *E. coli* monitoring results. For example, some PHUs may not close a beach is there is a single sample exceedance over the 400 *E. coli* per 100 mL, and will instead only close the beach if the geometric mean exceeded the 200 *E. coli* cfu per 100 mL limit. However, in order to ensure consistency in this report, Ontario beach data from 2010-2019 were re-analyzed against the 2018 Ontario *E. coli* thresholds. This was completed by analyzing the raw *E. coli* beach data from PHU monitoring events, and re-calculating the number of days during the swimming season that beaches would have been safe or unsafe for swimming, had the *E. coli* thresholds identified in the 2018 Ontario Operational Approaches been used. In previous SOGL Beach Advisories reports, the assessments for Canadian beaches were based on data as far back as 1999, and therefore trends longer than 10 years were assessed. For this report however, only data as far back as 2010 were reanalyzed against the same metric, so determining a trend based on data farther back then that year would be inconsistent and inaccurate.

The PHU monitoring frequency during the 2018-2019 swimming seasons also varied between beaches. Beaches in the lower Great Lakes were typically monitored more frequently compared to beaches in the upper Great Lakes. In Lake Superior, 0% of beaches were monitored daily, 42% were monitored weekly or more frequently (but not daily), and 58% were monitored monthly. In Lake Huron, 0% of beaches were monitored daily, 77% were monitored weekly or more frequently (but not daily), 2% were monitored once every 2-3 weeks, and 21% were monitored monthly. In Lake Erie, 3% of beaches were monitored daily, 77% were monitored weekly or more frequently (but not daily), and 20% were monitored monthly. In Lake Ontario, 20% of beaches were monitored daily, 76% were monitored weekly or more frequently (but not daily), and 4% were monitored between once every 2-3 weeks to monthly.

Status of Great Lakes Beach Advisories

US Great Lakes Beaches-Overall Assessment

Since 2007, U.S. Great Lakes beaches have been assessed as "Good," and both the long-term and 10-year trends in the percentage of monitored U.S. Great Lakes beaches open and safe for swimming is described as "Unchanging" (Figure 1). Overall, the U.S. Great Lakes monitored beaches were open and safe for 94% of the swimming season from 2018 to 2019. During this period, 883 (81%) of monitored U.S. Great Lakes beaches were safe for swimming greater than 90% of the season, 115 (11%) of beaches were safe for swimming greater than 80% but less than 90% of the season, and 89 (8%) of beaches were safe for swimming less than 80% of the beach season based on *E. coli* standards (Figure 2).

US Lake Superior Beaches

U.S. Lake Superior beaches are assessed as "Good," with monitored beaches considered safe for swimming 95% of the swimming seasons for 2018 and 2019. There has been little change in the status of monitored U.S. Lake Superior beaches since 2007, and both the 10-year and long-term trends are considered "Unchanging" (Figure 1). During the 2018 and 2019 swim seasons, 194 monitored U.S. Lake Superior beaches (86%) were safe for swimming greater than 90% of the season, 23 monitored U.S. Lake Superior beaches (10%) were safe for swimming greater than 80% but less than 90% of the season, and 9 monitored U.S. Lake Superior beaches (4%) were safe for less than 80% of the swimming season in Lake Superior based on *E. coli* standards (Figure 2).

US Lake Michigan Beaches

U.S. Lake Michigan beaches were assessed as "Good," with monitored beaches considered safe for swimming 96% of the seasons for 2018 and 2019. Since 2007, there is a slight increasing trend in the percentage of days beaches were open and safe for swimming during the swimming season. However, from 2010 through 2019, there was little change over time, so the 10-year trend is considered "Unchanging" while the long-term trend is "Improving" with a p-value of 0.03 (Figure 1). During the 2018 and 2019 swim seasons, 475 monitored U.S. Lake Michigan beaches (89%) were safe for swimming greater than 90% of the season, 40 monitored U.S. Lake Michigan beaches (7%) were safe for swimming greater than 80% but less than 90% of the season, and 20 monitored U.S. Lake Michigan beaches (4%) were safe for less than 80% of the swimming season in Lake Michigan based on *E. coli* standards (Figure 2).

US Lake Huron Beaches

U.S. Lake Huron beaches were assessed as "Good," with monitored beaches considered safe for swimming 97% of the seasons for 2018 and 2019. Since 2007, there has been relatively little change in the status of U.S. Lake Huron beaches. Based on regression analysis, both the long-term and 10-year trends are considered "Unchanging" (Figure 1). During the 2018 and 2019 swim seasons, 98 monitored U.S. Lake Huron beaches (92%) were safe for swimming greater than 90% of the season, 5 monitored U.S. Lake Huron beaches (5%) were safe for swimming greater than 80% but less than 90% of the season, and 4 monitored U.S. Lake Huron beaches (4%) were safe for less than 80% of the swimming season in Lake Huron based on *E. coli* standards (Figure 2).

US Lake Erie Beaches

U.S. Lake Erie beaches were assessed as "Fair," with monitored beaches considered safe for swimming 84% of the seasons for 2018 and 2019. Since 2007, there has been relatively little change in the status of U.S. Lake Erie beaches. Based on regression analysis, both the long-term and 10-year trends are considered "Unchanging" (Figure 1). During the 2018 and 2019 swim seasons, 71 monitored U.S. Lake Erie beaches (43%) were safe for swimming greater than 90% of the season, 39 monitored U.S. Lake Erie beaches (24%) were safe for swimming greater than 80% but less than 90% of the season, and 54 monitored U.S. Lake Erie beaches (33%) were safe for less than 80% of the swimming season in Lake Erie based on *E. coli* standards (Figure 2).

US Lake Ontario Beaches

U.S. Lake Ontario beaches were assessed as "Good," with monitored beaches considered safe for swimming 95% of the seasons for 2018 and 2019. Since 2007, there is a slight increasing trend in the percentage of days beaches were open and safe for swimming during the swimming season. A similar increase trend is also evident during the 2010 to 2019 period, so both the long-term and 10-year trends are considered "Improving" with p-values of 0.01 and 0.03, respectively (Figure 1). During the 2018 and 2019 swim seasons, 45 monitored U.S. Lake Ontario beaches (82%) were safe for swimming greater than 90% of the season, 8 monitored U.S. Lake Ontario beaches (14%) were safe for swimming greater than 80% but less than 90% of the season, and 2 monitored U.S. Lake

Ontario beaches (4%) were safe for less than 80% of the swimming season in Lake Ontario based on *E. coli* standards (Figure 2).

Canadian Great Lakes Beaches – Overall Assessment

From 2018 to 2019, monitored Canadian beaches on the Great Lakes were assessed as Good because beaches met Ontario bacterial thresholds and were open and safe for swimming an average of 90% of the swimming season (Figure 3). Results of the linear modeling for the 10-year trend indicate a trend of "Improving", with a p-value of 0.03. Monitored Canadian beaches on the Great Lakes were safe for swimming 86% of the swimming season from 2010–2019. The long-term trend is Undetermined because a reassessment of only 2010–2019 data was recently conducted to consider a change in Ontario *E. coli* thresholds in 2018.

During the 2018 and 2019 swimming seasons, 38% of Canadian beaches on the Great Lakes did not exceed the Ontario bacterial standards once and were open and safe for swimming the entire swimming season. Note this calculation is considering each beach within each year as one independent value. For example, the dataset for the 2018 swimming season consists of 171 monitored Canadian beaches on the Great Lakes, of which 55 were open and safe for swimming during the entire 2018 swimming season. The dataset for the 2019 swimming season consists of 168 monitored Canadian beaches on the Great Lakes (most of which were also included in the 2018 dataset), of which 73 were safe for swimming the entire swimming 2019 season. So therefore over both years, 128 out of 339 beaches (37%) were open and safe for swimming during the 2018 and 2019 swimming seasons. Also, over this same two-year period, 82% of beaches achieved a status of Good (open 80% or more of the swimming season), 11% achieved a status of Fair (open 70-79.9% of the swimming season), and 7% were assessed as Poor (open less than 70% of the swimming season) (Figure 4).

During the entire 2010-2019 swimming seasons, 31% of Canadian beaches on the Great Lakes did not exceed the Ontario bacterial thresholds once. Also, during this same 10-year period, 74% of Canadian beaches on the Great Lakes achieved a status of Good, 12% achieved a status of Fair, and 13% were assessed as Poor (Figure 4).

Canadian Lake Superior Beaches

From 2018 to 2019, monitored Canadian Lake Superior beaches were assessed as Good because beaches met Ontario bacterial thresholds and were open and safe for swimming an average of 99% of the swimming seasons (Figure 3). The results of the linear modeling for the 10-year trend indicate that the 10-year Canadian trend is Unchanging. Monitored Canadian Lake Superior beaches were open and safe for swimming 94% of the swimming seasons in 2010–2019.

During the 2018-2019 swimming seasons, 88% of Canadian Lake Superior beaches were open and safe for swimming the entire swimming seasons. Also over this same two-year period, 100% of Canadian Lake Superior beaches achieved a status of Good. There were no beaches that were Fair or Poor (Figure 4).

During the entire 2010-2019 swimming seasons, 66% of Canadian Lake Superior beaches did not exceed the Ontario bacterial thresholds once. Also during this same 10-year period, 92% of Canadian Lake Superior beaches achieved a status of Good, 3% achieved a status of Fair, and 5% were assessed as Poor (Figure 4).

Canadian Lake Huron Beaches (including St. Marys River)

From 2018 to 2019, monitored Canadian Lake Huron beaches were assessed as Good because beaches met Ontario bacterial thresholds and were safe for swimming an average of 93% of the swimming seasons (Figure 3). The results of the linear modeling to determine the 10-year trend indicate an "Improving" trend with a p-value of 0.049. Monitored Canadian Lake Huron beaches were open and safe for swimming 90% of the swimming seasons in 2010–2019.

During the 2018-2019 swimming seasons, 44% of Canadian Lake Huron beaches were open and safe for swimming the entire swimming seasons. Also over this same two-year period, 92% of Canadian Lake Huron Beaches achieved a status of Good, 6% achieved a status of Fair, and 2% were assessed as Poor (Figure 4).

During the entire 2010-2019 swimming seasons, 44% of Canadian Lake Huron beaches did not exceed the Ontario bacterial standards once. Also during this same 10-year period, 81% of Canadian Lake Huron beaches achieved a status of Good, 10% achieved a status of Fair, and 9% were assessed as Poor (Figure 4).

Canadian Lake Erie Beaches (including St. Clair-Detroit River Ecosystem)
From 2018 to 2019, monitored Canadian Lake Erie beaches were assessed as Fair because beaches met Ontario bacterial thresholds and were open and safe for swimming an average of 79.9% of the swimming seasons (Figure 3). The results of the linear modeling to determine a 10-year trend indicate that the 10-year Canadian trend is described as Unchanging. Monitored Canadian Lake Erie beaches were open and safe for swimming 79.6% of the swimming seasons in 2010–2019.

During the 2018-2019 swimming seasons, 17% of Canadian Lake Erie beaches were open and safe for swimming the entire swimming seasons. Also over this same two-year period, 52% of Canadian Lake Erie beaches achieved a status of Good, 26% achieved a status of Fair, and 22% were assessed as Poor (Figure 4).

During the entire 2010-2019 swimming seasons, 14% of Canadian Lake Erie beaches did not exceed the Ontario bacterial thresholds once. Also during this same 10-year period, 60% of Canadian Lake Erie beaches achieved a status of Good, 18% achieved a status of Fair, and 22% were assessed as Poor (Figure 4).

Canadian Lake Ontario Beaches (including Niagara River)

From 2018 to 2019, monitored Canadian Lake Ontario beaches were assessed as Good because beaches met Ontario bacterial standards and were open and safe for swimming an average of 91% of the swimming seasons (Figure 3). The results of the linear modeling indicate a 10-year trend of "Improving" with a p-value of 0.03. Monitored Canadian Lake Ontario beaches were open and safe for swimming 86% of the swimming seasons in 2010–2019.
During the 2018-2019 swimming seasons, 31% of Canadian Lake Ontario beaches were open and safe for swimming the entire swimming seasons. Also over this same two-year period, 84% of Canadian Lake Ontario beaches achieved a status of Good, 11% achieved a status of Fair, and 5% were assessed as Poor (Figure 4).

During the entire 2010-2019 swimming seasons, 21% of Canadian Lake Ontario beaches did not exceed the Ontario bacterial thresholds once. Also during this same 10-year period, 72% of Canadian Lake Ontario beaches achieved a status of Good, 14% achieved a status of Fair, and 14% were assessed as Poor (Figure 4).

Linkages

Water Levels

- High water levels may lead to flooding, high erosion rates, unsafe swimming conditions due to currents and waves, and loss of beach area (U.S. Army Corps of Engineers, 2003; Michigan Department of Natural Resources, 2021). In 2019, all the Great Lakes were well above average water levels. Lakes Erie and Ontario both had record high peak water levels in 2019 compared to each Lake's overall monthly record high since 1918. During this year, many Great Lakes Beaches retreated, losing recreational areas, and others were flooded (Frauhammer, 2019; Holland, 2019; Moore, 2019).

- The Great Lakes were in a period of high water levels starting in 2015. Lakes Superior, Michigan, Huron, and Erie were above monthly water level averages from 2015 through 2019. Lake Ontario water levels were above the monthly water level average from 2017 through 2019 except for three months within the time frame (U.S. Army Corps of Engineers, 2020).

Precipitation Amounts

- Heavy precipitation and storm water runoff can increase pathogens in recreational waters by flushing pathogens directly into nearby surface water. Storm water runoff can collect pathogens from soil, sand, streets, or agricultural land. Heavy precipitation can overwhelm combined sewer systems or municipalities with aging infrastructure resulting in the discharge of untreated wastewater or storm water into streams, rivers, drains, or lakes. Heavy precipitation and storm water runoff can also disturb and re-suspend pathogens in sediments and beach sand (Patz, Uejio, & McLellan, 2008; Staley et al. 2018a)
- Beach postings may be the result of bacterial loadings from tributaries and extreme precipitation events. Improved wastewater treatment in response to these pressures may reduce the number of beach postings. Implementation of best management practices and green infrastructure to reduce the volume of storm water runoff may also decrease the number of beach advisories (Podolsky and MacDonald, 2008).
- From the fall of 2018 through the spring of 2019, the Great Lakes Basin experienced a wet period with precipitation ranging from 109-154%, 91-101%, 101-118% of average precipitation for the basin during fall 2018, winter 2018-2019, and spring 2019 respectively. This was followed by a dryer summer in 2019 and another autumn with above average precipitation ranging from 88 to 125% of average autumn precipitation values (Great Lakes Quarterly Climate Impacts and Outlook: December 2018, March 2019, June 2019, September 2019, December 2019).

Cladophora/Harmful Algal Blooms

- Harmful Algal Blooms (HABs) may produce toxins, which can have adverse health effects. Thereby, a beach may be considered unsafe for swimming due to a HABs event, and a beach advisory may be posted due to HABs (Centers for Disease Control and Prevention, 2018). The U.S. BEACON database does not contain information on advisories due to a HABs event alone. This report (for both Canadian and U.S. portions) solely focuses on beach advisories due to the exceedance of *E. coli*.
- *E. coli* also has a unique, strong association with *Cladophora*. *Cladophora* provides a suitable habitat for indicator bacteria and potential pathogens to persist and potentially grow, which may, in turn, impact recreational beach water quality (Englebert et al., 2008; USGS, 2009).

Invasive Species – Dreissenid Mussels

- Dreissenid mussels have carpeted much of the bottom of the Great Lakes in high densities. Filtration by dreissenid mussels produces clearer waters, which allows for UV rays to penetrate the surface. Sunlight can cause cellular damage in bacteria, which may be linked to the decrease in *E. coli* levels across Lake Michigan (Weiskerger and Whitman, 2018). However, cellular damage can also impact "good" bacteria, which are responsible for sustaining the health of the ecosystem by recycling carbon and nutrients.

Traditional Ecological Knowledge (TEK), Citizen Science and other Bodies of Knowledge

With funding support from Environment and Climate Change Canada's (ECCC) Great Lakes Protection Initiative, Swim Drink Fish has established and supported monitoring hubs in six Great Lakes communities, including three First Nations. These hubs empower citizen scientists to educate their communities on water quality issues, including engaging them in collecting *E. coli* recreational water quality data on a weekly basis. Samples are analyzed in-house and data are made available to the public via www.theswimguide.org/. In addition, hubs engage citizens in water literacy awareness events. The long term goal of this project is to develop a model to be used by communities to initiate their own monitoring hubs (Swim Drink Fish, 2021a).

The Toronto Hub (launched in 2016) and the Kingston Hub (launched in 2020) are projects that have both been implemented by Lake Ontario Waterkeeper (Swim Drink Fish, 2021a). The Lake Erie-Niagara Hub (launched in 2019) is hosted by the Niagara Coastal Community Collaborative (Swim Drink Fish, 2021a). The Zhiibaahaasing First Nation Hub (launched in 2018) is hosted by the Zhiibaahaasing First Nation. This hub goes beyond monitoring Lake Huron beaches by increasing awareness on the science behind changing water quality and supporting the sharing of traditional knowledge with the youth of the community on the importance of water (Swim Drink Fish, 2021b). In 2021, monitoring hubs were initiated at Garden River First Nation in the St. Mary's River watershed and Biigtigong Nishnaabeg First Nation on Lake Superior.

These hubs represent valuable sources of information that could potentially be implemented into future SOGL Beach Advisories sub-indicator reporting. The feasibility of collaborating with and including data from these hubs, as well as additional Swim Drink Fish efforts, will be explored for future SOGL cycles.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: US: https://watersgeo.epa.gov/beacon2/reports.html Canadian data may be accessed from Public Health Units upon request.		

Data Limitations

- Variability in the data from year-to-year may result from the variations in monitoring and reporting and may not be solely attributable to actual increases or decreases in levels of microbial contaminants. In addition, variability of weather from year to year may affect the variability in bacterial counts.
- Viruses and parasites, although a concern in recreational waters, are difficult to isolate and quantify at present, and widely available measurement techniques have yet to be developed.
- Although considered reliable indicators of potential harm to human health, the presence of *E. coli* and/or *Enterococcus* may not necessarily be related to fecal contamination.
- Although data obtained from the U.S. and Canada are comparable in terms of quality of data from the source, the data are NOT comparable in terms of actual beach postings since each country uses different posting criteria and frequency of monitoring. The U.S. posts a beach as unsafe if *E. coli* levels are above 235 *E. coli* per 100 mL (State of Michigan uses 130 *E. coli* cfu per 100 mL as a 30-day geometric mean, and a maximum of 300 *E. coli* cfu per 100 mL), whereas Ontario (based on the 2018 guidelines) considers a beach as unsafe if *E. coli* levels exceed the threshold of 200 *E. coli* cfu per 100 mL (based on a geometric mean of at least 5 samples), or a single-sample maximum concentration of at least 400 *E. coli* cfu per 100 mL. As mentioned previously in this report however, some PHUs may use different thresholds.
- The U.S. long-term trend assesses the entire data set available for U.S Great Lakes beaches using the data available in the BEACON database, which is updated annually, and basin wide data entries started in 2007. The long-term trend for Canadian Great Lakes beaches is Undetermined. Older data are available, however it is not presently possible to determine a long-term trend, as it has not been reanalyzed using the 2018 Ontario *E. coli* thresholds, and therefore it is not comparable.
- Currently, it is difficult to report on beach advisories as they relate to Harmful Algal Blooms (HABs). Some health units are noting that a bloom is present while testing for *E. coli.* Still, specialized and expensive tests are needed to determine if the algae are toxic. Beach advisories/closures as a result of HABs/algal blooms may be a future component of this report, however this sub-indicator currently only assesses the percentage of days that beaches are open and safe for swimming during the beach season based on *E. coli* levels.
- There is variability in state beach monitoring frequencies. Information about developing state beach monitoring programs is included in EPA's National Beach Guidance and Performance Criteria for Grants, http://www2.epa.gov/beach-tech/national-beach-guidance-and-required-performance-criteria-grants. As a BEACH Act grant entity, states are required to develop a tiered monitoring plan which must adequately address the frequency and location of monitoring and the assessment of coastal recreation waters based on a review of existing monitoring data, periods of recreational use of the waters, the nature and extent of use of the waters, the proximity to known point and nonpoint sources of pollution, and the effect of stormwater runoff on the waters. The goal of a tiered monitoring plan is to define combinations of monitoring activities that align with identified priorities (tiers), are appropriate for the level of risk and use of a given beach, effectively allocate available monitoring resources and address site-specific circumstances. A BEACH Act grant-funded program must prioritize the use of grant funds for monitoring on the basis of the use of the waters and risk to human health. For example, Michigan which has hundreds of coastal beaches monitors its Tier 1 beaches once per week whereas Illinois monitors many of its beaches 7 days per week. Michigan's annual beach open status averages at being

open at approximately 97% of the time; whereas, Illinois averages more in the 89% range. Tier 1 or "high priority" beaches are monitored more frequently than Tier 3 beaches, some of which may not be monitored at all. The monitoring frequencies for each entity can be found at EPA's Beach Advisory and Closing Online Notification (BEACON) system (under Report Selection) at:. http://watersgeo.epa.gov/beacon2/reports.html.

- Most Canadian Great Lakes beaches are monitored weekly, but some may be monitored more or less frequently based on PHU resources, frequency of public use, exceedance history of a beach, and the current conditions. Monitored beaches may be closed because of exceedances for longer than necessary, or beaches may remain open if *E. coli* levels rise above the safe levels between monitoring days. Ontario PHUs are generally tasked with monitoring beaches in their county, but health units don't necessarily monitor all beaches in their area.
- This report includes information on the percentage of Canadian beaches that are open the entire swimming seasons, however it is important to consider that a relatively small portion of beaches are monitored daily. Most beaches are monitored weekly or less frequently, including many beaches that are monitored monthly. Therefore it is possible that there may have been *E. coli* exceedances in between monitoring events, however they would not have been acknowledged as the beach was not monitored when those exceedances occurred.
- At this time, the Canadian content in this report includes beach water quality data from Ontario PHUs, and does not include Provincial Park or other sources of beach water quality data. Opportunities to include alternative beach data sources will be explored for the next SOGL reporting cycle.
- The percentage of days that U.S. Great Lakes beaches were open and safe for swimming for 2007-2017 calculated in this report vary slightly from the calculations in the past report due to data management within the BEACON database. If errors in data are found, then the data are removed from the database. As a result, data accessed in 2017 may be different from data accessed in 2020. Data in this report were accessed from BEACON in December 2020-Janurary 2021.
- Beaches located on the Great Lakes connecting channels are included in this assessment, however beaches on the St. Lawrence River are not.
- Not all PHUs provide single sample data for Canadian beaches, some only provide geometric mean data. Therefore, during the reanalysis, it is possible that there may have been additional cases where the beaches would have exceeded the 2018 Ontario bacterial thresholds, which includes a max single sample count of 400 *E. coli* cfu/100 ml, but these weren't counted due to only having the geometric mean.
- It was not possible to reanalyze 2011 and 2012 swimming season data from beaches monitored by the Niagara Regional Area Health Unit, as the raw data was not provided in time to conduct the reanalysis. Data from those years are not included in this report.
- In Ontario, some PHUs may use predictive modelling to assist with decision to post beaches as unsafe for swimming, however this report only analyzed the raw *E. coli* PHU sampling results, and not modelling results.

Additional Information

Additional point and non-point source pollution at coastal areas due to human population growth and increased land use may result in additional beach postings, particularly during wet weather conditions, unless contaminant sources are reduced or removed (or new controls introduced). Great Lakes beach sample results generally contain similar bacteria levels after events with similar meteorological conditions (primarily wind direction and the volume and duration of rainfall). If episodes of poor recreational water quality can be associated with specific events (such as meteorological events of a certain threshold), then forecasting for episodes of elevated bacterial counts may become more accurate.

Research on Emerging Technologies at Toronto Beaches

In recent years, beaches located in Toronto and the surrounding area have served as a prime area for research, development and application of innovative technologies to identify sources of microbial contamination. In 2018, water samples from Toronto Harbour and the Don River watershed were analyzed for *E. coli*, wastewater chemicals and microbial source tracking using a digital PCR (dPCR) technique (Edge et al., 2020). Microbial DNA markers were found to be useful for interpreting the sources of elevated *E. coli* concentrations, and were used to identify bacteria found in the gut of seagulls to distinguish seagull fecal contamination from that of other birds, including Canada Geese. While wet weather events were correlated to increases in *E. coli* and human DNA markers, human DNA markers were also widely detected on dry sampling days, suggesting widespread sewage-cross connections into stormwater and dry weather combined sewer overflow systems (Edge et al., 2020). These results identified that the cumulative impact of urban cross-connections is likely greater than initially thought, and also that human sewer contamination is not just a wet weather combined sewer overflow concern.

At Marie Curtis Beach, Staley et al. (2018a) applied environmental DNA (eDNA) sequencing to analyze DNA extracted from water samples to improve the understanding of sources of fecal pollution. Water samples were analyzed for DNA sequences from human, as well as other mammals or bird species. The results of this study indicated not only that extreme rain events can significantly elevate *E. coli* concentration in beach waters, but also that they can cause an increase in the diversity of mammal and bird eDNA sequences (Staley et al., 2018a).

At Rouge Beach, located in the Toronto and Region Area of Concern, Staley et al. (2018b) assessed *E. coli*, as well as human-specific and gull-specific quantitative Polymerase Chain Reaction (qPCR) microbial source tracking markers. A preliminary comparison using dPCR methodologies for both human- and gull-specific microbial source tracking markers was conducted to assess sensitivity and specificity. The study found that the occurrence of human fecal contamination along Rouge beach was associated with rain events, and that during dry weather, the predominant source of fecal contamination was sourced from gull droppings. Additionally, this study determined that while dPCR and qPCR methodologies identified matching levels of human and gull markers in stormwater and beach locations respectively, the dPCR multiplex assay was more sensitive, and detected fecal contamination that was not detected by qPCR assays. These results indicate that dPCR assays could potentially be a valuable tool used by beach managers to identify of low levels of fecal contamination (Staley et al., 2018b).

There may be new indicators and new detection methods available through current research efforts occurring binationally in both public and private sectors and academia. Although currently a concern in recreational waters, viruses and parasites are difficult to isolate and quantify, and widely available measurement techniques have yet to be implemented. Although considered reliable indicators of potential harm to human health, the presence of *E. coli* and/or Enterococcus may not necessarily be related to fecal contamination.

New rapid detection methods are beginning to be used at several Great Lakes locations to provide the public with real time beach water quality information. The City of Racine Health Department is using the rapid quantitative Polymerase Chain Reaction (qPCR) method for *E. coli* at North Beach, along with the 18 hour culture method (Colilert), to validate the method. Racine was the first entity in the Great Lakes to use the rapid qPCR method for *E. coli*. The Wilmette, Illinois Water Utility and Milwaukee, Wisconsin Health Department have also done some culture/qPCR comparative testing for *E. coli* at some of their beaches. Various entities in Michigan are also beginning to use the rapid qPCR method for *E. coli* along with Colilert. EPA's Office of Research and Development in Cincinnati, Ohio has assisted Michigan Department of Environmental Quality (MDEQ) by providing training to multiple health departments in the state. Although this approach is feasible for beach water quality monitoring, it is very expensive.

Beach Management Success Stories

The status of water quality at Great Lakes beaches is impacted by harmful factors such as pathogens and *E. coli*. However, there have been instances where beaches with poor water quality and deemed unsafe for swimming, have shown remarkable improvement through remedial action efforts. As a result, some of these beaches now meet water quality thresholds, and are now open and safe for swimming 80-100% of the swimming season.

Bluffer's Park Beach

Bluffer's Park Beach, located within the Toronto and Region Area of Concern, has historically been impacted by bacterial contamination. Since the 1980's, beach postings would often exceed 80% of the swim season, including one year (2005) where the beach was unsafe for swimming for 93% of the swimming season (City of Toronto, 2009; Lake Ontario Waterkeeper, 2006). From 2005 to 2007, expanded *E. coli* surveillance and microbial source tracking techniques were applied to identify the sources of fecal pollution. Significant *E. coli* hotspots were identified in the beach sand pore water, as well, following rain events, overflow from nearby streams and a parking lot were found to contaminate beach waters. Microbial source tracking results indicated that the predominant source of contamination was animal fecal pollution, rather than human sewage. These findings were consistent with the observations of waterfowl (mainly gulls and Canada geese) and wildlife in the beach area, including observations of fecal droppings, which are direct sources of *E. coli* (Edge et al., 2018). As a remedial effort, the City of Toronto implemented a bird management program which included using trained dogs to deter waterfowl, as well as public education to deter feeding of waterfowl. Additionally, the wetland and dune system behind the beach was reengineered, increasing the water retention capacity of the marsh, as well as creating a barrier to prevent runoff onto the beach and direct stormwater overflows into infiltration basins. Community engagement was also a key player in this restoration process, as members of the public assisted with planting dune and wetland areas (Edge et al., 2018). Through these remedial efforts, water quality improved significantly at Bluffer's Park Beach, and in 2011 it was officially designated as a Blue Flag beach, thereby passing water quality tests over 80% of the time, and fulfilling environmental management, and safety standards (Toronto and Region Conservation Authority, 2017). As of 2021, Bluffer's Park Beach is still officially a Blue Flag Beach (Lake Ontario Waterkeeper, n.d.).

Niagara River Area of Concern

Since 1993, the *Beach Closings* Beneficial Use Impairment (BUI) has been designated as 'Impaired' for the Canadian side of the Niagara River Area of Concern due to water quality goals not being met at Queen's Royal Beach (Green, 2021); the only public beach on the Canadian side of the Area of Concern. Extensive microbial source tracking studies between 2010-2015 indicated the predominant source of bacterial contamination was from a nearby stormwater outlet. Further investigations identified the catchment area of this outlet had several issues with bacterial contamination from various sources. Remedial actions were undertaken to address these sources including the installation of raccoon gates, sewer infrastructure improvements, and construction of a bioswale to filter stormwater before it flows to the beach. Water quality monitoring has evaluated beach conditions following these remedial actions, and results indicate improvements such as an 82% reduction of animal sourced *E. coli* DNA markers in wet weather, and a 99% reduction in human DNA marker compared to highest recorded value in 2014 (Green, 2021). Moreover, monitoring at Queen's Royal Beach confirms that at least 80% of samples met the swimming guidelines from 2018-2020. As a result, in 2021, the Remedial Action Plan Team recommended that the *Beach Closings* BUI be changed to 'Not Impaired'.

Acknowledgements

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Information Sources

Great Lakes beach data provided by U.S. EPA http://watersgeo.epa.gov/beacon2/

Canadian Great Lakes Beach data provided by the following Ontario Health Units with beaches residing along the Great Lakes: Algoma; Chatham Kent; Durham Region; Elgin St. Thomas; Grey Bruce; Haliburton Kawartha Pine Ridge District; Halton Region; Hamilton; Hastings and Prince Edward Counties; Huron County; Kingston; Lambton County; Niagara Region; North Bay Parry Sound District; Peel Region; Simcoe Muskoka District; Sudbury & District; Thunder Bay District; Toronto; Windsor-Essex County

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State of the Great Lakes 2022 Report

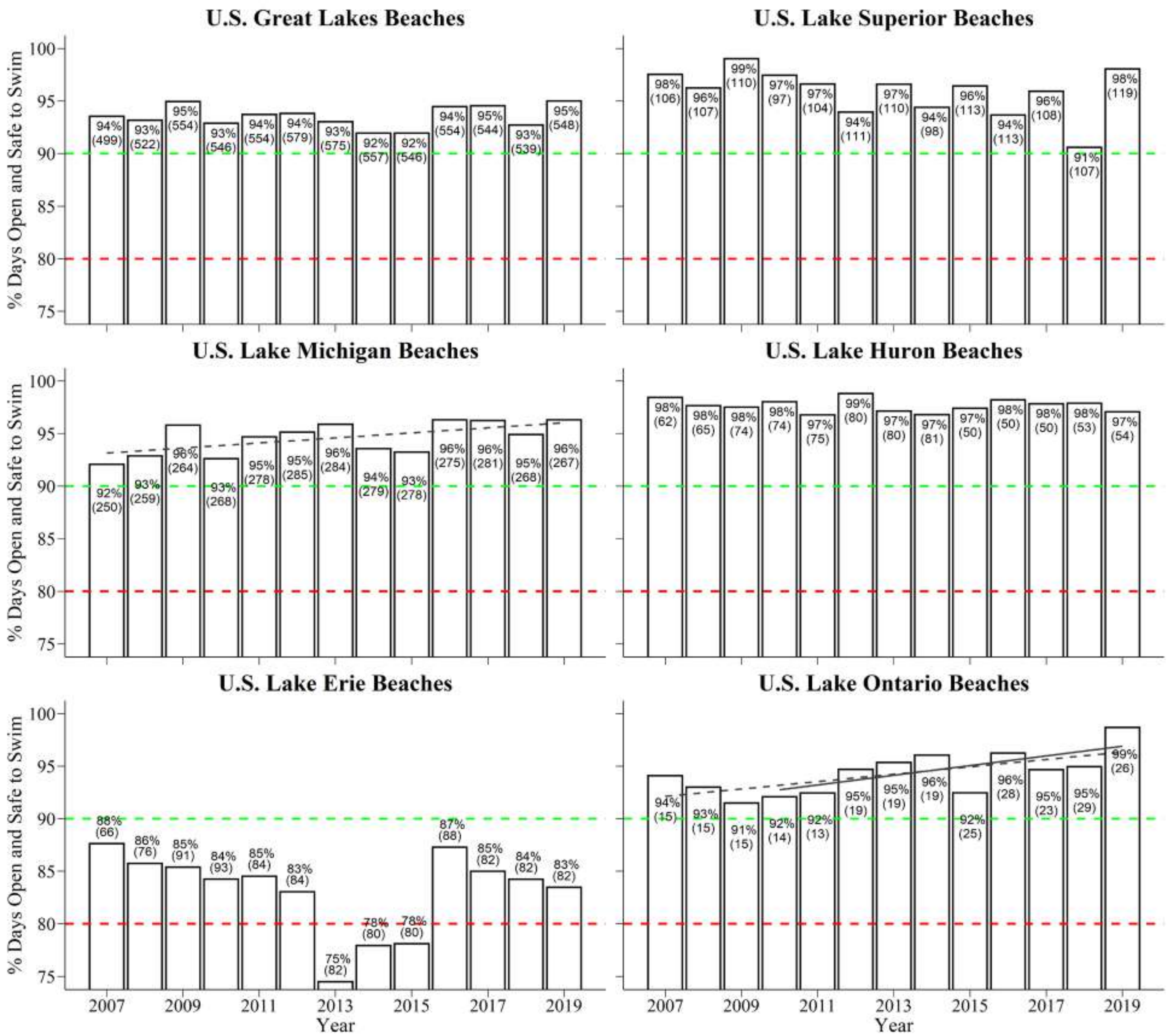

Figure 1. Percentage of days during the swimming season that U.S. Great Lakes beaches were open and safe for swimming from 2007-2019. A black dashed line is present if there was a significant (p-value ≤ 0.05) long-term trend in the data, and a solid black line is present if there was a significant (p-value ≤ 0.05) 10-year trend in the data. The value in the ( ) represents the number of beaches monitored each year. The dashed green line represents the US threshold for "Good" of 90% of beach days open, and the red dashed line represents the US threshold for "Poor" of 80% of beach days open.

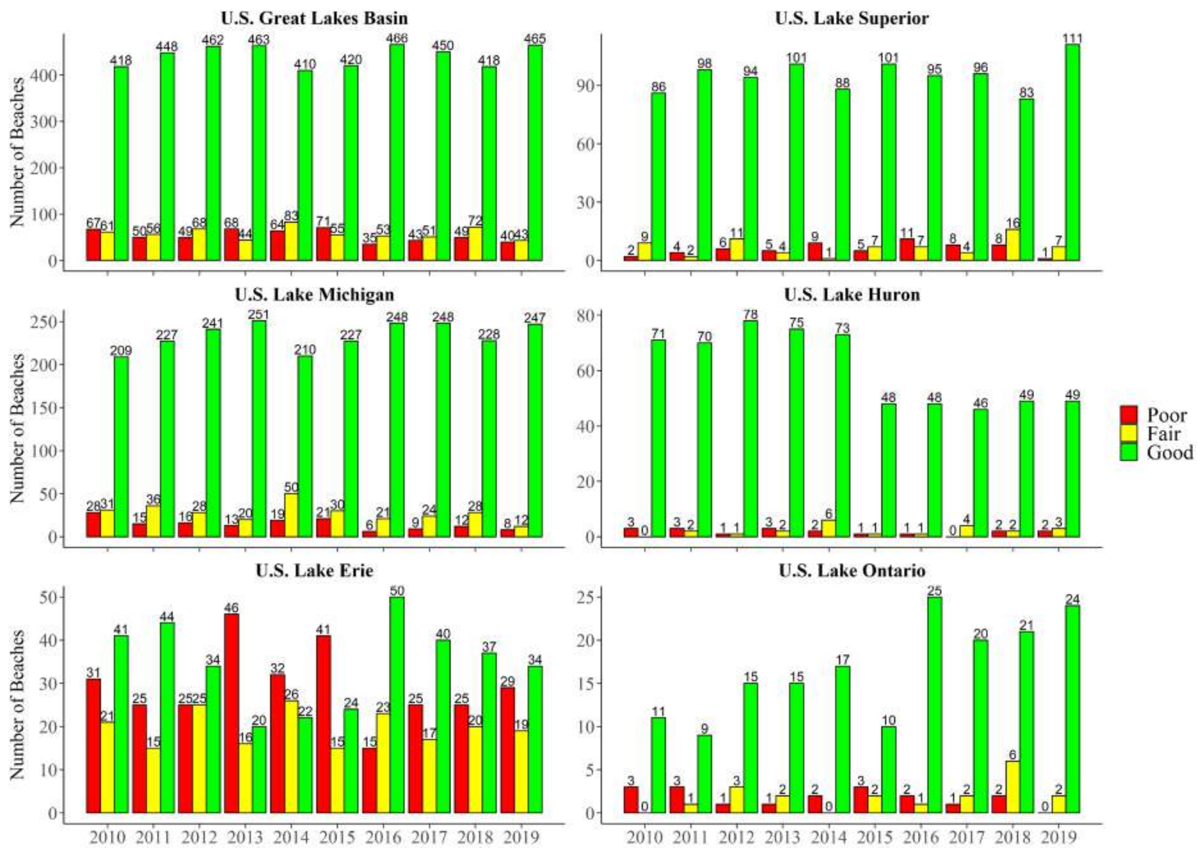

Figure 2. The number of U.S. Great Lakes beaches that fell into the "Good", "Fair", and "Poor" categories for the period 2010-2019. The value above the bars indicates the total number of beaches that fell into that category.

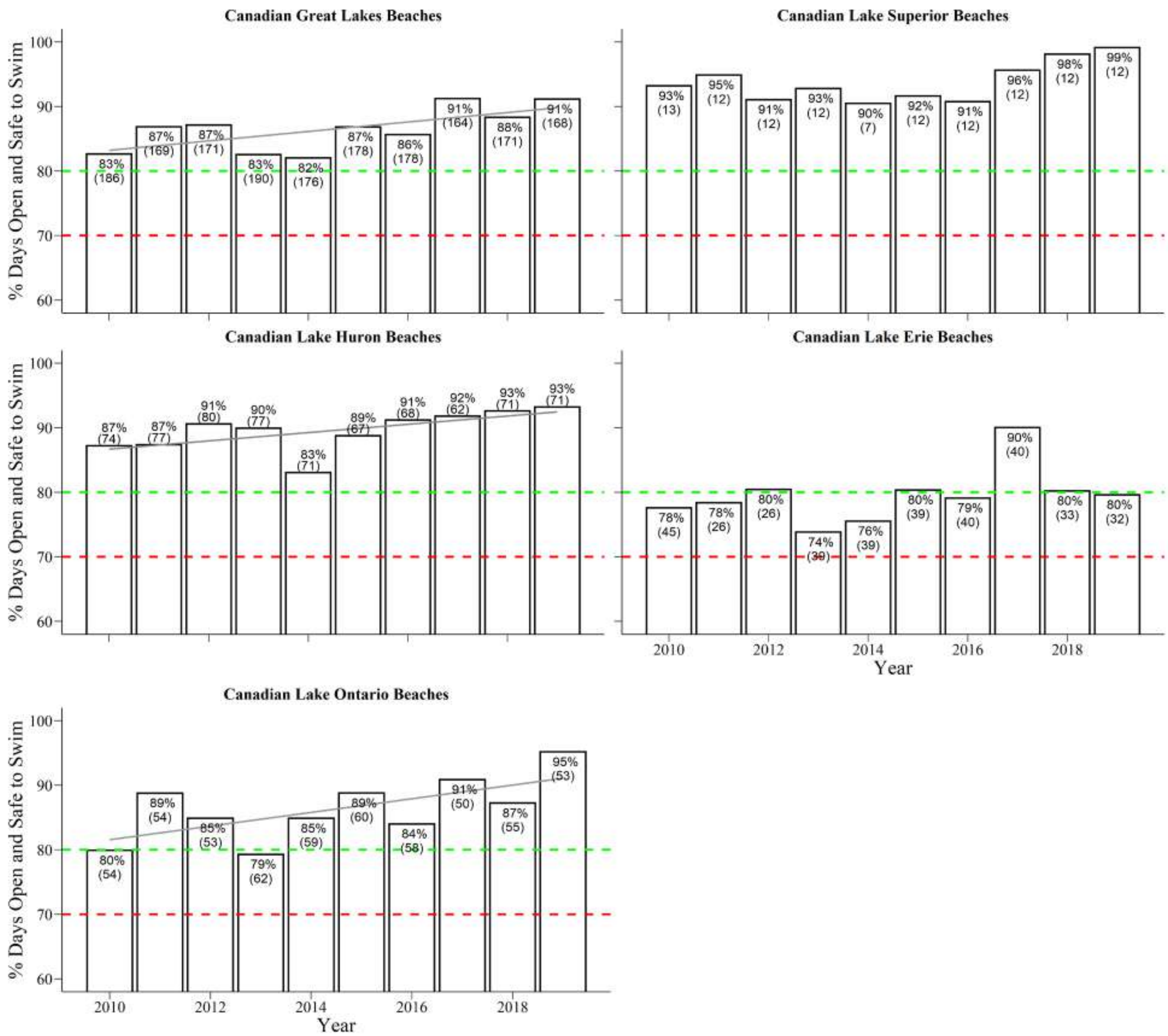

Figure 3. Percentage of days during the swimming season that Canadian Great Lakes beaches are open and safe for swimming from 2010-2019. A trend line is present if there was a significant (p-value $\leq$ 0.05) 10-year trend in the data. The value in the "()" represents the number of beaches monitored each year. The dashed green line represents the Canadian threshold for "Good" of (80% of beach days safe to swim), and the red dashed line represents the Canadian threshold for "Poor": (69.9% of beach days safe to swim). All bars above the green dashed line are considered Good, all bars between the green and red dashed lines are considered Fair, and all bars below the red dashed line are considered Poor.

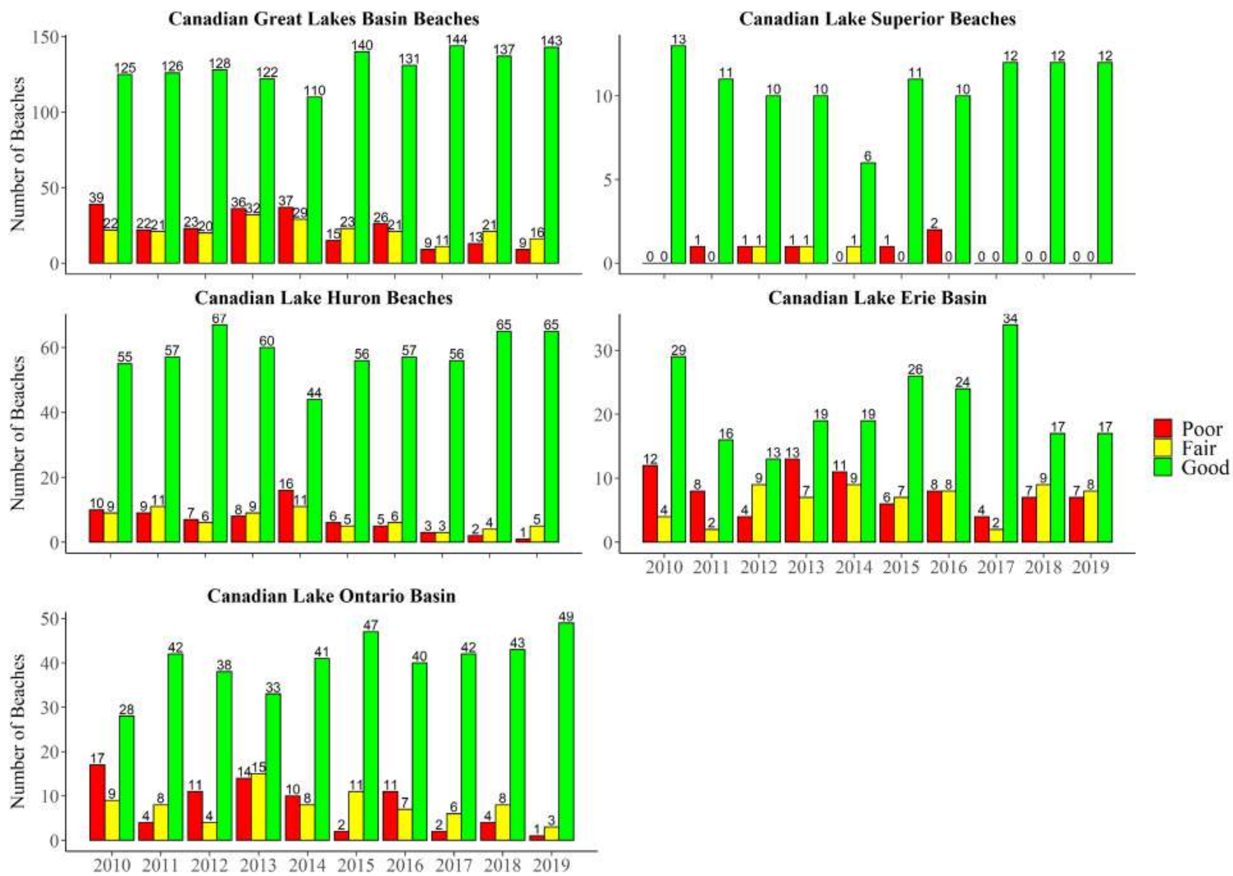

Figure 4. The number of Canadian Great Lakes beaches that fell into the "Good", "Fair", and "Poor" categories for the period 2010-2019. The value above the bars indicates the total number of beaches that fell into that category.

Sub-Indicator: Contaminants in Edible Fish

Overall Assessment

Status: Fair

Trends:

10-Year Trend: Improving
Long-term Trend (1975-2019): Improving

Rationale: Fish consumption advisories for all of the Great Lakes have typically been driven by polychlorinated biphenyls (PCBs). Levels of PCBs in the edible portion of Great Lakes fish have declined substantially (more than 90% in many cases) since regulatory and remedial actions have taken place since the 1970s (Figure 1). Mercury has been a secondary driver of the fish consumption advisories. Mercury levels in the edible portion of Great Lakes fish have also declined by about half during the last 40-50 years (Figure 2). Overall, long-term trends have shown improvements in the levels of both contaminants for all of the five Great Lakes. Looking at short-term trends for the past 10 years, PCB levels in the edible portion of fish tissue have also either improved or remained unchanged for all of the Great Lakes (Figure 1). However, recent levels of PCBs have generally remained in the "Fair" category (Figure 1, Table 1), as one or more of the fish species and sizes examined have had contaminant levels that exceeded the "Good" category criteria (Table 2). In contrast, short-term trends in mercury levels appeared to have largely remained stable or slightly improved during the last decade (Figure 2), and the recent mercury levels have generally maintained the status of "Good" (Table 2).

Status and trend assessments were based on data generated by the Province of Ontario for Lake Superior, Lake Huron, Lake Erie and Lake Ontario. Lake Michigan status and trend assessments were generated based on data provided by the U.S. EPA and States of Michigan, Wisconsin, Illinois, and Indiana. Reliance upon a single data source, such as the Province, allowed for more consistent comparison between reporting cycles but may not reflect the status and/or trends at the State level.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (1975-2017): Improving

Rationale: Lake Superior fish consumption advisories are primarily driven by PCBs and secondarily by mercury. Other fish consumption advisories for Lake Superior are due to elevated levels of dioxins/furans and toxaphene. Substantial improvements have been observed in the levels of all of these contaminants during the last 45 years; however, the levels appear to be unchanging in the recent years (e.g., Figures 1 and 2). Average levels of both PCBs (0.019-0.27 μg/g) and mercury (0.10-0.31 μg/g) mostly remain in the "Fair" category (Figures 1 and 2, Table 2). However, new monitoring data are needed to better assess the current status and short-term trend with confidence. Considering improved contaminant levels in edible fish tissue during the last 20 years, fish consumption advisories are expected to improve when more widespread new monitoring data are collected.

Lake Michigan

Status: Fair

10-Year Trend: Improving

Long-term Trend (1975-2019): Improving

Rationale: PCB levels in Lake Michigan fish were historically among the worst in the Great Lakes, but substantial improvements have now resulted in diminished differences when compared with the other Great Lakes. Nevertheless, the most recent Lake Michigan PCB measurements remain typically higher than the other Great Lakes (Figure 1), and have been primarily responsible for fish consumption advisories in Lake Michigan. During the last decade, Coho Salmon showed improved PCB levels; however, Chinook Salmon, Lake Trout, Walleye and Lake Whitefish showed worsening levels of PCBs (Figure 1). For mercury, Lake Trout levels improved, and Coho Salmon and Whitefish levels remained unchanged, but Chinook Salmon and Walleye levels deteriorated (Figure 2). A closer examination of the recent monitoring data indicated that differences in measurements collected by different states over the years and low number of measurements in some cases have contributed to the variability. In order to improve the understanding of the current status and trends for Lake Michigan, a coordinated lake-wide monitoring and comprehensive statistical analysis of fish PCB and mercury levels is recommended. Meanwhile, in the absence of reliable new data, previous assessments of “fair” status and “improving” recent trend have been continued in this update.

Lake Huron

Status: Good

10-Year Trend: Improving-Unchanging

Long-term Trend (1976-2018): Improving

Rationale: In the 2019 SOGL report, it was noted that although PCB and mercury levels in Lake Huron fish improved during the 1970s and 1980s, slower improvements have been observed since then (Figure 1). This slowdown was attributed to disturbances in the food web structure of Lake Huron due to invasive species. The most recent monitoring data considered in this reporting cycle has indicated continued decline in the level of PCB (Figure 1), which has been the prime contaminant responsible for the fish consumption advisories for Lake Huron. Mercury levels appear unchanging over the past couple of decades, but these levels have been considered generally low (Figure 2, Table 2). Overall, recent levels of PCB (26-110 ng/g) and mercury (0.09-0.25 µg/g) in edible fish tissue have on average been categorized as “Good” (Figures 1 and 2, Table 2), compared to “Fair” in the SOGL 2019 assessment.

Lake Erie

Status: Fair

10-Year Trend: Improving

Long-term Trend (1976-2019): Improving

Rationale: PCB and mercury levels in Lake Erie fish were historically among the lowest and improved further during the 1970s and 1980s (Figure 2). However, as reported in the SOGL 2019 report, these levels then stabilized and, in some cases, slightly increased (Figure 2). The increase in mercury and PCB levels were attributed to possible alterations of the food web from invasive species such as mussels and round goby (Hogan et al. 2007, Bhavsar et al. 2010). The recent measurements for this assessment have been generally lower and indicate improved trends on both long- and short-term basis (Figures 1 and 2). Recent mercury levels (0.01-0.21 μg/g) in edible fish tissue have been categorized as "Good", but PCB levels (89-436 ng/g) have been categorized as "Fair" in some cases (Figures 1 and 2, Table 2).

Lake Ontario

Status: Fair

10-Year Trend: Improving

Long-term Trend (1976-2019): Improving

Rationale: PCB levels in edible portions of Lake Ontario fish were among the worst in the Great Lakes but have generally shown over 90% improvements since the 1970s (Figure 1). Despite such substantial improvements, PCB levels still remain elevated compared to the advisory benchmarks causing the majority of restrictions advised on eating Lake Ontario fish. The levels (25-266 ng/g), however, are considered "Fair" and are closer to the "Good" category than "Poor" (Figure 1, Table 2). PCB levels during the last 10 years in all five types of fish examined suggest that Lake Ontario is continuing to improve. Mercury levels have also declined by about half during the last 40-50 years (Figure 2). The most recent mercury levels (0.09-0.17 μg/g) in limited sizes of all five types of fish examined are considered low (Figure 2, Table 2).

Status Assessment Definitions

Good: Recent average mercury concentration <0.25 parts per million (ppm or μg/g) and PCB concentration <0.1 ppm in edible fish tissue typically resulting in 8 or more meals per month advisories.

Fair: Recent average mercury concentration between 0.25 and 1 ppm or PCB concentration between 0.1 and 1 ppm in edible fish tissue typically resulting in 1 to 4 meals per month advisories.

Poor: Recent average mercury or PCB concentration >1 ppm in edible fish tissue typically resulting in <1 meal per month or "do not eat" advisories.

Undetermined: Contaminant data have not been available or have been insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Concentrations of PCBs and mercury are on average declining.

Unchanging: Concentrations of PCBs and mercury are not declining or increasing.

Deteriorating: Concentrations of PCBs and mercury are on average increasing.

Undetermined: Insufficient information to derive trends with confidence.

Endpoints and/or Targets

An assessment of contaminants of interest in edible fish tissue can help track progress towards meeting the Ecosystem Objective of "Fish in the Great Lakes ecosystem should be safe to eat. Consumption should not be limited by contaminants originated from human activities". Assessments include long term and short-term trend evaluation of levels of contaminants of concern in edible fish tissue. A declining trend for these contaminants represent an improvement in the environment and the potential for reduced exposure to contaminants from consumption of Great Lakes fish. The elimination of restrictive fish consumption advisories or a change from more restrictive to less restrictive fish consumption advice in the Great Lakes may also be considered to be an appropriate endpoint (See Tables 1 and 2).

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess levels of contaminants that pose a risk to human health and infer the potential harm through the consumption of Great Lakes fish. Special emphasis has been paid to contaminants that have triggered fish consumption advisories such as persistent, bioaccumulative, and toxic (PBT) contaminants, including mercury and PCB in edible fish tissue of Great Lakes fish.

This Contaminant in Edible Fish report includes contaminant information for the fillet of the fish that are typically consumed by people. The Toxic Chemicals in Whole Fish sub-indicator report includes contaminant information for the bones, organs, scales, blood, etc. of the fish. Other differences exist in the species of fish reported and the chemicals assessed in each report.

Ecosystem Objective

Fish in the Great Lakes ecosystem should be safe to eat. Consumption should not be limited by contaminants originating from human activities.

This sub-indicator best supports work towards General Objective #3 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the waters of the Great Lakes should "allow for human consumption of fish and wildlife unrestricted by concerns due to harmful pollutants"

Both the U.S. and Canadian agencies monitor contaminants in edible portions of Great Lakes fish to provide advice on safe consumption. This sub-indicator assesses the status of the ecosystem by comparing contaminant concentrations in fish to risk-based contaminant levels established by fish advisory programs that result in consumption advice (Tables 1 and 2). The outcome of this comparison is then used to relate the ecosystem status to General Objective #3 of the GLWQA.

Measure

Concentrations of contaminants (e.g., PCBs, mercury) from the edible fish tissue of species most consumed by Great Lakes basin citizens are used as an indicator of exposure to these contaminants. Temporal trends of contaminants and assessment of the risk-based advisory levels associated with these contaminants are used to track progress toward meeting the Ecosystem Objective. Levels of contaminants in fish are measured in individual or composited samples noting that they vary by type and size of fish and by location. Choosing appropriate indicator species is crucial and should be based on fish consumption patterns and availability of data, both of which can change over time. Therefore, there may not be complete consistency between reporting years.

For the Province of Ontario, fish are collected annually, in accordance with the Fish Contaminant Monitoring Program of the Ontario Ministry of the Environment, Conservation and Parks (OMECP). US samples are collected according to the Great Lakes State programs. For the status assessment, levels and trends of PBT chemicals in the most edible portion of fish (i.e., fillets) are assessed and compared against the concentration cut-off criteria derived from the advice categories used by the Province of Ontario and U.S. Great Lakes States identified in the Guide to Eating Ontario Fish and the Protocol for a Uniform Great Lakes Sport Fish Consumption Advisory (Tables 1 and 2).

Since the 1970s, there have been improvements in the levels of many PBT contaminants in the Great Lakes basin due to bans on their use and/or production and restrictions on emissions. However, because of their ability to persist in the environment and bioaccumulate, some PBT contaminants continue to be of concern. Historically, elevated levels of a variety of contaminants including PCB, mercury, dioxins/furans, mirex and toxaphene have restricted consumption of Great Lakes fish. However, concentrations of many PBTs, including toxaphene and mirex, have declined to levels such that they can be eliminated from regular monitoring to prioritize resources for other purposes such as monitoring contaminants of emerging concern (CECs, Gandhi et. al 2014, 2015). Because PCB and mercury are currently the most limiting contaminants in eating Great Lakes fish, the assessment is largely evaluated based on findings of PCB and mercury. PCB concentrations are high enough that they are the predominant chemical driving the fish consumption advice. When providing advice due to PCB and mercury, the advice is believed to be protective enough against all other monitored chemicals.

Monitoring of CECs continues to be a priority for Provincial, State, and Tribal programs as concentrations and toxicity of these contaminants continue to be assessed for inclusion into advice. At this time, however, risks due to the identified CECs do not exceed those from PCBs and mercury. For this assessment, per- and polyfluoroalkyl substances (PFASs) were considered, but polybrominated diphenyl ethers (PBDEs) and polychlorinated naphthalenes (PCNs) were omitted because fish consumption is not considered the primary route of exposure at present (Gandhi et al. 2017a, Gewurtz et al. 2018, Lorber 2008). Levels of PBDEs appear to have declined in edible fish tissue by 46–74% between 2006/07 and 2012, and although they will remain in-use in existing consumer items for a while, their accumulation in fish will not be substantial (Gandhi et al. 2017a). Similarly, the levels of PCNs are relatively low in Great Lakes fish, and important PCN contaminants (PCN-66/67) appear to have declined between 2006 and 2012 (Gewurtz et al. 2018). These trends are complemented by previously reported decrease in PCNs in Lake Ontario Lake Trout between 1979 and 2004 (Gewurtz et al. 2009).

Ecological Condition

History and Background

Fish contaminant monitoring data included in this assessment include those produced annually by Ontario's Fish Contaminant Monitoring Program, individual State monitoring programs, and results of the 2010 Great Lakes Human Health Fish Tissue Study. In 2009, U.S. EPA's Great Lakes National Program Office's Great Lakes Fish Monitoring and Surveillance Program eliminated the edible fish analysis portion of its program, refocused its efforts on identifying emerging contaminants in whole fish, and therefore could not contribute new data to this sub-indicator. The analysis for this sub-indicator was limited to fish species that are of interest for human consumption as well as are good indicators of contaminants of concern (i.e., PCB and mercury). Five selected fish species were Lake Trout, Walleye, Lake Whitefish, Coho Salmon, and Chinook Salmon. Fish contaminant levels can be influenced by age, and thereby size, of fish. To prepare spatial and temporal trends, narrow size ranges of 55-65 cm for Lake Trout, Coho Salmon and Chinook Salmon, and 45-55 cm for Walleye and Lake Whitefish were considered. Samples included in the analysis for this sub-indicator were selected to provide results from the widest temporal and spatial scale. This broad scale approach was accomplished by soliciting data generated by the Province of Ontario, the 8 Great Lakes State monitoring programs, and U.S. EPA's 2010 Great Lakes Human Health Fish Tissue Study.

Ontario's Fish Contaminant Monitoring Program

Ontario started monitoring contaminants in fish in the late 1960s. Ontario's Fish Contaminant Monitoring Program was formally established in 1976, and the first fish consumption advisories were issued in 1977. Staff from the Ontario Ministry of Natural Resources and Forestry and OMECP collect the fish, which are then analyzed by OMECP for a variety of substances, including mercury, PCBs, mirex, DDT, dioxins, and contaminants of emerging concern (e.g., PBDEs, PFASs). The results are used to develop the Guide to Eating Ontario Fish, which give size-specific consumption advice for each species and location tested. The Guide to Eating Ontario Fish gives advice to anglers, subsistence fishers and their families, and First Nations and Métis communities in determining which fish species and what size caught from Ontario water bodies can be consumed to minimize exposure to toxins. The Guide compiles information for more than 2,500 locations around the Province of Ontario, including about 60 regions covering the Canadian waters of the Great Lakes.

Great Lakes Human Health Fish Tissue Study

U.S. EPA's Office of Water, Great Lakes National Program Office, and Office of Research and Development collaborated to conduct the Great Lakes Human Health Fish Tissue Study. The Great Lakes Human Health Fish Tissue Study was initiated in 2010 under the Agency's National Coastal Condition Assessment (NCCA), and it is the first statistically based study of fish contamination in the Great Lakes. Fish samples were collected from 157 randomly selected sites in 2010 and 152 randomly selected sites in 2015 throughout the five U.S. Great Lakes. Fillet samples were analyzed for mercury, PCBs, PBDE, PFAS, and omega 3 fatty acids in 2010. Fillet samples were analyzed for mercury, PCBs, PFAS, Dioxin and Furans, Emerging Chemicals, and omega 3 & 6 fatty acids in 2015.

Great Lakes Consortium for Fish Consumption Advisories

The Great Lakes Consortium for Fish Consumption Advisories (Consortium) is a collaboration of fish advisory program managers from government health, water quality, and fisheries agencies in the eight U.S. states bordering the Great Lakes and Province of Ontario. The purpose of the Consortium is to share information about contaminants found in fish of the Great Lakes region, evaluate human health effects of those contaminants, and develop protocols and methods for determining fish consumption advice and communications. The Consortium has its roots in a taskforce formed in the early 1980s. Consortium membership is fluid but typically includes representatives from the eight U.S. states bordering the Great Lakes - Indiana, Illinois, Michigan, Minnesota, New York, Ohio, Pennsylvania and Wisconsin. Staff from the OMECP, and the Great Lakes Indian Fish and Wildlife Commission also participate. The following goals for fish consumption advisories guide the continuing work of the Consortium: 1) Develop common methods for determination of consumption advice and implement consistent advice for shared waters, 2) use, share, and advance credible data and science, 3) evaluate the risks and benefits of consuming Great Lakes fish to develop a shared understanding among Consortium members and incorporate these messages into fish consumption advice, and 4) establish and utilize best practices for communicating risks and benefits and influencing the behavior of fish consumers. Members provided contaminant concentration data for this sub-indicator.

PCB

Level of total PCB in fish ranged from a few hundred to thousands of nanograms per gram (ng/g) during the 1970s (Figure 1). In many cases, these historical levels were higher than the PCB advisory benchmark of about 2 μg/g or parts per million (ppm) at that time (Table 1). PCBs were banned during the late 1970s, which spurred declines in their environmental levels. PCB concentrations have declined substantially over the past four decades in all of the Great Lakes (Figure 1). The declines varied by fish type and lake, but were as much as >90% in many cases. Recent PCB levels in selected sizes of fish from the five Great Lakes are <0.5 μg/g.

Scientific studies conducted between the 1980s and 2000s highlighted greater toxicity of PCBs, which resulted in lower advisory benchmarks over time. At present, the advisory benchmarks for severe restriction on fish consumption (i.e., not more than 1-2 meals per month) are about an order of magnitude lower than the pre-1990s. As such, despite substantial declines in the fish PCB levels, PCBs continue to be of concern for health of humans consuming Great Lakes fish.

PCBs levels appear to be improving in most cases albeit at a slower rate compared to the historical declines during the 1970s and 1980s. A deteriorating trend observed for PCBs in Lake Michigan fish needs to be confirmed with more comprehensive monitoring as discussed above. A greater variability and slower declines at lower levels are typical and other stressors (e.g., invasive species, climate change) may be contributing to the slower improvements. These factors can affect food web dynamics and contaminant pathways to fish higher in the food chain. See Linkages section.

Mercury

Mercury concentrations in fish historically exceeded fish consumption advisory benchmarks more frequently than in current years (Figure 2). The levels have generally declined by about half over the last four decades, and the concentrations in the selected sizes of fish from the five Great Lakes are now below 0.25 µg/g (Figure 2).

PFASs

Emerging contaminants, such as PFASs, in edible portion of Great Lakes fish continue to be a priority for monitoring and surveillance for the Great Lakes States and the Province of Ontario. The Province of Ontario and the Great Lakes States have been examining levels of PFASs in fish, and have issued site-specific advice where fish have been tested for elevated perfluorooctane sulfonic acid (PFOS), a chemical of the PFAS family. A recent publication on PFAS from the National Coastal Conditions Assessment and the National Rivers and Streams Assessment identified that PFOS was the most dominant PFAS found in their samples and that maximum PFOS concentrations were 127 and 80 ng/g in urban river samples and Great Lakes samples, respectively (Stahl, et al. 2014). Recently, fish consumption advisories for select Great Lakes fish/locations were issued by the Province of Ontario and State of Wisconsin due to elevated PFAS levels in some select fish at certain Great Lakes locations. Overall, levels of PFAS have resulted in only about 1% of the fish consumption advisories issued. For this reason, a detailed assessment was not conducted for PFAS in this reporting cycle. However, it would be appropriate to continue monitoring PFASs in the edible portions of Great Lakes fish and shall be considered in the future reports as necessary.

Dioxins/furans, Toxaphene and Mirex

Previous reporting cycles identified that dioxins/furans, toxaphene and mirex are less of a concern than the dominant contaminants that drive consumption advice, such as PCBs and mercury, from the perspective of health risk to humans through fish consumption. Levels of these contaminants have declined over the last 30-40 years and continue to decline in recent years (Gandhi et al. 2014, 2015, 2019). The recent levels have resulted in relatively minor restrictions on eating Great Lakes fish. For this reason, dioxins/furans, toxaphene and mirex are no longer reported in this sub-indicator.

Omega-3 Fatty Acids

Omega-3 Information, Research, and Future work

Fish contain beneficial nutrients such as Omega-3 fatty acids, high quality lean protein, minerals and vitamins. Omega-3 fatty acids have been identified as important for development of the young brain, and have been associated with reductions in chronic diseases. It is important to consider both the risk of contaminants and the benefits of fatty acids when choosing fish for consumption. Contaminants of concern are generally greater in older fish and Omega-3 fatty acids are highest in cold water species. One can gain the most benefit while minimizing the risk by consuming a variety of smaller cold-water fish and by following the appropriate consumption advisory.

Omega-3 fatty acids are polyunsaturated fatty acids (PUFA) with three nutritionally important fats: α-linolenic acid (ALA), eicosapentaenoic acid (EPA), and docosahexaenoic acid (DHA). Humans are unable to synthesize Omega-3 fatty acids in the body, but can obtain them through diet. ALA is generally found in plant oils, while EPA and DHA are commonly found in fish oils and seaweed and phytoplankton. Benefits associated with consuming Omega-3 fatty acids include improved cognitive ability and cardiovascular health. However, depending on concentrations present, the benefit of Omega-3 fatty acids through the consumption of Great Lakes fish may not outweigh the risk of exposure to toxic chemicals, such as mercury and PCBs. Research regarding the risk and benefit relationship of consuming fish is ongoing. Researchers are attempting to add to this body of knowledge through 1) generation of fatty acid data for Great Lakes fish species, currently a significant gap, 2) comparing those fatty acid levels to contaminant concentrations, and 3) ultimately incorporating into fish consumption advice (Ginsberg et al 2009, Ginsberg et al, 2015, Groth et al, 2017, Neff et. al. 2014a, Stern et al 2011, Turyk et. al 2012, Williams et. al 2014).

In more recent years, State and Provincial governments responsible for issuing consumption advice have shifted their attention towards both the risks and benefits of consuming Great Lakes fish when setting fish consumption advisories. Although some quantitative analysis based on mercury and fatty acids has been completed (Ginsberg 2016), at present, this is achieved qualitatively by assessing both the contaminant burden of fish and their levels of fatty acids. While more monitoring data are needed to understand the levels of fatty acids in Great Lakes fish, there is evidence that Great Lakes fish can be a good source of beneficial long chain polyunsaturated fatty acids. For example, recent assessments by U.S. EPA's Office of Science and Technology, and the Province of Ontario indicate that concentrations of EPA and DHA in common species from the Great Lakes increase with fish length (Figure 3). This is supported by a recent assessment of 13 Wisconsin sport fish which found that fish length was positively correlated with total fatty acid for all of the fish assessed but that the correlation was not positive for any individual species (Williams et. al. 2014). Additionally, the study showed that of the species assessed, salmonids generally contained the highest total fatty acids while percids and centrarchids contained the lowest concentrations, and that fish diet was a better predictor of fatty acid concentration than taxonomic family (Williams et al. 2014).

EPA and DHA contents are generally higher in fatty, large fish; however, these fish also typically contain greater levels of PCBs (Neff et al. 2014a). Limited data have indicated that EPA and DHA content in Lake Erie fish are comparable to some commercially-sourced fish and shellfish such as Yellowfin Tuna, shrimp, Pacific Cod, halibut, lobster and scallops (Neff et al. 2014a). Based on concurrent measurements of contaminants and fatty acids, it was concluded that consumption of certain Lake Erie fish within the limits of the fish consumption advisories can be a good supplemental source of PUFA (Neff et al. 2014a). Further, cooking generally has little effect on Omega-3 fatty acid content of fish (Neff et al. 2014b). As such, cooking fish on a grill to let fat and associated organic contaminants such as PCB drip away is a good approach to enhance benefits over risk of eating Great Lakes fish. More comprehensive fatty acid and contaminant data are needed to provide consumption advice that not only considers the risk of consuming Great Lakes fish, but also the benefits.

It is well recognized that fish consumption can present both benefits and risks. The overall risk-benefit of eating fish is affected by various factors. Although scientific studies have begun to evaluate the health benefits against the risks of eating contaminated fish, our understanding is still limited due to differences in the benefits of various nutrients and health risks from different contaminants. This makes it challenging to compare benefits and risks in every case. Because of the current limitations, the fish consumption advisories issued for the Great Lakes continue to be based only on contaminant risk. Future reports would continue to show the change in contaminant levels in fish and may also show the benefit of consuming Great Lakes fish resulting in a more comprehensive assessment of "fish-ability."

Linkages

Sources of chemical contaminants, and their cycling through the ecosystem, vary among the lakes. Therefore, it is important to have an understanding of how contaminants arrive to the Great Lakes and ultimately into fish species through diet, in addition to the presence of contaminants and their potential harm. This sub-indicator can easily be linked to most of the sub-indicators in the Toxic Chemicals indicator:

- Toxic Chemicals in Whole Fish
    - Levels of toxic chemicals found in the edible portions of fish can be related to their corresponding whole body levels. As such, the trends presented for whole body Lake Trout and Walleye can be compared to the trends presented in this indicator for the edible portion of those species.
- Toxic Chemicals in Herring Gull Eggs
    - Toxic chemicals found in fish eating and colonial nesting waterbirds can be closely linked with the levels of those chemicals in whole fish and edible portion of fish that may be consumed by the waterbirds.
- Toxic Chemicals in Water
    - Fish are exposed to toxic chemicals via diet and gill uptake. The levels of toxics in water can therefore be an indicator of a partial exposure to fish and can be linked with the levels found in the edible portion of fish.
- Toxic Chemicals in Sediment
    - The levels of toxic chemicals in fish can be linked with the corresponding levels in sediment as fish can acquire toxic chemicals from lower trophic level species such as benthos that are in contact with sediment through food web processes.

It should be noted that other external factors such as invasive species and climate change can affect fish contaminant trends regardless of overall declines in active pools of contaminants in the system. For example, invasions of mussels and Round Goby in Lake Erie appear to have increased transfer of contaminants historically deposited in the sediments to higher trophic level fish (e.g., Bhavsar et al. 2010, Hogan et al. 2007). Similarly, climate change can alter the amount of mercury depositing in the Great Lakes and accumulating in the food web (Krabbenhoft and Sunderland, 2013).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	No			

Data Limitations

Data for use in developing indicators exist; however there are differences in extent, frequency and methods of sample collection as well as analytical techniques for contaminant measurements within and among jurisdictions. Status and trend assessments were based on data generated by the Province of Ontario for Lake Superior, Lake Huron, Lake Erie and Lake Ontario. Lake Michigan status and trend assessments were generated based on data provided by the U.S. EPA and States of Michigan, Wisconsin, Illinois, and Indiana. Reliance upon a single data source, such as the Province, allowed for more consistent comparison between reporting cycles but may not reflect the status and/or trends at the State level.

Additional Information

There are differences in the way fish samples are analyzed for contaminants and fish consumption advisories are developed in the U.S. and Canada. This means that the data and fish consumption advisories cannot be directly compared between the two countries. For this sub-indicator, more consistent data generated by the Province of Ontario for fish from the Canadian waters of the Great Lakes (1 provincial agency versus 8 states) were mostly utilized for Lakes Ontario, Erie, Huron and Superior, while data generated by the U.S. agencies were utilized for Lake Michigan. Since large bodied fish considered in this assessment have large home ranges and likely move across the border, utilization of only Ontario data for Lakes Superior, Huron, Erie and Ontario should not be a major concern. A comparison of the recent contaminant levels to the corresponding advisory benchmarks has been provided by considering similarities in the benchmarks used by the agencies on both sides of the border.

An increased focus on emerging contaminants is occurring in monitoring programs in the U.S. and Canada. While U.S. EPA's Great Lakes National Program Office no longer collects or analyzes edible fish tissue samples, the Office has instituted an Emerging Contaminants Surveillance Program in whole fish that looks to identify the presence or absence of emerging contaminants of interest and will inform State monitoring and advisory programs.

The Ontario Ministry of the Environment, Conservation and Parks continues to monitor contaminants of long term concern such as PCB, dioxins/furans, mercury and organochlorine pesticides. During the last decade, the Province has started analyzing some contaminants of emerging concern for the Great Lakes environment such as PBDEs, PFASs and PCNs in selected fish samples.

It should be noted that the analysis presented in this sub-indicator report is cursory and a more in-depth data analysis of the monitoring data is recommended to draw a firm conclusion on contaminant trends. Monitoring data for the connecting channels of the Great Lakes were not considered as fish captured from the channels could be migratory and the data may not reflect the local conditions.

Acknowledgments

Authors

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Information Sources

Ontario MECP Guide to Eating Ontario Fish – www.ontario.ca/fishguide

State Monitoring & Advisory programs – http://www.health.state.mn.us/divs/eh/fish/consortium/members.html

Great Lakes Human Health Fish Tissue Study - http://www2.epa.gov/fish-tech/studies-fish-contamination

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List of Tables

Table 1. Consumption limits set by the Guide to Eating Ontario Sport Fish (based on Health Canada TDIs) and the Sport Fish Advisory Taskforce. Sensitive population includes women of childbearing age and children under 15.

Source: Ontario Ministry of the Environment, Conservation and Parks and Great Lakes Sport Fish Advisory Task Force (PCB Protocol 1993, Mercury Protocol 2007)

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Source: Ontario Ministry of the Environment, Conservation and Parks, U.S. Environmental Protection Agency, Indiana Department of Environmental Management, Illinois Environmental Protection Agency, Wisconsin Department of Natural Resources, Michigan Department of Environment, Great Lakes and Energy

Figure 2. Total mercury concentrations (µg/g) in five fish species from the Great Lakes observed between 1975 and 2019. Lake Michigan measurements were for skin-on fillets, while skin-removed fillets for the other lakes. Dashed green lines represent the cut-off criteria below which the levels would be considered “good” as per Table 2.

Source: Ontario Ministry of the Environment, Conservation and Parks, U.S. Environmental Protection Agency, Indiana Department of Environmental Management, Illinois Environmental Protection Agency, Wisconsin Department of Natural Resources, Michigan Department of Environment, Great Lakes and Energy

Figure 3. EPA + DHA (mg/g) compared to the length (cm) of common species from the Great Lakes. The measurements are for the samples collected between 2010 and 2017.

Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Environmental Protection Agency

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Table 1. Contaminant Concentrations and Fish Consumption Guidelines set by the Guide to Eating Ontario Sport Fish (based on Health Canada TDIs) and the Sport Fish Advisory Taskforce. Sensitive population includes women of childbearing age and children under 15. The green category represents 8+ meals per month advisories that can be considered “unrestrictive”, yellow represents 1 to 4 meals per month or “partial restriction”, and red represents 0 to half meal per month or “severe restriction”. Source: Ontario Ministry of the Environment, Conservation and Parks and Great Lakes Sport Fish Advisory Task Force (PCB Protocol 1993, Mercury Protocol 2007).

Meals per month	PCB (ug/g)			Hg (ug/g)		
	Sensitive*	Sensitive**	General**	Sensitive*	Sensitive**	General**
32		<.026	<.026		<0.06	<0.15
16	0 - .05	.026-.053	.026-.053	0 <=.05	0.06-0.12	0.15-0.3
12		.053-.070	.053-.070		0.12-0.16	0.3-0.4
8		.070-.105	.070-.105	>0.05 <= .11	0.16-0.25	0.4-0.6
4	.06 - .2	.105-.211	.105-.211	.0.11 <= .22	0.25-0.5	0.6-1.2
2			.211-.422			1.2-1.8
1	.21 - 1.0		.422-.844	>.22 <= 0.95		
0.5	1.1 - 1.9					
0 (do not eat)	>1.9	>.211	>.844	>0.95	>0.5	>1.8

*Sport Fish Advisory Consortium Protocol

**Ontario Ministry of the Environment and Climate Change

Table 2. Contaminant Concentration criteria used for the status assessments. The criteria were based on the consumption limits presented in Table 1 and are used to determine the status assessments of Good, Fair and Poor. The concentrations are in µg/g or parts per million (ppm).

Status	PCB (µg/g)	Hg (µg/g)
Good	<0.1	<0.25
Fair	0.1 - 1	0.25 - 1
Poor	>1	>1

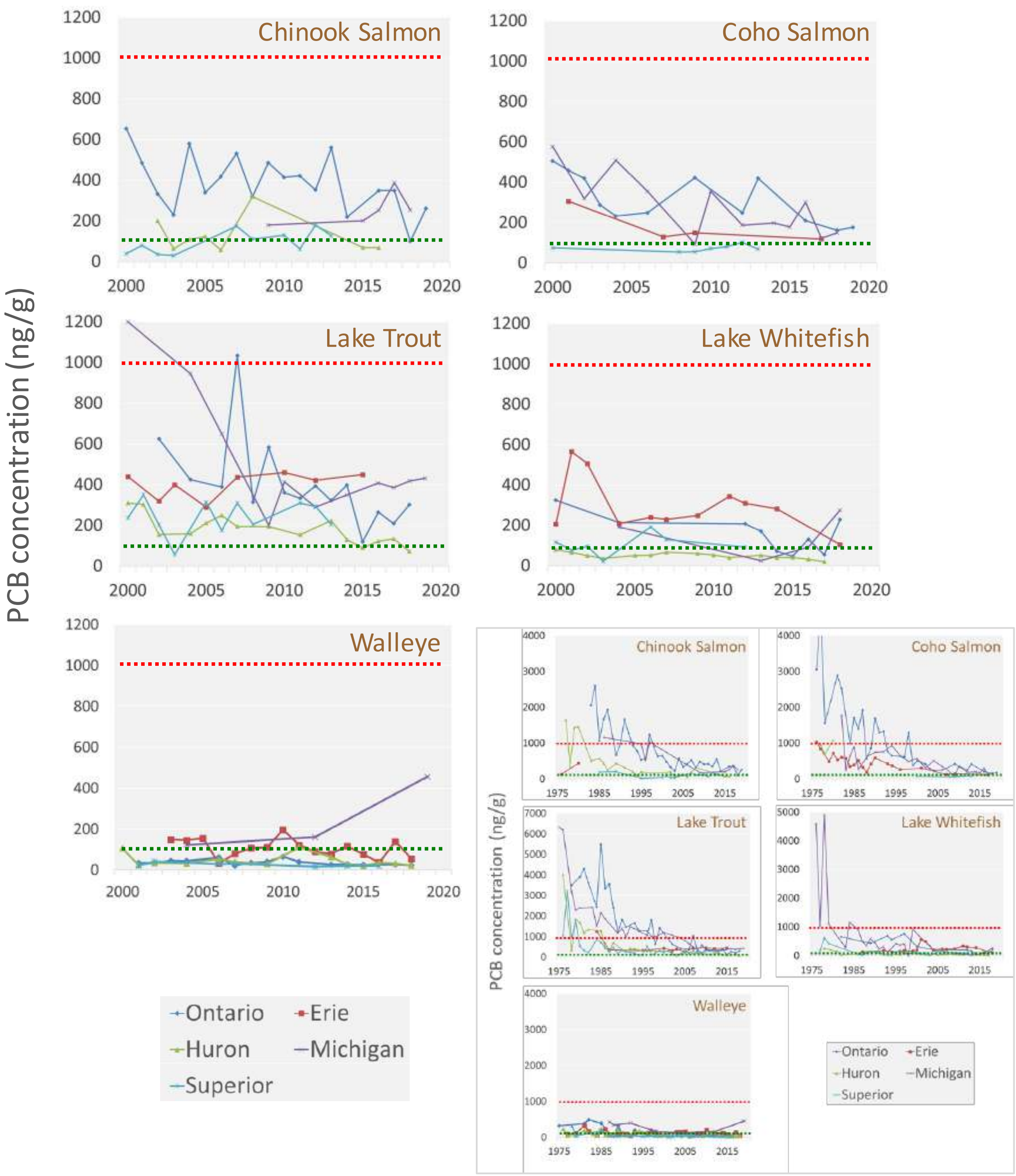

Figure 1. Total PCB concentrations (ng/g) in five fish species from the Great Lakes observed between 1975 and 2019 with focus on 2000 onwards. Lake Michigan measurements were for skin-on fillets, while skin-removed fillets for the other lakes. Conversion factor: 1000 ng/g = 1 µg/g or parts per million (ppm). Dashed red and green lines represent the cut-off criteria above/below which the levels would be considered “poor” and “good”, respectively, as per Table 2. Source: Ontario Ministry of the Environment, Conservation and Parks, U.S. Environmental Protection Agency, Indiana Department of Environmental Management, Illinois Environmental Protection Agency, Wisconsin Department of Natural Resources, Michigan Department of Environment, Great Lakes and Energy

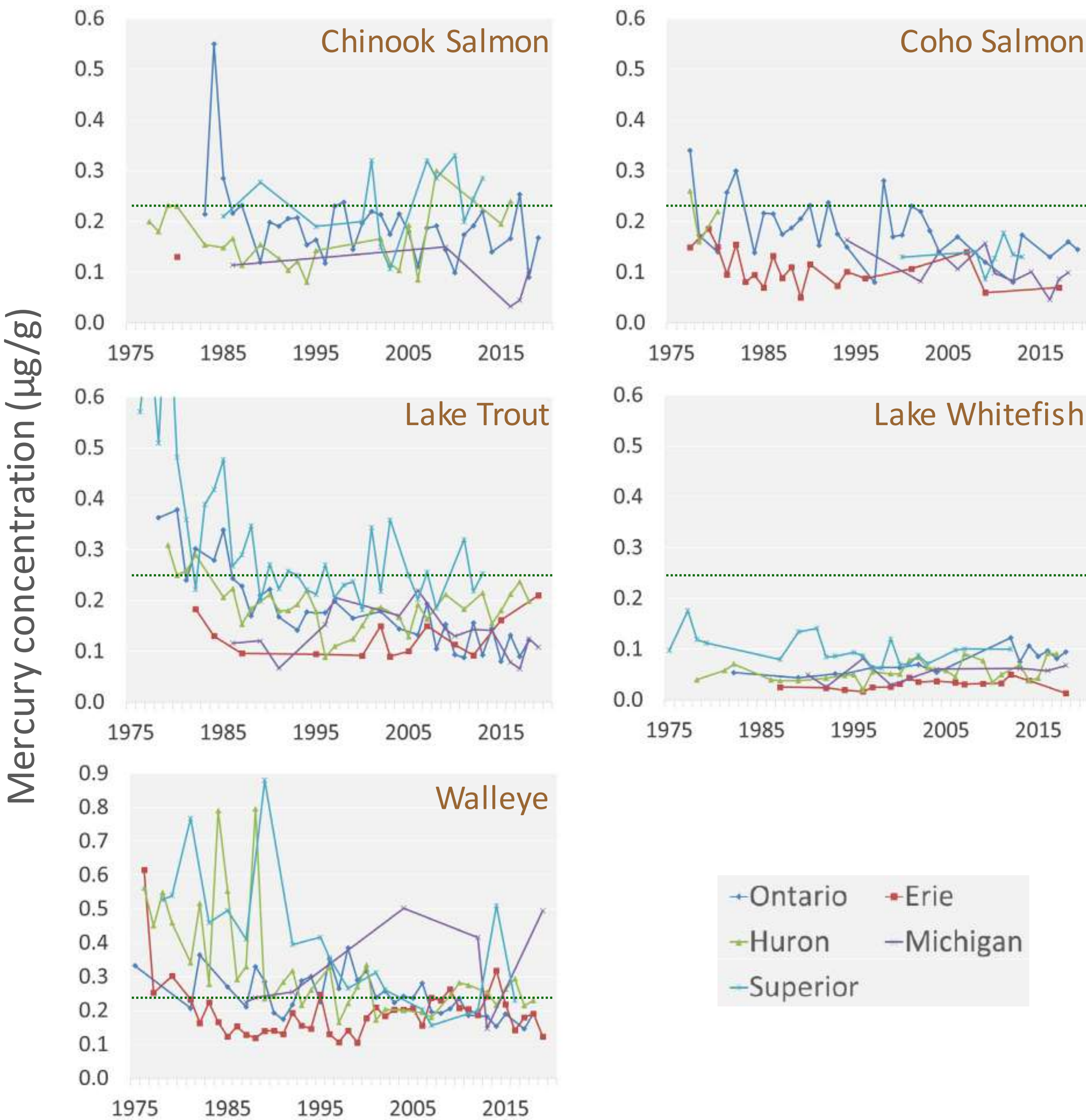

Figure 2. Total mercury concentrations (µg/g) in five fish species from the Great Lakes observed between 1975 and 2019. Lake Michigan measurements were for skin-on fillets, while skin-removed fillets for the other lakes. Dashed green lines represent the cut-off criteria below which the levels would be considered "good" as per Table 2. Source: Ontario Ministry of the Environment, Conservation and Parks, U.S. Environmental Protection Agency, Indiana Department of Environmental Management, Illinois Environmental Protection Agency, Wisconsin Department of Natural Resources, Michigan Department of Environment, Great Lakes and Energy.

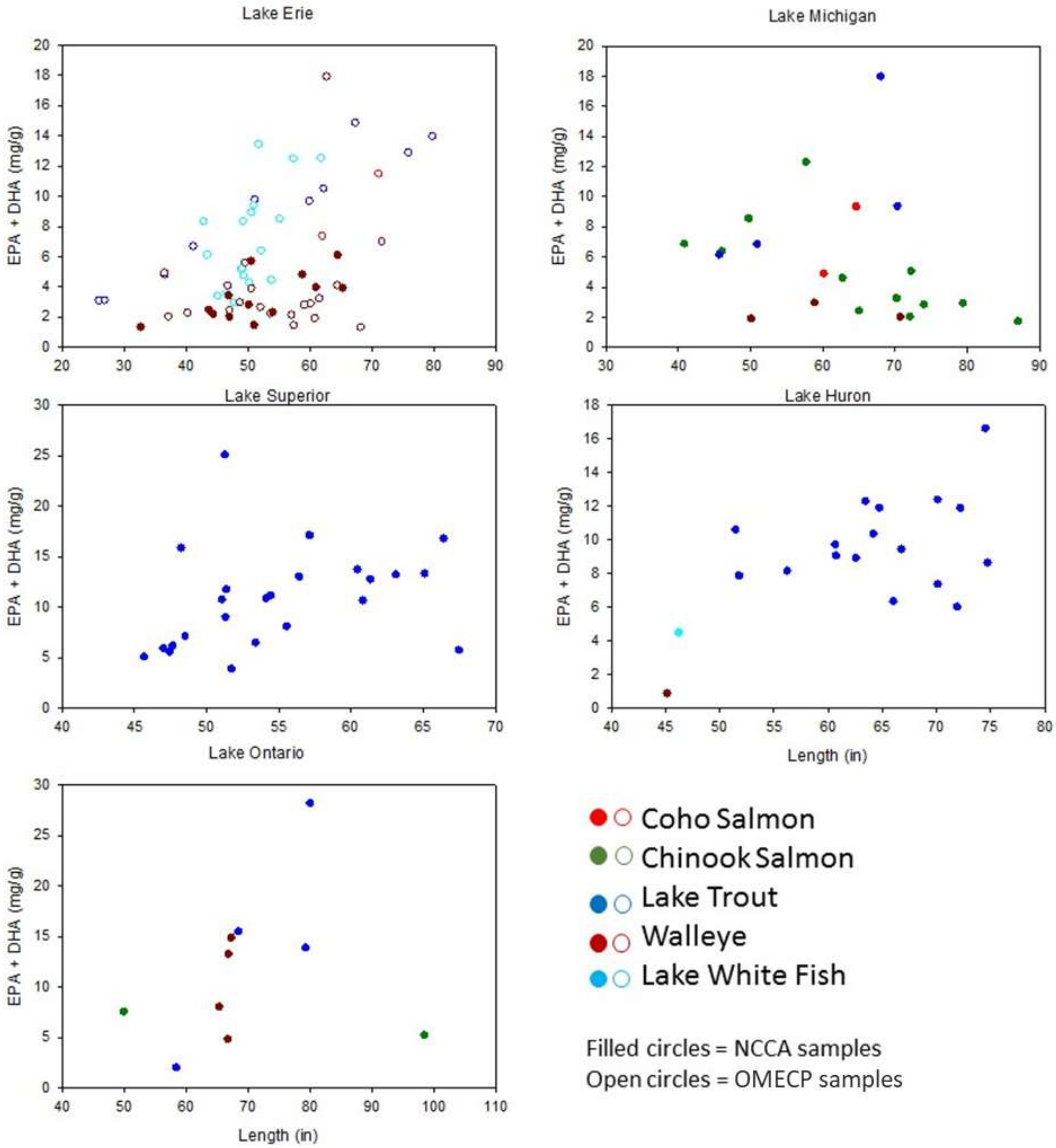

Figure 3. EPA + DHA (mg/g) compared to the length (cm) of common species from the Great Lakes. The measurements are for the samples collected between 2010 and 2017. Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Environmental Protection Agency.

Sub-Indicator: Toxic Chemicals in Sediment

Note: While the COVID-19 pandemic and subsequent lock-down has had impacts on the collection and dissemination of data used to assess toxic chemicals in Great Lakes sediment (described in detail below), the overall impact to the assessment of the Toxic Chemicals in Sediment sub-indicator is likely to be minimal. This sub-indicator indicator is used to report long-term trends in sediment contaminants, rather than short-term change. Impacts of the pandemic are not likely to affect trend analysis over this timeframe. Many of the Chemicals of Mutual Concern (CMCs) have been designated by Annex 3 specifically because they are persistent in the environment and bio-accumulate in the food chain. Legacy contaminants such as PCBs also have first-order half-lives significantly longer than the timeframe at which this indicator is applied (Li et al. 2009, Table 3). Given these factors, the assessments presented in the 2019 Toxic Chemicals in Sediment sub-indicator report are still valid indicators of Great Lakes status in 2021.

The global pandemic related to the emergence of the COVID-19 virus has had significant impacts on both the collection of data throughout the Great Lakes, as well as the laboratory processing of samples collected prior to pandemic lockdowns. Environment and Climate Change Canada (ECCC) sampling which normally would have taken place during the open-water seasons of 2020 and 2021 were completely suspended, which will lead to data gaps in the coming years. However, short-term trends are not the focus of this sub-indicator, and sediment in general is relatively slow in responding to changes in contaminant loadings. ECCC sediment sampling occurred in 2019 for the Cooperative Science and Monitoring Initiative (CSMI) cycle, but was not conducted in Lake Superior in 2021. It is hoped that ECCC CSMI sampling schedule for the other Great Lakes will continue as planned, as pandemic impacts are reduced. While planned Great Lakes Sediment Surveillance Program (GLSSP) sampling by the Environmental Protection Agency (EPA) and the United States Geological Survey (USGS) for CSMI in Lake Michigan was suspended in 2020, GLSSP sampling did occur in Lake Superior in 2021 and is planned for the 2022-2025 CSMI field years.

Pandemic lock-downs also had an impact on laboratory analysis of sediment samples collected in 2019 by ECCC. Samples taken in Lake Ontario were submitted to laboratories for analysis in the winter of 2020, but had not been analyzed before laboratories were closed. While ECCC laboratories are beginning to re-open, several factors will delay or reduce the amount of analysis which can be carried out. Specifically, capacity issues as laboratory work on backlogged samples, as well as new samples being collected, may slow the delivery of data into the following year.

The flow of data post-analysis was also significantly impacted during the pandemic. While data from samples analyzed prior to the lock-down were delivered to ECCC scientists, that data could not be released to the government Open Data portal. During the pandemic, the transfer of data was halted to permit software upgrades and re-development of the ECCC data catalog. This has resulted in a back-log of data for release, which is still awaiting the final release of the upgraded catalog, and the subsequent transfer to the government Open Data portal. Delays in data release should be eliminated in the near future.

Overall Assessment

Status: Fair

Trends:

10-Year Comparison: Unchanging

Long-term Trend (1970-2015): Improving

Rationale: In general, legacy contaminants that are persistent, bioaccumulative and/or toxic have decreased in Great Lakes sediment. Long-term trends for many legacy contaminants including mercury are exhibiting declines or no change. Legacy compounds including PCBs and DDT are generally below Canadian Council of Ministers of the Environment (CCME) sediment quality guideline values, while other contaminants including mercury, arsenic, polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs) and polybrominated diphenyl ethers (PBDEs) exhibit some exceedances of guidelines, particularly in Lake Ontario. Emerging and new contaminants are of interest as many exhibit trends toward increasing concentrations and need to be studied further to determine acceptable limits.

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Comparison: Unchanging

Long-term Trend (1970-2015): Improving

Rationale: Lake Superior is the largest, coldest and deepest of the Great Lakes. Physical processes such as longer contaminant cycling times and lower rates of volatilization have resulted in lower rates of decrease in concentrations for some legacy contaminants, compared to the other Great Lakes. However, typical offshore deep-water sediment contaminant concentrations are very low, with atmospheric deposition as the primary source. While still exhibiting the highest toxaphene concentrations in the Great Lakes, levels have declined by an order of magnitude since their peak in the 1980s. Concentrations of copper and lead often exceed the strictest sediment quality guidelines due to the geochemistry of the watershed (pre-Cambrian shield) while historical regional sources associated with mining and smelting are the likely cause of arsenic guideline exceedances. While the concentration of some of the brominated flame retardants (BFRs) including BDE 209, Dechlorane 604 and decabromodiphenylethane (DBDPE) are the lowest in the Great Lakes, they are increasing in concentration with doubling times of 7-24 years, 5-38 years and 5-16 years, respectively (Guo 2015), as a result of the same physical processes influencing concentrations of legacy contaminants.

Lake Michigan

Status: Fair

10-Year Comparison: Unchanging

Long-term Trend (1970-2015): Improving

Rationale: Lake Michigan consists of a cold, deep and forested northern basin, and a more urbanized southern basin. Atmospheric deposition is a primary source of most contaminants in sediments due to the lake's large surface area; however, inputs from tributaries and other local sources are also important (Lepak et al. 2015, Zhang et al. 2009, Eisenreich and Strachan 1992). Some chemicals exhibit elevated concentrations in sediment, in areas such as Green Bay, at sites on the eastern shores of the lake, and/or in the southern basin. Mercury concentrations are highest in Green Bay with higher contributions from industrial and watershed-derived sources (Lepak et al. 2015). Concentrations of some flame retardants are highest in Lake Michigan, compared to the upper Great Lakes (lower

Great Lakes not assessed), with the highest levels in the southeast portion of the lake and near Sleeping Bear Dunes (Guo 2015). PCBs concentrations are declining – albeit very slowly. In Lake Michigan, halving times in sediments are between 32 and 179 years (Li et al. 2009). PFCs that have replaced the more well-known PFOS and PFOA are now being found at comparable levels to PFOS and PFOA in Lake Michigan sediments (Codling et al. 2014).

Lake Huron (including St. Marys River)

Status: Good

10-Year Comparison: Unchanging

Long-term Trend (1970-2015): Improving

Rationale: Lake Huron is similar to Lake Superior from a sediment contamination viewpoint, as the lake is large, cold and deep with atmospheric deposition as the primary source of most contaminants. Typical sediment contaminant concentrations are very low; however PCDD/Fs, nickel and copper concentrations are above guidelines in areas of Spanish Harbour and the Whalesback Channel due to local historical industrial/mining activity. Arsenic concentrations are above guidelines across a third of the lake and may be increasing (Data source: ECCC). Very low sedimentation rates negatively impact natural recovery in the lake.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Fair

10-Year Comparison: Improving

Long-term Trend (1970-2015): Improving

Rationale: Lake Erie exhibits a definitive spatial gradient in contamination with decreasing concentrations from the western basin to the eastern basin, and from the southern area to the northern area of the central basin. This spatial distribution in Lake Erie is influenced by industrial activities in the watersheds of major tributaries, including the Detroit and St. Clair Rivers which, along with the Maumee River, hydrodynamically impact the southern shoreline. Sediment quality in the eastern basin continues to be classified as excellent. Lake wide decreases in sediment for legacy contaminants are impressive with declines of greater than 50% for mercury, PCBs, hexachlorobenzene (HCB), DDT and lead (Table 1). Government initiatives and remedial actions have effectively diminished point sources across the Great Lakes basin. Lake Erie has the highest sedimentation rate of the Great Lakes, and as a result exhibits the largest declines in bottom sediment legacy contaminant concentrations. Mean trace metal concentrations are no longer above the CCME federal threshold effects level (TEL) for all three basins of the lake; however, some exceedances in the probable effects level (PEL) still occur in the western basin (Table 2).

Lake Ontario (including Niagara River)

Status: Fair

10-Year Comparison: Improving

Long-term Trend (1970-2015): Improving

Rationale: Lake Ontario continues to exhibit the poorest sediment quality of all the Great Lakes. The greatest frequency and magnitude of exceedances of the CCME sediment quality guidelines is for PCDD/Fs and mercury (Table 2). This legacy contamination issue is the result of historical industrial activities in the Niagara River watershed and the influence of sources in the upstream lakes resulting in contaminants ultimately accumulating in the sedimentary record of Lake Ontario. However, current levels of PCDD/F contamination represent a 53% decline from peak levels in the 1970s. Mercury continues to have PEL exceedances in offshore depositional areas, however, declines of as much as 90% are evident lake wide. Trends in most legacy chemicals in Lake Ontario point toward improvements in sediment quality over time. While most BFR concentrations are low, dechlorane plus, also a result of historical industrial activity in the Niagara River watershed, is several orders of magnitude higher in Lake Ontario, compared to the other Great Lakes. Concentrations of bisphenol-A (BPA) are 5-10 times higher in Hamilton and Toronto Harbours, relative to the open lake sediments, indicating urbanized areas as primary sources and wastewater treatment plants as primary vectors (Data source: ECCC).

Status Assessment Definitions

Good: The metrics show that toxic chemical concentrations are meeting the ecosystem objectives or they are otherwise in an acceptable condition.

Fair: The metrics show that toxic chemical concentrations are not meeting the ecosystem objectives, but they are exhibiting minimally acceptable conditions.

Poor: The metrics show that toxic chemical concentrations are not displaying minimally acceptable conditions and are severely impacted.

Trend Assessment Definitions

Improving: Decrease in concentration or frequency of detection of toxic chemicals.

Unchanging: No change in the concentration or frequency of detection of toxic chemicals.

Deteriorating: Increased concentration or frequency of detection of toxic chemicals.

Undetermined: Data are not available or are insufficient to assess the trends or frequency of detection at this time, or the different groups of toxic chemicals are not trending in the same direction and an expert opinion of the overall direction of the trend cannot be agreed

Endpoints and/or Targets

The target or endpoint for this sub-indicator will have been met when the sediments of the Great Lakes are free from pollutants in quantities or concentrations that could be harmful to human health, wildlife or aquatic organisms, through direct exposure or indirect exposure through the food chain.

Status of surficial sediment will be determined by comparison with sediment quality criteria (e.g., CCME Probable Effect Level), or on a case-by-case basis when no guidelines exist. A weight-of-evidence approach will be taken in assessments, using factors including number of chemicals detected, comparison with sediment quality guidelines, or relative toxicity, if known. Status of temporal trends will be assessed using concentration profiles in sediment cores. Progress will be based on whether trends in chemicals are increasing or decreasing, the rate of change in concentrations, and the number of chemicals exhibiting changing trends.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the concentrations of toxic chemicals in sediments throughout the Great Lakes; to infer the potential for impairment to the quality of sediment of the Great Lakes by harmful pollutants; to infer progress toward virtual elimination of chemicals of mutual concern; to inform the risk assessment of toxic chemicals and the development of risk management strategies; to inform the development of environmental quality guidelines; and to report on environmental response (i.e., progress) toward the achievement of targets identified in action plans and risk management strategies for toxic chemicals in the Great Lakes basin.

Ecosystem Objective

This sub-indicator best supports work towards General Objective #4 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the Waters of the Great Lakes should "be free from pollutants in quantities or concentrations that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain."

Measure

The purpose of this sub-indicator is to assess the temporal trends and spatial distributions of toxic chemicals in sediments from the five Great Lakes. Each Great Lake will have a selection of chemicals assessed representing several chemical classes. The chemicals that will be assessed may include hexachlorobenzene (HCB), polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), dioxins, lead and mercury as well as PBDEs and non-brominated flame retardants and chlorinated paraffins. The sub-indicator report will include results of monitoring and surveillance activities for toxic chemicals of current and future concern. The monitoring data will be used to inform the selection of chemicals of mutual concern for Annex 3 of the GLWQA as well as monitor to assess the progress and effectiveness of pollution prevention and control measures for those compounds.

As a sub-indicator of long-term *temporal trends* the concentrations of toxic chemicals in sediment cores at selected sites within the Great Lakes will be measured. Sampling for each lake will follow the CSMI schedule. The chemicals of interest include chemicals of current and future concern which may be harmful to the Great Lakes ecosystem.

The sediment concentrations would be depicted using the standard tables and figures showing the change in concentration at different depths. Temporal trends may also be depicted using estimated fluxes to sediments for each core section.

Short-term trends (10 years or less) can not be measured in the Great Lakes using this sub-indicator. Short term (10 years or less) changes will be determined by comparing surficial sediment concentration measured over a 10-year period, or where this data does not exist, a comparison of the top two (1 cm) slices of a sediment core.

Assessment of temporal trends and concentrations of contaminants in sediment are influenced by a number of factors, including physical processes (e.g., current regime, sedimentation rate) and physical/chemical properties of chemicals (e.g., solubility, persistence). These same factors result in sediment having an overall slow response to changes in contaminant loadings; as a result, changes in concentrations may not be realized for a number of years.

As a sub-indicator of occurrence and *spatial distribution*, the concentrations of toxic chemicals in surficial sediments will be measured. Sampling will usually follow the CSMI schedule. Sampling locations will include not only the depositional zones of the lakes, but also near shore locations. Surficial sediments may either represent the top three centimeters in Lakes Michigan, Huron, Erie and Ontario and Lake Superior; or a homogenized sample collected with a ponar.

The number of stations sampled varies by lake, based on historical stations, sedimentology and bathymetry. A set of maps showing locations and concentrations would serve to illustrate the sub-indicator.

Because basin-wide information on bioaccumulative contaminants is not collected every year, the sub-indicator report will be a partial update every second State of the Great Lakes reporting cycle, i.e. every 6 years.

Ecological Condition

Status of Contaminants in Sediment

Sediments in the Great Lakes generally represent a primary sink for contaminants, but can also act as a source through resuspension and subsequent redistribution. Burial in sediments also represents a primary mechanism by which contaminants are sequestered and prevented from re-entering the water column. A new Environment and Climate Change Canada initiative (2014) which samples Great Lake sediment according to the CSMI schedule will provide a more extensive (spatially and temporally) assessment for both the connecting channels and the Great Lakes for future State of the Great Lakes reports.

Comparisons of surficial sediment contaminant concentrations with sub-surface maximum indicate that contaminant concentrations have generally decreased by more than 50 percent, and, in some cases, by as much as 90 percent in the lower Great Lakes over the past four decades (Table 1).

Sediment concentrations can also be assessed against guideline values established for the protection of aquatic biota, e.g., Canadian Sediment Quality Guidelines Probable Effect Level (PEL, CCME 1999). These guidelines can be applied as a screening tool in the assessment of potential risk, and for the determination of relative sediment quality concerns.

Status of Chemicals of Mutual Concern (CMCs)

The Parties to the GLWQA designated the first set of Chemicals of Mutual Concern (CMCs) in May 2016. For State of the Great Lakes reporting in 2019 and moving forward, the toxic chemical sub-indicators, where possible, will support reporting on CMCs in a more fulsome, consistent and transparent way. Information on additional chemicals of interest for the Great Lakes (non-CMCs) is valuable for inclusion in the report and will be included in a separate section below, as appropriate. For the 2019 Toxic Chemicals in Sediment sub-indicator report, the assessment is not based solely on CMCs.

The following chemicals were identified as the first set of CMCs:

- Hexabromocyclododecane (HBCD)
- Long-Chain Perfluorinated carboxylic acids (LC-PFCAs)
- Mercury (Hg)
- Perfluorooctanoic acid (PFOA)
- Perfluorooctane sulfonate (PFOS)
- Polybrominated Diphenyl Ethers (PBDEs)
- Polychlorinated Biphenyls (PCBs)

- Short-Chain Chlorinated Paraffins (SCCPs)

Mercury and Metals

The spatial distribution of mercury contamination in Great Lakes sediments generally represents those of other legacy toxic compounds, including other metals and organics such as PCBs, as accumulation of a broad range of contaminants on a lake-by-lake basis can be the result of common sources. The highest concentrations of mercury in sediments of Lakes Michigan, Erie and Ontario are observed in offshore depositional areas characterized by fine-grained sediments. Contaminant concentrations are generally correlated with particle size; hence the distribution of mercury is not only a function of loadings and proximity to sources, but of substrate type and bathymetry. Current research by ECCC confirms earlier work by Marvin et al. 2004 that mercury contamination is generally quite low in Lakes Huron, Michigan, Superior, and more recently Lake St. Clair, with higher concentrations in Lake Ontario and the western basin of Lake Erie (Figure 1). Levels of mercury in the Great Lakes indicate that most of Lake Ontario, western Lake Erie and areas in the north of Lake Superior still exceed the PEL (Figure 1). There is a gradient in contamination in Lake Erie toward decreasing concentrations from the western basin (mean 410 ng/g) to the central basin (190 ng/g) to the eastern basin (62 ng/g) (Data source: ECCC). The spatial distribution in Lake Erie is influenced by industrial activities in the watersheds of major tributaries, including the Detroit River, and areas along the southern shoreline (Marvin et al. 2004). Sources and loadings of mercury to Lake Huron appear to have been reduced to the point that no apparent spatial pattern exists. Current sediment contamination is substantially lower than peak levels that occurred in the mid – 1950s through the early 1970s for all of the lakes with concomitant reductions in the connecting channels including the Niagara, lower Detroit and upper St. Clair Rivers, all of which are associated with historical mercury contamination as a result of chlor-alkali production. These areas were also intensively industrialized and were primary sources of a variety of persistent toxics to the open lakes, including PCBs. Recent studies conducted from 2012 through 2014 (Lepak et al. 2015) and 2013-2018 (Figure 1; Data source: ECCC) are consistent with earlier studies showing:

- a wide total mercury concentration range across Great Lakes sediments;
- lowest total mercury concentrations observed offshore in Lakes Huron and Superior and higher concentrations in western Lake Erie and in Lake Ontario; and
- regional increases in mercury concentration relative to those offshore in Lake Michigan (Green Bay) and Lake Superior sediment (Thunder Bay).

For metals, PEL guideline exceedances were frequent in Lake Ontario for lead, cadmium, zinc and mercury. Guideline exceedances (PEL) were infrequent in all of the other lakes, with the exception of lead in Lake Michigan where the PEL (91.3 μg/g) was exceeded at over half of the sites, and arsenic in Lakes Huron and Ontario (Table 2).

PCBs

PCB results from Li et al. (2009), conducted during a similar time period to the study by Burniston et al. (2011), found a 30% reduction in PCB concentration across the Great Lakes, compared to results from (Eisenreich (1987), with the greatest decrease occurring in Lake Ontario. The comparison of PCB totals to historical studies is confounded by changes in analytical methodology. Comparing surficial sediment with subsurface maxima using similar analytical techniques may provide more reliable results. Reductions for PCBs across the Great Lakes, comparing lakewide average of surficial sediment with sub surface maxima, ranged from 5% in Lake Michigan to 94% in Lake Ontario. For PCBs, while decreased production contributes to this reduction, the decreased concentrations may also be the result of the loss of light congeners due to repeated resuspension of surficial sediment, desorption of light congeners and subsequent evaporation (in Lake Michigan; (Li et al. 2009)) or by anaerobic reductive dechlorination (in Lake Ontario; (Li et al. 2009)). Because of differences in toxicity between congeners the latter process could reduce the toxicity of the PCBs (Li et al. 2009). First order half-lives ($t_{1/2}$) vary from 44.9 years (Lake Huron) to 9.7 years (Lake Superior), see Table 3, with shorter half-lives found at sites (Ontario, Erie, Superior) closer to tributary sources and thus more responsive to PCB source reductions (Li et al. 2009). While PEL (277 ng/g total PCBs) guideline exceedances for PCBs are currently rare in any of the Great Lakes sediments, levels still remain elevated in areas of the St. Clair-Detroit River Ecosystem.

Flame Retardants

Flame retardants (FRs) are heavily used globally in the manufacturing of a wide range of consumer products and building materials. FRs have been found to be bioaccumulating in Great Lakes fish and in breast milk of North American women (Schecter et. al. 2003). While industrial discharges may not be responsible for ongoing contamination, modern urban/industrial centers can act as diffuse sources. Studies of sediment core profiles of PBDEs in Lake Ontario suggest accumulation of these chemicals recently peaked, or continues to increase (Marvin et al. 2007; Shen et al. 2010). The Lake Ontario total PBDE profile indicates a leveling off of accumulation in the past decade, presumably as a result of voluntary cessation of production of these compounds in North America. Recent core studies (Data source: ECCC) indicate total PBDE surface maximums (lakewide average 3.4 ng/g) in Lake Superior, while in Lake Huron levels have started to decline (Figure 2). Surficial sediment concentrations in Lake Huron are low and range from 0.1-14 ng/g and <0.1-12 ng/g for total PBDEs and BDE 209, respectively. While other contemporary studies (Guo, 2015; Zhu and Hites, 2005) have shown total PBDEs, and in particular the deca-substituted BDE 209, are continuing to increase across all five Great Lakes with doubling times ranging from 4 years to 74 years. BDE209 was produced in the U.S. as late as 2014, but still remains in many products and is the predominant congener in sediment, accounting for over 90% of total PBDEs. This is of concern because BDE209 can degrade in biota and sediment to less substituted BDEs that are more toxic and more bioaccumulative (Gauthier et al, 2008). A study of the upper lakes by Guo (2015) found the highest surficial concentrations for both total PBDE and BDE209 concentrations were in Lake Michigan (especially southeast and Sleeping Bear Dunes), and Lake Huron (especially Saginaw Bay and North Channel), and were comparable to Lake Erie concentrations, but lower than Lake Ontario.

Other FRs such as dechlorane plus (*anti* and *syn*) and related compounds Dec604 Dec602 are found at low levels throughout the upper Great Lakes, but are more elevated in Lake Erie and an order of magnitude higher in Lake Ontario (Figure 3.). Most FRs increased significantly after 1920 and have leveled off or decreased since 2000, but Dec 604 and DBDPE may still be increasing in Lake Ontario. For the upper Great Lakes, PBDEs and 1,2-Bis(2,4,6-tribromophenoxy) ethane (BTBPE) dominate in both southern and northern Lake Michigan, especially the southeast portion of the lake and the sites near Sleeping Bear Dune. Despite these trends, maximum concentrations of many FRs remain well below maximum concentrations of contaminants such as DDT and PCBs observed historically. In the lower Great Lakes, levels have shown a leveling off in recent years. A recent ECCC study detected few halogenated flame retardants in Lake Erie (n=10), including HBCD, even though detections of 2-Ethylhexyl-2,3,4,5-tetrabromobenzoate (EHTBB) and bis(2-ethyl-1-hexyl)tetrabromophthalate (BEHTBP) were frequent, albeit low, in the Detroit River (n=12).

Perfluoroalkyl Compounds

Perfluoroalkyl Compounds (PFCs) have attracted scientific and regulatory interest in recent years as a result of their detection globally in humans and wildlife. They are routinely detected in precipitation and air in urban and rural environments. These compounds have a myriad of applications, but have been primarily used as soil and liquid repellents for papers, textiles and carpeting. Two classes of PFCs, the perfluoroalkyl sulfonate acids (PFSAs), particularly perfluorooctane sulfonate (PFOS), and the perfluorocarboxylates (PFCAs), particulary perfluorooctanoic acid (PFOA), are the most commonly measured PFCs in sediment; these compounds are highly stable and persistent in the environment, and are potentially toxic. In surficial sediments, concentrations of perfluorobutane sulfonate (PFBS) and perfluoro-n-butanoic acid (PFBA) are now occurring at concentrations comparable to those of the PFCs which they replaced (PFOS and PFOA) (Codling et al. 2014). PFCs have been detected in environmental samples in remote areas such as the Canadian Arctic. The physical and chemical properties of PFCs are different from many other semi-volatile pollutants as they have both hydrophilic and hydrophobic properties. While persistent and bioaccumulative, PFCs are transported in both the aqueous and non-aqueous phase. As well, PFCs in bottom sediment may diffuse to the surface and become bioavailable.

There is a gradient toward increasing PFC contamination from the upper Great Lakes (Superior and Huron) to the lower Great Lakes (Erie and Ontario). This spatial trend is reflected in the concentrations of PFOS in surficial sediment, which range from 0.24 – 6.26 ng/g and 0.19 – 5.29 ng/g for Lakes Superior and Huron, respectively, to 0.66 – 15.3 ng/g and 0.65 – 46.0 ng/g for Lakes Erie and Ontario, respectively (Data Source, ECCC). When concentrations of PFCs in Lake St. Clair were compared with sediments (Marvin et al., 2004) and fishes (McGoldrick and Murphy, 2016) in other Great Lakes, concentrations of PFCs in Lakes Ontario and Erie were greater. Concentrations of PFCs in sediments are influenced not only by proximity to sources, but physical processes and bathymetry. The watersheds of Superior and Huron are less densely populated, and PFC concentrations are typically less. Most locations have concentrations that indicate non-point source contamination; however, concentrations of PFCs in sediments at a few sites are greater, which indicates influences from more local sources. In Lake Superior, the deeper sediment typically contained lesser concentrations of PFCs. Given the longer residence time of water in Lake Superior, there might be some breakdown or uptake of these compounds before reaching the sediment layer. Uses of PFBA and PFBS as replacements will likely result in greater releases to the environment.

Concentrations of surface sediment are dominated by PFBA and perfluoro-n-hexanoic acid (PFHxA), indicating these compounds might not be bound to the solid phase of sediments. Concentrations of PFBA in core samples were greater in deeper sediment than would have been predicted based on how much was being manufactured and used. Concentrations indicate contamination may enter the food chain (Codling et al, 2018 b). In general, PFC concentrations have increased over time, which largely corresponds to increased use. However, PFCs that have been subjected to targeted action have seen declines in concentrations in sediment. For example, concentrations of PFOS in Lake Ontario peaked in 2001 (Myers et al., 2012). PFBA and PFHxA were frequently determined in surface sediment and upper core samples indicating a shift in use patterns. Distributions of PFCs within dated cores largely corresponded with increase in use, but with physiochemical characteristics also affecting distribution. PFCs with chain lengths >7 that include perfluoro-n-octane sulfonate (PFOS) bind more strongly to sediment, which resulted in more accurate analyses of temporal trends. Shorter-chain PFCs, such as PFBA, which is the primary replacement for C8 compounds that were phased out, are more soluble and were identified in some core layers at depths corresponding to pre-production periods. Thus, analyses of temporal trends of these more soluble compounds in cores of sediments are less accurate. Based upon their physical-chemical properties, sediment might not be the best medium for monitoring (Codling et al, 2018 a).

Linkages

Sediment contamination affects both water quality and aquatic dependent life. Sediment can be a source of mercury and other toxic chemicals to enter the water column. These chemicals are components of the Toxic Chemicals and the Habitat and Species indicators including “Toxic Chemicals in Water” and “Toxic Chemicals in the Atmosphere”.

Linkages to other sub-indicators in the indicator suite include:

- Benthos/*Diporeia* – sediments serve as habitat for many benthic species and communities. However, sediments also serve as a reservoir for bioaccumulation and trophic transfer of contaminants, resulting in potential effects to the bottom-dwelling communities as well as implications for fish and other species in the food web.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources.	X			
Geographic coverage and scale of data are appropriate to the Great Lakes basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: Great Lakes Sediment Monitoring and Surveillance Data - Open Government Portal (canada.ca) Great Lakes Sediment Archive Database (1960-1975) - Open Government Portal (canada.ca)		

Data Limitations

Sediment cores may only be obtained every decade or so, limiting the ability to provide updates on a three-year

cycle. However, the updates of the sub-indicator when new information arises are applicable to past years (i.e., sediment cores will fill in the history for the previous decade). The thinnest increments in sediment cores from lakes with low sedimentation rates (e.g., Lake Superior) do not afford adequate temporal resolution in order to determine year-to-year changes in contaminant concentrations.

Additional Information

This sub-indicator will track whether concentrations of the toxic chemicals are, as a group, decreasing, staying the same, or increasing in Great Lake waters over time. The sub-indicator data will also demonstrate the magnitudes of the trends of the various chemicals. The magnitudes of the trends are expressed as halving time, or time to which the concentration of the chemical is decreased by a factor of two. The time which is most relevant to virtual elimination is the longest halving time of the measured chemicals.

Assessment of nearshore sediment quality will be done using surficial sediment collected every 5-10 years from sites either previously monitored for contaminants in sediment, water and/or fish, or determined by resource managers to be a high priority for surficial sediment information (e.g. tributary input). Sites would also be chosen based on sediment type, expected sedimentation rates, and proximity to potential sources. Cores would be sectioned, dated and analyzed for the toxic chemicals.

Efforts to control inputs of historical contaminants have resulted in decreasing contaminant concentrations in the Great Lakes open-water sediments for many of the legacy chemicals. However, chemicals such as FRs, current-use pesticides may represent emerging issues and potential future stressors to the ecosystem. These results corroborate observations made globally, which indicate that large urban centers act as diffuse sources of chemicals that are heavily used to support our modern societal lifestyle.

Long-term research and monitoring programs are valuable tools for demonstrating effectiveness of remedial actions and management initiatives, as well as acting as indicators of emerging issues. Enhanced Canadian Great Lakes studies now include the regular sampling of sediment to be collected following the CSMI schedule. The Great Lakes Sediment Surveillance Program is a complimentary program in the U.S. Comparison of contaminant results between studies and across lakes is currently difficult because of differences in sampling designs, sampling locations, and analytical procedures. Changes in contaminant deposition cannot be detected over time frames less than the temporal resolution of the surficial sediment samples, which can be from 3 to 220 years.

The selection of Chemicals of Mutual Concern (CMCs) has been completed by the Parties to the GLWQA. These identified CMCs and any additional CMCs added in the future, will be included in future reporting where possible in the sub-indicators included in the Toxic Chemicals Indicator category of the Great Lakes indicator suite.

Acknowledgments

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Table 1. Estimated percentage declines in sediment contamination in the Great Lakes (1970 – 2015) based on comparison of surface sediment concentrations with sub surface maximum concentration.

Source: Environment and Climate Change Canada; Li (2006); Marvin (2004)

Table 2. Occurrence of lead, arsenic and mercury in Great Lakes sediments in comparison to CCME guideline values. Sites with no sediment assigned “0” for all contaminants.

Data Source: Environment and Climate Change Canada

Table 3. First order half-life (t1/2) of PCBs in sediments of the Great Lakes

Source: Li et al. 2009

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Figure 2. Temporal trends of polybrominated diphenyl ethers in Lake Huron sediment.

Data Source: Environment and Climate Change Canada

Figure 3.Spatial distribution of BTBPE and dechlorane plus (sum of syn and anti) in Great Lake sediment (sampled in 2010-2014).

Source: Lakes Superior, Michigan and Huron-Guo (2015); St. Clair River, Lake St. Clair, Detroit R. and Lake Erie -

Environment and Climate Change Canada; Lake Ontario - Yang et al. 2011 and 2012

Figure 4. Total perfluorinated sulfonic acids (PFSAs) and Perfluorooctane sulfonic acid in Great lakes sediment.

Data Source: Environment and Climate Change Canada

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Table 1. Estimated percentage declines in sediment contamination in the Great Lakes (1970 – 2015) based on comparison of surface sediment concentrations with sub surface maximum concentration. Source: Environment and Climate Change Canada; Li (2006); Marvin (2004).

	Lake Superior	Lake Huron	Lake Michigan	Lake Erie	Lake Ontario
Mercury	4.5	11	49	60	94
PCBs	45	9	5	51	85
PCDD/Fs	NA	NA	NA	NA	53
HCB	NA	NA	NA	78	40
DDT	NA	93	NA	60	60
Lead	10	43	NA	71	65

Table 2. Occurrence of lead, arsenic and mercury in Great Lakes sediments in comparison to CCME guideline values. Sites with no sediment assigned "0" for all contaminants. Data Source: ECCC.

% of sites above guideline. Guideline values expressed as µg/g		**Lead**				**Arsenic**				**Mercury**			
	N	TEL (35)	PEL (91.3)	LEL (31)	SEL (250)	TEL (5.9)	PEL (17)	LEL (6)	SEL (33)	TEL (0.17)	PEL (0.486)	LEL (0.2)	SEL (2)
Lake Superior 2016	76	53	6.6	57	0	73.7	16	72	2.6	12	0	3.95	0
Lake Huron 2017	93	42	6.5	44	0	59	28	56	5.4	0	0	0	0
Lake Erie 2014	54	48	1.9	50	0	76	3.7	65	0	35	11.1	31	0
Lake Ontario 2018	42	95	45	100	0	100	71	100	4.8	86	57	86	0

Table 3. First order half-life (t1/2) of PCBs in sediments of the Great Lakes[a] Source: Li et al. 2009.

Lake	Location	Peak year	Half-life ($t_{1/2}$), Years
Superior	SU22	1993	9.7±7.9
Michigan	LM41[b]	1979	31.7±14.3
Huron	HU12[b]	1981	44.9±1.0
Erie	ER37	1981	16.6±2.2
Ontario	ON-30	1973	11.0±1.0
Ontario	ON-40	1963	17.0±4.4

[a] The first order $t_{1/2}$ values at other sampling locations cannot be obtained due to insufficient numbers (<3) of data points (SU08, SU12, SU16, HU38, HU48) or severe sediment mixing (ER09).[b] The top segment was excluded in $t_{1/2}$ calculation at these sites.

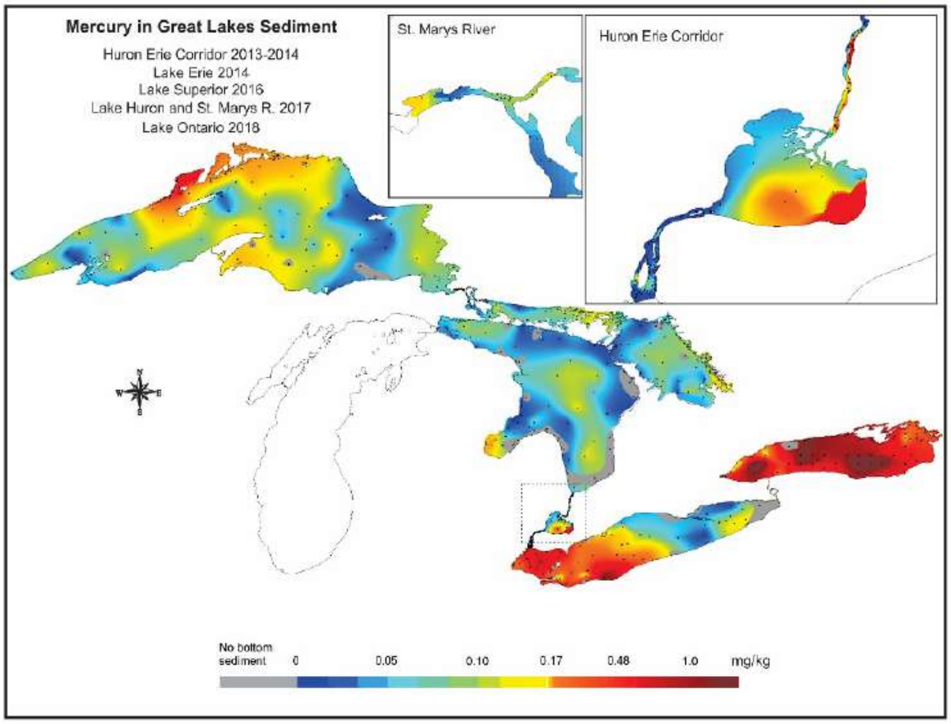

Figure 1. Spatial distribution of mercury contamination in surface sediments of open-lake areas and connecting channels of the Great Lakes. Source: Environment and Climate Change Canada.

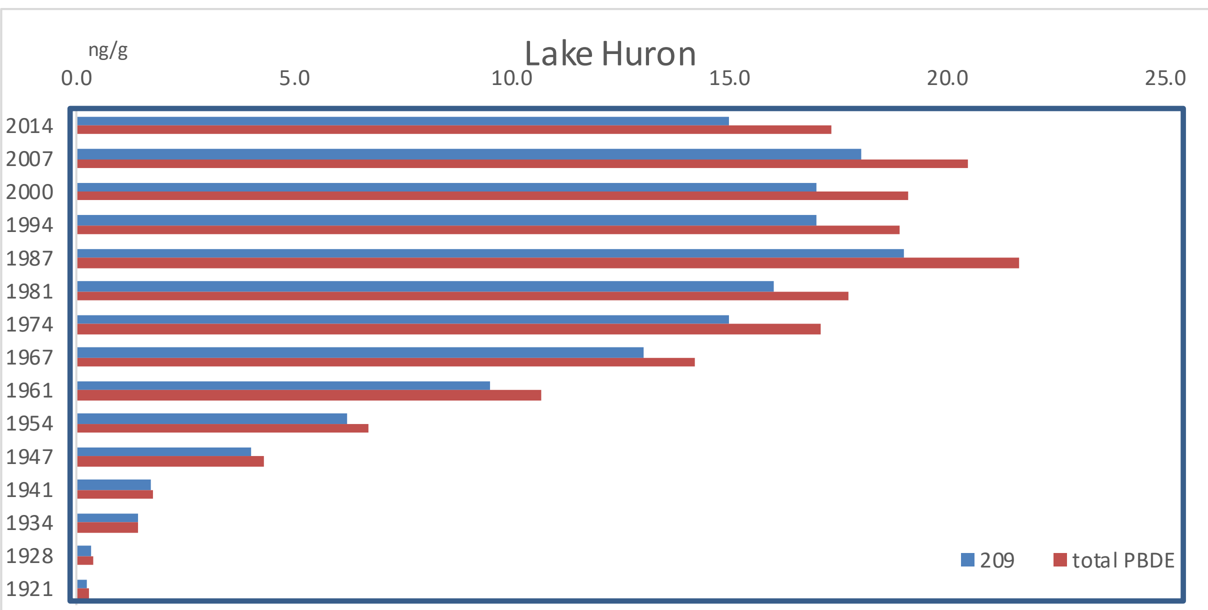

Figure 2. Temporal trends of polybrominated diphenyl ethers in Lake Huron sediment. Data Source: Environment and Climate Change Canada.

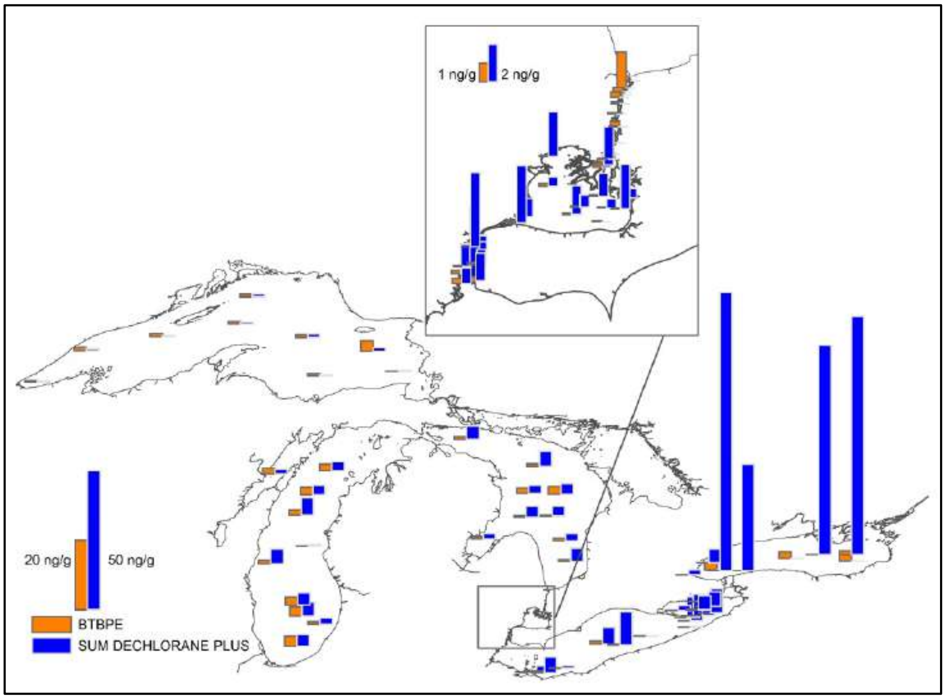

Figure 3. Spatial distribution of BTBPE and dechlorane plus (sum of syn and anti) in Great Lake sediment (sampled in 2010-2014). Source: Lakes Superior, Michigan and Huron-Guo (2015); St. Clair River, Lake St. Clair, Detroit R. and Lake Erie -Environment and Climate Change Canada; Lake Ontario -Yang et al. 2011 and 2012.

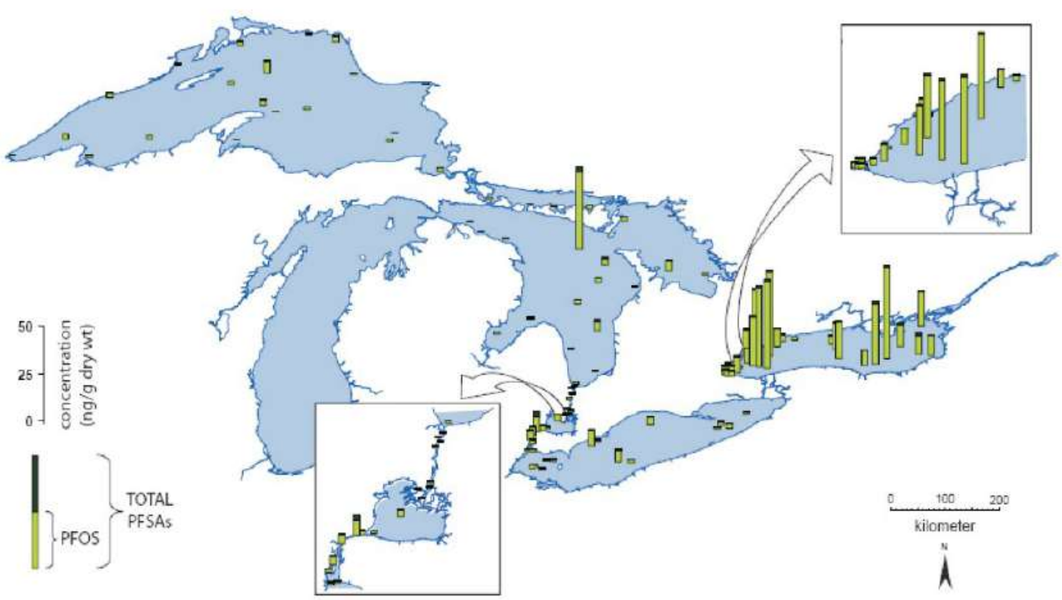

Figure 4. Total Perfluorinated sulfonic acids (PFSAs) and Perfluorooctane sulfonic acid in Great lakes sediment. Data Source: Environment and Climate Change Canada.

Sub-Indicator: Toxic Chemicals in Water

Note: This sub-indicator has not been updated since the previous (SOGL 2019) assessment. Field and laboratory issues, as well as the global COVID-19 pandemic, have had significant impacts on the collection of data in the Great Lakes. In 2018, Environment and Climate Change Canada's (ECCC) spring Lake Ontario water quality cruise was cancelled due to Canadian Coast Guard Ship (CCGS) LIMNOS staffing issues, and although some interagency studies took place with the United States Environmental Protection Agency (USEPA) and collaborators in that year, the vast majority of water quality samples for toxic contaminants could not be collected. ECCC conducted water quality sampling in Lake Erie in 2019 and this included some work for toxic chemicals, although unfortunately samples for some compounds such as PBDEs were lost at an external laboratory and those samples cannot be replaced. ECCC sampling during 2020 was completely suspended due to the pandemic. In the summer of 2021, sampling has resumed onboard the CCGS LIMNOS, and summer cruises on Lakes Erie and Ontario included some toxic chemicals such as currently used pesticides and perfluorinated compounds. We anticipate a return to collecting more routinely in the Great Lakes on a rotating basis starting in Lake Huron in spring 2022. Due primarily to the paucity of new data upon which to base an assessment, this sub-indicator cannot be updated at this time. The previous findings are considered relevant, but no new information has been collected at a scale supporting this basin-wide assessment. These data gaps will significantly impact our ability to assess and report on the trends of toxic chemicals in water in the subsequent report(s) as well.

Overall Assessment

Status: Fair

Trends

10-Year Trend/Long-term Trend (2004-2017)*: Unchanging

Rationale: Compounds considered in this report are the Chemicals of Mutual Concern (CMCs) or chemicals that could become CMCs due to their properties that make them persistent, bioaccumulative and/or toxic (PBT). The majority of legacy contaminants that are PBT have decreased over the long term in Great Lakes waters, with overall lower levels but little or no change in the more recent record. Occasional exceedances of water quality objectives are observed for total PCBs in Lakes Erie and Ontario. The number of compounds being monitored is increasing, thereby improving our base of knowledge. Trends are undetermined for certain CMCs (HBCDD, PBDEs, PFOS, PFOA) and although status is reported it is incomplete due to a lack of monitoring in all areas for all compounds. Status was previously assessed as Good. The change to Fair is based on recent detections of CMCs (noted above) in offshore waters.

* The long-term trend for the chemicals assessed in this report will be used for the 10-year trend determination.

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend/Long-term Trend (2005-2016)*: Improving

Rationale:

In general, the status of Lake Superior waters is fair, but it has the highest concentrations of certain compounds such as a-HCH, g-chlordane, lindane and toxaphene which accumulate in the cold, deep waters, and once present, are very slow to disappear due to the compounds' persistence and the long water residence time of the lake. Some compounds are showing declining concentrations over the long term but no change in recent years. Lake Superior has the lowest concentrations for a suite of new compounds, including perfluorinated surfactants and brominated flame retardants. Status was previously assessed as Good. The change to Fair is based on recent detections of CMCs in offshore waters; however, insufficient time has passed for an assessment of all CMC trends.

*The long-term trend for the chemicals assessed in this report will be used for the 10-year trend determination.

Lake Michigan

Status: Fair

10-Year Trend/Long-term Trend (2006-2015): Undetermined

Rationale: Fewer data are available for Lake Michigan; the available information indicates unchanging conditions for many compounds (declining for dieldrin) and no exceedances of available water quality guidelines are observed. Additional data for a suite of compounds will be made available by the U.S. EPA and included in future State of the Great Lakes reports. Trends cannot be determined because of the lack of data/information upon which to base the assessment.

Lake Huron

Status: Good

10-Year Trend/Long-term Trend (2004-2017)*: Unchanging

Rationale: Lake Huron has some of the lowest concentrations of many contaminants due to few sources and it is less subject to atmospheric deposition and retention of persistent compounds due to its geographical location. Some evidence of increasing PAHs is observed in Georgian Bay, although concentrations are low and no guidelines are exceeded at the monitored locations. Mercury and several important legacy organochlorines are showing declining trends over the long term.

*The long-term trend for the chemicals assessed in this report will be used for the 10-year trend determination.

Lake Erie

Status: Fair

10-Year Trend/Long-term Trend (2004-2014)*: Unchanging

Rationale: Lake Erie displays relatively high concentrations of certain legacy organochlorines and industrial byproducts due to its location downstream of historic sources. Some PAHs are also highest in Lake Erie. Current use pesticides are in general highest in Lake Erie and its tributaries but recent monitoring is lacking. Observed variability is highest in Lake Erie for most monitored parameters and few trends are discernible. A significant decline in total mercury is noted in the eastern basin only. In the most recent water quality surveys, no exceedances of available water quality guidelines were observed at the monitored locations.

*The long-term trend for the chemicals assessed in this report will be used for the 10-year trend determination.

Lake Ontario

Status: Fair

10-Year Trend/Long-term Trend (2004-2013)*: Unchanging

Rationale: Due to its position downstream of the other Great Lakes and in a highly populated region, relatively high concentrations of some contaminants such as PAHs are observed in Lake Ontario. An increase in total PAHs and some industrial compounds is observed. Other compounds indicative of consumer product sources (e.g., PBDEs, certain perfluorinated compounds) are also highest in Lake Ontario. Several organochlorines are declining, as they are in the other Great Lakes.

*The long-term trend for the chemicals assessed in this report will also be used for the 10-year trend determination.

Status Assessment Definitions

Good: The metrics show that the toxic chemical concentrations are meeting the ecosystem objectives or they are otherwise in an acceptable condition.

Fair: The metrics show that the toxic chemical concentrations are not meeting the ecosystem objectives, but they are exhibiting minimally acceptable conditions.

Poor: The metrics show that the toxic chemical concentrations are not displaying minimally acceptable conditions and are severely impacted.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Decrease in concentration or frequency of detection of toxic chemicals.

Unchanging: No change in the concentration or frequency of detection of toxic chemicals.

Deteriorating: Increased concentration or frequency of detection of toxic chemicals.

Undetermined: Data are not available or are insufficient to assess the trends or frequency of detection of toxic chemicals concentrations at this time, or the different groups of toxic chemicals are not trending in the same direction and an expert opinion of the overall direction of the trend cannot be agreed.

Endpoints and/or Targets

The target or endpoint for this sub-indicator will have been met when the Waters of the Great Lakes are free from pollutants in quantities or concentrations that could be harmful to human health, wildlife or aquatic organisms, through direct exposure or indirect exposure through the food chain. Because of the complexity (mixtures of compounds, sources and potential effects), and the presence of potentially conflicting trends (increases in some compounds amidst declines in others), status and trend determination takes a weight-of-evidence approach in making an expert assessment, including the number of compounds that are detectable and/or are below water quality guidelines (such as the CCME Water Quality Guidelines for the Protection of Aquatic Life, GLWQA Specific Objectives or other Great Lakes agency water quality guidelines, where available) and the relative effect of the compound, if known. Progress will be determined based on whether trends of the toxic chemicals are positive or negative, the rate of change in the concentrations, and by the number of chemicals which are doing so, with an emphasis on those compounds identified by the Parties as Chemicals of Mutual Concern.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the concentration of toxic chemicals in Great Lakes waters; to infer the potential for impairment to the quality of the waters of the Great Lakes by harmful pollutants; to infer progress toward virtual elimination of chemicals of mutual concern; to inform the risk assessment of toxic chemicals and the development of risk management strategies; to inform the development of environmental quality guidelines; and to report on environmental response (i.e., progress) toward the achievement of targets identified in action plans and risk management strategies for toxic chemicals in the Great Lakes basin.

Ecosystem Objective

This sub-indicator best supports work towards General Objective #4 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the Waters of the Great Lakes should "be free from pollutants in quantities or concentrations that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain".

Measure

This sub-indicator assesses the current status of toxic chemicals and will track whether concentrations are decreasing, staying the same, or increasing in Great Lakes waters over time. The chemicals included are the chemicals of mutual concern (CMCs) identified in Annex 3 of the Great Lakes Water Quality Agreement. Additional chemicals that are persistent, bioaccumulative and/or toxic (PBT) are also assessed in this report. These include, but are not limited to, other organochlorine pesticides, polycyclic aromatic hydrocarbons, chlorobenzenes, metals, pesticides in current use, and other toxic chemicals. Monitoring data are used to inform the continued assessment and potential selection of additional CMCs as well as monitor to assess the progress and effectiveness of pollution prevention and control measures for these and other compounds. The sub-indicator will primarily report offshore data because these are the focus for assessing trends; the status of this sub-indicator will consider the nearshore for those areas where the information is available (see data limitations section).

A suite of compounds is monitored on the Cooperative Science Monitoring Initiative (CSMI) rotation schedule (i.e., once every five years, per lake), with additional monitoring conducted in the intervening years, depending on appropriateness, feasibility and the availability of resources. The number of stations sampled varies by lake. This rotating schedule will require several decades to be able to detect potential trends for CMCs such as HCBD and PBDEs that were not monitored in water in a consistent way prior to 2011.

Ecological Condition

Programs and Methods

Toxic chemicals in Great Lakes waters are monitored by the Canadian and United States federal governments. Environment and Climate Change Canada (ECCC) conducts ship-based cruises to collect water quality samples as part of its Great Lakes Surveillance Program. Since 2004, this has included monitoring for the “legacy organic contaminants”, including organochlorines, PCBs, chlorinated benzenes and polycyclic aromatic hydrocarbons, using a specialized and improved technique that permits the accurate detection of concentrations in Great Lakes waters. Monitoring is generally conducted during spring cruises as this timing has been determined to be optimal to establish annual maxima for many of these compounds (Williams et al. 2001). From 2004 to 2013, monitoring for contaminants was conducted on each lake every two or three years. Since 2013, monitoring is coordinated with the CSMI, so that work is focused on one of the Great Lakes each year and the frequency has been somewhat reduced. For the CMCs PCBs and mercury, high quality and consistent data are available since 2004 for all of the Great Lakes that ECCC monitors.

Data from a subset of 18 stations distributed throughout all five of the Great Lakes were collected as part of a binational sampling effort in 2011 – 2012, using a technique to concentrate very large sample volumes (100-200 L) onto resin columns (Venier et al. 2014). These data have permitted assessment of the status of legacy contaminants as well as the CMCs HCBD and PBDEs, and certain PBDE replacement compounds that may not otherwise be detected in smaller volumes. Resin data collected since that time will be incorporated into future reporting; however, due to the low resampling frequency (once every five years), it may take up to several decades or more to detect any trends in Great Lakes waters for HBCD and PBDEs.

For the CMCs PFOA, PFOS, the first surveillance occurred in 2008 in Lakes Ontario and Superior, and more consistent monitoring has been initiated since 2012. Sampling has been more frequent than the CSMI schedule, but sufficient time has not yet passed to report on trends in water. The most recent assessment of perfluorinated compounds in water can be found in Gewurtz et al (2019).

Monitoring for toxic chemicals in Lake Michigan waters was conducted jointly by ECCC and US EPA in 2006, 2010 and 2012. Since that time, monitoring is conducted by Clarkson University under grant from the US EPA. Recent years of monitoring for CMCs are indicated in Table 1. Data will be included in future State of the Great Lakes reporting.

All major Great Lakes regions (nearshore, offshore and major embayments) are monitored in ECCC’s program. Tributary sampling, closer to sources and more informative with respect to maximum concentrations, is largely lacking; some information may exist but has not been incorporated in this report. Due to inherently high sample collection and laboratory costs, sample sizes are in general quite small, limiting our ability to assess all areas and parameters. Additional field techniques to reduce sample costs are being considered. The status of contaminants in the Great Lakes is performed using all available quality-assured data and the trends are based on data collected since 2004 because laboratory and field techniques improved greatly at that time, as described above.

Status of Chemicals of Mutual Concern (CMCs)

The Parties to the GLWQA designated the first set of CMCs in May 2016. For State of the Great Lakes reporting in 2019 and moving forward, the toxic chemical sub-indicators where possible, will support reporting on CMCs in a more fulsome, consistent and transparent way. Information on additional chemicals of interest for the Great Lakes (non-CMCs) is valuable for inclusion in the report and will be included in a separate section below, as appropriate. For the 2019 Toxic Chemicals in Water sub-indicator report, the assessment is not based solely on CMCs.

The CMCs are:

- Hexabromocyclododecane (HBCD)
- Long-Chain Perfluorinated carboxylic acids (LC-PFCAs)
- Mercury (Hg)
- Perfluorooctanoic acid (PFOA)
- Perfluorooctane sulfonate (PFOS)
- Polybrominated Diphenyl Ethers (PBDEs)
- Polychlorinated Biphenyls (PCBs)
- Short-Chain Chlorinated Paraffins (SCCPs)

Table 1 provides the monitoring record since 2011 for CMCs in each of the Great Lakes.

Polychlorinated biphenyls

PCBs are both a CMC and a legacy contaminant. Despite being banned in 1977 in the United States and Canada, PCBs continued to be used and stored. While inventories have been reduced over the past several decades, PCBs continue to be detected throughout the Great Lakes. Concentrations of total PCBs are observed according to the following spatial trend: Ontario ≈ Erie > Huron ≈ Michigan > Superior ($p < 0.001$; Venier et al. 2014). Within each lake, spatial distributions indicate higher concentrations in harbours and nearshore regions compared to offshore waters. The highest individual concentrations are observed in the western basin of Lake Erie and concentrations decrease as waters flow to the central and eastern basin of the lake. PCB concentrations in Lake Michigan waters are higher in Green Bay and near Chicago compared to the offshore. In Lake Huron, PCB concentrations are highest in Saginaw Bay and offshore concentrations are lower and appear to decline from south to north within the main body of the lake. There is no temporal trend in total PCBs since 2004, although we know from sediment core data (see Toxic Chemicals in Sediment sub-indicator) and fish tissue data (see Toxic Chemicals in Great Lakes Whole Fish sub-indicator) that PCB concentrations have declined over the longer term. The Ontario Provincial Water Quality Objective of 1 ng/L is used as a benchmark and it has been exceeded in some years in Lake Erie and Hamilton Harbour (Lake Ontario). The most recent data demonstrate the above-noted spatial distribution but no exceedances of the benchmark are observed.

Mercury

Concentrations of total mercury (i.e., unfiltered water) are highest in Lake Erie and significantly lower in offshore waters of the other Great Lakes (Figure 1). The Canadian Council of Ministers of the Environment (CCME 1999) guideline for mercury in water (26 ng/L for the protection of aquatic life) has not been exceeded although maximum concentrations in the western basin of Lake Erie in 2009 at the stations sampled (mean 13.2 and maximum 18.2 ng/L) approached the guideline. Higher concentrations of mercury have been noted in Lake Erie previously (Dove et al. 2011), due to the historic presence of chlor-alkali and other industries in the St. Clair River – Detroit River inter-connecting channel.

The overall decline in mercury from historic high levels is supported by long-term measurements in fish and sediment (for example, see the Toxic Chemicals in Great Lakes Whole Fish and Toxic Chemicals in Sediment sub-indicator reports). Mercury in water declined significantly between 2003 and 2009 (Dove et al. 2011); however, since that time, this decline may have slowed or halted (Figure 2). Since about 2000, Great Lakes predator fish have also recently experienced either flat or increasing trends of mercury (Bhavsar et al. 2010, Blukacz-Richards et al., 2017, Toxic Chemicals in Whole Fish sub-indicator report). The increase of mercury in fish, without a concurrent increase in water, implies that changes in mercury cycling may be occurring in the Great Lakes environment.

Flame Retardants

Recent work conducted on each of the Great Lakes sampled for polybrominated diphenyl ethers (PBDEs), and other flame retardants (Venier et al. 2014). The results showed higher concentrations in the lower Great Lakes and the spatial patterns were consistent with consumer products as a primary source (Figure 3). PBDE congener patterns reflected the Penta-BDE and Deca-BDE mixtures. Alternative brominated flame retardants were detected, reflecting the wide usage of these replacement products for the commercial Penta-BDE mixture. Dechlorane Plus and hexabromocyclododecane (HBCDD) concentrations were highest in Lake Ontario, reflecting manufacturing sources and usage patterns. The ubiquity of flame retardants reflects their widespread usage in commercial products, and it will remain important to continue risk assessment activities, monitor ambient levels and to track progress if and when the use of these compounds is regulated.

Perfluorocarbons (PFCs; PFOS and PFOA)

Results for PFCs are consistent with patterns of consumer product sources, with higher concentrations noted near urban regions (Gewurtz et al. 2013, 2019). Additional years of monitoring are required to determine any trends of PFCs in Great Lakes water; none are observed since surveillance began in 2008. Information about recent concentrations of these compounds is shared promptly with risk assessment and risk management agencies in order that decision making is based on the most recently available, best science.

Other Chemicals of Interest

Table 2 lists the parameters included in the legacy contaminant suite that have been routinely monitored since 2004 and indicates those that have been detected in more than 10% of recent samples for each of the lakes.

Organochlorine Pesticides and Industrial Byproducts

Organochlorine pesticides have been banned, restricted or discontinued but many remain ubiquitous in the Great Lakes. Overall, the most abundant organochlorines present in Great Lakes waters are alpha-HCH, dieldrin and lindane. Concentrations of alpha-HCH and gamma-HCH (Lindane; Figure 4) are highest in Lake Superior and dieldrin is generally highest in Lake Michigan although recent data show highest concentrations in the western basin of Lake Erie. Due to its large surface area, cold water temperature and long retention time, Lake Superior is most susceptible to accumulation of these compounds. All three of these compounds are declining over time. The decreasing trend for lindane is dramatic (Figure 5). The voluntary removal of lindane was announced by the Canola Council of Canada in 1998 (National Round Table on the Environment and the Economy, 2001). In 2006, the U.S. EPA banned the agricultural use of lindane and in 2009 the production and agricultural use of lindane was banned under the Stockholm Convention. Figure 5 demonstrates that the in-lake concentrations responded to these reductions with declines observed in each of the Great Lakes (statistically significant in lakes Erie and Ontario) since our measurements began in 2004.

For the industrial byproducts, the most abundant are hexachlorobenzene (HCB) and hexachlorobutadiene (HCBD). Concentrations are highest in the lower Great Lakes (lakes Erie and Ontario) because sources have historically been greater in the more industrial regions and these compounds are not as subject to atmospheric transport. Increasing trends are observed for both compounds in most lakes, although the trends are statistically significant ($p<0.05$) only for HCB in Lake Huron and the east basin of Lake Erie, and for HCBD in Lake Ontario.

Polycyclic aromatic hydrocarbons (PAHs)

The most abundant PAHs observed in Great Lakes waters include naphthalene, phenanthrene, fluoranthene, fluorene and pyrene. Higher molecular weight PAHs are less frequently detected because they tend to be less soluble in water and partition instead to sediment. Concentrations of total PAHs (the sum of 17 individual PAH compounds) are highest in lakes Erie and Ontario, intermediate in lakes Huron and Michigan, and lowest in Lake Superior. This spatial distribution follows the pattern of usage, with more intense industry and urbanization observed in the lower Great Lakes. Generally stable conditions or increases are observed for PAHs. Total PAHs are unchanged in most lakes although statistically significant increases are observed for Lake Ontario and Georgian Bay, largely driven by increasing naphthalene and fluorene concentrations. In an urban setting, PAHs in Lake Ontario were predominantly from tributary loading; therefore, source reductions must ultimately come from non-point sources (Melymuk et al. 2014).

In-Use Pesticides

Currently-used pesticides have been monitored in the Great Lakes since about 1994 and in high priority tributaries federally from about 2002-2016. This monitoring includes suites of compounds known as acid herbicides, neutral herbicides and organophosphorus insecticides (Struger et al. 2004). Toward the latter end of the monitoring record, additional compounds such as glyphosate and carbamates were monitored due to dramatic increases in their usage in the Great Lakes basin. The most commonly observed compounds were atrazine, metolachlor and 2,4-D. In Great Lakes waters, concentrations at the monitored locations have not exceeded CCME guidelines, indicating good status, and no temporal trends were observed. Concentrations of these compounds were highest in the lower Great Lakes (i.e., lakes Erie and Ontario), with maximum concentrations generally observed in the western basin of Lake Erie (e.g., for glyphosate). In tributaries, concentrations tend to be highest in agricultural and urban areas, although there has been a marked recent decline in the concentrations of urban pesticides in Ontario streams, primarily due to enhanced pesticide regulation at the provincial level (Todd and Struger 2014). Pesticide concentrations in monitored tributaries indicate occasional (at some sites, routine) exceedance of guidelines (e.g., 2,4-D, atrazine, metolachlor, chlorpyrifos) and the widespread presence of a longer list of pesticide compounds (Struger, pers. comm., Struger et al. 2016). The cumulative effect of chronic exposure to pesticide mixtures is also a gap requiring attention. Monitoring for pesticides has been reduced in the Great Lakes to the extent that status and trends may not be able to be reported in the future.

Toxaphene

Toxaphene is not routinely monitored but its discussion is merited due to its relevance to the Great Lakes. Toxaphene was banned almost 40 years ago, and its use in the Great Lakes basin was minimal, but atmospheric transport and deposition, combined with its high persistence and retention in cold environments, has resulted in its presence at relatively high concentrations in both Great Lakes water (Muir et al. 2006) and fish (Xia et al. 2012). Concentrations of toxaphene are highest in Lake Superior compared to the other lakes, where it is responsible for approximately 7% of the fish consumption advisories (Ontario Ministry of the Environment and Climate Change, 2015). Toxaphene concentrations in all the lakes are declining, with a modeled half-life of 9.2 years in Lake Superior (Xia et al. 2011). Similar rates of decline have been observed in Great Lakes fish (Xia et al. 2012). However, it may take 30 years for Lake Superior Lake Trout tissue concentrations to decline to concentrations observed in the other Great Lakes (Xia et al. 2011).

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Toxic Chemicals in Whole Fish – interpretation of status and trends is conducted jointly to determine the degree of concordance between the information sources – for example to determine if temporal trends observed in fish are due to water quality changes or due to biological mediation.

- Toxic Chemicals in Sediment – longer-term trends of Great Lakes toxics may be discerned from retrospective analyses in sediment cores. Trends of high molecular weight PAHs may be more accurately monitored in sediments; a disadvantage is that it may take a very long time to be able to track progress.
- Toxic Chemicals in the Atmosphere– water quality data are required for the calculation of fluxes, and temporal data are required to interpret trends in atmospheric concentrations and changes in deposition rates.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources.	X			
Geographic coverage and scale of data are appropriate to the Great Lakes basin			X	
Data obtained from sources within the U.S. are comparable to those from Canada		X		
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes*	URL: https://open.canada.ca/data/en/dataset/cfdafa0c-a644-47cc-ad54-460304facf2e		
Clarifying Notes: The scale of data is basin-wide using consistent methods and analysis. The geographic coverage is insufficient to be able to assess all areas and especially those closer to sources. Additional U.S. data are currently being gathered or generated and will be incorporated into future sub-indicator reports.				

*The content is currently incomplete, but will be made available in the future.

Data Limitations

- Currently, Canada is conducting routine monitoring for contaminants in the offshore in each of the Great Lakes. Some information is available from the U.S., and will be provided in the sub-indicator report when it is available.
- Although some nearshore information is available, inclusion of additional data from the States and the Province of Ontario and other potential contributors, where available, may be useful to assess nearshore status

and trends.
- Monitoring data from the Great Lakes Connecting Channels (e.g., Niagara River and St. Clair River) may be incorporated in the future for reporting long-term trends.

Additional Information

Monitoring by the Parties will continue to track the status of toxic chemicals in Great Lakes Waters. The assessment of organic contaminants in water can be challenging given the relatively complex field and laboratory requirements. The associated high costs limit the monitoring of toxic chemicals in Great Lakes waters; for example, monitoring of pesticides has largely been discontinued. Water provides a stable medium for the assessment of contaminants which may be more challenging in other media (e.g. for compounds having short residence times in air, those not bioaccumulating in fish tissue or binding to sediment, or those undergoing dramatic or complex transformations or biogeochemical cycling in the environment). Information from other media should be considered for certain compounds, especially those that are bioaccumulative and/or persistent. The assessment of contaminants in Great Lakes offshore waters is a valuable means to determine the status and trends for soluble and/or toxic compounds including many of the CMCs and to collect information for compounds that may be identified as CMCs in the future.

Several of the environmental quality guidelines that were previously available for legacy organic contaminants have been withdrawn. The Canadian Council of Ministers of the Environment has withdrawn guidelines for a-HCH and PCBs in water in favour of the use of fish and sediment guidelines as these compounds are hydrophobic and/or bioaccumulative. There is therefore a lack of benchmarks against which to gauge the lakes' status. Despite the dearth of guidelines, the assessment of toxic contaminants is important as the current status represents an important means of assessing exposure for biota and the offshore temporal data series provide a means of assessing trends.

The concentrations of many legacy organic contaminants are low in the offshore waters of the Great Lakes, are reduced from historical maxima, and are currently changing slowly. The realignment of the monitoring schedule with the CSMI will result in less frequent data collection for these compounds, and this is warranted. For potential CMCs requiring surveillance, and for those requiring more frequent assessment due to changing levels in the environment, the schedule may not follow the CSMI in order to effect adequate monitoring.

Acknowledgments

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Parameter	Lake Superior	Lake Michigan	Lake Huron	Georgian Bay	Lake Erie	Lake Ontario
Mercury	2011, 2013, 2016	2014 [a]	2012, 2014, 2015, 2017	2012, 2014, 2017	2012, 2014, 2017	2011, 2012, 2013, 2015
PFOA	2016		2012, 2014, 2017	2012, 2014, 2017	2012, 2013, 2015	2013, 2015
LC PFCAs						
HBCD	2011, 2016	2012	2012, 2017	2012, 2017	2012, 2014	2011, 2013
PCBs	2011, 2016	2012, 2015 [a]	2012, 2017	2012, 2017	2012, 2014	2011, 2013
PFOS	2016		2012, 2014, 2017	2012, 2014, 2017	2012, 2013, 2015	2013, 2015
PBDEs	2011, 2016	2012	2012, 2017	2012, 2017	2012, 2014	2011, 2013
SCCPs						

Table 1. Monitoring of Chemicals of Mutual Concern in Great Lakes waters, 2011 – 2017. Source: Environment and Climate Change Canada except [a)] from Clarkson University

Parameter	Lake Superior	Lake Michigan	Lake Huron	Georgia Bay	Lake Erie	Lake Ontario
Organochlorines						
Alpha-Chlordane	×	×			×	×
Alpha-Endosulfan		×	×		×	×
Alpha-HCH	×	×	×	×	×	×
Beta-Endosulfan	×	×			×	×
Dieldrin	×	×	×	×	×	×
Gamma-chlordane					×	×
Lindane	×	×	×	×	×	×
Mirex						
o'p'-DDT						
Octachlorostyrene						
p'p'-DDD					×	×
p'p'-DDE					×	×
p'p'-DDT						
Industrial byproducts						
Hexachlorobenzene	×	×	×	×	×	×
Hexachlorobutadiene					×	×
Pentachlorobenzene	×	×	×	×	×	×
Polychlorinated biphenyls[1]	×	×	×	×	×	×
Polycyclic Aromatic Hydrocarbons (PAHs)						
Acenaphthene		×	×		×	×
Acenaphthylene					×	×
Anthracene					×	×
Benzo(a)anthracene					×	×
Benzo(a)pyrene					×	
Benzo(b,k)fluoranthene					×	×
Benzo(e)pyrene					×	×
Benzo(ghi)perylene						
Chrysene	×				×	×
Dibenzo(ah)anthracene						
Fluoranthene	×	×	×	×	×	×
Fluorene	×	×	×	×	×	×
Indeno(1,2,3-cd)pyrene					×	
Napthalene	×	×	×	×	×	×
Perylene		×				
Phenanthrene	×	×	×	×	×	×
Pyrene	×	×	×		×	×

Table 2. Legacy organic contaminants monitored in Great Lakes waters. Parameters detected in more than 10% of samples are indicated with an "x". Source: Environment and Climate Change Canada

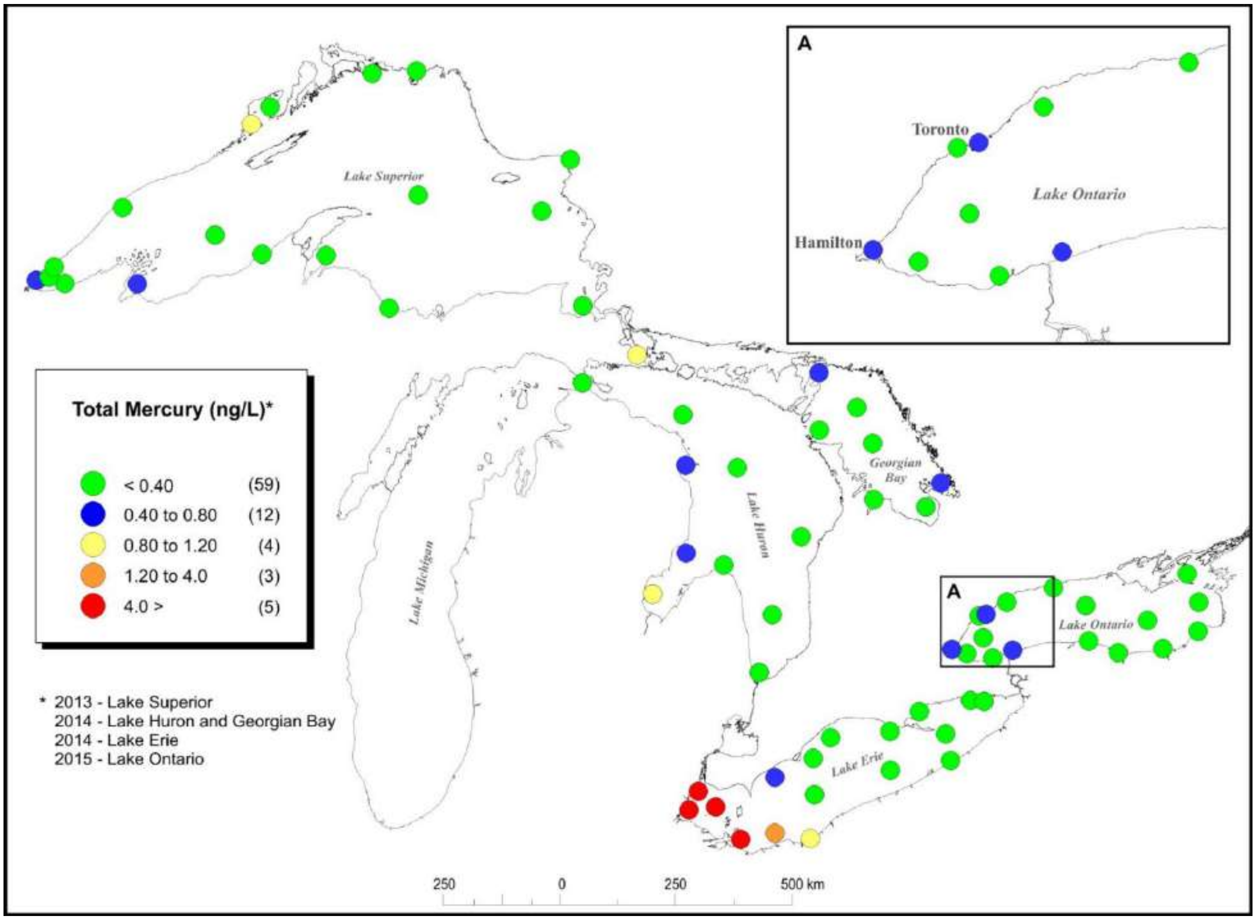
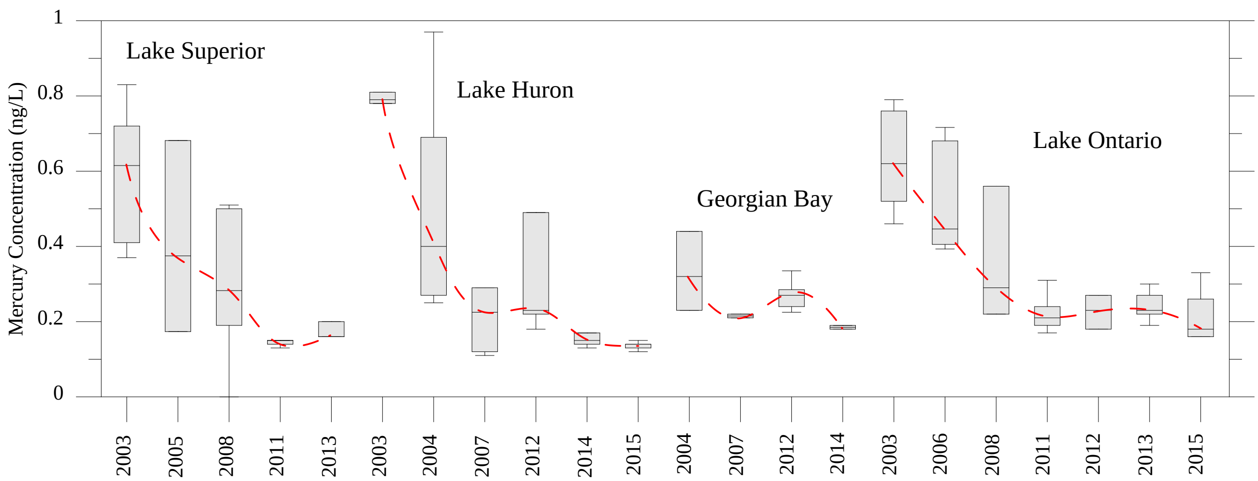


Figure 1. Spatial distribution of total mercury in the Great Lakes. Data are the most recent available spring, surface data for all stations. * = years of data included for total mercury analysis. Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance Program

a)



b)

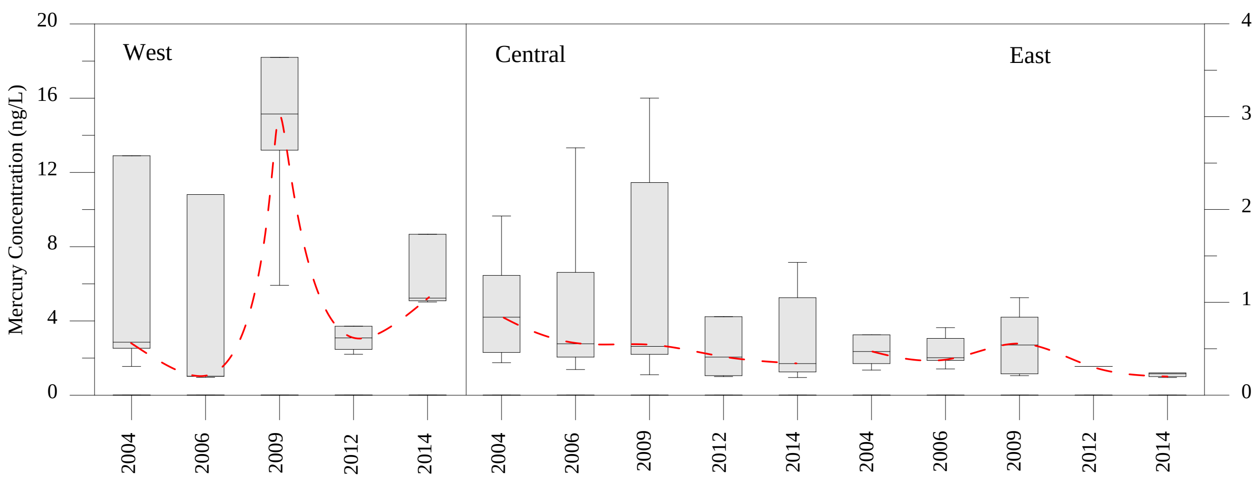


Figure 2. Temporal changes of total mercury in the Great Lakes. Data are a) Great Lakes spring, surface data from offshore stations and b) Lake Erie spring, surface data from all stations by basin. Lake Erie west basin data for are scaled using the left-hand vertical axis and central and east basin data are scaled using the vertical axis on the right. Boxes show central median and 25% and 75% values, whiskers show 1.5x interquartile range. Temporal trends indicate declines in all of the lakes (not statistically significant for Georgian Bay) with the exception of Lake Erie, where there is no significant change. Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance Program

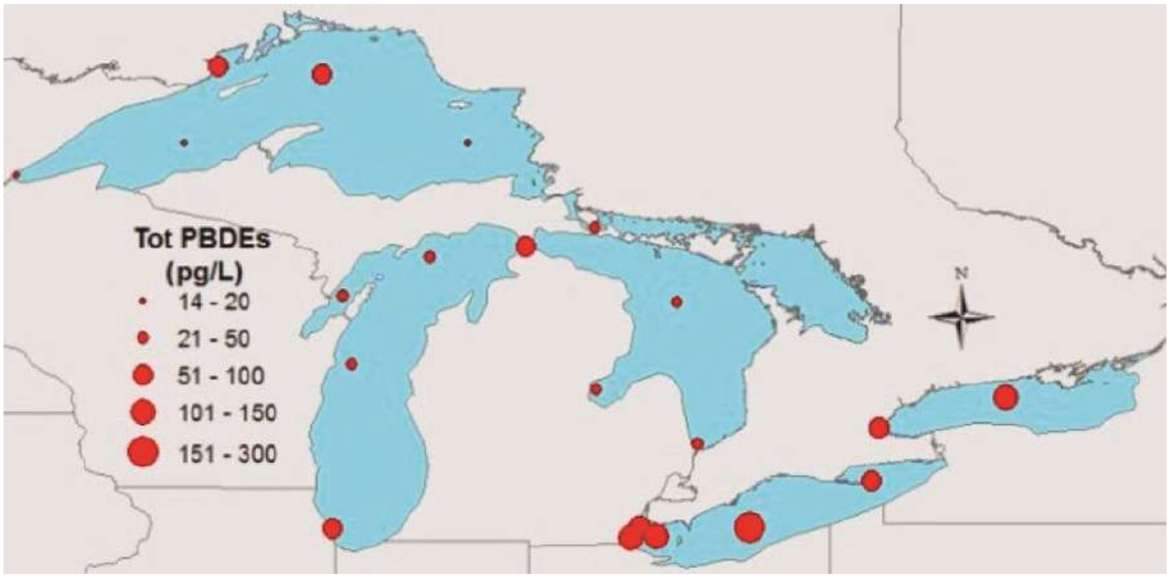


Figure 3. Spatial distribution of the concentrations of total PBDEs, pg/L. Source: Venier et al. 2014

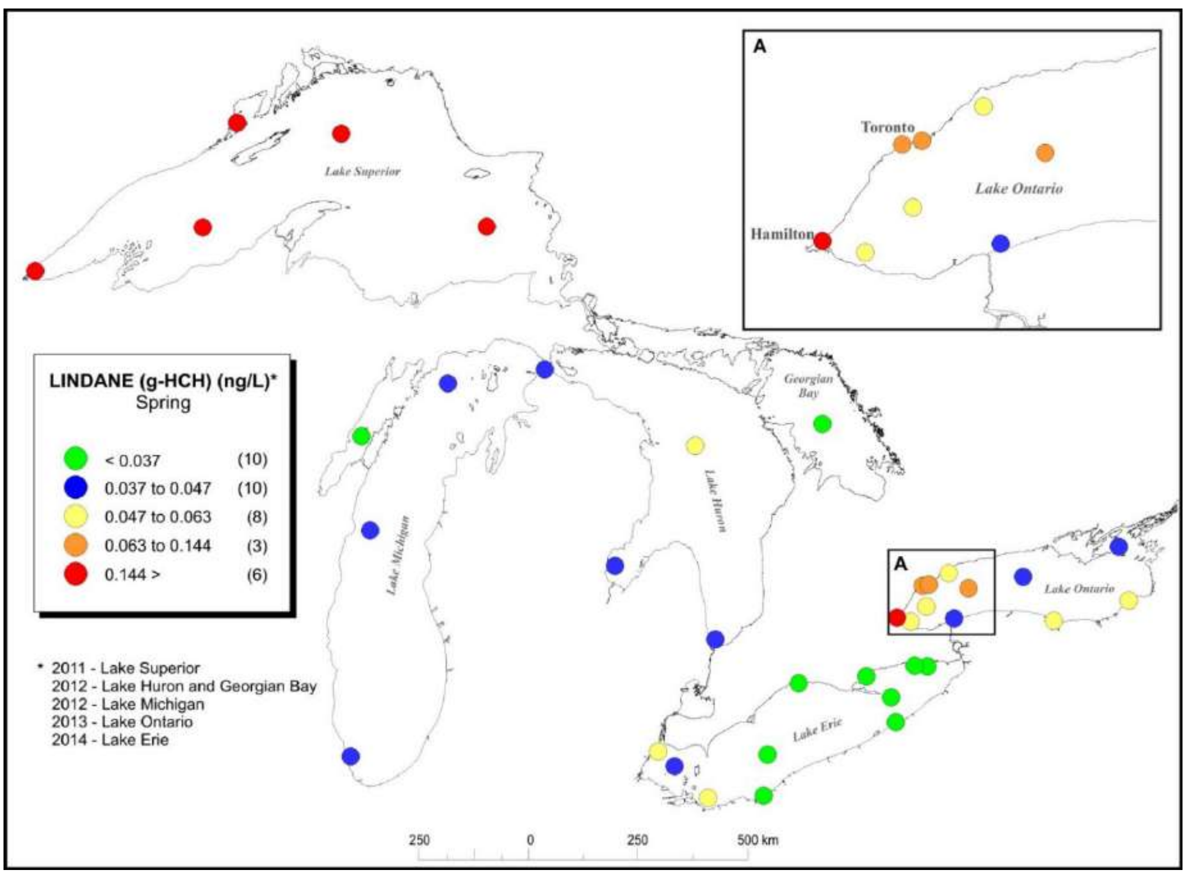


Figure 4. Spatial distribution of dissolved lindane (gamma hexachlorocyclohexane) in the Great Lakes. Data are the most recent available spring, surface data for all stations. * = years of data included for lindane analysis. Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance Program

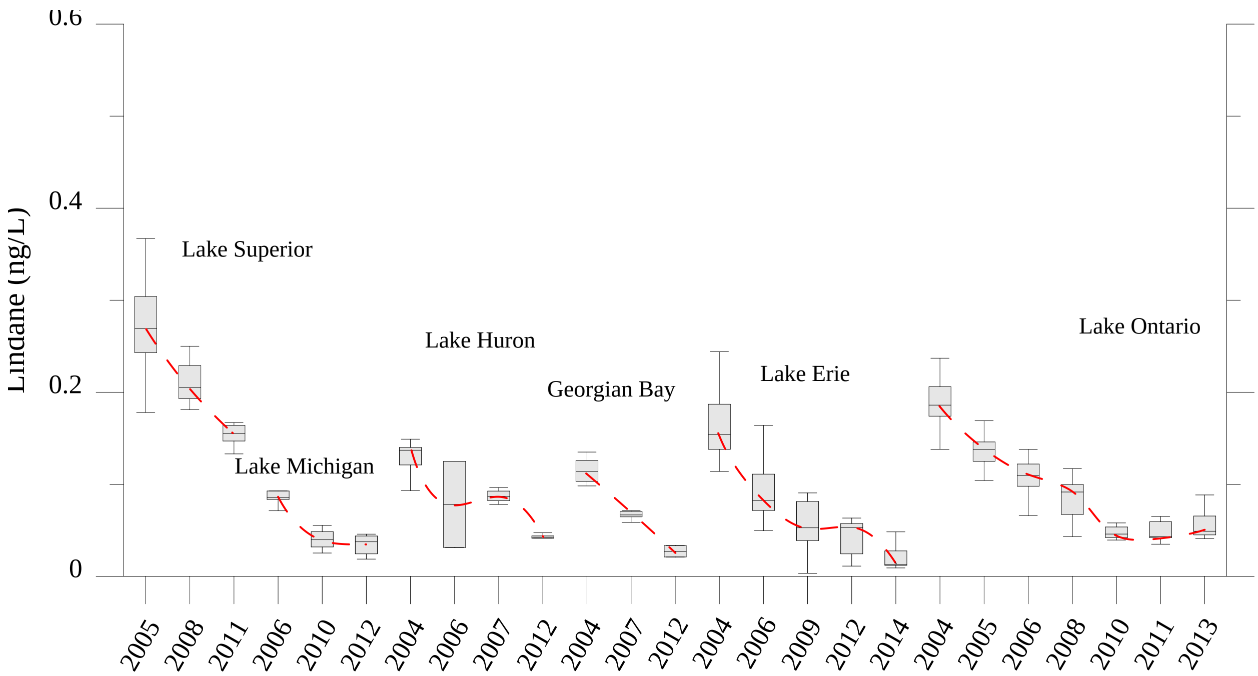


Figure 5. Temporal changes of dissolved lindane in the Great Lakes. Data are spring, surface data at offshore stations. Boxes show central median and 25% and 75% values, whiskers show 1.5x interquartile range. Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance Program

Sub-Indicator: Toxic Chemicals in Whole Fish

Overall Assessment

Status: Fair

Trends:

10-Year Trend (2009-2018): Unchanging

Long-term Trend (1977 – 2018): Improving

Rationale: While concentrations of some CMCs continue to exceed environmental guidelines or targets (Table 1 and Lake by Lake Assessments), on average the levels of chemicals of mutual concern (CMCs) are near their guidelines and the average Mean Deviation Ratio (MDR) for CMCs in whole fish remains in the yellow zone of the MDR chart, meaning the MDR is not significantly greater or less than zero and the overall status of the Great Lakes Basin is assessed as FAIR (Figure 1). The MDR is a unit-less composite measure of the extent to which contaminant concentrations in fish exceed guidelines. It is utilized as an indicator of the state of toxic chemicals in whole fish. Calculating the MDR factors in concentrations of Tetra-, Penta-, and Hexa-brominated diphenyl ethers (TeBDE, PeBDE, HxBDE), total mercury (Hg), total polychlorinated biphenyls (Total PCBs), hexabromocyclododecane (HBCD), and perfluorooctane sulphonate (PFOS). Long term conditions have IMPROVED since monitoring began in 1977 due mostly to declines in PCBs, mercury and, more recently, PBDEs. A statistically significant trend in the MDR is not present over the last 10-years of monitoring, indicating that the condition has remained UNCHANGED. Concentrations of PCBs, PBDEs, and PFOS still remain above environmental guidelines in all five Great Lakes. Because of the shift to focus on CMCs, this sub-indicator will no longer report out on legacy pollutants that are not CMCs (such as DDT). Through the Annex 3 process, new chemicals designated as CMCs will be further incorporated into monitoring and surveillance programs, when applicable.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

Trends:

10-Year Trend: (2009-2018): Unchanging

Long-Term Trend (1977 – 2018): Improving

Rationale: In Lake Superior, the MDR indicates that on average the concentrations of CMCs are below targets or guidelines. While this is true for mercury, TeBDE, HxBDE and HBCD, concentrations of the remaining CMCs used in this calculation (PCBs, PeBDE, and PFOS) are still above environmental guidelines or targets (Table 1) in Lake Superior. MDR values for Lake Superior fall within the yellow zone of the MDR chart indicating that the MDR is not significantly different from zero and that overall conditions remain as FAIR (Figure 2). Despite the increase in the MDR in ~2000, which was a result of the introduction of PBDEs into our monitoring programs, conditions have IMPROVED since monitoring began in 1977. A statistically significant trend was not present in the last 10-years of monitoring indicating that the condition has remained UNCHANGED.

Lake Michigan

Status: Fair

Trends:

10-Year Trend: (2009-2018): Unchanging

Long-Term Trend (1982 – 2018): Improving

Rationale: In Lake Michigan, the MDR indicates that on average the concentrations of CMCs are above environmental targets or guidelines (Table 1). Concentrations of mercury, TeBDE, and HBCD are below environmental guidelines or targets while those of PCBs, PeBDE, and PFOS are above guidelines or targets. The MDR values for Lake Michigan fall within the yellow zone of the MDR chart meaning that the MDR is not significantly different from zero and that overall conditions are FAIR (Figure 3). Since monitoring began, conditions have IMPROVED in Lake Michigan due to the continued declines of PCBs as well as Te- and PeBDEs. A statistically significant trend was not present in the last 10-years of monitoring indicating that the condition has remained UNCHANGED.

Lake Huron

Status: Fair

Trends:

10-Year Trend: (2009-2018): Unchanging

Long-Term Trend (1982 – 2018): Improving

Rationale: In Lake Huron, the MDR indicates that on average the concentrations of CMCs are below environmental targets or guidelines (Table 1). Mercury, TeBDE, HxBDE and HBCD concentrations are below guidelines or targets while those of PCBs, PeBDE, and PFOS were above environmental guidelines or targets. The MDR values for Lake Huron fall within the yellow zone of the MDR chart indicating that the MDR is not significantly different from zero and that overall conditions are FAIR (Figure 4). Since monitoring began, conditions have IMPROVED in Lake Huron due largely to declines of PCBs and TeBDE. A statistically significant trend was not present for the MDR in the last 10-years of monitoring indicating that the condition has remained UNCHANGED.

Lake Erie

Status: Fair

Trends:

10-Year Trend: (2009-2018): Unchanging

Long-Term Trend (1982 – 2018): Improving

Rationale: In Lake Erie, the MDR indicates that on average the concentrations of CMCs are below environmental targets or guidelines (Table 1). Mercury, TeBDE, HxBDE, and HBCD concentrations are below guidelines or targets while those of PCBs, PeBDE, and PFOS were above environmental guidelines or target. The MDR values for Lake Erie fall within the yellow zone of the MDR chart indicating that the MDRis not significantly different from zero and that overall conditions are FAIR (Figure 5). Since monitoring began, conditions have IMPROVED in Lake Erie due largely to declines of PCBs. A statistically significant trend was not present for the MDR in the last 10-years of monitoring indicating that the condition has remained UNCHANGED.

Lake Ontario

Status: Fair

Trends:

10-Year Trend: (2009-2018): Unchanging

Long-Term Trend (1982 – 2018): Improving

Rationale: In Lake Ontario, the MDR indicates that on average the concentrations of CMCs are higher than environmental targets or guidelines. PCBs, PeBDE, HxBDE, and PFOS concentrations are above guidelines or targets while those mercury, TeBDE, and HBCD were below environmental guidelines or targets. The MDR values for Lake Ontario fall within the yellow zone of the MDR chart indicating that the MDR is not significantly different from zero and that overall conditions are FAIR (Figure 6). Since monitoring began, conditions have IMPROVED in Lake Ontario due largely to declines of PCBs, mercury and PBDEs. A statistically significant trend was not present for the MDR in the last 10-years of monitoring indicating that the condition has remained UNCHANGED.

Status Assessment Definitions

Good: The MDR shows that, on average, levels of CMCs in fish are below ecosystem objectives or targets (Table 1) with 95% confidence or they are otherwise in an acceptable condition, in the green zone of the MDR chart.

Fair: The MDR shows that, on average, levels of CMCs in fish are at or near, but not significantly above or below, ecosystem objectives or targets (Table 1) with 95% confidence, in the yellow zone of the MDR chart

Poor: The MDRs show that on average, levels of CMCs in fish are above ecosystem objective or targets (Table 1) with 95% confidence, in the red zone of the MDR chart.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Concentrations of contaminants are declining with statistical significance according to the Mann-Kendall test

Unchanging: Concentrations of contaminants are not declining or increasing with statistical significance.

Deteriorating: Concentrations of contaminants are increasing with statistical significance

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Endpoints and/or Targets

The purpose of this sub-indicator is to identify the presence of CMCs and new and emerging chemicals of concern in Great Lakes whole fish and provide an interpretation and an explanation of what the presence of those chemicals means to the health of the Great Lakes ecosystem. In the absence of binational targets in the 2012 GLWQA, contaminant concentrations will be compared to environmental quality guidelines (ex. Canadian Federal Environmental Quality Guidelines - FEQGs) or other published ecotoxicological thresholds. The criteria used in this report are provided in Table 1. These criteria were selected by the authors based on availability, status (draft vs.

final), appropriateness, and best professional judgement to be most protective of the Great Lakes ecosystem. Criteria can be draft or interim for a while in the absence of a decision or concurrence or an alternative. For example, the 2012 GLWQA includes interim P concentrations and load targets.

Sub-Indicator Purpose

The purpose of this sub-indicator is to describe temporal and spatial trends of bioavailable contaminants in representative open water fish species from throughout the Great Lakes; to infer the effectiveness of remedial actions related to the management of critical pollutants; and to identify the nature and describe the trends of new and emerging chemicals of concern. The Toxic Chemicals in Whole Fish sub-indicator includes contaminant from whole body fish, which includes bones, organs, and blood of the fish. Concentrations presented in this report do not necessarily reflect contaminant concentrations in edible portions of fish tissue, please see Contaminant in Edible Fish sub-indicator for this information.

Ecosystem Objective

Great Lakes waters should be free of toxic substances that are harmful to fish and wildlife populations. This sub-indicator best supports work towards General Objective #4 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the Waters of the Great Lakes should "be free from pollutants in quantities or concentrations that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain." The programs used to develop this indicator directly impact and collaborate with Annex 3, Chemicals of Mutual Concern, and Annex 10, Science, of the 2012 Great Lakes Water Quality Agreement.

Measure

The assessment of status and trend incorporates chemical concentrations of multiple CMCs, from multiple fish species, in all 5 of the Great Lakes over time. The MDR has been refined from the previous reporting cycle and is used to estimate the overall status and trends based on multiple variables. The overall and lake specific assessments in the sub-indicator are limited to the CMCs monitored in whole fish and include: Tetra-, Penta-, and Hexa-brominated diphenyl ethers (TeBDE, PeBDE, HxBDE), total mercury (Hg), total polychlorinated biphenyls (Total PCBs), hexabromocyclododecane (HBCD), and perfluorooctane sulphonate (PFOS). Data and analyses for the sub-indicator report were provided by the United States Environmental Protection Agency (USEPA) and Environment and Climate Change Canada (ECCC) generated from existing and long-term fish contaminant monitoring and surveillance programs operating in the Great Lakes basin. Individual and composite samples of homogenized whole fish are analyzed to provide data on priority contaminants present in Great Lakes aquatic ecosystems. Data are statistically analysed to determine mean and variance for each fish species, chemical, lake and year as well as temporal trends.

Mean Deviation Ratio (MDR)

The MDR is a unit-less ratio that represents the average distance the contaminants included in the calculation are from the desired condition (ecosystem objective or targets). If all contaminants were present at their target or guideline the MDR calculation would result in a value of zero. MDR values significantly greater than zero indicate that average conditions are above the targets or guidelines, which for toxic chemicals would be interpreted as

"Poor" condition. MDR value that are significantly less than zero indicate that conditions are below the targets or guidelines and interpreted as "Good" condition. The MDR is calculated as presented in equation 1:

$$(1) \qquad MDR = \frac{\sum_{j=1}^{P}\left(\frac{\sum_{i=1}^{n}\log\left(\frac{C_{ij}}{T_j}\right)}{n_j}\right)}{P}$$

Where P is the number of parameters measured, nj is the number of measurements in a time period for each parameter j, Cij is the ith measured concentration of each parameter j, and T is the target or environmental guideline for parameter j. Over 95% of measured concentrations were above the detection limit; half the detection limit was substituted in for the small number of values which were below the detection limit. Briefly, each measured concentration is divided by its corresponding guideline, the resulting ratio is averaged for each contaminant in each collection year, and the overall mean of the individual contaminant means within each year is the annual MDR. The MDR was calculated annually for the entire period of record for each of the Great Lakes using the parameters and guidelines listed in Table 1. The variance in the MDR calculation for each lake was carried through all steps of the calculation in order to estimate the variance of each annual MDR value (SOGL 2019). The variance of the 10 years including and preceding the most current value was used to characterize the contemporary standard deviation of the MDR. A region above and below a value of zero bounded by 1.96 x standard deviation was deemed to represent a value of "Fair" ecological condition. This region would approximate the 95% confidence interval of the MDR and any value within this region would have an interval that straddles the boundary between "Good" and "Poor" ecological condition (zero).

Temporal trends of the status of this sub-indicator are assessed for each contaminant and for the MDR for the entire period of record (long-term) and for the last 10 years (recent). Trends are assessed using non-parametric Mann-Kendall test with α=0.05 using the package "trend" in r-stats (Pohlert 2018, trend: Non-Parametric Trend Tests and Change-Point Detection. R package version 1.1.0.), which is a commonly used technique to statistically analyze the significance of monotonic contaminant time trends. Mann-Kendall tests cannot be performed on time series with missing values hence, any missing values (i.e. years where concentrations of a parameter were not measured) were imputed using the "na.kalman" function from the package "imputeTS" in r-stats (Moritz and Bartz-Beielstein 2017).

This imputes annual means for each parameter for years where data is missing by fitting the existing data to a structural model and then imputing the missing means using a kalman smoothing function.

Ecological Condition

Background and Methods

Long-term (greater than 25 years), basin-wide monitoring programs that measure whole body concentrations of contaminants in top predator fish (Lake Trout (*Salvelinus namaycush*) and/or Walleye (*Sander vitreus*)) are conducted by both the USEPA Great Lakes National Program Office through the Great Lakes Fish Monitoring and Surveillance Program (GLFMSP), and the Fish Contaminants monitoring and surveillance activities included in ECCC's Fresh Water Quality Monitoring Program. These monitoring programs aim to identify risks posed from contaminants to fish and their wildlife consumers as well as to monitor trends in time as a measure of progress towards the Ecosystem Objectives of the 2012 GLWQA. The *Toxic Chemicals in Whole Fish* sub-indicator is included in the Toxic Chemicals indicator assessment for the Great Lakes since long-term trends of contaminants in biota provide valuable insights into the relative abundance of CMCs and other bioaccumulative contaminants in the environment. Fish integrate their exposure to contaminants over time and across their range and thus provide a broader assessment of environmental exposure than would a water sample taken at a single location at a point in time. Bioaccumulative contaminants are also found at higher concentrations in biota than they are in water, allowing for more accurate and cost-effective determination of levels in the environment. It is important to note, however, that contaminant levels in biota are not just influenced by contaminant concentrations in the water, but are the result of the integration of many biological, chemical and physical interactions (e.g. bioaccumulation and biomagnification processes, variations in diet and growth rates, and food web changes).

The GLFMSP completed an interlaboratory comparison study of multiple age structures to identify the most appropriate age estimation structure for the Program. The Lake Trout maxilla was selected, over the otolith, as the more precise, accurate, and rapidly assessed structure for the Program when compared between laboratories and against the known age from coded wire tags (CWTs). Age-normalization practices can now be implemented when assessing contaminant concentrations and trends for the GLFMSP (Murphy et. al 2018).

Fish Collection and Program Design

ECCC reports annually on contaminant burdens in similarly aged Lake Trout (4+ through 6+ year range) and Walleye (Lake Erie) as well as in Rainbow Smelt (*Osmerus mordax*), a common forage species. ECCC analyzes and reports chemical concentration on individual fish. The U.S. EPA monitors contaminant burdens in similarly sized Lake Trout (600-700 mm total length) and Walleye (western Lake Erie, 400-500 mm total length) annually from alternating locations by year in each lake. U.S. EPA collects fish in a standard size range and determines individual fish age prior to homogenizing samples into 5 fish composites. Monitoring stations for both ECCC and USEPA are shown in Figure 7. Environment and Climate Change Canada does not collect samples in Lake Michigan, and individual program contaminant lists are not identical (Table 2). Despite these differences in collection and analysis, the interpretation of results and trends by both programs are very similar.

Basin Wide Summary

Since the late 1970s, concentrations of legacy persistent organic pollutants such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (OC_{pest}) in most monitored fish species have declined. Long-term mercury trends show varied results across the basin. Concentrations of mercury continue to be below the target established in the 1987 GLWQA. In general, the levels of regulated compounds are slowly declining or have stabilized in the tissues of Great Lakes top predatory fish. Basin wide, the changes are often lake-specific as they are dependent, in part, on the physio-chemical characteristics of the contaminants, hydrological characteristics of the lake, and the biological composition of the fish community and associated food webs. Despite these declines, concentrations of some compounds, like PCBs, PFOS and PBDEs continue to exceed environmental quality guidelines and/or objectives. As reported in previous iterations of this sub-indicator report, these chemicals are generally on the decline and below environmental targets. It is important to note that both the U.S. and Canadian monitoring programs continue to monitor and report out on these chemicals to track trends and assess the health of the ecosystem. A section on “Other Chemicals of Interest” is included in this sub-indicator to help inform readers of the current state of the science and to aid the Annex 3 process in reviewing other chemicals in the future.

Status of Chemicals of Mutual Concern (CMCs)

The 2012 Great Lakes Water Quality Agreement requires the United States and Canada to identify Chemicals of Mutual Concern (CMCs) that are potentially harmful to human health or the environment and that originate from anthropogenic sources. Pursuant to the Agreement, these substances will be targeted for binational action. In May of 2016, the first group of Chemicals of Mutual Concern was identified through a binationally agreed upon multi-stakeholder process. This effort was led by the Chemicals of Mutual Concern Annex Sub-committee, based on advice from experts from government, industry, environmental non-government organizations, and academia. The following 8 chemicals were identified as the first set of CMCs:

- Polychlorinated Biphenyls (PCBs)
- Mercury (Hg)
- Polybrominated Diphenyl Ethers (PBDEs)
- Perfluorooctanoic acid (PFOA)
- Perfluorooctane sulfonate (PFOS)
- Hexabromocyclododecane (HBCDD/HBCD)
- Short-Chain Chlorinated Paraffins (SCCPs)
- Long-Chain Perfluorinated carboxylic acids (LC-PFCAs)

This toxic chemical sub-indicator reports on Chemicals of Mutual Concern (CMCs) in a fulsome, consistent and transparent way. Information on additional chemicals of interest for the Great Lakes (not CMCs) is valuable for inclusion in the report and will be included in a separate section below, as appropriate, but is not part of the formal assessment.

Total polychlorinated biphenyls (PCBs)

Historically, PCBs were used in many things such as electrical equipment, capacitors, oils, and insulation. They were banned in the U.S. in 1979. In general, total PCB (Arochlor 1254) concentrations in Great Lakes top predator fish have continuously declined since their phase-out in the 1970s (Figure 8) and remain above the GLWQA 1987 Amendment target of 100 ng/g wet weight (ww) (Table 1). Long-term concentration trends are declining for all lakes with significance (Table 3). Short-term concentration trends are generally declining but without significance (Table 4). The exceptions are Lake Huron which is showing a very small, non-significant short term increase and Lake Erie and Michigan which have seen significant declines over the past 10 years. Biological factors such as shifts in food webs, resulting in changes to growth rates of top predator species may be affecting concentrations and trends of PCB. To account for changes in growth rates, the GLFMSP has revised its compositing scheme to be based on age, rather than size, and age normalization is being incorporated into trend publications in peer reviewed literature (Zhou et. al 2018). Age is an important factor when analyzing contaminants in whole fish, since older fish usually have higher contaminant burdens due to longer life exposure. Recent studies have been conducted which compare temporal trends of PCBs in Great Lakes fish and air (Hites and Holsen, 2019). GLNPO also recently published a technical report which evaluates long term trends of PCBs, PBDEs, and Mercury in top predator fish through 2016 (U.S. EPA, 2020). Results from these studies show statistically significant declines in PCB concentrations at each long-term sampling site since 1992.

Since the last State of the Great Lakes (SOGL) sub-indicator report in 2019, PCB concentrations have been measured in Rainbow Smelt collected only from Lake Ontario, although more data are pending. Average total PCB concentrations in composited Rainbow Smelt measured by ECCC were 38.5 ng/g ww in Lake Ontario (2018). ECCC recently changed their lab services provider for PCB analysis in fish tissue to obtain lower detection limits (DL). Prior to 2016, the DL was 50 ng/g ww and thus it is not possible to determine whether the levels of PCBs in Rainbow Smelt in Lake Ontario have increased or decreased since the last assessment.

Total Mercury (Hg)

Mercury is used in chemical production, dental amalgam, fluorescent lighting, batteries, automotive switches and thermometers although some of these uses are being phased out. Concentration trends for total mercury in Great Lakes top predator fish have been variable. Their rates of decline and statistical significance have varied spatially across the basin and between the monitoring programs operated by ECCC and the U.S. EPA over the last few reporting cycles (Figure 9). Concentrations of mercury continue to be below the environmental target of 0.3 mg/kg ww which is set based on the ecological risk of methylmercury to piscivorous fish in the Great Lakes (Sandheinrich et al. 2011). Long-term concentration trends are declining in Lakes Ontario, Superior, Erie and Michigan, with Lake Ontario being the only statistically significant declining trend. There is a slight increase observed in Lake Huron, however this increase was not found to be significant (Table 3). Short term concentration trends are mixed (Table 4). As with PCBs, the GLFMSP is incorporating age and age normalization practices into its trend publications in peer reviewed literature (Zhou et al. 2017).

The GLFMSP has been collaborating with the USGS, Wisconsin Water Science Center, to identify sources of mercury from stable mercury isotope ratios in Great Lakes fish and sediment. Results indicate atmospheric sources dominate in Lakes Huron, Superior, and Michigan sediment while watershed derived and industrial sources dominate in Lakes Erie and Ontario sediment. Comparison of Ω^{200} mercury signatures in predatory fish from the upper Great lakes reveals that bioaccumulated mercury is more similar to atmospherically derived mercury than a lake's sediment. These data suggest that in some cases, atmospherically derived mercury may be a more important source of methylmercury to higher trophic levels than legacy sediments in the Great Lakes (Lepak et al. 2015). Further investigation into distinct mercury signatures driven by both food-web and water-quality characteristics has been completed in GLFMSP samples. Results indicate that sources of mercury, as assessed by stable mercury isotopes, vary by lake and indicate variability in the depth of the euphotic zone, where mercury is most likely incorporated into the food web. Mercury from precipitation, appears both disconnected from lake sedimentary sources and comparable in fish among the five lakes. Similar to the open ocean, water-column methylation occurs in the Great Lakes, possibly transforming recently deposited atmospheric mercury. The degree of photochemical processing of mercury is likely controlled by phytoplankton uptake rather than by dissolved organic carbon quantity among lakes (Lepak et al. 2018). Additional work was also done to identify changes in dietary mercury sources being altered by dreissenid mussel invasions in Lake Michigan. It was identified that mercury concentrations in fish can be affected by shifts in trophic structures and dietary preferences (Lepak et al. 2019).

ECCC completed an assessment of total mercury levels in aquatic bird and fish communities in the Canadian Great Lakes. These monitoring and surveillance programs have assessed chemical concentrations for over 42 years (1974 – 2015). These data (22 sites) were used to examine spatio-temporal variability of mercury levels in Herring Gull (*Larus argentatus*) eggs, Lake Trout, Walleye, and Rainbow Smelt. Trends were quantified with dynamic linear models, which provided time-variant rates of change of mercury concentrations. Lipid content (in both fish and eggs) and length in fish were used as covariates in all models. For the first three decades, mercury levels in gull eggs and fish declined at all stations. In the 2000s, trends for herring gull eggs reversed at two sites in Lake Erie and two sites in Lake Ontario. Similar trend reversals in the 2000s were observed for Lake Trout in Lake Superior and at a single station in Lake Ontario. Mercury levels in Lake Trout continued to slowly decline at all of the remaining stations, except for Lake Huron, where the levels remained stable. A post-hoc Bayesian regression analysis suggests strong trophic interactions between Herring Gulls and Rainbow Smelt in Lake Superior and Lake Ontario, but also pinpoints the likelihood of a trophic decoupling in Lake Huron and Lake Erie. Continued monitoring of mercury levels in Herring Gulls and fish is required to consolidate these trophic shifts and further evaluate their broader implications (Blukacz-Richards et. al 2016).

Similar temporal patterns in mercury concentrations are also observed in Rainbow Smelt, a common forage fish for many fish and birds in the Great Lakes basin (Figure 10. Concentration of mercury in smelt is highest in Lake

Superior and have been highly variable in all lakes since the last report. Continued monitoring of mercury levels in fish from all the lakes is warranted to adequately assess the future risk to wildlife consumers of fish in the Great Lakes basin, especially in areas where levels appear to be increasing.

Polybrominated Diphenyl Ethers (PBDEs)

PBDEs are commonly used in commercial formulations for fire retardants, plastics, consumer electronics, and coatings. In a national survey of PBDE concentrations in top predator fish from lakes across Canada, the highest concentrations were observed in fish from the Great Lakes and > 95% of the PBDE compounds in the fish were tetra-, penta-, or hexa-BDEs (Gewurtz et al. 2011). Concentrations of the tetra BDE mixture (BDE-47) in all five Great Lakes continue to remain below the Federal Environmental Quality Guideline (FEQG) target of 44 ng/g ww (Figure 11). Concentrations of the penta BDE mixture (BDE-99 + 100) in all five Great Lakes continue to remain above the FEQG of 1.0 ng/g ww (Figure 12). Concentrations of the hexa BDE mixture (BDE-153 + 154) in all five Great Lakes are generally at or below the FEQG of 4 ng/g ww (Figure 13. In general, long-term concentration trends for the 3 BDE mixtures are declining, with some small observed increases, with mixed significance (Table 3). In general, short term concentration trends for the 3 BDE mixtures are declining with limited significance (Table 4). As with PCBs, the GLFMSP is incorporating age and age normalization practices into its trend publications in peer reviewed literature.

Recent work on temporal trends in these 5 major PBDE congeners in age normalized Great Lakes fish showed that concentrations increased from 1979 to 2000 with subsequent declines until 2007 in response to phase-outs in production. Since that time the decreasing trends have leveled off and recently have become unchanging, as declines in BDE 47 are offset by increasing trends in the other four BDEs, potentially from the breakdown of higher brominated congeners (Zhou et al. 2019).

Perfluoroctane Sulfonate (PFOS) & Perfluorooctanoic Acid (PFOA)

Perfluorooctane sulfonate (PFOS) is a synthetic substance belonging to a larger class of organic fluorochemicals that are either partially or completely saturated with fluorine. These compounds have been used as water repellants and flame retardants. PFOA is generally not found in Great Lakes whole fish and will not be reported on through this sub-indicator. Concentration trends for PFOS in Great Lakes top predator fish have been variable spatially across the basin and between the monitoring programs (Figure 14). Both long- and short-term concentrations in all 5 Great Lakes remain above the FEQG of 4.6 ng/g ww for mammalian diet. Long and short-term concentration trends, and their significance, are varied across the five lakes (Tables 3 & 4). Recent work has been done to evaluate temporal trends of perfluoroalkyl acids in top predator fish from 2005-2016. Results show that generally analyte concentrations declined in fish basin-wide from 2005 to 2016, however increasing trends were observed at the odd-year GLFMSP sampling site in Lake Superior, and the two GLFMSP sampling sites in Lake Erie (Point et al. 2021).

Hexabromocyclododecane (HBCDD)

HBCDD is a high production flame retardant used mainly in polystyrene foams and is believed to have been used as a replacement alternative to PBDEs. HBCDD was added as a routine analyte to both the U.S. and Canadian monitoring and surveillance programs in 2012 as a result of it being designated as a CMC, however retrospective analysis resulted in data as far back as 2006. Current and retrospective analyses for this compound were completed in both fresh and archived fish tissues to allow for a more robust data set. In general, concentration trends are variable spatially across the basin and are well below the European Union's Environmental Quality Standard (EQS) for fish tissue of 167 ng/g lipid (Figure 15). Long- and short-term concentration trends, and their significance, are varied across the five Great Lakes (Tables 3 & 4).

A recent study of HBCDD in top predator fish (Lake Trout, Walleye, or Brook Trout) from across Canada identified the spatial distribution of HBCDD in 2013 (n = 165) from 19 sampling sites and in 2015 (n= 145) from 20 sites across Canada. HBCDD was measurable in at least one sample at each sampling site regardless of sampling year with the exception of Walleye from the south basin of Lake Winnipeg (2013). Sampling sites in or near the Laurentian Great Lakes had greater ∑HBCDD concentrations compared to locations to the west or east. HBCDD has 16 possible stereo-isomers with different biological activities. The greatest mean ∑HBCDD concentration was 72.6 ng/g lipid weight in fish from Lake Huron–Goderich (2015). Regardless of the sampling sites, α-HBCDD was the dominant congener followed by γ-HBCDD, whereas β-HBCDD was barely detectable. In fish from the same waterbody there were comparable α/γ isomer concentration ratios. The greatest ratio was 20.8 in fish from Lake Ontario, whereas the lowest ratio was 6.3 for fish from Lac Memphrémagog (Québec) likely related to more recent emissions of a technical HBCDD mixture (primarily composed of γ-HBCDD but also contains measurable α- and β-HBCDD). Temporal trends of HBCDD in Lake Trout from Lake Ontario showed a significant decreasing trend for γ-HBCDD with a half-life estimate of 10 years over a 36-year period (1979–2015), and for α-HBCDD with a half-life of 11 years over the years of 2008 to 2015. The proportion of α-HBCDD to ∑HBCDD increased significantly during 1979 to 2015. The study provided novel information on the isomer-specific HBCDDs in Canada freshwater fish. (Su et al. 2018).

Long-term HBCDD trends have been evaluated in recent studies. Researchers noted that recent trends of ∑HBCDD (2004-2016) show concentration increases in Lakes Erie and Ontario in Great Lakes fish. They also observed concentration decreases in Lakes Superior and Michigan and found that concentration in Lake Huron have been generally unchanged during this time period (2004-2016) (Parvizian et al. 2020).

Short-Chain Chlorinated Paraffins (SCCPs) & Medium Chain Chlorinated Paraffins (MCCPs)

Only data from 2010 to 2013 are currently available from the Canadian monitoring and surveillance program; additional data are pending once laboratories re-open post pandemic. These classes of compounds have been used as lubricants and coolants for metal cutting. Concentrations of total SCCPs in lake trout were 7.6 ng/g in Lake Ontario, 4.95 ng/g for Lake Superior and 5.25 ng/g in Lake Huron while concentrations of total SCCPs in walleye from Western Lake Erie were 1.1 ng/g. All were well below the FEQG for SCCPs of 2,700 ng/g lipid. No status or trend information can be provided on these compounds. A previous study from 2015, on the levels of chlorinated paraffins in fish from Canadian lakes, showed that fish from the Great Lakes had higher levels of the medium chain chlorinated paraffins (MCCPs) (C14-C17) than SCCPs (C10-C13) (Saborido Basconcillo et al. 2015). Medium chains may be an important component to monitor and/or research in the Great Lakes. The levels of MCCPs were very similar at approximately 12 ng/g in fish from Lakes Ontario, Erie and Huron and 4 ng/g in Lake Superior. In these same fish, SCCPs were measured to be between 3 and 5 ng/g.

Other Chemicals of Interest

Both the U.S. and Canadian monitoring and surveillance programs have invested in the identification and quantification of non-CMCs and emerging chemicals through the Great Lakes Restoration Initiative and Canada's Chemicals Management Plan. Some, such as dioxins and furans or polychlorinated napthalenes, are considered legacy compounds but may still be at concentrations harmful to fish or consumers of fish, or trends may have changed. There is also a need to survey emerging or unknown contaminants in order to gather enough data to determine if they may be causing harm, or have the potential to become harmful. Summaries of recent publications on identification, quantification and/or methods of non-CMCs and emerging chemicals are provided below. Because many of these chemicals are newly identified, or data have only recently become available, status and trend statements are not possible at this time. However, the authors feel it is important to highlight this work and if warranted, these compounds may appear in future State of the Great Lakes sub-indicator reports. It is also important to note that surveillance for emerging chemicals and continued tracking of legacy chemicals is an essential part of both countries' programs.

Dioxins and Furans (PCDD/F)

Dioxins and furans are primarily unintended byproducts of combustion or industrial processes such as herbicide production or the bleaching of wood pulp. The U.S. and Canadian Fish Monitoring and Surveillance Programs completed a trend concentration assessment of age corrected Lake Trout and Walleye for polychlorinated dibenzo-pdioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and coplanar polychlorinated biphenyls (CP-PCBs) from samples collected between 2004 and 2014. Age–contaminant corrections were developed to accurately report contaminant trends due to significant Lake Trout age structure changes, using the age-trend model (ATM). The ATM used a lake-specific age–contaminant regression to mitigate the effect of a fluctuating Lake Trout age structure to directly improve the log–linear regression models. ATM results indicated that half-life (t1/2) and percent decreases for PCDD/ Fs, CP-PCBs, and toxic equivalence (TEQ) (average −56 to 70%) were fairly uniform and consistent across the Great Lakes over the 2004–2014 period. The vast majority of TEQ associated with all Great Lakes Lake Trout and Walleye samples is due to the non-ortho CP-PCBs (average = 79%) as compared with PCDD/Fs (average = 21%). On average, CP-PCB-126 individually accounted for over 95% of the total CP-PCB TEQ. A retrospective analysis (1977–2014) of 2378-TCDF and 2378-TCDD raw concentrations in Lake Ontario Lake Trout revealed decreases of 94% and 96%, respectively. Tissue residue guidelines for wildlife protection based on Lake Trout and Walleye total TEQ were uniformly exceeded in all the Great Lakes (Pagano et al. 2018).

Novel Halogenated Organic Contaminants

The GLFMSP has developed non-targeted screening methodologies to identify new and emerging chemicals in whole fish samples. Halogenated chemicals were identified using a combination of authentic standards and library spectral matching, with molecular formula estimations provided by exact mass spectral interpretation. Halogenated organic compounds represent one of the largest groups of chemicals found in the environment and have been studied extensively over the past four decades due to their persistent, bioaccumulative, and toxic (PBT) properties. In addition to the halogenated chemicals currently being targeted by the GLFMSP, more than 60 non-targeted halogenated species were identified. Most appear to be metabolites or breakdown products of larger halogenated organics. The most abundant compound class was halomethoxyphenols accounting for more than 60% of the total concentration of halogenated compounds in top predator fish from all five Great Lakes. These results illustrate the need and utility of non-targeted halogenated screening of aquatic systems using this platform (Fernando et al. 2017).

More recently, non-targeted methodologies were used to compare novel halogenated features in Lake Trout between two time periods a decade apart, 2005/2006 and 2015/2016. Greater than 2000 unknown halogenated features were detected. Lake Superior had the lowest number of unknown halogen features while fish from Lake Ontario had the highest number. Most of these halogenated features changed in the Great Lakes between the two time periods, at a rate higher than changes in total PCBs, meaning generally they have decreased at a faster rate (Fakouri et al. 2020).

Additional Per- and Polyfluoroalkyl Substances (PFAS) compounds

Lake Michigan Lake Trout from the GLFMSP were analyzed for polyfluoroalkyl acids using a computer algorithm that would reference compounds against 3570 possible compounds including C4-C10 perfluoro- and polyfluoroalkyl, polyfluorochloroalkyl acids and sulfonates, and potential ether forms from chemical libraries. The results suggested the presence of 30 polyfluorinated chemical formulas which have not been previously reported in the literature. Little is known, at this time, regarding the toxicity of these novel PFAS compounds. As PFAS compounds are identified via this virtual library, and as they are quantified via neat standards (certified reference material), they will be considered for incorporation into the GLFMSP routine analyte list (Baygi et al. 2016).

Polyhalogenated carbazoles (PHCZs)

PHCZs were investigated in archived GLFMSP Lake Trout collected between 2004 and 2016. Median concentrations of Total PHCZs by lake ranged from 54.7 to 154 ng/g lipid weight (6.8-28.0 ng/g wet weight). Dominant congeners included 3,6-dichlorocarbazole, 1,3,6-tribromocarbazole, and 1,3,6,8-tetrachlorocarbazole. The highest Total PHCZs concentrations were found in Lakes Michigan and Ontario fish, followed by Lake Huron, whereas Lakes Erie and Superior fish contained the lowest concentrations. Congener profiles of PHCZs also exhibited spatial variations. After age normalization to minimize fish age influence on bioaccumulation rates, Total PHCZs' concentrations declined significantly over time in all lakes except Lake Erie, with slopes ranging from -10.24% to -3.85% per year. The median toxic equivalent (TEQ) of PHCZs due to their dioxin-like activity was determined to range from 8.7 to 25.7 pg/g lipid weight (lw) in Great Lakes fish. This study provides the first insight into the bioaccumulation and spatiotemporal trends of PHCZs in Great Lakes and suggests the need for further research on this group of chemicals (Wu et al.2018)

Halogenated Flame Retardants

The identification, persistence, accumulation and trophic transfer of 25 polybrominated diphenyl ether (PBDE) congeners, 23 non-PBDE halogenated flame retardants (NPHFRs), 4 polybrominated-diphenoxybenzenes (PB-DiPhOBzs) and 6 methoxylated (MeO–) PB-DiPhOBzs were investigated in predator and prey fish collected in 2010 from sites in Lakes Ontario (n = 26) and Erie (n = 39). Regardless of locations or species, 20 PBDEs and 12 NPHFRs were quantifiable in at least one of the 65 analyzed samples, and polybrominated-1,4- diphenoxybenzenes (PB-DiPhOBzs) and MeO-PB-DiPhOBzs were not detectable in any of analyzed samples. Among the Flame Retardants (FRs), the greatest concentrations were the ∑PBDE, ranging from 1.06 (Rainbow Smelt, Lake Erie) to 162 (Lake Trout, Lake Ontario) ng/g wet weight (ww), which was followed by mean HBCDD concentrations ranging ND to 17.3 (Lake Trout, Lake Ontario) ng/g ww. The remaining FRs were generally not detectable or at sub-ppb levels. In most of cases, FR concentrations in samples from Lake Ontario were greater than those from Lake Erie. Strong and significant positive linear relationships occurred between log-normalized FR concentrations (ww or lw) and ages of the top predator Lake Trout (n = 16, from Lake Ontario), and the estimated FR doubling ages (T2) were 2.9–6.4 years. For Walleye from Lake Erie, significantly positive linear relationships were also observed for some FRs, but the linear relationships generally became negative after FR concentrations were normalized with lipid weight. This study provides novel information on FR accumulation in aquatic organisms, and for the first time, significant positive linear relationships are reported between log-normalized FR concentrations (lw or ww) and ages of Lake Trout from the Great Lakes (Su et al. 2017).

Spatial and temporal trends in alternative flame retardants, including 18 dechlorane analogues and 20 alternative brominated FRs (ABFRs) were investigated in mega-composites of 50 Lake Trout or walleye from the Great Lakes, between 2004 and 2016. Concentration of sum dechlorane analogues ranged from 0.33 to 31.9 ng/g lipid weight, with highest concentrations in Lake Ontario and profiles there may be indicative of point sources. Other ABFRs measured included hexabromobenzene, pentabromotoluene, and tetrabromo-o-chlorotoluene. The total ABFRs ranged from 0.91 to 54.7 ng/g lipid weight, and were lowest in Lake Erie. The concentration of total dechloranes and total ABFRs declined in all lakes, with the exception of Lake Erie (Wu et al 2019).

Polychlorinated Naphthalenes (PCNs)

Polychlorinated naphthalenes (PCNs) are legacy contaminants, produced primarily as flame retardants and dielectrics until phased-out in Europe and North America in the 1970s. PCNs have been studied in whole fish, herring gull eggs, and sediment. Spatial and temporal trends (1979–2013) of PCN concentrations were studied throughout the Great Lakes and St. Lawrence River, whereas sediments were analyzed for 2011–2013 only. For both fish and gull eggs, concentrations of PCNs were highest in western Lake Erie (7660 & 3020 pg/g ww respectively), and declined downstream to St. Lawrence River (range: 34–2370 pg/g ww). For sediments, concentrations were highest in suspended sediments from the Detroit River (264,000 pg/g), and were lower in surficial sediments downstream to the St. Lawrence River (range: 440–19,300 pg/g). PCNs declined at all sites from ~1980 to 1995, but in Lake Erie concentrations of PCNs increased in gulls and fish from 1995 until 2005. The resurgence in PCNs in biota corresponded to the timing of remedial dredging of sediment highly contaminated with PCNs in the Detroit River, and the effects of this dredging appeared to be manifested downstream to Lake Ontario. Congener profiles of PCNs differed between Lake Erie and Lake Ontario until post-dredging, where PCN profiles of fish in both lakes became increasingly more similar. PCNs in gull eggs were mostly hepta-PCNs, whereas fish had higher concentrations of lower chlorinated PCNs. Patterns of PCNs in gulls and fish appear to be influenced by differences in not only routes of exposure and differential metabolic ability, but also resuspension of PCN contaminated sediments (McGoldrick et al. 2018).

Recent trends analysis for PCNs were performed from 2004-2018 in Lake Trout and Walleye in each of the Great Lakes. Trends generally showed significant decreases in total PCNs over this period, except for fish from Lake Erie, where concentrations increased. Total PCN concentrations in Great Lakes fish ranged from 5701 and 100 pg/g ww. In addition, researchers observed a prominent PCN concentration trend break point in Lake Ontario lake trout over the 2012-2016 period likely associated with hazard waste clean ups, channel dredging and spoil disposal in Detroit River and the western basin of Lake Erie (Pagano and Garner, 2021).

Substituted Diphenylamine Antioxidants (SDPAs)

SDPAs and benzotriazole UV stabilizers (BZT-UVs) are industrial additives of emerging environmental concern. However, the bioaccumulation, biomagnification, and spatial distribution of these contaminants in the Great Lakes are unknown. A 2018 study addressed these knowledge gaps by reporting SDPAs and BZT-UVs in Herring Gull eggs, Lake Trout, and their food web in the Great Lakes for the first time. Herring Gull eggs showed much higher detection frequency and concentrations of target SDPAs and 2-(2H-benzotriazol-2- yl)-4,6-di-tert-pentylphenol (UV328) than that of the whole-body fish homogenate. Interestingly, the predominant SDPAs in Herring Gull eggs were dinonyl- (C9C9) and mononoyl-diphenylamine (C9) which were previously shown to be less bioaccumulative than other SDPAs in fish. In contrast, dioctyl-diphenylamine (C8C8) was the major SDPA in Lake Trout, and biodilution of C8C8 was observed in a Lake Superior Lake Trout food web. Such variations in Herring Gull eggs and fish indicate the differences in accumulation and elimination pathways of SDPAs and BZT-UVs and require further clarification of these mechanisms (Lu et al. 2018).

Linkages

Contaminant levels in Lake Trout and Walleye are dependent on complex biological and physiochemical interactions both within and outside of the Great Lakes basin as these apex predators integrate contaminant inputs from water, air, sediment, and their food sources. A changing climate and associated changes to precipitation and wind currents will alter the influx of contaminants from sources outside of the basin and may alter food webs and the contaminant transfer through them. Aquatic invasive species also alter food webs and change energy and contaminant dynamics in the lakes. They also may introduce new pathways by which sediment contaminant pools could be mobilized and transferred to fish. Many new contaminants of concern are components of consumer products, personal care products, or pharmaceuticals. As a result, wastewater treatment effluents are an important source of contamination which is increasing along with the human population in the basin. Inferences on the effects of invasive species and climate change on the accumulation of contaminants in aquatic biota is not well understood. However, changes in contaminant trends in whole fish, as they relate to changes in the ecosystem, are considered in interpreting results.

It is important to note how chemical concentrations are behaving in the environment in various media. For example, a recent publication from the US Air and Fish Monitoring programs recently evaluated reported levels of PCBs and DDTs over time and calculated the rates at which age adjusted concentrations have decreased over time for comparison. In general, the halving times (9–17 years for PCBs and 7–10 years for DDTs) estimated from the full fish dataset are similar to those estimated from the atmospheric data, suggesting that the atmospheric and the fish levels are coupled. The more recent, age-adjusted rates are sometimes significantly faster than those from the full fish and atmospheric datasets, suggesting that the air-water dynamic may now be changing (Hites et al. 2018). These evaluations are important both to confirm observed trends across media but also to identify how chemicals can behave differently across media for a more accurate depiction of the health of the Great Lakes total environment.

Changes in the food webs of the Great Lakes are becoming much more important to understand and quantify. For example, the decline in age and older Alewife (*Alosa pseudoharengus*) abundance in Lake Huron is primarily responsible for slower growing Lake Trout (He et al. 2016), resulting in higher chemical concentrations of Persistent Bioaccumulative Toxics in the Lake Trout. Prior to the complete collapse of the Alewife population during 2003-2004, Alewives had been the mainstay of the diet of Lake Trout in Lake Huron. The decline in abundance of age and older Rainbow Smelt, which represented another component of Lake Trout diet, may have also contributed to the decrease in Lake Trout growth and condition. Food web assessments for chemical transfer fatty acid content, fatty acid ratios, and stable carbon and nitrogen isotopes are ongoing in the U.S. to assist with the interpretation of chemical results and trends.

Climate change may have an impact on contaminant body burdens in Great Lakes fish, but the effects are likely to be indirect with a changing climate accelerating or exacerbating existing processes. Increased temperatures may change the movement of contaminants within the ecosystem, their transformation into more harmful compounds and their availability to aquatic biota (Noyes et al 2009). For instance, mercury body burdens in fish may be impacted by warmer temperatures, fluctuating water levels and increased run-off that may effect the inputs of mercury to lakes, the methylation of mercury in the aquatic system and the rate of mercury uptake in fish (Grieb et al 2019). Climate warming may also increase or change the impact of contaminants to fish. As fish are exothermic animals, their ability to metabolize contaminants may change with increasing water temperatures, which may reduce or increase the toxicity of the compound, depending on the mechanism of action. For instance the toxicity of some pesticides has been shown to increase with temperature in fish (Noyes et al. 2009).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report		X		
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://cdx.epa.gov/ https://open.canada.ca/data/		

Data Limitations

This sub-indicator relies on U.S. and Canadian government monitoring programs for the majority of the information reported. The list of contaminants which require monitoring continues to grow in number which can be challenging for operating budgets and staffing resources. These limitations will require changes to the frequency and intensity of monitoring of some compounds as newer compounds are added to the priority list. As monitoring priorities for chemicals have shifted over time, not all CMCs have been measured in every lake, in every year or over the entire time span of ECCC and USEPA monitoring and surveillance programs. It is important to note that the ten-year time trend assessment may not be long enough to detect trends, especially when it is compared to long-term trend assessments. Trend detection may be more difficult if there are insufficient staffing or budgetary resources to measure CMCs in every lake, each year.

Additional Information

Reductions in contaminant levels in whole fish will reflect environmental change in the overall water quality of the Great Lakes. Reductions in contaminant loading with subsequent reductions in the concentration of contaminants in the water will pose less risk of harm to fish communities and fish-eating wildlife.

Identification of and revisions to environmental targets and guidelines will result in revised MDR development for status and trend evaluation. Additionally, the incorporation of new CMCs, should this be applicable, and additional existing CMC trend data will affect how the MDR is developed and interpreted.

Ancillary data collections have been established that include fish age, length, weight, sex, and lipid content of the fish collected. Detailed data on sample size, location and the complete suite of ancillary measurements is available to be sourced.

Environmental specimen banks containing tissue samples are a key component of both the U.S. and Canadian monitoring programs, allowing for retrospective analyses of newly identified chemicals of concern to be able to develop long-term trends in the short-term.

The authors have made efforts to improve the statistical rigor of this sub-indicator report through the inclusion of error bounds on estimated concentrations and trends through time. The authors have also focused on contaminants with defined environmental targets, guidelines and/or thresholds to put observed concentrations in context with risk to the environment. Other improvements to statistical rigor, such as, better methods to characterize dataset with censored values (i.e. non-detects) should be investigated and incorporated in future reports on this sub-indicator.

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Information Sources

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Data available upon request from authors or U.S. data can be accessed at: http://www.epa.gov/cdx/

Some Canadian data can be accessed at: https://open.canada.ca/data/

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Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

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Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

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Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

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Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

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Source: Environment and Climate Change Canada, U.S. Environmental Protection Agency

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Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie open dots) collected from each of the Great Lakes.

Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

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Table 1. Contaminant criteria for environmental monitoring and surveillance programs.

Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Source: Canadian Federal Environmental Quality Guidelines - https://www.canada.ca/en/health-canada/services/chemical-substances/fact-sheets/federal-environmental-quality-guidelines.html
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European Union Environmental Quality Standards - https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32013L0039

Contaminant	Criteria Source	Value (ng/g ww)
Sum-tetrabrominated diphenyl ethers (TeBDE) (BDE47)	FEQG* (wildlife diet)	44
Sum-pentabrominated diphenyl ethers (PeBDE) (BDE 99 + 100)	FEQG* (fish tissue)	1
Sum-hexabrominated diphenyl ethers (HxBDE) (BDE 153+154)	FEQG* (wildlife diet)	4
Perfluorooctane sulphonate (PFOS)	FEQG* (mammalian diet)	4.6
Total polychlorinated biphenyls (PCBs)	GLWQA 1987 Amendment	100
Sum DDT (DDT+DDD+DDE)	GLWQA 1987 Amendment	1000.0
Short Chain Chlorinated Alkanes (SCCA)	FEQG* (ng/g lipid)	2700.0
Total Hexabromocyclododecane (HBCDD)	EQS European Union (EQS)	167
Total Mercury	Sandeinrich et al. 2011 (LOER)	300

Contaminant Criteria for Environmental Monitoring and Surveillance

* Canadian Federal Environmental Quality Guidelines

Table 2. Chemicals detected greater than 10 % frequency identified through monitoring and surveillance programs. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Compound or Class	Agency		Great Lake				
	Environment and Climate Change Canada	US EPA	Ontario	Erie	Huron	Michigan	Superior
Hexabromocyclododecane (α-, γ-HBCD)	X	X	X	X	X	X	X
Polychlorinated naphthalenes (PCN)[1]	X	X	X	X	X	X	X
tris(2-butoxyethyl) phosphate (TBOEP)	X		X	X			
Chlorinated alkanes (short and medium chain)	X		X	X	X		X
Decamethylcyclopentasiloxane (D5)	X		X	X	X		X
Dodecamethylcyclohexasiloxane (D6)	X		X	X	X		X
Dodecamethylpentasiloxane (L5)	X		X	X	X		X
Hexamethylcyclotrisiloxane (D3)	X		X	X	X		X
Octamethylcyclotetrasiloxane (D4)	X		X	X	X		X
Perfluorooctane sulfonamide (PFOSA)	X		X	X	X		X
Perfluorooctanoic acid (PFOA)	X		X	X	X		X
Chlordane (α-, γ-)	X	X	X	X	X	X	X
Dieldrin	X	X	X	X	X	X	X
Heptachlor epoxide	X	X	X	X	X	X	X
Hexachlorobenzene (HCB)	X	X	X	X	X	X	X
Hexachlorocyclohexane (α-, γ-HCH)	X	X	X	X	X	X	X
Mercury	X	X	X	X	X	X	X
Mirex	X	X	X	X	X	X	X
p,p'-dichlorodiphenyldichloroethane (DDD)	X	X	X	X	X	X	X
p,p'-dichlorodiphenyldichloroethylene (DDE)	X	X	X	X	X	X	X
p,p'-dichlorodiphenyltrichloroethane (DDT)	X	X	X	X	X	X	X
Perfluorodecanesulfonate (PFDS)	X	X	X	X	X	X	X
Perfluorodecanoic acid (PFDA)	X	X	X	X	X	X	X
Perfluorododecanoic acid (PFDoA)	X	X	X	X	X	X	X
Perfluorononanoic acid (PFNA)	X	X	X	X	X	X	X
Perfluorooctanesulfonate (PFOS)	X	X	X	X	X	X	X
Perfluorotridecanoic acid (PFTrA)	X	X	X	X	X	X	X
Perfluoroundecanoic acid (PFUnA)	X	X	X	X	X	X	X

Polybrominated diphenyl ethers (PBDE)[1]	X	X	X	X	X	X	X
Polychlorinated biphenyls (PCB)[2]	X	X	X	X	X	X	X
Endrin		X	X	X	X	X	X
cis-nonachlor		X	X	X	X	X	X
Endosuflan (I, II)		X	X	X	X	X	X
Endosulfan sulfate		X	X	X	X	X	X
Hexachlorocyclohexane (β-, δ-HCH)		X	X	X	X	X	X
Octachlorostyrene		X	X	X	X	X	X
Oxychlordane		X	X	X	X	X	X
Total Dioxin TEQ (Mammal)		X	X	X	X	X	X
Toxaphene (Camphechlor)		X	X	X	X	X	X
trans-nonachlor		X	X	X	X	X	X

Table 3 Sen's slope and p-values from Mann-Kendall trend analysis of long term concentrations of CMCs in fish from each of the Great Lakes. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

Mercury	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	0.950	0.503	0.069	0.650
Sen's slope (%)	-0.52%	0.01%	-0.08%	-0.31%	-0.15%

LEGEND

P value	**Significant**	Not Sig
Sen's slope (%)	Increasing	Decreasing

HxBDE	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	0.075	0.974	**0.013**	0.112
Sen's slope (%)	-1.73%	-0.88%	0.13%	-2.90%	-0.84%

PeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	**0.004**	0.673	**<0.001**	**0.005**
Sen's slope (%)	-2.12%	-1.29%	-0.29%	-3.29%	-2.89%

TeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	**<0.001**	0.347	**0.006**	**<0.001**
Sen's slope (%)	-3.23%	-4.02%	-0.61%	-2.60%	-5.08%

HBCDD	Ontario	Huron	Erie	Superior	Michigan
P value	0.137	0.101	0.324	**0.004**	**0.038**
Sen's slope (%)	-0.68%	-0.02%	-0.91%	-3.77%	1.26%

PFOS	Ontario	Huron	Erie	Superior	Michigan
P value	0.583	**0.005**	0.631	0.584	0.917
Sen's slope (%)	-0.26%	-1.77%	0.74%	1.13%	-1.07%

PCB	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Sen's slope (%)	-3.11%	-2.74%	-1.31%	-2.11%	-2.43%

Table 4 Sen's slope and p-values from Mann-Kendall trend analysis of short term (10yr) concentrations of CMCs in fish from each of the Great Lakes. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

Mercury	Ontario	Huron	Erie	Superior	Michigan
P value	0.592	0.858	0.503	**0.049**	0.858
Sen's slope (%)	0.15%	-0.53%	-0.08%	-2.05%	0.38%

LEGEND

P value	**Significant**	Not Sig	
Sen's slope (%)	Increasing	Decreasing	Unchanged

HxBDE	Ontario	Huron	Erie	Superior	Michigan
P value	0.592	**0.049**	0.474	**0.020**	0.858
Sen's slope (%)	-0.36%	-0.02%	-0.54%	-4.01%	-0.14%

PeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	0.210	0.283	0.592	**0.020**	0.592
Sen's slope (%)	-1.58%	-1.60%	-0.75%	-5.61%	0.57%

TeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	0.107	**0.007**	0.152	0.074	0.152
Sen's slope (%)	-2.21%	-5.44%	-1.91%	-5.18%	-2.75%

HBCDD	Ontario	Huron	Erie	Superior	Michigan
P value	**0.032**	0.283	0.371	**0.049**	0.592
Sen's slope (%)	-1.60%	-3.37%	-4.14%	-4.98%	0.75%

PFOS	Ontario	Huron	Erie	Superior	Michigan
P value	1.000	0.152	1.000	0.592	0.371
Sen's slope (%)	0.78%	-4.43%	-0.06%	1.17%	5.44%

PCB	Ontario	Huron	Erie	Superior	Michigan
P value	0.474	0.592	**0.049**	0.152	**0.020**
Sen's slope (%)	-1.60%	0.54%	-3.43%	-4.14%	-3.08%

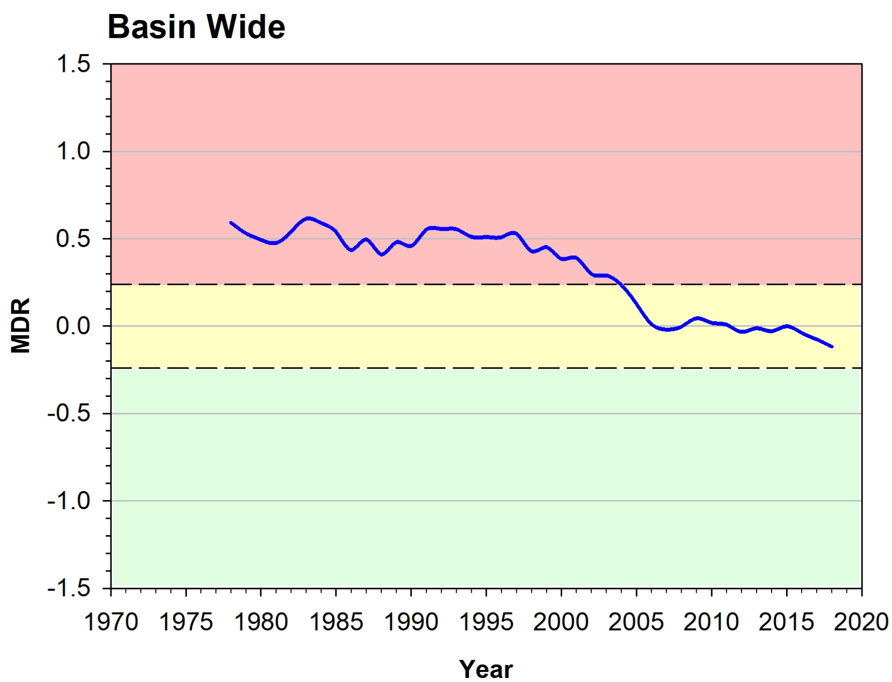


Figure 1. Mean Deviation Ratio (MDR) values (blue line) for the Great Lakes basin. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

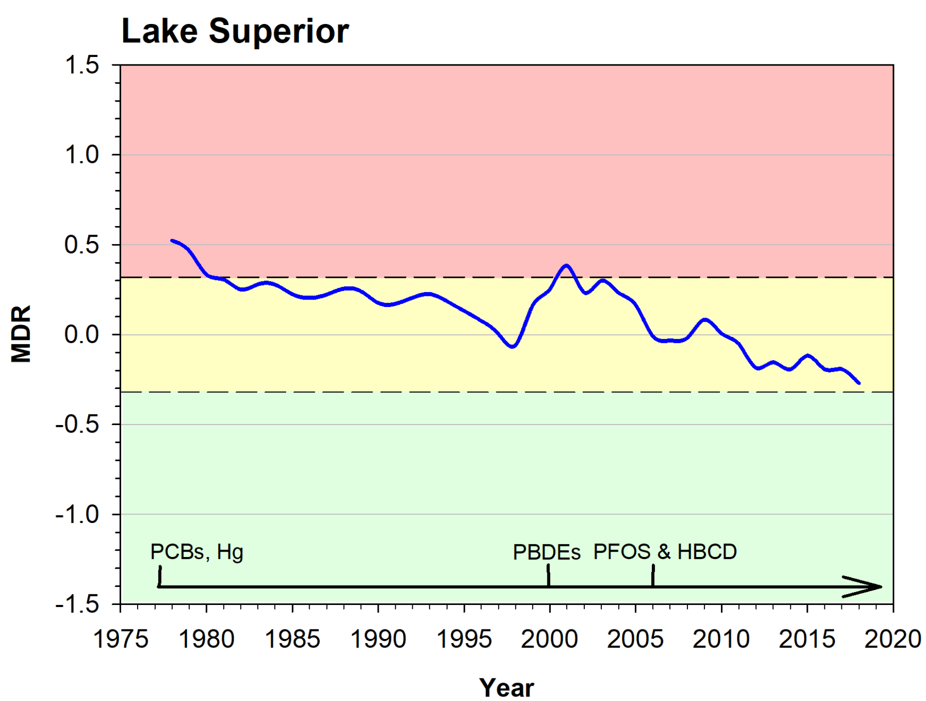


Figure 2. Mean Deviation Ratio (MDR) for Lake Superior. The timeline indicates the year in which different contaminants were added to the MDR calculation. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

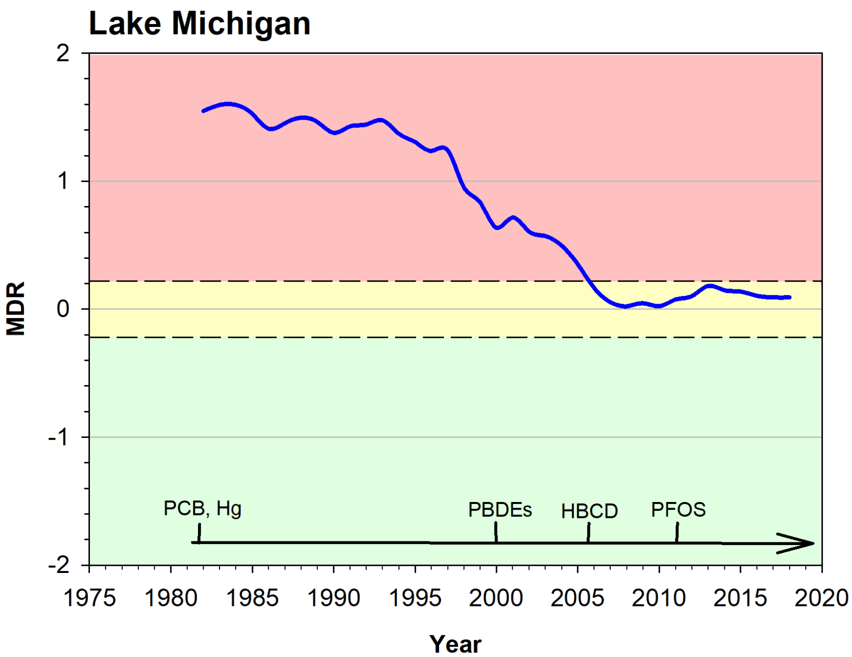


Figure 3. Mean Deviation Ratio (MDR) for Lake Michigan. The timeline indicates the year in which different contaminants were added to the MDR calculation. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

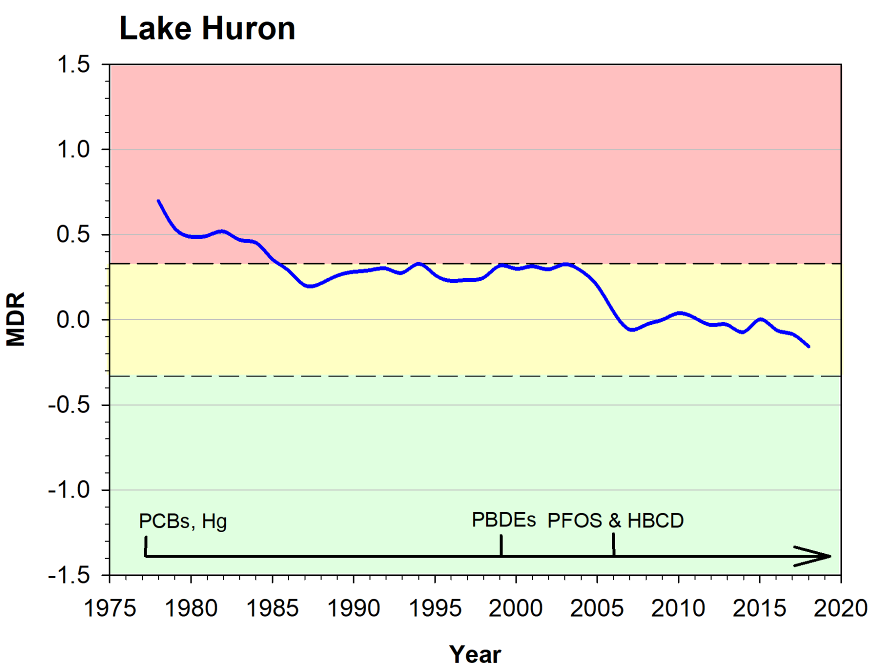


Figure 4. Mean Deviation Ratio (MDR) for Lake Huron. The timeline indicates the year in which different contaminants were added to the MDR calculation. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

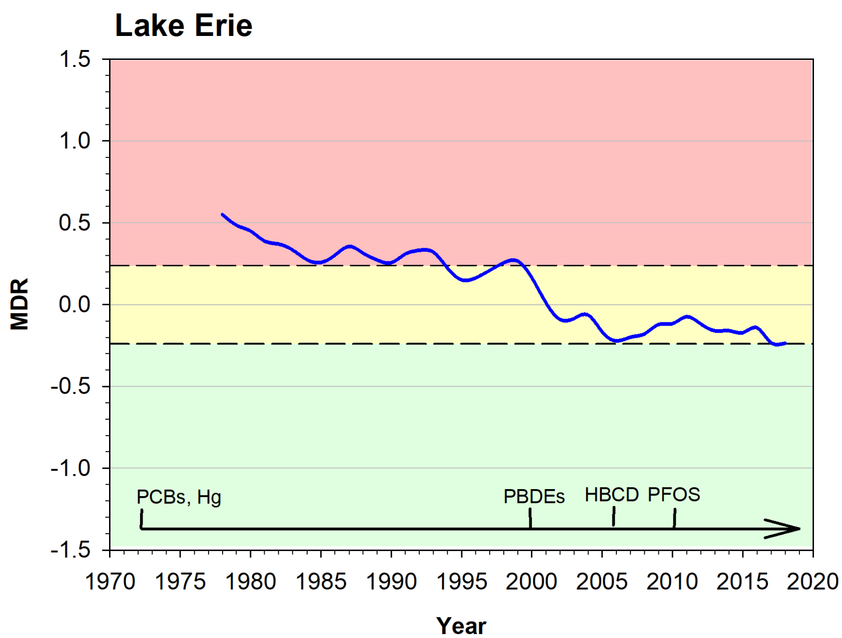


Figure 5. Mean Deviation Ratio (MDR) for Lake Erie. The timeline indicates the year in which different contaminants were added to the MDR calculation. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

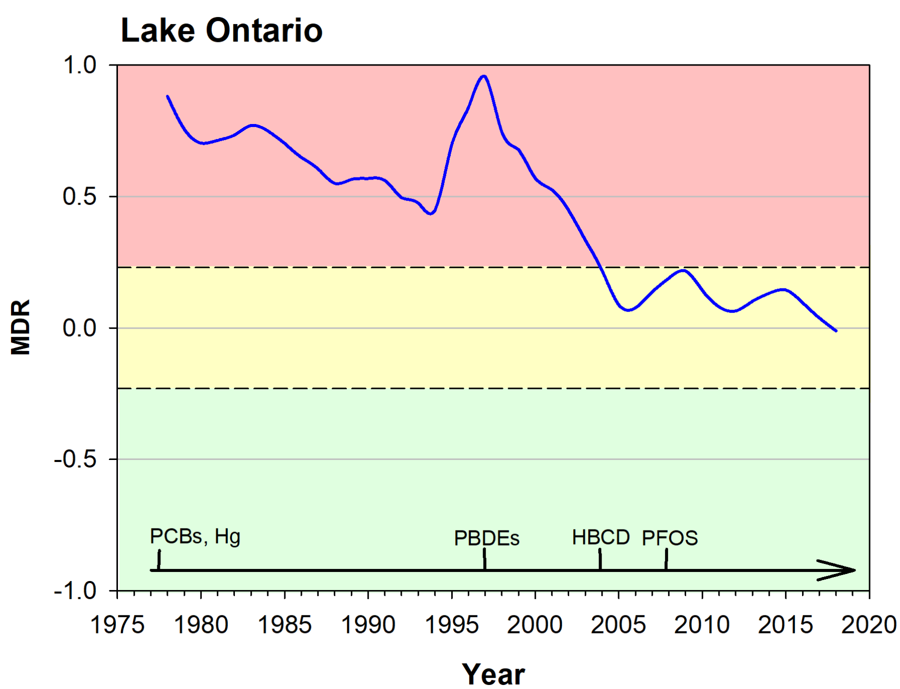


Figure 6. Mean Deviation Ratio (MDR) for Lake Ontario. The timeline indicates the year in which different contaminants were added to the MDR calculation. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

Great Lakes Fish Monitoring and Surveillance Program Collection Sites

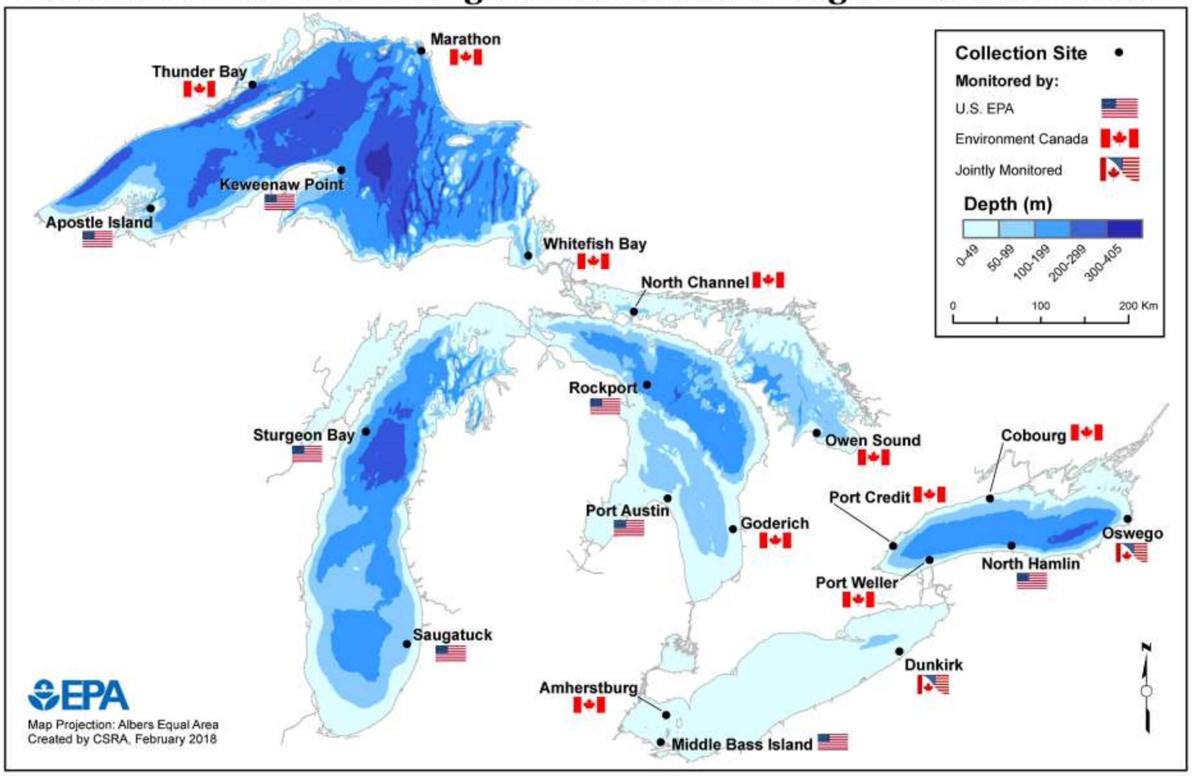
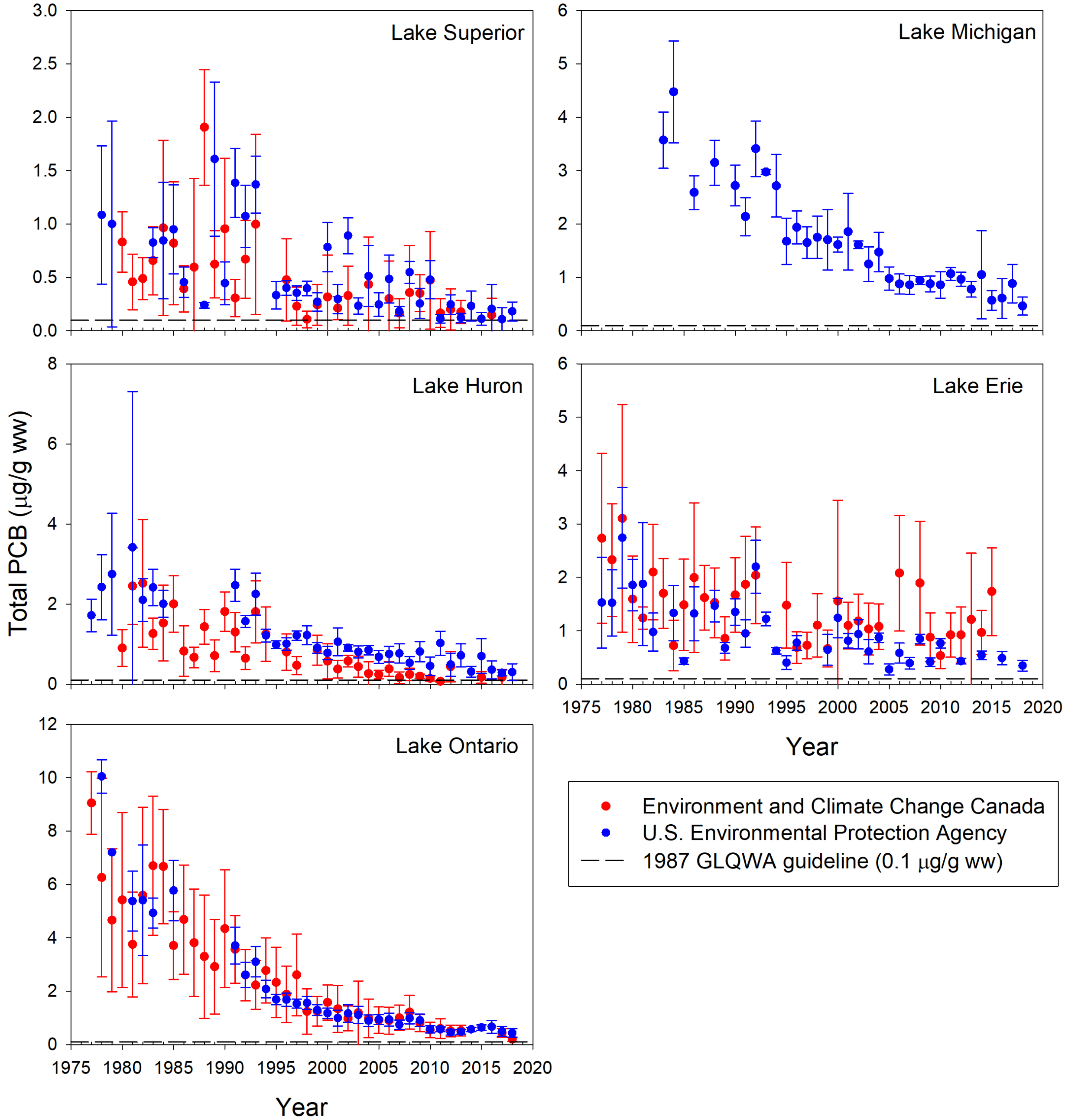


Figure 7. Map of Great Lakes showing Environment and Climate Change Canada and U.S. Environmental Protection Agency monitoring stations for fish contaminants. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.



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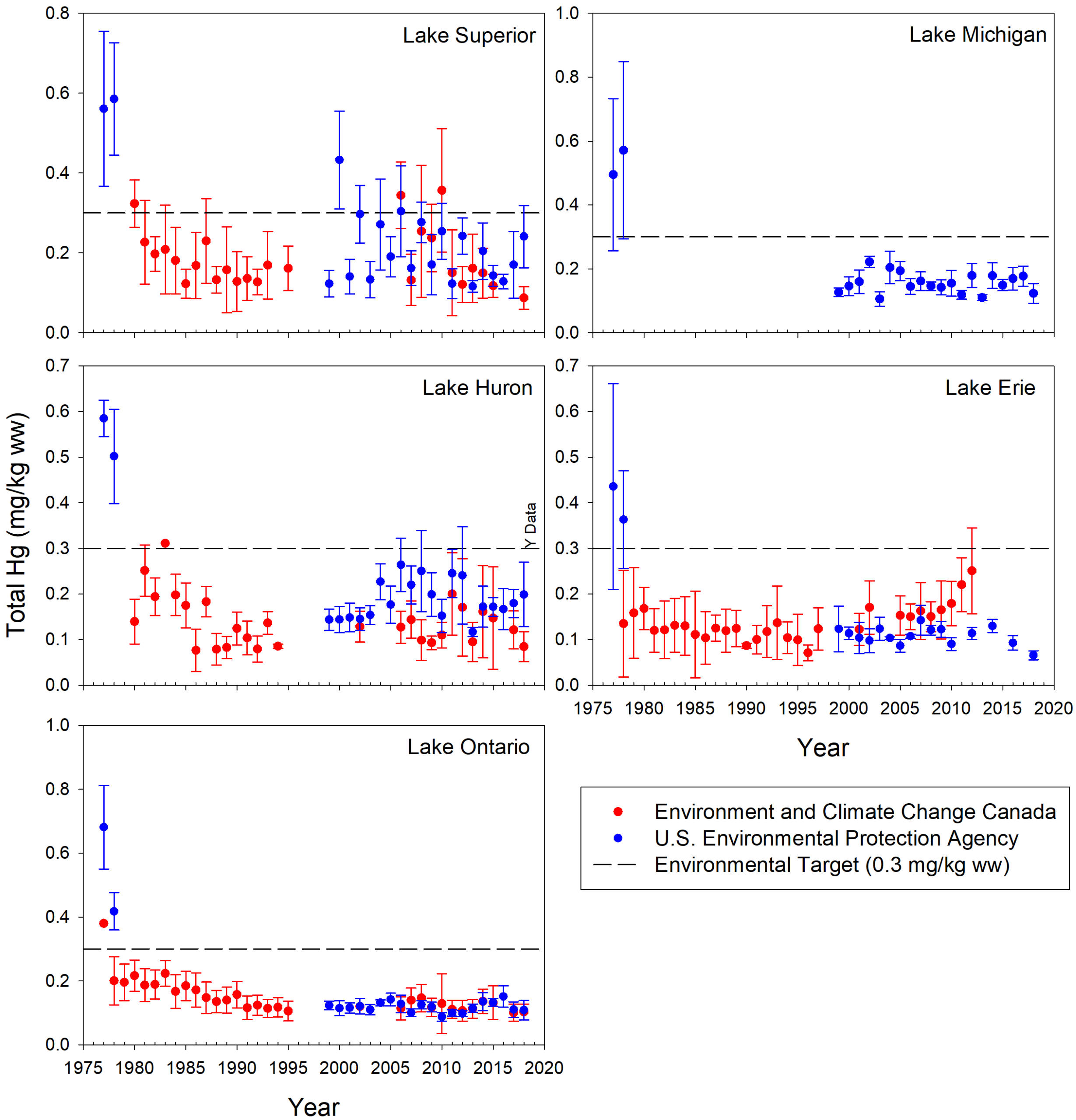

Figure 9. Mean annual total mercury concentrations (wet weight) for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie, EPA only, western basin) collected from each of the Great Lakes. Bars represent standard deviations around the mean. Source: Environment and Climate Change Canada, U.S. Environmental Protection Agency.

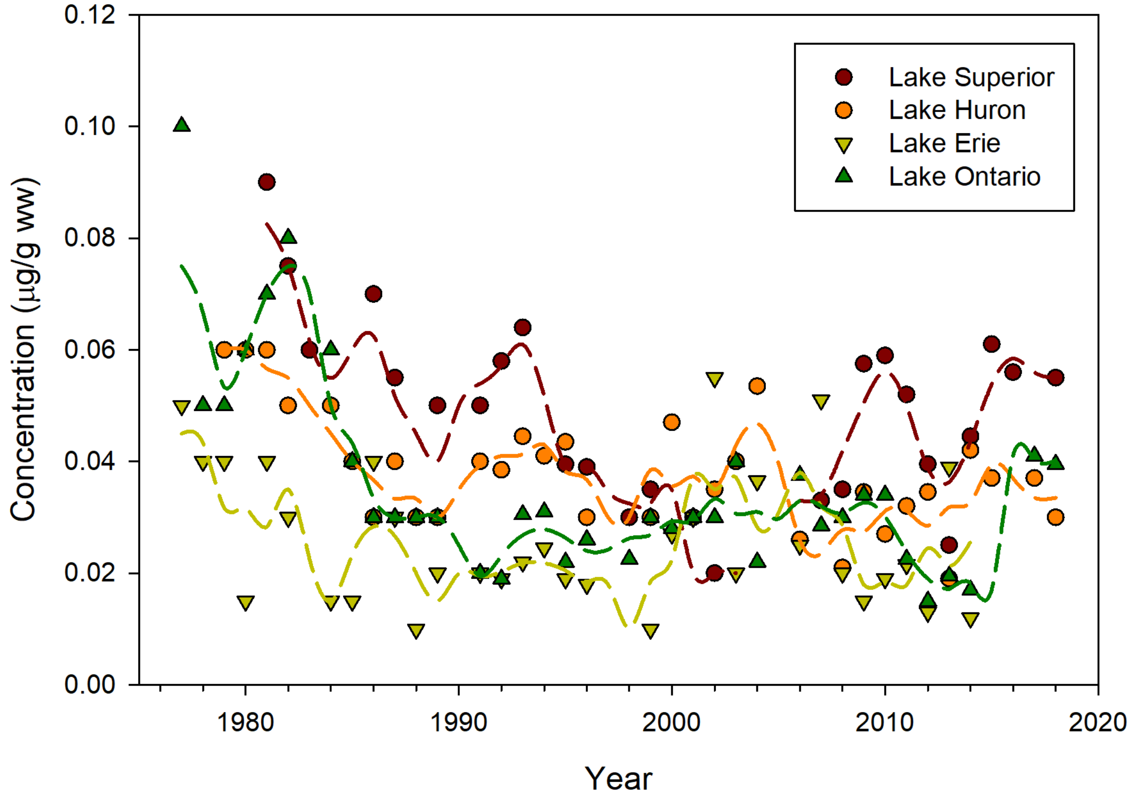

Figure 10. Average concentrations (wet weight) of total mercury (dots) measured in composite samples of Rainbow Smelt by Environment Canada. Lines show the three-year moving average. Source: Environment and Climate Change Canada.

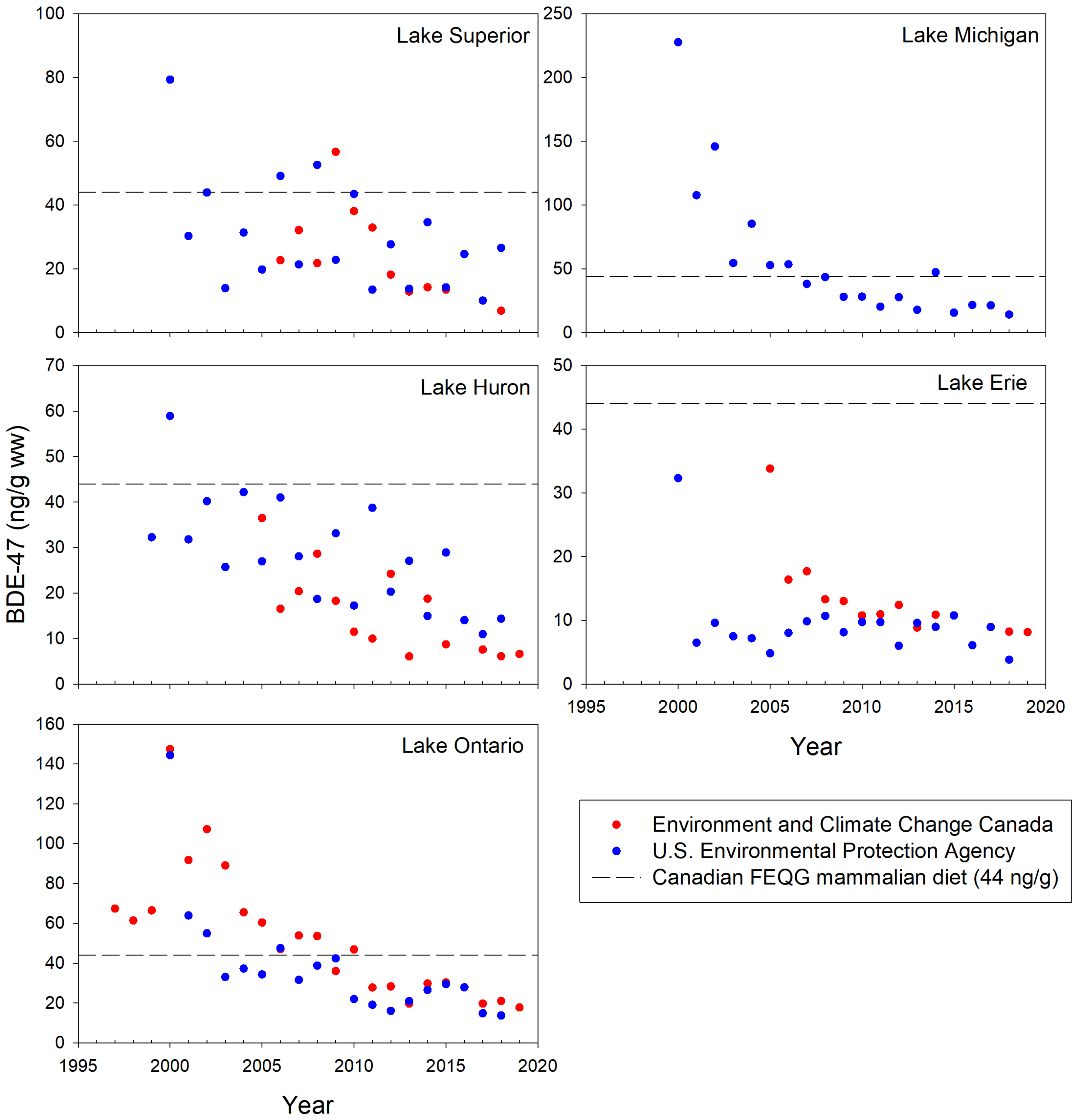

Figure 11. Mean annual tetra BDE-47 concentrations (weight weight) for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie, EPA only, western basin) collected from each of the Great Lakes. Canadian guidelines for mammalian diet are given for context. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

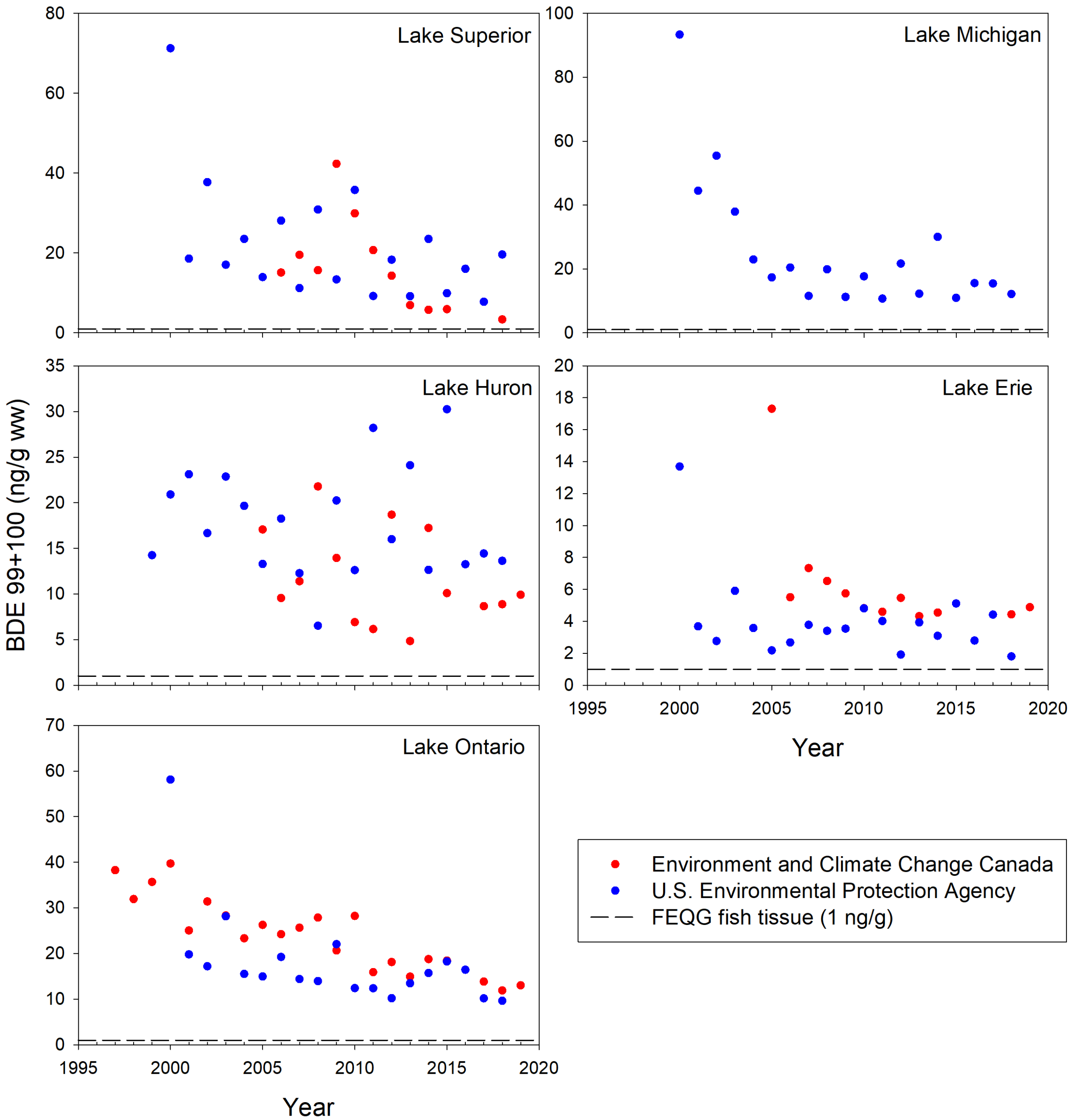

Figure 12. Mean annual penta BDE concentrations (wet weight) for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie, EPA only, western basin) collected from each of the Great Lakes. Canadian guidelines for fish tissue are given for context. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

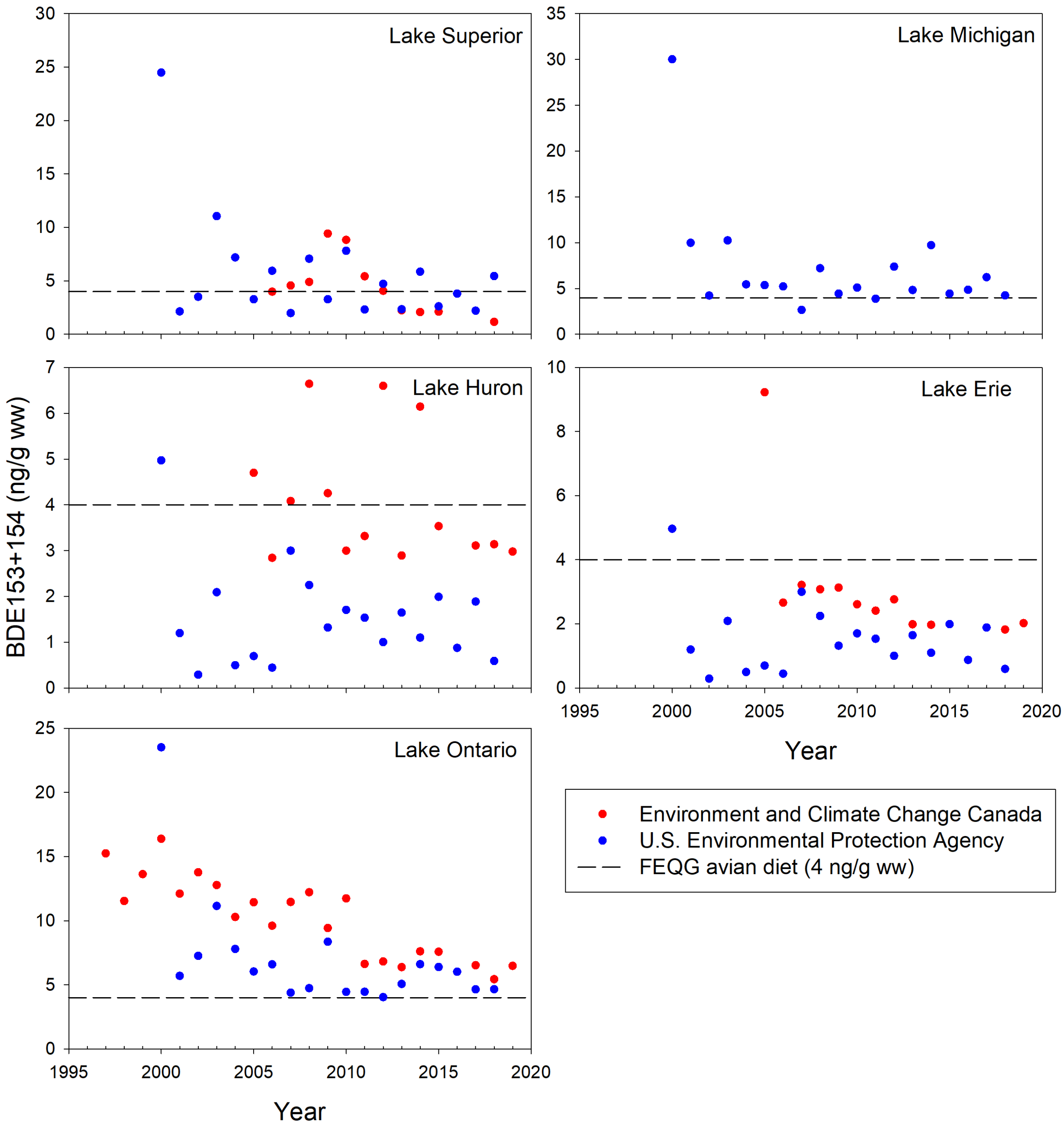

Figure 13. Mean annual HxBDE concentrations (wet weight) for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie, EPA only, western basin) collected from each of the Great Lakes. Canadian guidelines for avian diet are shown for context. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

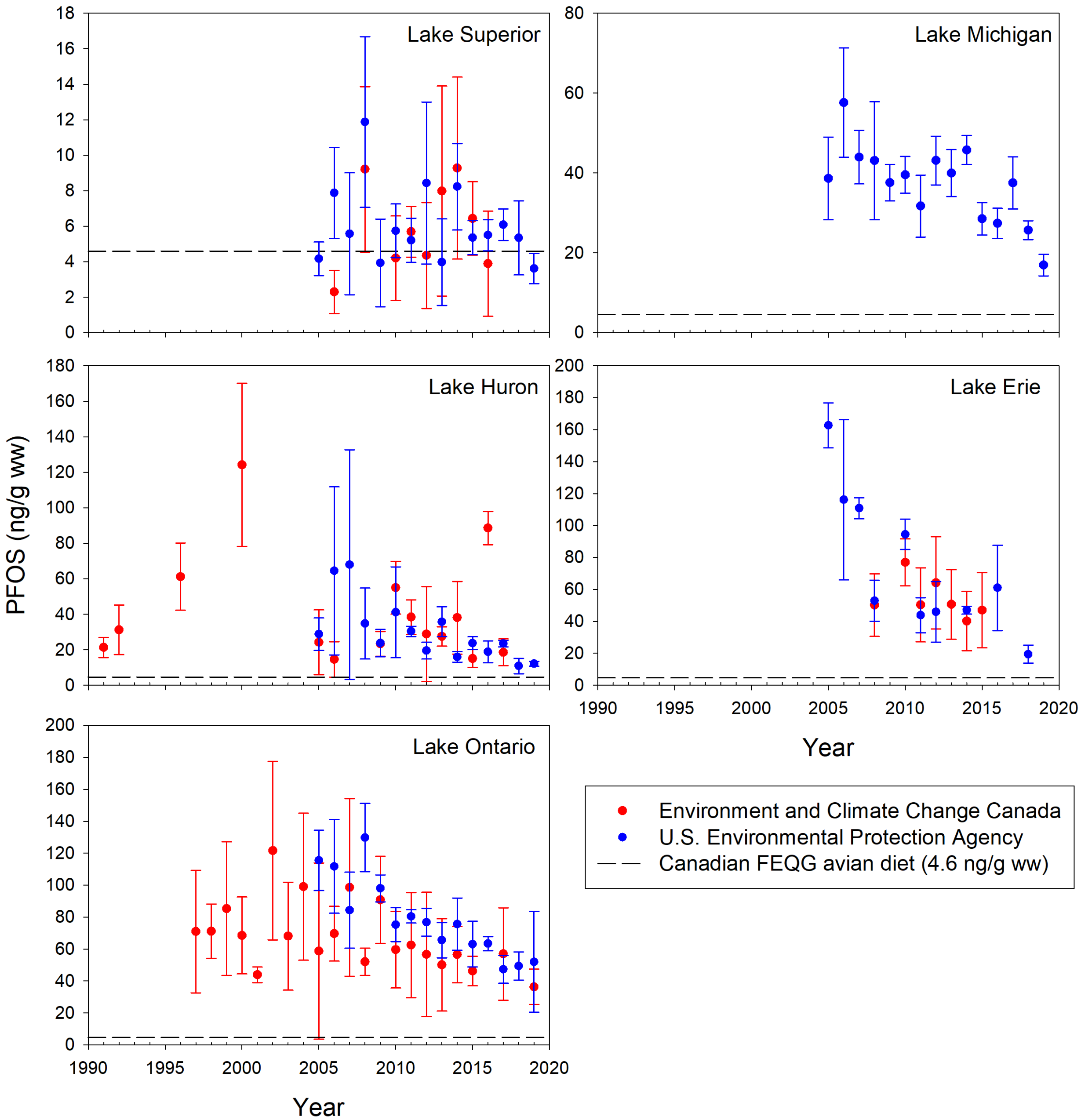

Figure 14. Mean annual PFOS concentrations (wet weight) for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency, Point et al. 2021.) whole body Lake Trout or Walleye (Lake Erie, EPA only, western basin) collected from each of the Great Lakes. Bars represent standard deviations around the mean. Canadian guidelines for avian diet are shown for context. Source: Environment and Climate Change Canada, U.S. Environmental Protection Agency.

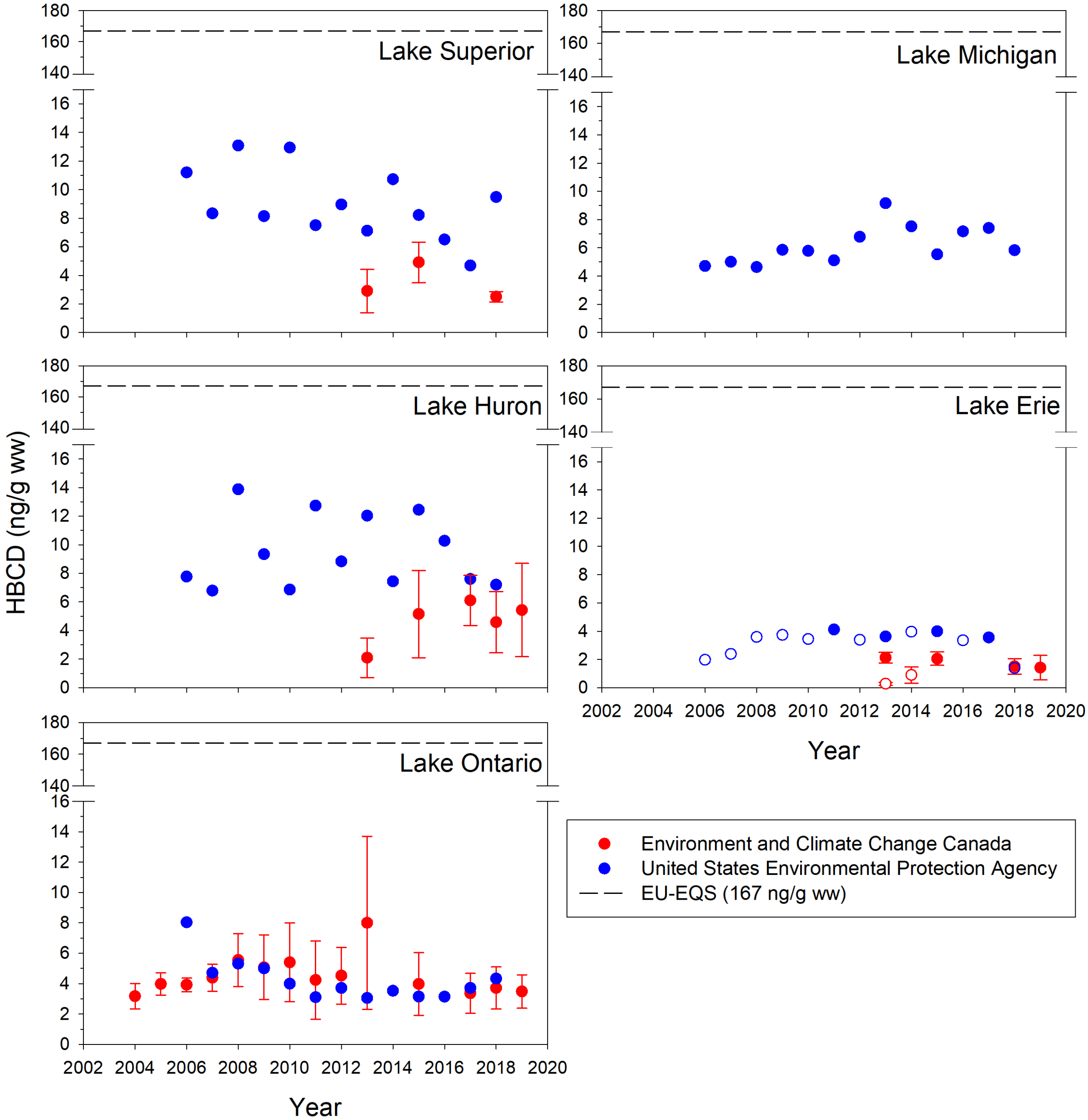

Figure 15. Mean annual total HBCD concentrations (wet weight) for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie open dots) collected from each of the Great Lakes. Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

Sub-Indicator: Toxic Chemicals in Herring Gull Eggs

Note: Due to the COVID-19 pandemic, restrictions in laboratories have delayed the analysis of some 2018 and most 2019 samples and there was limited data collection during the 2020 field season. However, due to the tendency of these data to not vary significantly over short time periods, it is the author's best professional judgement that the assessments presented in the 2019 Toxic Chemicals in Herring Gull Eggs sub-indicator report are still relevant in 2022. For this reason, the 2019 assessments for this sub indicator have been extended through 2022 for use in the Toxic Chemicals Indicator suite.

Overall Assessment

Status: Good

Trends:

10-Year Trend (2002-2017): Improving

Long-term Trend (1974-2017): Improving

Rationale: The long-term trends (1974 to present) of virtually all legacy contaminants (Polychlorinated Biphenyls-PCBs, dioxins and furans, organochlorine pesticides-OCs) are declining. Using a Wildlife Contaminant Index (WCI) based solely upon legacy organics, contaminant concentrations in most colonies of Herring Gulls have generally improved from 2002 to 2017, with a few exceptions. On a lake-wide basis, Lake Erie was an exception, as the WCI did not improve from 2002 to 2017. Most of the WCI scores were driven by PCBs in all lakes, with a smaller but roughly equal contribution of PCDD/Fs and Dichlorodiphenyltrichloroethane -DDT, although the relative contribution of PCDD/Fs was greater in some colonies in the United States compared to ones in Canada. The short-term trends of legacy contaminants (PCBs, OCs, mercury) over the last decade are a mixture of significant declines and no significant change. PBDEs have generally not changed in the last decade, and have increased at one colony (Granite Island) from 2008 to 2016. Concentrations of the flame retardant Dechlorane Plus (DDC-CO), although inconsistent, have increased in some colonies in recent years, particularly those in northern lakes. Although Polychlorinated naphthalenes (PCNs) has declined overall since 1980, there was a resurgence in the late 1990s to early 2000s from a source from the Detroit River, which appears to have influenced most colonies downstream into those from Lake Ontario. The Wildlife Contaminant Index indicates an improvement from 2002 to 2017, both lake-wide and for some individual colonies. The WCI has not improved at some U.S. colonies, but some of these may be due to lower power and the slopes were positive (hence improving).

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend (2002-2017): Improving

Long-term Trend (1974-2017): Improving

Rationale: The traditional legacy contaminants, DDE, Sum PCBs and TCDD, have declined significantly in the long term (1974-2017). Mercury has declined significantly in the long term but neither it, nor Sum PBDEs, has declined significantly in the short term. DCC-CO has increased from 1982 to 2015. At the Agawa Rocks colony, Sum PFCA have increased from 1990 to 2010. Sum PBDEs are unchanged, except for Granite Island, where concentrations have increased. The Wildlife Contaminant Index indicates an improvement from 2002 to 2017, both lake-wide and for some individual colonies. The WCI has not improved at some U.S. colonies, but some of these may be due to lower power; however, at Gull Island, the slopes were positive (hence improving). PCNs have declined since the last report in 2017.

Lake Michigan

Status: Good

10-Year Trend (2002-2017): Improving

Long-term Trend (1974-2017): Improving

Rationale: The traditional legacy contaminants, DDE, SUM PCBs and TCDD, have declined significantly since the 1970s (1974-2017). Mercury has declined significantly in the long term but neither it, nor Sum PBDE, has declined significantly in the short-term. DCC-CO has increased from 1982 to 2015. The Wildlife Contaminant Index indicates an improvement from 2002 to 2017, both lake-wide and for some individual colonies. The WCI has not improved at some U.S. colonies, but some of these may be due to lower power and the slopes were positive (hence improving).

Lake Huron (including St. Marys River)

Status: Good

10-Year Trend (2002-2017): Improving

Long-term Trend (1974-2017): Improving

Rationale: The traditional legacy contaminants, DDE, Sum PCBs and TCDD and mercury have declined significantly since the 1970s (1974-2017). No significant change for Sum PBDEs in the short-term was observed. DCC-CO has increased from 1982 to 2015, while Sum PFCAs have increased from 1990 to 2010 from the Detroit River colony. The Wildlife Contaminant Index indicates an improvement from 2002 to 2017, both lake-wide and for some individual colonies. The WCI has not improved at some US colonies, but some of these may be due to lower power and the slopes were positive (hence improving), except for Scarecrow Island where the slope for WCI was negative.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Good

10-Year Trend (2002-2017): Unchanging

Long-term Trend (1974-2017): Improving

Rationale: The legacy contaminants, DDE, Sum PCBs, TCDD and mercury, have all declined significantly since the 1970s (1974-2017). No significant change for Sum PBDEs in the short-term was observed. DCC-CO has increased from 1982 to 2015. Sum PFCA have increased from 1990 to 2010 in the Detroit River. The Wildlife Contaminant Index indicates an improvement from 2002 to 2017, both lake-wide and for some individual colonies. The WCI has not improved at some U.S. colonies, but some of these may be due to lower power and the slopes were positive (hence improving). Sum PCNs declined from 1979 to 2013, but in Lake Erie concentrations of PCNs increased in gulls from 1995 until 2000. Concentrations of PCNs were higher in gulls from Lake Erie than all other lakes.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Good

10-Year Trend (2002-2017): Improving

Long-term Trend (1974-2017): Improving

Rationale: The traditional legacy contaminants, DDE, sum PCBs and TCDD and mercury have declined significantly since the 1970s (1974-2017). No significant change for Sum PBDEs in the short-term was observed. DCC-CO has increased from 1982 to 2015. The Wildlife Contaminant Index indicates an improvement from 2002 to 2017, both lake-wide and for some individual colonies. The WCI has not improved at some US colonies, but some of these may be due to lower power and the slopes were positive (hence improving). Sum PCNs declined from 1979 to 2013.

Status Assessment Definitions

Good: The metrics show that the contaminant levels in fish-eating waterbirds eggs are meeting the ecosystem objectives/guidelines or they are otherwise in an acceptable condition.

Fair: The metrics show that the contaminant levels in fish-eating waterbirds eggs are not meeting the ecosystem objectives/guidelines, but they are exhibiting minimally acceptable conditions.

Poor: The metrics show that the contaminant levels in fish-eating waterbirds eggs are not displaying minimally acceptable conditions and are severely impacted.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Decrease in contaminant levels and other parameters of concern.

Unchanging: No change in the level of contaminants and/or other parameters of concern.

Deteriorating: Increased contaminant levels and other parameters of concern.

Undetermined: Data are not available or are insufficient to assess the trend of contaminant levels in fish-eating waterbirds at this time.

Endpoints and/or Targets

Chemical levels and biological measures in colonial nesting waterbirds are not different from those from reference sites in Atlantic Canada or from the Prairies. Decreasing contaminant trends.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess concentrations of chemical contaminants in a representative fish-eating colonial waterbird, and it will be used to infer the impact of these contaminants on the physiology of the colonial waterbird.

This sub-indicator will assess the current toxic chemical concentrations and trends in representative colonial waterbirds (gulls, terns, cormorants and/or herons) on the Great Lakes; infer and measure the impact of contaminants on the health (i.e. the physiology and breeding characteristics) of the waterbird populations; and assess ecological and physiological endpoints in representative colonial waterbirds on the Great Lakes. It can be used to describe temporal and spatial trends of bioavailable contaminants in representative biota throughout the Great Lakes; to infer the effectiveness of remedial actions related to the management of critical pollutants; and to document and describe the trends of chemicals of emerging concern.

Ecosystem Objective

Tracking progress of fish-eating colonial waterbirds on the Great Lakes toward an environmental condition in which there is no difference in contaminant levels and related biological endpoints between birds on and off the Great Lakes. As part of this sub-indicator, contaminant levels are also measured in Herring Gull eggs to ensure that levels continue to decline.

This sub-indicator best supports work towards General Objective #4 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from pollutants in quantities or concentrations that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain."

Measure

This sub-indicator will measure annual concentrations of Polychlorinated Biphenyls (PCBs), dioxins and furans, organochlorine pesticides such as Dichlorodiphenyltrichloroethane (DDT) and related metabolites, other organic contaminants, and trace metals including mercury in Herring Gull (*Larus argentatus*) eggs from 15 Environment Canada's Great Lakes Herring Gull Monitoring Program (GLHGMP) sites throughout the Great Lakes (U.S. and Canada). In addition, at some sites eggs from Double Crested Cormorants (*Phalacrocorax auritus*) have been collected to complement the Herring Gull data. The Michigan Department of Environmental Quality also measures a similar suite of chemicals in Herring Gull eggs from colonies from the U.S. side of the Laurentian Great Lakes.

Ecological Condition

Although there are Great Lakes wildlife species that are more sensitive to contaminants than Herring Gulls and colonial nesting waterbird species in general, there is no other species with a long-term dataset like the Herring Gull program. The Herring Gull egg contaminants dataset is the longest running continuous (annual) contaminants dataset for wildlife in the world**.** As contaminant levels continue to decline, the usefulness of the Herring Gull as a biological indicator species may lessen (due to its reduced sensitivity to low levels of contamination) but its value as a chemical indicator will remain and probably increase - as levels become harder and harder to measure in other media. It is an excellent accumulator tracker since many of the above biological measures are correlated with contaminant levels in their eggs. In other colonial waterbirds, there are similar correlations between contaminant levels in eggs and various biological measures. Contaminant levels in eggs of other colonial waterbirds are usually correlated with those in Herring Gulls. Adult Herring Gulls nest on all the Great Lakes and the connecting channels and remain on the Great Lakes year-round. Because their diet is primarily fish, they are an excellent terrestrially-nesting indicator of the aquatic community. The chemical related sub-indicators showing long-term trends of contaminants in biota provide valuable insight into the relative abundance of contaminants in the vicinity of fish and waterbird populations. They represent not just contaminants in water, but offer insight into how chemicals enter and move throughout the food web.

Status of Chemicals of Mutual Concern (CMCs)

The Parties to the GLWQA designated the first set of Chemicals of Mutual Concern in May 2016. For State of the Great Lakes reporting in 2019 and moving forward, the toxic chemical sub-indicators will support reporting on Chemicals of Mutual Concern (CMCs) in a more fulsome, consistent and transparent way. Information on additional chemicals of interest for the Great Lakes (not CMCs) is valuable for inclusion in the report and will be included in a separate section below, as appropriate. For the 2019 report, the assessment is not based solely on CMCs but PCBs are a driver of the WCI scores. PCBs are a designated Chemical of Mutual Concern and are a deleterious class of contaminant in the Great Lakes and can have negative effects on fish, wildlife and humans.

The following chemicals were identified as the first set of CMCs:

- Hexabromocyclododecane (HBCDD/HBCD)
- Long-Chain Perfluorinated carboxylic acids (LC-PFCAs)
- Mercury (Hg)
- Perfluorooctanoic acid (PFOA)
- Perfluorooctane sulfonate (PFOS)
- Polybrominated Diphenyl Ethers (PBDEs)
- Polychlorinated Biphenyls (PCBs)
- Short-Chain Chlorinated Paraffins (SCCPs)

Contaminant Burdens

Annual concentrations of legacy compounds, such as organochlorine pesticides, PCBs, PCDFs/PCDDs and other organic contaminants, and mercury and other metals are measured in Herring Gull eggs from 15 sites from the Great Lakes Herring Gull Monitoring Program, and 5 sites from the Michigan Department of Environmental Quality from throughout the Great Lakes (U.S. and Canada). The Herring Gull eggs are collected in a similar fashion between the two programs, and similar contaminant analyses are performed; the main difference between the two programs is the frequency of egg collection. On a less routine basis, measurements of brominated and non-brominated flame retardants, and perfluorinated sulfonates (PFSAs) and perfluorinated carboxylic acids (PFCAs) are also analyzed. PFSAs and PFCAs have not been measured recently, but will likely be analysed again in 2018.

At all colonies of Herring Gulls monitored in the Great Lakes Herring Gull Monitoring Program (Environment and Climate Change Canada), concentrations of PCBs, PCDD/Fs and organochlorine pesticides have fallen dramatically since the 1970s (de Solla et al., 2016). The range in concentrations of PCBs (Sum of 33 PCB congeners) in colonies monitored in both 2002 and 2017 was between 0.96 and 11.27 µg/g in 2002, whereas by 2017 the concentrations ranged between 0.9 to 7.81 µg/g (Table 1). In general, trends in contaminant burdens followed an exponential decline from the 1970s to 2013, i.e., the rate of decline is proportional to concentrations (de Solla et al. 2016). Although generally the declines were consistent with a first order exponential decay model, the rates of decline in Persistent Organic Pollutants (POPs) in Herring Gull eggs were generally lower in later years, and for many colonies, concentrations have stabilized in the last few years. When all colonies were pooled, the mean half-lives for POPs ranged from 5.5 to 13.7 years for PCBs, TCDD and the six organochlorine pesticides (de Solla et al. 2016). For Sum PCBs, the half-lives ranged from 9.9 to 24.3 years among colonies, with Middle Island having the longest half-life. Overall, Middle, Granite and Gull islands (Lakes Erie, Superior and Michigan, respectively) had the longest half-lives for POPs.

Although the Clean Michigan Initiative-Clean Water Fund (CMI-CWF; Michigan Department of Environmental Quality) have not monitored Herring Gulls for long as the Great Lakes Herring Gull Monitoring Program, there have been some declines in PCBs and OC pesticides. PCBs, p,p'-DDE and total mercury had declined from 2002/06 to 2016 for colonies from Lake Michigan and Lake Huron (Table 2). Generally, concentrations for the subset of colonies from the MDEQ were within the range for the 15 colonies of the GLHGMP.

Using the procedures initially developed for the derivation of the Canadian Water Quality Index (WQI) and then used for development of the Sediment Quality Index (SQI) as a means of summarizing data (Canadian Council of Ministers of the Environment, 2001; Grapentine et al., 2002), a wildlife contaminant index (WCI) was calculated to examine temporal trends in overall contaminant exposure as an indicator of protection to piscivorous wildlife (Figures 1, 2). This site-specific method combines individual contaminant data and calculates an integrated numerical score by incorporating counts of guideline exceedances (scope) and the magnitude of those exceedances (amplitude) for compounds of interest in a study area. Fish flesh criteria were selected as guidelines developed by Newell et al. (1987) which were designed to protect animals eating contaminated fish from adverse effects such as mortality, reproductive impairment, and organ damage. Using contaminant levels in eggs at each colony for each study year, an index was calculated using eight compounds for which guidelines are available and for which there was at least one exceedance (Table 3). These compounds, with corresponding fish flesh criteria guidelines shown in brackets, are as follows: Sum PCBs (110 ng/g), total DDT (200 ng/g), dieldrin (120 ng/g), mirex (330 ng/g), sum chlordane (500 ng/g), heptachlor epoxide (200 ng/g), octachlorostyrene (20 ng/g), and TEQs based on concentrations of dioxins and furans (3 pg/g). Unlike the applicable uses of the WQI and SQI which provide an assessment of water and sediment quality as these relate to protection of aquatic life, the value of the WCI has no meaning but is used here only to examine changes in exceedances of guidelines over time and thus assess temporal changes in the protection of piscivorous wildlife at a colony or Great Lake. Note that higher WCI scores are associated with better conditions versus lower scores for where there are more and/or larger exceedances.

The WCI generally improved at most colonies from 2002 to 2017, with a few exceptions. On a lake-wide basis, the WCI did not improve at Lake Erie from 2002 to 2017. Most of the WCI was driven by PCBs in all lakes, with a smaller but roughly equal contribution of PCDD/Fs and DDT, although the relative contribution of PCDD/Fs was greater in some colonies in the United States compared to ones in Canada.

These declines in legacy POPs are consistent with compounds whose production ceased in the 1970s (Figure 3); however, the temporal trends in other compounds whose production continued in the 2000s or later show different trajectories. For example, PBDEs in Herring Gull egg from 6 colonies generally showed rapid increases from 1982 to 2000, no further increasing trend from 2000 to 2006, and then declines by 2012 (Letcher et al. 2015); however, concentrations have largely stabilized since 2012 (Table 2). Conversely, syn- and anti-Dechlorane Plus (DDC-CO), have increased in recent years (Figure 4; Letcher et al. 2015). PCNs, although low compared to other legacy compounds, were elevated in eggs from the Detroit River and Western Lake Erie, putatively from dredging of PCN contaminated sediments in the Detroit River, and declined with distance downstream to eastern Lake Ontario and St Lawrence River (Figure 5). Concentrations declined from 1979 until the late 1990s, when they increased until 2000 (Figure 6), when they declined once more. Generally, concentrations of organic contaminants are highest in Herring Gull Eggs in colonies near large urban or industrial sources, but for substituted diphenylamines. (SDPAs) concentrations were highest in more northern colonies, in contrast to fish where concentrations were highest in lower Great Lakes (Figure 7). The discrepancy is believed to be due to a greater contribution of terrestrial dietary items of gulls in northern colonies, where landfills are a suspected main source of SPDAs.

Contaminant burdens varied among the 15 GLHGMP colonies, with concentrations generally highest in colonies with substantial urban or industrial influences nearby or upstream. Overall, Herring Gull eggs from the Detroit River or Western Lake Erie (Fighting Island, Middle Island, Detroit Edison), Toronto Harbour and Hamilton Harbour (Lakes Ontario), and Green Bay (Channel Shelter Island) were the most contaminated for legacy POPs. Conversely, the colonies from northern Lake Huron and Lake Superior tended to be the least contaminated (Table 1).

Assessment of Health of Colonial Waterbirds

The health of colonial waterbirds, particularly in relation to contaminant burdens or exposure, has been assessed at a number of colonies, primarily in Areas of Concern (AOCs). Contaminant burdens were examined in eggs of Herring Gulls and double-crested cormorants (*Phalacrocorax auritius*) collected from colonies in the vicinity of the Spanish Harbour AOC in Recovery (Lake Huron) and compared to reference colonies in 2011 and 2012. Concentrations of TCDD, PCBs, and mercury, were low in eggs and were not notably elevated in the Area in Recovery (AiR) relative to the reference colonies, and were considered to be below those associated with adverse effects on reproduction. Recent egg burdens appeared to be markedly lower to concentrations measured in earlier time periods (Hughes et al. 2014b). Similarly, Reproduction and development were examined in Herring Gulls and common terns (*Sterna hirundo*) breeding within the St. Marys River Area of Concern (Lake Huron) in 2011 and 2012. Freshly-laid eggs were collected from colonies within the AOC and from reference sites were artificially incubated in the laboratory and assessed for embryonic viability, incidence of embryonic deformities, contaminant burdens and other biochemical endpoints. Overall, embryonic viability of Herring Gulls and common terns was high at AOC colonies. Frequencies of embryonic deformities were comparable between AOC colonies and reference colonies for both species, were not associated with exposure to dioxin-like PCBs and dioxins, which did not differ between AOC and reference sites. Contaminants were not sufficiently elevated in embryos to adversely impact the reproductive success and development of Herring Gulls and common terns foraging in the St. Marys River AOC (Hughes et al. 2014a).

Breeding success of the Black-crowned Night-Heron (*Nycticorax nycticorax*) was examined at a colony on Turkey Island in the Detroit River Area of Concern (AOC) and an upstream non-AOC reference colony on Georgian Bay in 2009 and 2011. Breeding success was lower in night-herons from the AOC compared to the reference colony in both study years; at the AOC colony in 2009, productivity was below a range of thresholds considered to be typical for a stable population. Despite higher concentrations found overall at the AOC colony, concentrations of PCBs, other organochlorines and PBDEs in eggs and liver of nestlings were below concentrations associated with adverse reproductive effects. Mercury concentrations in eggs and livers of nestlings from the AOC colony were comparable to concentrations at the reference colony and were below those associated with adverse reproductive effects. Reduced breeding success in 2009 was likely not due to elevated concentrations of contaminants historically associated with the AOC, but likely to other stressors, such as predation, weather and disturbance. At both colonies, concentrations of DDT, PCBs and mercury in eggs and nestling livers exceeded tissue residue guidelines (Hughes et al. 2013).

Other Chemicals of Interest

Variable DNA microsatellites were used to screen for mutations in Double-crested Cormorants (*Phalacrocorax auritus*) families from two colonies in Hamilton Harbour AOC (Lake Ontario) and Mohawk Island (Lake Erie). Microsatellite mutation rates were 6 times higher at the Hamilton Harbour site closest to the industrial sources of Polycyclic Aromatic Hydrocarbons (PAHs) than the other Hamilton Harbour site, and both were higher than the reference colony (King et al. 2014). A Phase I metabolite of the PAH benzo[a]pyrene was identified in bile and liver from Hamilton Harbour cormorant chicks suggesting that these cormorants are exposed to and metabolizing PAHs, highlighting their potential to have caused the observed mutations (King et al. 2014).

The health of Herring Gulls is also being assessed at Thunder Bay (Lake Superior) and Hamilton Harbour AOCs.

Periodic measurements are made of biological features of gulls and other colonial waterbirds known to be directly or indirectly impacted by contaminants and other stressors. These include (but are not limited to): clutch size, eggshell thickness, hatching and fledging success, size and trends in breeding population, various physiological biomarkers including vitamin A, immune and thyroid function, stress (corticosterone) and growth hormone levels, liver enzyme induction, PAH levels in bile and porphyrins and genetic and chromosomal abnormalities. Additional monitoring considerations include: tracking porphyria, vitamin A deficiencies, and the evaluation of avian immune systems. Chemical burdens in eggs of colonial nesting waterbirds are assessed for temporal trends, and are compared to suitable reference sites.

Linkages

There are many linkages between the Toxic Chemicals in Great Lakes Herring Gull Eggs sub-indicator and many other sub-indicators within the Great Lakes (previously known as SOLEC) reporting suite. There is a link between Fish-Eating and Colonial Nesting Waterbirds and Toxic Chemicals in Whole Fish. Fish and gull eggs have shown similar trends in mercury both among lakes and within basins within each lake (McGoldrick et al., 2018; Blukacz-Richards et al, 2017).

There are also links to the Lake Sturgeon, Lake Trout and Prey fish sub-indicators. Changes in fish productivity of the Great Lakes have been reflected in fish-eating birds (Paterson et al. 2014); temporal changes in the energy density of forage fish eggs are reflected in those of both top predator fish (Lake Trout) and a fish-eating bird (Herring Gulls).

A link has also been shown by Dr. Craig Hebert between contaminant levels in Herring Gull eggs and Ice Cover. Inferences on the effects of climate change on the accumulation of contaminants in aquatic biota are beyond the scope of this sub-indicator as they would have to include changing food webs and energy cycling though them.

There is a direct link between Herring Gull contaminants and endocrine disruption.

In terms of the health of Great Lakes fish-eating birds, there is also a link between Herring Gulls and both botulism outbreaks and the occurrence of fish diseases.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources.	X			
Geographic coverage and scale of data are appropriate to the Great Lakes basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	No			

Data Limitations

Herring Gull eggs have been collected annually on all 5 lakes and connecting channels since 1973 (Weseloh and Moore, 2014) and are highly tolerant of persistent contamination and may underestimate biological effects occurring in other less monitored, more sensitive species.

Some adult Herring Gulls from the upper Lakes, especially Lake Superior, move to the lower Lakes, especially Lake Michigan, during harsh winters. This has the potential to confound the contaminant profile of a bird from the upper lakes. Most of the gull's time is still spent on its home lake and this has not been noted as a serious limitation up to this point. Using contaminant accumulation by young, flightless gulls would eliminate this problem, but their contaminant levels and effects would be less due to lower contaminant exposure/intake.

It is difficult to show consistent differences in biological effects among colony sites within the Great Lakes. This is probably due to the great overall reduction in contaminant levels as well as the lessening in differences among Great Lakes sites. The comparisons which show the greatest differences for biological effects of contaminants are between sites on and off the Great Lakes.

At one GLHGMP site (Detroit River), Herring Gulls no longer breed. The only colonial waterbird at that site is currently Double Crested Cormorants. Two other colonies – Toronto and Niagara River, are precariously small, but Cormorants are present at those colonies as well. Currently at those and other sites, Cormorant eggs are being collected to allow for continual monitoring of contaminants. At some sites, there may be a switch to Cormorants for long term monitoring (i.e. Detroit River), and for others the Cormorants will be assessed only for short term to aid in inter species comparisons. Having both species as long-term monitors may be prohibitively expensive, so the use of Cormorants will be limited in the long term.

Additional Information

Historical data on levels of chemical contamination in gull eggs are available, on an annual basis, for most sites in both the Canadian and U.S. Great Lakes dating back to the early 1970s. An immense database of chemical levels and biological measures from the Great Lakes, as well as many off-lake sites, is available from the Wildlife and Landscape Science Directorate at Environment and Climate Change Canada. Data on temporal trends, portrayed as annual contaminant levels over time, for 1974-present in most instances, are available for each site and each chemical compound. For example, DDE, from 1974-2013, is available for Toronto Harbour and could be displayed graphically. Geographical patterns in contaminant levels, showing all sites relative to one another, are also available for most years from 1974-present and for most chemical compounds. For example, PCBs, 2008, at 15 Great Lakes sites from Lake Superior to the St. Lawrence River (including U.S. sites) and could be displayed on both maps and graphs.

The bioavailability of POPs, and thus exposure to wildlife is not simply a function of the concentrations found in environmental matrices such as water, soil or sediment, but varies considerably with the myriad of factors that control the transport and fate of contaminants. Measurements of body burdens in waterbirds integrate the net effect of factors such as bioavailability, temperature, growth rates, food chain dynamics, and chemical partitioning behaviour. One of the advantages of using colonial waterbirds as indicators is that their rates of elimination of body burdens for POPs are generally much faster than the rates of environmental degradation; hence changes in body burdens reflect changes in the bioavailability of POPs.

Degradation half-lives in sediment of the PCB congeners typically found in Herring Gull eggs range between 10 to 19 years in sediment (Sinkkonen and Paasivirta 2000). Conversely, the half-life of p,p'-DDE in Herring Gulls was estimated to be 264 days (Norstrom et al. 1986), with half-lives for PCBs likely to be similar. The half-lives of PCBs fed to ring doves ranged from 7 to 53 days (Drouillard and Norstrom 2001). Hence, Herring Gulls respond faster to inputs of POPs through their diet than the degradation rate of POPs in the general environment. Although there were dramatic declines in contaminant burdens of legacy POPs in Herring Gull eggs from the 1970s to 2013, not all of the changes in egg burdens were due solely to the elimination of the contaminants in the environment. Changes in food web components affect dietary exposure and hence body burdens of POPs in wildlife. By using ecological tracers, Hebert and Weseloh (2006) found that not only did Herring Gull diets and trophic level change at many Great Lakes colonies between 1974 and 2003, but when the effect of changing trophic level was removed, the rates of contaminant declines were reduced. Hence, a proportion of the declines were due to reductions in dietary exposure from feeding at lower trophic levels.

Also, contaminant concentrations in most colonially-nesting, fish-eating birds are at levels where gross ecological effects, such as eggshell thinning, reduced hatching and fledging success, and population declines, are no longer apparent. Greater reliance for detecting biological effects of contaminants is being put upon physiological and genetic biomarkers.

Future consideration is to include contaminants in Bald Eagles as part of this sub-indicator. Bill Bowerman is collecting data on the U.S. side, however Canadian data are unknown. There are some differences in the diet of gulls and eagles, as eagles also feed on carrion more than Herring Gulls do. Cormorants are, on the other hand, obligate piscivores.

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Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality]

Table 2. Concentrations of brominated flame retardants (ng/g, wet weight) in Herring Gulls eggs from the Great Lakes in 2008 to 2016

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]

Table 3. Mean concentrations of PCBs and other organochlorines (ng/g) and total TEQs comprised of TEQ for non-ortho PCBs, dioxin/furans, and mono-ortho PCBs (pg/g) from 2002–2017. Colonies are ranked by mean sum PCB concentration. For years when individual eggs were analyzed, a mean concentration was calculated for the year. N represents the number of years.

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality]

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Figure 1. Temporal trends for the Wildlife Contaminant Index, based upon PCBs, PCDD/Fs and selected organochlorine pesticides, of Herring Gull colonies pooled by Great Lake. WCIs are significantly increasing (improving conditions) at all lakes, as designated by “**”, except for Lake Erie. Three colonies had regression slopes that differed significantly from the overall lake slope; Port Colborne (LE), Fighting Island (LE) and Scarecrow Is (LH).

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality]

Figure 2. Temporal trends for the Wildlife Contaminant Index of each Herring Gull colony within each Great Lake, based upon PCBs, PCDD/Fs and selected organochlorine pesticides. "**" indicates colonies where linear regression analysis is significantly different from zero.

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality]

Figure 3. Sum PCBs in herring gull eggs from Detroit River (Fighting Island) and western Lake Erie (Middle Island) from 1974-2018. The green bars indicate PCB concentrations in cormorant eggs from Fighting Island in 2016 and 2018 since no data are available for Herring Gulls.

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]

Figure 4. Concentrations of Dechlorane Plus (ng/g, wet weight) in Herring Gull eggs from selected colonies from the Great Lakes in 1982 to 2015.

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]

Figure 5. Concentrations of Polychlorinated naphthalenes (ng/g, wet weight) in Herring Gull eggs from selected colonies from the Great Lakes in 2011-2013 (unpublished data, ECCC; based upon McGoldrick et al, 2018). Colonies are ordered left-right from upstream-downstream; the peak corresponds to eggs from the Detroit River.

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]; McGoldrick et al., 2018

Figure 6. Concentrations of Polychlorinated naphthalenes (ng/g, wet weight) in Herring Gull eggs from the Detroit River and Western Lake Erie, and Walleye from Western Lake Erie, from 1980 to 2015. In the mid to late 1990s there was substantial dredging of PCN contaminated sediment from the Detroit River

Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]; McGoldrick et al., 2018

Figure 7. Concentration (mean ± standard error bar (ww) of ∑SDPAs in Herring Gull eggs (blue) and Lake Trout and Walleye whole body homogenate (red) in the Great Lakes of North America. Inset shows boxplot of lipid normalized ∑SDPAs concentrations.

Source: Chemical Monitoring Program [Environment and Climate Change Canada]; Liu et al., 2018

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Table 1. Concentrations of PCBs (sum of 33 congeners) and select organochlorine pesticides (µg/g, wet weight) in Herring Gull eggs from the Great Lakes in 2002 to 2017. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality]
HCB = hexachlorobenzene; PCBs = Polychlorinated biphenyls, DDT = Dichlorodiphenyltrichloroethane + isomers, HE = Heptachlor epoxide, OCS = Octachlorostyrene

Lake	Colony	Year	HCB	Σ33 PCBs	ΣChlordane	Σ DDT	dieldrin	mirex	HE	OCS
Superior	Agawa Rock	2002	0.009	2.38	0.082	1.237	0.027	0.021	0.026	0.002
		2017	0.018	0.92	0.044	0.287	0.023	0.017	0.018	0.002
Superior	Granite Is.	2002	0.010	2.68	0.076	1.141	0.040	0.020	0.022	0.002
		2017	0.016	1.28	0.041	0.366	0.022	0.015	0.015	0.002
Superior	Gull Island	2014	0.010	0.96	0.055	0.394	0.009	0.014	0.010	0.001
		2017	0.013	0.90	0.049	0.219	0.022	0.011	0.010	0.001
Superior	Huron Island	2002	0.008	3.61	0.133	2.066	0.023	0.023	0.037	0.001
Superior	Net Island	2002	0.013	3.94	0.175	2.608	0.045	0.066	0.041	0.005
Superior	Tahquamenon Island	2002	0.020	3.44	0.839	1.597	0.060	0.052	0.318	0.013
Michigan	Bellow Island	2002	0.009	6.39	0.125	2.624	0.030	0.050	0.040	0.002
		2017	0.013	0.94	0.039	0.293	0.023	0.006	0.010	0.001
Michigan	Big Sister Is.	2002	0.008	7.19	0.130	4.205	0.036	0.044	0.031	0.001
		2017	0.012	1.47	0.034	1.047	0.016	0.003	0.013	0.001
Michigan	Green Island	2002	0.024	5.70	0.241	3.150	0.058	0.093	0.056	0.003
Michigan	Gull Is.	2002	0.010	7.21	0.148	4.311	0.046	0.040	0.040	0.000
		2017	0.028	3.50	0.091	1.008	0.036	0.024	0.029	0.002
Huron	Channel-Shelter Is.	2002	0.009	11.27	0.055	2.734	0.016	0.020	0.016	0.010
		2017	0.019	3.95	0.024	0.767	0.019	0.005	0.008	0.039
Huron	Chantry Is.	2002	0.006	1.24	0.041	0.617	0.011	0.033	0.012	0.001
		2017	0.010	0.97	0.028	0.349	0.010	0.015	0.007	0.002
Huron	Double Is.	2002	0.007	1.55	0.060	0.957	0.028	0.029	0.017	0.001
		2017	0.010	1.04	0.023	0.281	0.013	0.009	0.008	0.003
Huron	Five Mile Island	2017	0.007	0.76	0.016	0.216	0.007	0.006	0.004	0.001
Huron	Little Charity Island	2002	0.011	4.79	0.052	1.168	0.013	0.038	0.016	0.003
		2017	0.013	1.82	0.031	0.268	0.018	0.012	0.007	0.010
Huron	Scarecrow Island	2002	0.018	3.41	0.139	1.559	0.047	0.029	0.036	0.002

Huron	West Twin Pipe	2002	0.009	2.34	0.064	0.957	0.017	0.027	0.020	0.001
Detroit R	Detroit Edison	2002	0.016	10.86	0.073	1.297	0.080	0.011	0.024	0.009
		2017	0.008	4.42	0.029	0.356	0.043	0.006	0.008	0.004
Detroit R	Fighting Is.	2002	0.012	11.74	0.046	1.478	0.026	0.015	0.012	0.010
Erie	Middle Is.	2002	0.008	7.11	0.049	0.719	0.040	0.006	0.017	0.005
		2017	0.009	6.04	0.034	0.454	0.019	0.006	0.010	0.004
Erie	Port Colborne	2002	0.003	2.50	0.022	0.517	0.015	0.025	0.007	0.000
		2017	0.011	3.34	0.032	0.559	0.019	0.017	0.009	0.002
Niagara	Weseloh Rocks	2002	0.008	2.81	0.040	0.800	0.022	0.068	0.011	0.002
Ontario	Hamilton Harbour	2002	0.010	5.20	0.053	2.028	0.025	0.280	0.014	0.003
		2017	0.013	3.78	0.037	0.891	0.028	0.083	0.008	0.004
Ontario	Toronto	2002	0.008	3.66	0.050	1.688	0.047	0.343	0.011	0.002
		2017	0.056	7.81	0.108	3.847	0.028	0.403	0.030	0.014
Ontario	Snake Is.	2002	0.007	3.51	0.035	1.625	0.014	0.309	0.010	0.003
		2017	0.014	2.23	0.031	0.709	0.014	0.104	0.010	0.003
St Lawrence	Strachan Is.	2002	0.005	5.62	0.031	1.448	0.011	0.265	0.007	0.002
		2017	0.008	1.55	0.014	0.386	0.005	0.073	0.003	0.002

Table 2. Concentrations of brominated flame retardants (ng/g, wet weight) in Herring Gull eggs from the Great Lakes in 2008 to 2016. DP = Declorane Plus; HBCDD = Hexabromocyclododecane; BTBPE = 1,2-bis-(2,4,6-tribromophenoxy)ethane, PBDEs = Polybrominated diphenyl ethers; ND = Not Detected. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality].

Lake	Colony	Year	ΣDP	HBCDD	BTBPE	Σ13 PBDEs
Superior	Agawa Rocks	2008	4.40	7.53		200.54
		2016	5.71	10.90	ND	379.46
Superior	Granite Is.	2008	3.22	7.31		347.17
		2016	1.85	13.30	0.49	546.02
Michigan	Gull Is.	2008	1.01	9.97		420.47
		2016	2.24	10.90	0.48	282.07
Michigan	Big Sister Is.	2008	5.47	10.23		265.38
		2016	5.50	18.60	0.48	460.83
Huron	Channel-Shelter Is.	2008	1.73	2.61		330.43
		2016	3.66	10.60	0.09	581.45
Huron	Chantry Is.	2008	2.28	6.78		153.36
		2016	4.34	17.00	ND	284.96
Huron	Double Is.	2008	1.28	7.30		237.55
		2016	3.55	12.20	0.50	318.11
Erie	Port Colborne	2008	0.72	1.41		136.62
		2016	4.32	7.15	0.64	355.83
Erie	Middle Is.	2008	1.69	2.70		185.43
		2016	3.64	15.00	0.51	382.85
Niagara	Weseloh Rocks	2008	3.37	2.60		254.83
Ontario	Toronto Harbour	2008	3.24	6.96		397.91
		2016	4.06	9.97	ND	341.75
Ontario	Hamilton Harbour	2008	2.06	2.77		296.89
		2016	2.82	8.30	0.45	359.46
Ontario	Snake Is.	2008	0.50	2.01		238.78
		2016	0.73	5.24	0.49	210.18
St Lawrence R	Strachan Is.	2008	1.09	7.04		246.70
		2016	1.66	8.57	ND	238.63

Table 3. Mean concentrations of PCBs and other organochlorines (ng/g) and total TEQs comprised of TEQ for non-ortho PCBs, dioxin/furans, and mono-ortho PCBs (pg/g) from 2002–2017. Colonies are ranked by mean sum PCB concentration. For years when individual eggs were analyzed, a mean concentration was calculated for the year. N represents the number of years. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality].

Colony	Years	N	Sum PCBs (33 Common)	TEQs NO-PCBs	TEQs-PDDD/Fs	TEQs-Mono PCBs	Total TEQs	Total DDT	Sum Chlordane	mirex	dieldrin	HE	HCB	OCS
Fighting Is.	2002-2015	8	8943.89	180.99	15.81	26.76	223.57	945.53	50.14	12.10	27.10	11.68	10.61	7.90
Detroit Edison	2002-2017	16	8426.88	413.09	40.40	25.77	479.26	953.02	47.05	10.72	47.00	14.12	9.69	7.47
Middle Is.	2002-2017	16	8317.77	255.11	17.37	24.73	297.21	838.64	55.45	12.67	37.19	16.79	11.39	6.54
Channel-Shelter Is.	2002-2017	16	6087.73	266.09	35.53	40.43	342.05	1389.73	43.45	15.07	21.33	10.36	15.15	17.55
Indiana Harbor	2010	1	4706.51	283.16	6.63	21.14	310.93	1211.82	71.13	9.69	20.50	14.90	7.18	0.64
Hamilton Harbour	2002-2017	16	4259.69	149.01	11.06	17.38	177.46	1000.68	35.49	136.45	18.12	7.56	9.83	3.00
Leslie St. Spit	2002-2017	16	3835.48	149.79	14.94	17.31	182.04	1604.33	70.48	257.46	38.27	15.46	17.89	4.78
Little Charity Is.	2002-2017	14	3687.94	299.94	48.14	16.32	364.39	779.18	41.50	18.44	20.65	12.80	14.33	6.16
Scarecrow Island	2002-2009	8	3246.62	161.82	99.04	15.47	276.34	1152.62	74.10	52.20	22.32	21.80	11.97	2.86
Strachan Is.	2002-2017	16	3183.69	124.22	9.27	20.94	154.43	672.59	25.80	146.90	11.43	5.96	6.23	2.04
Snake Is.	2002-2017	16	3097.83	126.78	16.89	15.05	158.72	947.65	42.16	149.05	18.67	10.67	10.85	3.72
Gull Is. (LM)	2002-2017	16	3080.49	183.55	6.66	19.35	209.56	1310.52	100.89	18.18	44.33	24.14	13.62	1.28
Net Island	2002-2008	6	3015.66	257.14	22.29	15.34	294.77	1559.45	119.22	43.32	41.12	37.10	11.81	2.20
Green Island	2002-2009	8	2962.17	256.46	43.33	14.14	313.93	1386.17	103.53	39.39	48.96	28.72	14.29	2.29
Tahquamenon Is.	2002-2009	7	2908.22	209.35	23.01	13.27	245.63	1290.00	201.19	27.05	53.49	76.45	14.06	3.30
Port Colborne	2002-2017	16	2750.83	78.40	6.17	8.96	93.53	416.82	26.69	21.07	14.05	6.63	7.38	1.58
West Twin Pipe	2002-2009	8	2647.39	295.86	43.31	12.64	351.81	1158.38	87.86	43.32	36.28	27.79	12.87	2.35
Big Sister Is.	2002-2017	16	2506.89	177.76	4.89	16.96	199.62	1502.74	62.09	12.34	28.12	15.77	7.64	0.79
Bellow Island	2002-2017	16	2367.43	278.81	26.14	11.78	316.72	1121.00	83.66	19.89	45.21	25.51	11.34	2.70
Weseloh Rocks	2002-2015	14	2273.88	85.79	9.03	9.19	104.01	481.29	32.17	46.06	15.02	7.54	16.16	2.41
Huron Island	2002-2012	5	2193.69	232.27	36.28	11.20	279.75	926.52	76.02	19.91	23.50	21.70	9.85	1.96
Five Mile Island	2003-2017	15	1548.25	112.04	53.41	6.90	172.36	543.86	38.45	17.64	15.20	11.83	8.02	1.19
Granite Is.	2002-2017	16	1534.75	83.33	6.15	8.41	97.88	641.69	58.48	17.39	24.83	16.47	10.24	1.60
Agawa Rock	2002-2017	16	1398.68	82.79	7.05	7.60	97.44	531.44	64.63	22.94	26.39	19.75	14.09	2.12
Chantry Is.	2002-2017	16	1394.56	64.60	8.86	6.61	80.06	490.95	31.83	29.54	15.47	9.06	8.74	2.07
Double Is.	2002-2017	16	1265.85	70.27	9.93	6.71	86.91	439.33	43.96	29.23	16.89	11.20	8.53	1.75
Gull Island (LS)	2013,2014 &2017	3	1185.52	59.96	49.01	5.85	114.83	411.26	63.41	16.15	15.58	12.07	12.05	1.10

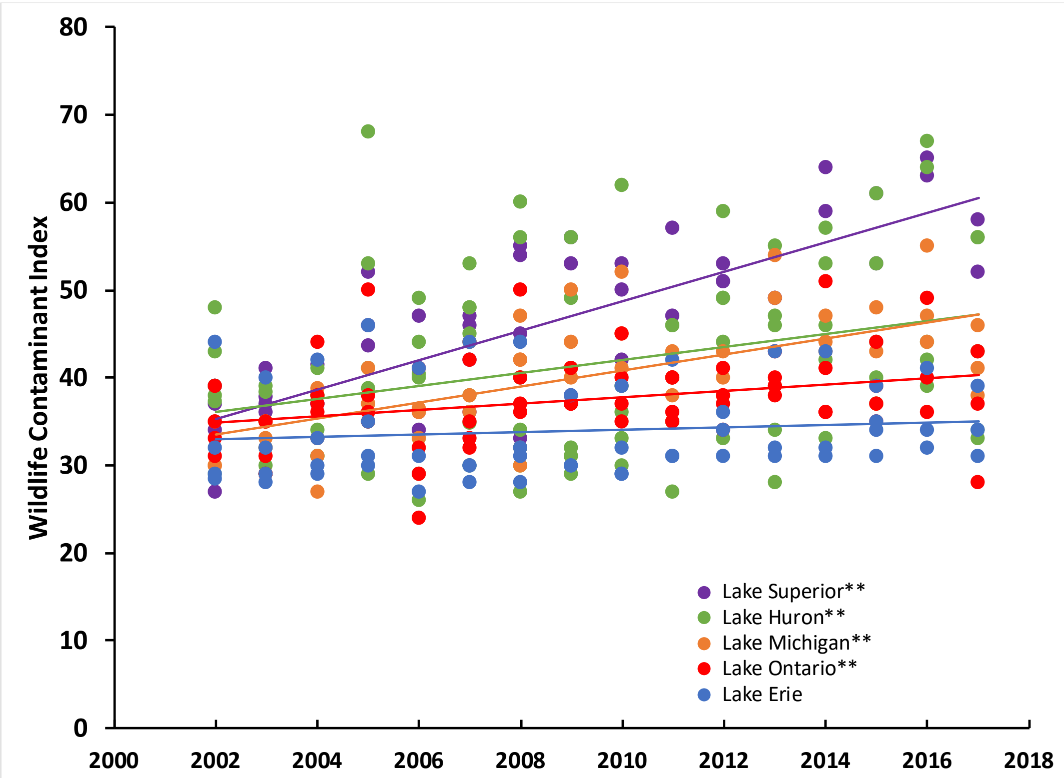
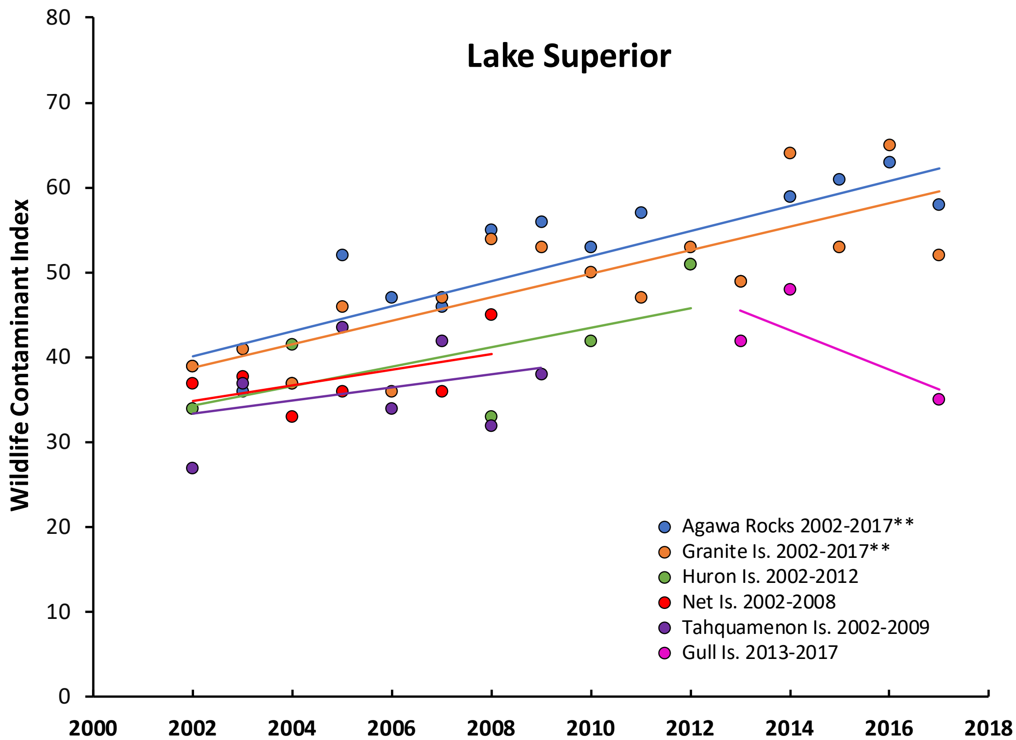
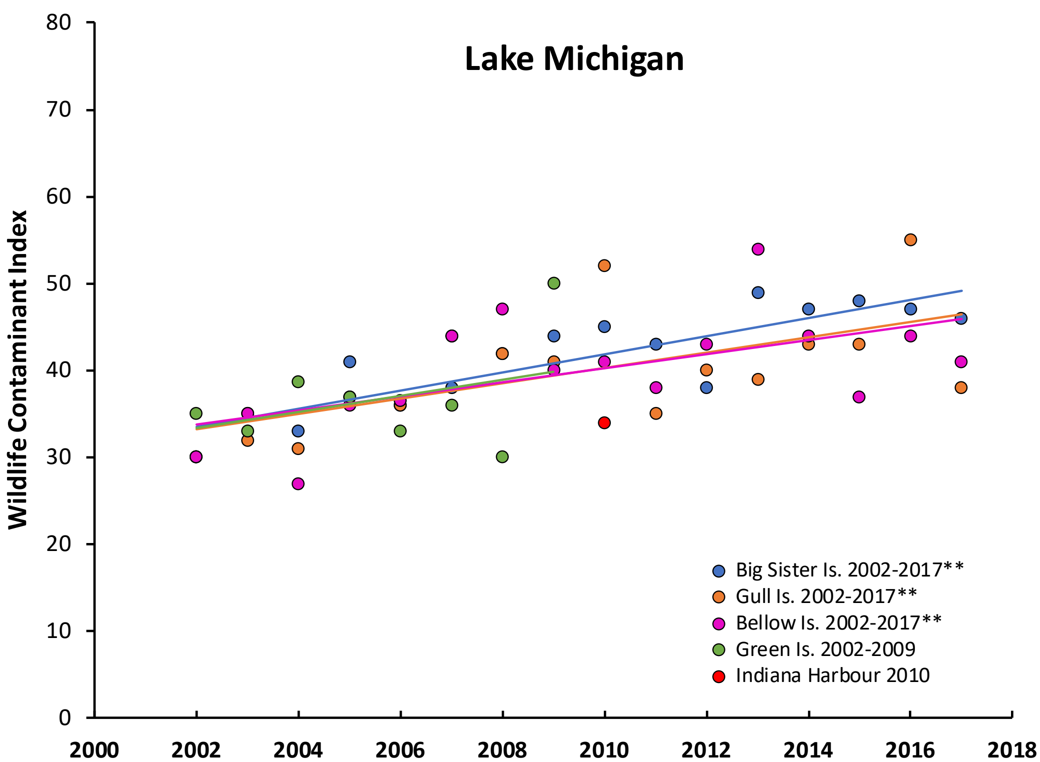
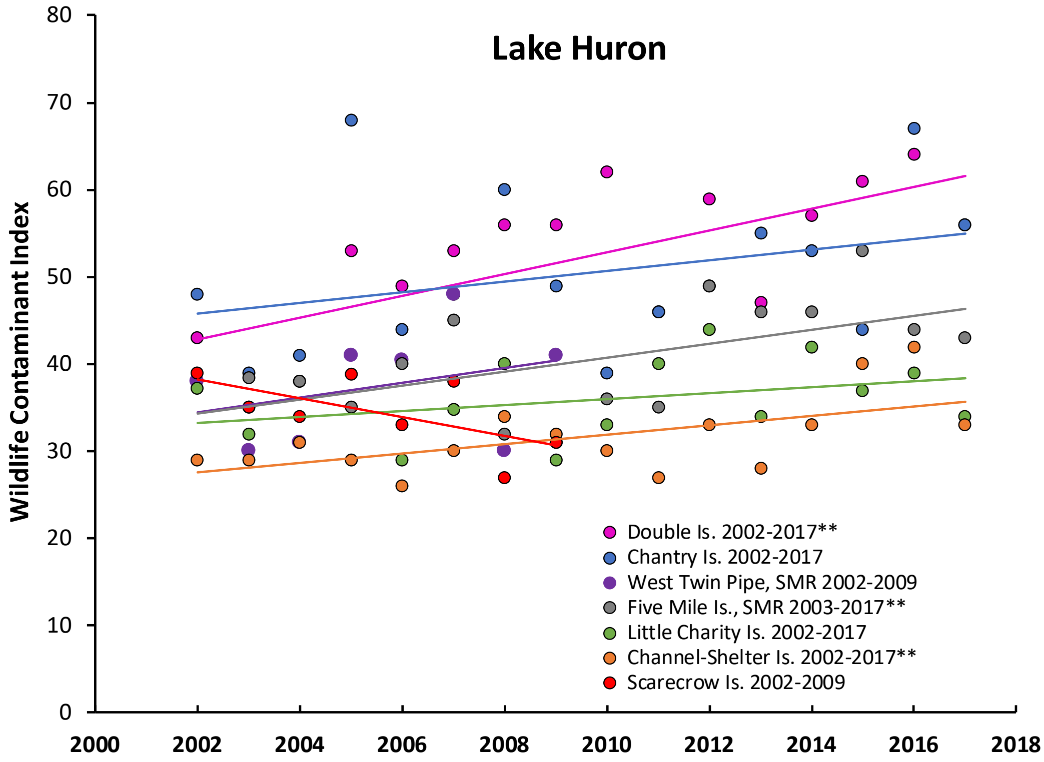


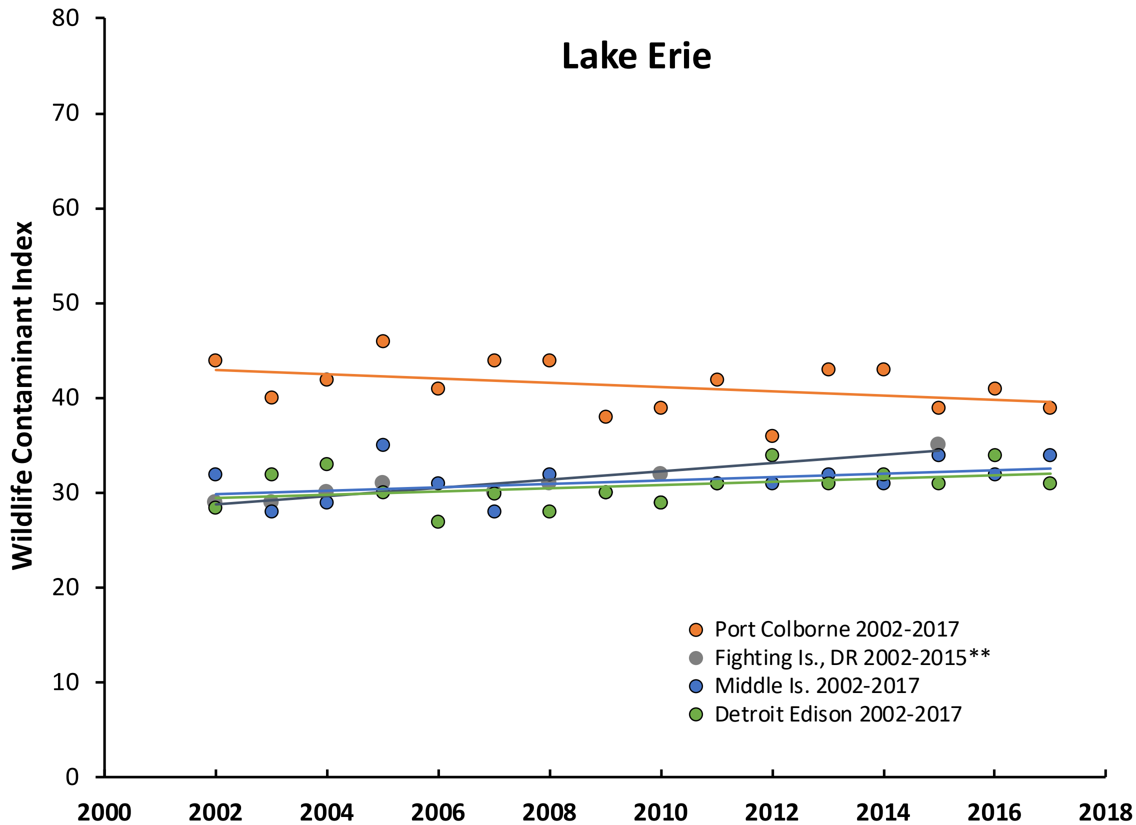
Figure 1. Temporal trends for the Wildlife Contaminant Index (WCI), based upon PCBs, PCDD/Fs and selected organochlorine pesticides, of Herring Gull colonies pooled by Great Lake. WCIs are significantly increasing which is indicative of improving conditions in all lakes as designated by "**", except for Lake Erie. Three colonies had regression slopes that differed significantly from the overall lake slope; Port Colborne (LE), Fighting Island (LE) and Scarecrow Is (LH). Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality].


Lake Superior
Wildlife Contaminant Index
0
10
20
30
40
50
60
70
80
2000
2002
2004
2006
2008
2010
2012
2014
2016
2018
Agawa Rocks 2002-2017**
Granite Is. 2002-2017**
Huron Is. 2002-2012
Net Is. 2002-2008
Tahquamenon Is. 2002-2009
Gull Is. 2013-2017


Lake Michigan
Wildlife Contaminant Index
0
10
20
30
40
50
60
70
80
2000
2002
2004
2006
2008
2010
2012
2014
2016
2018
Big Sister Is. 2002-2017**
Gull Is. 2002-2017**
Bellow Is. 2002-2017**
Green Is. 2002-2009
Indiana Harbour 2010



Lake Huron
Wildlife Contaminant Index
Double Is. 2002-2017**
Chantry Is. 2002-2017
West Twin Pipe, SMR 2002-2009
Five Mile Is., SMR 2003-2017**
Little Charity Is. 2002-2017
Channel-Shelter Is. 2002-2017**
Scarecrow Is. 2002-2009



Lake Erie
Wildlife Contaminant Index
Port Colborne 2002-2017
Fighting Is., DR 2002-2015**
Middle Is. 2002-2017
Detroit Edison 2002-2017

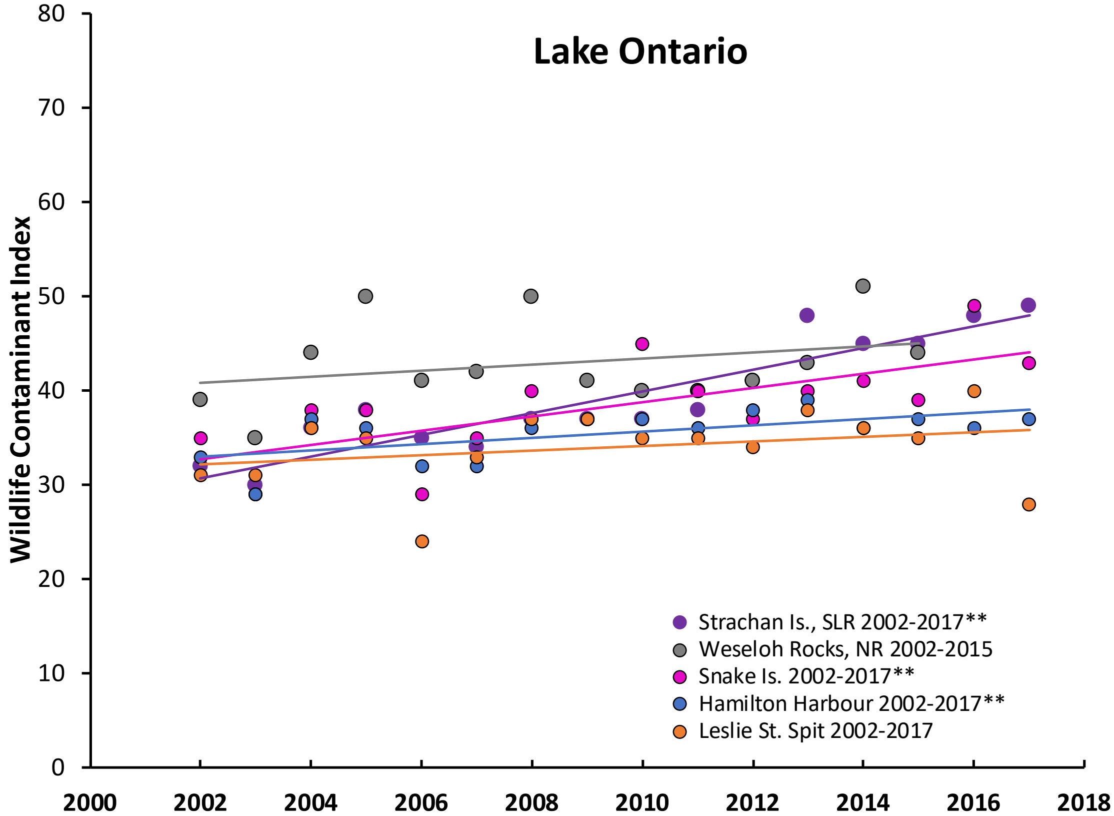


Figure 2. Temporal trends for the Wildlife Contaminant Index of each Herring Gull colony within each Great Lake, based upon PCBs, PCDD/Fs and selected organochlorine pesticides. "**" indicates colonies where linear regression analysis is significantly different from zero. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada] and Clean Michigan Initiative-Clean Water Fund [CMI-CWF; Michigan Department of Environmental Quality].

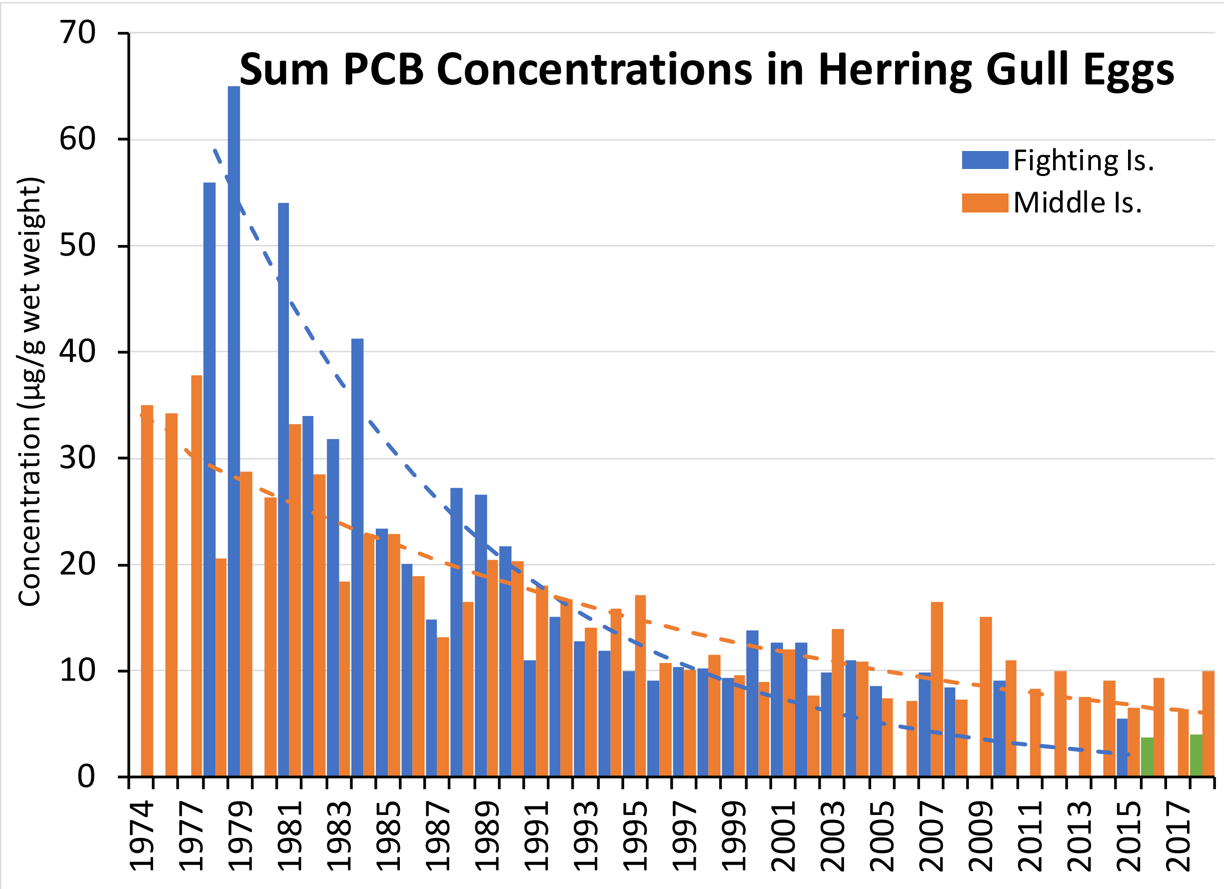


Figure 3. Sum PCBs in herring gull eggs from Detroit River (Fighting Island) and western Lake Erie (Middle Island) from 1974-2018. The green bars indicate PCB concentrations in cormorant eggs from Fighting Island in 2016 and 2018 since no data are available for Herring Gulls. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada].

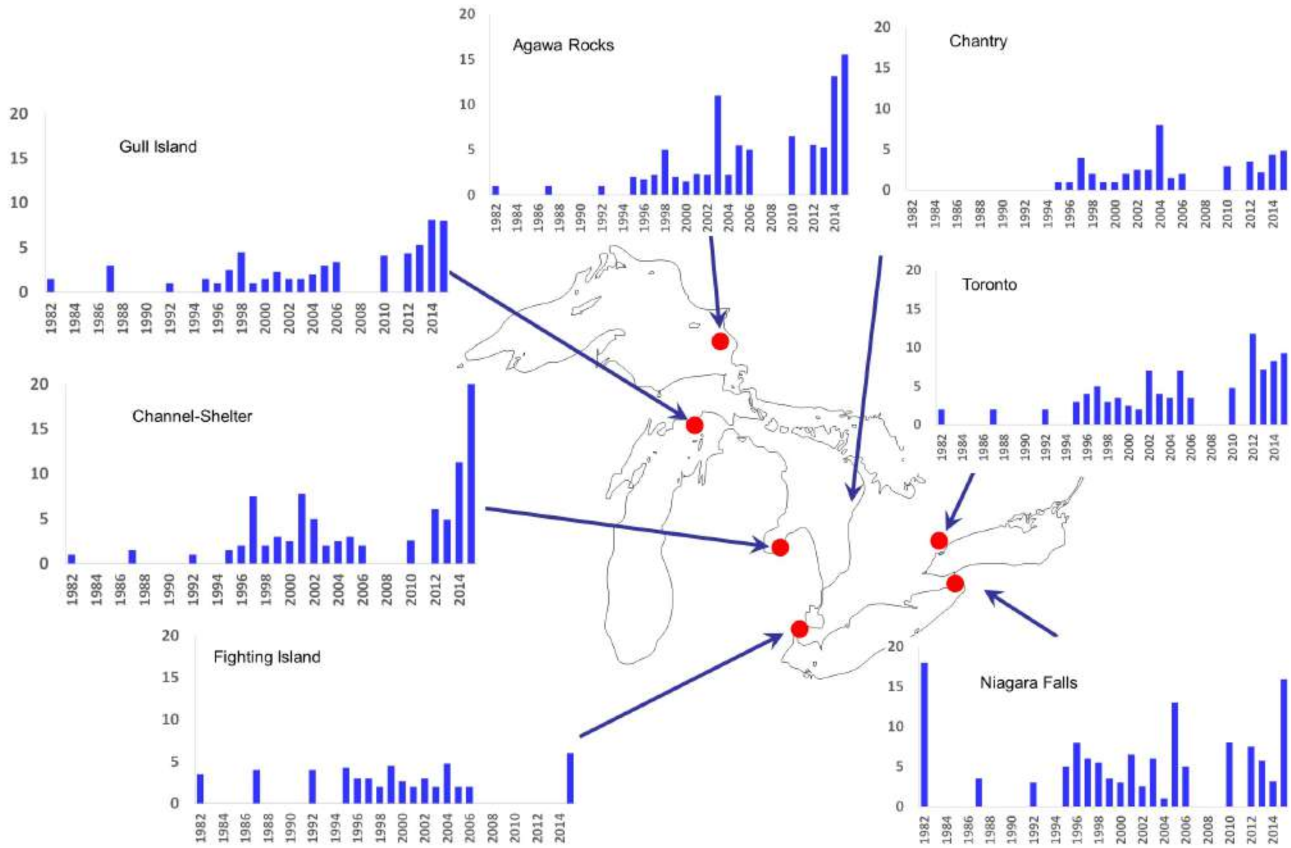


Figure 4. Concentrations of Dechlorane Plus (ng/g, wet weight) in Herring Gull eggs from selected colonies from the Great Lakes in 1982 to 2015. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada].

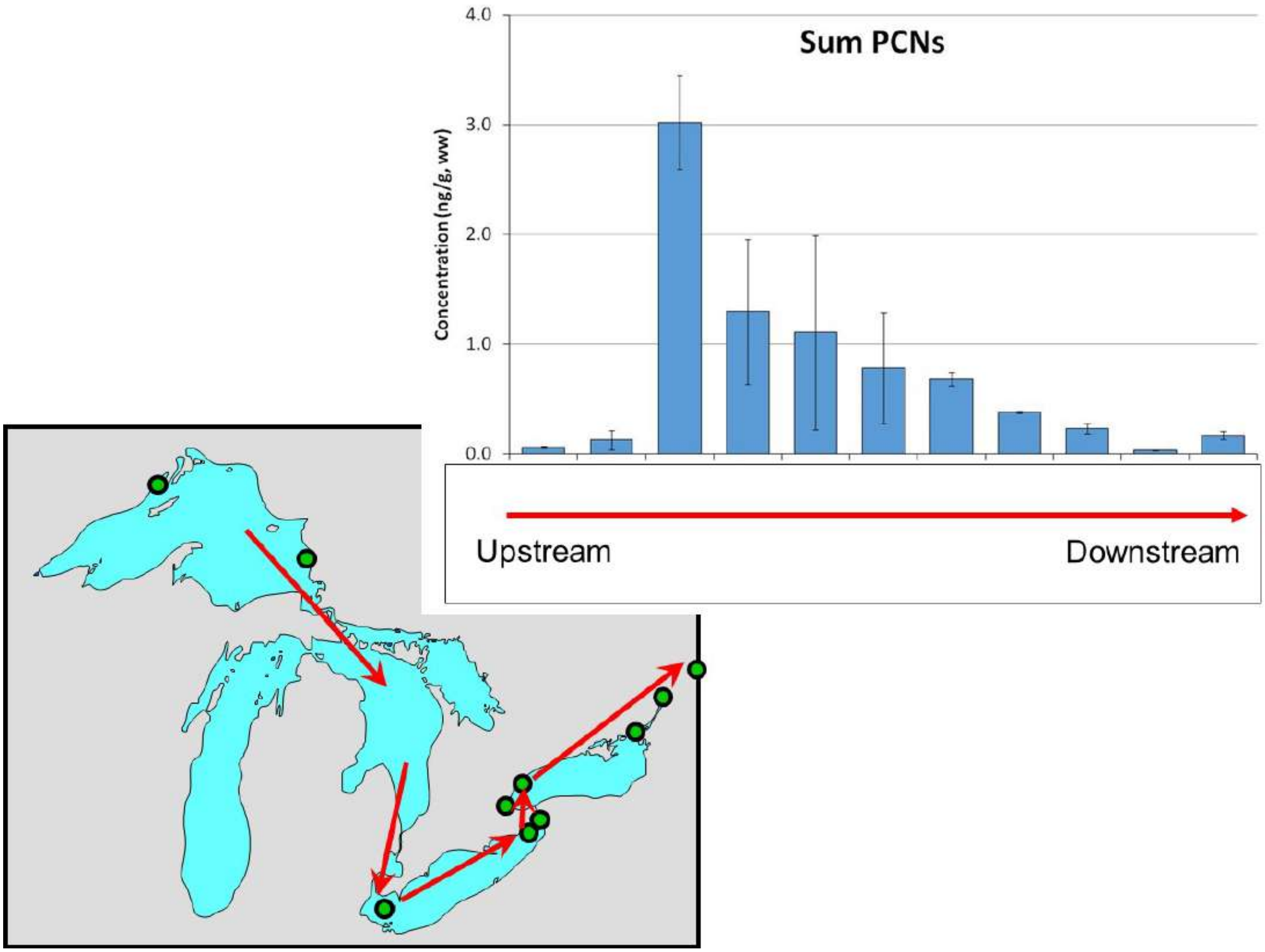


Figure 5. Concentrations of Polychlorinated naphthalenes (ng/g, wet weight) in Herring Gull eggs from selected colonies from the Great Lakes in 2011-2013 (unpublished data, ECCC; based upon McGoldrick et al, 2018). Colonies are ordered left-right from upstream-downstream; the peak corresponds to eggs from the Detroit River. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]; McGoldrick et al., 2018

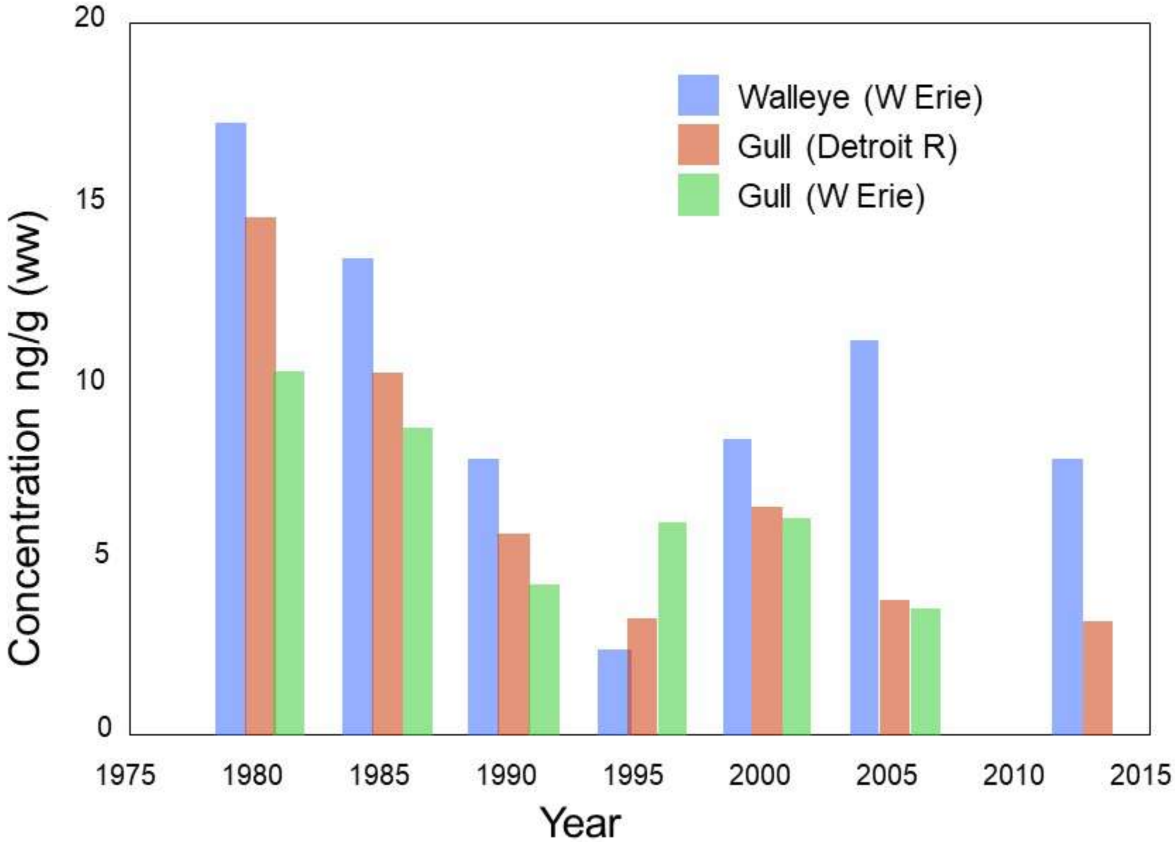


Figure 6. Concentrations of Polychlorinated naphthalenes (ng/g, wet weight) in Herring Gull eggs from the Detroit River and western Lake Erie, and Walleye from Western Lake Erie, from 1980 to 2015. In the mid to late-1990s, there was substantial dredging of PCN contaminated sediment from the Detroit River. Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]; McGoldrick et al., 2018.

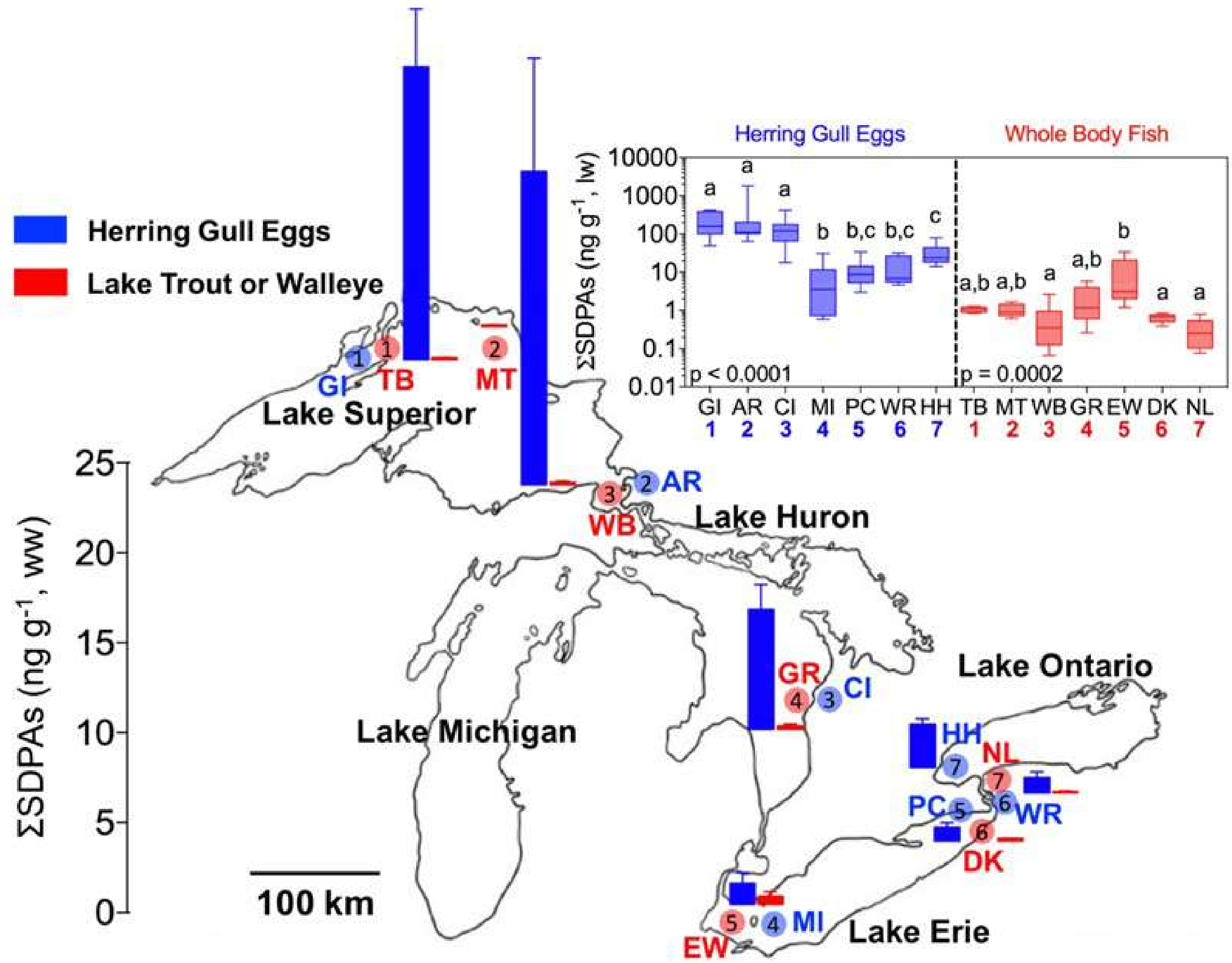


Figure 7. Concentration (mean ± standard error bar (ww) of ∑SDPAs in Herring Gull eggs (blue) and Lake Trout and Walleye whole body homogenate (red) in the Great Lakes of North America. Inset shows boxplot of lipid normalized ∑SDPAs concentrations. Source: Chemical Monitoring Program [Environment and Climate Change Canada]; Liu et al., 2018.

Sub-Indicator: Toxic Chemicals in the Atmosphere

Overall Assessment

Status: Fair

Trends:

10-Year Trend: Improving

Long-term Trend (1990s-2017): Improving

Rationale: The large surface area of the Great Lakes provides an opportunity for significant atmospheric inputs of toxic chemicals. While concentrations of some toxic chemicals are very low at rural sites, they are much higher in "hotspots" such as urban areas. Lake Michigan, Lake Erie, and Lake Ontario have greater inputs from urban areas. Of the rural sampling locations, the Lake Erie site (Sturgeon Point) tends to show higher levels than at other remote sites, most likely since it is located closer to an urban area (ex. Buffalo, NY), and may also receive some chemical input from the East Coast of the U.S.

The overall assessment is based on Chemicals of Mutual Concern (CMCs). The overall status and trend for CMCs in the atmosphere is Fair and Improving, which is consistent with the overall assessment for the 2019, 2017 and 2011 Toxic Chemicals in the Atmosphere reports (previously known as Atmospheric Deposition of Toxic Chemicals prior to 2019). Note: not all chemicals have been monitored over the full period noted. Six CMCs were used in the evaluation, with two not having enough data to establish a status or trend (see the listing of CMCs in the Ecological Condition section).

The majority of the compounds remain Fair and Improving. Deposition of PCBs (polychlorinated biphenyls), PBDEs (polybrominated diphenyl ethers), and mercury will continue to occur into the future due to on-going and legacy sources, and natural volatilization, but the rate of deposition is slightly decreasing. Per- and polyfluoroalkyl substances, or PFAS (including PFOA - perfluorooctanoic acid and PFOS - perfluorooctane sulfonate) compounds, have been assessed by Canada's Monitoring and Surveillance in the Great Lakes Basin (GLB) since 2005 (13 years of data available). Precipitation samples were collected, which provides good indications of atmospheric PFAS levels due to the polar character and water solubility of the compounds. PFOS and PFOA are showing slightly improving trends. However, there is an unchanging trend in many long-chain perfluorinated carboxylic acids (LC-PFCAs). Over the last several years hexabromocyclododecane (HBCDD, also referred to as HBCD) and Short Chained Chlorinated Paraffins (SCCPs) have become a higher priority for monitoring and assessment, but there is limited data for the Great Lakes region and not enough to make a status and trend assessment. Overall, the status and trends of toxic chemicals in the atmosphere of the Great Lakes has remained the same since the last report. HBCDD, SCCP, and PFAS compounds are now being monitored and included in the assessment. However, there are insufficient data to establish current status or temporal trends for SCCP and HBCDs.

Status and trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Each lake was not specifically categorized for status and trend because of limited sample stations for each lake basin to allow for a lake-by-lake assessment. Site specific trends for many chemicals (not all are CMCs) are available (Salamova et al. 2015; Shunthirasingham et al., 2016). There is no new loadings analysis since the last report.

Status Assessment Definitions

Good: The metrics show that the CMC concentrations and/or loadings are meeting the ecosystem objectives or they are otherwise in an acceptable condition.

Fair: The metrics show that the CMC concentrations and/or loadings are not meeting the ecosystem objectives, but they are exhibiting minimally acceptable conditions.

Poor: The metrics show that the CMC concentrations and/or loadings are not displaying minimally acceptable conditions and are severely impacted.

Undetermined: Data are not available or are insufficient to assess status of CMC concentrations at this time, or the different groups of CMC chemicals are assessed in Good and Poor rankings and an expert opinion of the overall status cannot be agreed on.

Trend Assessment Definitions

Improving: Decrease in concentrations or frequency of detection of CMCs.

Unchanging: No change in the concentration or frequency of detection of CMCs.

Deteriorating: Increased concentration or frequency of detection of CMCs.

Undetermined: Data are not available or are insufficient to assess the trends or frequency of detection of CMC concentrations at this time, or the different groups of CMCs are not trending in the same direction and an expert opinion of the overall direction of the trend cannot be agreed on.

The magnitudes of the trends are expressed as a "halving time", or time to which the concentration of the chemical is decreased by a factor of two. The halving time ($t_{1/2}$) was estimated by dividing *ln* 2 with the negative value of the linear regression slope of the trend line between the natural log of air concentrations, C (pg m^{-3}), and time, t (year, y). Positive halving time ($t_{1/2} > 0$) indicate a decreasing trend. Negative halving time (i.e. $t_{1/2} < 0$) (doubling time) indicates an increasing trend. Halving times are only used to compare the relative rates of decline between stations. Readers are advised to use the absolute values of these halving times with caution.

Time trends were developed using the Digital Filtration (DF) technique or linear regression. For the DF technique, an approximate long-term time trend and an average seasonal cycle were iteratively fitted to the measurement data using a smoothing Reinsch-type cubic spline and Fourier components, respectively. Outlier data points more than 3 standard errors away from the fitted curve were rejected after each iterative fit. When the fitted curves become unchangeable, digital filters are applied to extract the long- and short-term variations to refine the trend and seasonal cycle which are statistically significant with 95 % confidence. Details are described in Hung et al. (2005). For the linear regression, the log transformed concentrations (either as individual data points or annual medians) were regressed against time. Trends were considered significant at a 95% confidence level.

Endpoints and/or Targets

The target or endpoint for this sub-indicator will have been met when the Waters of the Great Lakes are free from pollutants in quantities or concentrations that could be harmful to human health, wildlife or aquatic organisms, and/or through direct exposure or indirect exposure through the food chain. The CMC status and trends will be determined on a case-by-case basis taking a weight-of-evidence approach in making an expert assessment.

Progress will be determined based on whether trends of the toxic chemicals are positive (i.e. increasing) or negative (decreasing), the rate of change in the concentrations, and by the number of chemicals which are doing so.

Sub-Indicator Purpose

- The purpose of this sub-indicator is to assess Chemicals of Mutual Concern (CMCs) in the atmosphere and in precipitation in the Great Lakes region.
- The sub-indicator will infer potential impacts of CMCs from atmospheric deposition on the Great Lakes aquatic ecosystem and progress toward virtual elimination of anthropogenic CMCs.
- The sub-indicator will also inform the risk assessment of potentially harmful chemicals and the development of risk management strategies for toxic substances other than CMCs, including persistent organic pollutants (POPs) and other harmful substances.

Ecosystem Objective

This sub-indicator is relevant to the General Objective #4 of the 2012 GLWQA that the waters of the Great Lakes "be free from pollutants in quantities or concentrations that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain." This sub-indicator is also relevant to Annex 3-Chemicals of Mutual Concern of the GLWQA, the purpose of which is to "reduce the anthropogenic release of chemicals of mutual concern, recognizing: (i) that chemicals of mutual concern released into the air, water, land, sediment, and biota should not result in impairment to the quality of the Waters of the Great Lakes; and (ii) the need to manage chemicals of mutual concern including, as appropriate, by implementing measures to achieve virtual elimination and zero discharge of these chemicals."

The Annex 3 further calls for the Parties to (i) monitor and evaluate the progress and effectiveness of pollution prevention and control measures; (ii) exchange, on a regular basis, information on monitoring, surveillance...; (iii) identify and assess the occurrence, sources, transport, and impact of chemicals of mutual concern, including spatial and temporal trends in the atmosphere...; (iv) identify and assess loadings ... from the atmosphere; and (v) coordinating research, monitoring, and surveillance activities as a means to provide early warning for chemicals that could become chemicals of mutual concern.

Measure

This sub-indicator measures the concentrations of Chemicals of Mutual Concern (CMCs) in the atmosphere and in precipitation of the Great Lakes region. This sub-indicator tracks whether concentrations are decreasing, staying the same, or increasing over time where data are available. The sub-indicator also assesses whether the spatial trends in concentrations are changing or staying the same. The monitoring data was used by the Parties to the GLWQA to inform the selection of CMCs for Annex 3 of the GLWQA in 2016. Monitoring data are used to inform the continued selection of chemicals of mutual concern as well as monitor to assess the progress and effectiveness of pollution prevention and control measures for those compounds.

The sub-indicator primarily reports using data from the Canada-US binational efforts in monitoring toxic substances in air and in precipitation in the Great Lakes region, including the Integrated Atmospheric Deposition Network (IADN), Great Lakes Basin (GLB) Monitoring and Surveillance Network under Canada's Chemicals Management

Plan (CMP) and the National Atmospheric Deposition Program's Mercury Deposition Network (MDN). These monitoring efforts follow well-established international monitoring protocols for sampling frequency, site selection, and sampling and analytical methods, with stations in the Great Lakes region.The sub-indicator also considers the most recent atmospheric loadings as estimated by IADN and GLB (see data limitations section).

CMCs that are relevant to this sub-indicator include PCBs, mercury, LC-PFCAs (C9-C20), PFOA, PFOS, PBDEs, HBCDD and SCCPs. Air concentrations, atmospheric deposition and loading information are reported in this sub-indicator report. For CMCs that do not have sufficient information to base a determination, air and atmospheric deposition information are gathered and presented in this sub-indicator but will not contribute to the Status and Trend Assessments. Air and atmospheric deposition data for Persistent Organic Pollutants (POPs) that are included under the United Nations Environment Programme (UNEP) Stockholm Convention on POPs and other chemicals that may be of concern in the future are also presented in this sub-indicator as additional information but only data on the CMCs are used for the status and trend assessments.

Precipitation samples are integrated over each month. Annual average concentrations are calculated and reported using these measurements. Spatial and temporal trend analyses are performed on the IADN and GLB data using a variety of statistical tools (Venier et al., 2012; Venier and Hites, 2010a; Venier and Hites, 2010b).

Weekly composite precipitation samples are collected and analyzed for mercury across the Great Lakes basin at MDN stations, using the same strict sampling and analytical protocols established under the network. Annual mercury concentrations, precipitation depths, and wet deposition are calculated and reported using this data. Spatial and temporal trend analyses are performed on the MDN data using a variety of statistical tools (Weiss et al., 2016; Risch et al., 2012; Prestbo and Gay, 2009).

Air concentrations respond rapidly to changes in emissions, ("environmental hysteresis") (Hites et al, 2021). This makes atmospheric measurements a very useful tool for tracking progress as a result of chemical control efforts. Since air has a short response time to changes in atmospheric emissions and is a relatively well-mixed environmental medium, this sub-indicator is also suitable in providing early warning for chemicals that could become CMCs. The occurrence of additional potentially harmful substances in Great Lakes air and precipitation are presented in this report but are not included in the Status and Trend Assessments.

Ecological Condition

The United States' Integrated Atmospheric Deposition Network (IADN) and Canada's Monitoring and Surveillance in the Great Lakes Basin (GLB) are the primary source of data for this sub-indicator report. IADN and GLB form a collaborative binational monitoring network that has been in operation since 1990, with five master monitoring stations, one near each of the Great Lakes, and several satellite stations (Figure 1). The Burnt Island Master station was closed by March 2013 and is replaced by the nearby Evansville Station for both air (started January 2014) and precipitation (started July 2013) measurements starting in 2014. Concentrations of Persistent Bioaccumulative Toxic (PBT) chemicals are measured in the atmospheric gas and particle phases and in precipitation. Spatial and temporal trends of these concentrations and atmospheric loadings to the Great Lakes can be examined using these data. CMC data from other networks and surveillance studies are used here to supplement the IADN and GLB data, particularly for mercury and SCCPs. In this assessment, only CMC data are used to assess this sub-indicator while in earlier assessments (prior to 2019), other contaminants (non-CMCs) were also considered in assessing the status and trend. Despite the difference in chemicals used for the 2022 and 2019 assessments as compared to that for 2017, the overall status and trends remain the same. However, some of the CMCs, e.g. HBCDD/HBCD and SCCPs, do not have sufficient data to assess their status and trends under this sub-indicator. Although the 2022 assessment focuses on CMCs, trends of non-CMCs are also included in this report for reference.

Status of Chemicals of Mutual Concern (CMCs)

The Parties to the GLWQA designated the first set of Chemicals of Mutual Concern in May 2016. For State of the Great Lakes reporting in 2019 and moving forward, the toxic chemical sub-indicators will support reporting on Chemicals of Mutual Concern (CMCs) in a more consistent and transparent way, based on available data. Information on additional chemicals of interest for the Great Lakes (non-CMCs) is valuable for inclusion in the report and will be included in a separate section below, as appropriate.

The following chemicals were identified as the first set of CMCs:

- Polychlorinated Biphenyls (PCBs)
- Polybrominated Diphenyl Ethers (PBDEs)
- Perfluorooctane sulfonate (PFOS)
- Perfluorooctanoic acid (PFOA)
- Long-Chain Perfluorinated carboxylic acids (LC-PFCAs)
- Mercury (Hg)
- Hexabromocyclododecane (HBCDD/HBCD)
- Short-Chain Chlorinated Paraffins (SCCPs)

Polychlorinated Biphenyls (PCBs)

Canadian data have been updated up to 2017 (Shunthirasingham et al., 2016; Hung et al. 2021). Burnt Island and Egbert ceased operation since 2013 and 2016 respectively, hence there is no new data for these two sites since the last report. US data have been updated to 2018. The half-lives of individual PCBs in the Canadian air are given in Table 1. The atmospheric concentrations of ∑PCBs (sum of 28 congeners, including 18, 28, 31, 44, 52, 70, 77+110, 100, 101+89, 105, 114, 118, 119, 126, 131, 138+163, 153, 156+171, 170, 172, 180, 199, 201, 202) continue to decline on both sides of the lakes with halving times of about 12 to 31 years on the US side and 17-74 on the Canadian side (Figures 2 and 3). There is no change in the halving times of PCBs in Point Petre to those in the last report. The rather long half-lives (especially at Cleveland, Burnt Island and Egbert) imply the tendency towards equilibrium with surrounding environmental surfaces. The 10-year trend at all sites (i.e. 2008-2017 for Point Petre and 2009-2018 for US sites) shows that PCB concentrations are decreasing, with halving times in range of 9 to 74 years and 6 to 12 at Canadian and US sites, respectively, which is similar to the overall trend for Canadian sites (i.e. 1992-2017) but faster for US sites.

Salamova et al. (2015) have found no differences in the halving times of PCBs among the five U.S. sites and Point Petre, suggesting a relatively homogeneous decrease rate in the Great Lakes region. The concentrations of PCBs were declining also in precipitation with a halving time of about 10 years (Venier et al., 2016).

Although PCB production was banned in the early 1970s in North America, the slow decline in air concentrations can be attributed to volatilization from the lakes themselves (Khairy et al. 2015), from building sealants (Robson et al., 2010; Shanahan et al. 2015), from drying sewage sludge (Shanahan et al. 2015), and from paints (Hu and Hornbuckle, 2010). In addition, there are continued emissions from older electrical and hydraulic equipment still in use and in the waste stream (Diamond et al., 2010). Urban areas are believed to be the main sources of PCBs to rural regions (Buehler et al. 2001; Hafner and Hites 2003; Cleverly et al. 2007; Melymuk et al., 2012; Shunthirasingham et al. 2016).

There is no new data about the loading of PCBs since the last report. Based on the last report, volatilization of PCBs from the lakes is shown by atmospheric loadings analysis (Shunthirasingham et al. 2016; Guo et al., 2018). The output flows of PCB-52, 101, and 118 due to volatilization from the lakes to the air are increasing exponentially with time (Figure 4 of Guo et al. (2018) as an example). In these three cases, the flows are doubling every 4–9 years. On the other hand, the input flows of the PCBs are generally small compared to the output flows. In addition, PCBs are likely being released from the sediment into the lake water from which they could volatilize into the atmosphere. Once in the atmosphere, PCBs could be rapidly degraded by reactions with hydroxyl radicals (Anderson and Hites, 1996). Clearly, the lakes are slowly releasing PCBs to the atmosphere, but it is not yet clear how long this process will take.

3,3'-Dichlorobiphenyl (PCB-11) is thought to be a byproduct of the production of yellow pigments, and thus, it has sources to the environment that differ both in type and magnitude compared to the PCBs that made up the, now banned, Aroclor commercial products. The atmospheric concentrations of the Aroclor-PCBs are decreasing with halving times of about 10-15 years, but the atmospheric concentrations of PCB-11 have not changed significantly over this time period (Figure 4). These results suggest that PCB-11 is still leaking into the environment, while at the same time sources of Aroclor-PCBs are decreasing. This effect is particularly notable at the most remote site on Lake Superior, where PCB-11 levels are, on average, 11% of those of total Aroclor-PCBs; indicating an abundance of a single PCB congener (Hites, 2018).

Guo et al. (2017) revisited and updated the Lake Michigan Mass Balance Project (LMMBP) for PCBs that was conducted in 1994–1995. Recent concentrations of PCBs in tributary and open lake water, air, and sediment were used to calculate an updated mass budget. Five of the eleven LMMBP tributaries were revisited in 2015. In these five tributaries, the geometric mean concentrations of ∑PCBs (sum of 85 congeners) ranged from 1.52 to 22.4 ng/L. The highest concentrations of PCBs were generally found in the lower Fox River and in the Indiana Harbor and Ship Canal. The input flows of ∑PCBs from wet deposition, dry deposition, tributary loading, and air to water exchange, and the output flows due to sediment burial, volatilization from water to air, and transport to Lake Huron and through the Chicago Diversion were calculated, as well as flows related to the internal processes of settling, resuspension, and sediment-water diffusion (Figure 5). The net transfer of ∑PCBs is 1240 ± 531 kg/yr out of the lake water. This net transfer is 46% lower than that estimated in 1994–1995. Overall, PCB concentrations in most matrices in the lake are decreasing, and as a result, the input and output flows are declining. Atmospheric deposition has become negligible, while volatilization from the water surface is still a major route of loss, releasing PCBs from the lake into the air. Large masses of PCBs remain in the water column and surface sediments and are likely to contribute to the future efflux of PCBs from the lake to the air.

Polybrominated Diphenyl Ethers (PBDEs)

The concentrations of halogenated flame retardants have been measured in IADN/GLB samples since January 2005. Liu et al. (2016) found that the atmospheric concentrations of polybrominated diphenyl ethers (PBDEs) were significantly higher in Chicago, Cleveland, and Sturgeon Point than at Sleeping Bear Dunes and Eagle Harbor. The concentrations of PBDEs are generally decreasing at the urban sites at Chicago and Cleveland, but unchanging at the remote sites, Sleeping Bear Dunes and Eagle Harbor.

In Chicago, the concentrations of BDE-47 and 99 decreased by a factor of two every 5.9 ± 0.9 and 8.0 ± 1.4 years, respectively, but the concentrations of BDE-209 doubled every 7.6 ± 1.8 years. In Cleveland, the concentrations of BDE-47 and 99 decreased by a factor of two every 5.1 ± 0.4 and 5.7 ± 0.5 years, respectively, and the concentrations of BDE-209 decreased by a factor of two every 9.2 ± 1.6 years. The delay in all these environmental responses relative to when production stopped is likely a result of decadal scale environmental hysteresis (Hites et al., 2021). Use and emissions from products are gradually diminishing as products are removed from use, resulting in a slower declining trend than expected. Over the ten year trend period, the halving time is in the range of 4 to 9 years for BDE-47, 3 to 8 for BDE-99 and 5 to 13 for total PBDEs. For BDE-209, the 10 years trend was significant only for Chicago.

At Point Petre, data was updated to 2017 (i. e. 2005-2017, based on (Free; Hung et al. 2021). There is no new data for Burnt Island and Egbert since the last report; hence the data reported here for these two sites are from 2005-2013. Half-lives of PBDEs in the Canadian sites are given in Table 2. At Point Petre, BDE-47, BDE-99, and BDE-209 were the dominant PBDE congeners. The annual average of the total concentrations of the 15 congeners (Σ_{15}PBDEs) ranged from 3.1-18 pg/m^3 and an annual median of 2.5-12 pg/m^3. At Burnt Island, the annual average of Σ_{15}PBDEs was 14-60 pg/m^3 and an annual median of 6.3-23 pg/m^3. At Egbert, the annual average of Σ_{15}PBDEs was 11-63 pg/m^3 and an annual median of 9.0–21 pg /m^3. High atmospheric concentrations of PBDEs were found at Egbert with a larger population (244, 446 within a 25 km radius of the site), while lower levels of PBDEs were detected at Point Petre, which is a remote site on Lake Ontario, with population of 15 966 within a 25 km radius) (Free map 2015). The time trends of BDE-28, -47, -99 and -209 in air at Point Petre were analyzed by the Digital Filtration method (Hung et al., 2016) and results are shown in Figure 6. The atmospheric PBDEs from 2005-2017 are decreasing slowly, in which with half-lives of BDE-47, 99, 209 congeners are, respectively, 7.7, 5.9 and 12 years. BDE-28 is not changing with a very long half-life of 31 years. The 10-year trend for Point Petre (i.e. 2008-2017) for the PBDE are the same as the overall trend with in which half-life of BDE-28 = -24 years, BDE-47 = 13 years, BDE-99 = 5.5 years, and BDE-209 = 6.6 years.

Perfluorooctane Sulfonate (PFOS)

Time trend of PFOS in GLB precipitation at Point Petre (Lake Ontario), Burnt Island/Evansville (Lake Huron) and Sibley (Lake Superior) between 2006 and 2018 was reported by Gewurtz et al. (2019). The median concentration of monthly-integrated precipitation for Point Petre was 1.4 ng/L, Burnt Island was 0.89 ng/L. PFOS in precipitation from Sibley was below detection limit. The halving time derived by the DF method at Point Petre is 12 years, Burnt Island is 6.2 years and Sibley is 8.3 years, which means PFOS was decreasing over time (Figure 7, right panel). The overall decreasing trends were likely in response to North American phase-outs and regulatory actions for PFOS long-chained perfluoroalkyl carboxylates (PFCAs) and their precursors, ongoing since the early 2000s (Interstate Technology and Regulatory Council PFAS; Government of Canada, 2012). Concentrations of PFOS in Great Lakes precipitation are greater in the urbanized/industrialized Lake Ontario compared to the more remote locations of Lakes Huron and Superior, indicating the effect of local anthropogenic sources on atmospheric concentrations of PFOS.

There is no new data for atmospheric PFOS in the GLB region since the last report. In 2009, PFOS concentrations (geometric mean) measured using a high-volume air sampler were more than three times higher in Toronto (1.5 pg/m^3) (Ahrens et al., 2011) compared with Lake Superior air (0.43 pg/m^3) (Gewurtz et al., 2013). Air sampling of PFAS in GLB has commenced in October 2018 at Point Petre (Lake Ontario) and in July 2019 at Evansville (Lake Huron) but no results are available yet during this review period.

PFAS samplers have been established on the US side in summer/fall 2020. However, there are no data updates for PFOS for US sites.

Perfluorooctanoic Acid (PFOA)

PFOA is measured in precipitation under GLB at Point Petre (Lake Ontario), Burnt Island/Evansville (Lake Huron) and Sibley (Lake Superior) from 2006-2018 (Gewurtz et al., 2019). The median concentrations of monthly integrated precipitation samples at Point Petre = 0.59 ng/L, Burnt Island = 0.42 ng/L, Sibley = 0.48 ng/L. Time trend analysis by the DF method showed that PFOA is declining at the 3 sites. The halving times are respectively 5.9, 6.5 and 11 years for Point Petre, Burnt Island/Evansville and Sibley (Gewurtz et al., 2019) (Figure 7, left panel).

Air sampling of PFAS in GLB has commenced in October 2018 at Point Petre (Lake Ontario) and in July 2019 at Evansville (Lake Huron) but there is no new data for atmospheric PFOA in the GLB region yet since the last report. The only air monitoring data for PFOA in the Great Lakes region which is currently available is at the urban site of Toronto. In 2010, PFOA was measured using an active air sampler with a reported concentration range of nd-1.4 pg/m^3 (Ahrens et al., 2012).

PFAS samplers have been established on the US side in summer/fall 2020. However, there are no data updates for PFOA for US sites.

Long-chain Perfluorinated Carboxylic Acids (LC-PFCAs)

Under GLB, LC-PFCAs (C9-12) concentrations in precipitation are available at Point Petre, Burnt Island/Evansville and Sibley. At the 3 sites, the medians of Σ_4LC-PFCA (C9-12) ranged from 0.49 to 0.57 ng/L. Perfluorononanoic acid (PFNA), perfluorodecanoic acid (PFDA) were significantly declining with halving times <15 years determined by the DF technique (Gewurtz et al., 2019). Although perfluoroundecanoic acid (PFUnA) and perfluorododecanoic acid (PFDoA) are also long-chained PFCAs, their concentrations did not monotonically decrease between 2006 and 2018. PFUnA and PFDoA are less water-soluble than PFNA, PFDA, and PFOA (Bhhatarai et al., 2011) and percent detection was relatively low (46% and 31%, respectively) for these chemicals in Great Lakes precipitation, which may be the reason behind their observed trends (Gewurtz et al., 2019).

Air sampling of PFAS in GLB has commenced in October 2018 at Point Petre (Lake Ontario) and in July 2019 at Evansville (Lake Huron) but there is no new LC-PFCA air data in the GLB region yet since the last report.

PFAS samplers have been established on the US side in summer/fall 2020. However, there are no data updates for LC-PFCAs for US sites.

Mercury

Atmospheric mercury has been measured in the Great Lakes Basin air shed in Ontario at three stations and one in Quebec, Canada. These stations include Egbert (~70 km north of Toronto, between Lake Huron and Lake Ontario; Figure 1), Point Petre (180 km east of Toronto on the shore of Lake Ontario) and Burnt Island (100 km north of Toronto on Georgian Bay off of Lake Huron). Egbert has been collecting Total Gaseous Mercury (TGM) since 1996 and continues today. Point Petre and Burnt Island measured TGM from 1996-2007 and 1998-2007, respectively. St. Anicet (located along the St Lawrence River, ~200 km east of eastern Lake Ontario) has been collecting mercury since 1994 and continues today.

Total Gaseous Mercury (TGM) at Egbert had a mean concentration of 1.51 ± 0.31 ng/m^3 from December 1996 - 2016. During this measurement period, the temporal trend decreased by -1.7 % per year (range of -1.9 to -1.5%) (GMA, 2018). Wet deposition measured at this site reported mean concentration of 8.0 ng/L between April 1998 and August 2007 (GMA, 2018). The annual trend for wet deposition was not found to be significant with a range of -1.2 % to +0.6 % per year (GMA, 2018). The annual Gaseous Elemental Mercury (GEM) concentration from St. Anicet was 1.53 ± 0.35 ng/m^3 from August 1994 to December 2016. During this measurement period, the temporal trend decreased by 1.5% per year (range -1.6 to -1.3%). Wet deposition measured at this site reported a mean concentration of 8.1 ng/L between March 2000 and December 2015 where the annual trend decreased by 3.0 % per year (range -5.3 to -1.0%) (GMA, 2018).

In earlier years, the average TGM concentrations reported from Point Petre and Burnt Island were 1.75 ± 0.33 ng/m^3 and 1.55 ± 0.22 ng/m^3, respectively, with declining trends of -1.7 % per year (range of -2.2 to -1.2%) and -2.5% per year (-3.4 to -1.6%), respectively. Wet deposition of mercury was measured at both Point Petre (Nov 2001 – Mar 2003) and Burnt Island (Dec 2001 – Mar 2003) with reported mean concentrations of 8.4 and 10.1 ng/L, respectively (Cole et al., 2014). Atmospheric gaseous elemental mercury (GEM) concentrations were measured at

Canada's Experimental Lakes Area (west of Lake Superior) from 2005 to 2013. The mean concentrations varied at this site and ranged between 1.21 and 1.62 ng/m^3 (±0.003). Trend analysis from this site showed a decrease of 2.2 ± 0.06 % per year during the same time period (St. Louis et al., 2019).

Wet deposition measurements of mercury from the National Atmospheric Deposition Network's (NADN) Mercury Deposition Network (MDN) have been made in many US states surrounding the Great Lakes since 1996. Figure 8 shows the annual concentration of mercury in precipitation and wet deposition of mercury for both 2003 (the first year of maps, top panel) and for 2019 (the most recent year for data available, bottom panel). Average concentrations in the Great Lakes areas have been generally reported low over time (Prestbo and Gay, 2009; Zhang et al., 2016). However, an in depth analysis of the wet deposition trends show that some trends in mercury concentration have increased over short periods at some sites within or surrounding the Great Lakes region (Weiss et al., 2016). In general, the MDN trends follow these Canadian trends with fluxes decreasing about 1.6% per year since 1996. All the NADP observations can be found online (http://nadp.slh.wisc.edu/MDN/). The NADP also makes atmospheric concentration measurements in its Atmospheric Mercury Network (AMNet) but have few sites in the Great Lakes area.

Lepak et al. (2015, 2018) used stable isotope signatures to determine sources of mercury in Great Lakes sediment and predatory fish. It was found that atmospheric sources dominate in the sediment of Lakes Huron, Superior, and Michigan, while watershed-derived and industrial sources dominate sediment in Lakes Erie and Ontario. However, isotope signatures in predatory fish, such as Δ200Hg, which is conserved during biogeochemical processing in Lakes Ontario, Superior and Michigan, showed that bioaccumulated mercury is more isotopically similar to atmospherically derived mercury than to a lake's sediment. This finding suggests that atmospherically derived mercury may be a more important source of methyl mercury (methyl mercury is a more toxic form that is biomagnified in aquatic food webs) to higher trophic levels than sediments in the Great Lakes.

Hexabromocyclododecane (HBCDD/HBCD)

In 2014, the median HBCDD concentrations measured in atmospheric particles in Chicago = 2.0 pg/m^3, Cleveland = 2.1 pg/m^3, Sturgeon Point = 1.7 pg/m^3 and Sleeping Bear Dunes = 5.2 pg/m^3. Higher levels of HBCDD were observed at the remote site of Sleeping Bear Dunes with comparable levels at the remaining three sites. Higher levels at the remote site of Sleeping Bear Dunes suggests either an unknown source nearby or site-specific long-range atmospheric transport of HBCDD to this site. α-HBCDD and γ-HBCDD were the dominant isomers with an average contribution of about 40% and 50% to total HBCDD concentrations, respectively (Olukunle, et al., 2018).

HBCDD is monitored in air samples collected at Point Petre and Burnt Island and is reported as the sum of the three isomers. Annual mean concentrations were found to be 0.17-8.7 pg/m^3 at Point Petre from 2009-2017 (Shunthirasingham et al. 2018, Wong et al., *in prep.*); and 0.054-0.38 pg/m^3 for Burnt Island from 2009-2013 respectively (Shunthirasingham et al. 2018). HBCDD tended to show an increasing trend with doubling time of 1.6 year but as it was only detected in 22% of all samples, this value is to be evaluated with caution as it may not be representative (Shunthirasingham et al. 2018, Wong et al., *in prep.*) (Figure 9).

Short-Chain Chlorinated Paraffins (SCCPs)

At Point Petre, selected air samples collected in 2013 (*n* = 4) were analyzed for SCCPs. The mean concentrations was 410 pg/m^3 with range of 175 to 613 pg/m^3 (Hung et al., 2021). Under GAPS, SCCPs have been measured in air using passive an air sampling method at the Toronto site. The concentrations found were 221.1 pg/m^3 in 2006, 5.5 pg/m^3 in 2009 Jan-Mar, 97.9 pg/m^3 in 2009 Oct-Dec and 77.7 pg/m^3 in 2011 (Harner et al., 2014; Jasmin Schuster, personal communication). The analysis was performed following the method of Tomy et al. (1997). Although SCCPs is a CMC, it is not included in the assessment of this indicator due to the lack of data.

Other Chemicals of Interest

Information on chemicals presented in this section are not included in the assessment of the status and trend of this sub-indicator as they are not CMCs.

Organochlorine Pesticides (OCPs)

Concentrations of OCPs that have been banned are generally declining in air in the Great Lakes Basin. There is no new OCP data for Burnt Island and Egbert of the Canadian sites. Data from the Canadian site of Point Petre are updated to 2017. Half-lives of OCPs in the Canadian sites are given in Table 3. Chlordanes, dieldrin, and DDT-related substances show halving times in the range of 7-13 years at the US sites (Salamova et al. 2015) and 7-12 years at Point Petre from 1992-2017 (Shunthirasingham et al. 2016, Hung et al. 2021). Concentrations of α-HCH and γ-HCH are decreasing rapidly in air, with halving times (by DF method) of approximately 4 years at Point Petre (Shunthirasingham et al. 2016, Hung et al. 2021) and about 4 years at U.S. sites (Salamova et al. 2015 and Venier, unpublished data); see Figures 10 and 11. The 10-year time trend (2008-2017 for Canadian sites or 2009-2018 for US sites) for these compounds is similar to those of the overall time trend (1992-2017/2018). At US sites the 10-year trend was significant only at the remote sites of Eagle Harbor, Sleeping Bear Dunes and Eagle Harbor with halving times of about 7 years, which indicate a slower decline than for the whole dataset. These are the most rapid halving times observed for any compound measured as part of IADN/GLB.

Endosulfan is the last insecticide among the legacy pesticides to be removed from the market. Its phase-out began in 2010, when the EPA signed a Memorandum of Agreement with this compound's registrants. All allowed uses were progressively eliminated since then, with the last four terminated in 2016. Endosulfan exists in two isomeric forms, endo and exo, known popularly as I and II. Endosulfan sulfate is an oxidation product of endosulfan. Therefore, sometimes, in reporting, concentrations of endosulfan I, endosulfan II and endosulfan sulfate are summed together (e.g. Figure 12). Their vapor phase atmospheric concentrations are decreasing with halving times respectively of 4.8 and 5.2 years from 1992-2017 at Point Petre (Shunthirasingham et al. 2016; Hung et al. 2021) and 6.4 to 8.3 years at US sites (Salamova et al., 2015 and Venier, unpublished). Figure 10 and 12 show the time trends for endosulfan. Its concentrations in precipitation are decreasing with halving times of about 8 years at all sites (Venier et al., 2016). The 10-year time trend for endosulfan at Point Petre showed rapid decreasing trend with half-life of about 1.5 year for both isomers. The 10-year trend for vapor phase concentrations at the US sites showed a faster decline than for the whole dataset (i.e 1995-2018) with halving times between 2 and 3 years. Based on estimated use rates of endosulfan in the U.S. from 1997 to 2009, Salamova et al. (2015) estimated that endosulfan has an atmospheric chemical degradation rate of about 4 years, which suggests that endosulfan is less persistent in the environment than related compounds.

The atmospheric concentrations of DDTs (reported as the sum of DDT, DDD and DDE congeners) continue to decline with halving times ranging from 10.3 ± 0.9 for Eagle Harbor to 17.8 ± 7.3 for Point Petre. The ten year trend for gas phase concentrations was significant only at Eagle Harbor and Cleveland, with halving times of 10.7 ± 5.4 and 8.7 ± 4.9, respectively (Figure 13).

There is no update on the loadings of OCPs to the lakes since the last report. Loadings calculations up to 2010 suggest that the atmosphere is a source of endosulfan and p,p'-DDT to the lakes and that the lakes are a source of p,p'-DDE to the atmosphere (Shunthirasingham et al. 2016).

The input and output flows to the lakes of both α- and γ-HCH are decreasing exponentially, and for both of these compounds we are soon approaching a time when the air and water concentrations will be at equilibrium, and there will be no net transfer across the air-water interface (Guo et al., 2018). The changes over the 1992-2015 period for the other pesticides are variable, but with the exception of t-nonachlor (for which there is the least water data), the input and output flows are decreasing or not changing significantly. This seems to suggest that air-water transfer for these compounds is slowly approaching equilibrium. Incidentally, of these pesticides, only endosulfan was still on the market in North America during the study period of Guo et al. (2018), i.e. 2010-2015. Between 1992 and 2015, its use in the U.S. decreased (USGS, 2017), before being completely phased out in 2016 (U.S. EPA, 2010). Canada started to phase out the use of endosulfan in 2010, and all registrations of endosulfan-containing pesticide products expired on December 31, 2016 (ECCC, 2012).

Current Use Pesticides (CUPs)

In the United States, pesticide use has increased over the last few decades, and atrazine, chlorothalonil, pendimethalin, chlorpyrifos, and metolachlor are now among the most commonly used pesticides in agriculture, whereas permethrin and other pyrethroid insecticides are now among the most commonly used pesticides in homes and gardens.

The concentrations of 15 CUPs, including nine pyrethroid insecticides, four herbicides, one organophosphate insecticide, and one fungicide, were measured in 24-hour air samples collected at IADN stations in 2017. Thirteen individual CUPs were detected at least once in both gas- and particle-phase samples, with chlorothalonil, trifluralin, metolachlor, λ-cyhalothrin, cypermethrin, and chlorpyrifos detected in >50% samples. Median total CUP concentrations were 339, 238, 84, 33, 60, and 6.0 pg/m3 at Chicago, Cleveland, Sturgeon Point, Point Petre, Sleeping Bear Dunes, and Eagle Harbor, respectively. The concentrations of total CUPs and most individual CUPs were generally higher at urban sites of Chicago and Cleveland than at rural/remote sites of Sturgeon Point, Point Petre, Sleeping Bear Dunes, and Eagle Harbor. Chlorothalonil, trifluralin, bifenthrin, and chlorpyrifos were the most abundant individual CUPs among fungicides, herbicides, pyrethroid insecticides, and other insecticides, respectively. The spatio-seasonal variation suggests that fungicides at Sturgeon Point and Sleeping Bear Dunes, with the highest fraction of agricultural land use, were associated with agricultural activities while pyrethroid insecticides were generally driven by non-agricultural human activities. (Wang et al., 2021).

Non-BDE Flame Retardants

The concentrations of halogenated flame retardants (HFRs) have been measured in IADN/GLB samples since January 2005. Specifically, the atmospheric concentrations of eight non-PBDE halogenated flame retardants (HFRs) [pentabromoethyl benzene (PBEB), hexabromobenzene (HBBz), 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB), *bis*(2-ethylhexyl)-tetrabromophthalate (BEH-TEBP), *syn*-Dechlorane Plus (*syn*-DDC-CO), *anti*-Dechlorane Plus (*anti*-DDC-CO), 1,2-*bis*(2,4,6-tribromophenoxy)ethane (BTBPE), and decabromodiphenylethane (DBDPE)] were measured in each IADN sample. The levels of almost all of these flame retardants, except for PBEB, HBBz, and DP, were significantly higher in Chicago, Cleveland, and Sturgeon Point. The concentrations of PBEB and HBBz were relatively high at Eagle Harbor and Sturgeon Point, respectively, for unknown reasons, and the concentrations of DP were relatively high at Cleveland and Sturgeon Point, the two sites closest to this compound's production site in Niagara Falls, New York (Lui et al., 2016).

The concentrations of PBEB were decreasing at almost all sites except for Eagle Harbor, where the highest PBEB levels were observed. HBBz concentrations were decreasing at all sites except for Sturgeon Point, where HBBz levels were highest. The reason for the relatively high levels of PBEB and HBBz at Eagle Harbor and Sturgeon Point are not clear. DP concentrations were increasing with doubling times of 3-9 years at all sites except Cleveland and Sturgeon Point, where the concentrations were largely unchanged.

EH-TBB and BEH-TEBP are the two main components of FireMaster 550, which is a replacement for the penta-BDE commercial mixture. These two main components (EH-TBB and BEH-TEBP) were included in the analyses of IADN samples in 2008. Because EH-TBB and BEH-TEBP together are the major components of FireMaster 550, their concentrations were summed (notated here as EH-TBB and BEH-TEBP), and this sum was regressed as a function of time. Initial assessments indicated the atmospheric EH-TBB and BEH-TEBP concentrations were significantly and rapidly increasing at all the five sites, with doubling times of 2–6 years (i.e. 2005-2018) but the 10 year (i.e. 2008-2018) trends now show a significant decline only at urban sites of Cleveland and Chicago.

Air samples collected at the Canadian sites are being analyzed for 13 non-PBDE HFRs [hexabromobenzene (HBBz), hexabromocyclododecane (HBCDD), 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB), bis(2-ethylhexyl)tetrabromophthalate (BEH-TEBP), allyl 2,4,6-tribromophenyl ether (TBP-AE), 2-bromoallyl-2,4,6-tribromophenyl ether (TBP-BAE), pentabromoethylbenzene (PBEB), pentabromotoluene (PBT), *anti*-dechlorane plus (*anti*-DDC-CO), *syn*-dechlorane plus (*syn*-DDC-CO), 2,3-dibromopropyl-2,4,6-tribromophenyl ether (TBP-DBPE), DBDPE, and 1,2-bis(2,4,6-tribromophenoxy) ethane (BTBPE)]. There is no new data for Burnt Island and data for Point Petre is updated until 2017. Figure 14 presents the updated for *anti*- and *syn*-DDC-CO, DBDPE, EH-TBB, HBB and PBT in air at Point Petre. The half-lives of HFRs for Point Petre and Burnt Island are presented in Table 4.

anti-DDC-CO at Point Petre showed an overall declining trend with $t_{1/2}$ = 5.4 years from 2008-2017. Its concentration fell slowly from 2008 to 2010, then it became stable from 2010 and onwards. *anti*-DDC-CO in the atmosphere is decreasing, with $t_{1/2}$ = 3.4 years at Burnt Island. *syn*-DDC-CO at Point Petre also showed declining trend with $t_{1/2}$ = 3.8 years, but the levels are increasing at Burnt Island, with a doubling time of 7.6 years from 2008-2014. The increasing trend of DBDPE seem to agree with the hypothesis that it is a replacement for deca-BDE. However, another study, Liu et al. (2016), found that the concentrations of both compounds were decreasing in the air from the Great Lakes region from 2005 to 2014, which provides different results. Concentrations of DDC-CO are not significantly declining at the US sites in the last 10 years.

For other non-BDE FRs, DBDPE at Point Petre appeared to be in a downward trend from 2016 and after but this may be driven by the exceptionally high measurement in 2016. The doubling time for DBDPE was 2.6 year. EH-TBB in air at Point Petre was slowly increasing with $t_{1/2}$ = -16 years. Liu et al. (2016) reported that EH-TBB+BEHTBP were increasing in air at the Great Lakes region and they suspected that PBDEs are being replaced by EH-TBB- and BEHTBP-containing alternatives. Air concentration of HBBz at Point Petre was slowly increasing from 2009 to 2011 and stabilized afterwards. It has an overall $t_{1/2}$ = -39 years which indicates its concentration was non-changing from 2009 to 2017. Pentabromotoluene concentrations are decreasing at Burnt Island, with a half-life of 2.1 years but seem to be increasing slowly at Point Petre, with a doubling time of 13 years. In previous report, doubling time of PBT at Point Petre was 6.8 years. It appears that the increasing rate has been slowed down. TBP-AE and HBBz concentrations at both Point Petre and Burnt Island were not changing. In general, the time trends of other non-BDE FRs in air at Point Petre were highly variable. There is no consistency in trends for the HFRs at Point Petre and there is no clear seasonal cycle. The highly scattered air concentrations are likely an indication of continuous emission and episodic transport from either local or long-range sources.

Organophosphate esters (OPE) are industrial chemicals that are widely used as flame retardants, plasticizers, antifoaming agents, and as additives in hydraulic liquids, lacquers, and floor polishes. These chemicals have been in heavy use for decades. OPEs include tris(2-chloroethyl) phosphate (TCEP), tris(2-chloroisopropyl) phosphate, and tris(1,3-dichloroisopropyl) phosphate, tri-*n*-butyl phosphate, triphenyl phosphate, and 2-ethylhexyl diphenyl phosphate (EHDP). Median total OPE concentrations (∑OPE) ranged from 93 pg/m^3 at Sleeping Bear Dunes to 1050 pg/m^3 at Chicago (Salamova et al., 2016). The total OPE levels were significantly higher at Chicago and Cleveland (the urban sites), compared to the rural and remote sites.

Most atmospheric ∑OPE concentrations were significantly decreasing over time, with halving times of about 3.5 years at the urban sites and about 1.5 years at the rural and remote sites. Interestingly, tris(2-chloroethyl) phosphate (TCEP) and 2-ethylhexyl diphenyl phosphate (EHDP) concentrations were increasing over time at the rural and remote sites with doubling times of 2.2 and 3.7 years, respectively (Salamova et al., 2016).

Per- and polyfluoroalkyl substances (PFASs) (other than PFOS, PFOA and LC-PFCAs)

The time trends of short-chain PFAS in precipitation of GLB from 2006-2018 was reported by Gewurtz et al., (2019). Figure 15 shows the time trends of PFBA and PFHxA in precipitation in Point Petre (Lake Ontario), Burnt Island/Evansville (Lake Huron) and Sibley (Lake Superior). PFBA, PFPeA, PFHxA, and PFHpA did not decrease monotonically at the three sites from 2006 to 2018 with the exceptions of PFBA at Burnt Island/Evansville and PFHpA at Point Petre. Furthermore, the DF trend line indicates that concentrations of PFBA and PFHxA appear to be increasing in the most recent years (~2010/2014 to 2018 at Point Petre and Burnt Island and ~2016 to 2018 at Sibley). These results may not be surprising considering that these short-chained PFCAs and their precursors have not been regulated in Canada (ECCC, 2018) or the US and are being used as replacement chemicals for PFOS, PFOA, and the longer-chain PFCAs (Wang et al., 2013).

There is no new data about the short-chain PFAS in air in the GLB region since the last report. At Toronto, levels of perfluorosulfonic acids (PFSAs, including PFOS) from an active air sampling campaign in 2010 of 0.38-3.1 pg/m^3 (Ahrens et al., 2012) were in line with levels measured using passive air samplers in 2009 (4.6 pg/m^3) and 2013 (0.29-4.7 pg/m^3) under GAPS, but lower than the levels seen in 2015 (8.1 pg/m^3).

Polycyclic Aromatic Hydrocarbons (PAH)

IADN data for total PAH concentrations show some significant decreases over time, with halving times ranging from 13 to 25 years (Salamova et al., 2015). PAH levels at Chicago and Cleveland are 10 times higher than the concentrations at the other IADN sites. However, the concentrations are also decreasing most rapidly at these stations. These declines can probably be attributed to emission reductions from the implementation of the Clean Air Act. PAH concentrations are decreasing at Eagle Harbor, the most remote IADN site in the US, but not at Sleeping Bear Dunes, the other remote site.

Concentrations of phenanthrene are decreasing at about the same rate as total PAH except at Sleeping Bear Dunes and Point Petre, where no significant decreases were observed (Salamova et al., 2015). Significant decreasing rates for benzo[a]pyrene were detected only at Chicago and Sturgeon Point, and the halving time at Chicago was about half that at Sturgeon Point (Salamova et al., 2015).

A passive air and water sampling study in Lake Superior in 2011 showed that surface water and atmospheric PAH concentrations were greatest at urban sites (Ruge et al., 2015). Net air-to-water deposition of PAHs was observed near populated areas, but the net exchange is near equilibrium off shore (Ruge et al., 2015). A similar study conducted in the lower Great Lakes using polyethylene passive samplers in air and water demonstrated that gaseous PAH concentrations were strongly correlated with population within 40 km of the sampling locations (McDonough et al., 2014). Source profiles differed for atmospheric and aqueous PAHs indicating that in addition to atmospheric deposition, runoff and sediment-water exchange contributed to dissolved concentrations.

Guo et al. (2018) assessed atmospheric flows to the Great Lakes for phenanthrene and benzo[a]pyrene as representative PAHs. These two compounds were selected because phenanthrene is relatively abundant and volatile and is used in several industries making dyes and explosives and because benzo[a]pyrene is a known human carcinogen. The overall region-wide input of phenanthrene to the five lakes has decreased slowly over the period 1992–2015. This is true despite the increasing inputs of phenanthrene into Lakes Superior, Michigan, Erie, and Huron. Presumably, some of these long-term decreases in PAH input flows are due to the wide-spread conversion from coal to cleaner fuels for energy production and due to the implementation of the Clean Air Act by the U.S. EPA (Liu et al., 2014) and of the U.S.-Canada Air Quality Agreement (U.S. EPA, 1991).

PAHs in air at Point Petre, Burnt Island and Egbert from 1997-2017 was reported (Li et al., 2021). The half-lives of the PAHs are presented in Figure 16. Li et al. (2021) reported that there were significant declining trends for most PAHs. For example, the concentrations of most of the higher mass PAHs, such as dibenz[a,c]anthracene (DacA) at the three sites, have decreased since 1997 with halving times of 4.1-5.4 years, which is much faster than the other PAHs. Other PAHs with halving times less than 10 years are anthracene at Burnt Island; benzo[g,h,i]fluoranthene (BghiF), chrysene (Chr), coronene (Coron), and retene (Ret) at Egbert; and benzo[e]pyrene (BeP), BghiF, dibenz[a,h]anthracene (DahA) at Point Petre, suggesting that these PAHs in the GLB atmosphere are decreasing rapidly. At Burnt Island, 6 PAHs are significantly decreasing, with the halving time in the range of 4.2-21 years. At Egbert, concentrations of 7 PAHs show significant decreasing trends from 1997 to 2006. At Point Petre, most of the PAHs in the atmosphere were significantly decreasing from 1997 to 2017, with half-lives in the ranges of 3.6-32 years. The declining trends are due to effective control of PAH emissions in the Great Lakes Basin. The main contributors of PAHs in the GLB are coal combustion. The potential source regions for most source sectors were identified south or southwest of the sampling sites. The authors stated that although health risks from coal combustion, liquid fossil fuel combustion and petrogenic sources significantly decreased, but risk from forest fire-related PAH emissions may play an increasing role in the future due to climate change.

At the US sites, the concentrations of PAHs are decreasing with halving times of 15 to 20 years when the entire sampling period is considered. When only the last 10 years are included in the analysis, the trends are not significant.

Diagnostic ratios (DRs) and positive matrix factorization (PMF) models can help in characterizing source profiles of polycyclic aromatic hydrocarbons (PAHs). In general, pyrogenic sources, including coal combustion and vehicular emissions, were the most important contributors to atmospheric profiles, in particular at the urban sites. Diesel emissions accounted for a larger portion of the traffic-oriented PAHs than gasoline emissions at urban sites but this signature was less obvious at the remote sites. Temporal analyses for DRs revealed that the relative contribution of petrogenic sources and volatilization from surfaces has been increasing gradually, and that the gaps in PAH emissions between diesel- and gasoline-engines appeared to be further amplified in recent years. In the PMF model, coal combustion and non-pyrogenic emissions were the main PAH sources for winter and summer air, respectively, but none of the DRs responded to these changes accordingly. Only two DRs, i.e., I123P/(I123P+BGP) and BAP/(BAP+BGP), which are characteristic of pyrogenic sources and traffic related emissions respectively, were relatively consistent between vapor and particle phases. Although our analysis highlighted some limitations, DRs can provide useful information on spatial and temporal trends and source characterization. (Wu et al., 2020)

Metals

Li et al., (2020) reported 19 elements in PM_{10} samples that were collected at Point Petre. Burnt Island and Egbert from 1998-2017. The 19 measured elements included aluminum (Al), arsenic (As), barium (Ba), bismuth (Bi), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), manganese (Mn), molybdenum (Mo), nickel (Ni), lead (Pb), antimony (Sb), selenium (Se), strontium (Sr), titanium (Ti), vanadium (V), and zinc (Zn). Results showed a strong relationship between element concentrations and local populations, which suggested that the emissions of trace elements were anthropogenically-related. The concentrations of most elements were significantly decreasing with halving times ranging from 10 to 48 years in response to national and international regulatory actions (Figure 17). The findings imply that metals in the GLB atmosphere were influenced by human settlement and industries not only from local emission but also through long-range atmospheric transport.

Dioxins and Furans (PCDD/Fs)

Areas with higher population generally showed higher annual mean air concentrations of PCDD/Fs in North America (CEC 2014; Venier et al. 2009; Cleverly et al. 2007). There are no new PCDD/F data since the last reporting period. Air concentration measurements under Canada's National Air Pollution Surveillance (NAPS) reported high toxic equivalency (TEQs) in air at the Walpole Island site (in Lake St. Clair) and the Windsor/University Ave. sites (Windsor, ON), where profiles were characterized by a lower contribution of octachlorodibenzo-p-dioxin and an increased contribution of dibenzofurans. This profile might indicate the impact of a local emission source (CEC 2014). PCDD/F levels at rural, suburban, and urban NAPS sites (including sites in the Great Lakes Basin) declined after the early 1990s and in the early 2000s. This decline can be attributed to control measures taken in Canada with respect to PCDD/F emission sources. After the year 2005, a clear trend is not evident (CEC 2014).

Linkages

Atmospheric deposition is a significant route by which persistent, bioaccumulative and toxic (PBT) chemicals, such as PCBs, currently enter the Great Lakes. Increases in the concentration and loadings of atmospheric chemicals of concern, including PBTs, may result in increased contamination in sediment, toxic chemicals in offshore waters, and contaminants in whole fish and waterbirds. Bioaccumulation of these PBTs in fish may result in fish consumption advisories. This sub-indicator links directly to the other sub-indicators in the Toxic Chemicals category, particularly Toxic Chemicals in Water.

Climate variabilities could affect contaminant exchange between air and water, for instance, increase in precipitation could result in greater wet/dry deposition of contaminants into the lakes. Conversely, decreased ice cover and increase in surface water temperatures would drive greater volatilization losses from the lakes. Further investigation is required to better understand the overall impact of climate variables on atmospheric exchange of contaminants in the lakes.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin		X		
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: IADN dataset: http://www.exchangenetwork.net/data-exchange/glenda/ https://iadnviz.iu.edu/ GLB dataset: https://open.canada.ca/data/en/dataset/5d8548c5-e284-4e85-aed4-c22536de615a NADP dataset: http://nadp.slh.wisc.edu/data/MDN/		

Data Limitations

- Since air (gas and particle) measurements were conducted using active air sampling method consisting of 24-hour integrated samples collected once every 12 or 36 days, it is not possible to assess toxic chemical levels during the time between two samples.
- Since most sampling sites are remote, samples may not be able to return to the laboratory until 3-4 months after sampling. Also, the sample extraction and analytical process for organic chemicals are very time consuming. Over 180 different individual chemicals need to be analysed and quantified in each sample. All resulting data need to undergo strict QA/QC evaluation and flagging to ensure data quality due to the low levels found in air and precipitation samples. It is expected that data can only be reported in peer-reviewed publications 3-4 years after the samples were collected.
- Some of the estimation parameters of the loadings calculations are poorly known at present and calculations are based on assumptions that air, precipitation and water concentrations measured at a few specific locations on the lakes apply to the entire lake. Thus, uncertainties in such loadings calculations are high. The trends in the atmospheric concentrations of toxic chemicals, however, are much better known and a much better sub-indicator of progress towards anthropogenic virtual elimination. Errors in these trends should be clearly stated and tested against the null hypothesis (things are not changing).
- To understand the pollutant concentration trends related to atmospheric deposition, additional information is needed in interpreting pollutant load estimates derived using the suggested calculation. For example, information on the yearly variations in the rain rate (dry years versus wet years) is needed to understand the pollutant loadings associated with wet deposition. Also, since it is known that the pollutant loads associated with atmospheric deposition have seasonality for some components, time trends and seasonal trends should be determined separately derived using statistical methods.

Additional Information

Many remaining sources of PCBs are located in urban areas, which is reflected by the higher levels of PCBs measured in Chicago and Cleveland by IADN, and in Toronto by other researchers (Diamond et al., 2010; Robson et al., 2010, Melymuk et al, 2012, Csiszar et al., 2014) and in other areas (Wethington and Hornbuckle 2005; Totten et al. 2001). Research to investigate the significance of these remaining sources is underway (Shanahan et al. 2015). This is important because fish consumption advisories for PCBs exist for all five Great Lakes.

Among the CMCs, PCBs, PBDEs [tetrabromodiphenyl ether and pentabromodiphenyl ether, hexabromodiphenyl ether and heptabromodiphenyl ether, decabromodiphenyl ether (commercial mixture, c-decaBDE)], HBCDD, SCCP and perfluorooctanoic acid (PFOA), its salts and related compounds are listed under Annex A (Elimination) under the UNEP Stockholm Convention on POPs. PFOS, its salts and perfluorooctane sulfonyl fluoride are listed under Annex B (Restriction) of the Stockholm Convention. Canada is a party of the Stockholm Convention.

The Minamata Convention on Mercury is a UNEP led multilateral environmental agreement between 131 Parties (as of April 2021) including the U.S. and Canada. The convention came into effect in August, 2017 and aims to protect human health and the environment from anthropogenic emissions and release of mercury and mercury compounds. The data collected through long term monitoring programs will provide critical information for the evaluation of how effective the current structure of the treaty is. Canada is a net recipient of mercury where over 95% of the anthropogenic mercury deposited comes from outside of the country.

Residential garbage burning (burn barrels) is now the largest current source of dioxins and furans (Environment Canada and U.S. Environmental Protection Agency, 2006). Basin and nationwide efforts are underway to eliminate emissions from burn barrels.

Regionally, many sources that emit mercury are reducing emissions. For instance, all coal-fired power plants in Ontario have ceased operation as of April 2014, being the first jurisdiction in North America to fully eliminate coal for producing electricity (Ontario Ministry of Energy, 2014).

Continued long-term monitoring of the atmosphere is necessary in order to measure progress brought about by toxic reduction efforts. Environment and Climate Change Canada and U.S. EPA recently added routine monitoring of PBDEs and some non-PBDE flame retardants to the IADN and GLB programs. Screening and method development for additional non-PBDE flame retardants, per- and polyfluoroalkyl substances (PFASs) and SCCPs is currently under way. These efforts ensure the early detection of potential chemicals of concern and allow for the development of temporal trends in advance of the implementation of regulatory measures to enable future effectiveness evaluation of regulations. Sample and extract archiving practices in monitoring programs also provide the opportunity for the retrospective trend analysis of newly identified chemicals of concern. Results from these monitoring efforts on emerging chemicals of concern will contribute to the scientific information needed for the risk assessment and identification of additional chemicals of mutual concern.

Acknowledgments

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IADN Data: https://iadnviz.iu.edu/#/about/
GLB Data: https://open.canada.ca/data/en/dataset/5d8548c5-e284-4e85-aed4-c22536de615a, or contact Helena Dryfhout-Clark, IADN/GLB Data Manager, Air Quality Processes Research Section, Environment and Climate Change Canada, 4905 Dufferin Street, Toronto, Ontario, M3H 5T4, Helena.Dryfhout-Clark@ec.gc.ca, 613-995-9031.
Link to IADN websites: http://epa.gov/greatlakes/monitoring/air2/index.html

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State of the Great Lakes 2022 Report

Table 1. Half-lives ($t_{1/2}$) of PCBs in air at Point Petre, Burnt Island and Egbert (Source: Hung et al., 2021)

PCBs	Period	$\sum_{28}$PCBs		18		28		31		52	
Station		$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2
Point Petre (Lake Ontario)	1992-2017	17	0.92	11	0.94	43	0.32	16	0.86	15	0.93
Burnt Island (Lake Huron)	1992-2013	65	0.17	9.8	0.93	160	0.024	-	-	29	0.63
Egbert (Lake Huron)	1995-2006	74	0.044	8.3	0.59	68	0.043	-	-	32	0.2
		101		**118**		**138**		**153**		**180**	
Station		$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2
Point Petre (Lake Ontario)	1992-2017	14	0.90	14	0.92	17	0.65	15	0.85	11	0.85
Burnt Island (Lake Huron)	1992-2012	34	0.56	22	0.44	17	0.33	31	0.59	8.9	0.75
Egbert (Lake Huron)	1995-2006	37	0.20	39	0.48	-	-	58	0.076	23	0.29

Table 2. Half-lives of PBDE in air at Point Petre and Burnt Island (Wong et al., *in prep*; Shunthirasingham et al., 2018)

	Point Petre (2005-2017)		**Burnt Island (2005-2013)**	
	$t_{1/2}$	r^2	$t_{1/2}$	r^2
BDE28	31	0.13	26	0.36
BDE47	7.7	0.81	-23	0.39
BDE99	5.9	0.81	13	0.44
BDE209	12	0.29	5.7	0.96

Table 3. Half-lives ($t_{1/2}$) of OCPs in air at Point Petre, Burnt Island and Egbert (Source: Hung et al., 2021)

OCPs	Period	aldrin		α-HCH		γ-HCH		c-chlordane		t-chlordane		t-nonachlor	
Station		$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2
Point Petre (Lake Ontario)	1992-2017	6.2	0.68	4.8	0.99	4.3	0.96	9.8	0.90	9.4	0.94	13	0.90
Burnt Island (Lake Huron)	1992-2012	7.6	0.61	4.9	0.99	4.7	0.94	11	0.83	12	0.95	13	0.92
Egbert (Lake Huron)	1995-2006	7.1	0.81	4.2	0.98	4.7	0.87	34	0.25	31	0.26	52	0.16
OCPs	**Period**	**p,p'-DDT**		**p,p'-DDE**		**p,p'-DDD**		**o,p'-DDT**		**o,p'-DDE**		**Dieldrin**	
Station		$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2
Point Petre (Lake Ontario)	1992-2017	9.1	0.83	12	0.92	7.5	0.90	9.0	0.87	11	0.84	10	0.94
Burnt Island (Lake Huron)	1992-2012	7.4	0.94	9.6	0.94	9.5	0.86	9.3	0.88	7.2	0.98	10	0.94
Egbert (Lake Huron)	1995-2006	12	0.49	38	0.10	12	0.75	28	0.22	14	0.87	18	0.41

OCPs	Period	α-endosulfan		β-endosulfan		endosulfan sulfate		heptachlor		heptachlor epoxide	
Station		$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2
Point Petre (Lake Ontario)	1992-2017	4.8	0.63	5.2	0.78	8.1	0.64	6.5	0.89	4.7	0.86
Burnt Island (Lake Huron)	1992-2012	13	0.62	11	0.78	10	0.61	8.6	0.52	5.8	0.87
Egbert (Lake Huron)	1995-2006	38	0.1	28	0.10	4.5	0.96	8.3	0.33	12	0.82

OCPs	Period	Oxychlordane		mirex		Photo-mirex	
Station		$t_{1/2}$	r^2	$t_{1/2}$	r^2	$t_{1/2}$	r^2
Point Petre (Lake Ontario)	1992-2017	8.5	0.91	13	0.89	9.4	0.82
Burnt Island (Lake Huron)	1992-2012	12	0.90	28	0.16	6.1	0.76
Egbert (Lake Huron)	1995-2006	71	0.10	8.0	0.80	4.6	0.88

Table 4. Half-lives of non-PBDE HFRs in air at Point Petre and Burnt Island (Wong et al., *in prep*; Shunthirasingham et al., 2018)

	Point Petre (2005-2017)		Burnt Island (2005-2013)	
	$t_{1/2}$	r^2	$t_{1/2}$	r^2
TBP-AE	28	0.019	24	0.10
TBP-BAE	90	0.074		
PBT	-13	0.30	2.1	0.99
TBP-DBPE	-3.9	0.78		
HBBz	-39	0.036	27	0.11
PBEB	7.7	0.82		
DBDPE	-2.6	0.53		
EH-TBB	-16	0.094		
HBCDD	-1.6	0.73		
anti-DCC-CO	5.4	0.47	3.4	0.36
syn-DCC-CO	3.8	0.66	-7.6	0.21

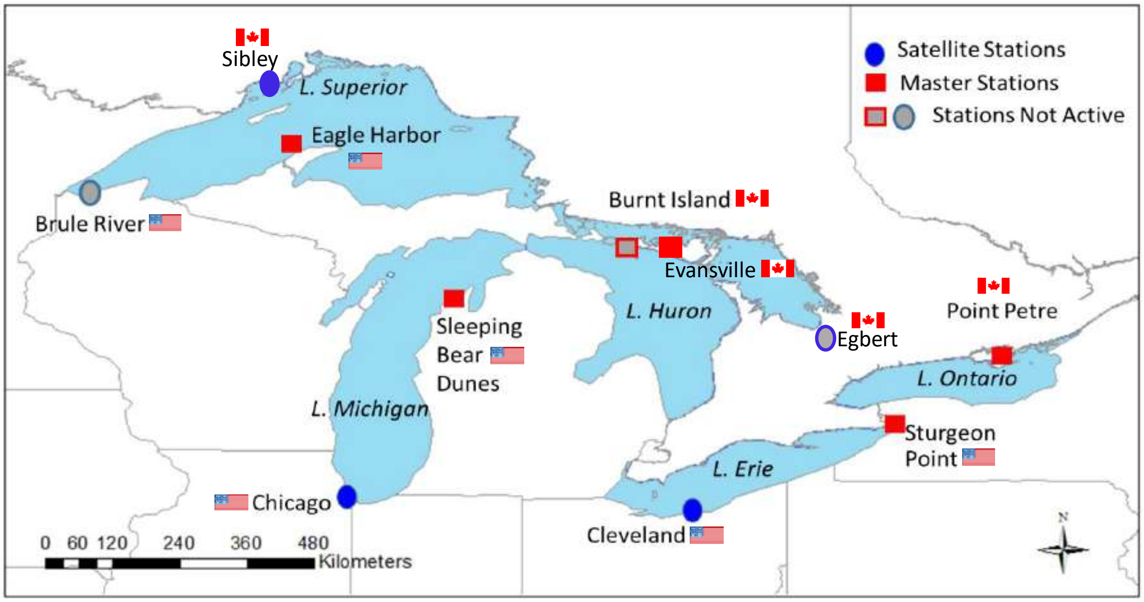
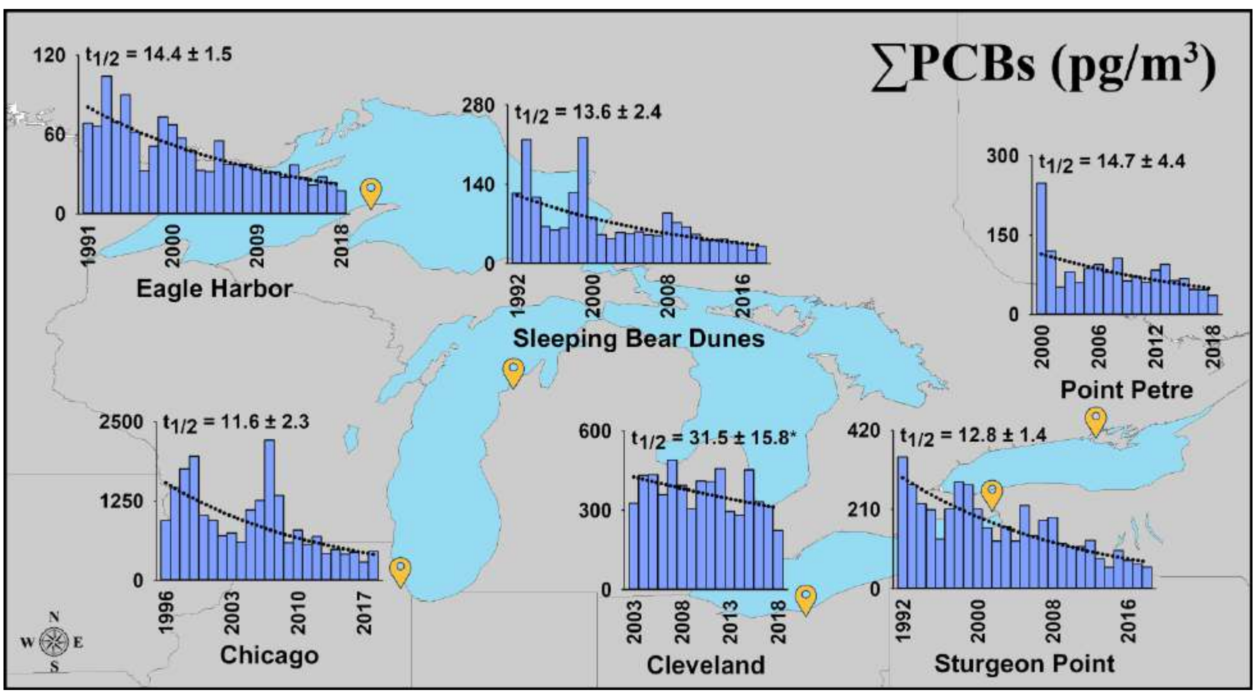


Figure 1. Site map of Integrated Atmospheric Deposition Network (IADN) and Great Lakes Basin (GLB). (Source: Modified from USEPA 2021)



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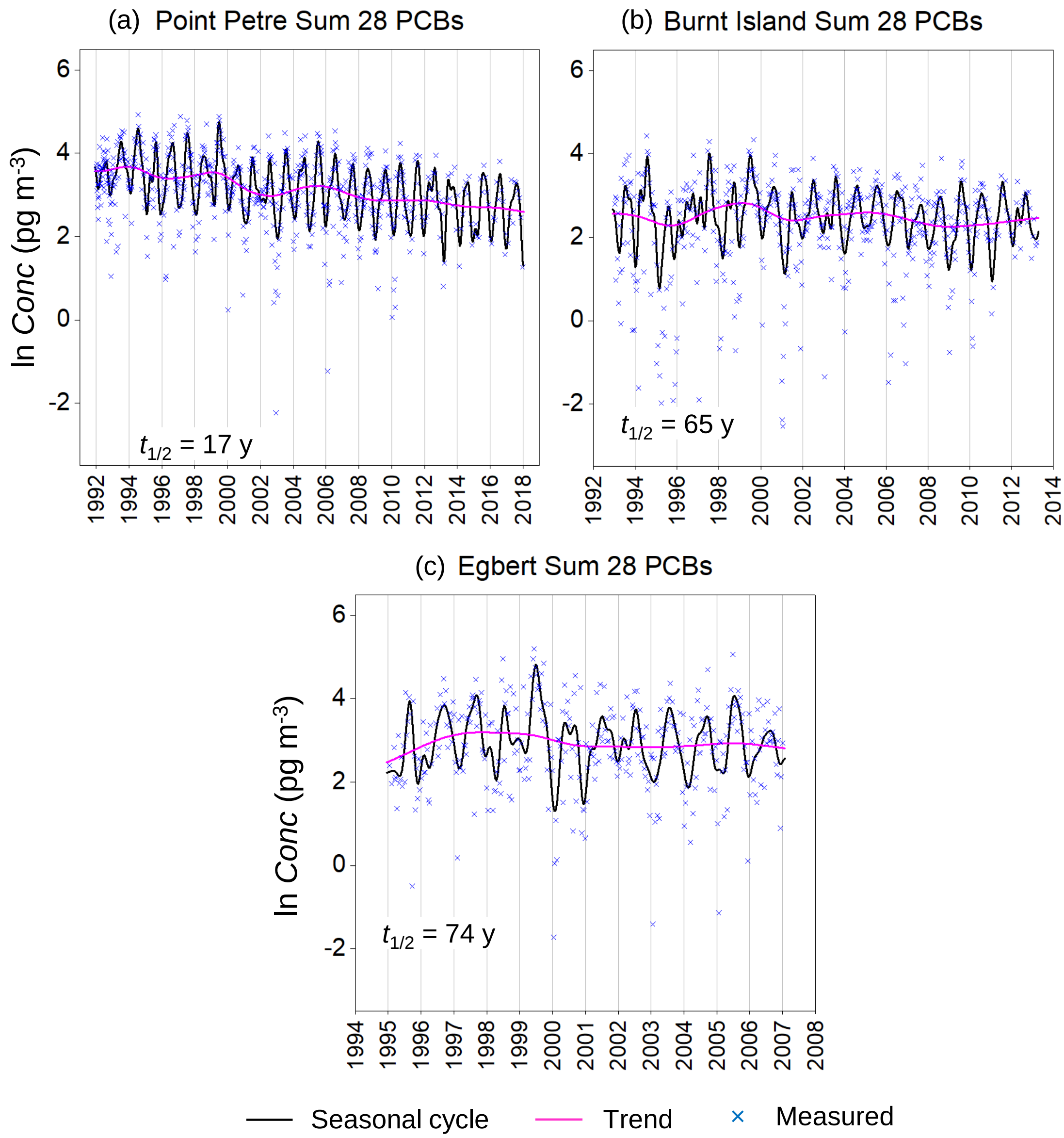

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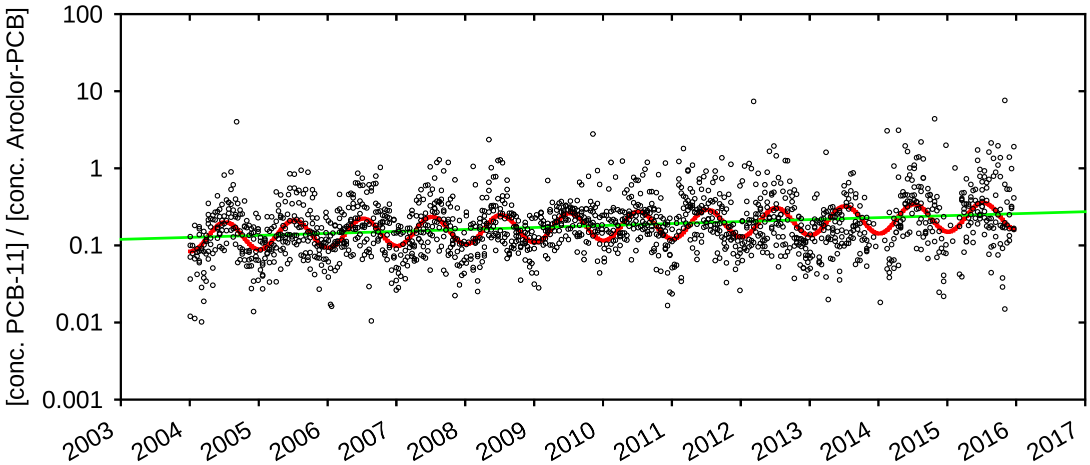

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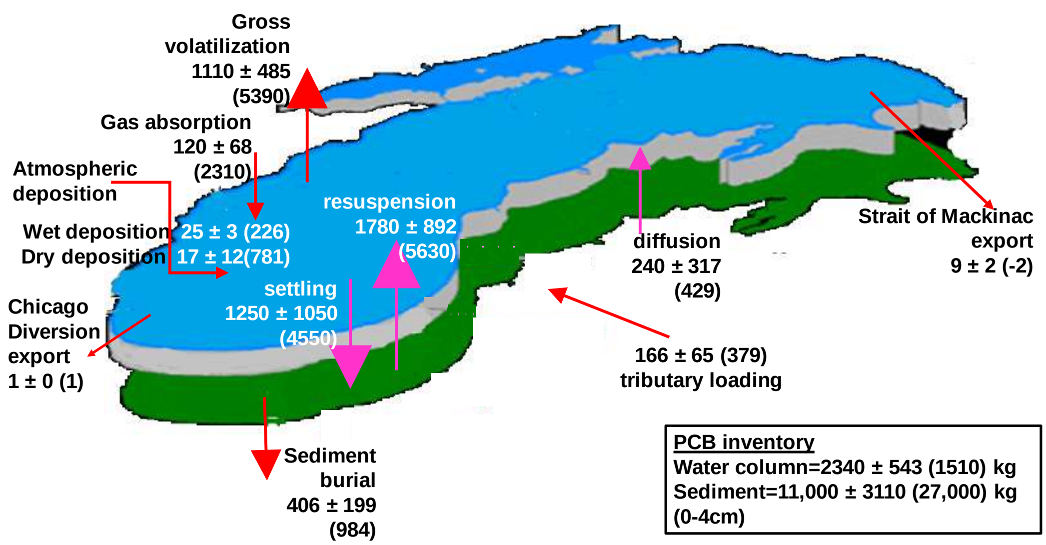

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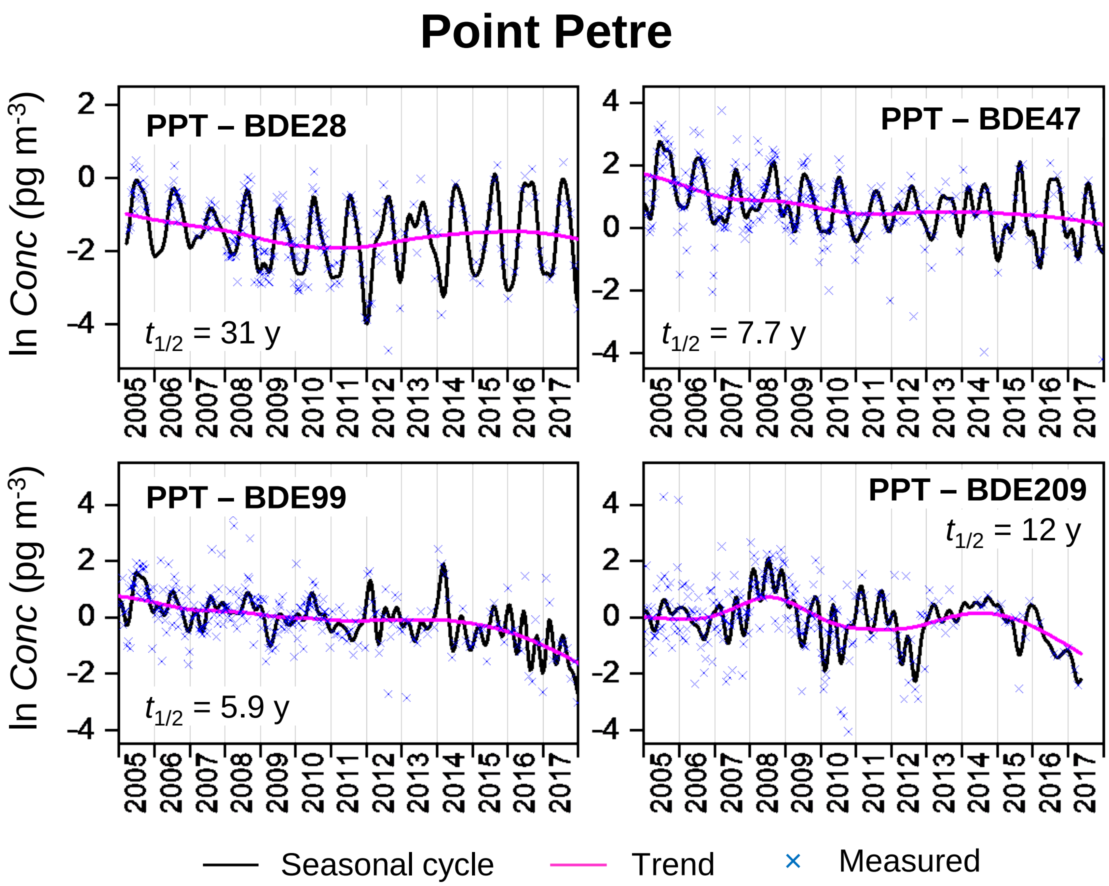

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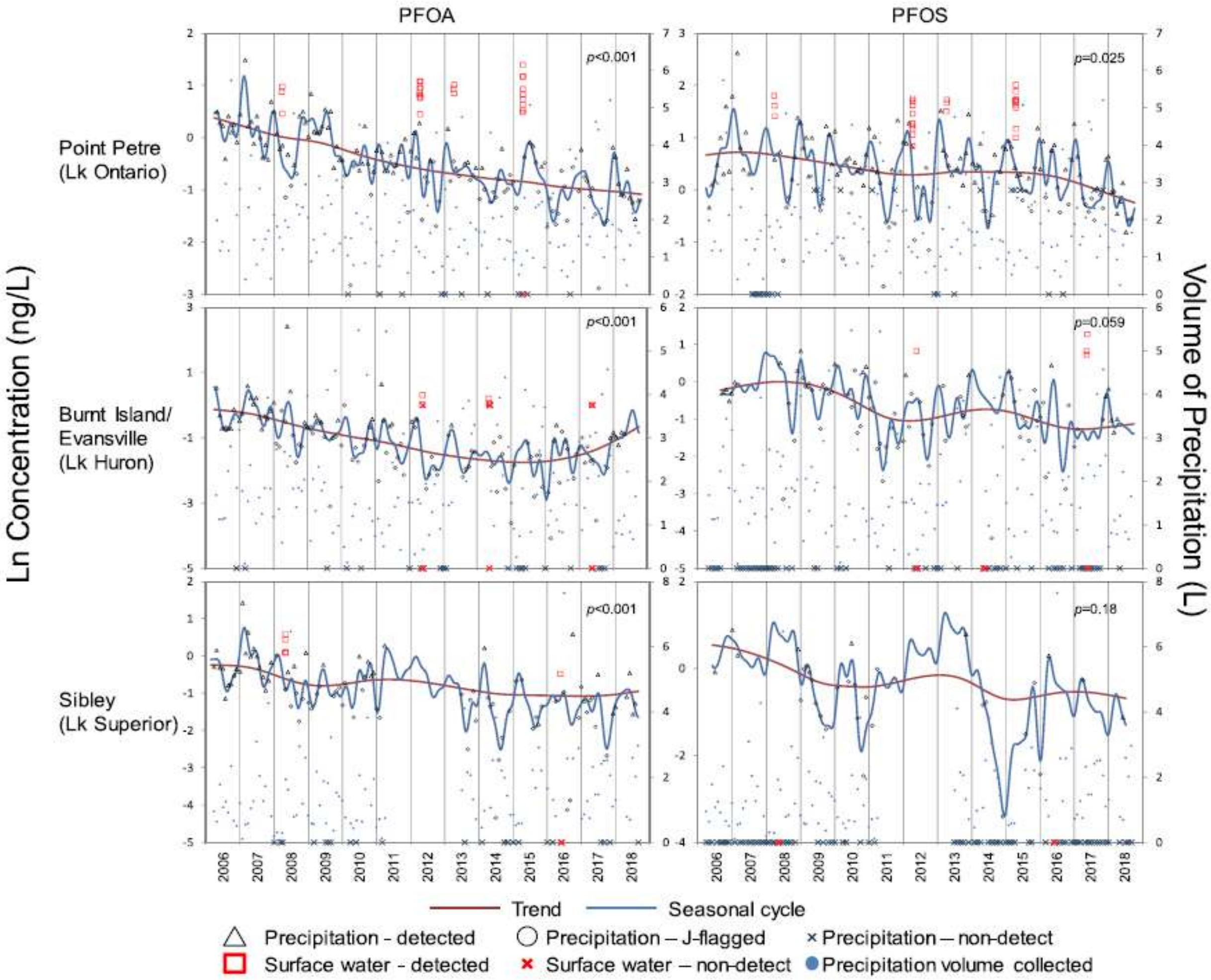

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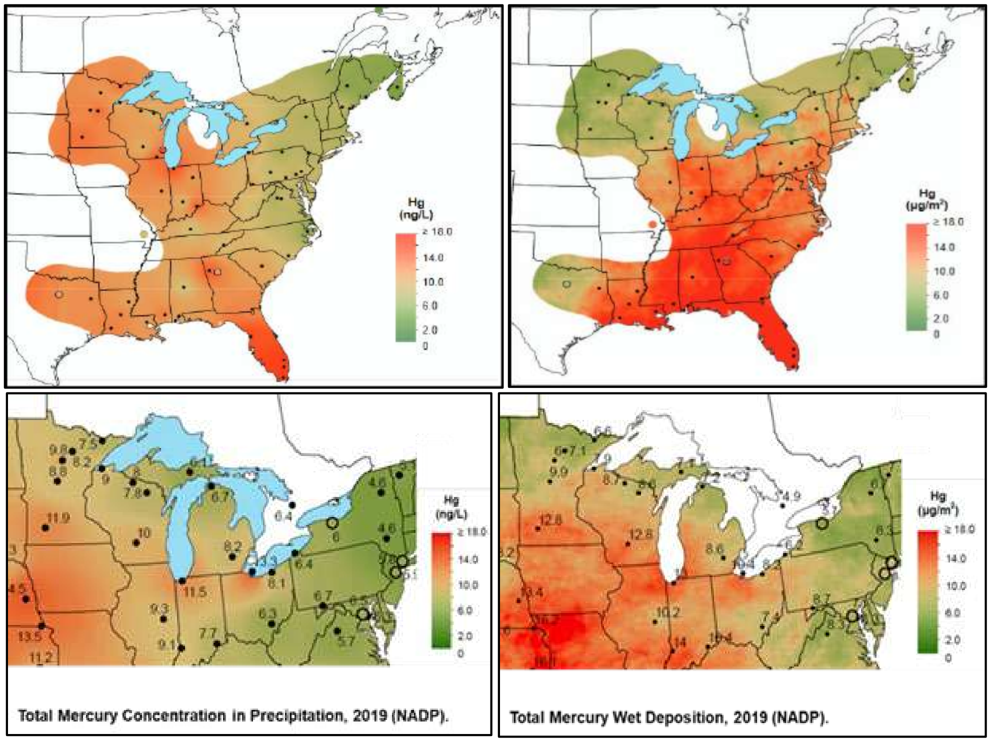

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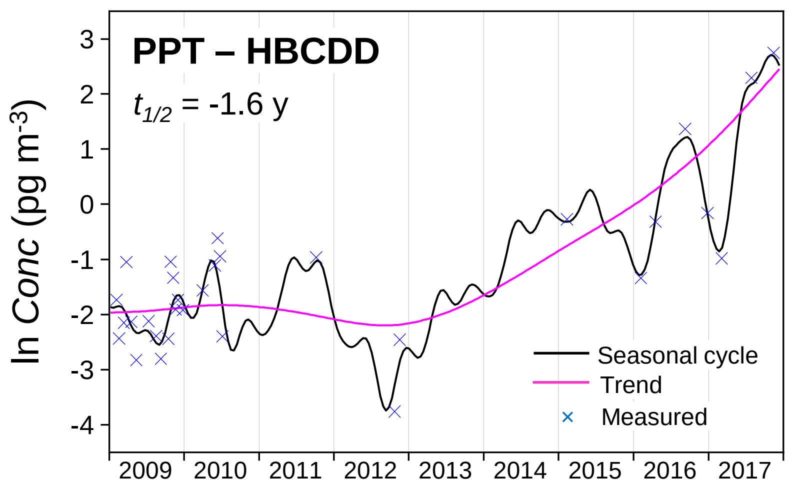

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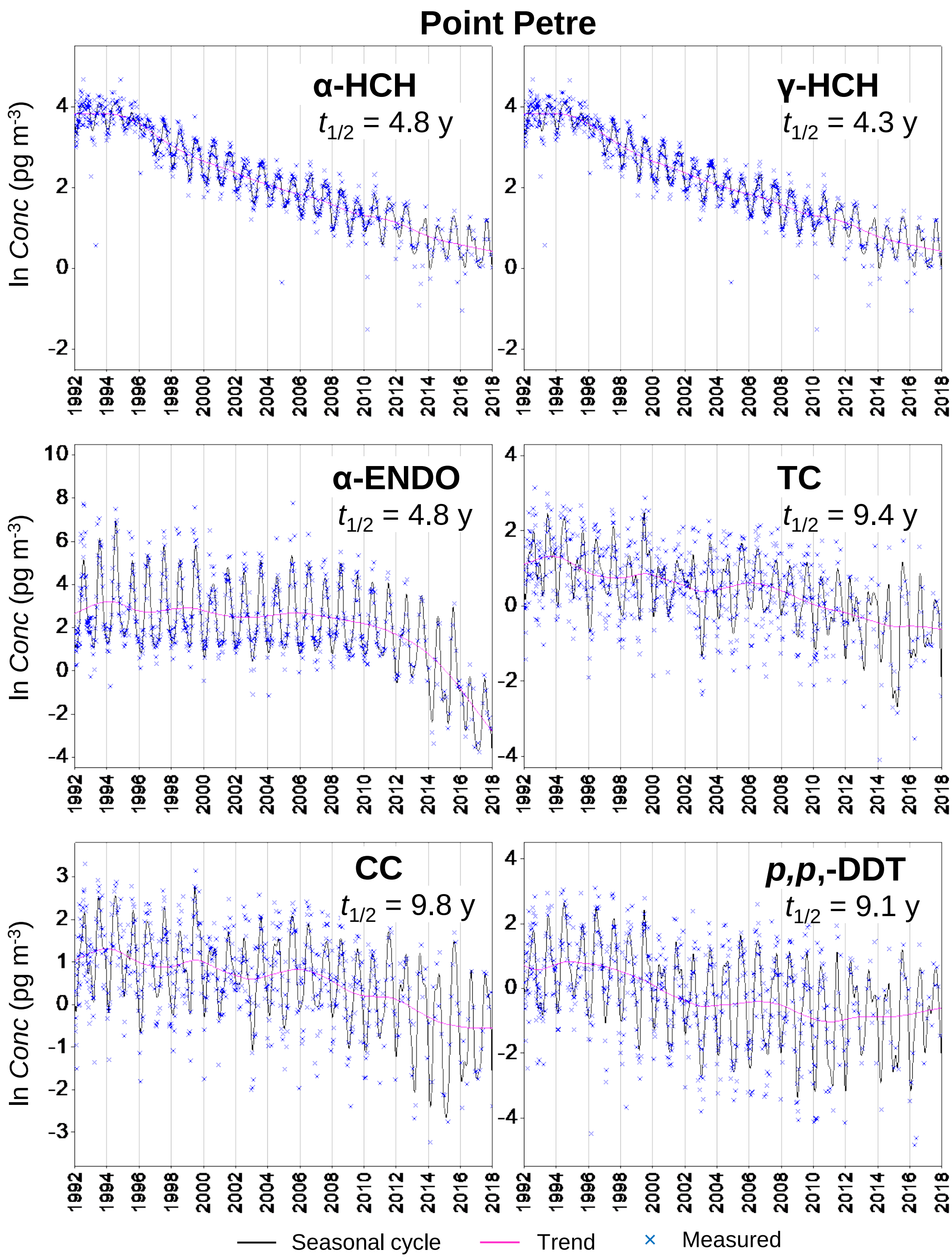

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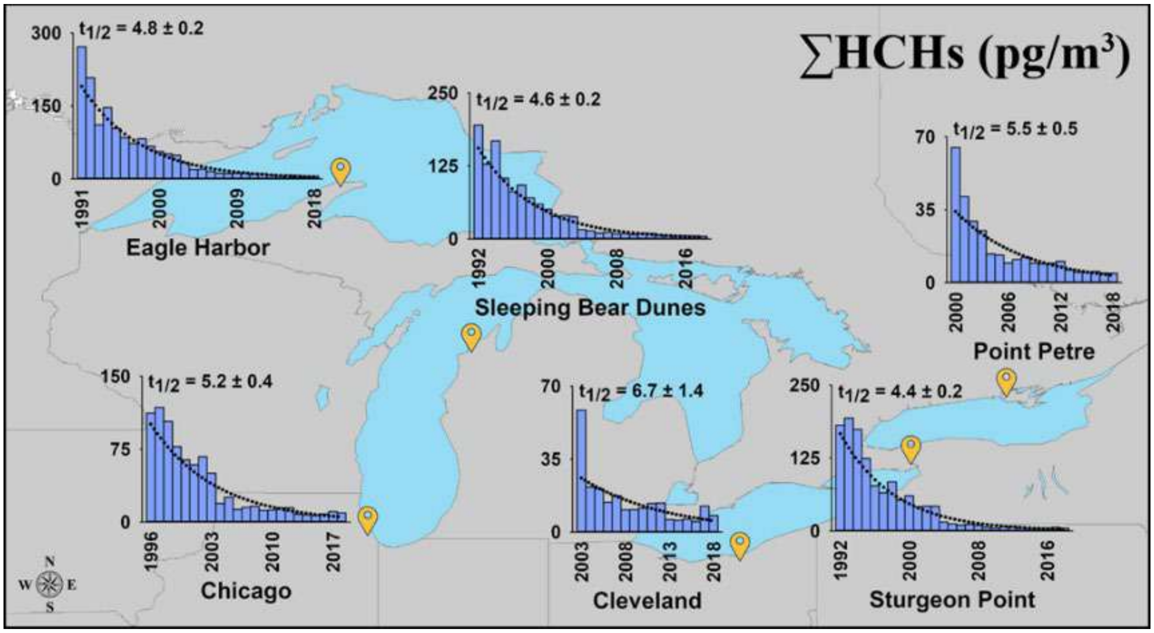

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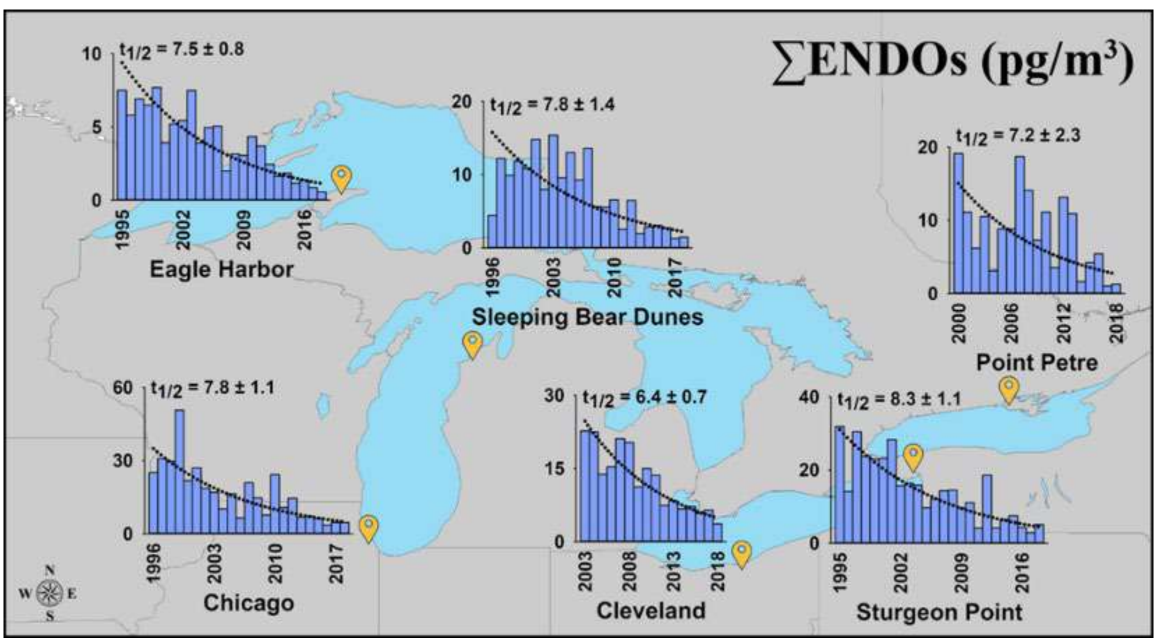

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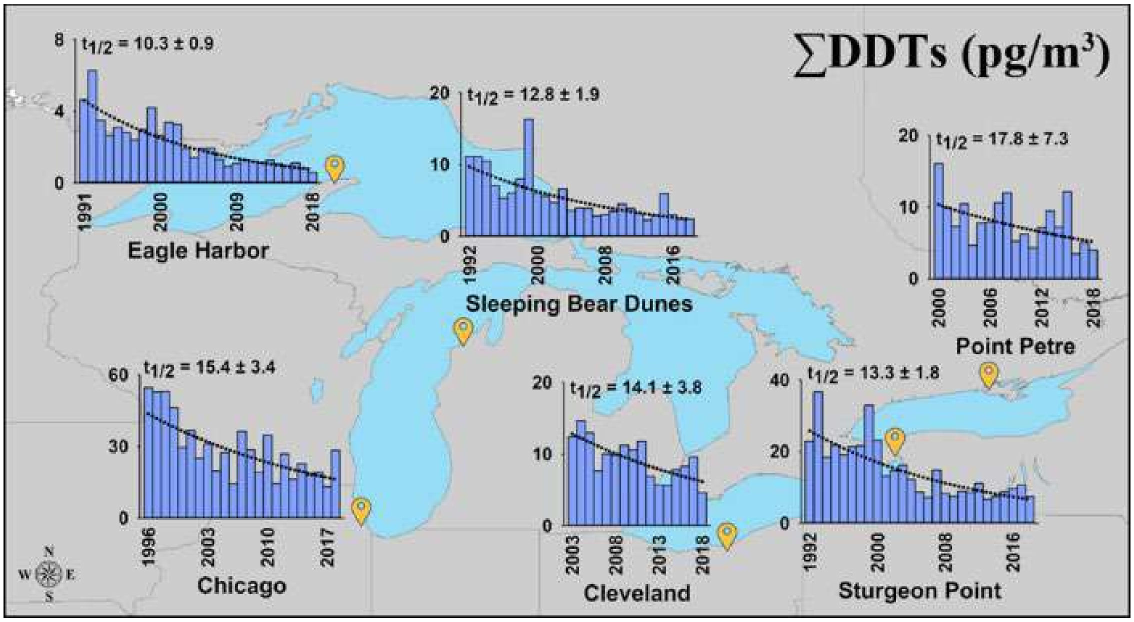

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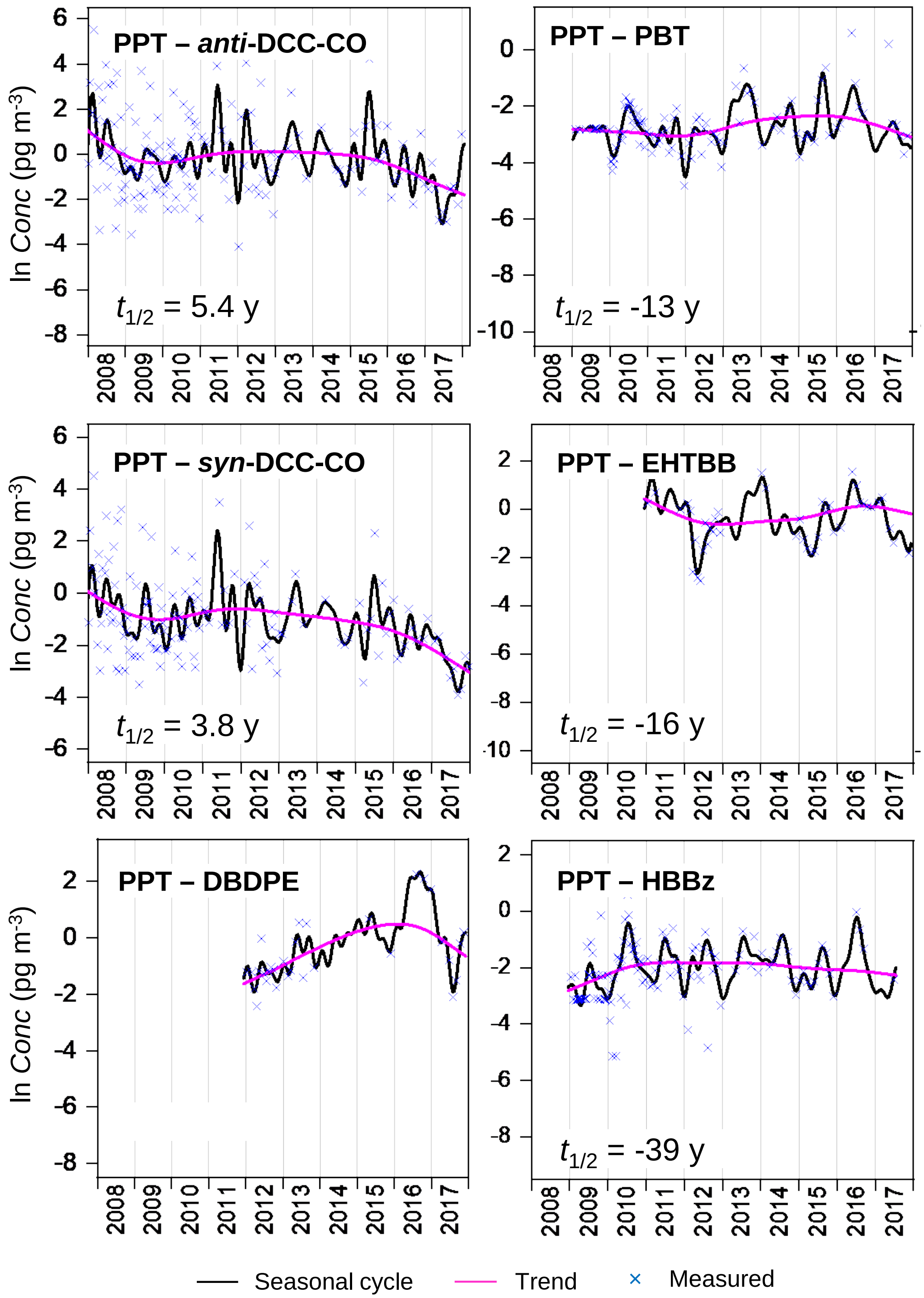

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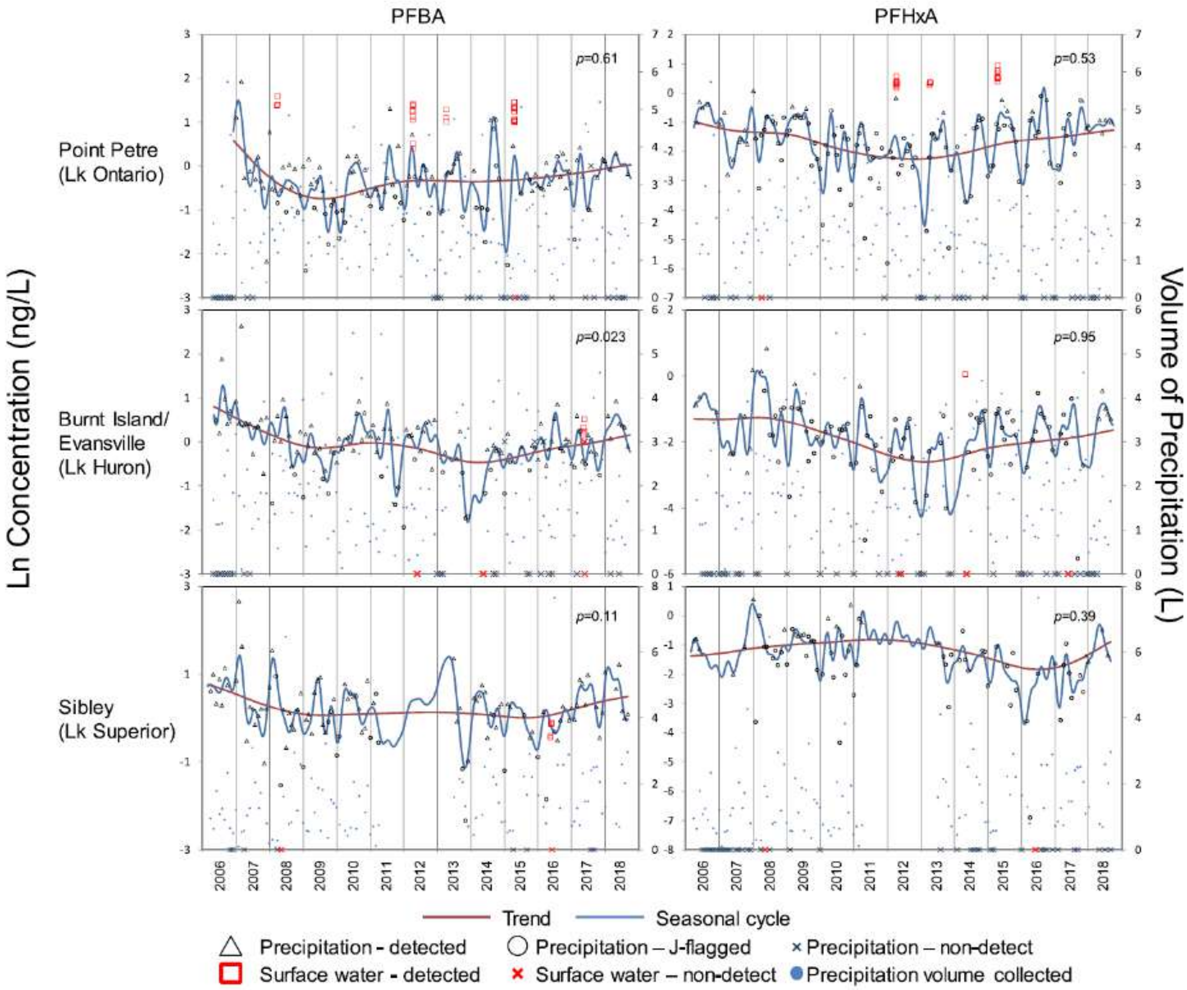

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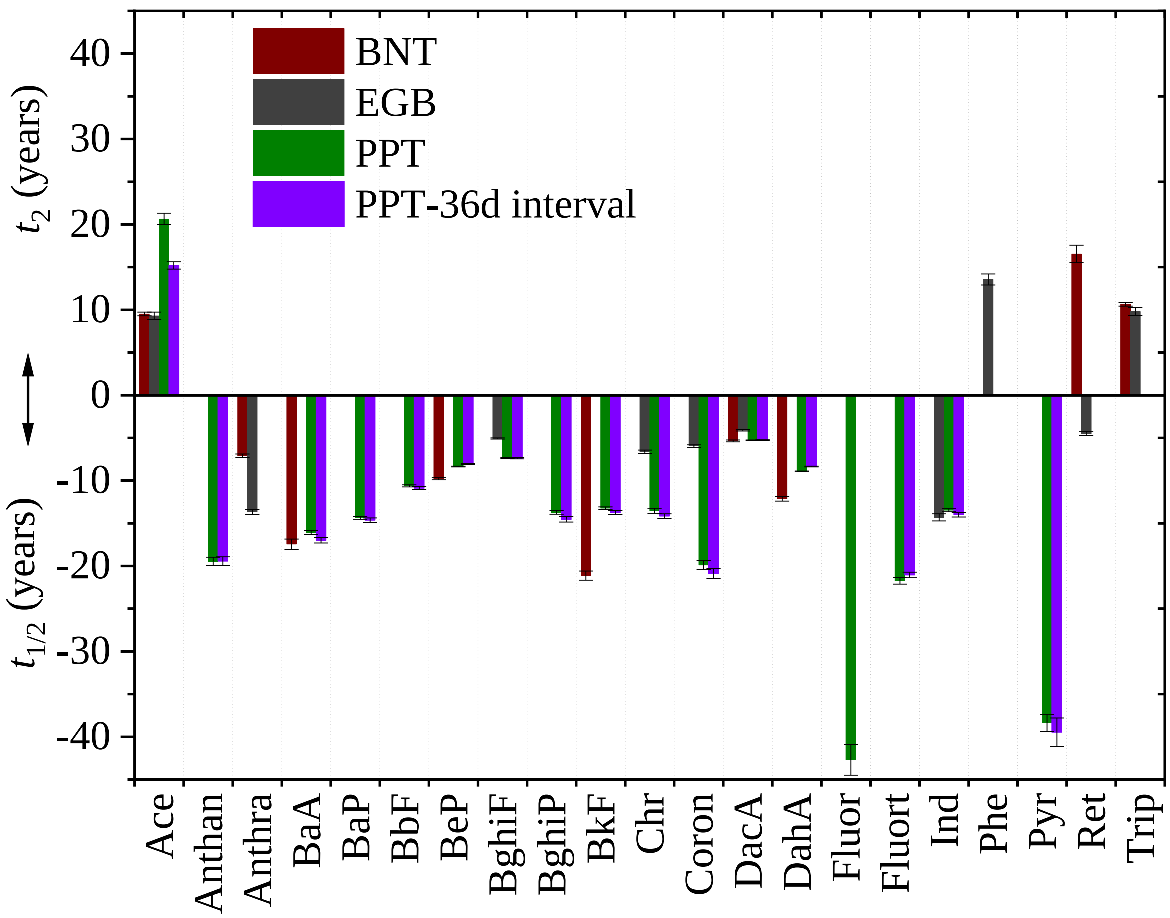

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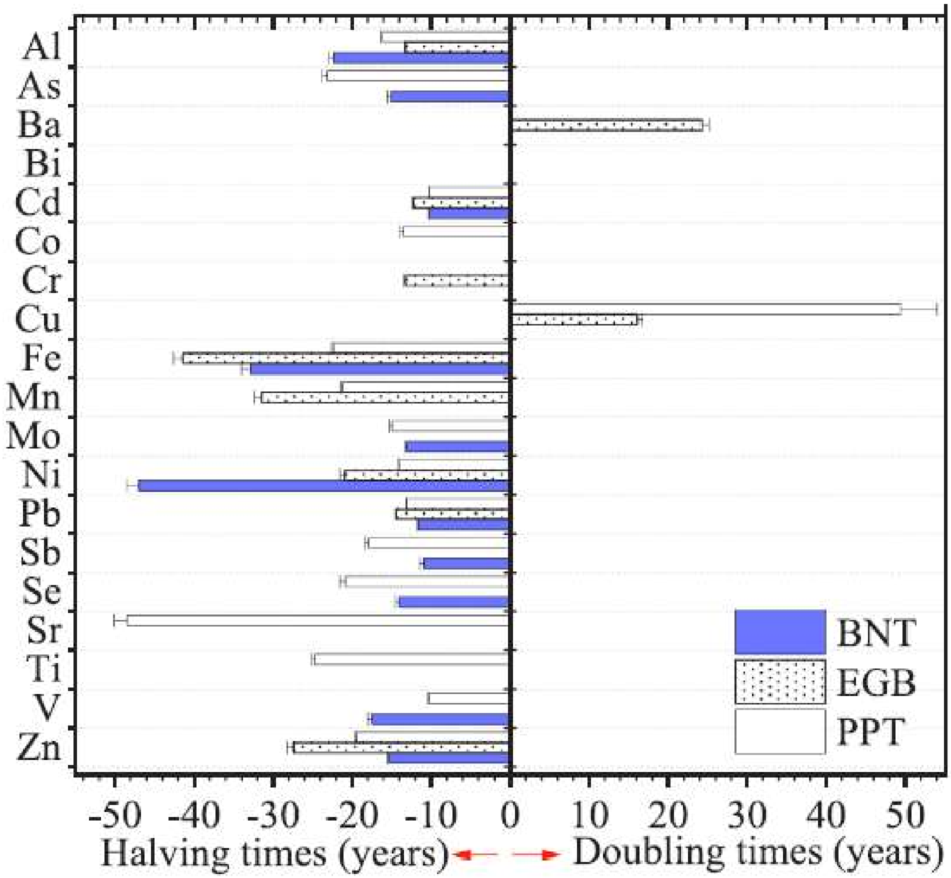

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Sub-Indicator: Coastal Wetland Invertebrates

Overall Assessment

Status: Fair

Trends:

10-Year Trend (2011-2019) *: Undetermined

Rationale: As of 2019, the majority of wetlands were not classified as degraded based on their Indices of Biotic Integrity (IBI) scores. From 2011 to 2019 the number of sites placed in the degraded categories (extremely degraded, degraded, and moderately degraded) remained between 12% and 30% of all sites that received macroinvertebrate IBI scores each year across the Great Lakes basin. There has been no consistent increase or decrease in any of the of IBI classifications at the basin level or the mean IBI score within lakes (Figures 1, 2 and 3) over the past nine years. Due to the low percentage of sampled wetlands in Lakes Ontario and Erie that are associated with IBI scores (11% and 7%, Table 2) the overall trend is undetermined.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been fully analyzed.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend (2011-2019) *: Undetermined

Rationale: The vast majority of Lake Superior coastal wetlands are not in the degraded categories based on IBI scores (Figure 5). From 2011 to 2019 the percentage of wetland sites in the degraded categories has remained between 0% and 25% of Lake Superior sites that received macroinvertebrate IBI scores annually. In 2015, 2016, 2017, and 2019 no sites were categorized as reference condition. Additionally, 2019 marked the first site placed in the "degraded" IBI rating in Lake Superior which is the lowest score in the lake to date (Figure 5). In 2014-2016 and in 2018 less than 50% of the sites sampled had the vegetation zones required for IBI scores, so the trend is undetermined.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been fully analyzed.

Lake Michigan

Status: Fair

10-Year Trend (2011-2019)*: Undetermined

Rationale: The majority of Lake Michigan coastal wetlands are not in the degraded categories (Figure 7) based on IBI scores. From 2011 to 2019 the percentage of sites in the degraded categories (moderately degraded, degraded, extremely degraded) has remained between 0% and 43% of the Lake Michigan sites that received macroinvertebrate IBI scores. There continues to be no sites categorized as extremely degraded. There is no obvious change in

the proportion of the sites in the degraded categories overtime and the trend is undetermined due to a low number of sites qualifying for an IBI calculation in in 2013 and 2018.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been fully analyzed.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend (2011-2019)*: Unchanging

Rationale: The majority of Lake Huron coastal wetlands are not in the degraded categories (Figure 9) based on their IBI scores. From 2011 to 2019 the percentage of sites in the degraded categories (moderately degraded, degraded, extremely degraded) has remained between 0% and 31% of the Lake Huron sites that received macroinvertebrate IBI scores. There is no trend in the average site IBI score for the lake, although in the past several years most of the sites are consistently in the middle range of site quality (moderately degraded, moderately impacted, mildly impacted). Approximately a third of the Lake Huron sites are located the St. Marys River. These river sites have a comparable range of site quality (moderately degraded to reference) to other areas in norther Lake Huron (Figure 7).

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been fully analyzed.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Undetermined

10-Year Trend (2011-2019)*: Undetermined

Rationale: Status and trends could not be determined for Lake Erie as only 8 sites (7% of sampled sites, Table 2) had the required vegetation zones for IBI calculations in the previous 9 years.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Undetermined

10-Year Trend (2011-2019)*: Undetermined

Rationale: Status and trends could not be determined for Ontario as only 24 sites (11% of sampled sites, Table 2) had the required vegetation zones for IBI calculations in the previous 9 years.

Status Assessment Definitions

Good: No wetlands are in the degraded categories but instead fall within reference conditions, mildly impacted, or moderately impacted.

Fair: The vast majority of the wetlands are not in the degraded categories.

Poor: The vast majority of the wetlands are in the degraded categories.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: This metric increased in score for the majority of the sites.

Unchanging: This metric score did not substantially change for the majority of the sites.

Deteriorating: This metric decreased in score for the majority of the sites.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

The endpoint for this sub-indicator will need to be established, based on a literature search of current and historical data, if available, or from data gathered from measuring this sub-indicator. Data would be evaluated for patterns by lake, wetland type, and ecoregion, and then calibrated against the monitoring objectives based on the professional judgement of those with expertise in the field. An endpoint for this sub-indicator is not possible at this time, but the planned expansion of the macroinvertebrate IBI to additional vegetation zones will be an important step in establishing targets. Currently, researchers associated with the Great Lakes Coastal Wetland Monitoring Program are working to establish macroinvertebrate IBI scoring systems for wetlands that are dominated by common vegetation types, including *Typha*, *Phragmites*, and floating leaf species.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the diversity of the invertebrate community, especially aquatic insects; to track the trends of Great Lakes coastal wetland ecosystem condition by measuring relative community composition of macroinvertebrates; and to infer water quality, habitat suitability, and biological integrity of Great Lakes coastal wetlands.

Ecosystem Objective

Coastal Wetland habitats are critical areas for many invertebrate species of ecological importance. Conservation of remaining coastal wetlands and restoration of previously destroyed wetlands are vital components of restoring the Great Lakes ecosystem and this sub-indicator can be used to report progress toward such an objective.
This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species."

Significant wetland areas in the Great Lakes system that are threatened by urban development, agricultural development and waste disposal activities should be identified, preserved and, where necessary, rehabilitated. Conducting monitoring activities will gather definitive information on the location, severity, extent, and frequency of degradation of aquatic habitat in coastal wetlands. This sub-indicator supports the restoration and maintenance of the chemical, physical and biological integrity of the Great Lakes basin and beneficial uses dependent on healthy wetlands (Annex 1 GLWQA). Wetland macroinvertebrate IBI scores can be used for identifying wetlands that have sensitive taxa or high diversity, and therefore, should be considered for preservation. IBI scores can also identify areas that are good candidates for restoration, for example, a wetland with a low IBI score that has other desirable qualities (high recreation potential, a diverse vegetative community, etc.) may benefit from interventions to improve water quality.

Measure

This sub-indicator uses the relative abundance and richness of sensitive taxa, tolerant taxa, and community diversity of aquatic macroinvertebrates in an index of biotic integrity to assess wetland condition. The IBI was developed originally by the Great Lakes Coastal Wetland Consortium (GLCWC) in 2002, published in 2004 (Uzarski et al. 2004) and protocols were finalized in 2008 (GLCWC 2008). In effort to control for variation in macroinvertebrate communities due to natural differences in habitat structure, the IBI is based on macroinvertebrate sampling at the vegetation zone level. Currently, the IBI can be applied to wetlands that have mono-dominant zones of *Schoenoplectus* spp. (formerly *Scripus* spp.) and/or zones wet meadow species (*Poaceae* and *Carex* spp.). IBI metrics for other common vegetation zones including *Typha* and *Phragmites* are in development. Table 3 shows the IBI scoring system for the current published zones, which were used in this report. For more information on the development of the metrics used in the IBI see Uzarski et al 2004.

The scores presented here were calculated using the data collected through the Great Lakes Coastal Wetland Monitoring Program. A sub-set of all coastal wetlands that meet program size and hydrologic criteria are sampled each year, with the majority of qualifying sites sampled over a 5-year period. The annual site selection scheme is designed in order to capture lake, regional, and geomorphic variation within each year. In addition to the sites selected through the stratified selection scheme, a number of benchmark sites are sampled for either long term reference or to assist in restoration efforts. More information on the sampling design and data collection methods is found in the Great Lakes Coastal Wetland Monitoring Program Quality Assurance Project Plan (GLCWMP QAPP) at greatlakeswetlands.org/Sampling-protocols.

Macroinvertebrate samples are collected at three replicate locations within up to four mono-dominant vegetation zones (wet meadow, *Typha* spp., *Phragmites*, Lily, *Schoenoplectus* spp., submergent aquatic vegetation, etc.) in each wetland site. Invertebrates are collected by sweeping the water column and sediment with dip nets, placing net contents into gridded trays, and picking out the invertebrates in set time in accordance with standard Great Lakes Coastal Wetlands Monitoring Program (GLCWMP) protocols, which can be found at greatlakeswetlands.org/Sampling-protocols. Invertebrates are preserved in ethanol and identified to a predetermined lowest operational taxonomic unit, which is genus for most insects, and family or order for most non-insects.

After identification, invertebrate data are uploading into the project database housed at GreatLakesWetlands.org. Metrics and IBI scores are calculated for each site that has *Schoenoplectus* spp. and/or wet meadow zones.. Depending which zones are present an IBI score between 1-6 and qualitative category “extremely degraded” to “reference” is calculated based on the sum of the scores of all zones compared to the total possible scores (GLCWMP QAPP).

Many sites that are sampled do not have *Schoenoplectus* spp. or wet meadow zones, and therefore, are not associated with an IBI score. For a lake to be assigned a status in this report at least 50% of the sites sampled by the GLCWMP needed to be given an IBI score, and to be assigned a trend at least 50% of the sites sampled each year needed to be associated with an IBI score.

Ecological Condition

Coastal wetlands trap, process, and remove nutrients and sediment from Great Lakes nearshore waters. However, over half of all Great Lakes coastal wetlands have been destroyed by human activities and many remaining coastal wetlands suffer from anthropogenic stressors such as nutrient and sediment loading, fragmentation, invasive species, shoreline alteration, and water level control, as documented by a binational Great Lakes-wide mapping project (Albert and Simonson 2004; Ingram and Potter 2004).

To restore/maintain the overall biological integrity of Great Lakes coastal wetlands, both biotic and physical function need represented through monitoring efforts. Coastal wetland macroinvertebrates are an important component to both wetland and pelagic food webs (Sierszen et al. 2014). Their community composition can carry signs of intermittent anthropogenic disturbances that may not be visible at the time of sampling. Macroinvertebrate communities often correlate with factors such as water depth, vegetation, and sediment type (Cooper et al 2014) and vary greatly within a single wetland providing insight into the diversity of habitats present.

Since 2011 the GLCWMP has sampled approximately 200 wetlands annually, funded by the Great Lakes Restoration Initiative (GLRI) through 2025 (about $2 million per year). On average 120 of those are sampled for macroinvertebrates because they meet protocol requirements, are accessible to crews and are selected as part of the statistical sampling design. In 2018, 117 sites were sampled and in 2019, 119 sites were sampled. As of 2019, more than 550 distinct wetland sites have been sampled for macroinvertebrates, many have been sampled more than one time meaning that there have been over 900 macroinvertebrate sampling events in total. This report focuses on sampling events that occurred within the GLCWMP statistically designed sampling scheme, with 553 distinct wetland sites and 904 sampling events (Tables 1 and 2, Figure 1).

As of 2019, the average number of macroinvertebrate taxa (taxa richness) per site was 38 for Canadian sites and 39 for US sites, but wetlands with high richness had more than twice this number (Table 1). The average number of non-native invertebrate taxa in coastal wetlands was less than 1, with a maximum of no more than 5. It is important to note that the one-time sampling method used at coastal wetland sites may not be capturing all of the non-native taxa and it is not necessarily intended to. Furthermore, some non-native macroinvertebrates are very cryptic, may resemble native taxa, and may not yet be recognized as invaders to the Great Lakes.

Comparing taxa richness within lakes, Lake Ontario and Lake Erie the average taxa richness was approximately 34, while Lakes Huron and Michigan averaged approximately 41 taxa, and Lake Superior 43 (Table 2). While Lake Ontario and Lake Erie have few sites with IBI scores, they appear to have consistently lower taxa richness. It is predicted that when IBI are developed for the vegetation zones that dominate these lakes (*Typha* spp.) that they will on average score lower than Lakes Huron, Michigan, and Superior.

Patterns are likely driven by differences in habitat complexity, which may in part be due to the loss of wetland habitats on Lakes Erie and Ontario from diking and water level control, respectively. There is little variability among lakes in non-native taxa occurrence, although Lakes Erie and Huron had wetlands with 4-5 non-native taxa, which was slightly higher than other lakes. In each lake, a portion of wetlands had only native taxa; however, as noted above, this does not necessarily mean that these sites do not harbor non-native macroinvertebrates.

Lakes Michigan and Huron both have a wide distribution of site quality. Both lakes have multiple degraded sites, but also the some of the most taxa rich sites and those with high IBI scores (Figure 6 and Figure 8). The maximum number of invertebrate taxa was higher in Lakes Huron and Michigan wetlands (>80) than for the most invertebrate-rich wetlands in other lakes, which have a maximum closer to 70 taxa (Table 2).

Invertebrates are less mobile than fish and reflect a more localized scale of disturbance. Invertebrates are also associated with more zones that fish cannot access, and therefore, coastal wetland invertebrates and fish cannot be expected to show the same trends.

Many of the sites in Lakes Erie and Ontario only contained submersed aquatic vegetation, *Phragmites*, or *Typha* and therefore the IBIs cannot be used in those areas at this time, but scores for other zones will be available for all years of the project once IBIs for these vegetation zones are developed.

Linkages

Physical alteration and eutrophication of wetland ecosystems continue to be a threat to invertebrates of Great Lakes coastal wetlands. Both can promote establishment of non-native vegetation, and physical alteration can destroy plant communities altogether while changing the natural hydrology of the system. Invertebrate community composition is directly related to vegetation type and densities; changing either of these components will negatively impact the invertebrate communities.

Linkages to other sub-indicators in the indicator suite include:

- Hardened Shorelines – physical modifications to the shoreline have disrupted coastal and nearshore processes, flow and littoral circulatory patterns, altered or eliminated connectivity to coastal wetlands/dunes, and have altered nearshore and coastal habitat structure.
- Along many coastal wetlands, residential development has altered wetlands by nutrient enrichment from fertilizers and septic systems, shoreline alterations for docks and boat slips, filling, and shoreline hardening. Shoreline hardening can eliminate wetland vegetation, particularly during periods of high-water levels when vegetation communities cannot move landward. Robust vegetation stands are key for wetland macroinvertebrate habitat.
- Mechanical alteration takes a diversity of forms, including diking, ditching, dredging, filling, and shoreline hardening. With all of these alterations, non-native species are introduced by construction equipment or in introduced sediments. Changes in shoreline gradients and sediment conditions are often adequate to allow non-native species to become established (see below).
- Land Use – Agriculture degrades wetlands in several ways, including nutrient enrichment from fertilizers, increased sediments from erosion, increased rapid runoff from drainage ditches, introduction of agricultural non-native species and destruction of inland wet meadow zone by plowing and diking, and addition of herbicides.

  Great Lakes Wetland invertebrate communities have been shown vary with the percent agriculture in the surrounding watershed (Cooper et al. 2014) as well as with proximity to agricultural drainage ditches (Schock et al. 2014). It is currently hypothesized that nutrient and sediment pollution from agricultural runoff impact macroinvertebrate community structure by changing the food web structure or favoring tolerant taxa. Agricultural runoff may also promote the establishment of invasive vegetation like *Typha angustifolia* and *Phragmites australis* which also may impact aquatic macroinvertebrate structure.
- Urban development degrades wetlands by hardening shoreline, filling wetland, adding a broad diversity of chemical pollutants, increasing stream runoff, adding sediments, and increased nutrient loading from sewage treatment plants. In most urban settings, almost complete wetland loss has occurred along the shoreline. Because of this the impacts of urban or industrial development on coastal wetland macroinvertebrates is less understood.

- Impact of Aquatic Invasive Species – Non-native species are introduced in many ways. Some were purposefully introduced as agricultural crops or ornamentals, later colonizing in native landscapes. Others came in as weeds in agricultural seed. Increased sediment and nutrient enrichment allow many of the worst aquatic weeds to out-compete native species. Most of the worst non-native species are either prolific seed producers or reproduce from fragments of root or rhizome. The transformation of wetlands, like those in Saginaw Bay, from majority native *Schoenoplectus spp*. to majority invasive *Phragmites*, will have changed physical habitat structure and food web dynamics. It has been shown that patches of invasive *Typha spp* have differences in macroinvertebrate community structure and lower total macroinvertebrate biomass relative to surrounding native vegetation in the same wetland complex, possibly due to changes in habitat complexity, water chemistry characteristics, or food web changes (Lawrence et al. 2016).

  Non-native animals have also been responsible for increased degradation of coastal wetlands. The faucet snail (*Bithynia tentaculata*) is an example of a prolific macroinvertebrate invader of particular interest to U.S. Fish and Wildlife Service and others because it carries parasites that can cause disease and die-offs of waterfowl.

- Precipitation Amounts in the Great Lakes Basin/Tributary Flashiness – Change in atmospheric temperature will potentially affect the number of extreme storms in the Great Lakes region which will, in turn, affect coastal wetlands. Extreme storms would likely affect riverine systems the most through increased sediment movements, particularly in flashy watersheds. Increased sedimentation can interfere with respiratory organs of more sensitive macroinvertebrate taxa.
- Water Levels – Water level change has strong influences on Great Lakes habitat and biological communities associated with Coastal Wetlands. Lake levels have a major influence on undiked coastal wetlands and are basic to any analysis of wetland trends. Water levels influence the vegetation structure in wetlands which in turn structure the habitat available for macroinvertebrates. Altered water levels will also influence the spread of invasive plant species, such as *Phragmites*, that can take over native vegetation stands and alter habitat available to macroinvertebrates and other organisms.
- This sub-indicator links directly to the other sub-indicators in the Habitats and Species category, particularly the other Coastal Wetlands-related sub-indicators. Individual IBIs, such as the fish IBI used in this assessment, are useful as independent indicators, but evaluation of this indicator in combination with the other coastal wetland indicators is key to an overall assessment of Great Lakes coastal wetland health. This is because the different sub-indicators function and indicate anthropogenic disturbance at different spatial and temporal scales and have varying resolution of detection. See greatlakeswetlands.org/Sampling-protocols for details on indicator metrics.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: www.greatlakeswetlands.org		

Data Limitations

The greatest limitation to the Great Lakes coastal wetland macroinvertebrate indicator is that the current published index of biotic integrity only covers sites that have a wet meadow or bulrush vegetation zones. Table 2 shows that only around half of all sampling events have an associated IBI score, and in Lake Erie and Lake Ontario the ratio is even lower. This has been even more problematic as high-water levels and colonization by invasive plant species both can lower the number of *Schoenoplectus* and wet meadow zones. This limitation is currently being addressed with efforts to expand the IBI to include other common vegetation zones, which is expected to be completed before the next indicator report.

Additional Information

The invertebrate IBI is a multi-metric indicator, developed from a composite of specific parameters ("metrics") used to describe the invertebrate community structure, function, and abundance. The IBI provides a rigorous approach that quantifies the biological condition of the invertebrate community based on data from wetlands across gradient of anthropogenic disturbance and stratified by region and wetland type. It is important to note that the invertebrate IBI has been developed for coastal wetlands that are directly connected to the Great Lakes, not for those wetlands that are connected hydrologically via groundwater only.

Work to update and expand the macroinvertebrate IBI is underway. These efforts will expand the vegetation zones for which IBIs can be calculated and increase the number of wetlands that can be evaluated based on macroinvertebrate communities.

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Figure 2. The count of all sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=402.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 3. The mean IBI score for Lakes Erie and Ontario (A), Lakes Michigan and Huron (B), and Lake Superior (C) from 2011 to 2019, error bars show 95% confidence interval. The highest possible score is 6 which indicates reference conditions, and lowest possible score is 1 which indicates extreme degradation.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 4. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across Lake Superior.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 5. The count of all Lake Superior sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=57.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 6. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A and B) and from 2016 to 2019 (C and D) across Lake Michigan.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 7. The count of all Lake Michigan sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=72.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 8. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A and B) and 2016 to 2019 (C and D) across Lake Huron.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

Figure 9. The count of all Lake Huron sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=241.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 10. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across Lake Erie.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 11. The count of all Lake Erie sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=8.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 12. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across Lake Ontario.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 13. The count of all Lake Ontario sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=24.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

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Table 1. The number of distinct sites sampled, number of sampling events (includes repeat sampling of sites), total macroinvertebrate taxa in Great Lakes coastal wetlands, and non-native species; summary statistics by country. Data from wetlands sampled 2011 through 2019. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

Country	# Sites	#Samples	Mean	Max	Min	St. Dev
Overall						
Canada	198	333	38.0	76	16	11.3
U.S.	355	572	39.0	86	12	13.0
Non-natives						
Canada			0.6	4	0	0.9
U.S.			0.6	5	0	0.9

Table 2. The number of distinct sites sampled, number of sampling events, number of sampling events related to IBI scores macroinvertebrate total taxa and non-native species found in Great Lakes coastal wetlands by lake. Mean, maximum, and minimum number of taxa per wetland. Data from wetlands sampled 2011 through 2019.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

	Total Macoinvertebrate Taxa						Non-native Taxa		
Lake	*# Sites*	*# Samples*	*# Samples w/IBI*	*Mean*	*Max*	*Min*	*Mean*	*Max*	*Min*
Erie	65	113	8	34	69	12	0.8	4	0
Huron	198	320	241	41	80	12	0.5	5	0
Michigan	88	146	72	41	86	14	0.8	3	0
Ontario	137	221	24	34	71	12	0.8	4	0
Superior	65	105	57	43	69	15	0.1	2	0

Table 3. Indices of biotic integrity metrics for dense *Schoenoplectus*, sparse *Schoenoplectus*, and wet meadow zones. Source: Uzarski et al. 2004.

Metric	*Score = 1*	*Score = 3*	*Score = 5*	*Score = 7*
Odonata richness (genera)	0	>0 to <1	1 to 2	>2
Relative abundance Odonata (%)	0	>0 to <1	1 to 2	>2
Crustacea plus Mollusca richness (genera)	0 to 2	>2 to 4	>4 to 5	>5
Total richness (genera)	<8	8 to 13	>13 to 17	>17
Relative abundance Gastropoda (%)	0	>0 to 3	>3 to 5	>5
Relative abundance Sphaeriidae (%)	0	>0 to 0.05	>0.05	NA
Total number of families	0 to 7	>7 to 12	>12	NA
Relative abundance Crustacea + Mollusca (%)	<8	8 to 30	>30	NA
Evenness	0 to 0.4	>0.4 to 0.7	>0.7	NA
Shannon diversity index	0 to 0.4	>0.4 to 0.9	>0.9	NA
Simpson index	> 0.3	>0.15 to 0.3	0 to 0.15	NA

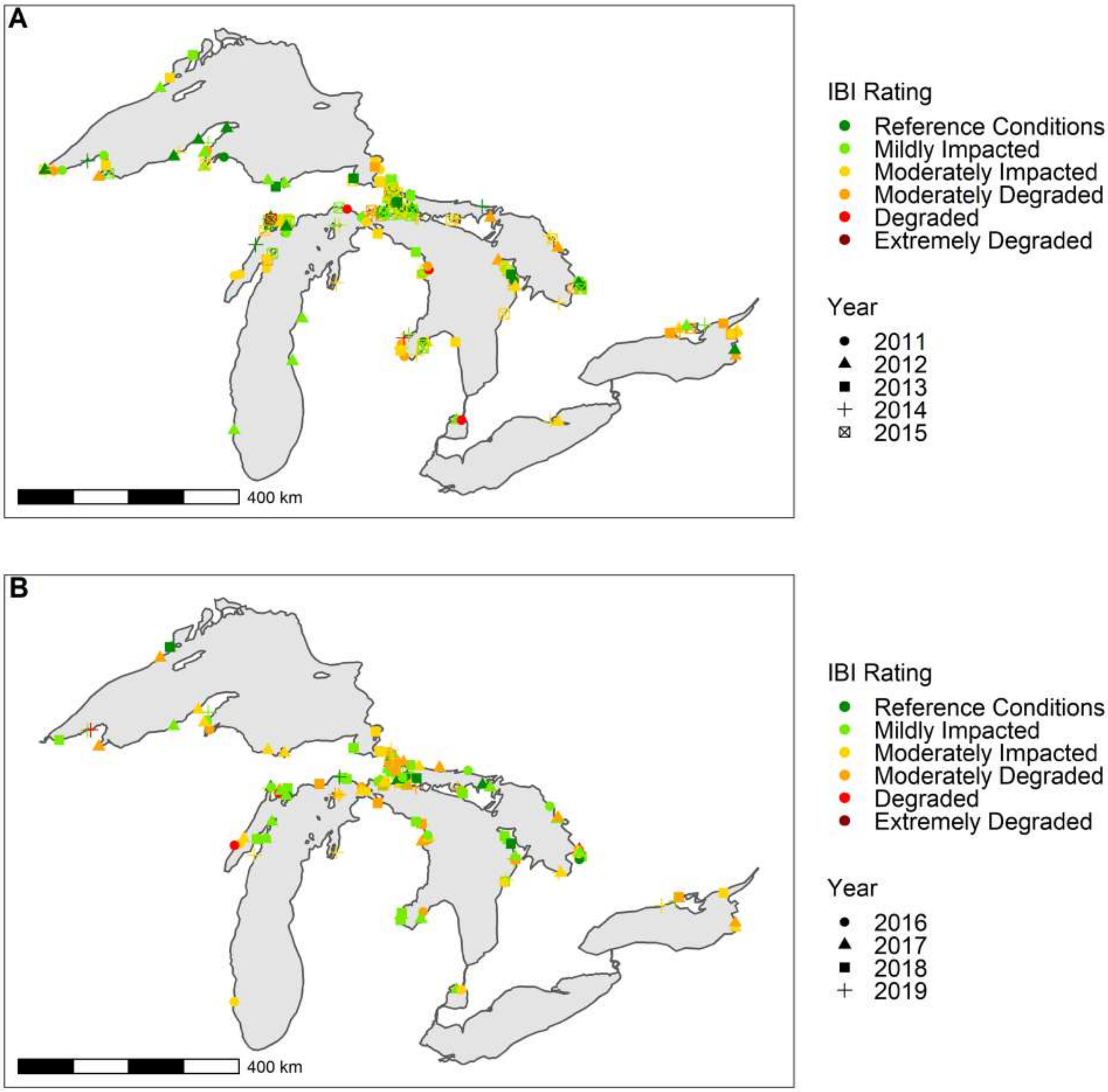

Figure 1. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across the Great Lakes basin.

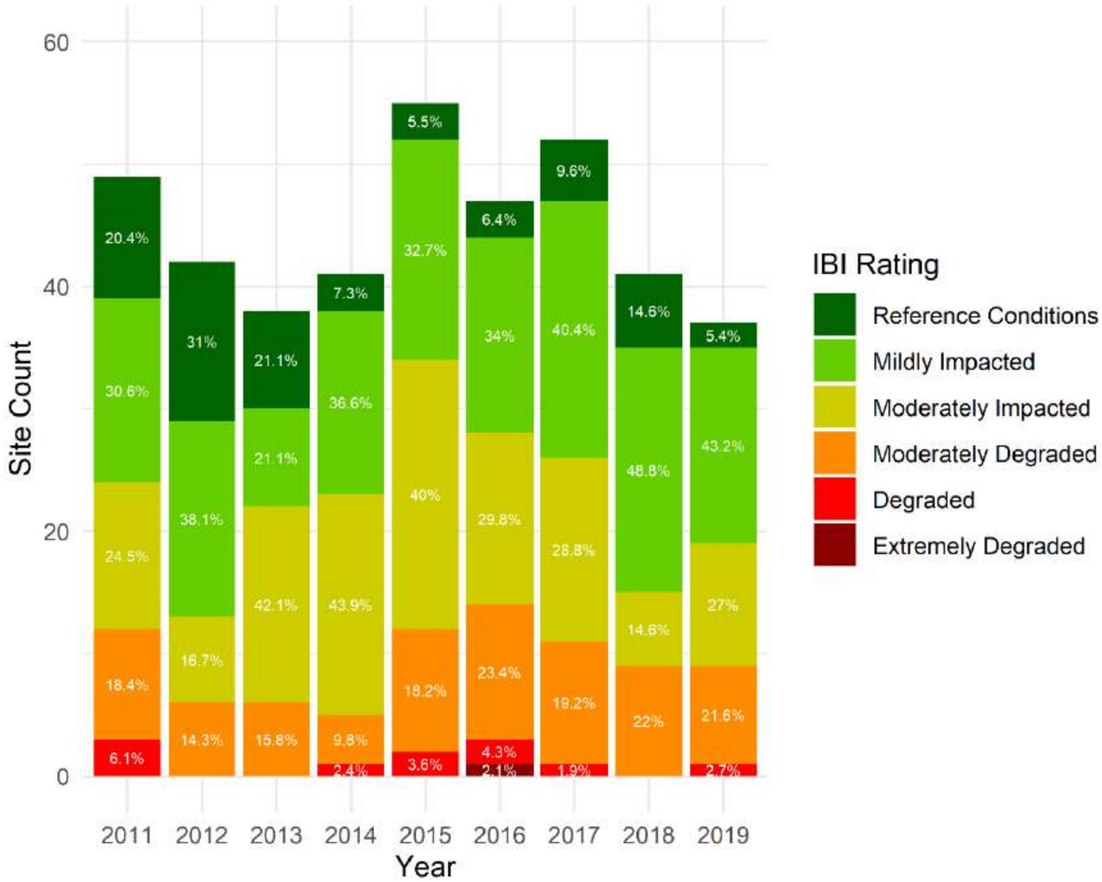

Figure 2. The count of all sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=402. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

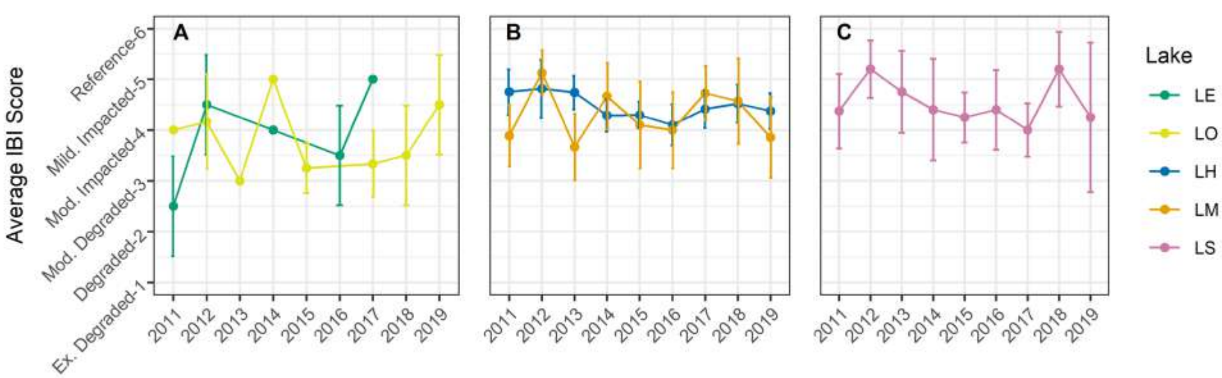

Figure 3. The mean IBI score for Lakes Erie and Ontario (A), Lakes Michigan and Huron (B), and Lake Superior (C) from 2011 to 2019, error bars show 95% confidence interval. The highest possible score is 6 which indicates reference conditions, and lowest possible score is 1 which indicates extreme degradation. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

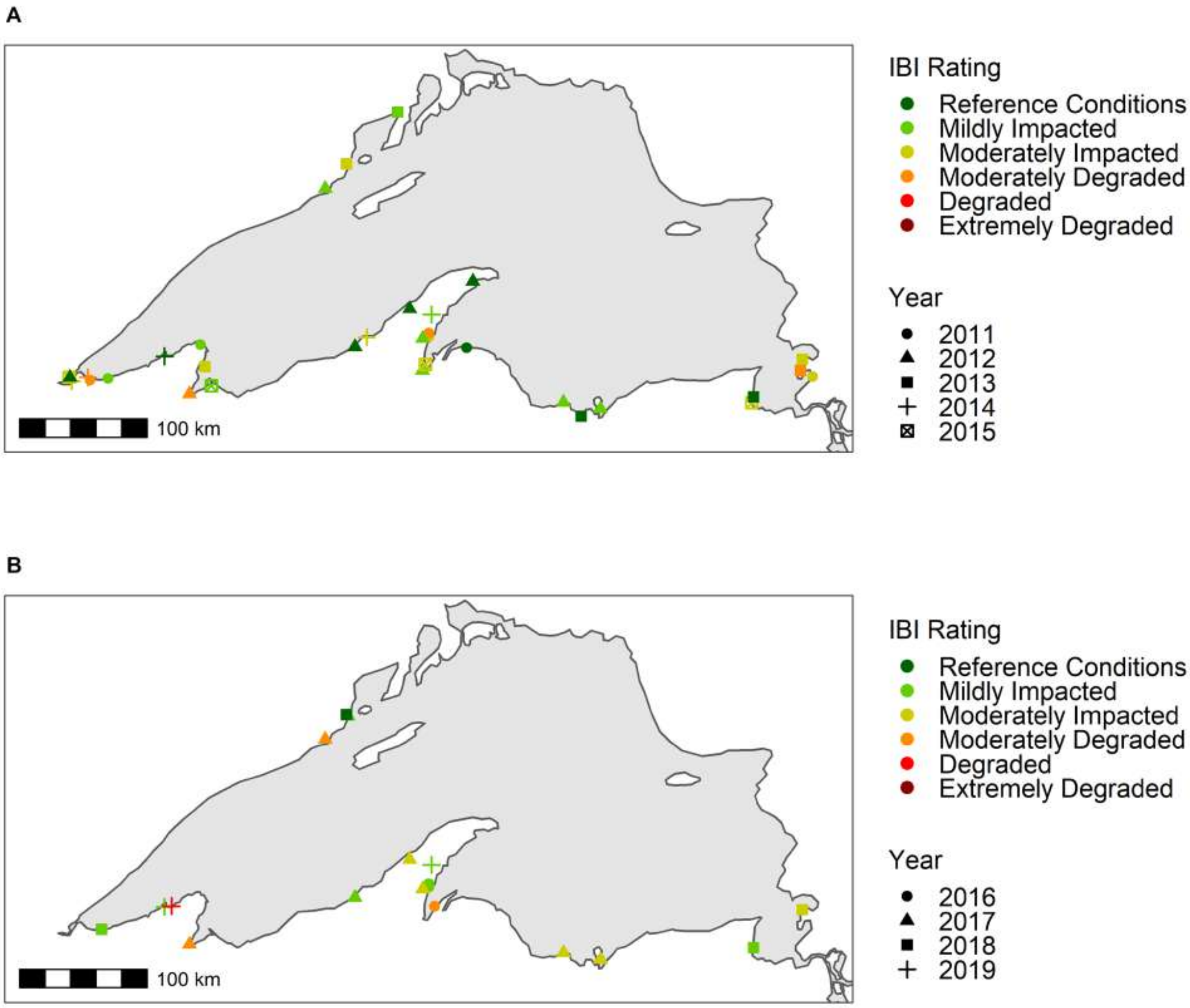

Figure 4. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across Lake Superior. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

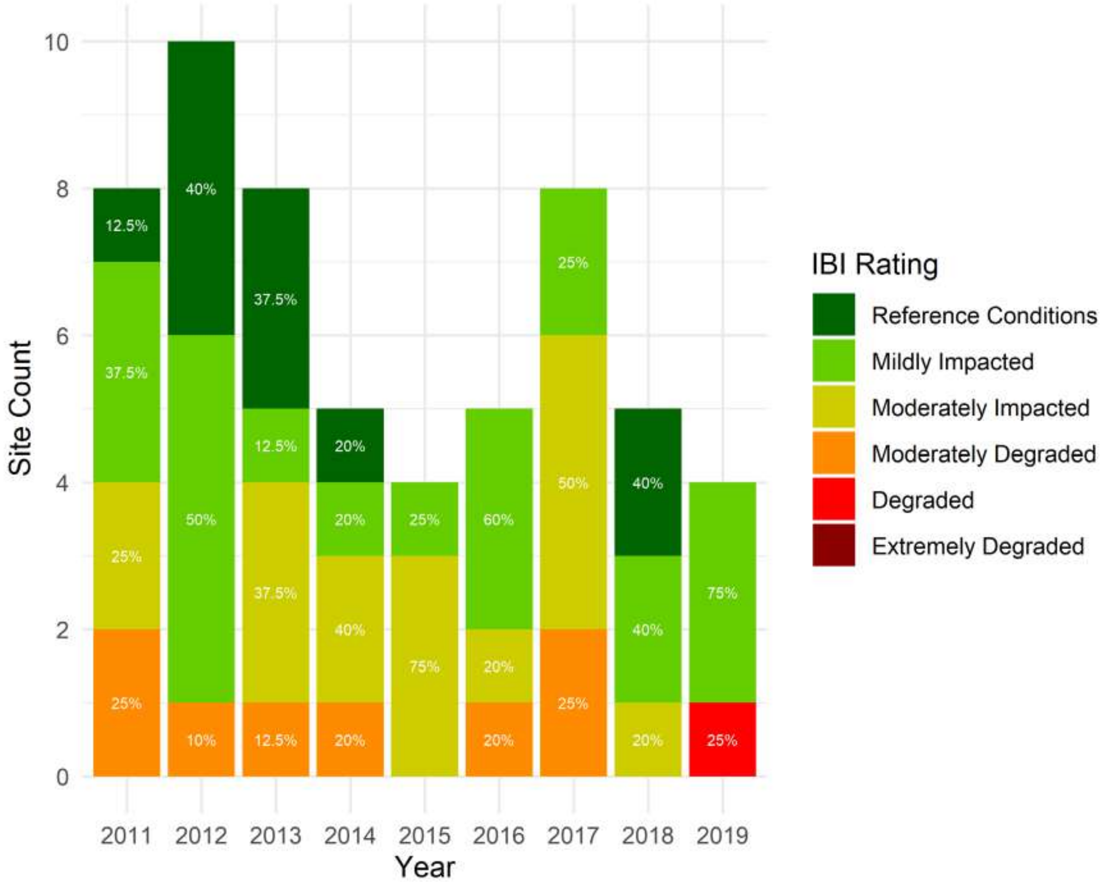

Figure 5. The count of all Lake Superior sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=57. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

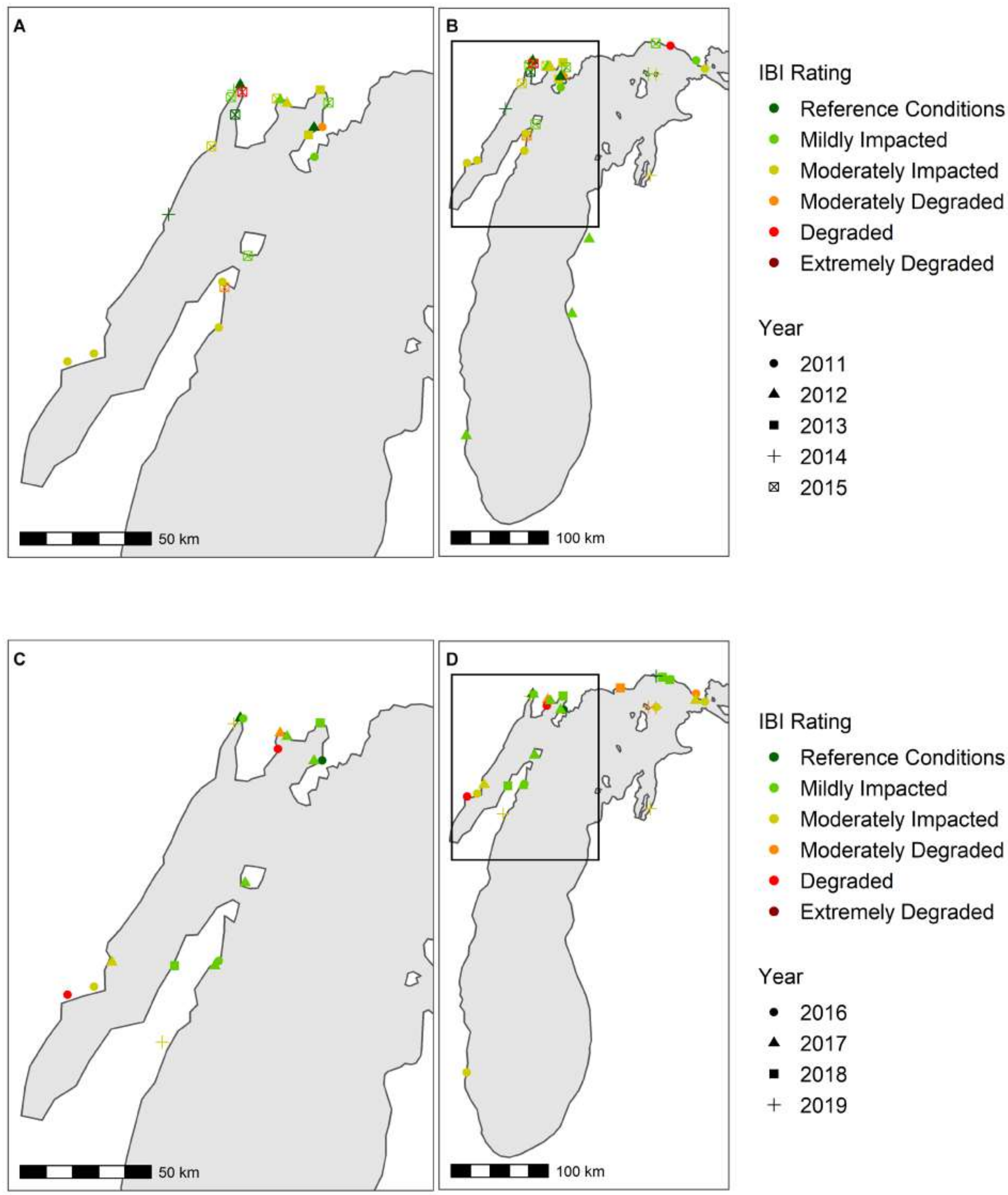

Figure 6. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A and B) and from 2016 to 2019 (C and D) across Lake Michigan. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

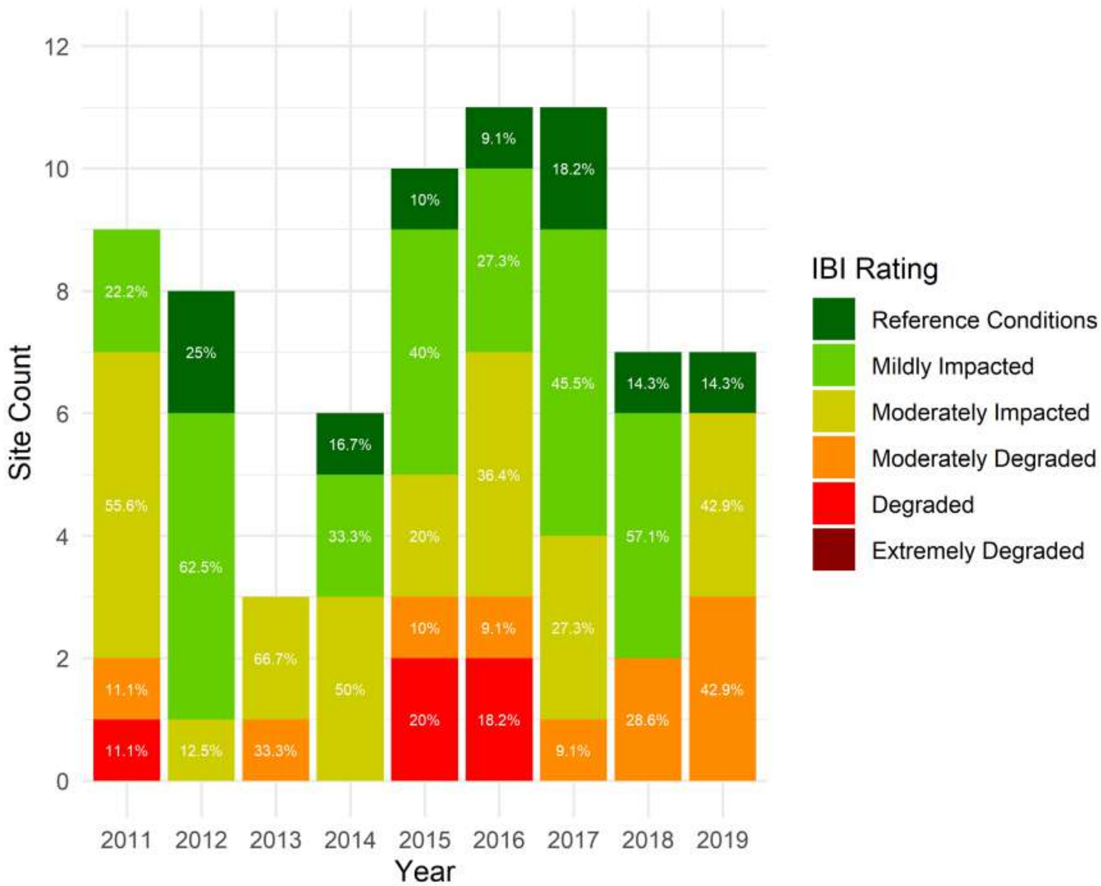

Figure 7. The count of all Lake Michigan sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=72. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

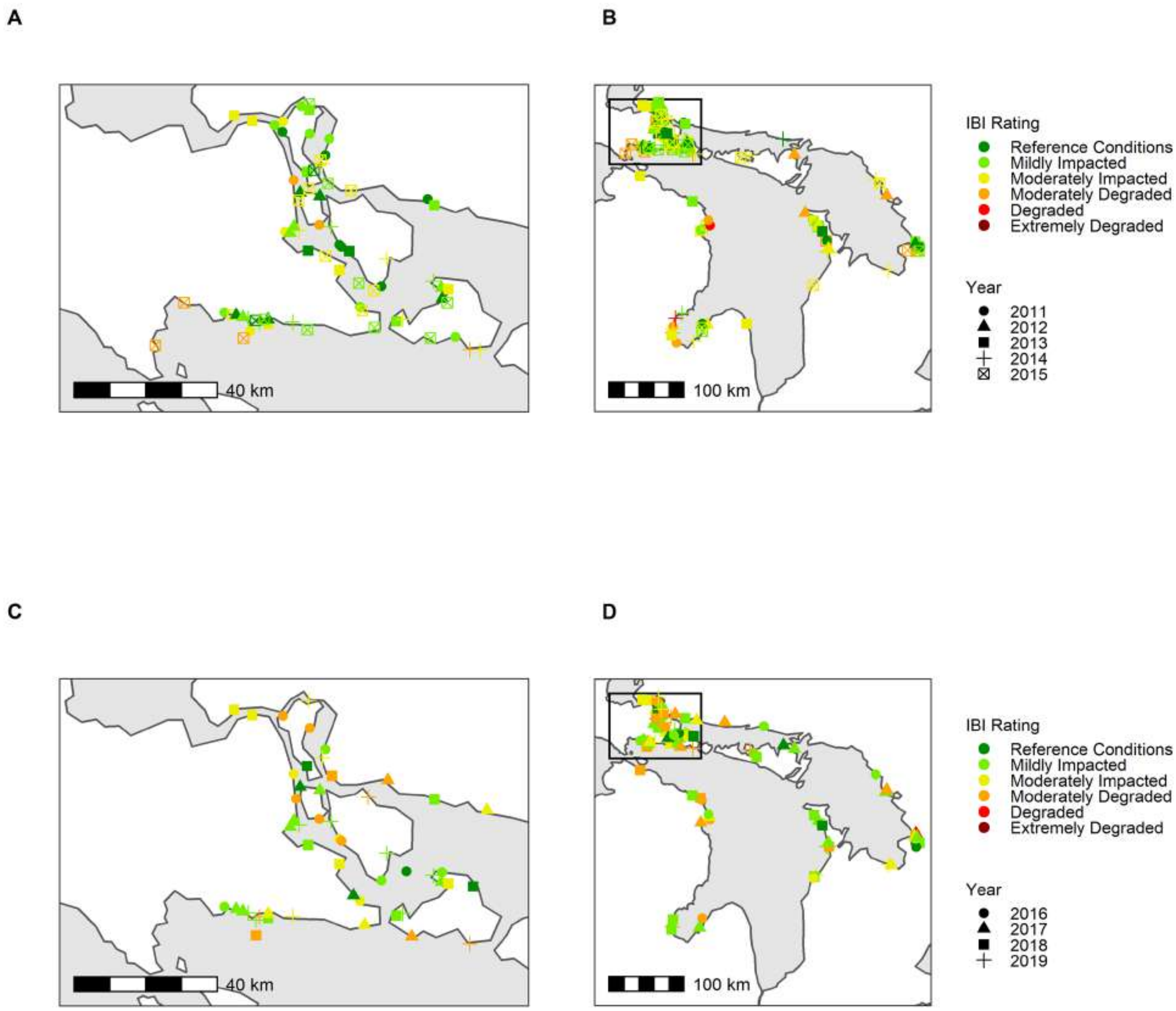

Figure 8. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A and B) and 2016 to 2019 (C and D) across Lake Huron. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

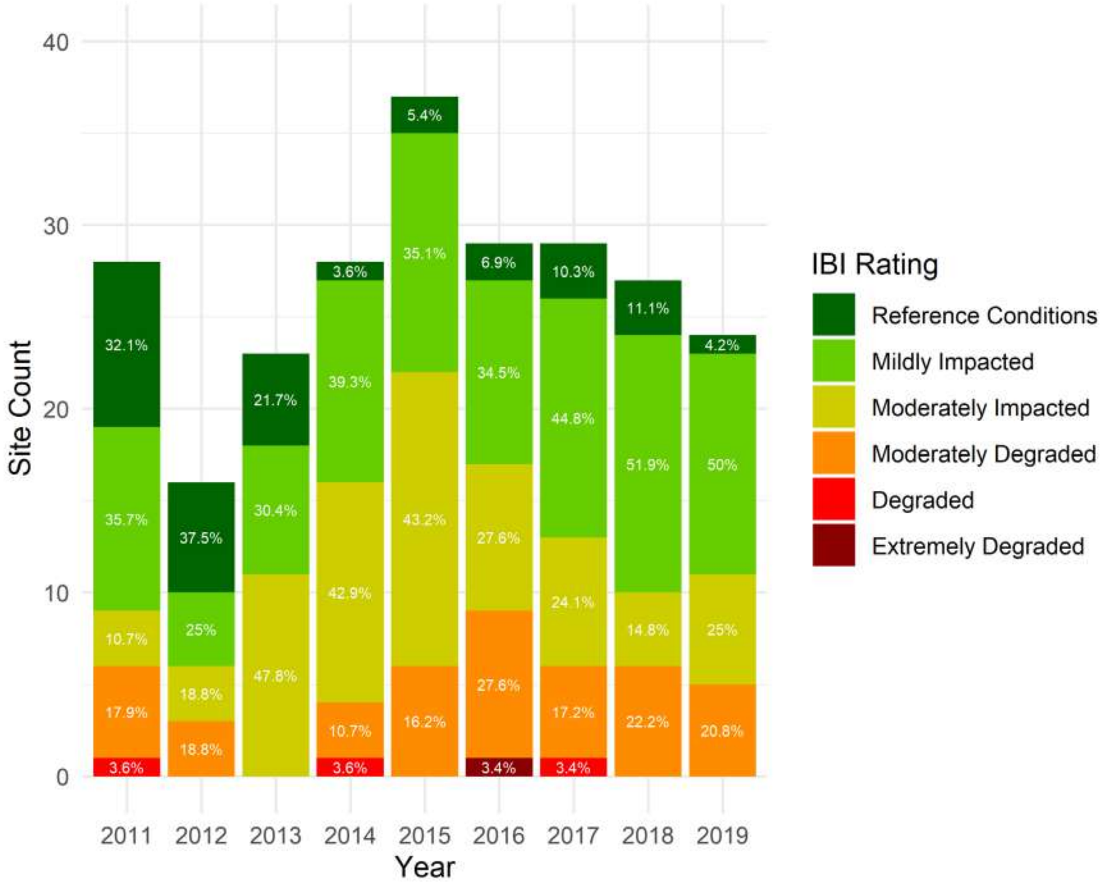

Figure 9. The count of all Lake Huron sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=241. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

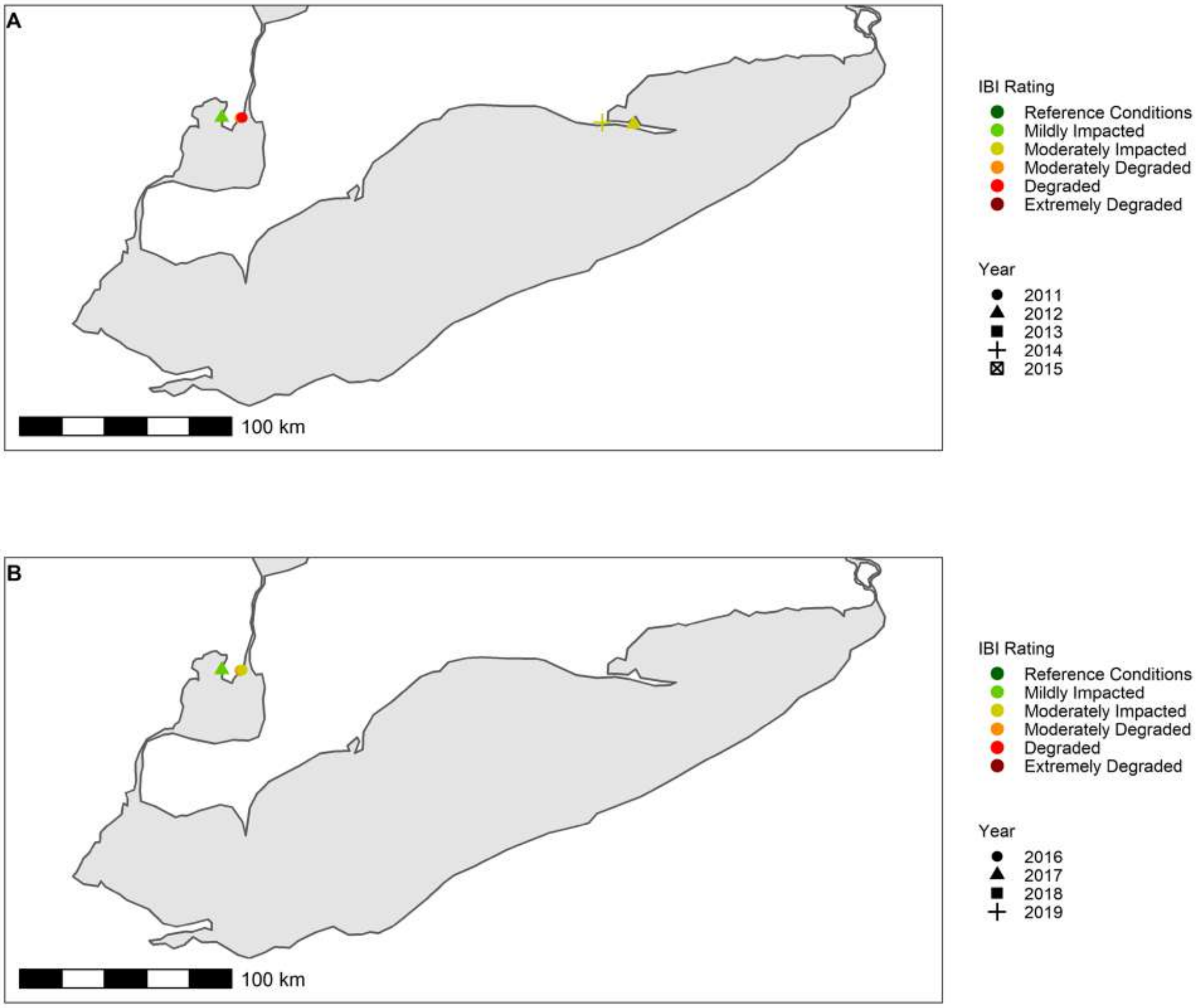

Figure 10. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across Lake Erie. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

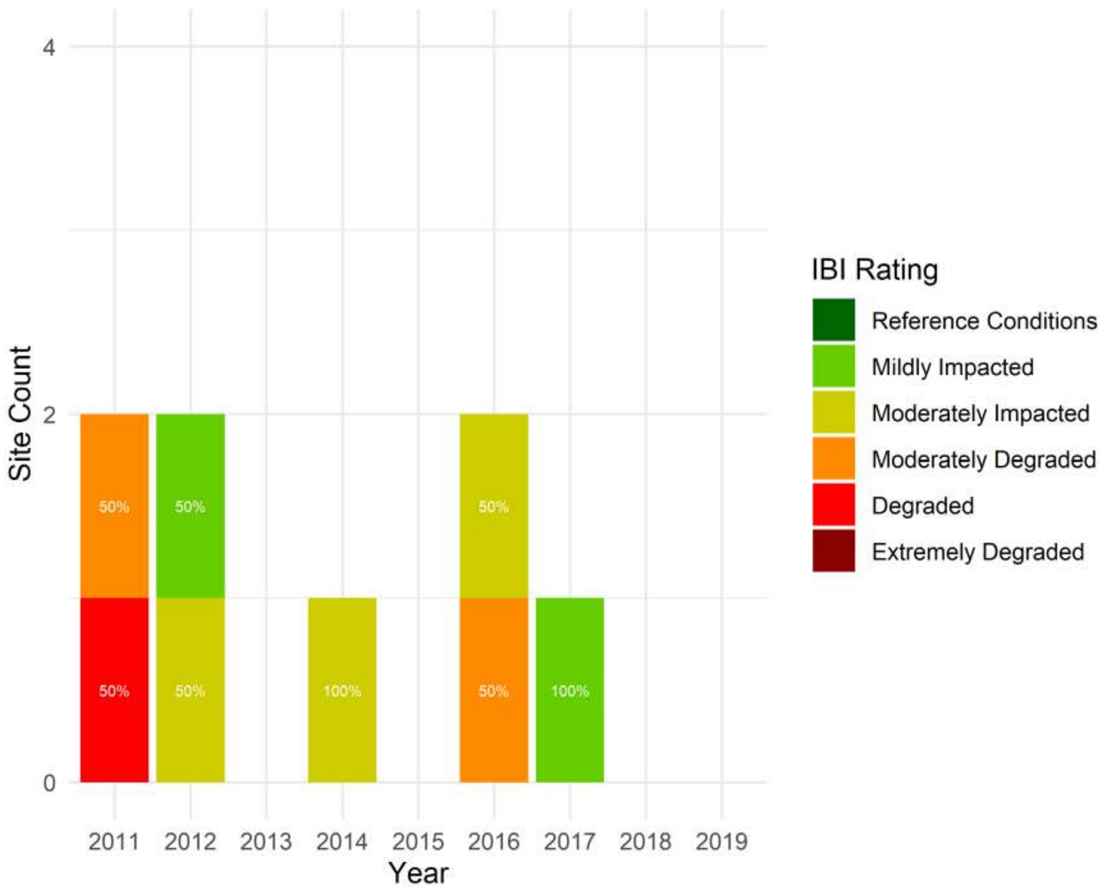

Figure 11. The count of all Lake Erie sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=8. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

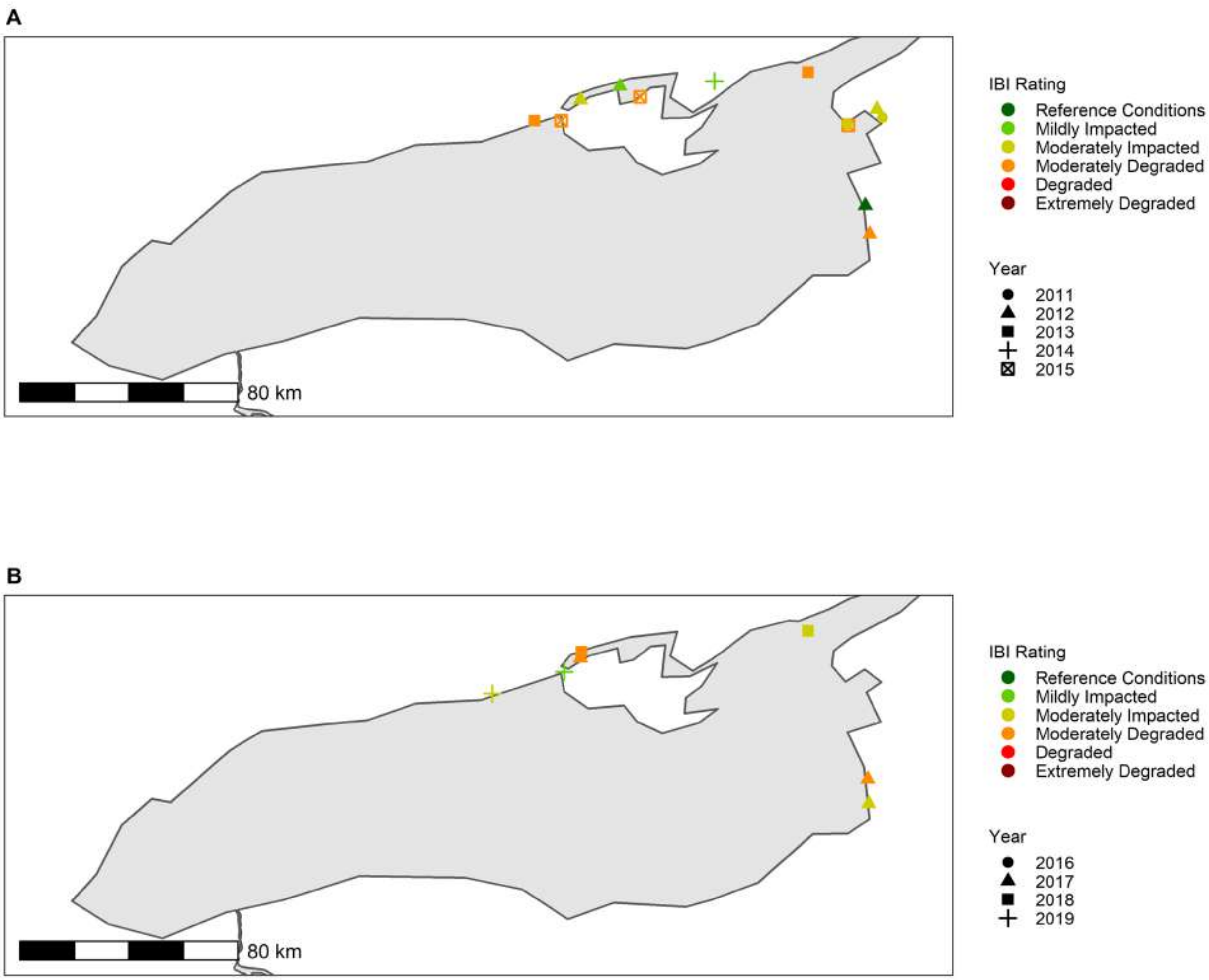

Figure 12. The condition of coastal wetland aquatic macroinvertebrate communities from 2011 to 2015 (A) and from 2016 to 2019 (B) across Lake Ontario. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

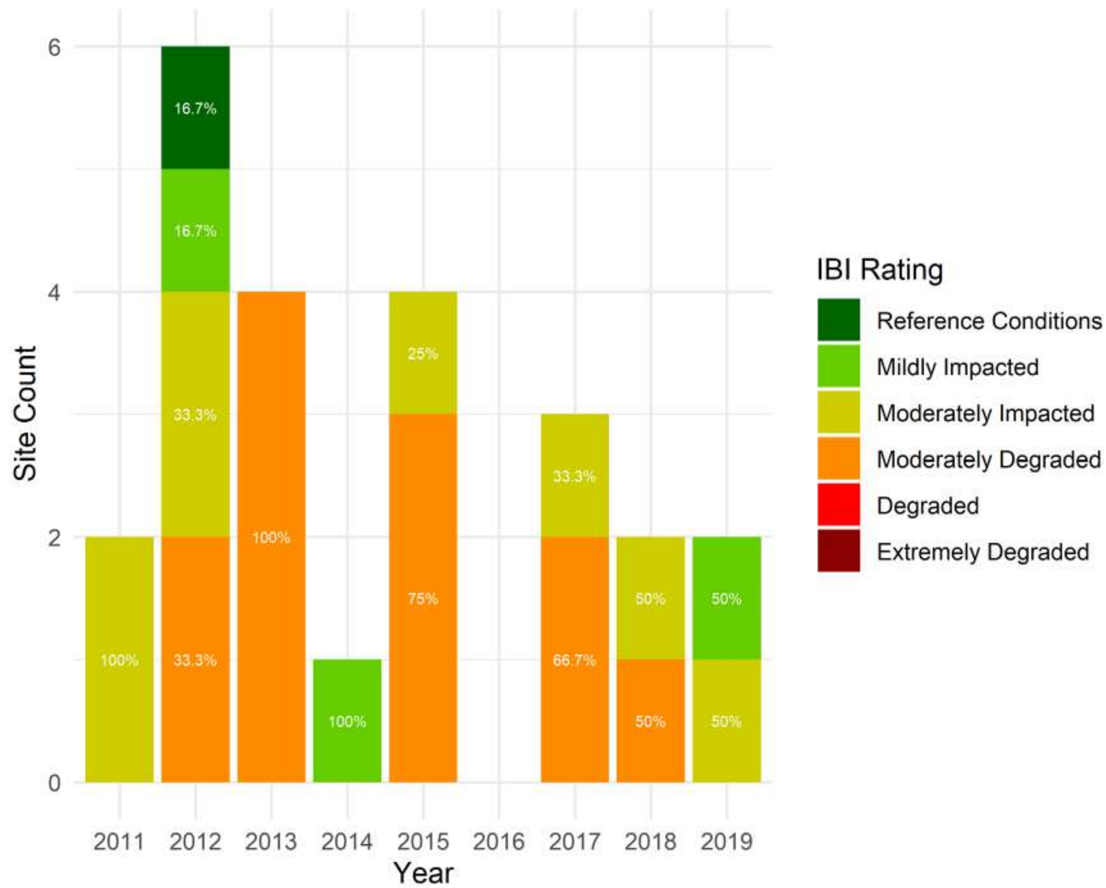

Figure 13. The count of all Lake Ontario sites that fall into each IBI category based on the aquatic macroinvertebrate community each year between 2011 and 2019, n=24. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

Sub-Indicator: Coastal Wetland Fish

Overall Assessment

Status: Fair

Trends:

10-Year Trend (2011-2019)*: Unchanging

Rationale: As of 2019, the majority of wetlands were not degraded or moderately degraded with approximately 30% of the wetlands sampled in all years falling in these degraded categories based on fish community metrics (Figure 1). The remaining sites were moderately impacted, mildly impacted, or reference quality with 20 to 25% of annual sampling events falling into each category. Between 2011 and 2019, 19 to 32% of wetlands were in the reference score category with no directional trend over time in the proportion of reference quality sites. No time-trend over the years was evident for each condition category nor mean IBI scores for each lake (Figures 2 and 3).

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend (2011-2019)*: Unchanging

Rationale: As of 2019, none of the Lake Superior wetlands sampled were classified as degraded and 23% were classified as moderately degraded based on fish community metrics (Figure 4). The trend is evaluated by comparing the status of coastal wetland fish communities over the 9-year period. Between 2011 and 2019, 0 to 50% of Lake Superior wetlands were in the reference category with no temporal trend in the proportion of reference quality sites. The lack of a trend in proportions over the years was consistent for each condition category (Figure 5). Over the 9-year period 12.5 to 55.6% of Lake Superior wetlands were mildly impacted, 0 to 37.5% were moderately impacted, 0 to 50% were moderately degraded and none were degraded.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Michigan

Status: Fair

10-Year Trend (2011-2019)*: Undetermined

Rationale: As of 2019, the majority of Lake Michigan wetlands were not degraded with 33% of sampled wetlands in the degraded categories based on fish community metrics over the 9-year period (Figure 6). The trend is determined by comparing the status of coastal wetland fish communities over the 9-year period. Between 2011 and 2019, 0 to 50% of Lake Michigan wetlands were in the reference score category with no temporal trend in the proportion of reference quality sites. The lack of a temporal trend over the years was consistent for each condition category (Figure 7). Over the 9-year period, year to year variation was high: values ranged from 0 to 35.7% of Lake

Michigan wetlands categorized as mildly impacted, 16.7 to 58.3% moderately impacted, 0 to 50% moderately degraded and 0 to 12.5% degraded.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend (2011-2019)*: Unchanging

Rationale: As of 2019, the majority of sampled Lake Huron wetlands were not degraded, with 28.2% of the

Lake Huron wetlands falling in the degraded categories based on fish community metrics over the 9-year period (Figure 8). The trend is determined by comparing the status of coastal wetland fish over the 9-year period. Between 2011 and 2019, 20.7 to 47.6% of Lake Huron wetlands were in the reference score category with no temporal trend in the proportion of reference quality sites. The lack of a temporal trend over the years was consistent for each condition category (Figure 9). Over the 9-year period 13.8 to 31.6% of Lake Huron wetlands were mildly impacted, 9.1 to 37.9% were moderately impacted, 0 to32.1% were moderately degraded and 0 to 23.8% were degraded.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend (2011-2019)*: Undetermined

Rationale: As of 2019, 59% of the Lake Erie wetlands sampled fell into the degraded categories based on fish community metrics (Figure 10). The trend is determined by comparing the status of coastal wetland fish over the 9-year period. Between 2011 and 2019, 0 to 33.3% of Lake Erie wetlands were in the reference score category with no temporal trend in the proportion of reference quality sites. The lack of a temporal trend over the years was consistent for each condition category, however there is a high level of variation in the proportion of sites that fall into the degraded categories each year (Figure 11). Over the 9-year period 0 to 22.2% of Lake Erie wetlands were mildly impacted, 0 to 37.5% were moderately impacted, 0 to 55.6% were moderately degraded and 0 to 44.4% were degraded.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Fair

10-Year Trend (2011-2019)*: Unchanging

Rationale: As of 2019, the majority of Lake Ontario wetlands were not degraded, with 21% of the

sampled wetlands falling into the degraded categories based on fish community metrics (Figure 12). The trend is determined by comparing the status of coastal wetland fish over the 9-year period. Between 2011 and 2019, 9.1to 52.6% of Lake Ontario wetlands were in the reference score category with no temporal trend in the proportion of reference quality sites. The lack of a temporal trend over the years was consistent for each condition category (Figure 13). Over the 9-year period 15 to 42.9% of Lake Ontario wetlands were mildly impacted, 4.8 to 31.8% were moderately impacted, 7.1 to 35% were moderately degraded and 0 to 13.6% were degraded.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Status Assessment Definitions

Good: No wetlands are in the degraded categories but instead fall within reference conditions, mildly impacted, or moderately impacted.

Fair: The majority of the wetlands are not in the degraded categories.

Poor: The majority of the wetlands are in the degraded categories.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: This metric increased in score for the majority of the sites.

Unchanging: This metric did not substantially change for the majority of the sites.

Deteriorating: This metric decreased in score for the majority of the sites.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

An endpoint for this sub-indicator will be established based on fish communities of reference systems. Data are being evaluated for patterns by lake, ecoregion, wetland type, and vegetation zone. An endpoint for this sub-indicator is not possible at this time as the data are still being collected and analyzed.

Sub-Indicator Purpose

The purpose of this sub-indicator is to track the trends of Great Lakes coastal wetland ecosystem health by measuring the composition of fish communities, and to infer suitability of habitat and water quality for Great Lakes coastal wetland fish communities.

Ecosystem Objective

Coastal wetland habitats are critical spawning, nursery, forage, and shelter areas for many fish species of ecological and economic importance. Conservation of the remaining coastal wetlands and restoration of previously destroyed and degraded wetlands, are vital actions for restoring the Great Lakes ecosystem. This sub-indicator can be used to report progress toward such an objective.

This sub-indicator supports General Objective #5 of the 2012 Great Lakes Water Quality Agreement stating that waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species." Sub-indicator work supports Annex 7 of GLWQA which calls for restoration and maintenance of the diversity of the fish community of Great Lakes coastal wetlands while indicating overall ecosystem health. Significant wetland areas in the Great Lakes system that are threatened by urban and agricultural development and waste disposal activities should be identified, preserved and, where necessary, rehabilitated. This sub-indicator supports the restoration and maintenance of the chemical, physical and biological integrity of the Great Lakes basin and beneficial uses dependent on healthy wetlands (Annex1 GLWQA).

Measure

Coastal wetland fish communities are monitored by the Great Lakes Coastal Wetland Monitoring Program (GLCWMP), which uses consistent methods across the whole Great Lakes Basin. Fish are sampled using three replicate fyke nets of 4.8 mm mesh in each major plant zone (wet meadow, *Typha spp.*, *Phragmites*, Lily, *Schoenoplectus* spp., etc.) in each wetland for one net-night (Uzarski et al., 2005; Uzarski et al. 2016). Sampling locations correspond with macroinvertebrate and water quality sampling. The timing of sampling corresponds with the maturity of the vegetation in each system (no earlier than mid-June and no later than August due to migration patterns of the fish communities). Dominant vegetation zones were identified because different zones support different fishes (Uzarski et al. 2005). There are two sizes of fyke nets that can be used depending on depth: 0.5-m x 1-m opening and 1-m x 1-m opening. The smaller nets are placed in water that is 0.20-0.5 m deep and the larger fyke nets are placed in water that is greater than 0.50 m deep. The leads are 7.3 m long with 1.8 m long wings. Nets are haphazardly placed a minimum of 20 m apart in each vegetation zone. The fyke nets are placed perpendicular to the vegetation zone being sampled, so that fish swimming within the vegetation zone are captured.

Any fish collected that are 20 mm or longer are identified to species and counts are recorded per net. Fish abundance by taxon is used to calculate the Great Lakes Coastal Wetland Monitoring Program (GLCWMP) IBI scores (Cooper et al. 2018). The Great Lakes Coastal Wetland Consortium (GLCWC) developed indices of biological integrity (IBIs) in 2002 and protocols were finalized in 2008 (GLCWC 2008). The GLCWMP Index of Biotic Integrity (IBI) was developed based on measures of richness and abundance, percent exotic species, functional feeding groups, and other species-level parameters. The current version of the GLCWMP IBI developed by Cooper et al. (2018) can be used for evaluation in bulrush, cattail, water lily, and submerged aquatic vegetation zones.

The GLCWMP IBI was the metric of focus for this report for both the status and trends evaluation.

Ecological Condition

Coastal wetlands trap, process, and remove nutrients and sediment from Great Lakes nearshore waters and recharge groundwater supplies. However, over half of all Great Lakes coastal wetlands have been destroyed by human activities, and many remaining coastal wetlands suffer from anthropogenic stressors such as excessive nutrient and sediment loading, habitat fragmentation, invasive species, shoreline alteration and infilling, and water level control, as documented by a binational Great Lakes-wide mapping and attribution project (Albert and Simonson 2004; Ingram and Potter 2004).

Wetland sampling was conducted no earlier than mid-June and no later than August due to migration patterns of the fish communities.

Given the importance of coastal wetlands and the continued threat to these ecosystems, the Great Lakes Restoration Initiative (GLRI) funds the GLCWMP; this group monitors water quality and the status of plant, anuran, bird, fish, and invertebrate communities in wetlands across the Great Lakes Basin in the US and Canada. Under the GLCWMP, approximately 200 wetlands have been sampled annually for at least one taxonomic group, with approximately 100 of those sampled to assess fish communities each year. A total of 109 wetlands were sampled for fish in 2011, 100 in 2012, 94 in 2013, 106 in 2014, 113 in 2015, 103 in 2016, 93 in 2017, 101 in 2018, and 104 in 2019. This report focused on the sites that were selected as part of the statistically designed sampling scheme; in total 760 sampling events at 486 different sites. As of 2019, 103 fish sampling events have taken place in Lake Erie, 255 in Lake Huron, 122 in Lake Michigan, 194 in Lake Ontario, and 86 in Lake Superior (Table 2). As of

2019, nearly 100% of the medium and large (> 4 hectares), hydrologically-connected coastal wetlands in the Great Lakes have been sampled.

An average of 9 and 12 fish species were collected in Canadian and U.S. Great Lakes coastal wetlands, respectively between 2011 and 2019 (Table 1). These data include sites in need of restoration, and some had very few species. However, wetlands with the highest richness had as many as 20 (CA) or 26 (US) fish species. The average number of non-native fish species per wetland was approximately one, though some wetlands had as many as 6 (U.S.). There are several wetlands in which only native fish species were caught in fyke nets, although some non-native fish are adept at net avoidance (e.g. Common Carp). Non-native species lists were compiled by lake from the Great Lakes Aquatic Nonindigenous Species Information System (GLANSIS).

Total fish species captured did not differ greatly by lake, averaging 10 to 12 species per wetland across lakes (Table 2). Lake Ontario wetlands had the lowest maximum number of species with 20 and Lake Michigan had the highest maximum number of species caught (26). Lake Huron and Lake Superior wetlands averaged the lowest mean number of non-native fish taxa (0.5 and 0.6) and Lake Erie had the highest (1.4). All other lakes had a similar average number of non-native fish species per wetland, roughly 1 species.

Richness of invasive species is particularly important in the Fish IBI, as it is one of two metrics that is used in all zone types, the other being percent richness of tolerant taxa (Cooper et al. 2018). Species that contribute to the invasive species richness metric commonly include Common Carp (*Cyprinus carpio*), Round Goby (*Neogobius melanostomus*), Alewife (*Alosa pseudoharengus*), Goldfish (*Carassius auratus*) and Tubenose Goby (*Proterorhinus semilunaris*). Many of these are also tolerant to human disturbance like agricultural input and urban development. This is likely why sites near Green Bay (urban hub), Saginaw Bay (agricultural hub), and western Lake Erie (urban and agricultural hub) consistently score poorly (Figures 6, 8 and 10).

When the fish communities of reference wetlands are compared across the entire Great Lakes, the most similar sites come from the same ecological province, or region, rather than from any single Great Lake or specific wetland types. Fish communities will likely be more sensitive to water chemistry and hydrology related alterations than the vegetation or bird metrics. Kovalenko et al. 2018 found that in the southern Great Lakes, chlorophyll concentration and turbidity of wetland surface water and wave are predictors of fish species presence/absence, while air temperature is the strongest predictor in the northern Great Lakes. Fish are also mobile meaning their community structure reflects multiple connected wetlands as well as adjacent pelagic areas. This may affect our ability to detect the impact of degradation to a single wetland and led to more consistent scores within regions and across years (Figure 1).

There are a number of carp introductions that have the potential for substantial impact on Great Lakes fish communities, including coastal wetlands. Goldfish (*Carassius auratus*) are common in some shallow habitats, and they occurred along with Common Carp young-of-the-year in many of the wetlands sampled along Green Bay. In addition, there are several other carp species, e.g., Grass Carp (*Ctenopharyngodon idella)*, Bighead Carp (*Hypophthalmichthys nobilis)* and Silver Carp (*Hypophthalmichthys molitrix)* that escaped aquaculture operations and are now in the Illinois River and migrating toward the Great Lakes through the Chicago Sanitary and Ship Canal. To prevent the spread of these species into the Great Lakes, from 2002 to 2011 electrical barriers were constructed in the CSSC. One Bighead Carp and one Silver Carp have been found on the Great Lakes side of the barrier, both in the 2010s. Three big head carp were captured in Lake Erie, but all before the year 2000 (Chapman et al. 2021). Grass carp have now been collected in all Great Lakes except Lake Superior, and reproduction of grass carp has been documented in Lake Erie tributaries (Chapman et al. 2021). These species attain large sizes, some are planktivorous, but also eat snails and mussels, while the grass carp eats vegetation. These species represent yet another substantial threat to food webs in wetlands and nearshore habitats with macrophytes (U.S. Fish and

Wildlife Service (USFWS) 2002). If these carp species become established in the Great Lakes, they could negatively impact the status of coastal wetland fish communities by both competing with native fish for resources and by degrading the habitat. The effect that the introduction of these species will have on the coastal wetland fish health is difficult to predict because it will depend on their population densities, distribution, and a variety of other factors.

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Hardened Shorelines – physical modifications to the shoreline have disrupted coastal and nearshore processes, flow and littoral circulatory patterns, altered or eliminated connectivity to coastal wetlands/dunes, and have altered near-shore and coastal habitat structure.
- Along many coastal wetlands, residential development has altered wetlands by nutrient enrichment from fertilizers and septic systems, shoreline alterations for docks and boat slips, filling, and shoreline hardening. Shoreline hardening can completely eliminate wetland vegetation, which results in degradation of fish habitat. It appears that when a wetland becomes affected by human development, the fish community changes to that typical of a warmer, and more southerly wetland. This finding may help researchers anticipate the likely effects of regional climate change on the fish communities of Great Lakes coastal wetlands.
- Mechanical alteration takes a diversity of forms, including diking, ditching, dredging, filling, and shoreline hardening. With all of these alterations, non-native species may be introduced by construction equipment or in introduced sediments. Changes in shoreline gradients and sediment conditions are often adequate to allow non-native species to become established.
- Land Cover – Agriculture degrades wetlands in several ways, including nutrient enrichment from fertilizers, increased sediment deposition from erosion, increased rapid runoff from drainage ditches, introduction of non-native species (reed canary grass), destruction of inland wet meadow habitats by plowing and diking, and addition of herbicides. In the southern lakes, Saginaw Bay, and Green Bay, agricultural sediments have resulted in highly turbid waters which support few or no submerged plants.
- Physical modifications to the shoreline have disrupted coastal and nearshore processes, flow and littoral circulatory patterns, altered or eliminated connectivity to coastal wetlands/dunes, and have altered nearshore and coastal habitat structure. Urban development degrades wetlands by hardening shoreline, filling wetland, adding a broad diversity of chemical pollutants, increasing stream runoff, adding sediments, and increased nutrient loading from sewage treatment plants. In most urban settings, almost complete wetland loss has occurred along the shoreline. Thoma (1999) and Johnson et al. (2006) were unable to find coastal wetlands on the U.S. side of Lake Erie that experienced minimal anthropogenic disturbances. According to Seilheimer and Chow-Fraser (2006; 2007), there has been accelerated loss of wetland fish habitat in Lake Ontario, Lake Erie and Lake Michigan near urban areas and agriculture.
- Impact of Aquatic Invasive Species – Non-native species are introduced in many ways. Some were purposefully introduced as agricultural crops or ornamentals, later colonizing in native landscapes. Others came in as weeds in agricultural seed. Increased sediment and nutrient enrichment allow many of the worst aquatic weeds to out-compete native species. Many nuisance non-native species are either prolific seed producers or reproduce from fragments of root or rhizome. Non-native animals have also been responsible for increased degradation of coastal wetlands. Common and grass carp reproductive and feeding behavior results in loss of submerged vegetation in shallow marsh waters.

- Precipitation Amounts in the Great Lakes Basin/Tributary Flashiness – change in atmospheric temperature will potentially affect the number of extreme storms in the Great Lakes region which will, in turn, affect coastal wetlands; extreme storms would likely affect riverine systems mostly through increased sediment movements, particularly in flashy modified watersheds.
- Water Levels – water level change has strong influences on Great Lakes habitat and biological communities associated with coastal wetlands. Lake levels have a major influence on undiked coastal wetlands and are basic to any analysis of wetland change trends; water levels influence the vegetation structure in wetlands which in turn structure the habitat available for fish. Altered water levels will also influence the spread of invasive plant species, such as *Phragmites*, that can take over native vegetation stands and alter habitat available to fish and other organisms. This sub-indicator links directly to the other sub-indicators in the Habitats and Species category, particularly the other Coastal Wetlands-related sub-indicators. Individual IBIs, such as the fish IBI used in this assessment, are useful as independent indicators, but evaluation of this indicator in combination with the other coastal wetland indicators is key to an overall assessment of Great Lakes coastal wetland health. This is because the different sub-indicators function and indicate anthropogenic disturbance at different spatial and temporal scales and have varying resolution of detection. See greatlakeswetlands.org/Sampling-protocols for details on indicator metrics.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: www.greatlakeswetlands.org		

Data Limitations

This sub-indicator can only be used where there is sufficient water depth to use fyke nets. Annually approximately 200 sites are sampled by the GLCWMP, but on average only around 100 are sampled for fish. There are a number of sites sampled only by bird and anuran crews, because these crews can complete their site sampling more quickly and thus have the capacity to sample more sites than do the fish, macroinvertebrate, and vegetation crews. Therefore, ecosystem health cannot be assessed for all sampled wetlands using fish IBIs. Furthermore, fish IBIs have been developed for a subset of monodominant vegetation zones in coastal wetlands, including *Typha*, *Schoenoplectus*, submerged aquatic vegetation (SAV), and water lily. Fish IBI scores can only be calculated for wetlands where fish were captured in these vegetation stands. Additionally, fyke nets may not be equally effective for capturing all species and size classes of wetland fish (Clement et al. 2014).

Acknowledgments

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Table 1. The number of distinct sites sampled, number of sampling events (includes repeat sampling of sites), total fish species and non-native species found in Great Lakes coastal wetlands and summary statistics by country for sites sampled from 2011 through 2019. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP), Uzarski et al. 2016.

			Total Fish Taxa				*Non-native taxa*			
Country	*# Sites*	*#Samples*	*Mean*	*Max*	*Min*	*St. Dev*	*Mean*	*Max*	*Min*	*St. Dev*
Canada	178	276	8.9	20	1	3.6	0.7	4	0	0.8
U.S.	308	484	12.0	26	2	4.7	0.9	6	0	1.0

Table 2. The number of distinct sites sampled, number of sampling events, number of sampling events related to IBI scores, total fish species and non-native species found in Great Lakes coastal wetlands and summary statistics by lake for sites sampled from 2011 through 2019. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP), Uzarski et al. 2016.

			Total Fish Taxa				*Non-native Taxa*		
Lake	*# Sites*	*# Samples*	*# Samples w/IBI*	*Mean*	*Max*	*Min*	*Mean*	*Max*	*Min*
Erie	61	103	83	10	23	2	1.4	5	0
Huron	168	255	206	11	25	1	0.5	4	0
Michigan	76	122	109	12	26	3	1	5	0
Ontario	127	194	183	10	20	2	0.8	4	0
Superior	54	86	68	12	24	3	0.6	6	0

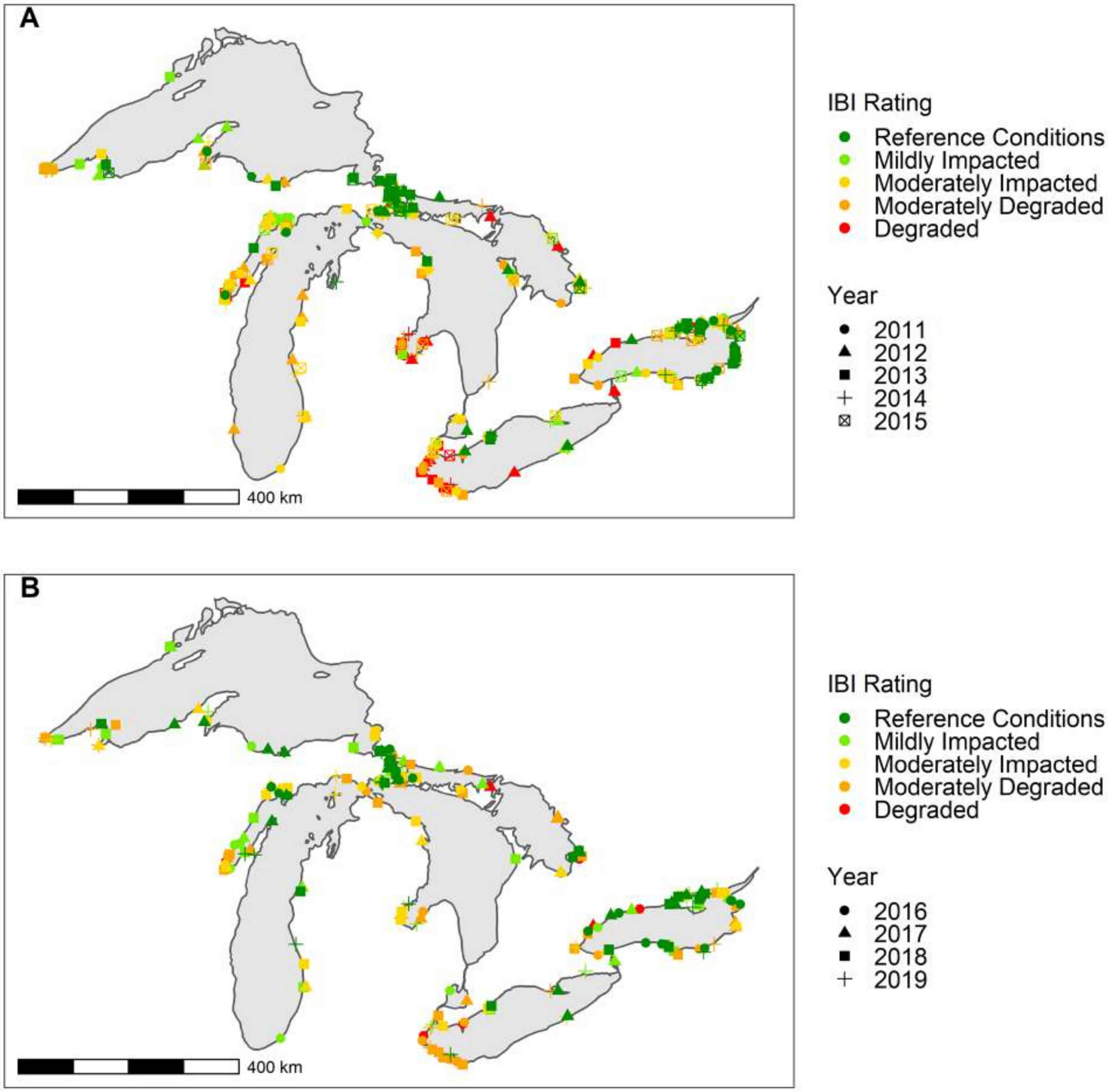


Figure 1. The condition of coastal wetland fish communities from 2011 to 2015 (A) compared to the condition from 2016 to 2019 (B) across the Great Lakes basin. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

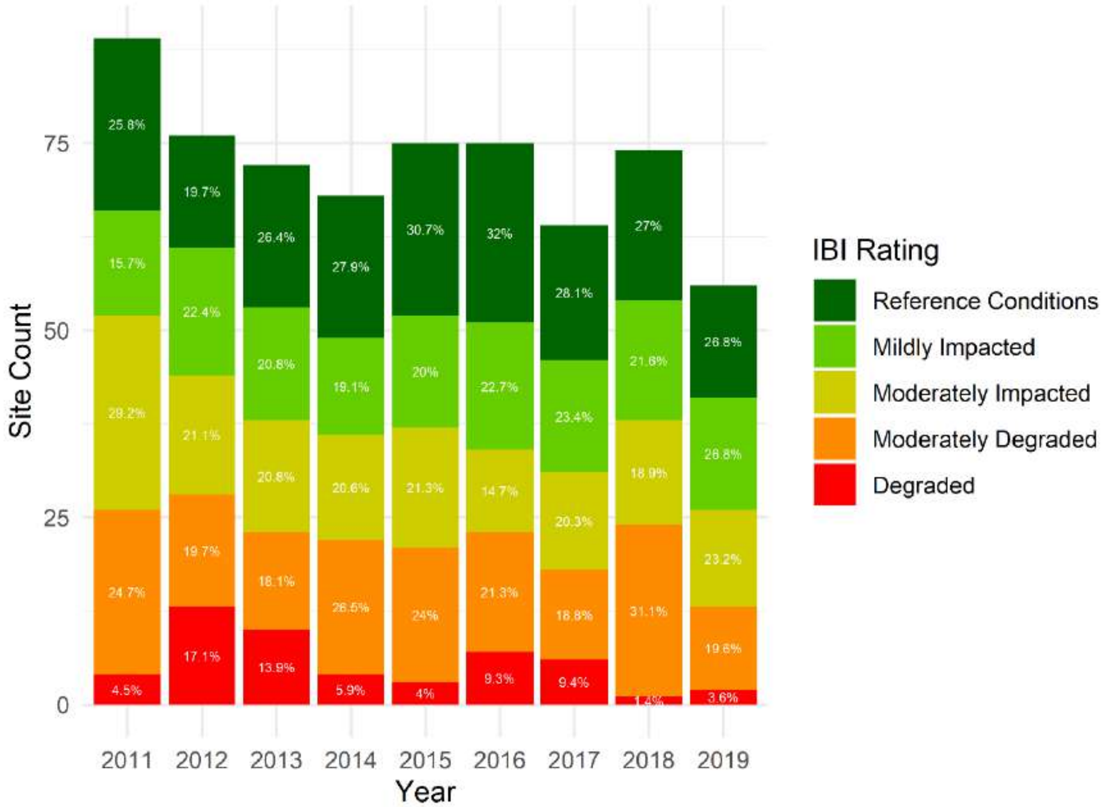
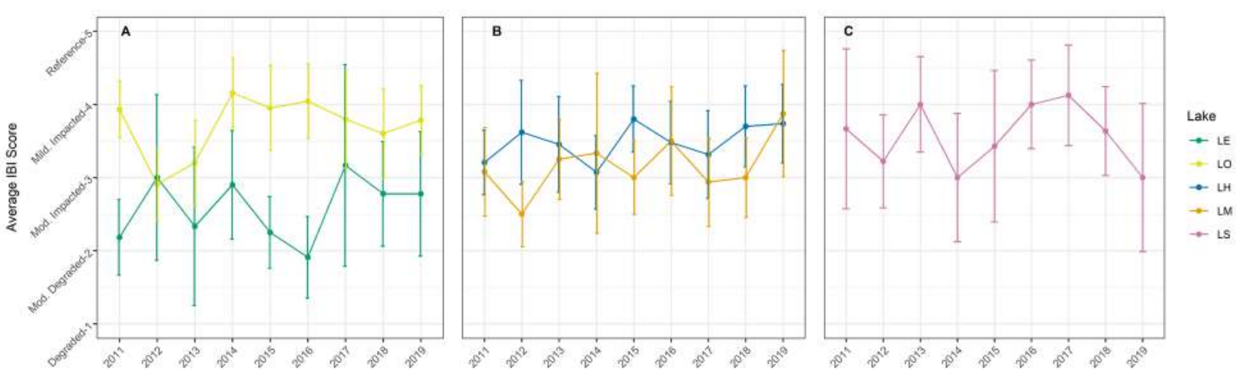


Figure 2. The count of all sites that fall into each IBI category based on the coastal wetland fish community each year between 2011 and 2019, n=649. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)



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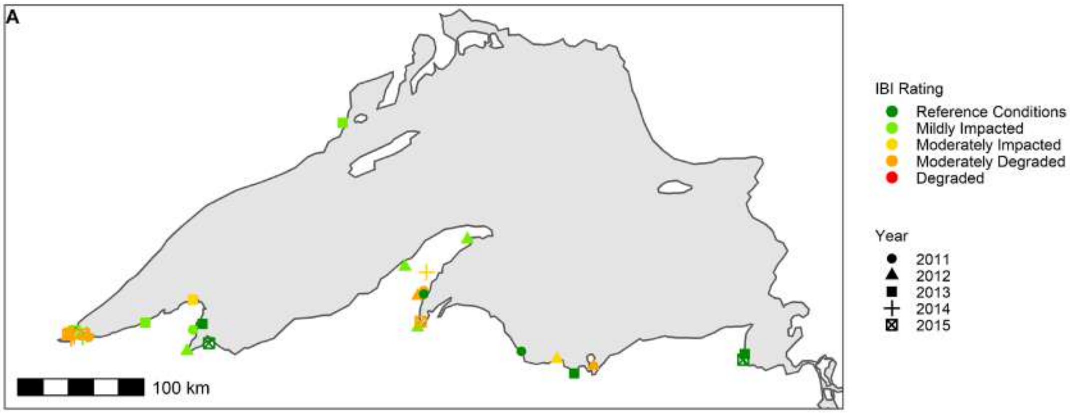
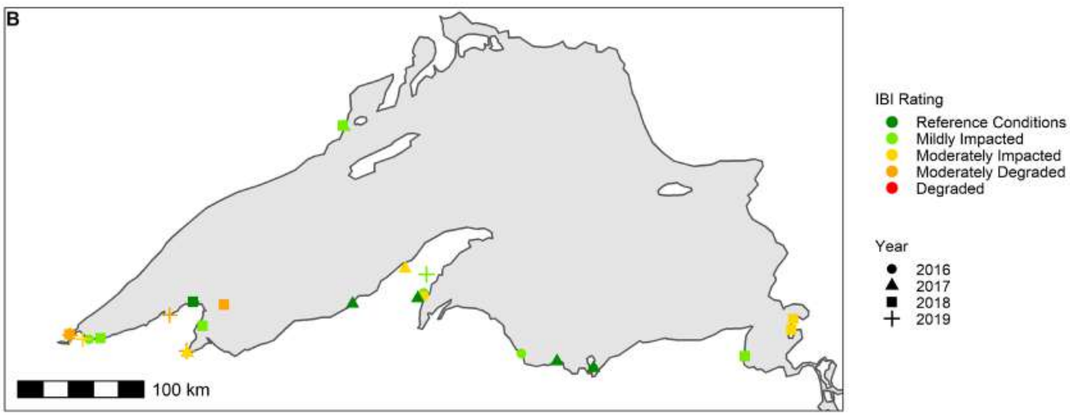
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)





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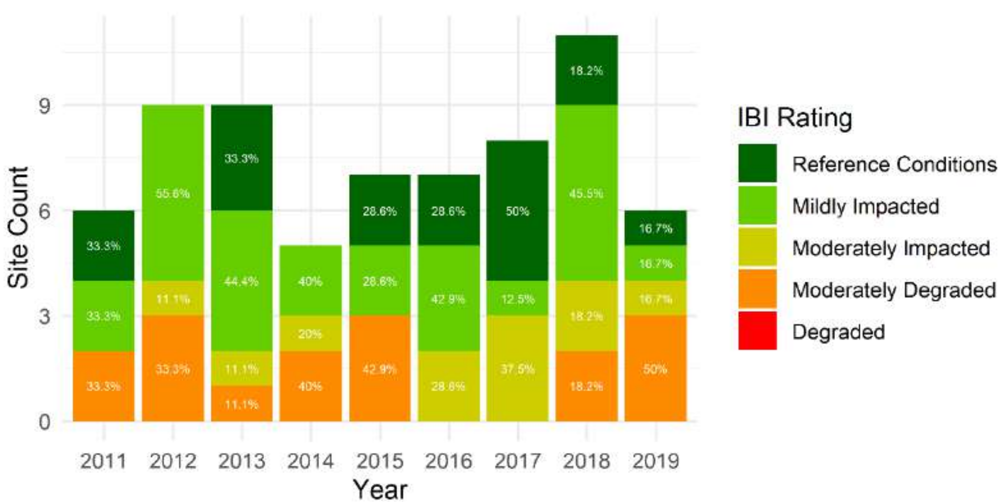

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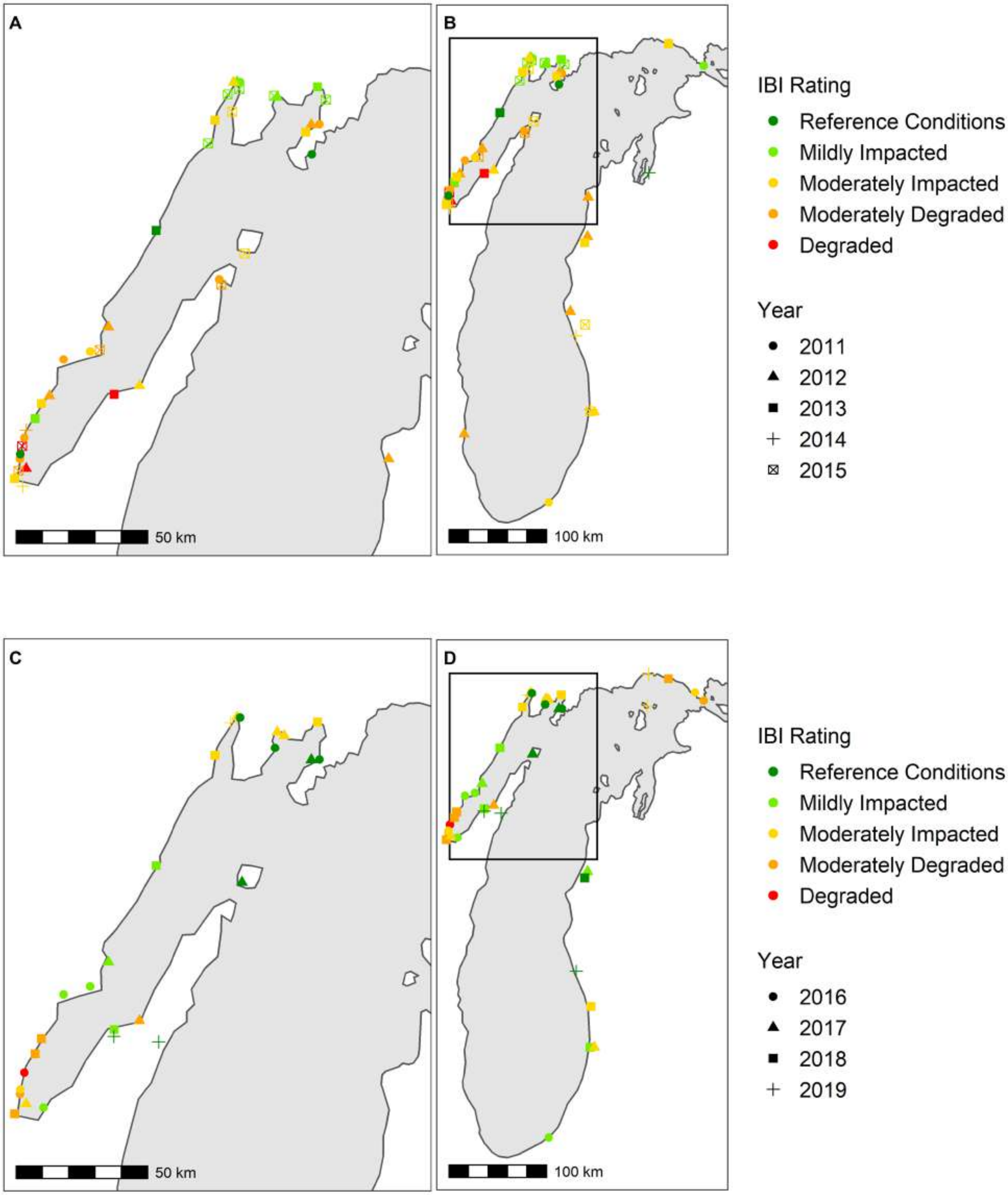
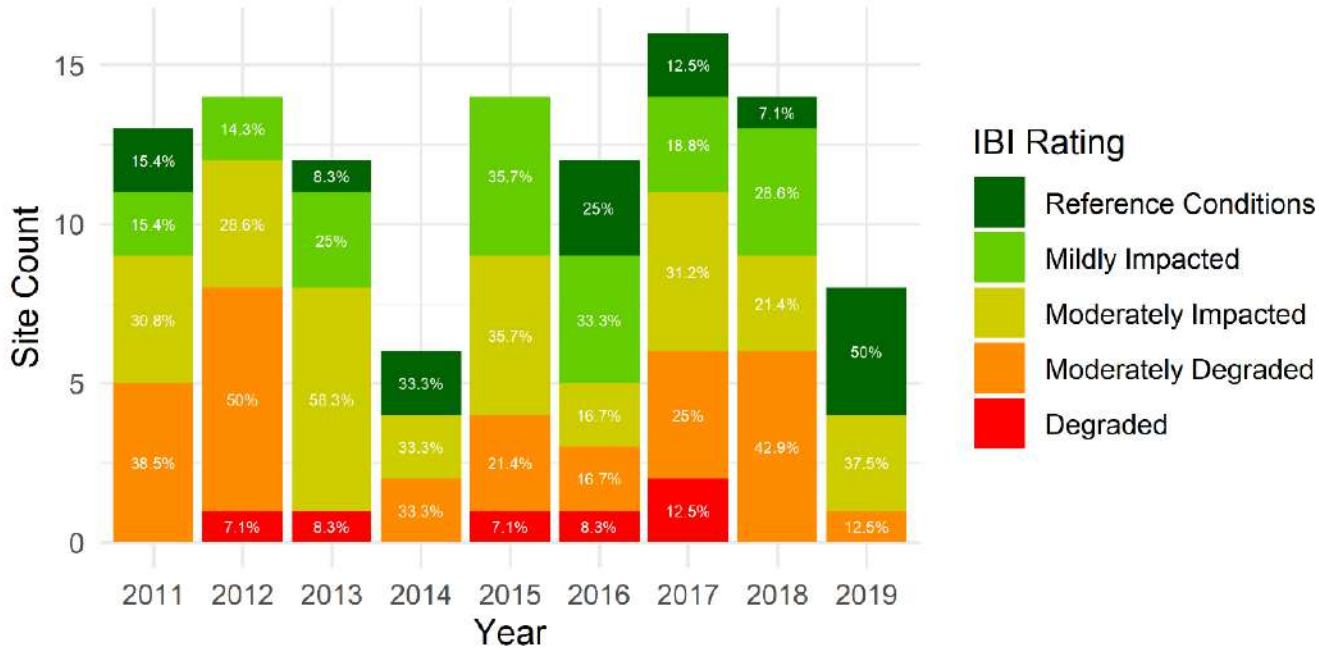


Figure 6. The condition of coastal wetland fish communities from 2011 to 2015 (A and B) compared to the condition from 2016 to 2019 (C and D) across Lake Michigan. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).



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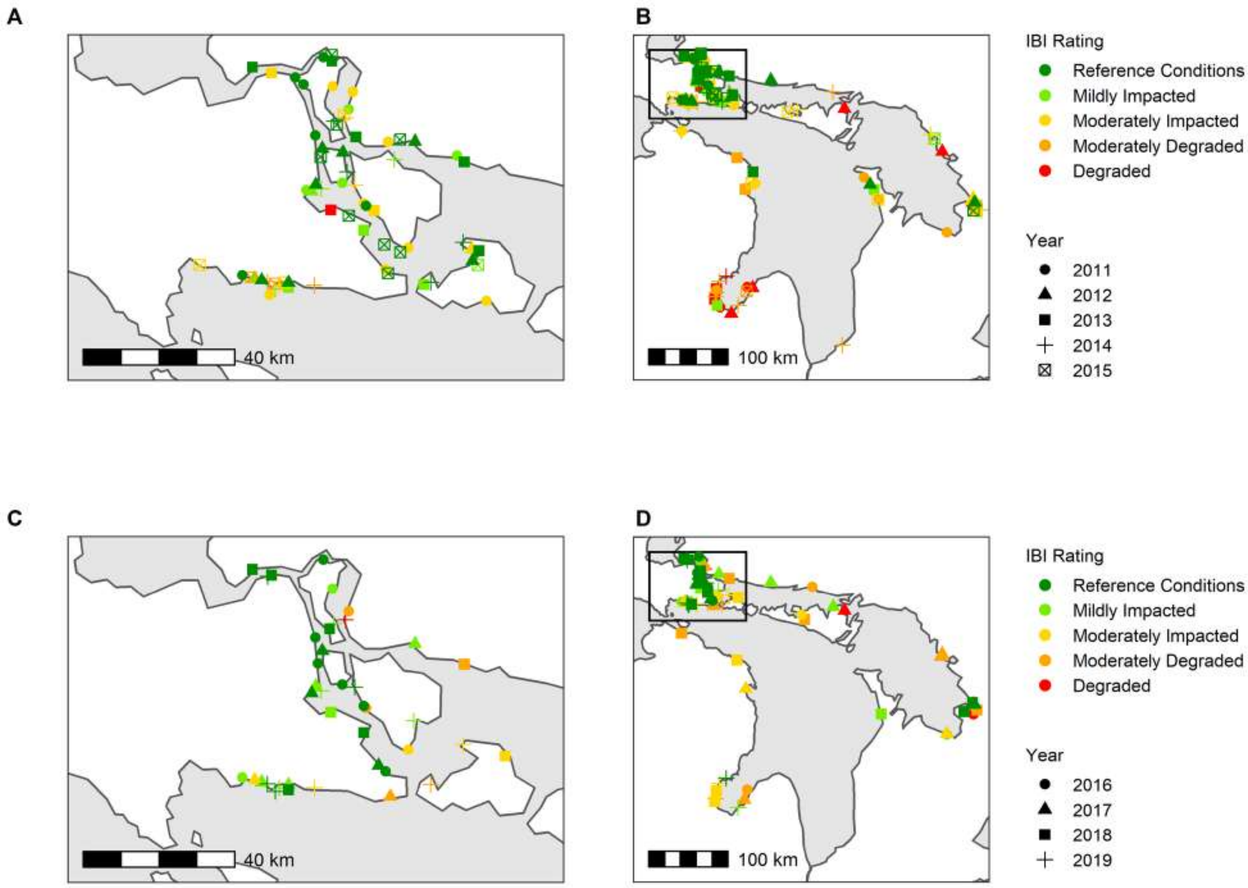

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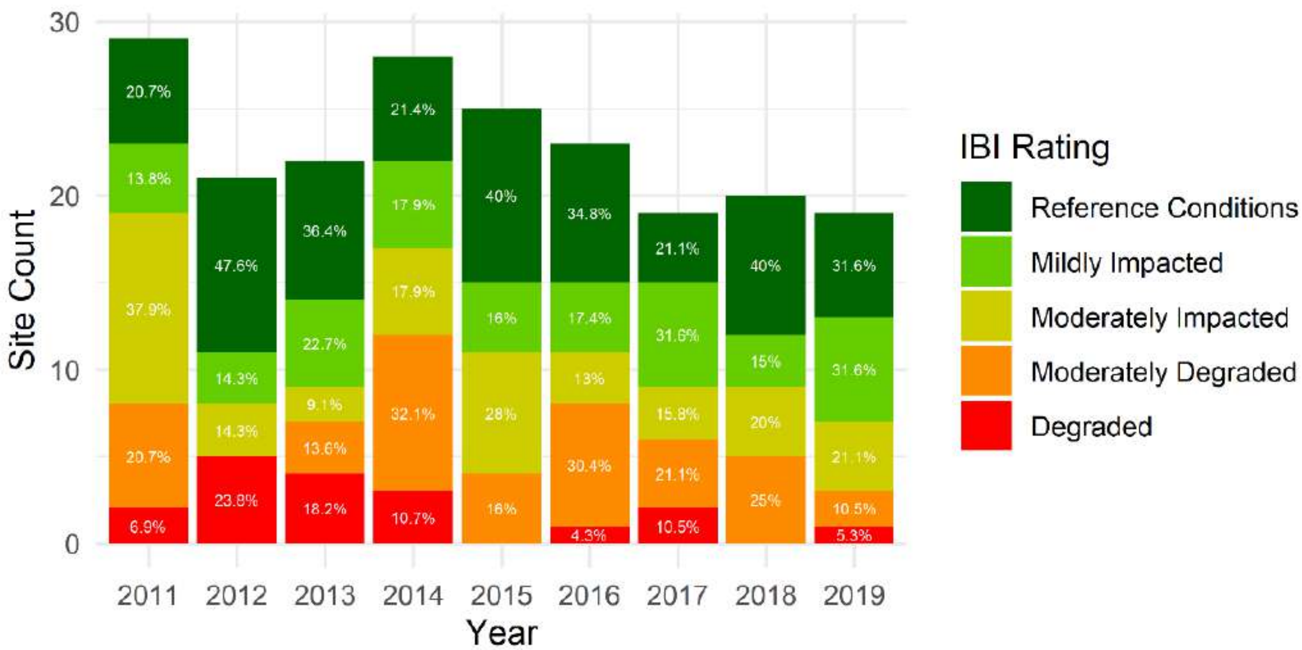

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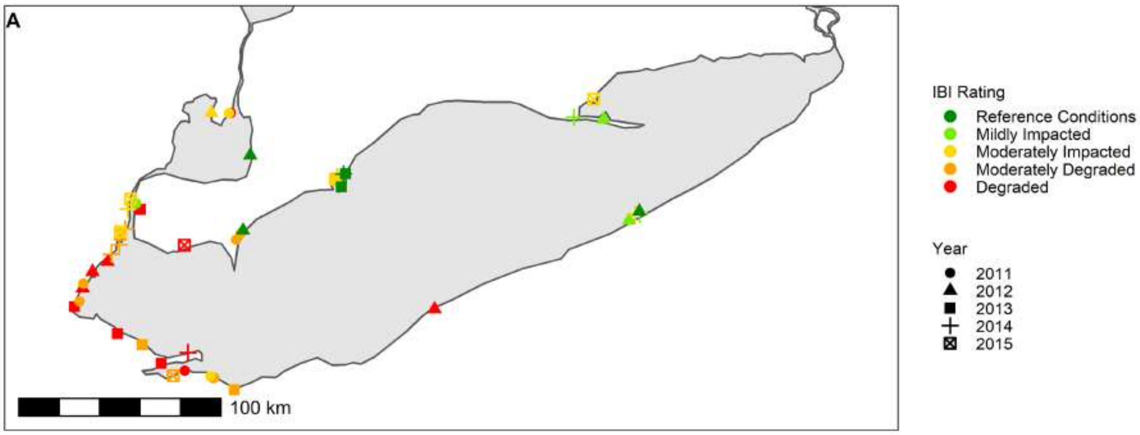

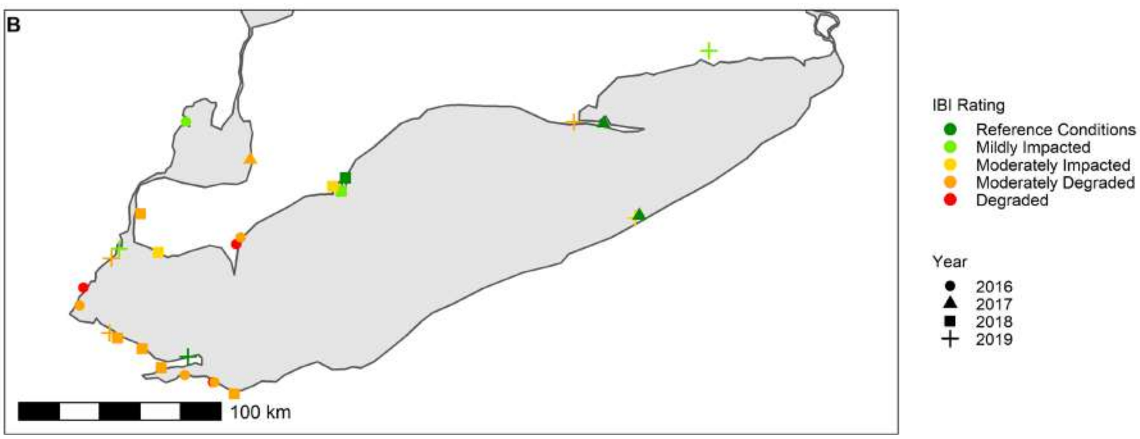

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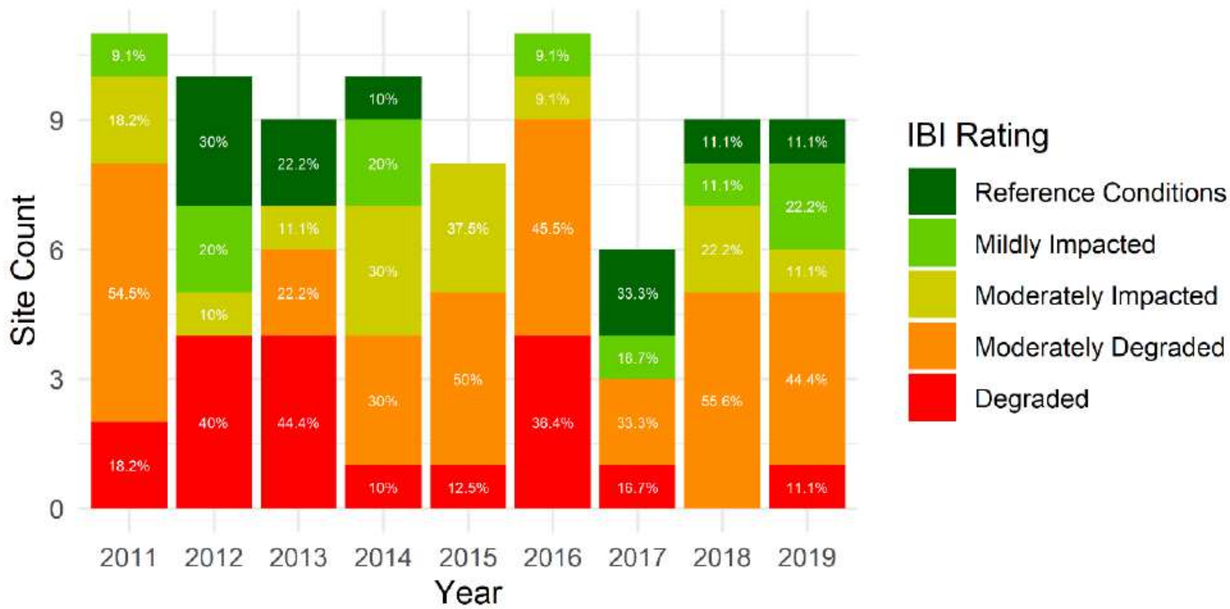

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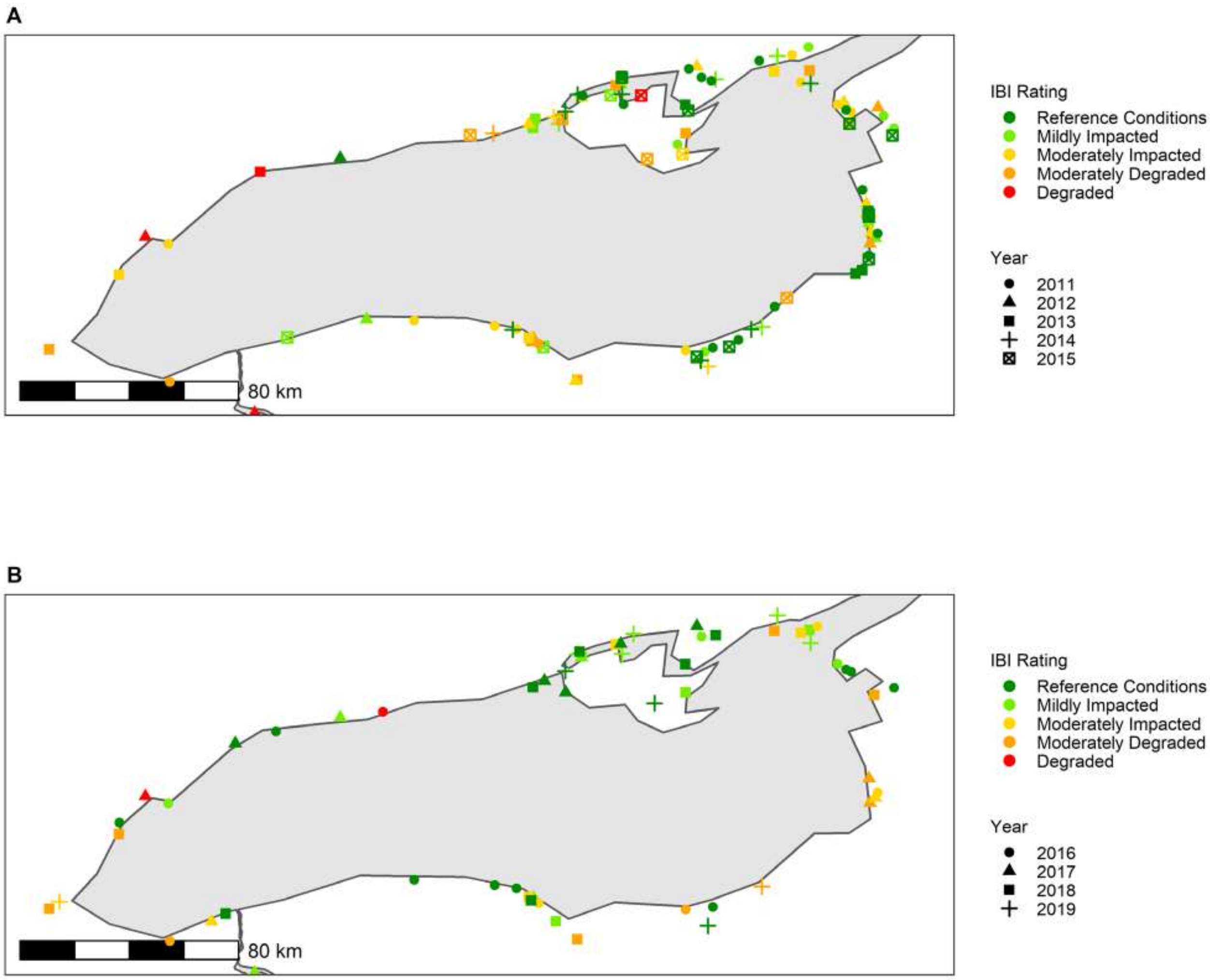

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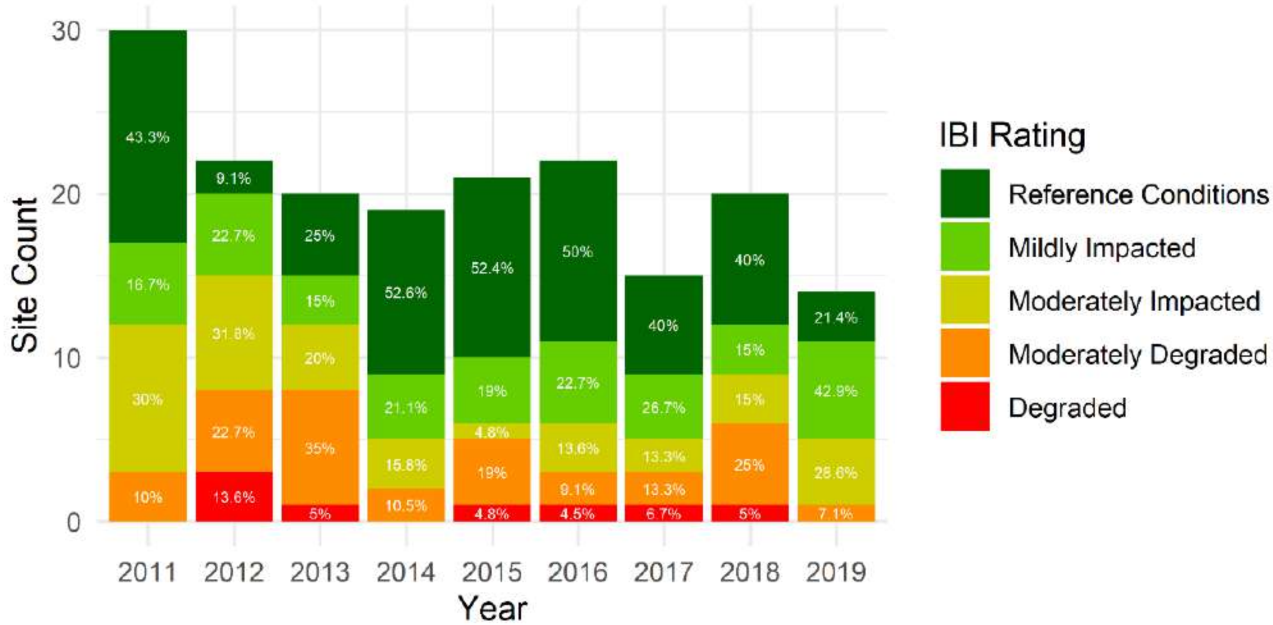

Figure 13. The count of all Lake Ontario sites that fall into each IBI category based on the coastal wetland fish community each year between 2011 and 2019, n=183. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP).

Sub-Indicator: Coastal Wetland Amphibians

Overall Assessment

Status: Fair

Trends:

10-Year Trend (2010-2020): Improving

Long-term Trend (1995-2020): Improving

Rationale: Median of a modified Index of Ecological Condition (IEC), an objective biotic indicator summarizing standardized observations of wetland breeding anurans (i.e., frogs and toads; Order Anura) in coastal wetlands (see Analysis section below for details), was 5.8 (out of 10) based on data from 2019-20, while the median IEC ranged from 3.3 to 6.5 from 1995-2020. IECs showed a significant increase of 4.2%/year over the past 10 years (2010-2020). Note that a separate determination of status and trend was made for the Overall Assessment based on data from all of the individual lake basins, rather than averaging across the lake-by-lake assessments given below. IECs at coastal wetlands were notably lower throughout the western portion of Lake Ontario and Lake Erie, the southern portion of Lake Michigan, and the southern portion of Green Bay, Lake Michigan, where the overall footprint of human development is substantial compared with most of the rest of the lakes and where, generally, a mix of low and high IECs occurred. One exception being that low IECs occurred along parts of the Lake Superior shoreline, where the human footprint is relatively low, but anuran occurrence is more limited, probably due to natural environmental factors.

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: Median IEC for coastal wetlands based on anuran communities was 5.3 using data from 2019-2020 and ranged from 2.1 to 6.9 from 1995-2020. Although the landscape in the coastal zone of Lake Superior is generally non-agricultural and minimally-developed, the coastal wetlands of Lake Superior (with a few notable exceptions) are relatively small in area, tend to have lower productivity (bogs), and are ecologically quite different compared to the rest of the lakes. Being farthest north on the relatively unproductive Canadian Shield, they also experience a cooler climate compared to most of the more southern lakes, which may limit anuran distributions. These factors may account at least partially for some of the modest scores in coastal wetlands in Lake Superior compared with those from other lakes.

Lake Michigan

Status: Fair

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: Median IEC for coastal wetlands based on anuran communities was 6.1 based on data from 2019-2020 and ranged from 1.2 to 6.5 from 1995-2020. Some of the highest quality coastal wetlands with respect to anurans occur in Lake Michigan, even though development and agricultural stressors are fairly strong in parts of the coastal zones of this lake. The lower IECs at coastal wetlands in the southern portion of the lake and in the southern portion of Green Bay compared to the rest of the lake is likely due to strong development-related stressors.

Lake Huron (including St. Marys River)

Status: Good

10-Year Trend (2010-2020): Unchanging

Long-term Trend: Undetermined

Rationale: Median IEC for coastal wetlands based on anuran communities was 6.5 based on data from 2019-2020 and ranged from 3.2 to 7.2 from 1995-2020. IECs did not significantly increase or decrease over the most recent 10 years (2010-2020). Some of the highest quality coastal wetlands with respect to anurans occur in Lake Huron, even though development and agricultural stressors are fairly strong in parts of the coastal zones of this lake.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Fair

10-Year Trend (2010-2020): Unchanging

Long-term Trend (1995-2020): Unchanging

Rationale: Median IEC for coastal wetlands based on anuran communities was 4.8 based on data from 2019-2020 and ranged from 3.5 to 5.5 from 1995-2020. IECs did not significantly increase or decrease over the most recent 10 years (2010-2020). IECs at coastal wetlands were notably lower throughout the western portion of the lake where the overall footprint of human development is substantial. Out of all of the lake basins, Lake Erie had the lowest median IEC in coastal wetlands, slightly lower than Lake Superior and Lake Ontario.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Fair

10-Year Trend (2010-2020): Improving

Long-term Trend (1995-2020): Improving

Rationale: Median IEC for coastal wetlands based on anuran communities was 6.2 based on data from 2019-2020 and ranged from 2.6 to 7.2 from 1995-2020. IECs significantly increased by 5.6%/year over the past 10 years (2010-2020). IECs at coastal wetlands were notably lower throughout the western portion of the lake where the overall footprint of human development is substantial.

Status Assessment Definitions

Standardized indices (IEC values/scores) range between 0 (Poor condition or most degraded) and 10 (Good condition or least degraded). Status will be determined based on median IEC in coastal wetlands in the most recent year (for 2022 reporting, status is based on data from 2019-2020 because sample sizes were relatively low in 2020) compared to percentiles based on data from all coastal wetlands in all years since and including 2011. Coverage of coastal wetlands throughout the Great Lakes has been best since 2011 with the implementation of the Great Lakes Coastal Wetland Monitoring Program (CWMP). Note that the values defining Poor, Fair, and Good below have changed since the previous report (Tozer et al. 2019) because of: 1) improved sensitivity of the metrics used to calculate the IEC since the last report for reflecting the true condition of a wetland and 2) more years of data used to calculate the values since the last report.

Good: Most or all ecosystem components are in acceptable condition; IEC > 66th percentile (for 2022 reporting, IEC > 6.4).

Fair: Some ecosystem components are in acceptable condition; 33rd percentile ≤ IEC ≤ 66th percentile (for 2022 reporting, 4.4 ≤ IEC ≤ 6.4).

Poor: Very few or no ecosystem components are in acceptable condition; IEC < 33rd percentile (for 2022 reporting, IEC < 4.4).

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Both regional and local wetland breeding anuran populations naturally fluctuate over time; therefore, several years of monitoring data with suitable geographic coverage throughout targeted areas will be required to detect all but the most dramatic trends. Interpretation of this sub-indicator may be improved if coupled with patterns observed in other wetland sub-indicators.

The terms Improving, Unchanging, and Deteriorating were applied based on geometric mean rates of change (%/year) of a modified version of a standardized wetland breeding anuran-based index of ecosystem health, the Index of Ecological Condition or IEC (Howe et al. 2007a, 2007b), using equation 4 in Smith et al. (2014; see Analysis section below for details). The statistical significance of trends was assessed via parametric bootstrapping in R (R Core Team 2020) with package “boot” (Canty and Ripley 2013) and 1,000 bootstrap replicates. Bootstrapping in this manner was necessary to account for the varying precision of the beginning annual estimate and the ending annual estimate used to calculate each trend. For 10-year trends, the start year was 2010 and the end year was 2020, whereas for long-term trends, the start year was 1995 and the end year was 2020. The year 2010 was chosen as the start year for 10-year trends because during 2010-2020 there are 10 possible transitions between years, which was taken to be most representative of the trend during the most recent decade (IUCN Standards and Petitions Committee 2019). Trend estimates with 95% confidence intervals that did not overlap zero were considered statistically significant.

Improving: A statistically significant increase in IEC over the most recent 10 years.

Unchanging: No statistically significant increase or decrease in IEC over the most recent 10 years.

Deteriorating: A statistically significant decrease in IEC over the most recent 10 years.

Undetermined: Data are not available or are insufficient to report on a trend.

Endpoints and/or Targets

Endpoints will be established based on existing data. Alternative endpoints may be used based on historical records or expert opinion in certain instances as determined and justified by the authors. Endpoints for this sub-indicator were left undefined for the 2022 report due to uncertainty around the best way to calculate meaningful endpoints. Ongoing work by the authors will strive to develop useful endpoints.

Sub-Indicator Purpose

The purpose is to assess the status and trends of Great Lakes coastal wetland ecosystem health by directly measuring the composition and occurrence of wetland breeding anurans. The sub-indicator thereby infers the condition of coastal wetland habitat as it relates to the health of this ecologically and culturally important component of wetland communities.

Ecosystem Objective

Coastal wetlands provide critical habitat for various life stages of many wildlife species including anurans, which are known to be sensitive to environmental contamination and habitat degradation, and which provide an important element of food webs in the Great Lakes coastal zone (Knutson et al. 1999; Price et al. 2004, 2007; Gnass Giese et al. 2018). Conservation of remaining coastal wetlands and restoration of previously degraded or destroyed wetlands are vital components for restoring the Great Lakes ecosystem, and this sub-indicator can be used to report progress toward such an objective.

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should “support healthy and productive wetlands and other habitats to sustain resilient populations of native species.”

Measure

Background —Wetland breeding anurans are influenced by the physical, chemical, and biological components of wetlands and surrounding landscapes. For example, the occurrence and/or reproductive success of multiple species in the Great Lakes basin decline as (1) wetland size decreases; (2) wetland habitat and natural cover in the surrounding landscape decrease or degrade in quality; and (3) pollution from pesticide, herbicide, and sediment runoff increases (Hecnar 1995; Hecnar and M’Closkey 1996, 1998; Bishop et al. 1999; Crosbie and Chow Fraser 1999; Kolozsvary and Swihart 1999; Houlahan and Findlay 2003; Price et al. 2004; Brazner et al. 2007a, 2007b; Gagné and Fahrig 2007; Eigenbrod et al. 2008a, 2008b). Thus, the occurrence or abundance of sensitive wetland breeding anurans can be a valuable indicator of the health of wetlands and the surrounding landscape.

Data—Several initiatives monitor Great Lakes wetland breeding anurans. One of the longest running is the Great Lakes Marsh Monitoring Program (GLMMP) coordinated by Birds Canada, which started in 1995 and has operated every year since then at coastal and inland wetlands throughout much of the Great Lakes basin (Tozer 2013, 2016, 2020). Some of the previous reports for this sub-indicator are based solely on data from this ongoing broad scale program (e.g., Tozer 2014). From 2001 to 2005, the University of Minnesota Duluth’s Natural Resource Research Institute (NRRI) led an ambitious multi-institutional Great Lakes Environmental Indicator Project (GLEI) aimed at assessing the overall biotic health of coastal wetlands in the U.S. portion of the Great Lakes (Howe et al. 2007a, 2007b; Hanowski et al. 2007a, 2007b). More recently, the CWMP, which is a partnership between USEPA and Central Michigan University, was initiated in 2011 and currently is scheduled to operate until at least 2025 throughout both the U.S. and Canadian Great Lakes coastal zones (Uzarski et al. 2017). These projects/programs have somewhat different study designs, but rely on standardized, fixed duration point counts that can be adjusted to maximize cross-project compatibility. To garner large numbers of trained volunteer participants to achieve large sample sizes at relatively low cost, the GLMMP allows participants to select sample wetlands and sample points within them—a justifiable approach if one assumes that the sample locations are representative of wetlands across a region of interest. By contrast, GLEI and CWMP select sample wetlands via stratified random sampling and survey sample points within them via paid professional staff. Nonetheless, all of the projects/programs target wetlands dominated by non-woody emergent plants, such as cattails (*Typha* spp.) and sedges (e.g., *Carex* spp.), with sample points located within wetlands. In this report, the datasets listed above were brought together to generate a comprehensive analysis of the status and trend of Great Lakes coastal wetland breeding anurans and associated wetland health.

Surveys—Anurans were sampled to an unlimited distance from a point (hereafter "sample point") or up to 6 points located near the upland/wetland interface (shoreline) of a wetland depending on its size. Each sample point was surveyed for 3 minutes on three visits separated by at least 10 or 15 days during the main anuran breeding season, typically between late March and early July. Surveys occurred at night starting at least 0.5 hr after local sunset to 4.5 hr after local sunset and only under weather conditions that were favourable for detecting all species present (no persistent or heavy precipitation; wind: Beaufort 0-3, 0-19 km/hr). The first survey in the season was conducted when night-time air temperature had reached > ~5°C, the second when > ~10°C had been reached, and the third when > ~17°C had been reached. With few exceptions, only shoreline locations were sampled due to night-time, over-water safety issues. The survey protocols of each of the projects were similar to the North American Amphibian Monitoring Program protocol (Weir et al. 2009, 2014).

Analysis—Numerous methods are available for analyzing Great Lakes coastal wetland breeding anuran data. Some previous analyses for this report were based on the separate status and trend of the occurrence of 8 wetland breeding anuran species (e.g., Tozer 2014). Alternative approaches include various indices of wetland health, which combine data from suites of species (e.g., Chin et al. 2014). The latter approach is likely more objective and more practical for the purposes of State of the Great Lakes because it provides a single comprehensive metric that represents the collective responses of breeding anuran species to wetland condition. Multi-species metrics, like the widely used index of biotic integrity for fishes (Karr and Chu 1999) and the mean coefficient of conservatism for plants (Taft et al. 1997), are robust when they are tested against known stressor gradients and include enough species to allow calculations even if some species are absent due to extraneous factors. For example, a wide-ranging resident species might go undetected because, by chance, all local individuals of that species happen to be absent or inactive within the survey plot during the sampling period. Similarly, a high quality wetland might be missing a species because of factors unrelated to habitat like a regional epidemic that affects individuals regardless of wetland condition.

Developing a robust, multi-species ecological indicator of Great Lakes coastal wetlands based on anurans has been challenging for several reasons. First, only 12 species of anurans are found in the Great Lakes basin, and only 8 species are widely encountered (Table 1). When one or only a few key species are present at a site, likelihood metrics like the Index of Ecological Condition (IEC; Howe et al. 2007a, 2007b, Gnass Giese et al. 2015, Jung et al. 2020, and Howe et al. 2021) are often unstable and may provide misleading results. Second, large numbers of frogs and toads are often impossible to count during night time auditory surveys, so analyses are limited to presence/absence data or highly subjective estimates of abundance. Finally, all anuran species regularly found in Great Lakes coastal wetlands are native, and all are at least somewhat sensitive to habitat degradation. Some species clearly are more vulnerable to human disturbance than others, but presence of any anuran species generally reflects positively on the quality of a wetland.

We therefore developed a metric that is correlated with species richness (the number of anuran species), but "weights" species according to 1) their sensitivity to wetland disturbance and 2) their likelihood of occurrence and detection in the highest quality wetlands. Species weights were derived from parameters of a stressor response curve (biotic response [BR] function) representing each of the targeted anuran species, like the approach underlying the IEC (Howe et al. 2007a, 2007b, Gnass Giese et al. 2015). Our BR functions were based on a multivariate gradient representing wetland area and the "human footprint" impacts on the wetland, described in detail in the two paragraphs immediately below. These BR functions resemble the pre-determined coefficients of conservatism often used for plant species indicator metrics (Bourdaghs et al. 2006) except that, in this case, the BR functions represent an empirically derived functional response rather than a single number determined by experts.

To generate BR functions, we assembled quantitative information for seven minimally correlated ($-0.30 \leq r \leq 0.31$) environmental stressors at 2,236 single-year CWMP point samples (i.e., the combined 1st, 2nd, and 3rd point counts during a given year at a given point) located within 616 coastal wetlands (Elliott 2019): 1) EmWetArea, the area of emergent, non-woody vegetation within 500 m of the point + ½ the emergent wetland area between 500-1,000 m; 2) %Dev2km, the percent cover of developed land (industrial buildings, residential sites, golf courses, etc.) within 2 km of the sample point (excluding open water); 3) %Crop2km, the percent cover of agricultural cropland within 2 km of the sample point (excluding open water); 4) Roads2km, total meters of primary and secondary roads within 2 km of the sample point; 5) %DevWatershed, the percent developed land within the watershed draining into the wetland; 6) %CropWatershed, the percent cropland within the watershed draining into the wetland; and 7) PopWatershed, total human population within the watershed. We note that the sizes of the buffers were based on the range of distances found to most influence anuran occurrence during previous research in the Great Lakes (Price et al. 2004). The watershed variables were calculated by Brazner et al. (2007a,b) as part of the GLEI project (Danz et al. 2005, 2007). We note that the two agricultural land cover variables (%Crop2km and %CropWatershed, $r = -0.01$) and the two land development variables (%Dev2km and %DevWatershed, $r = 0.02$) were largely independent. We log-transformed variables when needed to improve normality and then conducted a Principal Components Analysis (PCA; McCune and Mefford 1999) to reduce the dimensionality of the multivariate dataset. The first 4 PCA axes explained 87% of the variation. Scores for these 4 axes were scaled from worst (most disturbed) to best (least disturbed) condition and summed to produce a raw index of environmental condition that approximates the "human footprint," which was then converted to C_{env}, an environmental condition score on a 0 to 10 scale, where 0 represents the most disturbed and 10 the least disturbed.

Each of the 2,236 single-year CWMP point samples were placed within bins of 15 points each (or 16 in the last case) with similar measures of environmental stress. In other words, the 15 sites with the worst "human footprint" scores were placed in the first bin, then the 15 sites with the next best "human footprint" scores were placed in the second bin, and so on, until the last of the 149 bins was reached, which contained the 16 sites with the very best "human footprint" scores. This created a new gradient of 149 bins, characterized by the mean "human footprint" score for the sites within each bin, ranging from most impacted (0) to least impacted (10). For each of eight anuran species or groups of anuran species, a BR function was estimated from the probabilities of occurrence in each of the 149 bins (Figure 1a). The probability values were defined as the proportion of the 15 points where the species was detected during at least one survey at a sample point during a sample year. The environmental condition score for each bin was the mean value for the 15 points. The probabilities of occurrence for each species or species group across the binned environmental stress ("human footprint") gradient can be described by a three-parameter function (μ = mean; σ = standard deviation; and h = scale factor) depicting a normal bell-shaped curve within the range of 0-10 (e.g., Gnass Giese et al. 2015). We used the best-fit parameters of the normal bell-shaped curve (following the familiar normal probability density function, R function dnorm). The best-fit parameters were estimated by an R script (R Core Team 2020) using the nlminb function developed by Bates and Sarkar (Gay 1990). These BR functions (Figure 1a) provided the basis for scoring new sites based on the presence/absence of 8 species/species groups (). Parameters of the best-fit mathematical function were estimated by computer iteration in R (R Core Team 2020) with package "iec" (https://github.com/ngwalton/iec).

Our measure of coastal wetland condition based on anuran communities is a modified IEC (Howe et al. 2007a, 2007b, Gnass Giese et al. 2015, Jung et al. 2020, and Howe et al. 2021), which for simplicity, is referred to as IEC in this report. Weights for each species were calculated as the product of two parameters: 1) the sensitivity of the species to wetland disturbance, calculated as the mean of the BR function $\underline{x}_i$ and 2) the probability of occurrence or detection, calculated as the value of the BR function when the reference (environmental) gradient = 10 ($x(10)_i$). The BR function mean ($\underline{x}_i$) reflects the sensitivity of the species to the reference gradient; the more sensitive the species, the larger will be the mean. We limited the range of this parameter from 0 (least sensitive) to 10 (most sensitive), although the shape of BR functions might reflect normal distribution functions whose means extend beyond these limits (Gnass Giese et al. 2015). The second parameter ($x(10)_i$) reflects the probability of encountering the species at the highest quality sites, influenced by the ubiquity (commonness) of the species and the probability of detecting individuals when the species is present. The IEC was calculated as the sum of the weights for all species present at the wetland during the three seasonal counts, scaled from 0 (when no species are present) to 10 (when all species are present):

$$IEC = \left[\sum_{i=1}^{present} \underline{x}_i \; x(10)_i \right] * 10/W_{max}$$

$\underline{x}_i$ = mean of the BR function for species i

$x(10)_i$ = value of the BR function at highest quality reference wetland (i.e., reference gradient = 10) for species i

W_{max} = sum of weights of all species possible in the sampling region:

$$\sum_{i=1}^{all\,species\,possible} \underline{x}_i \; x(10)_i$$

In all cases, IEC values range from 0 (most disturbed or lowest possible quality) to 10 (least disturbed or highest quality). See Figure 1 for a schematic representation of how our IEC was calculated.

IECs for each sample point during a given year were based on species detected during one or more of the field visits. Basing IECs on species' presence or absence (rather than abundance) is desirable because it minimizes the effects of differences in detectability. Next, the point-level IECs were averaged across all sample points within each wetland or wetland complex in each year, which standardizes wetlands containing differing numbers of sample points. Box-and-whisker plots of these wetland-level mean IECs for coastal wetlands in each basin and throughout the entire Great Lakes basin (hereafter "overall") are reported in each year. These wetland-level means form the basis for the status and trend assessments. IEC metrics were also calculated for inland wetlands for comparison to the Great Lakes coastal wetlands but are not used for any assessments of status or trends. In addition, box-and-whisker plots of wetland-level mean IECs for coastal wetlands in each basin and overall for the most recent years (2019-2020) are reported to illustrate current status. The most recent two years of data were used rather than the most recent year because sample sizes were relatively low in 2020. Density plots of wetland-level mean IECs for coastal and inland wetlands in each basin (but not inland for Lake Superior or Lake Michigan due to insufficient sample size) based on the most recent 5 years of data (2016-2020) are reported to illustrate variation in distributions. To further illustrate distributions, these same data for coastal wetlands are mapped throughout each of the basins. In the calculations for the density plots and the map, wetland-level mean IEC values were averaged across years for wetlands that were sampled in multiple years using the most recent 5 years of data (2016-2020) to boost sample sizes. To better illustrate general patterns on the map, the point-to-raster tool in ArcGIS (ESRI 2013) was used to calculate mean values within 3,000 m cells, and the focal statistics tool was used on the resulting means to calculate average values within a 2-cell circular window, which is referred to as "smoothed IEC." Values within 3,000 m cells and a 2-cell circular window were used because through trial and error these values produced maps with desired visual resolution.

Ecological Condition

Data coverage—The dataset available for scoring sites consisted of mean annual wetland-level IECs based on 58,602 point counts conducted at 5,087 sample points in 1,850 wetlands over 26 years from 1995-2020 throughout coastal and inland areas of the Great Lakes basin (Figure 2). The number of years that each wetland was surveyed varied from 1 to 26, with a mean of 3.7 ± 3.9 (SD). Spatial patterns among locations of sampled points were due mainly to natural variation in the distribution of Great Lakes coastal wetlands and differences in observer participation in the long running, broad scale GLMMP (Figure 2). The majority of the surveyed wetlands were coastal (*n* = 1,182; 64%) rather than inland (*n* = 668; 36%) because both the GLEI and CWMP projects focus entirely on coastal wetlands, whereas the GLMMP surveys both coastal and inland wetlands (Figure 2).

The number of coastal and inland wetlands surveyed per year (274 ± 123 [mean ± SD]) ranged from 113 (1995) to 450 (2015), with only 208 wetlands surveyed in 2020 due to complications from COVID-19. Substantially more wetlands were surveyed from 2011 and onwards due to the CWMP operating during those years (Figure 3). Annual coverage was also higher in Lake Erie and Lake Ontario compared to the upper Great Lakes mostly because GLMMP coverage is more extensive in the lower lakes, and annual coverage was higher at coastal compared to inland wetlands (Figure 3). The number of wetlands available for surveys also varied depending on location, as some sections of the Great Lakes shoreline naturally have no, or almost no, wetlands.

Overall—Median IEC in coastal wetlands in the Great Lakes overall ranged from 3.3 (2000) to 6.5 (2020) from 1995-2020, significantly increased by 4.2%/year (3.1, 5.3) [lower, upper 95% confidence limits] over the past 10 years (2010-2020; Table 2, Figure 4), and was 5.8 (out of 10) based on data from 2019-20 (Figure 5). Based on these patterns, the status of coastal wetland health based on anuran communities in the Great Lakes overall is Fair and the trend is Improving. By contrast, IECs at inland wetlands showed little change across years (Table 2; Figure 4, 7). IECs at coastal wetlands were notably lower throughout the western portion of Lake Ontario and Lake Erie, the southern portion of Lake Michigan, and the southern portion of Green Bay, Lake Michigan (Gnass Giese et al. 2018) where the human development footprint is substantial (Bourgeau-Chavez et al. 2015) compared with most of the rest of the lakes where, generally, a mix of low and high IECs occurred (Figure 6). One exception being that low IECs occurred along parts of the Lake Superior shoreline, where the human footprint is relatively low (Bourgeau-Chavez et al. 2015), although anuran occurrence is likely more limited in these locations due to natural environmental factors. The status and trend of coastal wetlands in the Great Lakes overall are the same as in the last report (Tozer et al. 2019).

Lake Superior—Median IEC in coastal wetlands ranged from 2.1 (1997) to 6.9 (2016) from 1995-2020 (Figure 4) and was 5.3 based on data from 2019-20 (Figure 5). For this report, it was concluded that there was not enough information to perform reliable statistical significance testing for trends. Based on these patterns, the status of coastal wetland health based on anuran communities in Lake Superior is Fair, and the trend is Undetermined. Similar patterns appeared to occur at inland wetlands in the Lake Superior watershed, although it was concluded that sample sizes were too low to be certain (Figures 4, 7). Although the landscape in the coastal zone of Lake

Superior is generally non-agricultural and minimally-developed (Bourgeau-Chavez et al. 2015), the coastal wetlands of Lake Superior (with a few notable exceptions) are relatively small in area, tend to have lower productivity (bogs), and are ecologically quite different compared to the rest of the lakes (Mayer et al. 2004). Being farthest north on the relatively unproductive Canadian Shield, they also experience a cooler climate compared to most of the more southern lakes (Crins et al. 2009), which may limit anuran distributions (Hecnar 2004). These factors may account at least partially for some of the modest scores in coastal wetlands in Lake Superior compared with those from other lakes.

Since the last report, the status of coastal wetlands in Lake Superior has changed from Good to Fair. The difference in status may be due to improved sensitivity of the metrics used to calculate the IEC since the last report for reflecting the true condition of a wetland or other factors (see discussion below for further explanation).

Lake Michigan—Median IEC in coastal wetlands ranged from 1.2 (2000) to 6.5 (2020) from 1995-2020 (Figure 4) and was 6.1 based on data from 2019-20 (Figure 5). For this report, we concluded that there was not enough information to perform reliable statistical significance testing for trends. Based on these patterns, the status of coastal wetland health based on anuran communities in Lake Michigan is Fair, and the trend is Undetermined. Somewhat contrasting patterns appeared to occur at inland wetlands in the Lake Michigan watershed, although it was concluded that sample sizes were too low to be certain (Figure 4, 7). Some of the highest quality coastal wetlands with respect to anurans occur in Lake Michigan, even though development and agricultural stressors are fairly strong in parts of the coastal zones of this lake (Figure 4-7; Allan et al. 2013, Bourgeau-Chavez et al. 2015). The lower IECs at coastal wetlands in the southern portion of the lake and in the southern portion of Green Bay compared to the rest of the lake are likely due to strong development-related stressors (Figure 6; Allan et al. 2013, Bourgeau-Chavez et al. 2015).

Since the last report, the status of coastal wetlands in Lake Michigan has changed from Good to Fair. The difference in status may be due to improved sensitivity of the metrics used to calculate the IEC since the last report for reflecting the true condition of a wetland or other factors (see discussion below for further explanation).

Lake Huron (including St. Marys River)—Median IEC in coastal wetlands ranged from 3.2 (2001) to 7.2 (2013) from 1995-2020, did not significantly increase or decrease over the past 10 years (Table 2, Figure 4), and was 6.5 based on data from 2019-20 (Figure 5). Based on these patterns, the status of coastal wetland health based on anuran communities in Lake Huron is Good, and the trend is Unchanging. By contrast, IECs at inland wetlands showed little change across years (Table 2; Figure 4, 7). Some of the highest quality coastal wetlands with respect to anurans occur in Lake Huron, even though development and agricultural stressors are fairly strong in parts of the coastal zones of this lake (Figure 4-7; Allan et al. 2013, Bourgeau-Chavez et al. 2015). The status and trend of coastal wetlands in Lake Huron are the same as in the last report (Tozer et al. 2019).

Lake Erie (including St. Clair-Detroit River Ecosystem)—Median IEC in coastal wetlands ranged from 3.5 (2008) to 5.5 (2002) from 1995-2020, did not significantly increase or decrease over the past 10 years (Table 2, Figure 4), and was 4.8 based on data from 2019-20 (Figure 5). Based on these patterns, the status of coastal wetland health based on anuran communities in Lake Erie is Fair, and the trend is Unchanging. A similar trend occurred at inland wetlands in the Lake Erie watershed (Table 2; Figures 4, 7). IECs at coastal wetlands were notably lower throughout the western portion of the lake (Figure 6) where the human development footprint is substantial (Bourgeau-Chavez et al. 2015). Out of all of the lake basins, Lake Erie had the lowest median IEC in coastal wetlands, being only slightly worse than Lake Superior and Lake Ontario (Figures 5-7).

Since the last report, the status of coastal wetlands in Lake Erie has changed from Poor to Fair. The difference in status may be due to improved sensitivity of the metrics used to calculate the IEC since the last report for reflecting the true condition of a wetland or other factors (see discussion below for further explanation).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)—Median IEC in coastal wetlands ranged from 2.6 (1995) to 7.2 (2013) from 1995-2020, significantly increased by 5.6%/year (3.7, 7.5) [lower, upper 95% confidence limits] over the past 10 years (2010-2020) (Table 2, Figure 4), and was 6.2 based on data from 2019-20 (Figure 5). Based on these patterns, the status of coastal wetland health based on anuran communities in Lake Ontario is Fair, and the trend is Improving. By contrast, IECs at inland wetlands showed little change across years (Table 2; Figures 4, 7). IECs at coastal wetlands were notably lower throughout the western portion of the lake (Figure 6) where the human development footprint is substantial (Bourgeau-Chavez et al. 2015). The status and trend of coastal wetlands in Lake Ontario are the same as in the last report (Tozer et al. 2019).

Discussion—Throughout the Great Lakes basin, the current status of coastal wetland health based on wetland breeding anurans is Fair, with the current status of Lake Huron being Good, and Lake Superior, Lake Michigan, Lake Erie, and Lake Ontario being Fair (Note that a separate determination of status and trend was made for the Overall Assessment based on data from all of the individual lake basins, rather than averaging across the lake-by-lake assessments). In addition, it was found that coastal IECs located towards the Poor end of the ecological condition gradient are more common in the western portion of Lake Ontario and Lake Erie; the southern portion of Lake Michigan; the southern portion of Green Bay, Lake Michigan; and in parts of Lake Superior compared to elsewhere throughout the Great Lakes (Figure 6). The low IECs are probably due to greater anthropogenic stress from agriculture, development, and perhaps wetland loss in Lake Michigan south of the Canadian Shield and in all of Lake Erie and Lake Ontario compared to elsewhere (Allan et al. 2013, Bourgeau-Chavez et al. 2015, Danz et al. 2007, Niemi et al. 2009). By contrast, the low IECs in parts of Lake Superior are likely mostly due to many of the wetlands being relatively small in area, tending to have lower productivity (bogs), and being ecologically quite different compared to the rest of the lakes. Despite the predominantly Fair status throughout the Great Lakes basin, some high quality coastal wetlands are still present in all of the Great Lakes (Figures 4-7). By illustrating and documenting differences in wetland health, the analysis provides a unique baseline for assessing long-term changes in wetland quality and for quantifying the success of restoration efforts in individual wetlands, regions, and the entire Great Lakes basin. A more detailed analysis of species' responses to individual stressors is available but is beyond the scope of this report. Instead, the condition of wetlands based on a multivariate "human footprint" stressor that incorporates measures of seven stressor variables (including development, agriculture, human population density) was reported.

Throughout the Great Lakes basin, coastal wetland health based on wetland breeding anurans significantly increased over the past 10 years (2009-2019), with non-significant positive trends in most individual lake basins during the same period (Table 2, Figure 4). The cause of recent increases in IECs is unclear, although it may be at least partially related to increased survey coverage in coastal wetlands following the implementation of the CWMP starting in 2011 (Figure 3). As the monitoring and sophistication of the IEC analysis program matures, we anticipate that our understanding of the inherent variability of wetland anuran species will improve over time.

Alternatively, the recent improvement in coastal IECs over the past 10 years may be at least partially due to real improvement in coastal wetland condition. An index of water quality significantly improved across 22 wetlands along the Canadian shore of Lake Ontario between 2003 and 2014 (Croft-White et al. 2017). The improvement in water quality, which is indicative of concurrent improvement in wetland breeding anuran habitat (Boyer and Grue 1995, Bishop et al. 1999, Macecek and Grabas 2011), may have also contributed to the increasing coastal wetland breeding anuran IECs that were observed in Lake Ontario and perhaps elsewhere over the same period.

In addition to assessing status and trend of the health of coastal wetlands, status and trend of inland wetlands were examined for comparison (Table 2; Figure 4, 7). Due to differences in sample sizes, the ability to compare coastal and inland wetlands was best for Lake Erie and Lake Ontario, whereas it was more limited for the other lake basins.

It is worth noting that patterns in IECs tended to differ between coastal and inland wetlands. In general, where and when IECs in coastal wetlands increased or decreased, there were no corresponding changes in IECs at inland wetlands (Table 2, Figure 4). Thus, wetland health as represented by wetland breeding anurans may be responding to different intensities of stressors in coastal versus inland wetlands within certain watersheds. This idea is supported by previous analyses using only the GLMMP dataset, which showed that occupancy of certain wetland breeding anuran species was different at coastal wetlands compared to inland wetlands (Tozer 2013, 2020). Thus, continued sampling of both coastal and inland wetlands throughout the Great Lakes basin will be useful for generating a more complete understanding of the health of wetlands based on birds throughout the entire Great Lakes watershed.

The overall Fair status and Improving trend reported for coastal wetlands throughout the Great Lakes basin contrasts with some of the previous reports for this sub-indicator (Tozer 2014, Tozer et al. 2017). The differences are at least partially due to differences in sampling. Some of the previous reports were based predominantly on data from the southern portion of the Great Lakes basin due to reliance on the mostly southern GLMMP dataset. The current report provides a more balanced assessment throughout the entire Great Lakes basin by bringing GLMMP data together with data from the southern and northern GLEI and CWMP projects. Nevertheless, the patterns reported here are based on a comprehensive IEC metric, which represents the collective responses of 8 wetland breeding anuran species or groups of species to wetland condition. Yet at least one species, the Western Chorus Frog, has experienced long-term declines in occupancy in the southern portion of the Great Lakes (e.g., Tozer 2013, 2020). This species may be responding to environmental stressors in species-specific ways, warranting unique actions or presenting unique opportunities for restoration to improve wetland health.

Linkages

Coastal wetland breeding anurans are influenced by numerous local and landscape-level characteristics, some of which are monitored by other State of the Great Lakes sub-indicators. For instance, coastal wetland breeding anurans are known to be influenced by water levels (Gnass Giese et al. 2018). Thus, the Coastal Wetland Amphibians sub-indicator can be expected to co-vary with the Water Levels sub-indicator, and in turn, with climate-related impacts that indirectly affect water levels, such as the Surface Water Temperatures, Ice Cover, and Precipitation Amounts sub-indicators. By contrast, we note that we saw little evidence within individual lake basins or overall across the entire Great Lakes basin for an effect of changing lake levels on IECs for coastal wetlands based on anuran communities. For instance, coastal IECs based on breeding anurans changed very little during the period of rapid lake level rise from 2013 to 2018 (Figure 4; mean increase in lake level from 2013-2018: 0.16-0.97 m depending on the lake; Hohman et al. 2021) (see Water Levels Sub-Indicator for further details including historical context back to 1918). We did, however, observe far fewer anurans at some wetlands where water levels were especially high; uncovering why lake levels influence anuran occurrence at some scales and not others is an area for future research.

In addition, timing of breeding of wetland anurans can be influenced by altered temperatures due to climate change (Walpole et al. 2012), although whether this influence results in changes in anuran occurrence and abundance is unknown, so it may be possible that the Coastal Wetland Amphibians sub-indicator can be expected to directly co-vary with the Surface Water Temperatures sub-indicator. As well, coastal wetland breeding anurans are influenced by various water pollutants, particularly nitrates (e.g., Rouse et al. 1999). Thus, the Coastal Wetland Amphibians sub-indicator can be expected to co-vary with all of the Toxic Chemicals and Nutrients sub-indicators. The Coastal Wetland Amphibians sub-indicator can also be expected to co-vary with sub-indicators that track the extent and spatial arrangement of wetland breeding anuran habitat (e.g., Coastal Wetlands: Extent and Composition, Aquatic

Habitat Connectivity) and prey (Coastal Wetland Invertebrate; Coastal Wetland Fish). And finally, the occurrence of multiple species of anurans is known to be lower in developed and urban landscapes (Price et al. 2004, 2007), as reflected in the pattern of IECs based on anuran communities presented here (Figure 6). Thus, the Coastal Wetland Amphibians sub-indicator can be expected to co-vary with the Human Population sub-indicator.

Traditional Ecological Knowledge (TEK), Citizen Science and Other Bodies of Knowledge

The tremendous power to report robust status and trends for this sub-indicator are made possible by large sample sizes and extensive survey coverage accomplished mainly by two ongoing broad-scale anuran monitoring programs operating throughout the Great Lakes. The GLMMP, which draws upon the dedication and skills of hundreds of volunteer surveyors annually, provided much of the data used here from 1995 to 2011, as well as all of the inland data used for comparison to coastal wetlands (Tozer 2013, 2020). The CWMP has provided much needed expansion of coverage in coastal wetlands particularly in the northern portion of the Great Lakes since 2011 (Uzarski et al. 2017). Ongoing support of both of these programs will ensure that these quality data continue to be available for understanding the condition of the health of the Great Lakes ecosystem.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: CWMP and GLEI: https://www.greatlakeswetlands.org/Home.vbhtml GLMMP: https://www.birdscanada.org/birdmon/default/main.jsp		

Data Limitations

This sub-indicator focuses on anurans because they are more readily detected than other amphibians. Other amphibian species, such as salamanders, are not surveyed. Nonetheless, monitoring results for anurans likely provide an indication of habitat suitability for other amphibians dependent on similar habitats.

This sub-indicator may be more powerful if evaluated as part of an overall analysis of biological communities of Great Lakes coastal wetlands and nearshore aquatic systems. This can be done by considering the coastal wetland sub-indicators in combination, because they function and indicate anthropogenic disturbance at different spatial and temporal scales, and have varying resolution of detection. However, the geographic scale of disturbance for coastal wetland breeding anurans is not yet determined, nor is the resolution for detecting ecosystem health. This is a fruitful area for further development.

Great Lakes coastal wetlands are diverse. Some are protected; some are open to the lake. Some are associated with river mouths, others are not. Additional variation is imposed by underlying geomorphology and climate differences across different regions. This variation in wetland types carries over to influence anurans. Thus, the most powerful data for estimating anuran-based indices of ecosystem health at broad scales, such as across individual Great Lakes basins, are those collected via spatially-balanced randomized experimental designs, which account for and capture the variation. Most, but not all, of the existing wetland breeding anuran datasets suitable for use with this sub-indicator address these sampling issues in their experimental designs, although there is room for improvement.

Additional Information

Over half of all Great Lakes coastal wetlands have been lost or changed by human activities and many remaining coastal wetlands suffer from anthropogenic stressors, such as nutrient and sediment loading, fragmentation, invasive species, shoreline alteration, and water-level control, as documented by a binational Great Lakes-wide mapping and attribution project (Albert and Simonson, 2004; Ingram and Potter, 2004). Such wetland loss and stress are especially apparent in certain portions of the Great Lakes basin (Allan et al. 2013, Bourgeau-Chavez et al. 2015). Indeed, it was found that IECs based on anuran community data were especially poor at coastal wetlands throughout the western portion of Lake Ontario and Lake Erie, the southern portion of Lake Michigan, and the southern portion of Green Bay, Lake Michigan, where stress due to human development is substantial.

One approach to identify appropriate actions and opportunities to improve the health of coastal wetlands is to identify factors that are positively associated with the occurrence of anuran species. Factors related to increased probability of occurrence then translate into actions and opportunities that will help increase occurrence of declining species and ultimately improve wetland health. This approach has been completed using the GLEI component of the larger dataset analyzed in this report. Using step-wise logistic regression models and data from 279 GLEI point counts conducted at 93 sample points, Price et al. (2004) identified important local, wetland, and landscape-scale factors influencing occupancy of five wetland breeding anuran species in coastal wetlands throughout U.S. portion of Lake Michigan and Lake Huron. The results of the study suggest that most wetland-breeding anuran species benefit from landscape factors such as conserving, restoring, or creating wetlands surrounded by limited urban land use and increased forest cover. In addition, individual or smaller groups of species also benefit from conserving, restoring, or creating robust-emergent-dominated but interspersed, *Phragmites*-free wetlands surrounded by higher proportions of wetland cover in the surrounding landscape. Rouse et al. (1999) found that nitrate runoff into Great Lakes wetlands was high enough to cause sub-lethal effects in amphibians in 20% of widespread samples and recommended that natural vegetated buffer strips around wetlands could help mitigate the effects. These actions will help promote occupancy among anuran species, which will also ultimately help improve associated coastal wetland health across the southern portion of the Great Lakes basin.

The status and trend assessment of coastal wetland health based on wetland breeding anurans is based on BR functions developed using CWMP data only (since landscape variables were readily available for CWMP sites). The BR functions were also developed based on information from multiple stressor gradients related to development, agriculture, and human population density. The ability of the IEC to capture the health of coastal wetlands based on anuran data might be improved by expanding the development of the BR functions to include all of the wetland breeding anuran data that are available from the GLMMP, GLEI, and CWMP projects. The performance of the IEC might also be improved by incorporating other known wetland breeding anuran stressors in the development of BR functions, particularly within-wetland attributes like relative dominance of invasive plant species. These ideas are fruitful areas for future expansion.

Three large wetland breeding anuran datasets were brought together, specifically the GLMMP, GLEI, and CWMP project datasets to perform the analyses summarized in this report. This provided tremendous analytical power at many different scales compared to using only one of the datasets. However, it was evident that the combined dataset is deficient in information from healthy wetlands. Future collection of anuran data from wetlands located towards the pristine end of the degraded-pristine gradient might improve the performance of the IEC.

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Table 1. Wetland anuran species and groups of species (*n* = 8) used to generate biotic response functions for calculating the Index of Ecological Condition (IEC) as an estimate of the health of Great Lakes wetlands. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

No.	Taxon	Species (scientific name)
1	American Toad	American Toad (*Anaxyrus americanus*)
2	Bullfrog	Bullfrog (*Rana catesbeianus*)
3	Gray Treefrog	Eastern Gray Treefrog (*Hyla versicolor*), Cope's Gray Treefrog (*Hyla chrysoscelis*)
4	Chorus Frog	Boreal Chorus Frog (*Pseudacris maculata*), Western Chorus Frog (*Pseudacris triseriata*)
5	Green Frog	Green Frog (*Rana clamitans*)
6	Northern Leopard Frog	Northern Leopard Frog (*Rana pipiens*)
7	Spring Peeper	Spring Peeper (*Pseudacris crucifer*)
8	Wood Frog	Wood Frog (*Rana sylvaticus*)

Table 2. Long-term (1995-2020) and 10-year (2010-2020) trends in median Index of Ecological Condition (IEC) throughout coastal and inland wetlands of the entire Great Lakes (overall) and within each basin (e.g., Superior). Shown are geometric mean rates of change (Trend: %/year) based on differences between the first and last year in the trend, along with "lower" and "upper" 95% confidence limits. Instances where sample sizes were too low to calculate reliable trends are shown with dashes. Trends calculated using equation 4 in Smith et al. (2014) with 1,000 bootstrap replicates. Statistically significant trends are shown in bold, based on confidence intervals that do not overlap zero. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

Wetland type	Basin	Start year	End year	Trend	Lower	Upper
Coastal	**Overall**	**1995**	**2020**	**1.9**	**1.2**	**2.5**
Coastal	**Overall**	**2009**	**2020**	**4.2**	**3.1**	**5.3**
Coastal	Superior	1995	2020	–	–	–
Coastal	Superior	2009	2020	–	–	–
Coastal	Huron	1995	2020	–	–	–
Coastal	Huron	2009	2020	1.3	-1.5	4.2
Coastal	Michigan	1995	2020	–	–	–
Coastal	Michigan	2009	2020	–	–	–
Coastal	Erie	1995	2020	0.7	-0.4	1.8
Coastal	Erie	2009	2020	2.1	-0.5	4.6
Coastal	**Ontario**	**1995**	**2020**	**3.1**	**1.9**	**4.3**
Coastal	**Ontario**	**2009**	**2020**	**5.6**	**3.7**	**7.5**
Inland	**Overall**	**1995**	**2020**	**-0.8**	**-1.3**	**-0.2**
Inland	Overall	2009	2020	-0.6	-1.9	0.7
Inland	Superior	1995	2020	–	–	–
Inland	Superior	2009	2020	–	–	–
Inland	Huron	1995	2020	-0.2	-1	0.6
Inland	Huron	2009	2020	1.1	-1.7	3.9
Inland	Michigan	1995	2020	–	–	–
Inland	Michigan	2009	2020	–	–	–
Inland	Erie	1995	2020	-1.1	-2.4	0.2
Inland	Erie	2009	2020	-1.4	-4.3	1.4
Inland	Ontario	1995	2020	-0.6	-1.6	0.4
Inland	Ontario	2009	2020	-0.5	-2.3	1.2

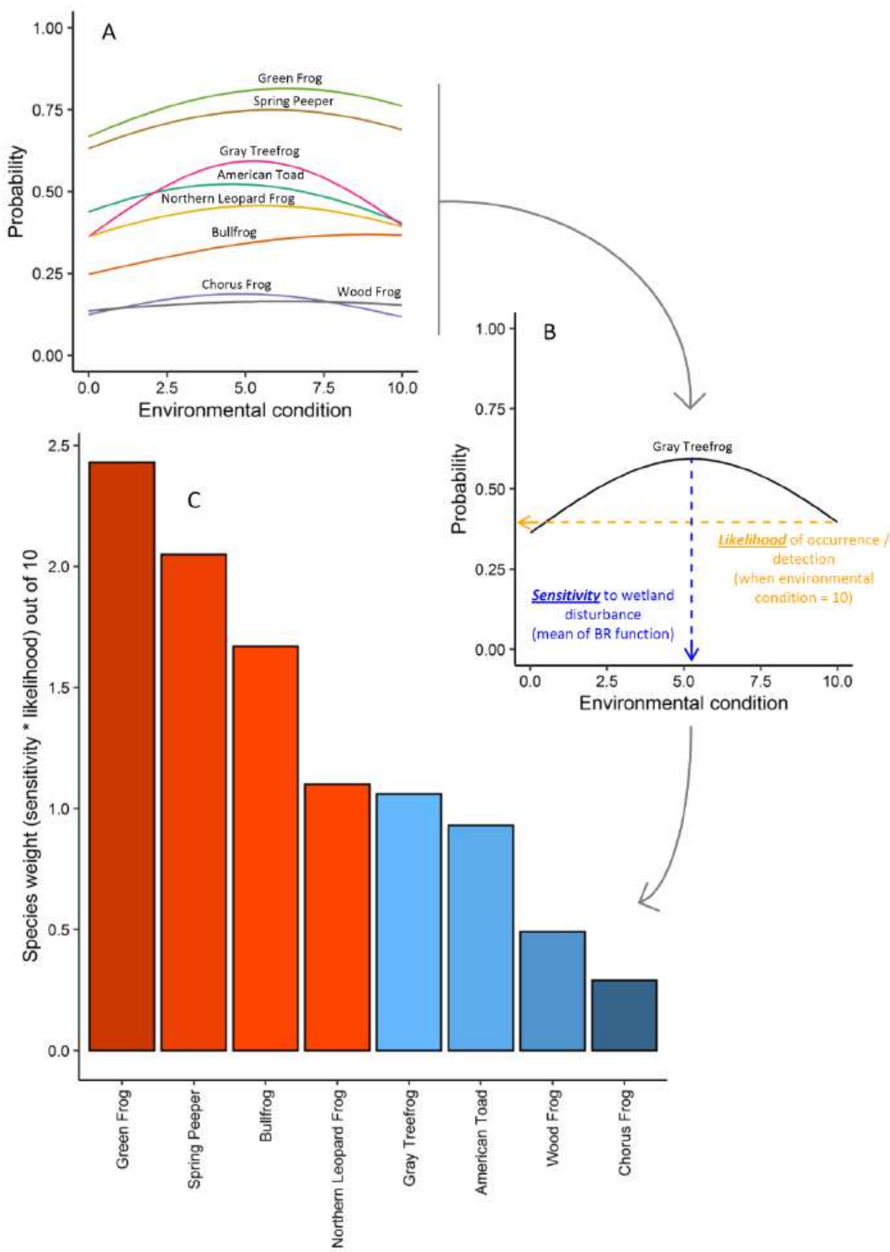

Figure 1. Schematic representation of the steps required to calculate the Index of Ecological Condition (IEC) in this report based on anuran data from coastal wetlands throughout the Great Lakes basin. For each of 8 anuran species or groups of species (see Table 1 for details), a biotic response function (A) depicting the species' probability of occurrence as a function of a combined "human footprint" variable (Environmental condition) based on wetland area, development, agriculture, and human population density (0 = Poor condition, 10 = Good condition) was used to calculate the species' sensitivity to wetland disturbance and likelihood of occurrence / detection (B), which were then multiplied together and scaled out of 10 to derive the species' weight for use in generating IECs for particular wetlands. For example, a wetland where only Green Frog was present received an IEC of 2.4; a wetland where Green Frog and Spring Peeper were present received an IEC of 4.5; and a wetland where all 8 species were present received an IEC of 10. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

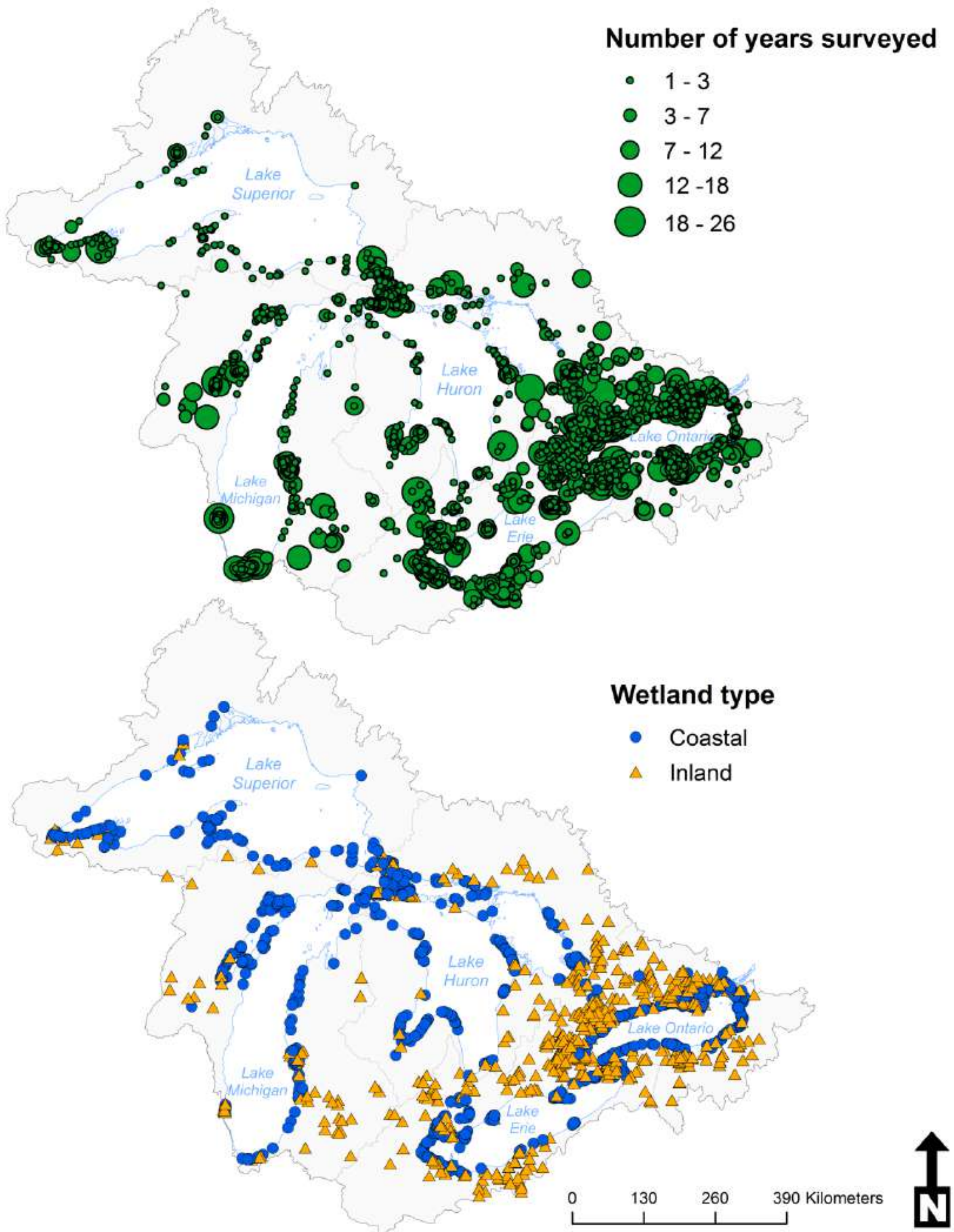

Figure 2. Wetlands surveyed for anurans from 1995-2020 throughout the Great Lakes basin for the purpose of estimating anuran-based indices of wetland health. Shown are wetlands as a function of the number of years that each wetland was surveyed (top) and as a function of coastal versus inland (bottom). Note that coastal wetlands (*n* = 1,164) far outnumber inland wetlands (*n* = 660), although this does not appear to be the case due to tightly overlapping symbols. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

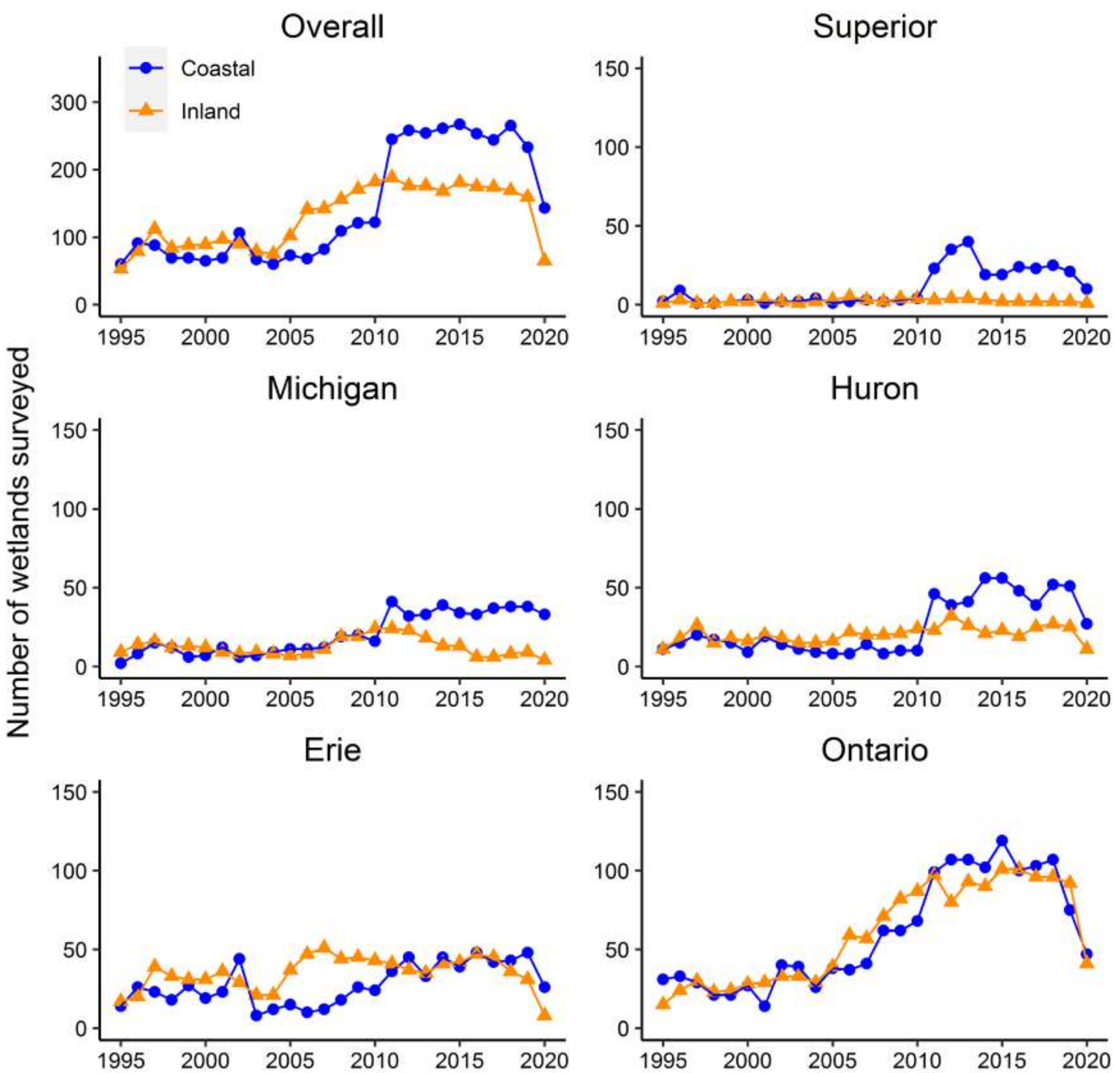

Figure 3. Number of wetlands surveyed for anurans per year from 1995-2020 throughout the Great Lakes basin for the purpose of estimating anuran-based indices of wetland health. Shown are wetlands surveyed as a function of the entire Great Lakes basin (overall) and each individual lake basin for coastal and inland wetlands. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

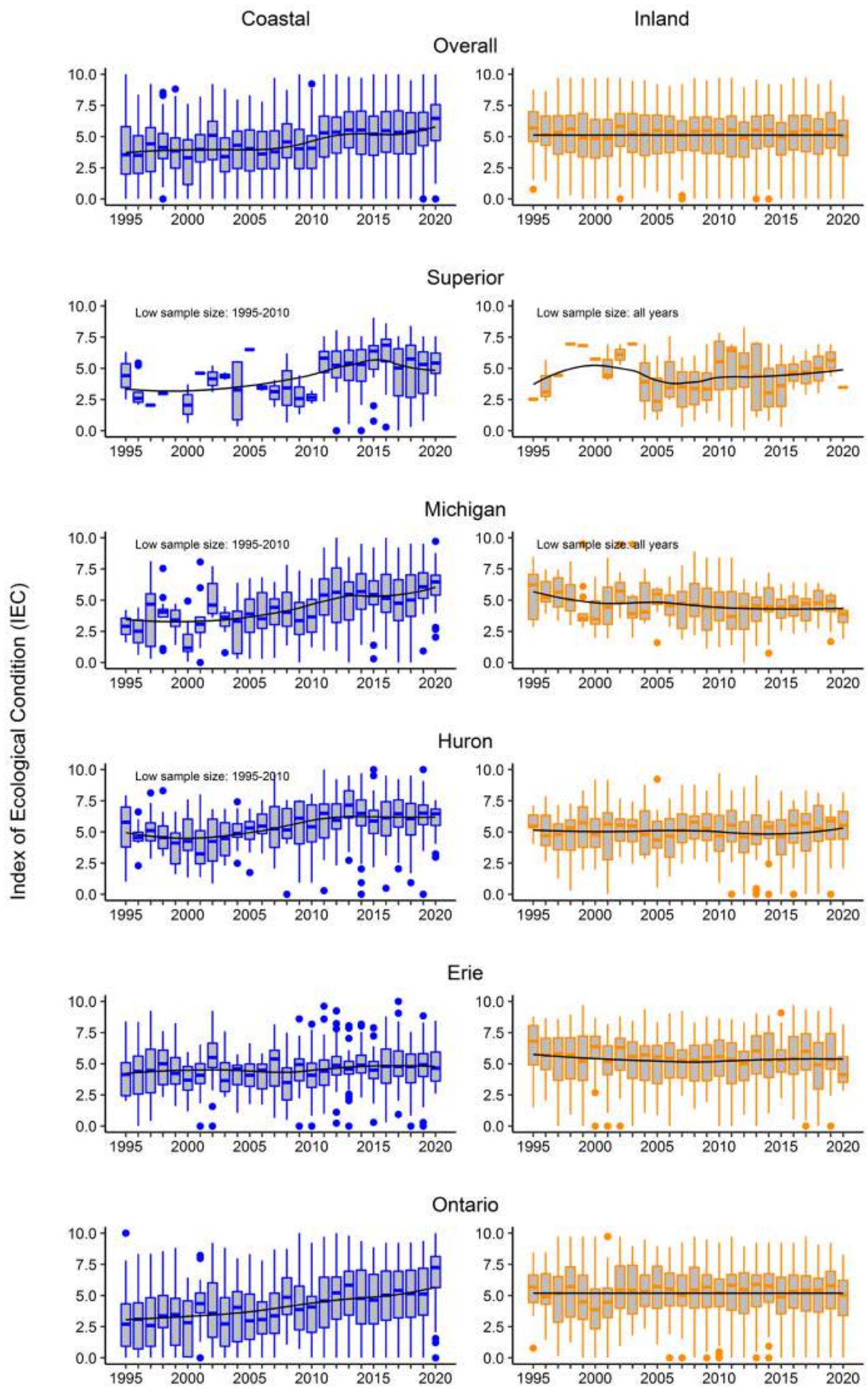

Figure 4. Temporal trends in the Index of Ecological Condition (IEC) based on anuran community data from 1995-2020 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal (blue) and inland (orange) wetlands. Shown are box-and-whisker plots of wetland-level means in each year along with loess smoother lines of best fit (solid black lines). Whiskers represent 1.5 times the inter-quartile range; dots are outliers. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

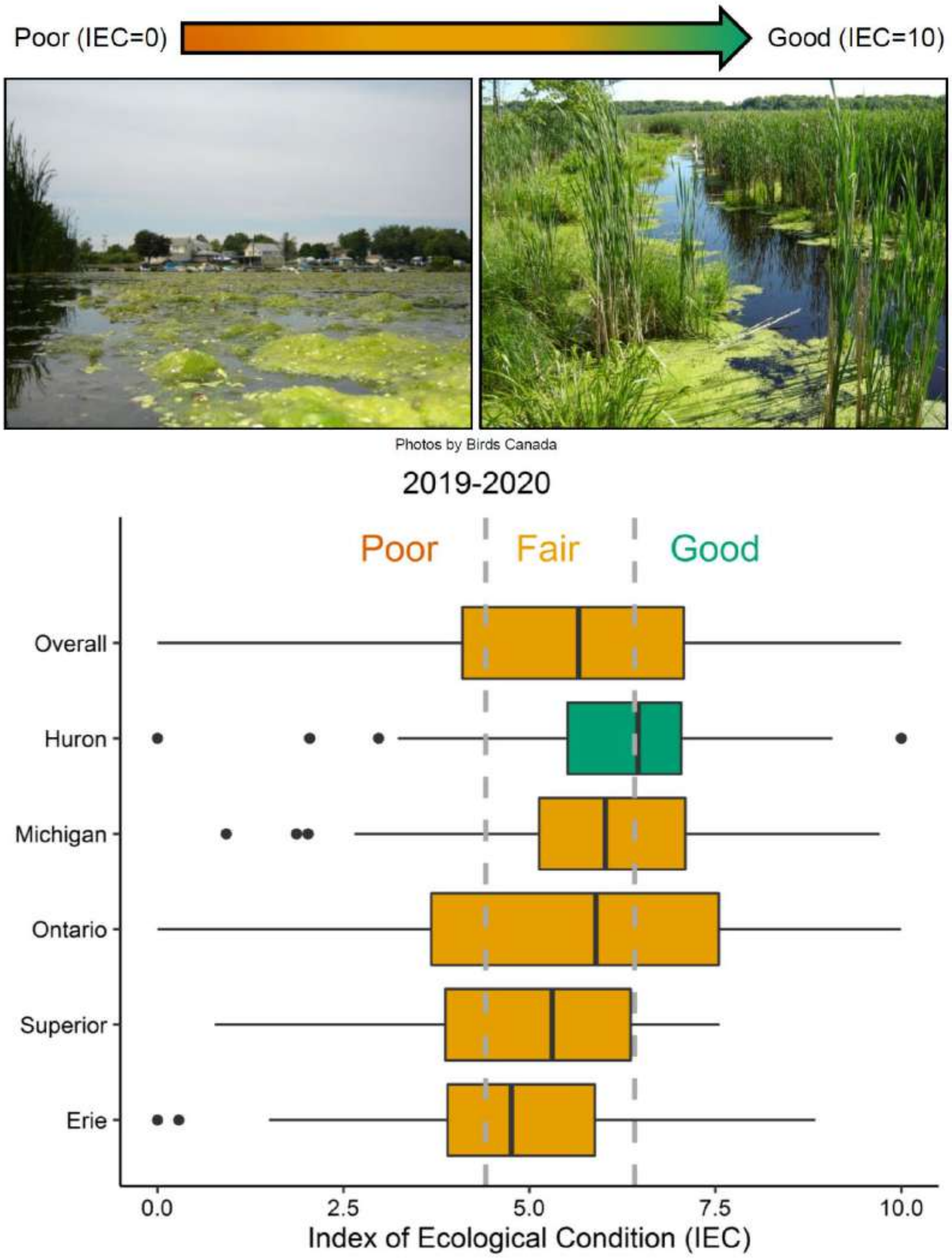

Figure 5. Distribution of the Index of Ecological Condition (IEC) based on anuran community data in 2019-2020 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal wetlands. Shown are box-and-whisker plots of wetland-level means. Whiskers represent 1.5 times the inter-quartile range; dots are outliers. Vertical dashed grey lines show defining values for Poor, Fair, and Good status. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

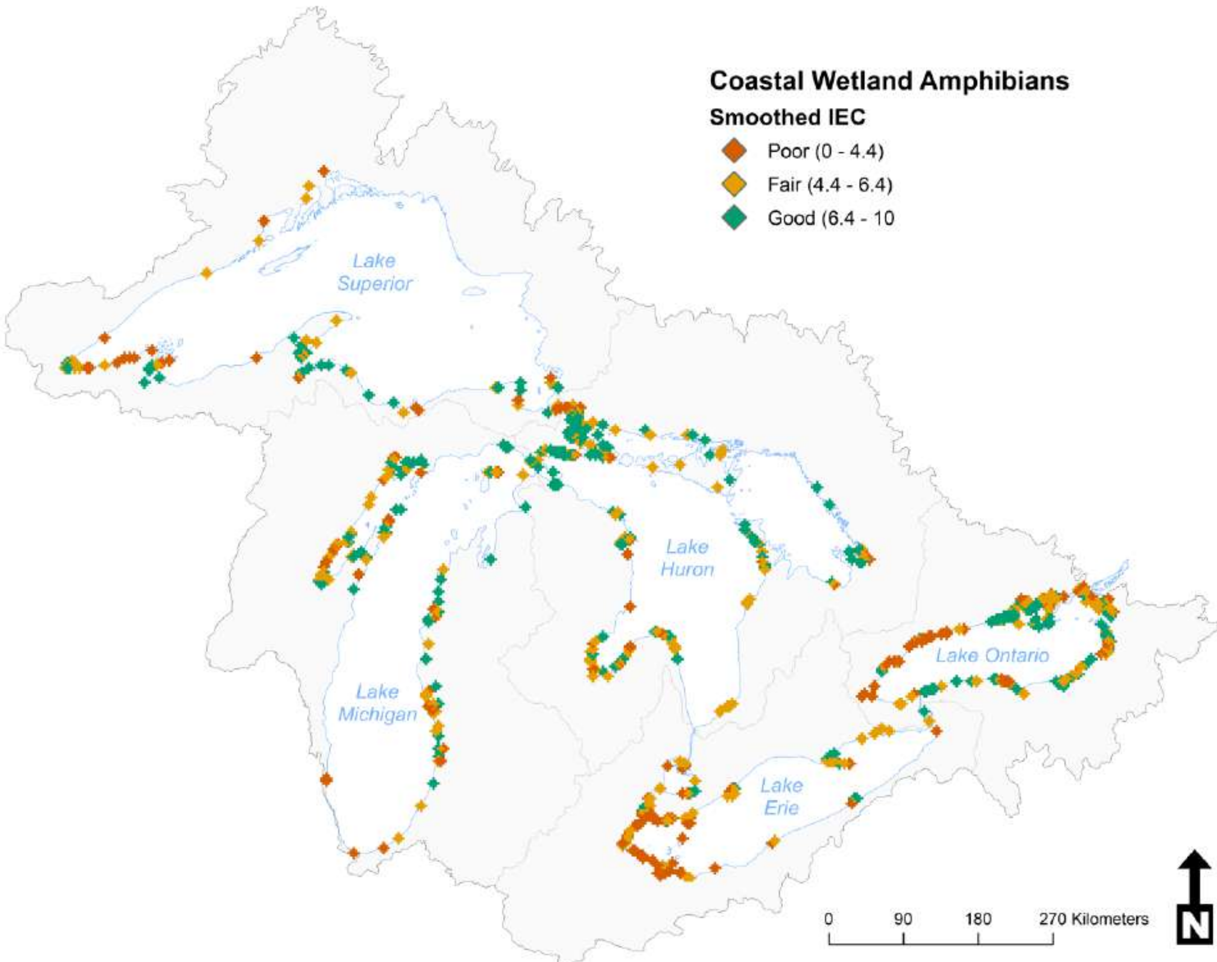

Figure 6. Index of Ecological Condition (IEC) throughout coastal wetlands of the Great Lakes based on anuran community data from the past 5 years (2016-2020). Shown are smoothed wetland-level mean IEC values, averaged across years for wetlands that were sampled in multiple years. To smooth the data, the point-to-raster tool in ArcGIS was used to calculate mean values within 3,000 m cells, and the focal statistics tool was used on the resulting means to calculate average values within a 2-cell circular window, which is referred to as "smoothed IEC." The most recent 5 years of data were used to boost sample sizes. Defining values for colours correspond to Poor, Fair, and Good status. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

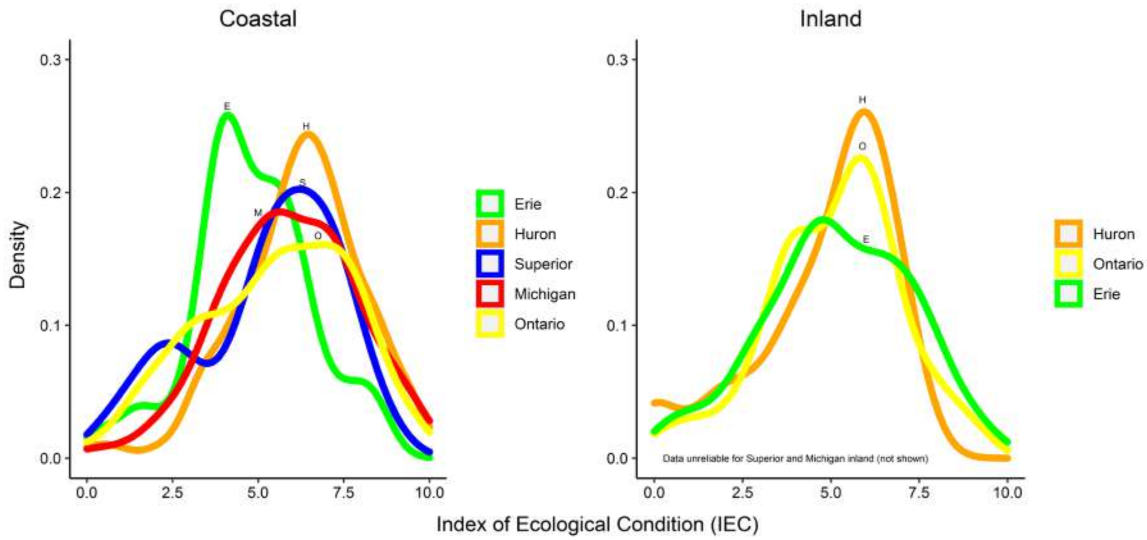

Figure 7. Density plots of the Index of Ecological Condition (IEC) based on anuran community data from the past 5 years (2016-2020) throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal and inland wetlands. Shown are plots based on wetland-level mean IEC values, averaged across years for wetlands that were sampled in multiple years. Inland data for Superior are not shown due to small sample sizes. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

Sub-Indicator: Coastal Wetland Birds

Overall Assessment

Status: Fair

Trends:

10-Year Trend (2010-2020): Improving

Long-term Trend (1995-2020): Improving

Rationale: Median Index of Ecological Condition (IEC, a measure of coastal wetland health using observations of coastal wetland birds) in coastal wetlands was 3.7 (out of 10) based on data from 2019-2020, while the median IEC ranged from 1.3 to 4.5 from 1995-2020. IECs showed a significant increase of 4.9%/year over the past 10 years (2010-2020). The status of coastal wetland health based on bird communities in the Great Lakes overall is Fair, and the trend is Improving. Note that a separate determination of status and trend was made for the Overall Assessment based on data from all of the individual lake basins, rather than averaging across the lake-by-lake assessments given below. IECs at coastal wetlands were notably lower throughout the western portion of Lake Ontario where the human development footprint is substantial compared with most of the rest of the lakes where, generally, a mix of low and high IECs occurred.

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: Median IEC for coastal wetlands based on bird communities was 2.4 using data from 2019-2020 and ranged from 0.5 to 6.0 from 1995-2020. There appeared to be an increase in median IEC in coastal wetlands over the past decade followed by a decrease in the most recent few years, although the trend is not statistically significant. Although landscapes in the coastal zone of Lake Superior are generally non-agricultural and minimally developed compared with wetlands in the more southern lakes, coastal wetlands of Lake Superior (with a few notable exceptions) are relatively small in area, tend to have lower productivity (bogs), and are ecologically quite different compared to the rest of the lakes, which may account at least partially for some of the modest scores in comparison with those from other lakes.

Lake Michigan

Status: Fair

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: Median IEC for coastal wetlands based on bird communities was 2.8 using data from 2019-2020 and ranged from 0.6 to 5.9 from 1995-2020. There appeared to be an increase in median IEC in coastal wetlands over the past decade or so followed by a decrease in the most recent few years, although the trend is not statistically significant. Some of the highest quality coastal wetlands with respect to birds occur in Lake Michigan, even though development and agricultural stressors are common to extensive in the coastal zones of this lake. For example, the lower IECs at coastal wetlands in the southern third of the lake compared to the rest of the lake could be due to strong development-related stressors.

Lake Huron (including St. Marys River)

Status: Good

10-Year Trend (2010-2020): Unchanging

Long-term Trend (1995-2020): Undetermined

Rationale: Median IEC for coastal wetlands based on bird communities was 4.9 using data from 2019-2020 and ranged from 1.2 to 7.1 from 1995-2020. There appeared to be an increase in median IEC in coastal wetlands over the past decade or so followed by a decrease in the most recent few years, although the trend is not statistically significant. Some of the highest quality coastal wetlands with respect to birds occur in Lake Huron, even though development and agricultural stressors are fairly strong in parts of the coastal zones of this lake .

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Fair

10-Year Trend (2010-2020): Unchanging

Long-term Trend (1995-2020): Unchanging

Rationale: Median IEC for coastal wetlands based on bird communities was 4.7 using data from 2019-2020 and ranged from 0.9 to 5.4 from 1995-2020. The trend did not significantly increase or decrease over the past 10 years (2010-2020).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Fair

10-Year Trend (2010-2020): Improving

Long-term Trend (1995-2020): Improving

Rationale: Median IEC for coastal wetland based on bird communities was 3.1 using data from 2019-2020 and ranged from 0.5 to 4.0 from 1995-2020. The trend significantly increased by 7.5%/year over the past 10 years (2010-2020). IECs were lower at coastal wetlands throughout the western portion of the lake compared to the rest of the lake, most likely due to especially strong development-related stressors.

Status Assessment Definitions

Standardized indices (IEC values/scores) range between 0 (Poor condition or most degraded) and 10 (Good condition or least degraded). Status will be determined based on median IEC in coastal wetlands in the most recent year (for 2022 reporting, status is based on data from 2019-2020 because sample sizes were relatively low in 2020) compared to percentiles based on data from 2011-2020 in all coastal wetlands. Coverage of coastal wetlands throughout the Great Lakes has been most extensive since 2011 with the implementation of the Great Lakes Coastal Wetland Monitoring Program (CWMP). Note that the values defining Poor, Fair, and Good below have changed since the previous report (Tozer et al. 2019) because of: 1) improved sensitivity of the metrics used to calculate the IEC since the last report and 2) more years of data used to calculate the values since the last report.

Good: Most or all ecosystem components are in acceptable condition; median IEC > 66^{th} percentile (for 2022 reporting, median IEC > 4.8).

Fair: Some ecosystem components are in acceptable condition; 33rd percentile ≤ median IEC ≤ 66^{th} percentile (for 2022 reporting, 1.9 ≤ median IEC ≤ 4.8).

Poor: Very few or no ecosystem components are in acceptable condition; median IEC < 33^{rd} percentile (for 2022 reporting, median IEC < 1.9).

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Both regional and local marsh bird populations naturally fluctuate over time, particularly due to fluctuations in Great Lakes water levels; therefore, several years of monitoring data with suitable geographic coverage throughout targeted areas are required to detect all but the most dramatic trends. Interpretation of this sub-indicator may be improved if coupled with patterns observed in other wetland sub-indicators.

The terms Improving, Unchanging, and Deteriorating were applied based on geometric mean rates of change (%/year) of a standardized bird-based index of ecosystem health, the Index of Ecological Condition or IEC (Howe et al. 2007a, 2007b), using equation 4 in Smith et al. (2014). The statistical significance of trends was assessed via parametric bootstrapping in R (R Core Team 2020) with package “boot” (Canty and Ripley 2013) and 1,000 bootstrap replicates. Bootstrapping in this manner was necessary to account for the varying precision of the annual estimate in the start and end year used to calculate each trend. For 10-year trends, the start year was 2010 and the end year was 2020, whereas for long-term trends, the start year was 1995 and the end year was 2020. The year 2010 was chosen as the start year for 10-year trends because during 2010-2020 there are 10 possible transitions between years, which was taken to be most representative of the trend during the most recent decade (IUCN Standards and Petitions Committee 2019). Trend estimates with 95% confidence intervals that did not overlap zero were considered statistically significant.

Improving: A statistically significant increase in IEC.

Unchanging: No statistically significant increase or decrease in IEC.

Deteriorating: A statistically significant decrease in IEC.

Undetermined: Data are not available or are insufficient to report on a trend.

Endpoints and/or Targets

Endpoints will be established based on existing data. Alternative endpoints may be used based on historical records or expert opinion in certain instances as determined and justified by the authors. Endpoints for this sub-indicator were left undefined for the 2022 report due to uncertainty around the best way to calculate meaningful endpoints.

Ongoing work by the authors will strive to develop useful endpoints.

Sub-Indicator Purpose

To assess the status and trend of Great Lakes coastal wetland ecosystem health by directly measuring the composition and occurrence of wetland birds, and thereby inferring the condition of coastal wetland habitat as it relates to the health of this ecologically and culturally important component of wetland communities.

Ecosystem Objective

Coastal wetlands provide critical breeding and migratory habitat for wildlife such as birds. Conservation of remaining coastal wetlands and restoration of previously degraded or destroyed wetlands are vital components of restoring the Great Lakes ecosystem. Birds are effective ecological indicators and can be used to report progress toward such an objective.

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species."

Measure

Background—Wetland birds are influenced by the physical, chemical, and biological components of wetlands and surrounding landscapes. For example, the occurrence, abundance, and/or reproductive success of multiple bird species in the Great Lakes basin decline as (1) wetland size decreases; (2) wetland habitat and natural cover in the surrounding landscape decrease or degrade in quality; (3) pollution from pesticides, herbicides, and sediment runoff increases; (4) generalist predators (e.g., northern raccoon [*Procyon lotor*]) associated with anthropogenic habitats in the surrounding landscape increase; and 5) lake levels decrease (Brazner et al. 2007a, 2007b; Crosbie and Chow-Fraser 1999; Hohman et al. 2021; Howe et al. 2007a; Gnass Giese et al. 2018; Grandmaison and Niemi 2007; Naugle et al. 2000; Smith and Chow-Fraser 2010 a, 2010b; Timmermans et al. 2008; Tozer et al. 2010, 2020). Thus, the occurrence or abundance of sensitive wetland birds can be a valuable indicator of the health of wetlands and the surrounding landscape.

Data—Several initiatives monitor Great Lakes wetland birds. One of the longest running is the Great Lakes Marsh Monitoring Program (GLMMP) coordinated by Birds Canada, which started in 1995 and has operated every year since then at coastal and inland wetlands throughout much of the Great Lakes basin (Tozer 2013, 2016, 2020). Some of the previous reports for this sub-indicator are based solely on data from this ongoing broad scale program (e.g., Tozer 2014). From 2001 to 2005, the University of Minnesota Duluth's Natural Resource Research Institute (NRRI) led an ambitious multi-institutional Great Lakes Environmental Indicator Project (GLEI) aimed at assessing the overall biotic health of coastal wetlands in the U.S. portion of the Great Lakes (Howe et al. 2007a, 2007b; Hanowski et al. 2007a, 2007b; Niemi et al. 2007). More recently, the CWMP, which is a partnership between USEPA and Central Michigan University, was initiated in 2011 and currently is scheduled to operate until at least 2025 throughout both the U.S. and Canadian Great Lakes coastal zones (Uzarski et al. 2017). These projects/programs have somewhat different study designs but rely on standardized, fixed duration point counts that can be adjusted to maximize cross-project compatibility. To garner large numbers of skilled, trained, volunteer surveyors to achieve large sample sizes at relatively low cost, the GLMMP allows participants to select sample wetlands and sample points within them—a justifiable approach if one assumes that the sample locations are representative of wetlands across a region of interest. By contrast, GLEI and CWMP select sample wetlands via stratified random sampling and survey sample points within them via paid professional staff. Nonetheless, all of the projects/programs target wetlands dominated by non-woody emergent plants, such as cattails (*Typha* spp.) and sedges (e.g., *Carex* spp.), with sample points located within wetlands. Furthermore, Tozer (2020) showed that in southern Ontario annual indices of population abundance for 9 of 10 marsh-breeding bird species were statistically comparable in each of three years when based on data collected by GLMMP participants, as compared to data collected by paid professional staff at stratified random locations throughout the same study area. In this report, the datasets listed above were brought together to generate a comprehensive analysis of the status and trend of Great Lakes coastal wetland birds and associated wetland health.

Surveys—Wetland birds were sampled to an unlimited distance from 1 point (hereafter "sample point"), or up to 8 points, located at the edge of or within a wetland depending on its size. In most large wetlands, sample points were placed both near the upland/wetland interface (shoreline) and in the interior of the wetland, including the open lake/wetland interface. In most small wetlands, only shoreline points were sampled. Sample points were > 250 m (GLMMP) or > 400 m (GLEI, CWMP) apart to avoid multiple detections of the same individuals. Depending on the year and project/program, each sample point was surveyed for 10 or 15 minutes on 1-3 visits separated by at least 10 or 15 days during the main avian breeding season, typically between late May and early July. Differences among the field protocols have been shown to have minor influence on the probability of detecting most of the species considered in this report (e.g., Tozer et al. 2016, 2017). Surveys occurred in either the morning (30 minutes before local sunrise to 10:00 h local time) or evening (4 hours before local sunset to dark) and only under weather conditions that were favourable for detecting all species and individuals present (little to no precipitation; wind: Beaufort 0-3, 0-19 km/hr). Observers broadcasted calls during surveys to entice vocal response by individuals of especially secretive species. The broadcast calls occurred during a 5-minute portion of each 10- or 15-minute survey and consisted of 30 seconds of vocalizations followed by 30 seconds of silence for each of the following species: Least Bittern (*Ixobrychus exilis*), Sora (*Porzana carolina*), Virginia Rail (*Rallus limicola*), a mixture of American Coot (*Fulica americana*) and Common Gallinule (*Gallinula galeata*), and Pied-billed Grebe (*Podilymbus podiceps*), in that order. The survey protocols of each of the projects/programs closely follow the Standardized North American Marsh Bird Monitoring Program protocol (Conway 2011). The survey protocol for the GLMMP is summarized in Tozer (2016), whereas the survey protocol for the CWMP is summarized in Uzarski et al. (2017).

Analysis—Numerous methods are available for analyzing Great Lakes coastal wetland bird data. Some previous analyses for this report were based on the separate status and trends of the relative abundance of approximately 20 wetland bird species (e.g., Tozer 2014). Alternative approaches include various bird-based indices of wetland health, which combine data from suites of species (e.g., Chin et al. 2014). The latter approach is likely more objective and more practical for the purposes of State of the Great Lakes because it provides a single comprehensive metric that represents the collective responses of bird species to wetland condition. Multi-species metrics, like the widely used index of biotic integrity for fishes (Karr and Chu 1999) and mean coefficient of conservatism for plants (Taft et al. 1997), are robust when they are tested against known stressor gradients and include enough species to allow calculations even if some species are absent due to extraneous factors. For example, a wide-ranging resident species might go undetected because, by chance, all local individuals of that species happen to be absent or inactive within the survey plot during the sampling period. Similarly, a high quality wetland might be missing a species because of factors unrelated to habitat, such as a regional epidemic that affects individuals regardless of wetland condition or bird declines caused by mortality during migration.

In this report, we assessed bird community health based on multi-species data collected at thousands of sample points throughout the Great Lakes basin in both the U.S. and Canada (Howe et al. 2007a, 2007b; Hanowski et al.

2007a, 2007b; Tozer 2013, 2016, 2020). We assembled quantitative information for seven minimally correlated ($-0.30 \leq r \leq 0.31$) environmental stressors at 2,236 single-year CWMP point samples (i.e., the combined early + late season point counts during a given year at a given point sampled between 2011 and 2018) located within 616 coastal wetlands (Elliott 2019): 1) EmWetArea, the area of emergent, non-woody vegetation within 500 m of the point + ½ the emergent wetland area between 500-1,000 m; 2) %Dev2km, the percent cover of developed land (industrial buildings, residential sites, golf courses, etc.) within 2 km of the sample point (excluding open water); 3) %Crop2km, the percent cover of agricultural cropland within 2 km of the sample point (excluding open water); 4) Roads2km, total meters of primary and secondary roads within 2 km of the sample point; 5) %DevWatershed, the percent developed land within the watershed draining into the wetland; 6) %CropWatershed, the percent cropland within the watershed draining into the wetland; and 7) PopWatershed, total human population within the watershed. The watershed variables were calculated by Brazner et al. (2007a,b) as part of the GLEI project (Danz et al. 2005, 2007). We note that the two agricultural land cover variables (%Crop2km and %CropWatershed, $r = -0.01$) and the two land development variables (%Dev2km and %DevWatershed, $r = 0.02$) were largely independent. We log-transformed variables when needed to improve normality and then conducted a Principal Components Analysis (PCA; McCune and Mefford 1999) to reduce the dimensionality of the multivariate data set. The first 4 PCA axes explained 87% of the variation. Scores for these 4 axes were scaled from worst (most disturbed) to best (least disturbed) condition and summed to produce a raw index of environmental condition that approximates the "human footprint," which was then converted to C_{env}, an environmental condition score on a 0 to 10 scale, where 0 represents the most disturbed and 10 the least disturbed.

Each of the 2,236 single-year CWMP point samples were placed within bins of 15 points each (or 16 in the last case) with similar measures of environmental stress. In other words, the 15 sites with the worst "human footprint" scores were placed in the first bin, then the 15 sites with the next best "human footprint" scores were placed in the second bin, and so on, until the last of the 149 bins was reached, which contained the 16 sites with the very best "human footprint" scores. This created a new gradient of 149 bins, characterized by the mean "human footprint" score for the sites within each bin, ranging from most impacted (0) to least impacted (10). For each of several bird species or groups of bird species, including wetland obligates (i.e., bird species that exclusively depend on wetland habitat for breeding) and several informative non-obligate wetland users (i.e., bird species that use wetland habitat, but also use other non-wetland habitats), a stressor response curve (biotic response [BR] function) was estimated from the probabilities of occurrence in each of the 149 bins (Figure 1). The probability values were defined as the proportion of the 15 points where the species was detected during at least one survey at a sample point during a sample year. The environmental condition score for each bin was the mean value for the 15 points. The probabilities of occurrence for each species or species group across the binned environmental stress ("human footprint") gradient can be described by a three-parameter function (μ = mean; σ = standard deviation; and h = scale factor) depicting either a normal bell-shaped or asymptotic curve within the range of 0-10 (e.g., Gnass Giese et al. 2015). We used the best-fit parameters of either the normal bell-shaped curve (following the familiar normal probability density function, R function *dnorm*) or an asymptotic curve (following the cumulative normal distribution, R function *pnorm*). The best-fit parameters were estimated by an R script (R Core Team 2020) using the *nlminb* function developed by Bates and Sarkar (Gay 1990). These BR functions (Figure 1) provided the basis for scoring new sites based on the presence/absence of 15 species/species groups (Table 1). Parameters of the best-fit mathematical function were estimated by computer iteration in R (R Core Team 2020) with package "iec" (https://github.com/ngwalton/iec).

The final suite of 15 BR functions for calculating IECs included wetland obligate species (e.g., Pied-billed Grebe; American Bittern; Marsh Wren, *Cistothorus palustris*), wetland user species (e.g., Bald Eagle, *Haliaeetus leucocephalus*; Great Blue Heron, *Ardea herodias*; Red-winged Blackbird, *Agelaius phoeniceus*), and species groups such as rails, dabbling ducks, and terns. Considering a broad suite of wetland bird species is important because it gives a better representation of overall wetland health. For instance, some disturbance-tolerant marsh user species are more common in degraded wetlands (e.g., European Starling, *Sturnus vulgaris*), whereas some relatively disturbance intolerant wetland obligate species are more common in less degraded wetlands (e.g., Sandhill Crane, *Antigone canadensis*; Figure 1), plus Chin et al. (2015) recommended using indicators that both increase and decrease along the environmental disturbance gradient. Unless included in taxonomic groups (e.g., bitterns and wrens), uncommon species and species rarely found in wetland habitats were excluded, resulting in an indicator metric that directly represents the bird assemblage associated with a coastal wetland. The 15 species or species groups used to generate BR functions for calculating IECs are listed in Table 1.

Geographic ranges of bird taxa used in our analyses extend across the Great Lakes basin, yet local abundances of these taxa are not evenly distributed. For example, large herons (Great Blue Heron and Great Egret, *Ardea alba*) are much more frequent in the southern and eastern Great Lakes than in Lake Superior. Sedge Wrens (*Cistothorus platensis*) are more frequent in the northern lakes. Combining species into multi-species groups (e.g., Sedge Wren + Marsh Wren = Wren; Least Bittern + American Bittern = Bittern) mitigates the effects of some geographic patterns because at least one of the combined species can be expected in any given Great Lakes region. These combined groups enabled us to validly compare IEC estimates across the basin.

Despite our efforts to develop basin-wide IEC estimates, regional differences were still evident in the distributions of our selected taxa. We used Dufrene and Legendre's (1997) indicator analysis to compare frequencies and abundances of the 15 taxa among 4 geographic regions: Lake Ontario (LO); Lake Erie, southern Lake Huron, and southern Michigan (LEsHM); northern Lake Huron and northern Lake Michigan (nLHM); and Lake Superior (LS). All but one taxon (Bald Eagle-Osprey) showed a statistically significant affinity to one or more of these regions. For example, Bittern, Wren, and Dabbling duck were far more frequent in Lake Ontario; European Starling and Heron-Egret were far more frequent in LEsHM; Tern, Sandhill Crane, Rail, and Rare were far more frequent in nLHM; and Common Yellowthroat was significantly more frequent in Lake Superior and nLHM.

In order to compare IEC values without the confounding effects of geographic differences in bird distributions, we applied a second approach to generate the results presented in this report. All 15 taxa were well represented in LEsHM and nLHM so we included the full list of species and species groups for these regions. However, we removed 2 seldom-encountered taxa (Tern and Sandhill Crane) from Lake Ontario and 5 taxa (Tern, Dabbling duck, Heron-Egret, Coot-Gallinule, and Pied-billed Grebe) from the regional IEC analysis for Lake Superior. Results give a regional IEC that uses the same analytical framework but does not "penalize" geographic regions for taxa that are at the margins of their geographic distribution in the Great Lakes basin.

The health of coastal and inland wetlands was subsequently evaluated using GLMMP, GLEI, and CWMP data and the Index of Ecological Condition (IEC), an objective biotic indicator introduced by Howe et al. (2007a, 2007b), improved by Gnass Giese et al. (2015), compared to other similar indices for wetland birds by Chin et al. (2015), and revised for use in the lower Great Lakes by Jung et al. (2020). By recording the species present at a wetland, one can identify the best fit IEC from the previously generated BR functions. The computer-intensive process maximizes a likelihood function consisting of the sum of probabilities (from BR functions) of species or species groups that were detected at a sample point, plus the sum of one minus the probabilities of species or species groups that were not detected (i.e., the sum of one minus the probabilities of not finding species that were absent at the sample point). The presence of species or species groups that have been shown previously to be more common in minimally-stressed wetlands will indicate ecologically healthy conditions and high IEC scores. By contrast, the presence of species or species groups that are more common in highly-stressed wetlands will indicate ecologically unhealthy or degraded conditions and low IEC scores. This method resembles other approaches to environmental indicator development, but the IEC framework establishes an explicit connection between stressors and biotic variables, providing a clear picture about what the indicator truly "indicates." A more detailed description of IEC methodology is available in a separate document (Howe et al. *in prep.*) and at http://www.uwgb.edu/BIODIVERSITY/forest-index/iec.asp.

IECs for each sample point during a given year were based on species detected during one or more of the field visits. Basing IECs on species' presence or absence (rather than counts) is desirable because it minimizes the effects of differences in detectability. Next, the point-level IECs were averaged across all sample points within each wetland or wetland complex in each year, which standardizes wetlands containing differing numbers of sample points. Box-and-whisker plots of these wetland-level mean IECs for coastal wetlands in each basin and throughout the entire Great Lakes basin (hereafter "overall") are reported in each year. These wetland-level means form the basis for the status and trend assessments. IEC metrics were also calculated for inland wetlands for comparison to the Great Lakes coastal wetlands but are not used for any assessments of status or trends. In addition, box-and-whisker plots of wetland-level mean IECs for coastal wetlands in each basin and overall for the most recent years (2019-2020) are reported to illustrate current status. The most recent two years of data were used rather than the most recent year because sample sizes were relatively low in 2020 due to COVID-19 travel restrictions. Density plots of wetland-level mean IECs for coastal and inland wetlands in each basin (but not inland for Lake Superior due to insufficient sample size) based on the most recent 5 years of data (2016-2020) are reported to illustrate variation in distributions. These plots can be thought of as smoothed histograms, which are useful for continuous data that come from an underlying smooth distribution. To further illustrate distributions, these same data for coastal wetlands are mapped throughout each of the basins. In the calculations for the density plots and the map, wetland-level mean IEC values were averaged across years for wetlands that were sampled in multiple years using the most recent 5 years of data (2016-2020) to boost sample sizes. To better illustrate general patterns on the map, the point-to-raster tool in ArcGIS was used to calculate mean values within 3,000 m cells, and the focal statistics tool was used on the resulting means to calculate average values within a 2-cell circular window, which is referred to as "smoothed IEC." Values within 3,000 m cells and a 2-cell circular window were used because through trial and error these values produced maps with desired visual resolution.

Ecological Condition

Data coverage—The dataset available for generating IECs consisted of 43,280 point counts conducted at 5,611 sample points in 1,857 wetlands over 26 years from 1995-2020 throughout coastal and inland areas of the Great Lakes basin (Figure 2). The number of years that each wetland was surveyed varied from 1 to 25, with a mean of 3.9 ± 4.2 (SD), due mostly to large differences in observer participation in the long running, broad scale GLMMP (Figure 2). The majority of the surveyed wetlands were coastal (*n* = 1,265; 68%) rather than inland (*n* = 592; 32%) because both the GLEI and CWMP projects focused entirely on coastal wetlands, whereas the GLMMP surveyed both coastal and inland wetlands (Figure 2).

The number of coastal and inland wetlands surveyed per year (275 ± 136 [mean ± SD]) ranged from 118 to 489, with substantially more wetlands surveyed from 2011-2020 due to the CWMP operating during those years (Figure 3). Annual coverage was higher in Lake Erie and Lake Ontario compared to the upper Great Lakes mostly because GLMMP coverage is more extensive in the lower lakes. Annual coverage was higher at coastal compared to inland wetlands (Figure 3). The number of wetlands available for surveys also varied depending on location, as some sections of the Great Lakes shoreline naturally have no, or almost no, wetlands.

Overall—Median IEC in coastal wetlands in the Great Lakes overall ranged from 1.3 to 4.5 from 1995-2020, significantly increased by 4.9%/year (2.8, 7.0) [lower, upper 95% confidence limits for % annual change] over the past 10 years (2010-2020; Table 2, Figure 4), and was 3.7 (out of 10) based on data from 2019-20 (Figure 5). Based on these patterns, the status of coastal wetland health based on bird communities in the Great Lakes overall is Fair, and the trend is Improving. By contrast, IECs at inland wetlands showed little change across years, and were lower (2.2) compared to coastal wetlands (3.7) based on data from 2019-2020 (Table 2; Figure 4, 7). IECs at coastal wetlands were notably lower throughout the western portion of Lake Ontario where the human development footprint is substantial (Bourgeau-Chavez et al. 2015) compared with most of the rest of the lakes where, generally, a mix of low and high IECs occurred (Figure 6). The status and trend of coastal wetlands in the Great Lakes overall are the same as in the last report (Tozer et al. 2019).

Lake Superior—Median IEC in coastal wetlands ranged from 0.5 to 6.0 from 1995-2020 (Figure 4) and was 2.4 based on data from 2019-2020 (Figure 5). For this report, we concluded that there was not enough information to perform reliable statistical significance testing for trends due to low sample size during 1995-2010, although we note that there appeared to be an increase in median IEC in coastal wetlands over the past decade followed by a decrease in the most recent few years, a pattern also noted for Lake Michigan and Lake Huron (Figure 4). Based on these patterns, the status of coastal wetland health based on bird communities in Lake Superior is Fair, and the trend is Undetermined. IECs at inland wetlands in the Lake Superior watershed were too sparse for meaningful comparison with coastal wetlands (Figure 4). Although landscapes in the coastal zone of Lake Superior are generally non-agricultural and minimally developed compared with wetlands in the more southern lakes (Bourgeau-Chavez et al. 2015), coastal wetlands of Lake Superior (with a few notable exceptions) are relatively small in area, tend to have lower productivity (bogs), and are ecologically quite different compared to the rest of the lakes, which may account at least partially for some of the modest scores in comparison with those from other lakes (Figure 4-7). Since the last report, the status of coastal wetland in Lake Superior has changed from Good to Fair. The difference in status may be due to improved sensitivity of the metrics used to calculate the IEC since the last report, or changes in lake levels, which may lead to dramatic changes in the area and/or quality of coastal wetland bird habitat and associated IECs (see discussion below for further explanation).

Lake Michigan—Median IEC in coastal wetlands ranged from 0.6 to 5.9 from 1995-2020 (Figure 4) and was 2.8 based on data from 2019-2020 (Figure 5). For this report, we concluded that there was not enough information to perform reliable statistical significance testing for trends, although we note that there appeared to be an increase in median IEC in coastal wetlands over the past decade or so followed by a decrease in the most recent few years, a pattern also noted for Lake Superior and Lake Huron (Figure 4). Based on these patterns, the status of coastal wetland health based on bird communities in Lake Michigan is Fair, and the trend is Undetermined. By contrast, IECs at inland wetlands in the Lake Michigan watershed showed little change across years (Figure 4, 7). Some of the highest quality coastal wetlands with respect to birds occur in Lake Michigan, even though development and agricultural stressors are common to extensive in the coastal zones of this lake (Figure 4-7; Allan et al. 2013, Bourgeau-Chavez et al. 2015). For example, the lower IECs at coastal wetlands in the southern third of the lake compared to the rest of the lake could be due to strong development-related stressors (Figure 6).

Since the last report, the status of coastal wetlands in Lake Michigan has changed from Good to Fair. The difference in status may be due to improved sensitivity of the metrics used to calculate the IEC since the last report, or changes in lake levels, which may lead to dramatic changes in the area and/or quality of coastal wetland bird habitat and associated IECs (see discussion below for further explanation).

Lake Huron (including St. Marys River)—Median IEC in coastal wetlands ranged from 1.2 to 7.1 from 1995-2020, did not significantly increase or decrease over the past 10 years (2010-2020; Table 2, Figure 4), and was 4.9 based on data from 2019-2020 (Figure 5). We note that even though the recent 10-year trend (2010-2020) was not significant, there did appear to be an increase in median IEC in coastal wetlands over the past decade or so followed by a decrease in the most recent few years, a pattern also noted for Lake Superior and Lake Michigan (Figure 4). Based on these patterns, the status of coastal wetland health based on bird communities in Lake Huron is Good, and the trend is Unchanging. By contrast, IECs at inland wetlands in the Lake Huron watershed showed little change across years (Table 2; Figure 4, 7). Some of the highest quality coastal wetlands with respect to birds occur in Lake Huron, even though development and agricultural stressors are fairly strong in parts of the coastal zones of this lake (Figure 4-7; Allan et al. 2013, Bourgeau-Chavez et al. 2015). The status and trend of coastal wetlands in Lake Huron are the same as in the last report (Tozer et al. 2019).

Lake Erie (including St. Clair-Detroit River Ecosystem)—Median IEC in coastal wetlands ranged from 0.9 to 5.4 from 1995-2020, did not significantly increase or decrease over the past 10 years (2010-2020; Table 2, Figure 4), and was 4.7 based on data from 2019-2020 (Figure 5). Based on these patterns, the status of coastal wetland health based on bird communities in Lake Erie is Fair, and the trend is Unchanging. Similar trends occurred at inland wetlands in the Lake Erie watershed, although IECs were lower (3.0) compared to coastal wetlands (4.7) based on data from 2019-2020 (Table 2; Figure 4, 7). The status and trend of coastal wetlands in Lake Erie are the same as in the last report (Tozer et al. 2019).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)—Median IEC in coastal wetlands ranged from 0.5 to 4.0 from 1995-2020, significantly increased by 7.5%/year (4.5, 10.5) over the past 10 years (2010-2020; Table 2, Figure 4), and was 3.1 based on data from 2019-2020 (Figure 5). Based on these patterns, the status of coastal wetland health based on bird communities in Lake Ontario is Fair, and the trend is Improving. By contrast, IECs at inland wetlands in the Lake Ontario watershed showed little change across years, and were lower (2.0) compared to coastal wetlands (3.1) based on data from 2019-2020 (Table 2; Figure 4, 7). IECs were lower at coastal wetlands throughout the western portion of the lake compared to the rest of the lake, most likely due to especially strong development-related stressors (Bourgeau-Chavez et al. 2015; Figure 6). The status and trend of coastal wetlands in Lake Ontario are the same as in the last report (Tozer et al. 2019).

Discussion—Throughout the Great Lakes basin, the current status of coastal wetland health based on wetland birds is Fair, with the current status of Lake Superior, Lake Michigan, Lake Erie, and Lake Ontario also being Fair, and only the status of Lake Huron being Good (note that a separate determination of status and trend was made for the Overall Assessment based on data from all of the individual lake basins, rather than averaging across the lake-by-lake assessments). In addition, the status of coastal wetlands was poor throughout the western portion of Lake Ontario, most likely due to especially strong development-related stressors in the region (Bourgeau-Chavez et al. 2015). Despite the predominantly Fair status throughout the Great Lakes basin, it is important to note that high-quality coastal wetlands are present throughout each of the Great Lakes (Figure 4-7). By illustrating and documenting differences in wetland health, the analysis provides a unique baseline for assessing long-term changes in wetland quality and for quantifying the success of restoration efforts in individual wetlands, regions, and the entire Great Lakes basin. A more detailed analysis of species' responses to individual stressors is available but is beyond the scope of this report (see also e.g., Saunders et al. 2019, Tozer et al. 2020). Instead, the condition of wetlands based on a multivariate "human footprint" stressor that incorporates measures of seven stressor variables (including development, agriculture, human population density) was reported.

Throughout the Great Lakes basin, coastal wetland health based on wetland birds significantly increased over the past 10 years (2010-2020), with a significant positive trend in Lake Ontario (Table 2, Figure 4). The cause of recent increases in IECs is unclear, although it may be due to concurrent changes in lake levels for some bird species in some wetlands (e.g., Gnass Giese et al. 2018) or in combination with other factors (see next paragraph). Marsh bird species richness and abundance of many wetland-obligate species decreased significantly with falling lake levels between 1995 and 2002 at coastal wetlands in Lake Michigan, Lake Huron, and Lake Erie (mean decrease in lake level from 1995-2002: 0.06-0.41 m depending on the lake; Timmermans et al. 2008), and then increased significantly with rising lake levels between 2013 and 2018 at coastal wetlands throughout all of the lakes, apparently due to concurrent increases in open water extent and open water-vegetation interspersion (mean increase in lake level from 2013-2018: 0.16-0.97 m depending on the lake; Hohman et al. 2021) (see Water Levels

Sub-Indicator for further details including historical context back to 1918). The increase in marsh birds with increasing lake levels between 2013 and 2018 was especially strong in Lake Michigan and Lake Huron compared to the other lakes (Hohman et al. 2021), which is reflected in the IECs reported here (Figure 4). It appears, therefore, that the overall health and ecological condition of Great Lakes coastal wetlands based on wetland birds is strongly driven by low (generally poor condition) and high (generally good condition) lake levels, although there is considerable variation in this generality across individual wetlands (Hohman et al. 2021). Plus, the true relationship might be more complicated. In Lake Superior, Lake Michigan, and Lake Huron, there was an increase in median IEC in coastal wetlands over the past decade followed by a decrease in the most recent years, even though lake levels have remained high. It is possible that periods of sustained high water eventually result in a reduction in the health of Great Lakes coastal wetlands based on wetland birds due to concurrent reductions in preferred vegetation patches for nesting and feeding (Smith et al. 2021). More years of data documenting the response of the marsh bird community across rising and falling lake levels will help reveal how this wetland health-marsh bird dynamic truly operates.

A contributing factor that may further explain recent increases in coastal IECs over the past 10 years (2010-2020) is the increased survey coverage in coastal wetlands following the implementation of the CWMP starting in 2011 (Figure 3). As the monitoring and sophistication of the IEC analysis program matures, we anticipate that our understanding of such issues as the high variability of water levels and the inherent variability of wetland bird species will improve over time.

Alternatively, the recent improvement in coastal IECs over the past 10 years (2010-2020) may be at least partially due to real improvement in coastal wetland condition. An index of water quality significantly improved across 22 wetlands along the Canadian shore of Lake Ontario between 2003 and 2014 (Croft-White et al. 2017). The improvement in water quality is indicative of concurrent improvement in wetland bird habitat (Grabas et al. 2012, Grabas and Rokitnicki-Wojcik 2015, Tozer 2016) and was reflected in the increasing coastal wetland bird IECs observed in Lake Ontario over the same period.

In addition to assessing status and trend of the health of coastal wetlands, status and trend of inland wetlands were examined for comparison (Table 2; Figure 4, 7). Due to differences in sample sizes, the ability to compare coastal and inland wetlands was best for Lake Erie and Lake Ontario, whereas it was more limited for the other lake basins. It is worth noting that patterns in IECs tended to differ between coastal and inland wetlands. In general, where and when IECs in coastal wetlands increased or decreased, there were no corresponding changes in IECs at inland wetlands (Table 2, Figure 4). In addition, IECs at inland wetlands tended to be lower than IECs at coastal wetlands within any particular lake basin, although this was not always the case (Figure 4). Thus, wetland health as represented by wetland birds may be responding to different intensities of stressors in coastal versus inland wetlands within certain watersheds. This idea is supported by previous analyses using only the GLMMP dataset, which showed that mean abundance of certain wetland-dependent bird species was different at coastal wetlands compared to inland wetlands (Tozer 2013, 2020). Thus, continued sampling of both coastal and inland wetlands throughout the Great Lakes basin will be useful for generating a more complete understanding of the health of wetlands based on birds throughout the entire Great Lakes watershed.

The overall Fair status and Improving trend reported for coastal wetlands throughout the Great Lakes basin contrasts with some previous reports (i.e., before 2019) for this sub-indicator. The differences are at least partially due to differences in sampling. Some of the previous reports were based predominantly on data from the southern portion of the Great Lakes basin due to reliance on the mostly southern GLMMP dataset. The current report provides a more balanced assessment throughout the entire Great Lakes basin by bringing GLMMP data together with data from the southern and northern GLEI and CWMP projects. Nevertheless, the patterns reported here are based on a comprehensive IEC metric, which represents the collective responses of many wetland bird species to wetland condition. Yet many species, such as bitterns (e.g., *Botaurus*), shallow- (e.g., *Porzana*) and deep-water rails (e.g., *Gallinula*), and marsh-nesting terns (e.g., *Chlidonias*), have experienced long-term declines in populations in the Great Lakes (e.g., Tozer 2013, 2016, 2020). These taxa may be responding to environmental stressors in species-specific ways, warranting unique actions or presenting unique opportunities for restoration to improve wetland health.

Linkages

Coastal wetland birds are influenced by numerous local and landscape-level characteristics, some of which are monitored by other State of the Great Lakes indicators. For instance, coastal wetland birds are known to be influenced by changing water levels at local and individual Great Lakes basin scales (e.g., Timmermans et al. 2008, Jobin et al. 2009, Gnass Giese et al. 2018, Hohman et al. 2021). Thus, the Coastal Wetland Birds sub-indicator can be expected to co-vary with the Water Levels sub-indicator (e.g., Chin et al. 2014; see discussion above for details). As well, the Coastal Wetland Birds sub-indicator can be expected to co-vary with sub-indicators that track the extent and spatial arrangement of wetland bird habitat (e.g., Coastal Wetlands: Extent and Composition, Aquatic Habitat Connectivity) and prey (Coastal Wetland Invertebrate; Coastal Wetland Fish). It can also be expected to co-vary with invasive plant species (e.g., *Phragmites australis*) that encroach upon preferred native vegetation (e.g., Aquatic and Terrestrial Non-Native Species) and pollution that may reduce prey abundance and/or availability (e.g., all of the Toxic Chemicals and Nutrients and Algae indicators). Finally, it can be expected to be influenced by climate-related impacts and associated interactions with other factors, such as through direct mortality of certain coastal wetland bird species as a result of climate-related outbreaks of type-E botulism (Princé et al. 2018) and through indirect climate-related changes in water levels that in turn influence wetland bird habitat (citations above). The Coastal Wetland Birds sub-indicator can be expected to co-vary with the *Cladophora* sub-indicator (rotting mats of algae enhanced by climate-related factors are known to be starting points for botulism; Chun et al. 2015), the Dreissenid Mussels sub-indicator (beds of these mussels enhanced by climate-related factors are also known to be starting points for botulism; Getchell and Bowser 2006), and with all of the Climate Trends indicators (including Water Levels, Surface Water Temperature, Ice Cover, and Amount of Precipitation). And finally, the occurrence of multiple species of wetland birds is known to be lower in developed and urban landscapes (Saunders et al. 2019, Smith and Chow-Fraser 2010a, Tozer et al. 2020), as reflected in the pattern of IECs based on wetland bird communities presented here (Figure 6). Thus, the Coastal Wetland Birds sub-indicator can be expected to co-vary with the Human Population sub-indicator.

Traditional Ecological Knowledge (TEK), Citizen Science and Other Bodies of Knowledge

The tremendous power to report robust status and trends for this sub-indicator are made possible by a rigorous standardized bird count protocol, large sample sizes, and extensive survey coverage accomplished mainly by two bird monitoring programs operating throughout the Great Lakes. The Great Lakes Marsh Monitoring Program of Birds Canada, which draws upon the dedication and skills of hundreds of volunteer surveyors annually, provided much of the data used here from 1995 to 2011, as well as all of the inland data used for comparison to coastal wetlands (Tozer 2013, 2016, 2020). Central Michigan University's Great Lakes Coastal Wetland Monitoring Program has provided much needed expansion of coverage in coastal wetlands particularly in the northern portion of the Great Lakes since 2011 (Uzarski et al. 2017). Ongoing support of both of these programs will ensure that these quality data continue to be available for understanding the condition of the health of the Great Lakes ecosystem.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: CWMP and GLEI: https://www.greatlakeswetlands.org/Home.vbhtml GLMMP: https://www.birdscanada.org/birdmon/default/main.jsp		

Data Limitations

This sub-indicator may be more powerful if evaluated as part of an overall analysis of biological communities of Great Lakes coastal wetlands and nearshore aquatic systems. This can be done by considering the coastal wetland sub-indicators in combination because they function and indicate anthropogenic disturbance at different spatial and temporal scales and have varying resolution of detection. However, the geographic scale of disturbance for coastal wetland birds is not yet determined, nor is the resolution for detecting ecosystem health. This is a fruitful area for further development.

Great Lakes coastal wetlands are diverse. Some are protected from wind and wave action; some are open to the lake. Some are associated with river mouths, others are not. Additional variation is imposed by underlying geomorphology and climate differences across different regions. This variation in wetland types carries over to influence wetland birds. Thus, the most powerful data for estimating bird-based indices of ecosystem health at broad scales, such as across individual Great Lakes basins, are those collected via spatially-balanced randomized experimental designs, which account for and capture this variation. Most, but not all, of the existing wetland bird datasets suitable for use with this sub-indicator address these sampling issues in their experimental designs, although there is room for improvement.

Additional Information

Over half of all Great Lakes coastal wetlands have been lost or changed by human activities and many remaining coastal wetlands suffer from anthropogenic stressors, such as nutrient and sediment loading, fragmentation, invasive species, shoreline alteration, and water-level control, as documented by a binational Great Lakes-wide mapping and attribution project (Albert and Simonson, 2004; Ingram and Potter, 2004). Such wetland loss and stress are especially apparent in certain portions of the Great Lakes basin (Allan et al. 2013, Bourgeau-Chavez et al. 2015). Indeed, it was found that IECs based on bird community data were especially poor throughout the western portion of Lake Ontario, as well as in other locations distributed throughout each of the Great Lakes where stress due to human development is substantial.

One approach to identifying appropriate actions and opportunities for improving the health of coastal wetlands is to identify factors that will promote colonization or reduce extinction or extirpation of wetland bird species that are decreasing over time. Factors related to colonization and extinction in this manner then translate into actions and opportunities that will help increase occurrence of declining species and ultimately improve wetland health. This approach has been completed using the GLMMP component of the larger dataset analyzed in this report. Using multi-season site occupancy models and data from 21,546 GLMMP point counts conducted at 2,149 sample points, Tozer (2016) determined important local, wetland, and landscape-scale factors influencing occupancy of 15 marsh bird species in wetlands throughout the southern portion of the Great Lakes basin. The results of the study suggest that most decreasing marsh-dependent bird species benefit from conserving, restoring, or creating large wetlands surrounded by limited urban land use, and from addressing issues within Great Lakes Areas of Concern (Tozer 2016). In addition, individual or smaller groups of decreasing species also benefit from conserving, restoring, or creating robust emergent-dominated but interspersed, purple loosestrife (*Lythrum salicaria*)-free, *Phragmites*-free wetlands surrounded by higher proportions of wetland cover in the surrounding landscape, and from addressing issues within Great Lakes coastal wetlands (Tozer 2016). These actions will promote colonization or reduce extinction and help slow or perhaps reverse declining trends in site occupancy among decreasing species across the southern portion of the Great Lakes basin. These actions will also ultimately help improve associated coastal wetland health across the southern portion of the Great Lakes basin.

The status and trend assessment of coastal wetland health based on wetland birds is based on BR functions developed using CWMP data only. The BR functions were also developed based on information from multiple stressor gradients related to development, agriculture, and human population density. The ability of the IEC to capture the health of coastal wetlands based on bird data might be improved by expanding the development of the BR functions to include all of the marsh bird data that are available from the GLMMP, GLEI, and CWMP projects. The performance of the IEC might also be improved by incorporating other known wetland bird stressors in the development of BR functions, particularly within-wetland attributes like relative dominance of invasive plant species such as *Phragmites* and *Typha* spp. These ideas are fruitful areas for future expansion.

Three large marsh bird datasets were brought together, specifically the GLMMP, GLEI, and CWMP datasets, to perform the analyses summarized in this report. This provided tremendous analytical power at many different scales compared to using only one of the datasets. However, it was evident that the combined dataset is deficient in information from healthy wetlands. Future collection of bird data from wetlands located towards the pristine end of the degraded-pristine gradient might improve the performance of the IEC.

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Table 1. Wetland bird species and groups of species (*n* = 15) used to generate biotic response functions for calculating the index of ecological condition (IEC) as an estimate of the health of Great Lakes wetlands. LEsHM = Lake Erie, southern Lake Huron, and southern Michigan; LO = Lake Ontario; LS = Lake Superior; nLHM = northern Lake Huron and northern Lake Michigan. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

No.	Taxon	Species (scientific name)	Region
1	Bittern	American Bittern (*Botaurus lentiginosus*) and Least Bittern (*Ixobrychus exilis*)	All
2	Common Yellowtroat	Common Yellowthroat (*Geothlypis trichas*)	All
3	Dabbling duck	Dabbling (marsh) ducks (*Anas* spp., *Mareca* spp., *Aix sponsa*), excluding Mallard (*Anas platyrhynchos*).	LEsHM, LO, nLHM
4	Eagle-Osprey	Bald Eagle (*Haliaeetus leucocephalus*) and Osprey (*Pandion haliaetus*)	All
5	European Starling	European Starling (*Sturnus vulgaris*)	All
6	Heron-Egret	Great Blue Heron (*Ardea herodias*) and Great Egret (*Ardea alba*). Note: not included in Lake Superior analyses.	LEsHM, LO, nLHM
7	Coot-Gallinule	American Coot (*Fulica americana*) and Common Gallinule (*Gallinula galeata*)	LEsHM, LO, nLHM
8	Pied-billed Grebe	Pied-billed Grebe (*Podilymbus podiceps*).	LEsHM, LO, nLHM
9	Rail	Sora (*Porzana carolina*), Virginia Rail (*Rallus limicola*), King Rail (*Rallus elegans*), and Yellow Rail (*Coturnicops noveboracensis*)	All
10	Rare	Rare/seldom recorded marsh obligates: Wilson's Snipe (*Gallinago delicata*), Northern Harrier (*Circus hudsonius*), Black-crowned Night Heron (*Nycticorax nycticorax*)	All
11	Red-winged Blackbird	Red-winged Blackbird (*Agelaius phoeniceus*)	All
12	Sandhill Crane	Sandhill Crane (*Antigone canadensis*).	LEsHM, LS, nLHM
13	Swamp Sparrow	Swamp Sparrow (*Melospiza georgiana*)	
14	Tern	Black Tern (*Chlidonias niger*), Common Tern (*Sterna hirundo*), and Forster's Tern (*Sterna forsteri*).	LEsHM, nLHM
15	Wren	Marsh Wren (*Cistothorus palustris*) and Sedge Wren (*Cistothorus platensis*)	All

Table 2. Long-term (1995-2020) and 10-year (2010-2020) trends in median index of ecological condition (IEC) throughout coastal and inland wetlands of the entire Great Lakes (overall) and within each basin (e.g., Superior). Shown are geometric mean rates of change (Trend: %/year) based on differences between the first and last year in the trend, along with "lower" and "upper" 95% confidence limits. Instances where sample sizes were too low to calculate reliable trends are shown with dashes. Trends calculated using equation 4 in Smith et al. (2014) with 1,000 bootstrap replicates. Statistically significant trends are shown in bold, based on confidence intervals that do not overlap zero. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

Wetland type	Basin	Start year	End year	Trend	Lower	Upper
Coastal	Overall	**1995**	**2020**	**0.9**	**0.0**	**1.8**
		2010	**2020**	**4.9**	**2.8**	**7.0**
	Superior	1995	2020	–	–	–
		2010	2020	–	–	–
	Huron	1995	2020	–	–	–
		2010	2020	4.1	-2.9	10.9
	Michigan	1995	2020	–	–	–
		2010	2020	–	–	–
	Erie	1995	2020	0.7	-0.4	1.9
		2010	2020	3.2	-0.3	6.7
	Ontario	**1995**	**2020**	**2.4**	**0.4**	**4.5**
		2010	**2020**	**7.5**	**4.5**	**10.5**
Inland	Overall	1995	2020	0.1	-1.1	1.4
		2010	2020	2.0	-1.0	5.0
	Superior	1995	2020	–	–	–
		2010	2020	–	–	–
	Huron	1995	2020	1.3	-1.3	3.8
		2010	2020	0.5	-5.8	6.8
	Michigan	1995	2020	–	–	–
		2010	2020	–	–	–
	Erie	1995	2020	2.1	-1.1	5.3
		2010	2020	8.2	-0.2	16.7
	Ontario	1995	2020	-1.6	-3.6	0.3
		2010	2020	-0.3	-4.2	3.7

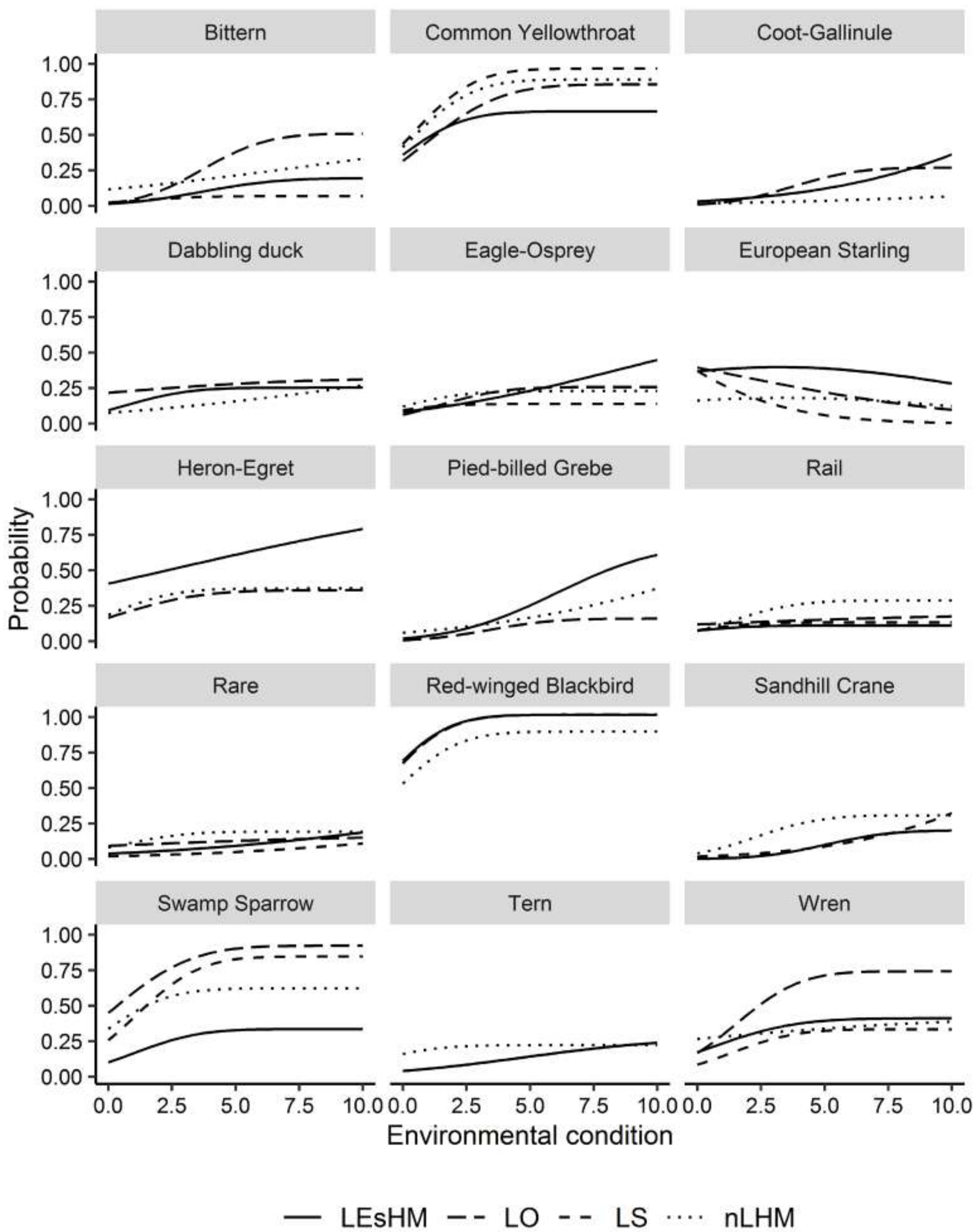

Figure 1. Biotic response functions (lines) for bird species and groups of species based on data from coastal wetlands throughout the Great Lakes basin, and used for the purpose of estimating bird-based indices of wetland health. Shown is the probability of occurrence as a function of a combined "human footprint" variable (Environmental condition) based on wetland area, development, agriculture, and human population density (0 = Poor condition, 10 = Good condition). See Table 1 for lists of species that comprise groups and for scientific names. Biotic response functions are shown for multiple regions: LEsHM = Lake Erie, southern Lake Huron, and southern Lake Michigan; LO = Lake Ontario; LS = Lake Superior; and nLHM = northern Lake Huron and northern Lake Michigan. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

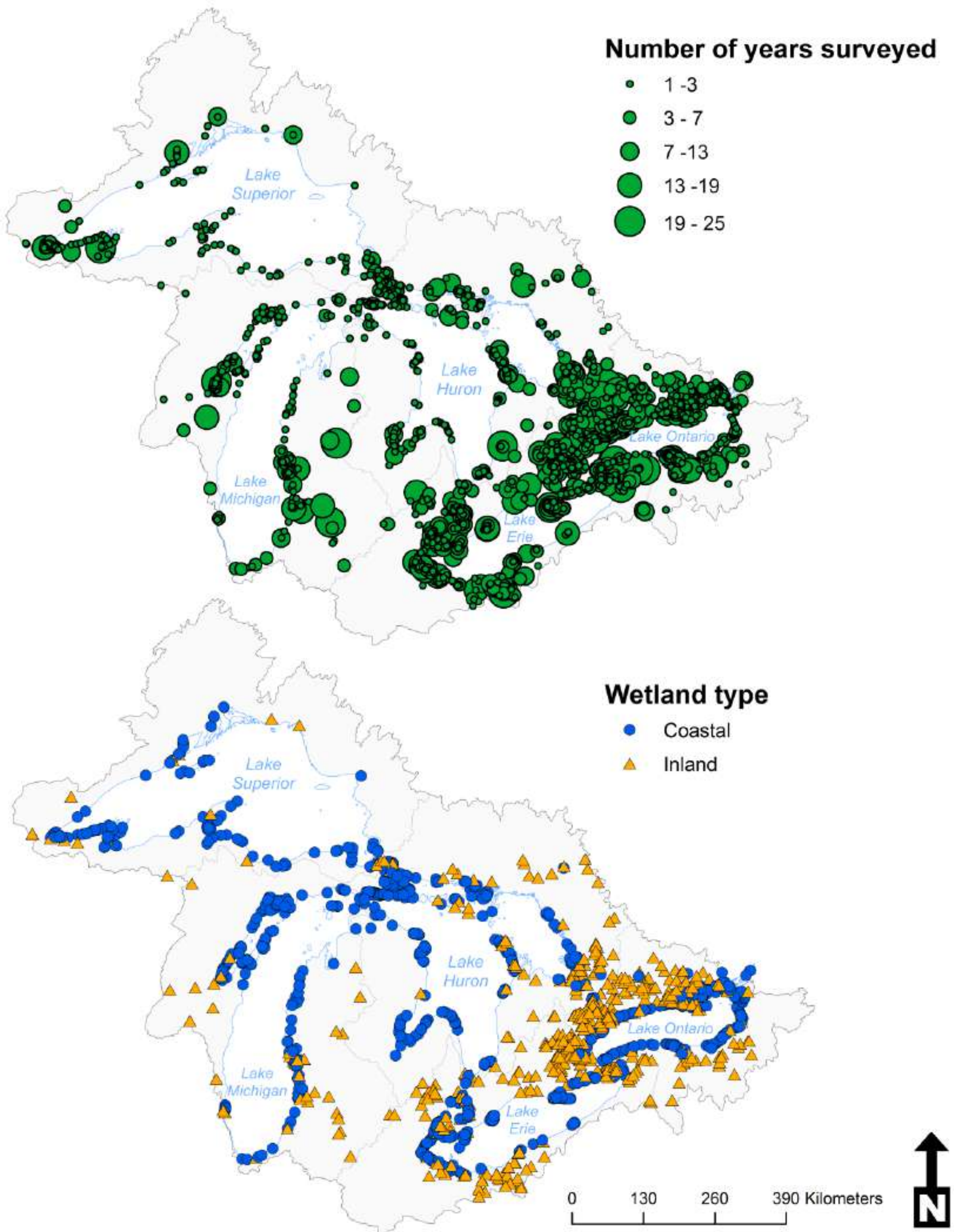

Figure 2. Wetlands surveyed for birds from 1995-2020 throughout the Great Lakes basin for the purpose of estimating bird-based indices of wetland health. Shown are wetlands as a function of the number of years that each wetland was surveyed (top) and as a function of coastal versus inland (bottom). Note that coastal wetlands (*n* = 1,265) far outnumber inland wetlands (*n* = 592), although this does not appear to be the case due to tightly overlapping symbols. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

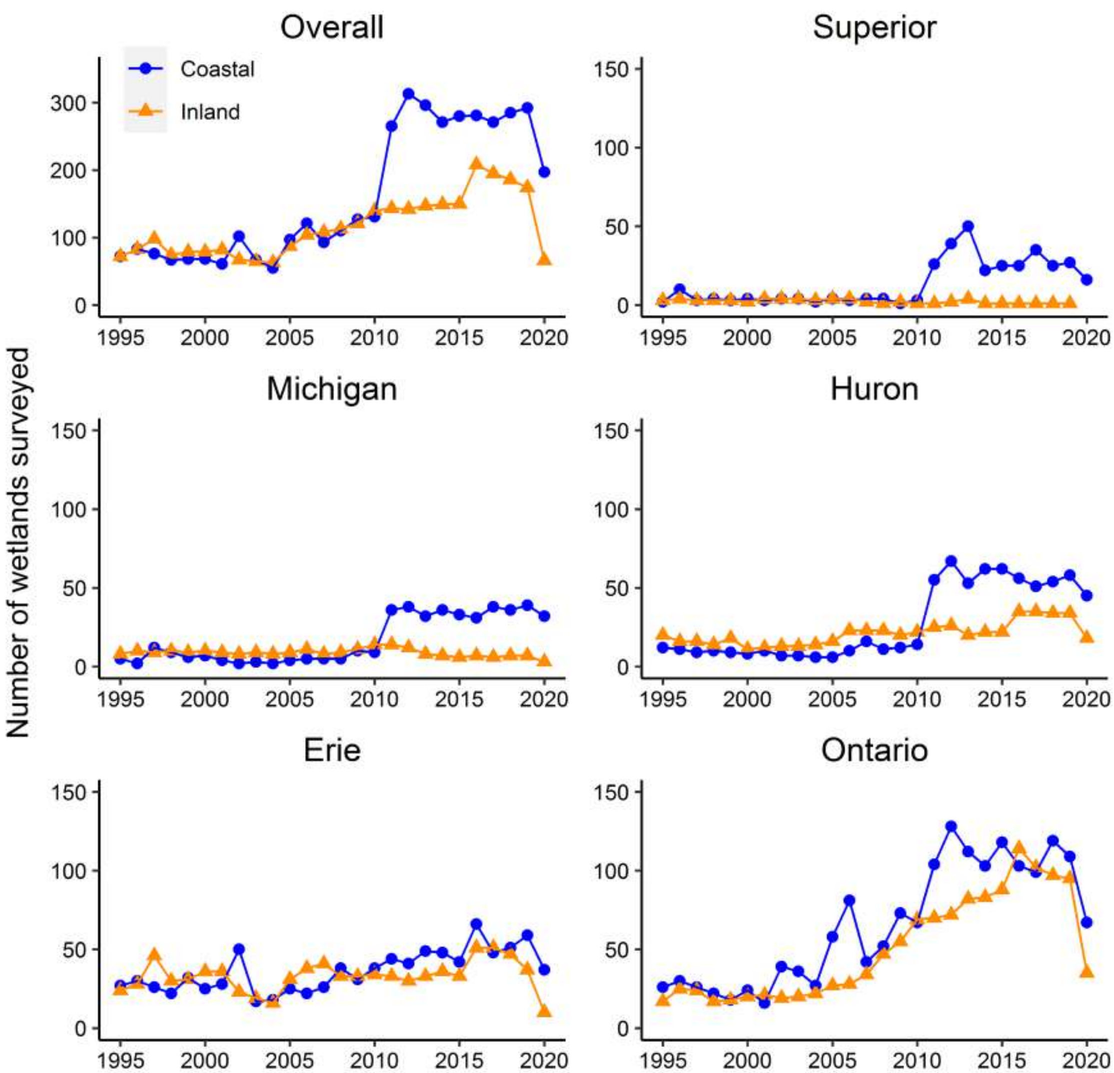

Figure 3. Number of wetlands surveyed for birds per year from 1995-2020 throughout the Great Lakes basin for the purpose of estimating bird-based indices of wetland health. Shown are wetlands surveyed as a function of the entire Great Lakes basin (overall) and each individual lake basin for coastal and inland wetlands. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

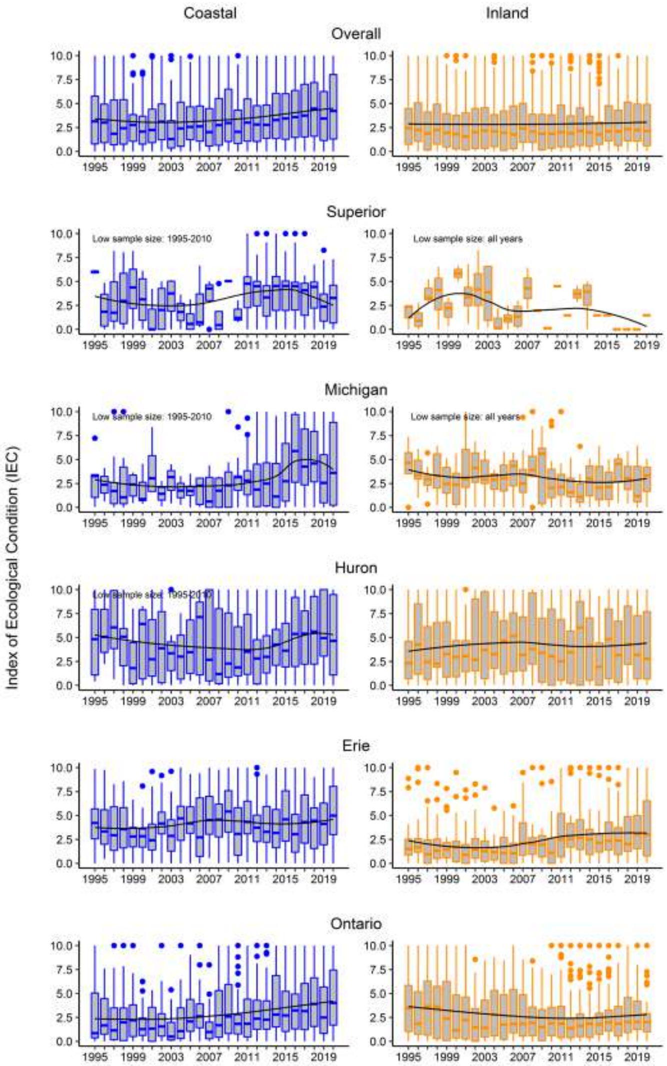

Figure 4. Temporal trends in the index of ecological condition (IEC) based on bird community data from 1995-2020 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal (blue) and inland (orange) wetlands. Shown are box-and-whisker plots of wetland-level means in each year along with loess smoother lines of best fit (solid black lines). Whiskers represent 1.5 times the inter-quartile range; dots are outliers. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

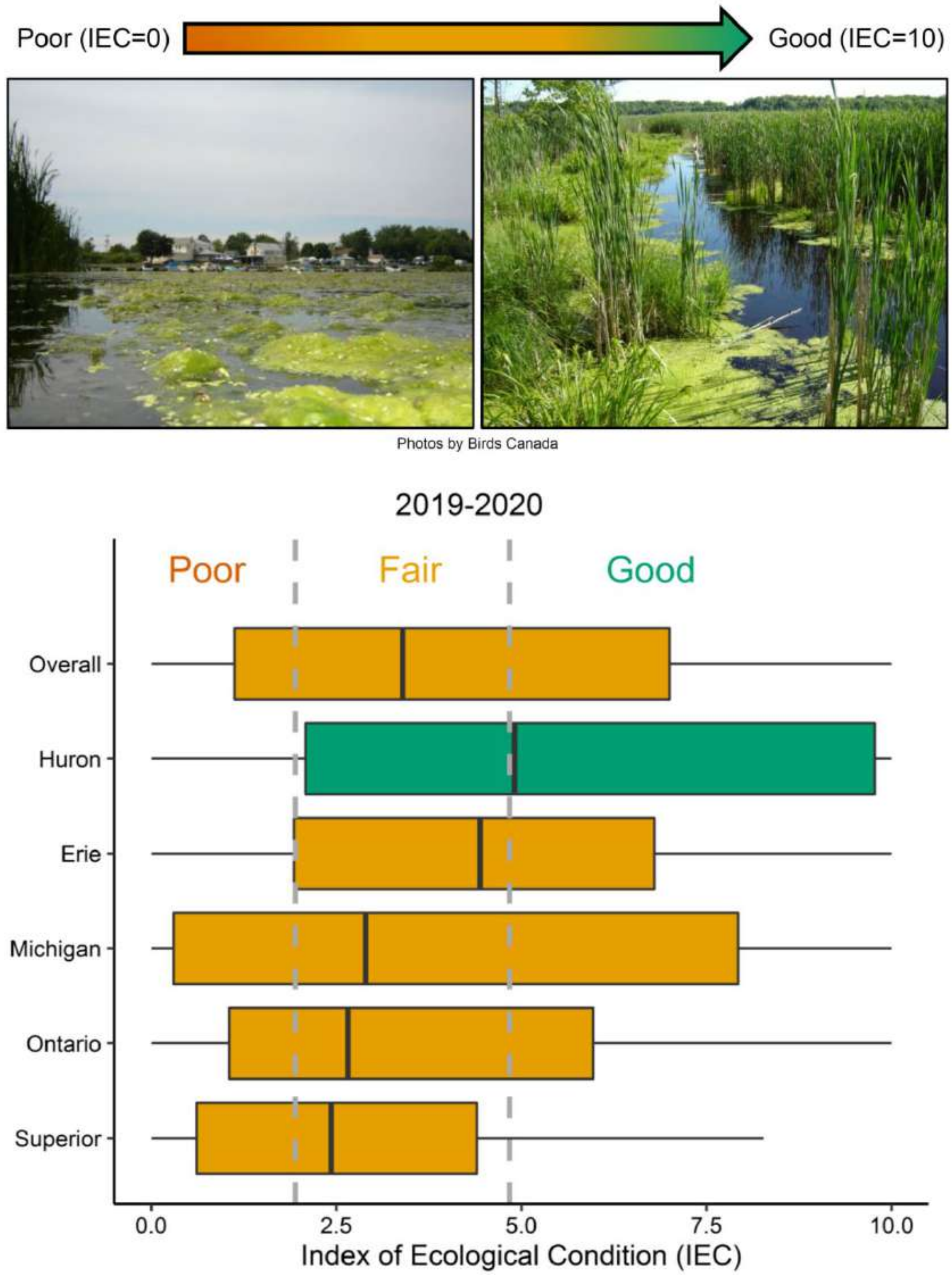

Figure 5. Distribution of the index of ecological condition (IEC) based on bird community data in 2019-2020 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal wetlands. Shown are box-and-whisker plots of wetland-level means. Whiskers represent 1.5 times the inter-quartile range. Vertical dashed grey lines show defining values for Poor, Fair, and Good status. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program.

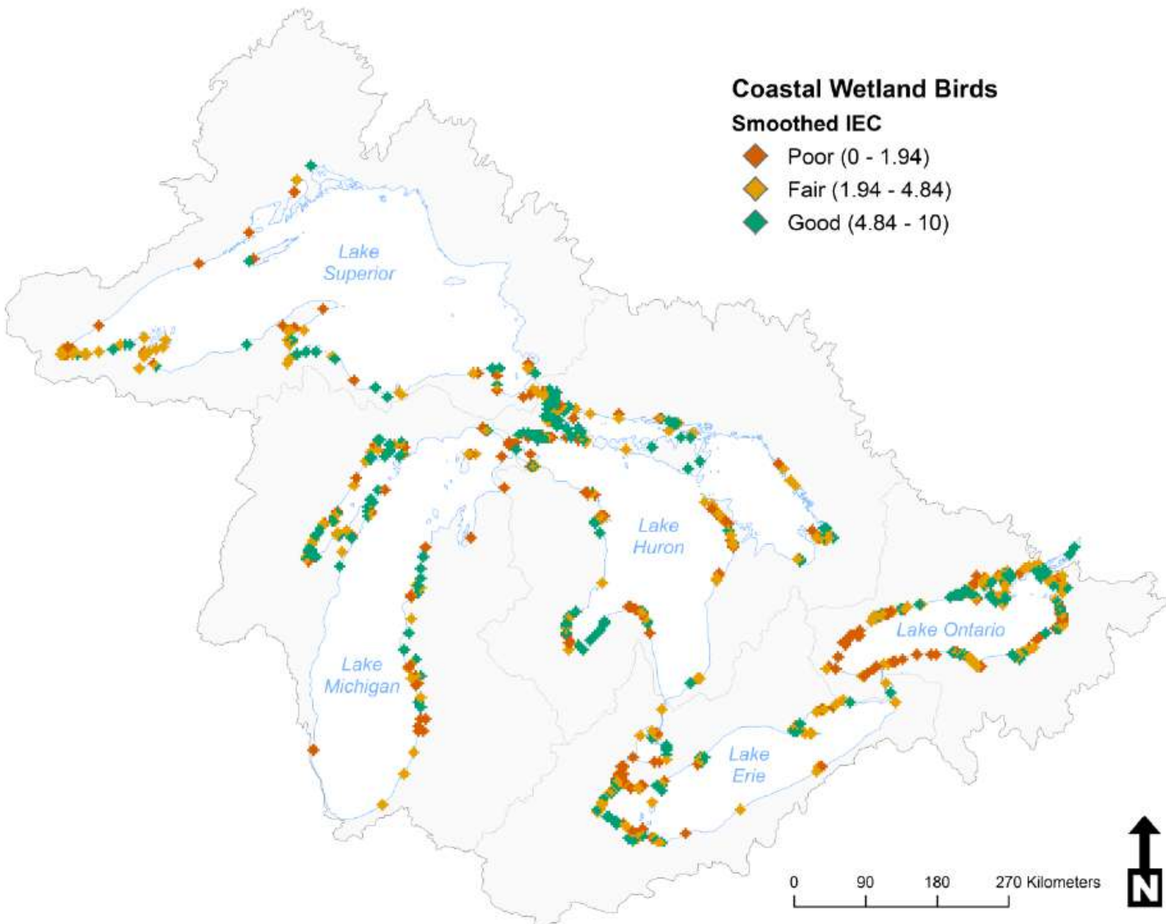

Figure 6. Index of ecological condition (IEC) throughout coastal wetlands of the Great Lakes based on bird community data from the past 5 years (2016-2020). Shown are smoothed wetland-level mean IEC values, averaged across years for wetlands that were sampled in multiple years. To smooth the data, the point-to-raster tool in ArcGIS was used to calculate mean values within 3,000 m cells, and the focal statistics tool was used on the resulting means to calculate average values within a 2-cell circular window, which is referred to as "smoothed IEC." The most recent 5 years of data were used to boost sample sizes. Defining values for colours correspond to Poor, Fair, and Good status. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program

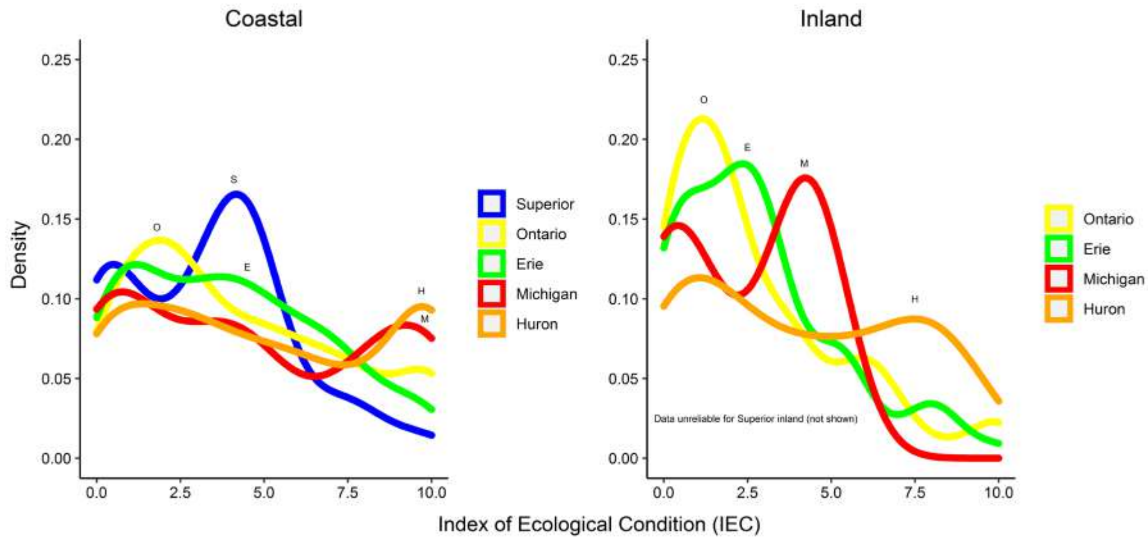

Figure 7. Density plots of the index of ecological condition (IEC) based on bird community data from the past 5 years (2016-2020) throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal and inland wetlands. Shown are probability density functions based on kernel density estimates, which can be thought of as smoothed histograms. Density plots are useful alternatives to histograms for continuous data that come from an underlying smooth distribution, as is the case here. Shown are plots based on wetland-level mean IEC values, averaged across years for wetlands that were sampled in multiple years. Inland data for Superior are not shown due to small sample sizes. Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator Project, Great Lakes Coastal Wetland Monitoring Program

Sub-Indicator: Coastal Wetland Plants

Overall Assessment

Status: Moderately Degraded **(**Fair) [NOTE: We have modified our 3 status categories to 6, with Good equaling Reference and Mildly Impacted, Fair equaling Moderated Impacted to Moderately Degraded, and Poor equaling Degraded and Extremely Degraded. See Table 1 for a summary of scores and categories.]

Trends:

10-Year Trend (2011-2019)*: Unchanging

Rationale: Trends in this report are based on simplified index of biotic integrity (IBI) scores based on a combination of invasive plant dominance and Mean C scores (Albert, 2008; see more detailed description under the heading Measure below) (Table 1, Figure 1), although weighted Mean C (wC) (Bourdaghs et al., 2006) was also calculated for comparison (Table 2, Figure 2); both plant community measures were calculated from the Coastal Wetland Monitoring Program** inventory data collected between 2011 and 2019. Mean C (Herman et al., 2001), included in earlier reports, is not provided, as the Mean C score alone does not provide insights into status of wetlands with abundant invasive species. Status of the coastal wetland plant community for the entire Great Lakes is Fair (Moderately Degraded = 2.58), based on this simplified IBI (Albert, 2008), and Fair (Moderately Degraded = 3.98) based on wC. The simplified IBI scores based on the combination of invasive plant dominance and Mean C (Albert, 2008) express a broader range of conditions than wC scores (see Tables 1 and 2). On average, wetlands in Lakes Huron, Michigan, and Superior generally harbor fair [Moderately Degraded to Moderately Impacted] or good [Mildly Impacted to Reference] wetland plant communities with lower numbers of poor [Degraded to Extremely Degraded] sites. Wetlands in Lakes Erie and Ontario tend to be of more uniformly poor quality [Degraded to Extremely Degraded], with only scattered high quality fair [Moderately Degraded to Moderately Impacted] or good [Mildly Impacted to Reference] sites

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake-by-Lake Assessment

Lake Superior

Status: Good [Mildly impacted]

10-Year Trend (2011-2019): Unchanging

Rationale: Lakewide average values for IBI (2011-2019) are in the 'Good' [Mildly Impacted] category with score of 3.80. Lakewide average wC category is 'Fair' [Moderately Impacted = 5.42]. Sixty-two percent of the surveyed wetland sites in Lake Superior have overall site scores categorized as Good, 35% Fair, while Poor quality sites near urban centers comprise roughly 3% of the wetlands sampled. IBI scores range from 1.5 to 5.0. Individual Lake Superior wetland IBI scores are mapped and summarized in Figure 3. The highest quality wetlands in Lake Superior tend to be barrier-protected poor fens (poor fens are acidic rather than basic), with these wetlands supporting habitat specialists with high conservatism values.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Michigan

Status: Fair [Moderately Impacted]

10-Year Trend (2011-2019): Unchanging [Moderately impacted to Moderately degraded]

Rationale: Lakewide average values for IBI are in the 'fair' [Moderately Impacted] category with score of 2.60. Lakewide average wC category is 'Fair' [Moderately Degraded =3.84]. Among all Great Lakes, Lake Michigan has the second widest distribution of site scores across the gradients, ranging from IBIs of 1.2 to 4.8. Sixty-eight percent of site IBI scores are fair [Moderately Degraded to Moderately Impacted], 24% are good [Mildly Impacted to Reference], and 12% are poor [Degraded]. Individual Lake Michigan wetland IBI scores are mapped and summarized in Figure 4. The higher quality wetlands include open lacustrine, riverine, and barrier-protected wetlands in the northern part of the lake and are associated with surrounding forest cover. Riverine wetlands, especially those in the south with extreme urban and agricultural nutrient enrichment, are among those with the lowest quality. Many wetlands in the Green Bay, WI region have experienced similar severe wetland degradation resulting from long-term agricultural and urban nutrient enrichment and more recent low water levels and associated invasion by reed (*Phragmites australis*). Restoration efforts in this region are improving wetland plant condition. High Lake Michigan water levels in 2014 through 2019 have resulted in erosion of wetland vegetation from the more exposed open lacustrine marshes.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Huron (including St. Marys River)

Status: Fair [Moderately Impacted]

10-Year Trend (2011-2019): Unchanging [Moderately Impacted]

Rationale: Lakewide average values for IBI are in the 'Fair' [Moderately Impacted] category with score of 3.21. Lakewide average wC category is 'Fair' [Moderately Impacted = 4.91]. Wetlands in Lake Huron occur across a wide gradient in plant community condition; IBI scores include 44% Good [Reference and Mildly impacted], 52% Fair [Moderately impacted and Moderately degraded], and 4% Poor [Degraded]. Individual Lake Huron and St. Marys River wetland IBI scores are mapped and summarized in Figure 5. Sites in the northern and eastern portion of Lake Huron tend to be higher quality barrier-protected, lacustrine, and riverine wetlands that reflect surrounding forest cover and management. In Saginaw Bay and on the southern shore of Lake Huron, extensive plowing, raking, and mowing during recent low water periods has led to vast areas of native wetland vegetation in open lacustrine wetlands being replaced by *Phragmites australis* and *Typha x glauca*. This long-term change was documented by observed changes between surveys conducted in the mid-1990s and those conducted between 2011-2015 (Carson et al., 2018). During the recent extended low-water conditions, *Phragmites australis* has expanded lakeward beyond native emergent vegetation on Ontario's Bruce Peninsula and eastern shoreline of Lake Huron, although perhaps recent high water conditions will erode these extensive *Phragmites* beds, although erosion of *Phragmites* beds has not been observed on Saginaw Bay. Loss of emergent vegetation has also occurred in wetlands bordering the St. Marys River, the connecting river between Lakes Superior and Huron during the 1999 to 2013 low-water conditions, probably the result of both winter ice and ship wakes on exposed sediments and vegetation beds. This long-term change is based on surveys conducted in the late 1980s, mid 1990s (summarized in Minc 1997), and between 2011 and 2015. Another major invasive plant that now occupies large stretches of Saginaw Bay and the St. Marys River is European frogbit (*Hydrocharis morsus-ranae*), which has continued to expand coverage even in recent high-water years (Monks et al., 2019). Wetlands in eastern Georgian Bay are susceptible to nutrient enrichment from runoff through shallow soils or on exposed bedrock; in this area, increasing pressures from development and changing water levels are expected to have the greatest impacts in the near future.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor [Degraded]

10-Year Trend (2011-2019: Unchanging [Degraded]

Rationale: Lakewide average values for IBI are in the 'Poor' [Degraded] category with score of 1.47. Lakewide average wC category is 'Poor' [Degraded = 2.60]. IBI scores are among the lowest in the Great Lakes, ranging from 0.5 to 3.1. No sampled sites were in the 'good' [Mildly Impacted to Reference] category on the basis of IBI scores, while 29% were considered Fair [Moderately Degraded to Moderately Impacted], and 71% Poor [Extremely Degraded to Degraded]. Individual Lake Erie and St. Clair and Detroit River wetland IBI scores are mapped and summarized in Figure 6. Some of the higher quality sites are Presque Isle, Pennsylvania, and several large Ontario sites along the north shore, including Long Point, Turkey Point, Rondeau, and Point Pelee, while restoration activities have recently improved Metzger Marsh, Ohio. Overall, the coastal wetland plant communities of Lake Erie are also classified as deteriorating based on historical data from 1975 in Lake Erie (Stuckey, 1989). In Lake Erie, riverine wetlands have slightly lower average quality than barrier-protected or lacustrine wetlands.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Poor [Degraded]

10-Year Trend (2011-2019): Unchanging [Degraded]

Rationale: Lakewide average values for IBI are in the 'poor' [Degraded] category with score of 1.60. Lakewide average wC category is 'poor' [Degraded = 2.76]. Only one Lake Ontario coastal wetland is within the 'good' [Mildly Impacted or Reference] category based on IBI scores, while 43% are considered fair [Moderately degraded or Moderately impacted] and 57% poor [Extremely Degraded or Degraded]. Low wC and IBI scores are a result of the dominance by invasive cattails (*Typha angustifolia* and *Typha* x *glauca*) and other invasive species. Individual Lake Ontario and Niagara and Upper St. Lawrence River wetland IBI scores are mapped and summarized in Figure 7. There is a slight increase west to east in condition, due largely from high levels of urbanization in the western portion of the basin.

* Note: A 9-year trend will serve as an estimate for the 10-year trend as only 9 years of data have been collected.

Status Assessment Definitions

Lake Assessment Scale for IBI

Reference (Good): 4.2-5.0

Mildly Impacted (Good): 3.4-4.19

Moderately Impacted (Fair): 2.60-3.39

Moderately Degraded (Fair): 1.70-2.59

Degraded (Poor): 0.80-1.69

Extremely Degraded (Poor): 0.0-0.79

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Lake Assessment Scale for wC

Reference (Good): 6.91-8.30

Mildly Impacted (Good): 5.53-6.90

Moderately Impacted (Fair): 4.15-5.52

Moderately Degraded (Fair): 2.77-4.14

Degraded (Poor): 1.39–2.76

Extremely Degraded (Poor): 0.0-1.38

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Note: At this time, the assessment scales used to define categories Good, Fair, and Poor are also based on best professional judgment. Work has begun to develop defensible scales that incorporate statistical analysis.

Trend Assessment Definitions

Improving: IBI scores per lake have increased by at least 0.5 unit.

Unchanging: IBI scores per lake have experienced low amounts of natural variation within a range of -0.5 to + 0.5 units.

Deteriorating: IBI scores per lake have decreased by at least 0.5 unit.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend. Trends cannot be determined because of the lack of pre-existing benchmark data of the measures. Because we have recently gathered benchmark and lake-wide data for all of the Great Lakes, we should be able to limit the use of the undetermined category in the future.

Note: At this time, the assessment scales used to define categories Improving, Unchanging and Deteriorating are also based on best professional judgment. Work has begun to develop defensible scales that incorporate statistical analysis.

Endpoints and/or Targets

Values of the two wetland plant quality measures (wC and IBI) are known to be affected by several natural factors, including wetland geomorphic type, lake, and geographical location in addition to human effects (Albert et al., 2005; Brazner et al., 2007). Statistical analyses to parse these components of variation and develop assessment endpoints using data from the Coastal Wetland Monitoring Program (CWMP) are in progress and not completed at this time.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the quality of the vegetation as an integral component of the condition of coastal wetlands.

Ecosystem Objective

Coastal wetlands throughout the Great Lakes basin are influenced by coastal manipulations and the input of sediments, nutrients, and pollutants. About half of coastal wetlands have been lost basin-wide since pre-settlement. Remaining wetlands should be dominated by native vegetation with low numbers of invasive plant species at low levels of coverage. Conservation of these wetlands and restoration of previously destroyed wetlands are vital components of restoring the Great Lakes ecosystem and this sub-indicator can be used to report progress toward such objectives.

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement that states the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species."

Measure

This sub-indicator incorporates information on the presence, abundance, and diversity of aquatic macrophytes within Great Lakes coastal wetlands.

As part of the basin wide CWMP 2011-2020, aquatic plants from each wetland are sampled from three transects perpendicular to depth contours crossing wetland vegetation zones during July and August. The number of vegetation zones varies depending on each particular wetland. Operationally-defined vegetation zones are wet meadow, emergent vegetation, and submergent vegetation. Plant abundance data collected in 1-m^2 quadrats along the transects are used to calculate three measures of wetland plant quality for the entire site:

1. Mean Coefficient of Conservatism (*C*)
2. Weighted Mean Coefficient of Conservatism (*wC*)
3. Vegetation Index of Biological Integrity (IBI)

The first two measures, C and wC are based on the concept of Coefficient of Conservatism developed by Swink and Wilhelm (1994). Plant species are assigned a C-score 0-10 to reflect their specificity to natural, undisturbed habitats of a region. Ubiquitous species are assigned a low score, while species rare and only found in specific habitats are assigned a high score. Scores were assigned by Herman et al. (2001) for the Great Lakes basin region and recently updated by Reznicek et al. (2014). For an overall wetland site, C is calculated by averaging first within each 1-m^2 quadrat and second across all quadrats at the site. For wC, an additional step includes weighting species C-values by observed species abundances within quadrats (Bourdaghs et al., 2006). In this approach, species with high cover values in a wetland will have greater influence on the overall wetland wC score.

While C scores were calculated for all of the wetland sites sampled, they are not summarized in this report, since the scores do not capture quality differences resulting from high levels of invasive species dominance. In contrast, wC (Table 2) captures the degradation resulting from invasive plant dominance and is referenced in this report. Because wC scores calculated from 2011-2019 range from 0.06 to 8.27, the scale for wC-score ranking has been modified to 0.0 to 8.30 instead of the 0.0 to 10.0 scale used for C calculations as developed by Herman et al. (2001) Ranking is expanded from three categories (Poor, Fair, and Good) to six scoring categories (Extremely Degraded, Degraded, Moderately Degraded, Moderately Impacted, and Reference) (Table 2).

The IBI was developed by the Great Lakes Coastal Wetlands Consortium (Albert, 2008) and is being used by several agencies throughout the basin to rank sites based on wetland plant quality. The IBI consists of 10 metrics combined into an overall index of ecosystem condition. The metrics are primarily comprised of Mean C scores for individual wetland zones and the abundance of invasive species in zones. While three categories were used in earlier reports (Poor, Fair, and Good), these have been expanded to six scoring categories (Extremely Degraded, Degraded, Moderately Degraded, Moderately Impacted, Mildly Impacted, and Reference) (see Table 1). The 6 Descriptive Scores in Table 1 were based on the full range of Transformed Numeric IBI Scores from individual wetland sites across the entire Great Lakes, and the Descriptive Scores represent a roughly equal partitioning of 0.0 to 5.0 scores into 6 categories. While individual sampling sites have Numeric and Descriptive Scores in all six categories, the averaged scores of the individual lakes do not include Extremely Degraded, with both Lake Erie and Lake Ontario having average lakewide scores of Degraded (Poor). Figures 3 through 7 show the distribution of individual wetlands in each of the six IBI classes, and it can be seen that both Lakes Erie and Ontario have wetlands in the Extremely Degraded class.

Ecological Condition

Across the entire Great Lakes basin, the state of the wetland plant community is quite variable, ranging from Good to Poor (Further refined by dividing each of these categories into two categories: Poor = Extremely Degraded or Degraded; Fair = Moderately Degraded or Moderately Impacted; and Good = Moderately Impacted or Reference) depending primarily on local land use history, nearshore management, and the prevalence of invasive plant species. Plant communities in some wetlands have deteriorated rapidly in recent years due to extremely low water levels that have allowed invasion and dominance by non-native species. With water levels rebounding in 2014-2019, it will be critical to evaluate how these wetlands respond. In other wetlands, there have been recent improvements to plant community condition. For example, the turbidity of the southern Great Lakes has reduced with expansion of zebra mussels, resulting in improved submergent plant diversity in many wetlands. Moreover, wetland restoration activities have been undertaken throughout the basin over the past 10 years, especially targeting wetlands dominated by invasive plants.

Short and long-term trends in wetland condition based on plants have not been well-established in the Great Lakes. Almost all wetlands in Lake Erie, Lake Ontario, and the Upper St. Lawrence River are degraded by nutrient enrichment and sedimentation, while wetlands in Lake Ontario and the Upper St. Lawrence River are further degraded by water-level control. Probably the strongest demonstration of this is the prevalence of broad zones of cattails, reduced submergent diversity and coverage, and prevalence of non-native plants, including common reed (*Phragmites australis*), reed canary grass (*Phalaris arundinacea*), purple loosestrife (*Lythrum salicaria*), curly pondweed (*Potamogeton crispus*), Eurasian milfoil (*Myriophyllum spicatum*), frogbit (*Hydrocharis morsus-ranae*), and water chestnut (*Trapa natans*).

In the remaining Great Lakes (Lake St. Clair, Lake Huron, Lake Michigan, Georgian Bay, Lake Superior, and their connecting rivers), intact, diverse wetlands can be found for most geomorphic wetland types. However, low water conditions have resulted in the explosive expansion of common reed and hybrid cattail (*Typha x glauca*) in many wetlands, especially in Lake St. Clair and southern Lake Huron, including Saginaw Bay (Albert and Brown 2008) as well as Green Bay in Lake Michigan. As water levels rise, the response of common reed, hybrid cattail, and narrow-leaved cattail should be monitored. In at least a couple northern Lake Huron wetlands, hybrid cattail was noted to be expanding in the extremely high water levels that were simultaneously causing die back of native emergent plant.

One of the disturbing trends is the expansion of frogbit, a floating plant that forms dense mats capable of eliminating submergent plants, from the St. Lawrence River and Lake Ontario into Lake Erie, Lake St. Clair, Lake Huron, and the St. Marys River. This expansion will probably continue into all of the remaining Great Lakes, although frogbit was still not encountered in either Lake Michigan or Lake Superior in 2019. In addition, our sampling has shown water chestnut to be expanding rapidly in Lake Ontario—increasing in both distribution and density.

Studies in the northern Great Lakes have demonstrated that non-native invasive species like common reed, reed canary grass, and purple loosestrife have become established throughout the Great Lakes but that the abundance of these species is low, often restricted to only local disturbances such as docks and boat channels. It appears that undisturbed marshes are not easily colonized by these species. However, as these species become locally established, seeds or fragments of plants may be able to establish themselves when water-level changes create appropriate sediment conditions. Hybrid cattail (*Typha x glauca*) expansion has also been recently documented in northern Lakes Michigan and Huron and the St. Marys River (Lishawa et al., 2010).

Regional Wetland Types

The conditions of the plant community in coastal wetlands naturally differ across the Great Lakes basin, due to differences in geomorphic and climatic conditions. The characteristic size and plant diversity of coastal wetlands vary by wetland type, lake, and latitude; in this document these differences will be described broadly as “regional wetland types.”

Coastal wetlands are divided into three main categories based on the hydrology of the area. Lacustrine wetlands are connected to the Great Lakes, and they are largely impacted by fluctuations in lake levels. Riverine wetlands occur in the lower reaches of rivers that flow into the Great Lakes basin. Typically, the quality of riverine wetlands is influenced by the river drainage system; however, coastal processes cause lakes to flood back into these wetlands, which control water levels. The last type of coastal wetlands is barrier-protected. Barrier-protected wetlands are derived from coastal processes that deposit sediment to create barrier beaches that separate wetlands from the Great Lakes. Coastal wetlands contain different vegetation zones (treed or shrub swamp, meadow, emergent, submergent and floating), some of which may be absent in certain types of wetlands and under different water-level conditions. Great Lakes wetlands were classified and mapped in 2004 (see https://greatlakeswetlands.org/Home.vbhtml, Albert et al., 2006).

Lake Variations

Physical properties such as the type of shoreline, substrate, bedrock, and chemical and physical water quality parameters vary between Great Lakes. Variation in nutrient levels creates both a north to south gradient, and an increase in nutrient levels from Lake Erie in the west to Lake Ontario and the upper St. Lawrence River in the east. Lake Superior is the most distinct Great Lake due to its low alkalinity and prevalence of bedrock shoreline.

Differences in Latitude

Latitudinal variations result in different climatic conditions based on the location of the coastal wetlands. Temperature differences between the north and south lead to differences in the species of plants found in coastal wetlands. Watersheds in the southern portion of the Great Lakes also have increased agricultural activity, resulting in increased nutrient loads, sedimentation, and non-native species introductions.

Linkages

There are characteristics of coastal wetlands that make use of plants as indicators difficult in certain conditions. Among these are:

Water-level fluctuation

Great Lakes water levels fluctuate greatly from year-to-year. Either an increase or decrease in water level can result in changes in numbers of species or overall species composition in the entire wetland or in specific zones with change in level of human disturbance. Such changes make it difficult to monitor change over time. Changes are great in two zones: the wet meadow, where grasses and sedges may disappear in high water or new annuals may appear in low water, and in shallow emergent or submergent zones, where submergent and floating plants may disappear when water levels drop rapidly. Recent studies indicate that prolonged periods of low water favor rapid expansion of invasive species like *Phragmites australis* (Albert and Brown, 2008; Lishawa et al., 2010; Wilcox, 2012). In addition, water levels of Lakes Superior and Ontario are regulated, which has altered plant community dynamics. This is most obvious in Lake Ontario, where cattails have displaced sedge/grass meadow (Wilcox et al., 2008). Extreme high water conditions in 2018 and 2019 have resulted in wide-spread erosion of the vegetation in both the emergent and the meadow zones, with both zones being eliminated or greatly reduced in area in many open lacustrine wetlands. Protected lacustrine and riverine wetlands have displayed changes in species composition resulting from high water conditions, but most have not been heavily damaged by wave action. While our sampling of barrier-protected wetlands has been less intensive than in lacustrine and riverine wetlands, it has been observed that the outer barriers in many barrier-protected wetlands has been eliminated by storm waves, resulting in burial of vegetation in swales behind the outermost barrier by sand and gravel.
With climate change, some modelers have projected that water-level fluctuations will be more extreme, with higher highs and lower lows (Mailhot et al., 2019). As part of these models, it is also projected that there will be more frequent extreme rain events and frequency and extent of droughts. The greatest impact would be in riverine wetlands, where increased currents could change river channels and erode vegetation, but riverine wetland plant distributions already reflect this type of disturbance. Drought is likely the more problematic change, especially in a landscape with more drought-tolerant invasive plants (common reed, purple loosestrife, reed canary grass, hybrid cattail).

Pressures

Lake-wide alterations

For the southern lakes, most wetlands have been dramatically altered by both intensive agriculture and urban development of the shoreline. Alterations of coastal wetland especially in the wet meadow and upper emergent zone will lead to drier conditions which may allow non-native species to establish.

Agriculture

Agriculture degrades wetlands in several ways, including nutrient enrichment from fertilizers, increased sediments from erosion, increased rapid runoff from drainage ditches, introduction of agricultural non-native species (reed canary grass), destruction of inland wet meadow zone by plowing and diking, and addition of herbicides. In the southern lakes, Saginaw Bay, and Green Bay, agricultural sediments have resulted in highly turbid waters that support few or no submergent plants.

Lake-level regulation

Regulation of Lake Ontario water levels since 1960 has reduced the range of fluctuations. The most evident effect has been the elimination of low lake-level periods, even when water supplies are low. The competitive advantage of sedges and grasses at higher elevations due to their tolerance of low water levels and low soil moisture has been lost, and they have been displaced by larger cattails that are no longer limited by their need for more water. A new regulation plan for Lake Ontario (Plan 2014) was implemented by the International Joint Commission in January 2017 that will allow more natural water level fluctuation patterns.

Urban development

Urban development degrades wetlands by hardening shoreline, filling wetland, adding a broad diversity of chemical pollutants, increasing stream runoff, adding sediments, and increased nutrient loading from sewage treatment plants. In most urban settings, almost all wetlands have been lost along the shoreline.

Residential shoreline development

Along many coastal wetlands, residential development has altered wetlands by nutrient enrichment from fertilizers and septic systems, shoreline alterations for docks and boat slips, filling, and shoreline hardening. Agriculture and urban development are usually less intense than local physical alteration, which often results in the introduction of non-native species. Shoreline hardening typically eliminates wetland vegetation.

Mechanical alteration of shoreline

Mechanical alteration takes a diversity of forms, including diking, ditching, dredging, filling, shoreline hardening, and disking and plowing of coastal vegetation by private landowners. With all of these alterations, non-native species are introduced by construction equipment or in introduced sediments. Changes in shoreline gradients and sediment conditions are often adequate to allow non-native species to become established. Disking (harrowing) and plowing of coastal wetlands continued through 2011 in exposed coastal marshes along Saginaw Bay, Grand Traverse Bay, and on islands within the St. Clair River delta.

Introduction of non-native species

Non-native species are introduced in many ways. Some were purposefully introduced as agricultural crops or ornamentals, later colonizing in native landscapes. Others came in as weeds in agricultural seed. Increased sediment and nutrient enrichment allow many of the worst aquatic weeds to out-compete native species. Most of the worst non-native species are either prolific seed producers or reproduce from fragments of root or rhizome. Non-native animals have also been responsible for increased degradation of coastal wetlands. One of the worst invasive species has been common carp, whose mating and feeding habits result in loss of submergent vegetation in shallow marsh waters. The most prevalent non-native plants include common reed (*Phragmites australis*), reed canary grass (*Phalaris arundinacea*), purple loosestrife (*Lythrum salicaria*), curly pondweed (*Potamogeton crispus*), and Eurasian milfoil (*Myriophyllum spicatum*). Low water conditions have resulted in an explosive expansion of common reed in many wetlands, especially in Lake St. Clair and southern Lake Huron, including Saginaw Bay (Albert and Brown 2008). One of the disturbing recent trends is the expansion of European frogbit (*Hydrocharis morsus-ranae*), a free-floating plant that forms dense mats along the emergent margin capable of eliminating submergent and emergent plants, from the St. Lawrence River and Lake Ontario into Lake Erie, Lake St. Clair, Lake Huron, and the St. Marys River (Monks et al., 2019). Continued expansion of frogbit into many additional coastal wetlands has been observed throughout the high water years of 2014-2019. This expansion will likely continue to all of the remaining Great Lakes. In addition, our sampling has shown water chestnut (*Trapa natans*) to be expanding rapidly in Lake

Ontario—increasing in both distribution and density. The recent rediscovery of a non-native macroalgae, starry stonewort (*Nitellopsis obtusa*), is of conservation concern because of its long-term establishment since the 1970s and its current distribution within better quality wetlands in northeastern Lake Ontario as well as wetlands in Saginaw Bay, Lake St. Clair, and the Detroit River. Starry stonewort grows rapidly, reducing habitat value and clogging boat propellers.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://www.greatlakeswetlands.org/Home.vbhtml		

Data Limitations

The characteristic presence and abundance of coastal wetland native plants has been adequately documented across the Great Lakes basin by the Great Lakes Coastal Wetland Monitoring Program (CWMP), whose URL is attached above. An addition, most regional wetland types have been adequately described in existing studies (Minc, 1997; Minc and Albert, 1998; Albert and Minc, 2001; Albert et al., 2006; Lemein et al., 2017). The changes in species composition and dominance related to Great Lakes water-level fluctuations have not been adequately determined for many regional wetland types. This is an important task, as natural water-level fluctuations can introduce changes in wetland vegetation that could falsely be attributed to either increased wetland degradation or improved management. Laboratory studies are also needed to identify wetland plant species response to different types of degradation, including turbidity, sedimentation, heavy metal and organic chemical introduction, pH change, erosion, exotic plant competition, and increased herbivory by non-native fauna.

Additional Information

**The CWMP was funded by the Great Lakes Restoration Initiative 2011-2020 to implement statistically sound basin-wide monitoring of select physical and biotic components (Uzarski et al., 2016) and the program's funding has been extended to include 2021-2025. This binational program involved a consortium of universities and agencies with the goal of producing scientifically-defensible information on status and trends of Great Lakes coastal wetlands. As of 2015, the majority of coastal wetlands ≥4 ha with a surface water connection to the lakes have been surveyed at least once since 2011. Data from 2011-2019 were included in the analysis reported here. In each wetland, data from up to three wetland zones (wet meadow, emergent, submergent) are included if all zones are present.

The tables in this document summarize data collected between 2011 and 2019 on three broad hydrogeomorphic wetland types: barrier-protected, lacustrine, and coastal wetlands that were characterized for each separate Great Lake. All three types were lumped for the analysis conducted in this report, but in subsequent analyses these types may be further divided into recognized hydrogeomorphic subtypes (Albert et al., 2006) that are subject to different environmental and human stresses, and thus characterized by different status and potential for restoration. This sub-indicator incorporates information on the presence, abundance, and diversity of aquatic macrophytes within Great Lakes coastal wetlands. Plant abundance data are used to calculate two measures of wetland plant quality including: 1. Vegetation Index of Biological Integrity (IBI); and 2. Weighted Mean Coefficient of Conservatism (wC). A third approach, the Mean C approach was calculated, but is not adequate for Great Lakes coastal wetlands, because it provides little understanding of the overall condition of Great Lakes coastal wetlands because of the prevalence of invasive plants in many wetlands. Either an IBI or Weighted Mean C (wC) better capture the influence of invasive plants on wetland condition.

It has been estimated that approximately half of the coastal wetlands have been lost basin wide, but this estimate does not include degraded wetlands, just those that have been lost by shoreline hardening or complete erosion of vegetation from an area. There is no agreed on approach to providing a more accurate estimate for several reasons, the most important of which are 1) The original land surveys, the basis of many original plant community area estimates, did not consistently reference herbaceous wetland vegetation along the shoreline, 2) Emergent wetland vegetation is not easily seen in aerial photos limiting the use of 1930s and 1940s early aerial photos to estimate original wetland sites, and 3) the earliest Great Lakes-wide surveys of coastal wetlands were conducted in the late 1970s and early 1980s, well after most of the coastal wetland destruction had occurred due to a combination of shoreline hardening, dredging, agricultural planting, and destruction by invasive fish [carp].

While no Great Lakes-wide surveys of coastal wetland with a focus on vegetation were conducted before the 1980s, cluster analyses of physical and vegetation data from field surveys conducted in the 1980s and 1990s identify several distinct native plant communities, as well as some plant communities dominated by invasive plants, that show strong relationships to regional climatic, sediment, and hydro-geomorphic conditions (Minc, 1997; Albert and Minc, 2001; Albert et al., 2006) that can justifiably be used as the basis for assuming there are predictable regional wetland vegetation types or communities. Vegetation sampling from the CWMP (Uzarski et al., 2016) was utilized to update the distribution of native and invasive dominated plant communities (Lemein et al., 2017).

Cattails have been noted as a major source of degradation because the expansion of cattails into wetlands following nutrient enrichment and water-level manipulation had been documented in numerous studies (Prince and D'Itri, 1985; Stuckey, 1989; Wilcox, 1993; Minc, 1997; Wilcox et al., 2008; Lishawa et al., 2010; and Robert Humphreys, refuge manager for MI DNR), personal communications). The native cattail in Great Lakes coastal wetlands was *Typha latifolia* (common or wide-leaved cattail) a species that was limited in distribution by characteristic fluctuations in Great Lakes water levels. *Typha angustifolia* (narrow-leaved cattail) has expanded into Great Lakes wetlands, where it tolerates deeper water levels than common cattail, expanding its range rapidly through the eastern U.S. and the Midwest along roadside ditches (Carson et al., 2018; Bansal et al., 2019). Common and narrow-leaved cattails hybridized, forming *Typha* x *glauca* (hybrid cattail), a larger and more aggressive plant that along with narrow-leaved cattail created broad, dense monocultures that did not meet the habitat needs of many native waterbirds and waterfowl. The dense mats of narrow-leaved and hybrid cattails were also able to float in drowned river mouth wetlands, eliminating important fish habitat as well.

Damage to Great Lakes wetlands by non-native invasive plants during the most recent low-water event (1999-2013 in Lakes Michigan and Huron) is considered to be linked to anthropogenic degradation because all of the invasive plants that have expanded dramatically into Great Lakes coastal wetlands were introduced into the Great Lakes by humans and respond aggressively to agricultural and urban nutrient enrichment and/or sedimentation. Earlier surveys of Great Lakes wetlands in low-water conditions in the 1980s and 1990s documented existing large-scale or localized expansions of these invasive plants in Lakes Ontario, Erie, and Lake St. Clair, but the expansion of these same plants was much greater than the extended low-water conditions in Lakes Huron and Michigan between 1999 and 2013. Prior to the 1970s, our most aggressive invasive plants (*Phragmites australis*, *Typha angustifolia*, *Typha* x *glauca*, *Lythrum salicaria*, *Hydrocharis morsus-ranae*, etc.) that respond to low-water conditions were not widespread along the Great Lakes shoreline, but since the early 1970s and into the future prolonged periods of low-water can be expected to result in at least localized expansions of invasive wetland plants.

Baseline condition in biological or restoration studies has typically been based on characteristic native flora and fauna in an ecosystem. Several examples of wetlands with no extensive populations of invasive plants were inventoried during 2011-2015 (Uzarski et al., 2016) and provide the basis for defining baseline condition and providing a goal for restoration. These high-quality wetlands will remain the basis for monitoring wetland condition and guiding restoration efforts, even if it is determined in the future that returning degraded wetlands to these conditions is impossible.

Acknowledgments

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Dennis Albert.

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Dennis Albert

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

Figure 3. Distribution of wetland IBI scores along Lake Superior over sampling years 2011 through 2019.

Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

Figure 4. Distribution of wetland IBI scores along Lake Michigan over sampling years 2011 through 2019.

Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

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Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

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Table 1. IBI Score Categories calculated from Great Lakes Coastal Wetland Monitoring Program vegetation data (Albert 2008). The original 3-division descriptive codes are shown in parentheses (). Source: GLRI Coastal Wetland Monitoring Program, analysis by Dennis Albert.

Raw Numeric Score	Transformed Numeric Scores	Descriptive Scores
0-7	0-0.79	Extremely Degraded (Poor)
8-16	0.80-1.69	Degraded (Poor)
17-25	1.70-2.59	Moderately Degraded (Fair)
26-33	2.60-3.39	Moderately Impacted (Fair)
34-41	3.40-4.19	Mildly Impacted (Good)
42-50	4.20-5.00	Reference (Good)

1. Actual individual site IBI scores range from 0.40 to 5.0, which included Extremely Degraded to Reference conditions over the period of 2011-2019.
2. Annual Mean IBI scores by lake range from 1.28 to 4.26, which included Degraded to Reference conditions over the period of 2011-2019. The range of Mean IBI scores is less than the range of Actual individual site IBI scores.

Table 2. Modified Mean Weighted C (wC) Score Categories calculated from Great Lakes Coastal Wetland Monitoring program vegetation data. The original 3-division descriptive codes are shown in parentheses (). While Mean Weighted C Scores are based on a 0-10 scale, the actual vegetation scores did not exceed 8.27, so a Modified 0-8.30 score is being utilized. Source: GLRI Coastal Wetland Monitoring Program, analysis by Dennis Albert.

Modified Numeric Score	Descriptive Scores
0-1.38	Extremely Degraded (Poor)
1.39-2.76	Degraded (Poor)
2.77-4.14	Moderately Degraded (Fair)
4.15-5.52	Moderately Impacted (Fair)
5.53-6.90	Mildly Impacted (Good)
6.91-8.30	Reference (Good)

1. Actual Weighted Mean C scores for individual sites ranges from 0.06 to 8.27, which included Extremely Degraded to Reference conditions over the period of 2011-2019.
2. Annual Average Weighed Mean C scores by lake range from 1.72 to 6.11, which included Degraded to Mildly Impacted over the period of 2011-2019. The range of Annual Average Weighted Mean C scores by lake is less than the range of Actual Weighted Mean C scores for individual sites.

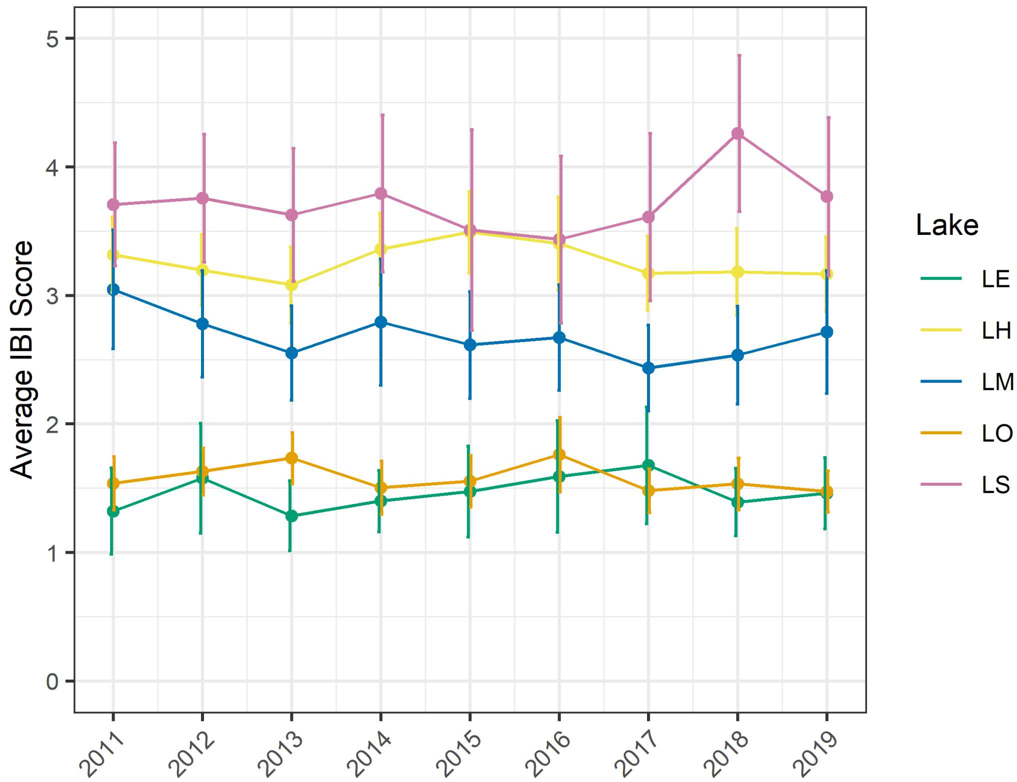

Figure 1. Average IBI Scores by year and lake. The scores were computed from transect data collected as part of the Great Lakes Coastal Wetland Monitoring Program. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel.

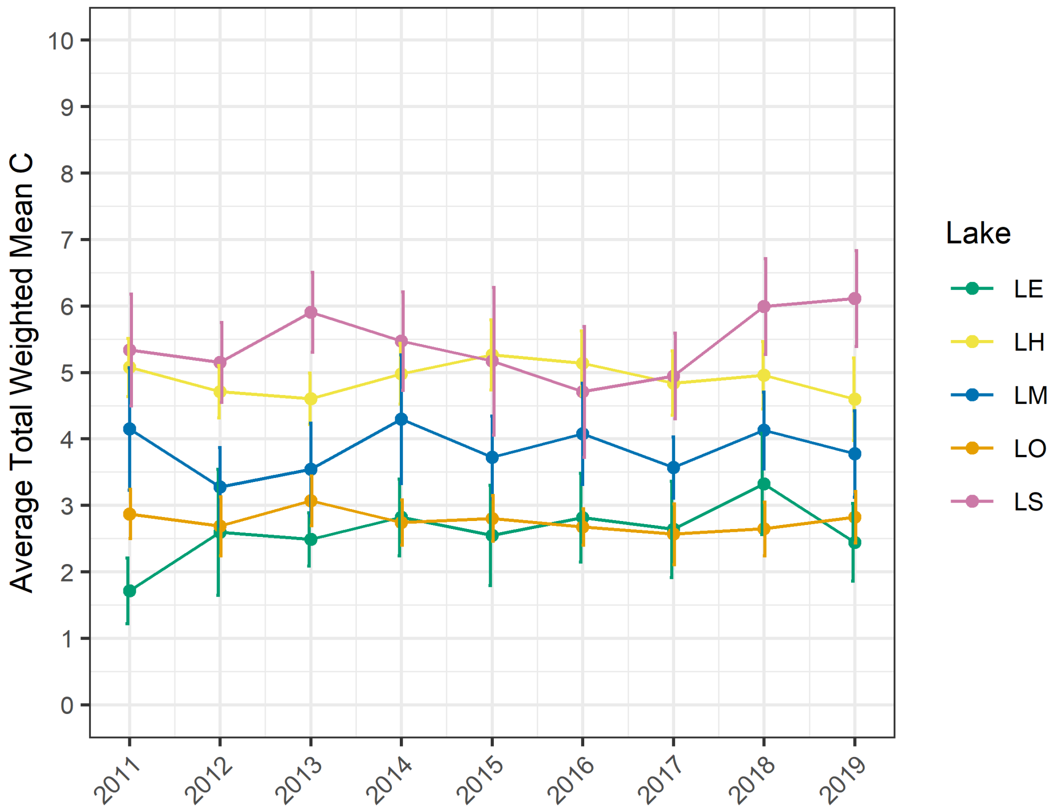

Figure 2. Average Weighted Mean C (wC) Score by year and lake. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel.

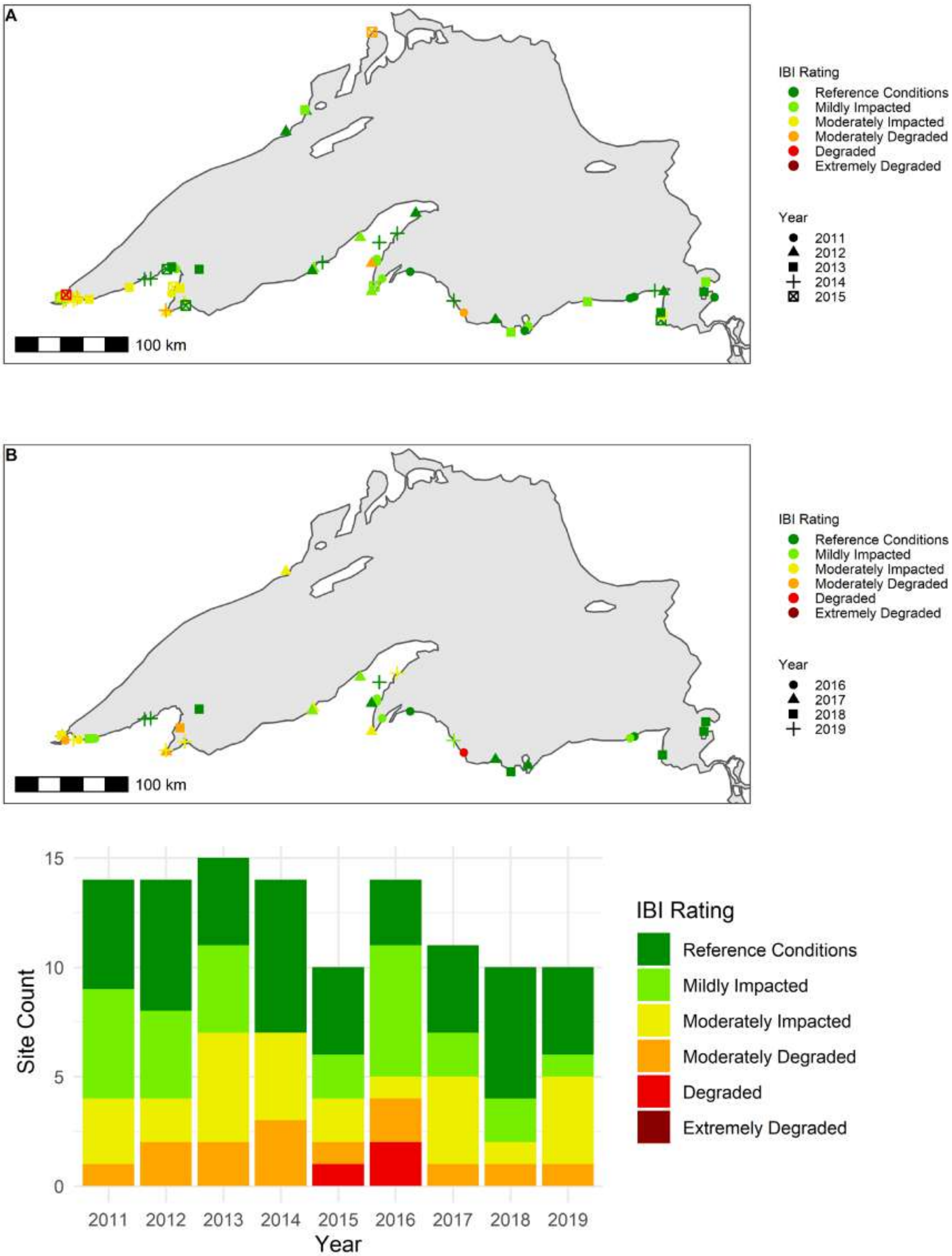

Figure 3. Distribution of wetland IBI scores along Lake Superior over sampling years 2011 through 2019. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

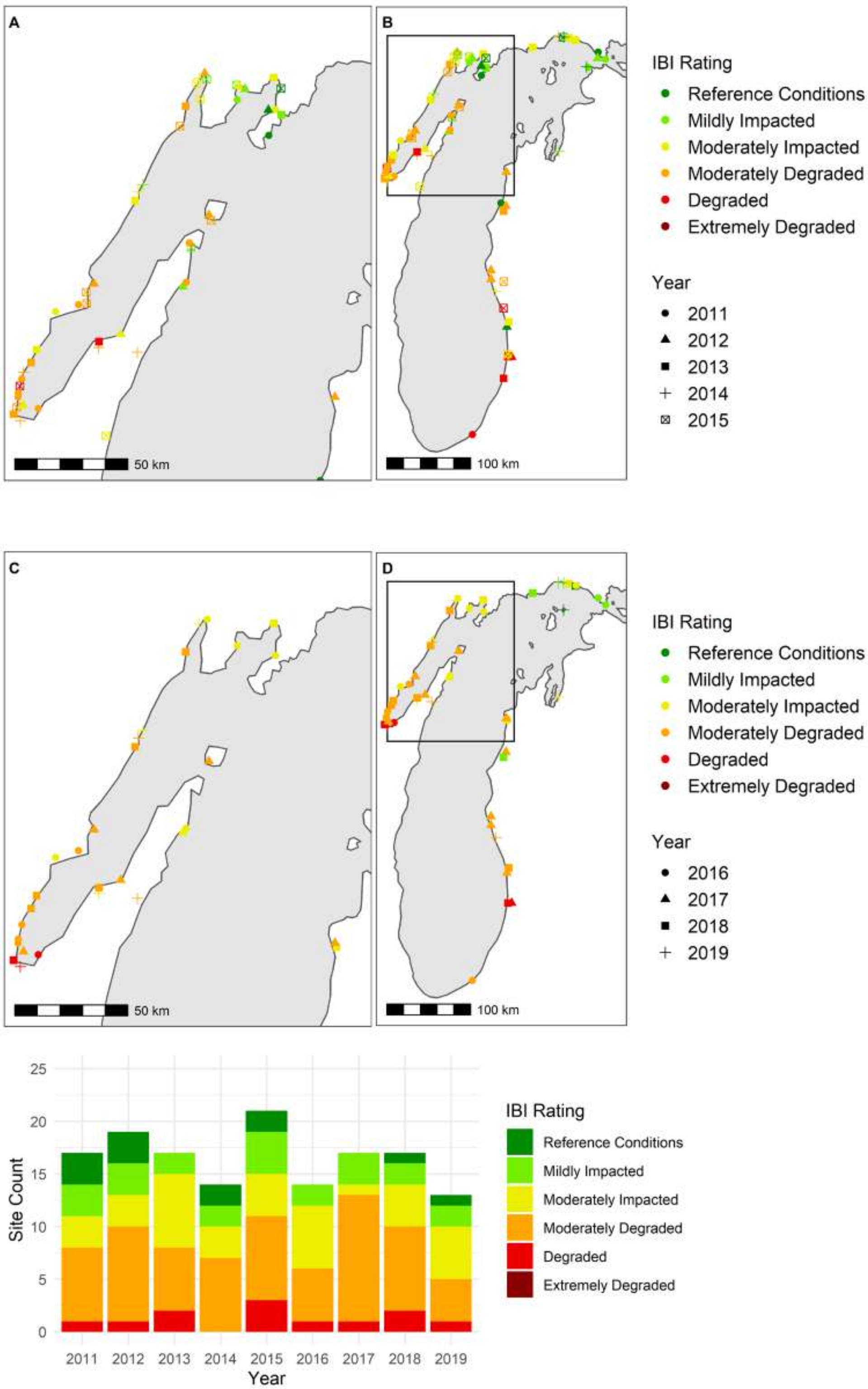

Figure 4. Distribution of wetland IBI scores along Lake Michigan over sampling years 2011 through 2019. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel.

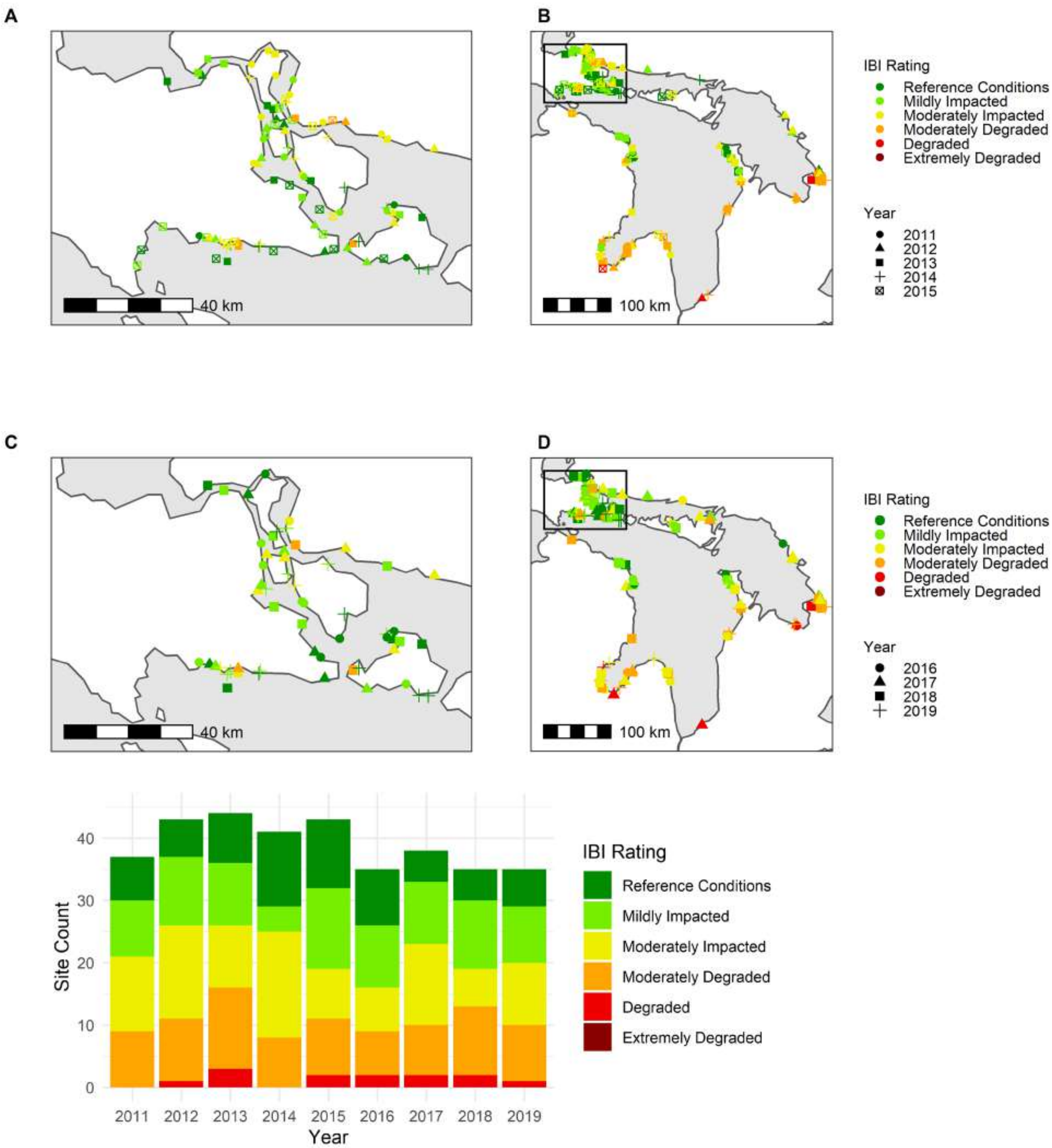

Figure 5. Distribution of wetland IBI scores along Lake Huron over sampling years 2011 through 2019. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel.

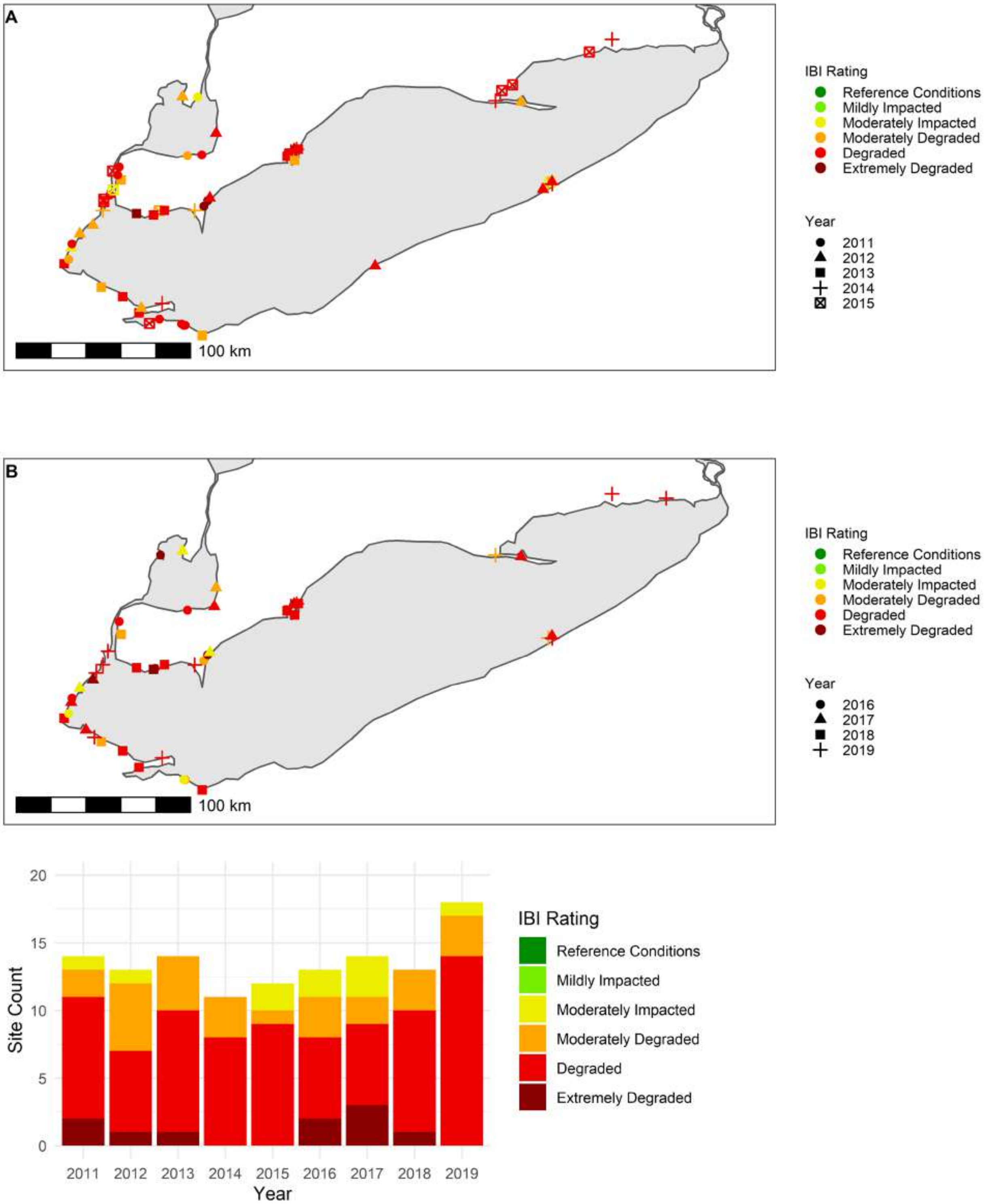

Figure 6. Distribution of wetland IBI scores along Lake Erie/Lake St. Clair over sampling years 2011 through 2019. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel

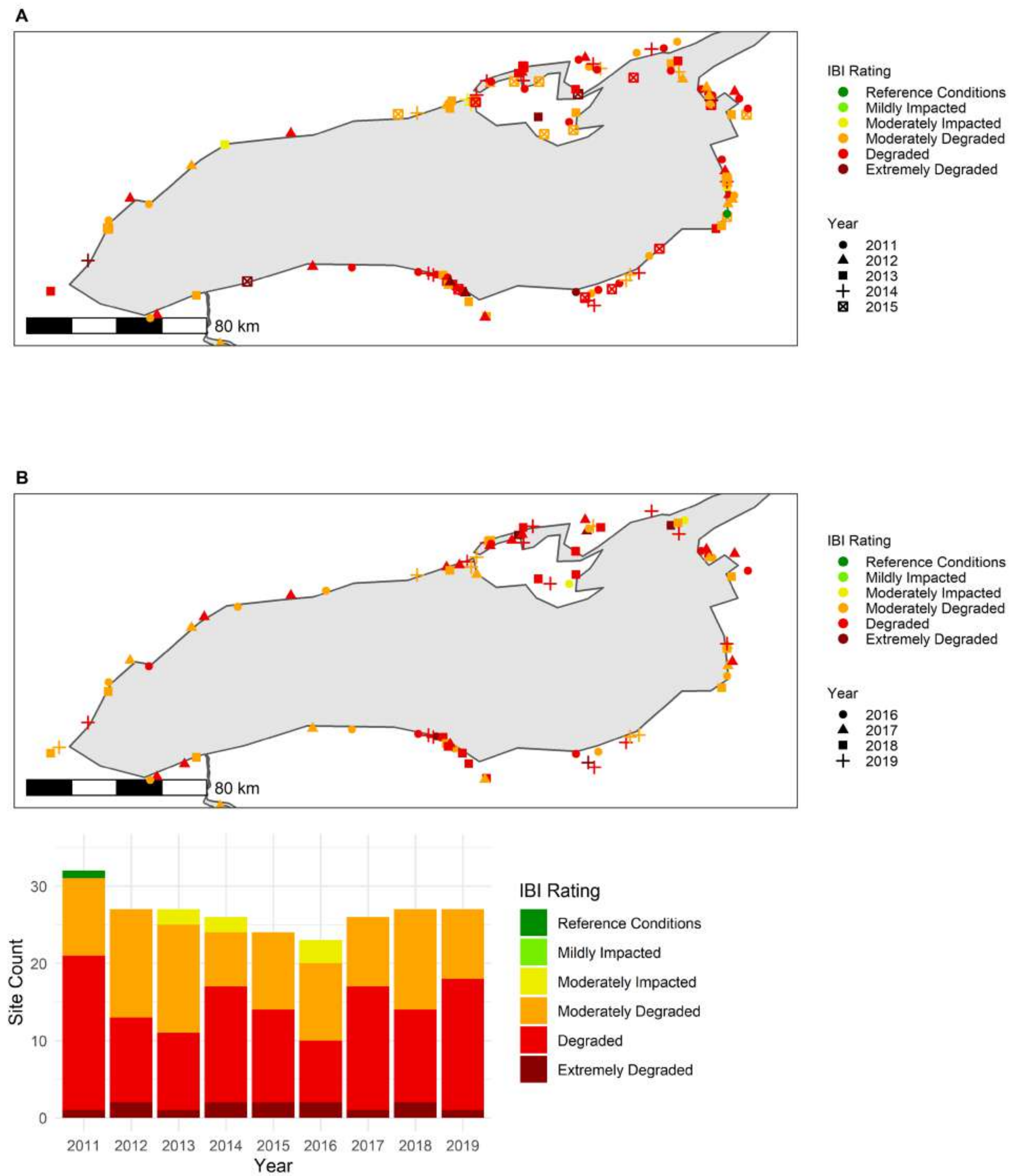

Figure 7. Distribution of wetland IBI scores along Lake Ontario and the Niagara and Upper St. Lawrence Rivers over sampling years 2011 through 2019. Source: GLRI Coastal Wetland Monitoring Program, analysis by Allison Kneisel.

Sub-Indicator: Aquatic Habitat Connectivity

Overall Assessment

Status: Fair

Trends:

10-Year Trend: Improving

Long-term Trend (1950-2022): Improving

Rationale: Approximately one-third (32.5%) of the tributaries that were historically connected to the Great Lakes remain accessible to migratory fishes (Figure 1). The amount of tributary habitat that is accessible ranges from 53% in Lake Superior to less than 20% in Lake Michigan (Khoury et al., 2018). These barriers include dams, but also road-stream crossings. In some parts of the Great Lakes lack of access to spawning habitats in rivers and streams is limiting fish populations. Over the past decade there has been an increasing number of projects to remove barriers or improve fish passage. The Great Lakes Fisheries Commission have identified priority barriers where actions to restore connectivity would support the recovery of fish populations (Table 1, Figure 2). Barrier removal in the Great Lakes must be coordinated with efforts to limit the access of the invasive Sea Lamprey to riverine spawning habitats (Figure 3).

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend: Improving

Long-term Trend (1950-2022): Improving

Rationale: Just over half (53.8%) of the tributaries that were historically connected to Lake Superior remain accessible to migratory fishes. There are continued efforts to restore aquatic habitat connectivity in the Lake Superior basin. These projects range from culvert replacement on Gehrman Creek at the Highway 169 crossing that will restore 1.6 km of passage for Brook Trout (Superior Rivers Watershed Partnership, WI). . The recent decision by the Ontario Ministry of Environment, Conservation and Parks to repair, rather than remove the failing Camp 43 dam on Ontario's Black Sturgeon is an example of the challenges in balancing fish passage, infrastructure and Sea Lamprey control.

Lake Michigan

Status: Poor

10-Year Trend: Improving

Long-term Trend (1950-2022): Improving

Rationale: Less than one-fifth (18.9%) of the tributaries that were historically connected to Lake Michigan remain accessible to migratory fishes. There are continued efforts to restore aquatic habitat connectivity in the Lake

Michigan basin. This range in scale from replacing under-sized culverts on the Crystal River (Grand Traverse Band of Ottawa and Chippewa Indians, MI) to dam removal on the Boardman River that will connect more than 250 km of tributary habitat back to Lake Michigan (Conservation Resource Alliance, MI). The sturgeon "elevator" on the Menominee River that has operated since 2015 is an example of the innovative approaches that are being applied in improve aquatic habitat connectivity.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend: Improving

Long-term Trend (1950-2022): Improving

Rationale: Almost one-third (30.0%) of the tributaries that were historically connected to Lake Huron remain accessible to migratory fishes. There are continued efforts to restore aquatic habitat connectivity in the Lake Huron basin. These include the removal of the dam on Black Ash Creek (Nottawasaga Valley Conservation Authority, ON) and the continued efforts of the Huron Pines Resource Conservation and Development Council in Michigan that recently removed two fish passage barriers in the Thunder Bay River, reconnecting 37 km of upstream habitat for Brook Trout and other fishes. The Thunder Bay project is an example of the many barrier removal projects across the basin that are not contributing to immediate gains in connectivity to the Great Lakes but could result in substantial increases as barriers further downstream are mitigated.

Lake Erie (including St. Clair-Detroit River watershed)

Status: Fair

10-Year Trend: Improving

Long-term Trend (1950-2022): Improving

Rationale: Approximately one-third (33.4%) of the tributaries that were historically connected to Lake Erie remain accessible to migratory fishes. There are continued efforts to restore aquatic habitat connectivity in the Lake Erie basin. Current efforts include the Henry Ford Estate Dam Fishway Restoration Rouge River Area of Concern (MI) that will reconnect 80.5 km of river and 174 km of tributary to the Great Lakes for the first time in over 100 years via a naturalized by-pass channel and the Springville Dam removal project on Cattaraugus Creek (NY) will restore approximately 106 km of spawning habitat for Rainbow Trout and improve connectivity for Brook Trout while simultaneously blocking Sea Lamprey from the upper watershed (currently on hold due to COVID).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Fair

10-Year Trend: Improving

Long-term Trend (1950-2022): Improving

Rationale: Over one-third (39.4%) of the tributaries that were historically connected to Lake Ontario remain accessible to migratory fishes. There are continued efforts to restore aquatic habitat connectivity in the Lake Ontario basin. These range from restoration of road-stream crossings on Spring Mills Creek and Indian Creek (Steuben County Soil & Water Conservation District (NY) to the creating on-stream pond by-passes and the removal of perched culverts on Twelve Mile Creek (ON) by Trout Unlimited Canada.

Status Assessment Definitions

Good: >75% aquatic habitat connectivity compared to historical conditions. The state of aquatic habitat connectivity and access to spawning habitat is likely sufficient to maintain migratory fish populations.

Fair: 25% to 75% aquatic habitat connectivity compared to historical conditions. The state of aquatic habitat connectivity is stressed in some locations and access to spawning habitat is likely limiting populations of some migratory fish species.

Poor: < 25% aquatic habitat connectivity. The state of aquatic habitat connectivity severely impaired and access to spawning habitat is likely limiting populations of many migratory fish species Undetermined: The percentage of aquatic habitat connectivity is insufficient to assess conditions of the ecosystem components or data are not available.

Trend Assessment Definitions

Improving: Increase in the amount of riverine habitat connected to the Great Lakes based on an assessment of new mitigation projects and new barriers.

Unchanging: No measurable change in the amount of aquatic connectivity.

Deteriorating: Decrease in the amount of riverine habitat connected to the Great Lakes based on an assessment of new mitigation projects and new barriers.

Undetermined: Information on barrier mitigation does not indicate a clear overall trend or data are not available to report on a trend.

Endpoints and/or Targets

1. Percent of rivers/streams connected to the Great Lakes compared to historical conditions. Overall objective to increase connectivity, with a focus on priority barriers, while managing for invasive species.
2. Progress in mitigating priority barriers (Table 1).
3. Examples of key initiatives to improve aquatic habitat connectivity.

Sub-Indicator Purpose

The purpose of this sub-indicator is to track the amount of accessible tributary habitat for migratory Great Lakes fishes by lake, to summarize key initiatives to improve the connectivity of aquatic habitat with a focus on priority sites; and highlight some of the issues related to barrier mitigation.

Ecosystem Objective

This sub-indicator supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species." The objective is to increase aquatic habitat connectivity in a manner that enhances native fish populations while supporting the control of invasive species.

Measure

This sub-indicator measures tributary connectivity as the percent of mainstem channel length that is currently connected to the Great Lakes compared to historical conditions. It has been calculated for the entire Great Lake basin and each lake (Khoury et al., 2018). Increased aquatic connectivity would benefit fish passage, as well as ecosystem function, however, in many cases the risk of increased Sea Lamprey production prohibits removal of most of the lowest barriers to Great Lakes migration at this time. These barriers are beneficial to some native fishes by limiting the threat of Sea Lamprey predation. Although direct measures of the population viability of migratory fishes by watershed is the optimal indicator on the status of aquatic connectivity, the amount of available habitat can be applied as a surrogate measure. Results for this sub-indicator have been drawn from an analysis on barriers that is now available for the Great Lakes basin and case studies on barrier removal projects. Status thresholds on aquatic habitat connectivity have been developed for the Biodiversity Conservation Strategies for each lake (Franks Taylor et al., 2010; Lake Ontario Biodiversity Conservation Strategy Working Group, 2009; Lake Superior Lakewide Action and Management Plan, 2013; Pearsall et al., 2012a; Pearsall et al., 2012b). Data on barriers is available through Fishwerks (University of Wisconsin 2018). Fishwerks is an online GIS tool that maps barriers and provides the basis for a decision-support tool to guide barrier removal and provide a systematic framework for comparing costs (economic costs, species invasions) and benefits (connectivity, focal fish species) (Januchowski-Hartley et al., 2013).

Ecological Condition

Dams have impacted over half of the world's drainage basins (Barbarossa et al., 2020; Nilsson et al., 2005; Su et al., 2021). Globally only 37 per cent of rivers longer than 1000 km remain free-flowing over their entire length (Grill et al., 2019). Dams and barriers have been impacting the health of aquatic ecosystems in the Great Lakes basin for over two centuries. These barriers have contributed to extirpation of species such as Atlantic Salmon and limit the recovery of some fish populations. The construction of new dams and barriers on Great Lakes tributaries peaked over a century ago when waterpower was the primary energy source in the basin. Many of the larger dams were built in the 20th century for hydroelectric power generation. Over the last few decades, very few new dams have been built, and there has been a recent trend to remove old dams. The potential impacts of road-stream crossings are now better understood, and there have been several regional initiatives to identify and mitigate culverts that act as barriers.

Aquatic connectivity provides chemically and physically unobstructed routes to fulfill life history requirements of aquatic species, including access to intact refugia and opportunities for genetic exchange. Barriers threaten the diversity and abundance of native Great Lakes fishes by restricting or eliminating connectivity between the lakes and critical spawning, nursery, and overwintering habitats (Januchowski-Hartley et al., 2013) and compromise production and harvests. Two-thirds of the river and stream habitats that were historically connected to the Great Lakes are now inaccessible to migratory fishes (Figure 1). This loss of tributary habitat has resulted in significant declines in native fish populations in the Great Lakes, such as Lake Herring, Yellow Perch, Walleye, Lake Sturgeon, River Redhorse, Black Redhorse, Eastern Sand Darter, and Channel Darter (Great Lakes Fishery Commission. 2007; Bredin 2002).

For most fishes (including almost all native species), traditional passage facilities (e.g. fish ladders) will not mitigate these effects. For example, salmonids can pass through the fishway at the Flat Rock Dam on the Huron River but it remains a barrier for Walleye (Leonardi & Thomas, 2000). In addition to impacting the fishes that migrate from the Great Lakes into tributaries, many stream-dwelling species of fish (e.g. Brook Trout) suffer discontinuity in their ranges because of barriers.

In addition to limiting access of fishes to spawning and nursery habitats, loss of aquatic connectivity impacts nutrient flows, and riparian and coastal processes (Childress et. al, 2014; Jansson et. al, 2007).

This indicator on aquatic habitat connectivity primarily focusses on the connectivity of the Great Lakes to its tributaries for fishes. While natural barriers to fish migration, such as rapids and waterfalls, are not the same as dams as they do not disrupt the flow and temperatures of tributaries, they still are a barrier to the upstream migration of many fishes. However, their presence has not resulted in decline in the diversity and abundance of Great Lakes fishes from historical conditions. Mapping of these naturally disconnected watershed has been completed since the last indicator report (Khoury et al., 2018), and these watersheds removed from the analysis. As a result, tributaries that were naturally not connected to the Great Lakes have now been excluded from the indicator.

This new information resulted in some non-genuine status changes for Lake Huron and Ontario. Both of these lakes have significant portions of their watersheds that have natural barriers (e.g. Moon River Falls on the Moon River in Georgian Bay). This change in approach helps to emphasize the locations and importance of artificial barriers that continue to impede the recovery of many fish stocks. While there is a positive improving trend in all the Great Lakes, genuine change in status will take decades of continued efforts. These gains will come through both the mitigation of priority barriers (Table 1) and the cumulative impact of the thousands of smaller scale projects that are happening throughout the basin. Mitigation of first barriers will require new and creative approaches to improve connectivity while limiting access by Sea Lamprey. The Fishpass project and other initiatives that are employing new technologies could greatly increase the pace of progress for restoring Great Lakes connectivity Zielinski & Freiburger, 2020).

Evaluating aquatic habitat connectivity and prioritizing projects to increase it are complex matters. Some dams and barriers impede the movement of some fish species but not others. Many dams and reservoirs have been in place for over a century and there are strong social pressures to maintain them. Climate change and aging infrastructure are resulting in the failure of dams and culverts. Removing dams can be complicated by contaminated sediments or endangered species that require careful consideration and planning. Dams and barriers have also been identified as both an opportunity to control the spread of aquatic invasive species, and as a potential source of aquatic invasive species due to releases of bait fish in impoundments and by acting as 'stepping stones' for aquatic invasive species (Anas & Mandrak, 2021; Comte et al., 2021; Johnson et al., 2008). Impoundments can also change fish assemblages in adjacent river habitats through the proliferation of habitat generalists species and declines in fluvial specialists.

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Climate Change – Increasing frequency and magnitude of storm events will continue to put pressure on aging infrastructure. This increases the urgency for surveys to identify under-sized culverts at road-stream crossings, and opportunities to mitigate barriers to fish migration during replacement. Many older dams will also come under increasing pressure. For example, on May 19, 2020, following heavy rainfall in the Gladwin and Midland County areas in Michigan, the privately-owned Edenville Dam failed, releasing a torrent of water that caused the downstream Sanford Dam to fail. With aging dam infrastructure and climate change come an increasing probability of dam failure across the basin. There is an urgent need to identify these risks and weigh the costs and benefits of dam replacement with removal.

- Aquatic Invasive Species – There are many examples across the Great Lakes where the first barrier is protecting native stream assemblages from competition and physical disturbance of substrates from non-native salmonids (Bredin 2002; Zorn et. al 2020). Hence, decisions about removal of dams and barriers must balance competing interests and goals, which may not always be explicit. Some dams and barriers may also play a role in limiting the spread of other invasive species such as Round Goby and Sea Lamprey (see below) and disease (e.g. Viral Hemorrhagic Septicemia). There is also recent evidence of dams facilitating the invasion of Round Goby (Raab et. al 2018).
- Lake Sturgeon – Loss of aquatic connectivity has contributed to the decline of the species.
- Lake Trout – Removed barriers that result in more parasitic Sea Lamprey would likely cause declines in numbers of Lake Trout and slow progress towards restoration. However, Lake Trout were also a historic migrant into river systems and loss of connectivity contributed to the loss of most riverine Lake Trout spawning runs (Loftus 1958, Khoury 2018).
- Sea Lamprey – Dams and barriers currently limit the spread of some Great Lakes invaders. For example, Lake Huron supports the largest population of Sea Lamprey in the Great Lakes (Liskauskas et al., 2007), and dams and low-head barriers are a major control mechanism used by managers. By managing Sea Lamprey through barriers, non-target species, such as native lampreys, are not impacted by lampricide treatments.
- Walleye – Loss of aquatic connectivity has contributed to the decline of some populations.
- Water Quality in Tributaries – Barrier removal could improve water quality by restoring natural flow patterns and helping to reduce stream temperatures.
- Precipitation Amounts – An increase in extreme rainfall events may encourage replacement of aging infrastructure, including culverts. Infrastructure adaption projects could potentially improve aquatic habitat connectivity at road-stream crossings (Neeson et. al 2015).

This sub-indicator also links directly to the other sub-indicators in the Habitats and Species indicator.

Traditional Ecological Knowledge (TEK), Citizen Science and other Bodies of Knowledge

There are several excellent examples from around the Great Lakes of citizen science being used to help confirm potential barriers to aquatic habitat connectivity. These programs typically engage volunteers to visit road-stream crossings and collect data to identify if the crossing represents a barrier, such as a perched culvert. Volunteers use standardized protocols to collect this data (e.g. U.S. Forest Service et. al 2011). The impacts of these projects can difficult to measure, but are significant (see Data Limitations). There remain opportunities to better incorporate Indigenous knowledge and perspectives on lamprey control efforts into current management practices (Mattes & Kitson,2021).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://greatlakesconnectivity.org/		

Data Limitations

Although there have been significant improvements in the cataloging of dams and barriers across the basin in the last few years, some dams are undocumented. Our current picture of barriers across the Great Lakes is not complete. Some watersheds are better sampled than others, potentially resulting in data bias (Khoury et al. 2018). There is potential that watersheds that have more sampling may be artificially elevated in priority. Spatial analysis of connectivity can be challenging if dam coordinates do not intersect with the hydrology layer. Road stream crossings can highlight potential barriers, but these need to be ground truthed to assess their actual impact. Recent efforts by energy companies to re-license hydropower dams in the United States have led to a reconsideration of the habitat losses associated with these dams and a useful picture is emerging which allows an assessment of the adverse impacts of habitat fragmentation on migratory and resident stream-fish communities. Data for tributary habitat are being developed in connection with Federal Energy Regulatory Commission (FERC) dam relicensing procedures in the United States. Data are presently available for Michigan, New York State, and Wisconsin. The Ontario Dam Inventory was released in 2020, however it does not include small dams and water-control structures. The Canadian Wildlife Federation has also recently launched the Canadian Aquatic Barriers Database that includes pilot areas from the Great Lakes basin.

This sub-indicator does not measure lateral hydrological connectivity (LHC) (Liu and Wang 2018). LHC includes the connectivity between aquatic ecosystems and adjacent habitats, such as floodplains. A decrease in LHC can result from flow regulation on rivers and physical structures and have an impact on fish communities. In the context of the Great Lakes LHC could include access to spawning habitat in floodplains along rivers that are connected to the Great Lakes, and aquatic habitat connectivity to coastal marshes. Data to measure LHC has not been adequately developed throughout the Great Lakes. Land cover and dams may provide a proxy, although connectivity between rivers and streams and their floodplains in sub-watersheds dominated by natural cover could be impacted by water level regulation from upstream dams. Connectivity of the Great Lakes to coastal marshes has also not been analyzed and mapped. This sub-indicator does not fully measure all the smaller scale changes to connectivity that are happening around the basin. While the many smaller scale projects that are happening around the basin are difficult to fully report on, their impact is undoubtedly significant, and likely exceeds the impact of larger single barrier removal projects. For example, in just the last five years, Huron Pine in Michigan has improved over 90 road/stream crossings, removed 5 dams and reconnected over 720 km of river.

Additional Information

Examples of barrier removal projects were based on a search of GLRI projects database (search terms: 'connectivity', 'culvert', 'dam', 'passage' and 'tributary'). For Canadian projects a web search was using these same terms, plus Great Lake name. There are also recent examples of emerging decision-support tools to support the removal of the most critical barriers while considering economic costs and invasive species (e.g. Milt et al., 2018) and the use of indicator species to guide and measure restoration efforts (Fitzpatrick et al., 2021).

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Reviewers

Matt Hebert (The Nature Conservancy) and Sarah Matchett (Department of Fisheries and Oceans)

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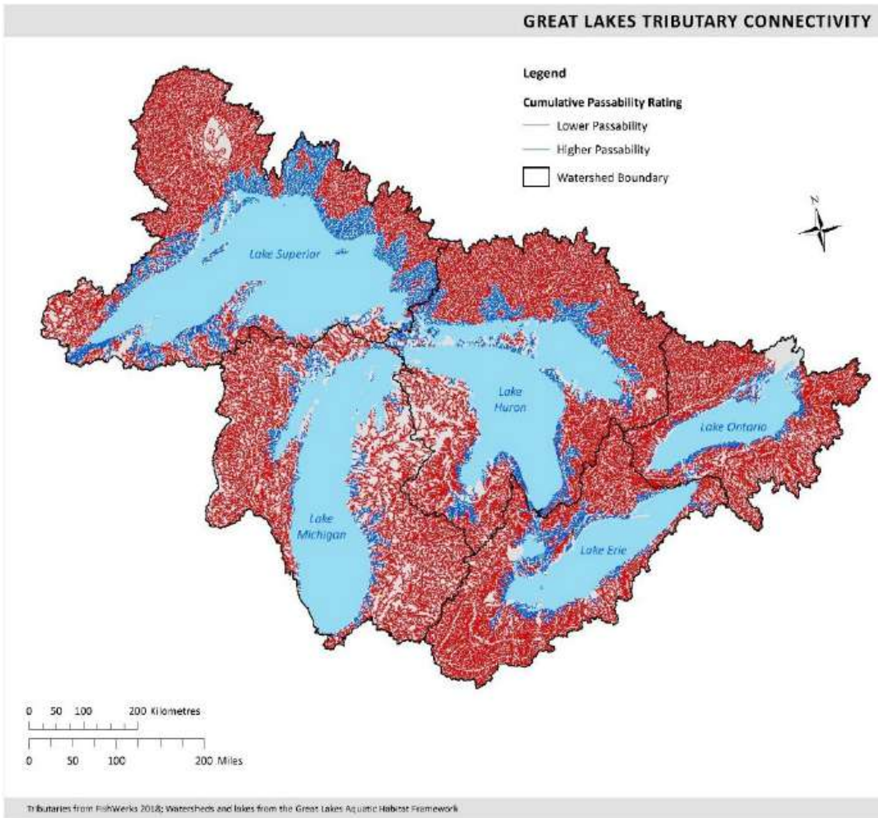
Table 1. Priority rivers and barriers (Great Lakes Fisheries Commission, 2021)

Lake	State/ Province	Priority River	Dam	Current Fish Passage
Superior	ON	Montreal	Andrews Generating Station	None
Superior	MN, WI	St. Louis	Fond du Lac Dam	None (Schram et al., 1999)
Superior	ON	Michipicoten	Michipicoten River Dam	None
Michigan	WI	Peshtigo	Peshtigo Dam	None (Daugherty et al., 2009)
Michigan	WI	Oconto	Stiles Dam	None (Daugherty et al., 2009)
Huron	Tbd[1]			
Huron	Tbd			
Huron	Tbd			
Erie	MI	Black River	Wingford Dam	None
Erie	MI	Huron River	Flat Rock Dam	Fish ladder installed in 1997 (Leonardi & Thomas, 2000)
Erie	ON	Grand River	Dunnville Dam	Fishway (Bunt et al., 2000)
Ontario	ON	Cobourg Brook	Pratt Dam	Fishway (Pratt et al., 2009)
Ontario	ON	Credit River	Streetsville Dam Norval Dam	Fish ladders. http://www.riverwatcherdaily.is/Migration

A)



B)

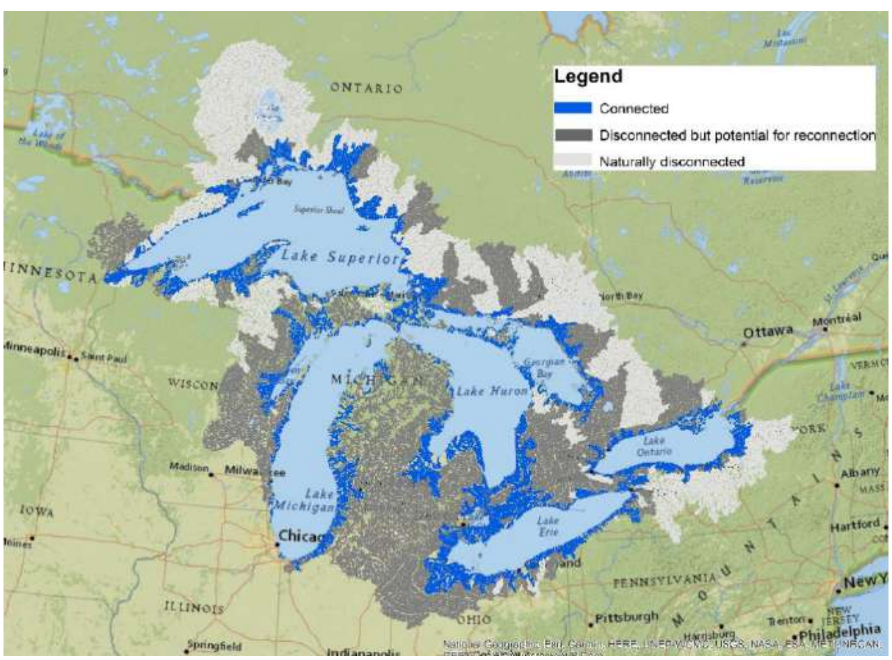


Figure 1. A) Dams and barriers in the Great Lakes basin (University of Wisconsin, 2018), and B) Watersheds that were historically disconnected from the Great Lake due to natural barriers (Khoury et al., 2018)

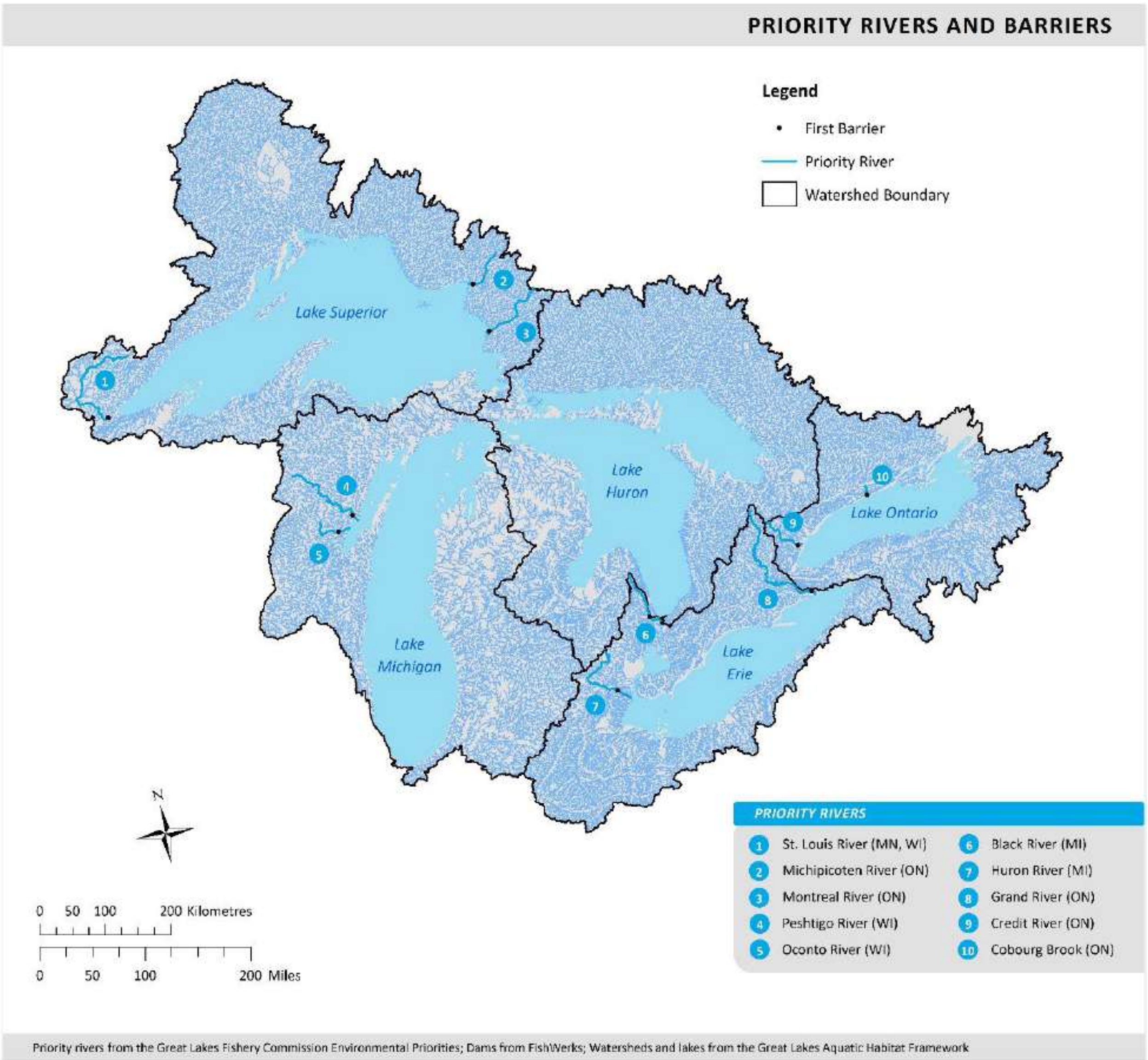


Figure 2. Priority rivers and barriers (Great Lakes Fisheries Commission, 2021)

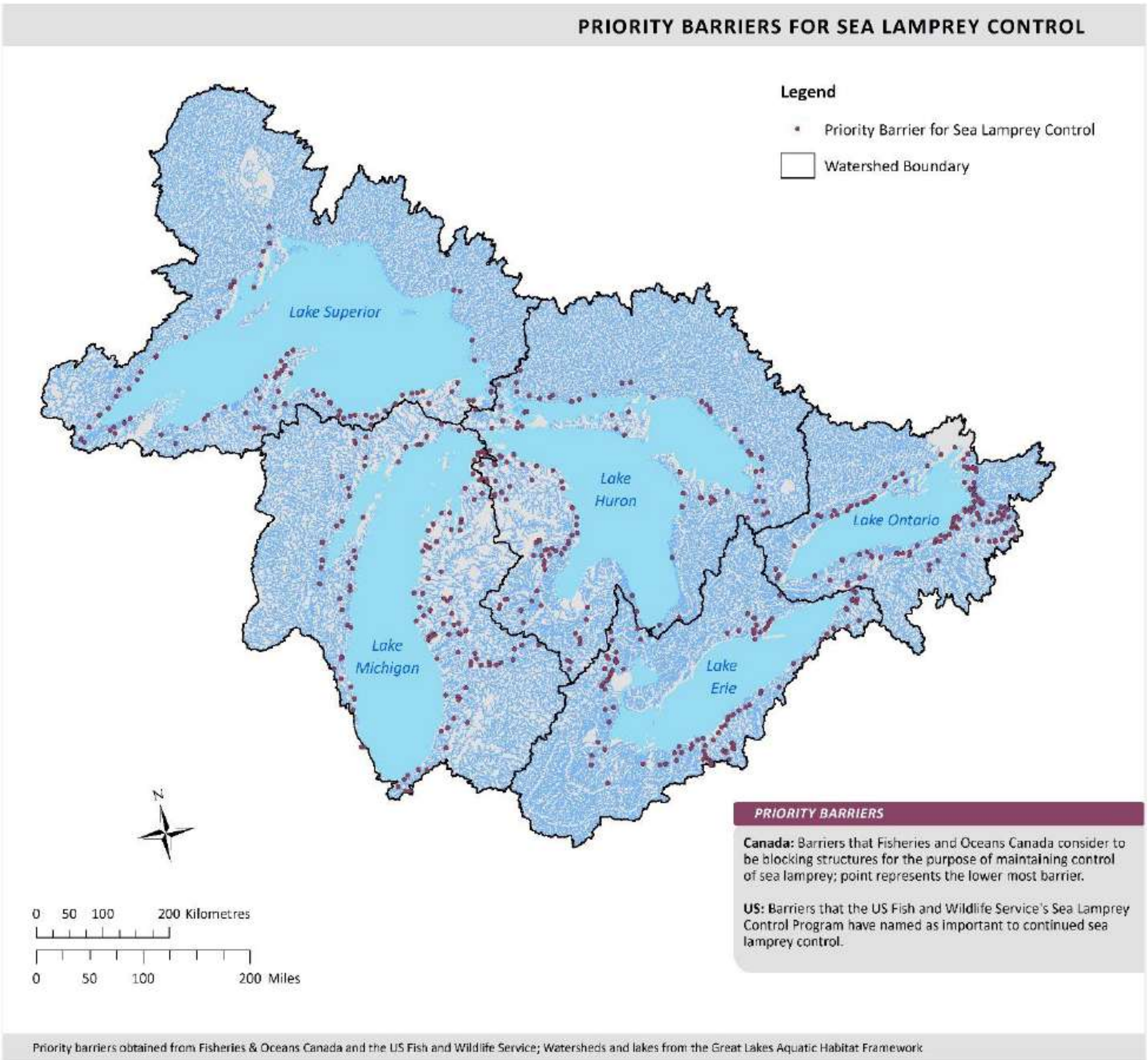


Figure 3. Priority barriers for Sea Lamprey control (DFO & USFWS, 2021)

Sub-Indicator: Phytoplankton

Overall Assessment

Status: Fair

Trends

10-Year Trend: Deteriorating

Long-term Trend (1950-2019): Deteriorating

Rationale: Phytoplankton are a critical food resource for zooplankton and small fish. Invasive mussels have caused algal reductions in Lake Michigan and Lake Huron, negatively impacting food webs of those lakes. Re-eutrophication has occurred in Lake Erie. Changes in Lake Superior and Lake Ontario are more subtle.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Deteriorating

Long-term Trend (1950-2019): Unchanging

Rationale: The lake has maintained a phytoplankton assemblage reflecting oligotrophic conditions, though nearshore occurrences of algal blooms are being recognized. Invasive species are not notably affecting phytoplankton, but there is evidence from paleolimnological data of gradual assemblage reorganization due to recent climate changes.

Lake Michigan

Status: Fair

10-Year Trend: Deteriorating

Long-term Trend (1950-2019): Deteriorating

Rationale: A reduction in phytoplankton and consequent diminution in seasonality has occurred. Lower levels of primary production could be reducing resources for higher trophic levels. Although the lake has a phytoplankton assemblage reflecting oligotrophic conditions, the designation of “Fair” is appropriate based on these observations.

Lake Huron

Status: Fair

10-Year Trend: Deteriorating

Long-term Trend (1950-2019): Deteriorating

Rationale: The lake has a phytoplankton assemblage reflecting oligotrophic conditions, which may appear to be "good" however, the change is likely due to the recent invasion by mussels that have reduced pelagic primary producers (negatively affecting invertebrate grazers).

Lake Erie

Status: Poor

10-Year Trend: Deteriorating

Long-term Trend (1950-2019): Deteriorating

Rationale: Re-eutrophication and proliferation of undesirable cyanobacteria is an increasing problem, particularly in the western basin. The central basin exhibits substantial spring diatom blooms indicating periodic eutrophic or mesotrophic conditions and contributing to summer hypoxia in the central basin.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1950-2019): Improving

Rationale: The lake has a phytoplankton assemblage reflecting mesotrophic to oligotrophic conditions. There is some evidence of assemblage changes due to invasive dreissenids, but the "unchanging" trend is assigned relative to overall water quality and health risk. Sporadic harmful blooms have been noted in the St. Lawrence River, though more data are needed to determine whether this is an increasing trend.

Status Assessment Definitions

Good: the phytoplankton assemblage reflects the appropriate trophic structure for the lake.

(Good = Oligotrophic for Lakes Superior, Huron and Michigan; Oligotrophic to Mesotrophic conditions for Lakes Ontario and Erie.)

Fair: the phytoplankton assemblage is moderately/periodically reflecting the appropriate trophic structure for the lake.

Poor: the phytoplankton assemblage does not reflect the appropriate trophic structure for the lake.

An appropriate condition is defined relative to known or inferred baseline conditions for a lake, including natural levels of nutrients and algae.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: changes in phytoplankton biovolume, density and species composition in support of a healthy food web.

Unchanging: phytoplankton characteristics as they relate to food web and water quality are not showing an overall change in trend.

Deteriorating: changes in phytoplankton biovolume, density and species composition resulting in an unhealthy food web.

Undetermined: phytoplankton data are reflecting conflicting trends or data are insufficient to make an assessment.

Endpoints and/or Targets

For the purposes of SOGL phytoplankton as an indicator largely refers to inferred trophic status or algal abundance. Lake-specific trophic status that reflects baseline conditions has been determined from paleolimnological analyses (oligotrophic conditions for Lakes Superior, Huron and Michigan; oligo-mesotrophic conditions for Lakes Erie and Ontario) so there is potential to set remedial targets. However, re-attaining historical phytoplankton assemblages (or "desirable" species) is not realistic because of the fundamental reorganization of the algal communities that are likely not recoverable under current physical, chemical, and biological regimes. More general terms like "oligotrophic" are suitable, though geographic context (i.e., lake) is often important to identify more specific issues that are driven by stressors beyond nutrient pollution. It is not realistic to derive endpoints for stressor impacts related to invasive species impacts and climate change. Instead, scientific inquiry is required to interpret the unique, lake-specific trends related to these variables, and base current conditions on whether trajectories are desirable. For instance, the oligotrophication and reorganization of phytoplankton in Lake Huron may be interpreted as lower nutrient stress, but instead is a negative food web impact resulting from the invasion of profundal dreissenids and likely a warming climate.

Sub-Indicator Purpose

The purpose of this sub-indicator is to directly assess phytoplankton species composition, biomass, and primary productivity in the Great Lakes, and to indirectly assess the impact of stressors on Great Lakes water quality, lower food webs, and potential threats to human and environmental health. This includes inferring impacts from pollution, invasive non-native species, climate change, and harmful algal blooms.

Ecosystem Objective

1. Maintain trophic states with phytoplankton biomass and composition consistent with a healthy aquatic ecosystem in open waters of the Great Lakes. Desired objectives are phytoplankton biomass and community structure indicative of oligotrophic conditions (i.e. a state of low biological productivity, as is generally found in the cold open waters of large lakes) for Lakes Superior, Huron and Michigan; and of mesotrophic (or oligotrophic) conditions for Lakes Erie and Ontario.
2. Qualitatively and quantitatively detect and predict changes in phytoplankton biomass and composition and apply those changes to stressor impacts or recovery. Desired outcomes are maintenance of good condition over several years or a detectable transition to healthy conditions.
3. This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement (The Government of the United States of America and the Government of Canada 2012) which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species." Also, as an indicator at the bottom of the food chain phytoplankton are capable of detecting subtler ecosystem changes, so Article 2(1)(b) of the GLWQA ("develop programs, practices and technology necessary for a better understanding of the Great Lakes Basin Ecosystem") applies.

Measure

The measures used to compile indicator results will include the following.

- Biovolume and density of phytoplankton and taxonomic composition for spring and late summer in the offshore, isothermal waters (e.g., Reavie et al. 2014a). Taxonomic identification is to the best detail possible (e.g., species) to identify atypical or non-native taxa. Abundances of taxa (e.g., cyanobacterial or diatom blooms) will reflect conditions relative to known historical condition within a lake. Sampling and analysis are standardized across all lakes.
- All available historical data on phytoplankton that are sufficiently comparable to allow for long-term reconstructions of trends and trajectories of primary producers. Such data are available from past monitoring (e.g. Munawar & Munawar 1978) and paleolimnological (e.g. Reavie et al. 2017, 2021) assessments and may include bulk biomass measurements (e.g. chlorophyll a), biovolume (e.g. μm^3/mL) and assemblage composition. Paleolimnological data allow for historical inferences as far back as the 19th century, and such data were used to support the newer "long-term trend" requirement in this report.

Ecological Condition

Phytoplankton composition is an important indicator of water quality and biological condition. Several qualitative indicators also exist: the abundance of cyanobacteria is a clear indicator for nutrient stress; reductions in algal abundance signal dreissenid-driven oligotrophication; and phytoplankton assemblage changes reflect changes in pelagic ecology due to climate change, grazer composition and other factors. Phytoplankton play a critical role in food web processes. Production (energy) sinks from the surface euphotic zone to nourish the benthos. It may flow efficiently, with high productivity across the size-spectrum, or it may accumulate, negatively affecting water quality while little energy reaches top predators; i.e., the current situation in summer in the western basin of Lake Erie.

Phytoplankton are photosynthesizing organisms that inhabit the sunlit layer of aquatic systems. They are microscopic algae that typically form a major component of a lake's primary producers, thereby sustaining the aquatic food web. The amount and taxonomic structure of phytoplankton populations can be related to anthropogenic stressors, thereby permitting inferences to be made about lake condition and change (Stoermer 1978). Recently, the most important, comprehensive data sources for phytoplankton-based assessments have been time series data on phytoplankton community size and composition (e.g., Reavie et al. 2014a; Figure 1), satellite-based measurements of chlorophyll (e.g., Barbiero et al. 2012, Binding et al. 2021, Lekki et al. 2019) and recent paleolimnological studies of fossil phytoplankton (e.g., Shaw Chraïbi et al. 2014, Sgro et al. 2018, Reavie et al. 2021; Figure 2). Additional phytoplankton data have been collected by The US Coast Guard (e.g., Oyserman et al. 2012; winter collections from icebreakers) and Canadian agencies, such as that for Lake Erie winter conditions (Twiss et al. 2012; Environment and Climate Change Canada 2015).

Status of the Great Lakes ecosystem as a whole is characterized as *fair* although condition and trends vary among lakes. Invasive mussels have caused reductions in phytoplankton (algae) in Lake Michigan and Lake Huron, negatively impacting food webs of those lakes. Re-eutrophication has occurred in Lake Erie in the last 1.5 decades, mainly indicated by cyanobacterial blooms that are occurring with greater frequency in the western basin of Lake Erie and large spring diatom blooms in the central basin that support hypolimnetic hypoxia (Reavie et al. 2016). Slower, long-term changes are occurring in Lake Superior and Lake Ontario, but these changes are not yet well understood. However, with the exception of Lake Erie, trophic status across the basin would generally be considered good. For the most part, trends herein reflect compiled datasets from 2001 through 2016. "Long-term" inferences are supported by older collections and paleolimnology.

Assigning firm condition assessments was also complicated in individual lakes. Consider Lake Michigan and Lake Huron, for instance: if trophic status was the only factor considered their low phytoplankton abundance would superficially reflect good conditions. However, the periodic, mussel-driven depletion of phytoplankton in these lakes represents food web stress. Further, paleolimnological trends indicate a longer-term shift to small-celled diatoms in these lakes (Reavie et al. 2017), a likely climate-driven food web disruption though the impacts of this shift on ecosystem health is not yet understood. From an ecological perspective that simultaneously considers multiple parameters *fair* remains an appropriate assessment. It is also worth noting that longer-term data from paleolimnological assessments of phytoplankton pigments (Figure 2) indicate that some of the Great Lakes system has improved on a 50-year scale.

The previous State of the Great Lakes report noted the rapid changes that occurred in the phytoplankton community of several Great Lakes in the decade prior. In general, these changes are continuing, or the lakes remain in the previously reported "changed" state. In association with the dreissenid advance, the spring phytoplankton bloom in Lake Huron, which practically disappeared in 2003 (Barbiero et al. 2011) remains absent, though monitoring data (Figure 1d,f) indicate that summer algal abundance may be returning to pre-invasion levels. Declines in the spring bloom were also seen in Lake Michigan (Reavie et al. 2014a; Figure 1g,i). Such trends of oligotrophication can be viewed positively, but it likely also represents an overall reduction in the carrying capacity of the two lakes, as evidenced by coinciding losses of invertebrates and reductions in fish energy content (Pothoven and Fahnenstiel 2014).

It is unlikely that the deep waters of Lake Superior will be anything but oligotrophic, so in that context it will remain in good condition. However, the lake's phytoplankton assemblage continues to change over decadal timescales, likely associated with atmospheric warming that is changing the physical properties of the lake (Shaw Chraïbi et al. 2014). Such a shift has now been recognized across all of the Great Lakes and their sub-basins (Reavie et al. 2017), so such longer-term changes in primary producers should continue to be observed to determine future impacts on food webs. Recent observations of nearshore blooms (Sterner et al. 2020, Alexson et al. 2018) and increases in chlorophyll a in surface sediments (Figure 2) suggest there may be increasing threats from algal blooms that are not apparent in pelagic collections.

In the western basin of Lake Erie, blooms of the nuisance algae *Microcystis* (among other cyanobacteria) have become prominent in late summer most years (Michalak et al. 2013). The spring algal bloom in the central basin, largely attributed to filamentous, centric diatoms (Reavie et al. 2016, Twiss et al. 2012, Figure 1m) is likely contributing substantial organic biomass to the hypolimnion and exacerbating hypoxia, though a reduction in this bloom is apparent in the most recent few years, possibly a result of extreme winter conditions that may not have been conducive to a robust winter-spring diatom bloom.

Over the last decade in Lake Ontario spring chlorophyll levels have remained stable, but there is evidence of a slight summer chlorophyll increase (Figure 2, and USEPA, unpub. data) since declines seen in the 1980s (Johengen et al. 1994). This corresponds with recent changes in Lake Erie, albeit at a smaller scale for Lake Ontario. While current trophic condition in Lake Ontario remains better than that during the mid- to late-20th century, future conditions in Lake Ontario should be observed carefully to ensure re-eutrophication does not occur, as may be suggested by very recent sedimentary concentrations of chlorophyll a (Figure 2), and as has been the case in Lake Erie. The St. Lawrence River, adjacent to Lake Ontario, has had periodic harmful algal blooms of cyanobacteria (Hudon et al. 2014), a phenomenon that should be observed closely to determine whether it represents an increasing threat.

Linkages

Linkages to other indicators in the indicator suite include:

- Nutrients and Dreissenid Mussels – it is well known that the phytoplankton population and its productivity changes with anthropogenic pollution. The ecosystem changes are reflected by the change of phytoplankton composition and productivity. For example, Lake Superior represents an oligotrophic ecosystem and is widely considered to be in the best condition of the Great Lakes. Similarly, Lake Erie's phytoplankton composition, which was once eutrophic, dramatically changed to seasonally meso-oligotrophic status due to phosphorous abatement and the invasion of zebra mussels, a trophic trend that has since reversed to indicate re-eutrophication. A great deal of recent data are available for phytoplankton biomass, composition and primary productivity which will reflect the overall ecosystem health including grazing pressures of non-native filter-feeders and bottom-up influences from nutrients.
- This sub-indicator also links directly to the other sub-indicators in the Habitat and Species indicator, such as invertebrate grazers that rely on phytoplankton as a primary food resource. The cycling of phosphorus is being driven by catchment inputs and sedimentary processes, impacting the food web and having implications on many forms of aquatic life, especially benthos, zooplankton, and phytoplankton. Effects on fish communities are less direct but must also be considered.
- Climate is driving changes in Great Lakes phytoplankton. This is not surprising given the known long-term changes in the surface temperatures of all of the lakes. Warming surface waters alter lake physical structure, especially the length of the ice-free period and the size and integrity of the summer epilimnion. The phytoplankton are the first biological component of the Great Lakes that have been noticeably affected by recent atmospheric warming (Reavie et al. 2017) but we have yet to determine the future implications of these changes. Recent climate warming-driven reorganization of Great Lakes phytoplankton communities should be further studied to identify possible outcomes for higher organisms that rely on stable food resources.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada		X		
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://cdx.epa.gov/		

Data Limitations

Phytoplankton taxonomy (microscopic identification and enumeration) is a highly specialized and time-consuming activity that requires intensive training and experience. However, if properly done the phytoplankton analysis generates scientific, precise, and reliable species data that helps to identify current community composition and when compared to historic data, it can be used to determine community shifts or the sensitivity of phytoplankton to anthropogenic stressors. Expanding phytoplankton sampling would be beneficial as the majority of interpretations are based on two synoptic surveys each year. For instance, we are missing winter, mid-summer, and late fall conditions, all of which are likely to be important to fully understand year-round food web characteristics.

The study of lower trophic levels and their use as indicators has been largely ignored in Great Lakes coastal environments, though localized assessments in Canada (e.g. Currie et al. 2015) and the US (e.g. Luo et al. 2017) exist. To understand the role of phytoplankton in such a rapidly changing ecosystem further evaluation of the microbial loop may be beneficial, including the base of the food chain ranging from bacteria, viruses, fungi (e.g., chytrids), heterotrophic nanoflagellates, autotrophic picoplankton, and ciliates up to phytoplankton (nanoplankton and microplankton-netplankton). This could be further supplemented with eDNA and fluorometric pigment analysis (currently being explored by the USEPA during sampling cruises) that could identify, for instance, changing proportions of cyanophytes. Also, now that climate-driven changes have been observed in the phytoplankton, the mechanisms of the climate-biology relationships need to be clarified to make predictions of future conditions. Furthermore, extension of long-term characterization of primary producers (monitoring, paleoecology) to coastal environments (nearshore, embayments, wetlands) would help to identify links between anthropogenic activities in catchments and lake condition. The prevailing observation of algal blooms in shallower, nearshore environments, such as Lake Huron that has dramatically dichotomous deep (oligotrophication) and embayment (algal bloom) conditions, suggests a great deal of further study is needed to characterize phytoplankton as an indicator of ecosystem health.

While there is a well-established long-term monitoring program for the deep waters of the lakes, data are lacking for the connecting channels and their associated lacustrine systems: St Marys River, Lake George, and Lake Nicolet upstream of Lake Huron; Lake St. Clair, the Detroit River, and the St. Clair River upstream of Lake Erie; The Niagara River upstream of Lake Ontario. Intermittent reporting on paleolimnological or short-term monitoring provide some insight into the condition of the connecting channels. Baustian et al. (2020) determined via paleolimnology at two locations in Lake St. Clair that phytoplankton history was profoundly affected by decades of agricultural inputs and the more recent mussel invasion. As of 1993 the phytoplankton community of Lake George was considered to be in decline (Reavie et al. 2005). Rozon et al. (2018) determined that no impairments were apparent based on plankton collections in the Niagara River. In most cases, however, river collections of phytoplankton have been sporadic or not performed for decades. Given the population centers associated with these connecting waters a more thorough phytoplankton monitoring program may be a beneficial contribution to the overall SOGL reporting.

Additional Information

Objective, quantitative mechanisms for evaluating ecosystem health from phytoplankton are gradually being developed. For instance, nutrient optima and tolerances for indicator species are now available for the Great Lakes (Reavie et al. 2014b), thereby allowing quantitative reconstructions of water quality variables from assemblage data.

The U.S. EPA has an active program for phytoplankton collection and analysis in the pelagic regions of all Great Lakes in spring and summer and other, more localized programs are ongoing (e.g. Fahnenstiel et al. 2010). Satellite imagery has also enabled the detection of chlorophyll trends in the surface waters of the Great Lakes (e.g., Kerfoot et al., 2010), and these data can provide a broad overview of algal abundance. A greater synthesis of long-term satellite data across the Great Lakes basin is needed to supplement concurrent phytoplankton collections and paleolimnological results.

The main purposes of this sub-indicator are to: (1) measure lake condition based on the biological character of primary producers; (2) measure condition relative to driving stressors including changing water quality, nutrient loading and cycling, and invasive species abundance; (3) evaluate direct problems (e.g., blooms) associated with phytoplankton; (4) indirectly evaluate the trophic efficiency of the food web at transferring algal production to fish. As a sensitive indicator of changes in primary producers due to various drivers (invasive species effects, nutrients, climate, etc.), phytoplankton provide information on the effects of multiple stressors.

To enhance this sub-indicator temporal gaps should be filled by year-round collections similar to those collected by the Lake Guardian, the U.S. Environmental Protection Agency's research vessel. Some lake-specific collections are occurring at times not previously sampled (e.g., Twiss et al. 2012 for Lake Erie), and in specific nearshore areas going back to the 1960s (i.e., water intakes; Ontario by the Ministry of Environment, Conservation and Parks) (Winter et al. 2015) but remote, automated sampling stations could play an important role in more comprehensive year-round sampling.

Acknowledgments

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Figure 1. Histograms of phytoplankton biovolume and community composition in the Great Lakes basins from 2001 through 2018. Bars represent averages across all pelagic samples, with error bars calculated from sample totals. Bars are divided according to algal groups. Major noteworthy trends include: early-2000s declines in phytoplankton abundance in Lake Huron and Lake Michigan (particularly in spring and attributed to diatom loss); and increases in spring and summer phytoplankton in central and western Lake Erie (mainly attributed to increases in spring diatoms and summer cyanobacteria). Revised and updated from Reavie et al (2014a).

Figure 2. Sedimentary chlorophyll a from 12 cores collected throughout the Great Lakes. Colored lines represented core profiles are generalized additive models of downcore measurements. Profiles indicate the prevailing eutrophication in the late 20th century (mainly lakes Michigan, Erie and Ontario), followed by reductions in phytoplankton largely starting in the 1990s, and spikes in chlorophyll a during the uppermost ~15 years in lakes Erie, Ontario and Superior. Figure is modified from (Reavie et al. 2021). Divisions in the lakes denote "basins" with unique phytoplankton and water quality conditions.

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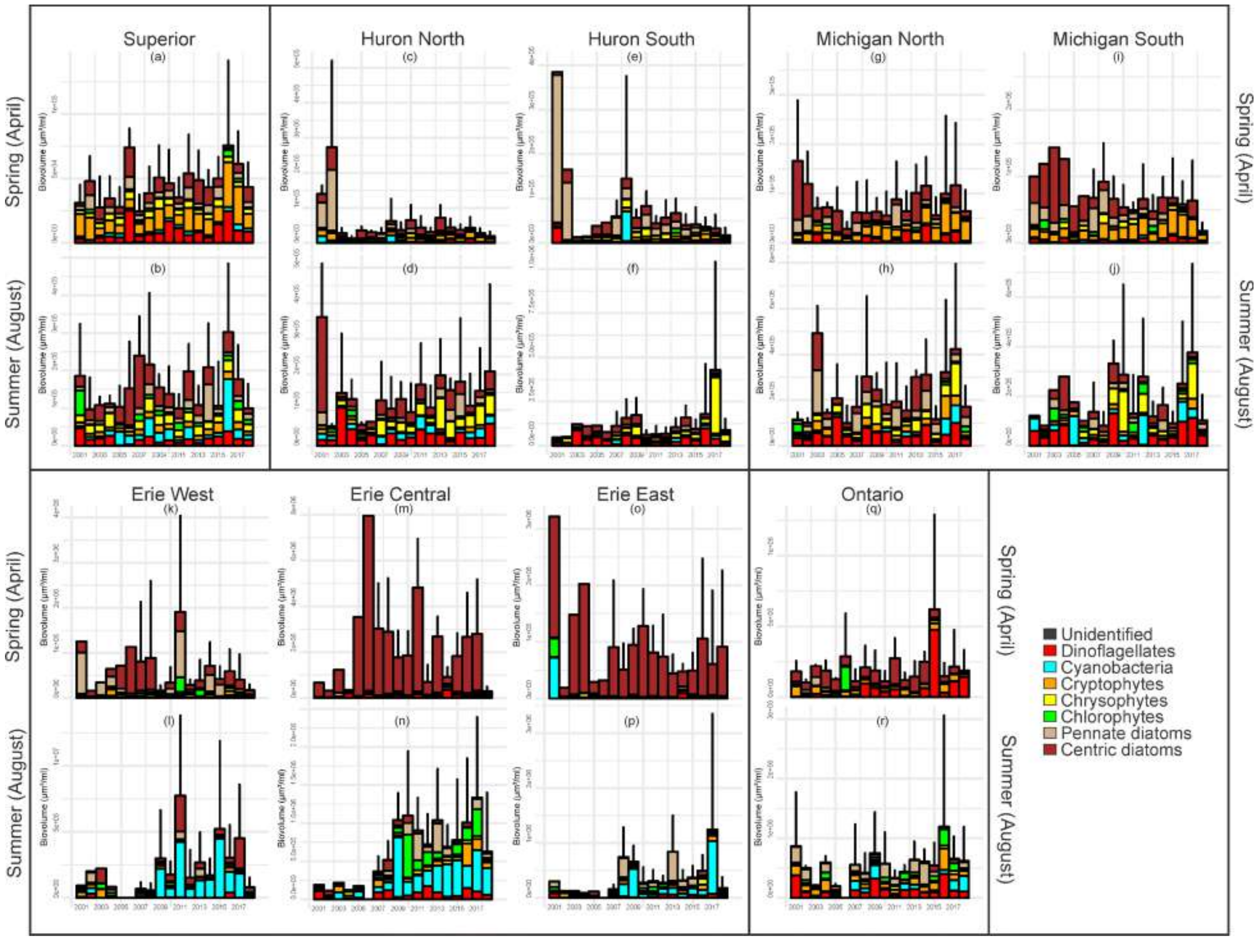

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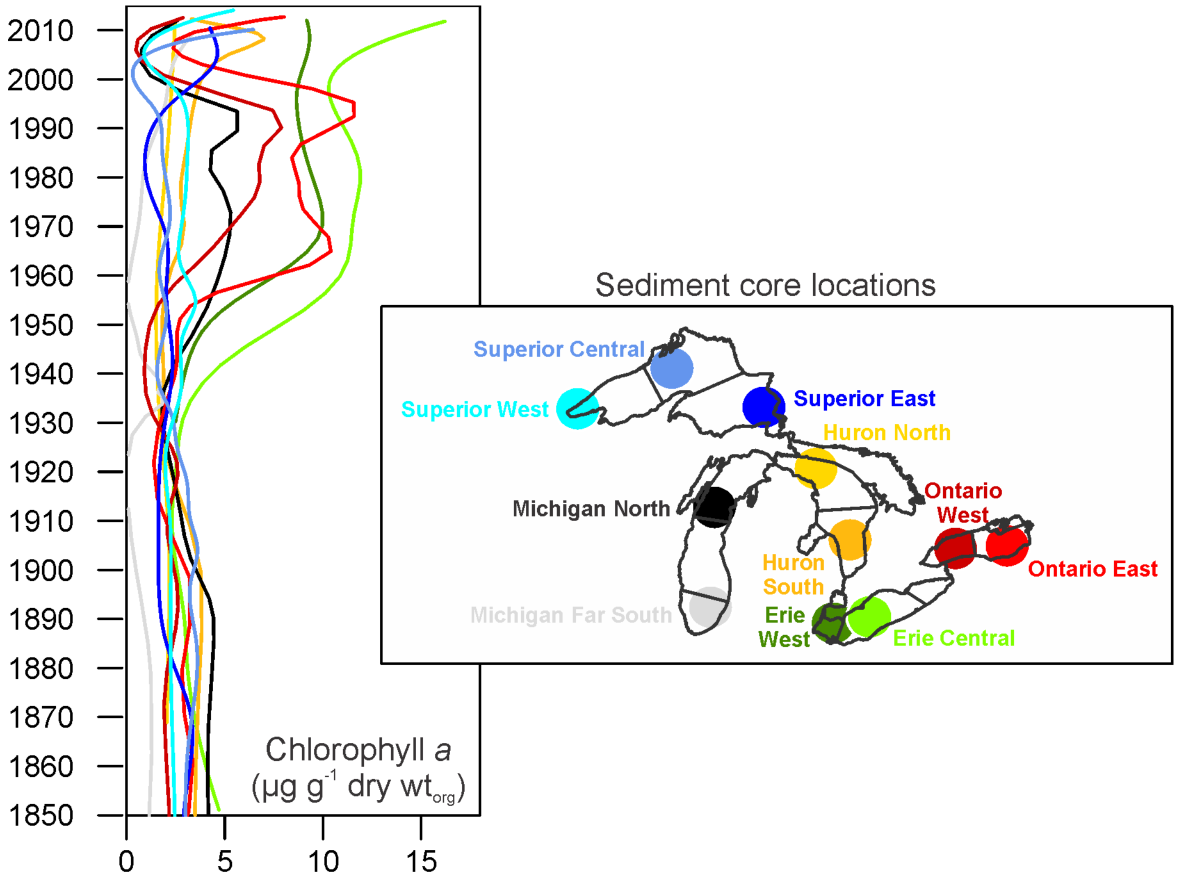

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Sub-Indicator: Zooplankton

Overall Assessment

Status: Good

Trends:

10-Year Trend: Unchanging

Long-term Trend (1997-2019): Undetermined

Rationale: The oligotrophic zooplankton community of Lake Superior did not change from 1997 to 2019. Lake Huron experienced a well-documented decrease in zooplankton biomass in 2003, particularly for cladocerans. Similar but smaller decreases in biomass and community shifts occurred at this time for Lakes Michigan and Ontario. Lake Erie zooplankton biomass has been highly variable over the time series but recently (since 2013) biomass increased in the Central and Eastern basins including *Daphnia*.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1997-2019): Unchanging

Rationale: Consistent oligotrophic zooplankton community dominated by calanoid copepods and maintenance of low but sustained zooplankton biomass near 2-3 g m^{-2}

Lake Michigan

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1997-2019): Deteriorating

Rationale: The oligotrophic zooplankton community has been dominated by calanoid copepods since the early 2000s. Decreases in zooplankton biomass with loss of cladocerans was evident in 2004.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (1997-2019): Deteriorating

Rationale: The decrease in zooplankton biomass in 2003, particularly of cladocerans, to levels below that of Lake Superior (< 2 g m^{-2}) was sudden and likely represents levels limiting to forage fish. The decadal status has not changed, although the long-term trend remains of concern.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Good

10-Year Trend: Improving

Long-term Trend (1997-2019): Unchanging

Rationale: Zooplankton biomass has steadily increased, including important *Daphnia* in the Central Basin and the Eastern Basin, since 2013.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1997-2019): Unchanging

Rationale: Over the past decade, overall zooplankton biomass and community structure has been unchanging. Over the entire time series, biomass has decreased slightly over time and the community has shifted toward calanoid copepods, with losses of cyclopoid copepods and variable abundance of cladocerans. The decrease in biomass has not reached levels of concern for forage fish.

Status Assessment Definitions

The current available GLNPO time series is 1997-2019. New data added in this report is for 2017-2019.

Good: Zooplankton biomass and community structure consistent with target TP concentrations of specific deep lakes (higher than 2 g m^{-2} in Lakes Superior, Huron and Michigan, 3 g m^{-2} in Lake Ontario). For shallow Lake Erie, 2 g m^{-2} in Eastern and Central Lake Erie, and 1 g m^{-2} in Western Lake Erie.

Fair: Evidence of change in biomass, average size, or community structure away from the desired lake-specific goals.

Poor: Offshore zooplankton biomass below 1 g m^{-2} in Lakes Superior, Huron and Michigan and 2 g m^{-2} in Lake Ontario. For Lake Erie, below 1 g m^{-2} in Eastern and Central Basin and 0.5 g m^{-2} in the Western Basin. Such low values may limit forage fish populations.

Undetermined: data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: recovery to sustaining levels of biomass and return to original native community composition.

Unchanging: no significant changes in biomass or composition, generally a good thing unless in restoration process.

Deteriorating: zooplankton biomass declining well below goals, low prevalence of large *Daphnia* species (and perhaps small overall average size) due to high fish predation, high prevalence of non-native species such as predatory cladocerans.

Undetermined: metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

Due to the bottom-up connection of nutrients (Total Phosphorus - TP) with algal concentration (Chl a) and zooplankton carrying capacity, targets for areal zooplankton biomass (measured in dry weight) should reflect the International Joint Commission (IJC) goals for spring total phosphorus concentrations (Chapra and Dolan 2012). Thus, lakes with lower target TP concentrations (e.g. Lake Superior and Huron at phosphorus concentrations of 5 µg l^{-1} and Lake Michigan at 7 µg l^{-1}) will have a lower target zooplankton biomass (2 g m^{-2}) than lakes with higher target TP concentrations (e.g. Lake Ontario at 10 µg P l^{-1}) which will have a target offshore zooplankton biomass of 3 g m^{-2}. Although Lake Erie has a similar TP target as Lake Ontario, it is a much shallower lake. Therefore goals set for whole water column zooplankton biomass are lower with 2 g m^{-2} for the Eastern (40 m) and Central Basin (20 m) and 1 g m^{-2} for the Western Basin (10 m).

Sub-Indicator Purpose

The offshore zooplankton biomass sub-indicator assesses the standing stock and community composition of zooplankton in the Great Lakes over time and space.

Changes in the offshore zooplankton biomass sub-indicator reflect influences from both bottom-up (primary production) and top-down (vertebrate or invertebrate predation) mechanisms as well as energy transfer across trophic levels. The purpose of this sub-indicator is to contribute to the measurement of the stepwise trophic efficiency of the food web at transferring algal production to fish. Zooplankton biomass has often been used to explain deviations in the relationship between nutrients (total phosphorus, TP) and phytoplankton biomass (Chl a) (Taylor and Carter 1997).

Ecosystem Objective

Maintain and support a healthy and diverse fishery; maintain trophic states consistent with the lake-specific goals – oligotrophic Lakes Superior, Huron, Michigan, and Ontario, and mesotrophic Lake Erie. Zooplankton represent an important trophic link from primary production to fish and thus abundant zooplankton tend to improve water quality and produce more fish.

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement that states that the Waters of the Great Lakes should “support healthy and productive wetlands and other habitats to sustain resilient populations of native species.”

Measure

The primary offshore zooplankton index is average lakewide or basinwide areal biomass (g dry weight/m^2). This value can be expressed as summer mean biomass (July-August) or as growing season mean biomass (e.g. April 1-October 31) if more frequent sampling is available. Standard U.S. EPA Great Lakes National Program Office (GLNPO) collection protocols call for vertical net tows from 2 m above the bottom or the top 100 m, whichever is less, with a metered 0.5-m diameter mouth net and 153-µm mesh (U.S. EPA GLNPO LG 402). Zooplankton dry weight biomass is estimated using length measurements in standard sets of length-dry weight equations available for each taxa (U.S. EPA GLNPO LG 403). Within the “Additional Information” section of this report, we propose the future addition of a mysid indicator. GLNPO samples for analysis of mysids were collected at stations > 30 m that are visited during night (1 hour after sunset to 30 min before sunrise) using whole water column tows with a 1-m diameter net with 500 µm mesh at the top and 250µm mesh at the bottom and cod end. Mysid areal dry weight biomass is reported herein as additional context, although it is not explicitly included in the indicator assessment.

Several Canadian/US state/provincial and federal agencies routinely collect offshore zooplankton samples in each Great Lake. U.S. EPA GLNPO collects offshore samples for all five Great Lakes in April and August. GLNPO samples at eight to twenty offshore stations in each lake using the same method and has been the preferred data source for this sub-indicator. Data are now available for the period 1997-2019.

Ecological Condition

Summer biomass of crustacean zooplankton communities in the offshore waters of Lake Superior has remained at a relatively low but stable level near 2-3 g m^{-2} since at least 1997 (Figure 1). The plankton community is dominated by large calanoid copepods (*Leptodiaptomus sicilis* and *Limnocalanus macrurus*) that are characteristic of oligotrophic, coldwater ecosystems.

Changes observed in the zooplankton communities of Lakes Huron and Michigan, and to a lesser extent Lake Ontario (Figures 2, 3 and 4), are consistent with expected responses to the reductions in nutrient levels seen in all three lakes. These changes could represent a consequence of nutrient reduction activities, perhaps compounded by effects of dreissenid mussels. The reductions in cladocerans in Lakes Huron and Michigan, along with continued declines in populations of the benthic amphipod *Diporeia*, could represent a decreasing food base for forage fish. However, exact mechanisms of these declines, and the relative strength of bottom-up versus top-down forcing, have yet to be determined.

Over time, zooplankton indicators of Lakes Huron and Michigan have converged towards Lake Superior in terms of biomass levels and community composition (Barbiero et al. 2012 and 2019). The community shift towards dominance by calanoid copepods is consistent with increased oligotrophication (Gannon and Stemberger ,1978). In 2003, zooplankton biomass in Lake Huron fell below that of ultra-oligotrophic Lake Superior (Barbiero et al. 2011). Bunnell et al. (2014) highlighted this decline as a potential explanation for the concurrent collapse of alewife and reduced growth of salmonids (Riley et al. 2008). Here we show that there has been little additional change since 2003 in Lake Huron.

Lake Ontario has not experienced recent declines in primary production, suggesting that top-down control may better explain the observed zooplankton community shifts in this lake. There has also been a change away from a long-term community of cyclopoid copepods and *Daphnia retrocurva* to a community composed of calanoid copepods and *Daphnia mendotae*, which may be related to low alewife abundance and a rise in invertebrate predation by the predatory cladoceran *Bythotrephes* (Barbiero et al. 2014, Rudstam et al. 2015).

The zooplankton community of Lake Erie is taxonomically diverse and rich in native and non-native cladoceran species (Figures 5-7). The low abundance of deep dwelling calanoid *Limnocalanus macrurus*, and the overall maintenance of cladocerans relative to calanoids in Lake Erie, can be attributed to the shallow bathymetry as well as the lake's mesotrophic state. Zooplankton biomass has been highly variable among years in all three basins, although there have been recent increases in biomass in the central and eastern basins.

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Other Habitat and Species sub-indicators (phytoplankton, benthic community, and prey fish diversity). Zooplankton consume phytoplankton, and thus respond to changes in algal biomass. Zooplankton also respond to changes in their predators that include both fish and invertebrates. Both zooplankton and benthic invertebrate community structure influences the prey fish community biomass and composition.
- Nutrients in Lakes (open water) – phosphorus levels regulate primary productivity by phytoplankton and thus impact food availability for zooplankton.
- Increased water clarity shifts primary production to deeper depths in the form of deep chlorophyll layers (DCL, Scofield et al. 2020), which affects food availability for zooplankton.
- Interannual changes in water temperatures and ice cover associated with climate change could lead to impacts on the distribution of cold-water fauna such as large bodied calanoid copepods that have become increasingly prevalent in the upper Great Lakes. Water levels and changes in precipitation are less likely to affect zooplankton.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://cdx.epa.gov/		

Data Limitations

Both U.S. and Canadian agencies conduct zooplankton monitoring programs. Note that sampling and analysis methodologies and biomass calculations differ, highlighting the importance in consolidating efforts across agencies to increase the ability for data sharing in the future.

Many historic time series are based only on epilimnetic sampling that miss deeper dwelling zooplankton, particularly for increasingly oligotrophic offshore settings and in lakes with higher water clarity that have zooplankton residing in deep water during the day and migrating to surface water during the night. Those data are not being utilized in this assessment; only deeper net tow data were used (100 m or 2-m off the bottom, whichever is shallower), which limits the time series to years with deep tow data.

Several taxa-based ratios (e.g. calanoids to cyclopoids+cladocerans) have been proposed but are not always consistently presented (i.e. by density or biomass). Several of the proposed indices have not been fully tested for Great Lakes or in long term datasets across many lakes, and thus are not appropriate to use at this time; such alternate indices may be used in the future if additional validation suggests they would add value to this assessment.

The biomass of dreissenid mussel larvae (veligers) in the zooplankton community can vary dramatically in time and space. Many current monitoring programs focus only on crustacean zooplankton. The small size of veliger larvae requires sampling with a 64 micron net to place this additional biomass in the context of the rest of the zooplankton community.

Additional Information

Note that we use areal biomass (g m^{-2}) rather than volumetric (g m^{-3}) units to better evaluate the overall standing biomass of these lakes for connecting to fish production potential (Bunnell et al. 2014). A change to using areal units as indicators was done in 2015. Whole water column (in this case maximum of 100 m) tows in deep lakes include large strata of hypolimnion that have few zooplankton. Volumetric biomass estimates from 100 m net tows are thus “diluted” relative to shallower lakes that have less hypolimnion. Areal biomass represents the zooplankton biomass found within a one meter square water column. Note that for Lakes Superior, Michigan, and Ontario, most offshore GLNPO sites are > 100 m but many of the sites for Lake Huron are < 100 m. In Lake Erie, depths range from 10 m in the Western to 20 m in the Central to 50 m in the Eastern basins.

More information could be included in this sub-indicator in the future. Including a sub-indicator based on crustacean zooplankton community structure, focusing on calanoid copepods, would be useful. Mean body size and species composition of zooplankton are also sensitive indicators of predatory pressure by planktivorous fish and large invertebrates (*Mysis* and predatory cladocerans). Nearshore measures for the Zooplankton sub-indicator have also been proposed. Several long-term nearshore biomonitoring programs that focus on zooplankton exist (e.g. Lakes Erie and Ontario) and time series have detected changes that can complement offshore trends in biomass and community composition. For example, the Lake Ontario Biomonitoring Effort (NY DEC Regional Offices, USGS, USFWS and Cornell) collects nearshore (10 m depth) samples biweekly throughout the growing season at several sites along the south shore of Lake Ontario (Holeck et al. 2020).

A future *Mysis* sub-indicator could be supported using the GLNPO time series but need further development. Mysids represent a large proportion of overall zooplankton biomass and thus represent an important prey of planktivorous fish. For example, mysids can represent up to 30% of the total crustacean zooplankton biomass in Lake Ontario, 15% in Lake Superior, 10% in Lake Michigan, 3% in Lake Huron, and less than 1% in Lake Erie (Jude et al. 2018). These animals reside on the bottom during the day and therefore are not sampled with daytime net tows. Existing monitoring programs for mysids are based on night net collection and have only been conducted consistently since 2006 or 2007, depending on the lake (Jude et al. 2018). U.S. EPA GLNPO mysid data for 2006 to 2019 are shown in Figure 8, although these data are not directly incorporated into this zooplankton indicator assessment.

An important threat to the zooplankton communities of the Great Lakes is posed by invasive species. The continued proliferation of dreissenid populations can be expected to impact zooplankton communities through the alteration of the structure and abundance of the phytoplankton community that many zooplankton depend on for food. Predation from the non-native cladocerans *Bythotrephes longimanus* and *Cercopagis pengoi* may also have an impact on zooplankton abundance and community composition. These invasive predatory cladocerans have been shown to have a major impact on zooplankton community structure in the Great Lakes (Lehman 1991; Barbiero and Tuchman 2004; Warner et al. 2006). Four new non-native zooplankton species were recently detected in Western Lake Erie: the rotifer *Brachionus leydigii*, the cyclopoid copepods *Thermocyclops crassus* and *Mesocyclops pehpeiensis*, and the cladoceran *Diaphanosoma fluviatile*. All of these species were detected at very low abundances in at least one year between 2015-2018, but they have not rapidly expanded and thus their potential impacts are not likely to be significant.

Some of the other measures useful for the interpretation of the zooplankton data include: total phosphorus, chlorophyll a, temperature, oxygen (seasonal depletion in upper hypolimnion, including anoxia in central basin of Lake Erie), primary production, and phytoplankton composition and biomass.

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Data Source:
U.S. EPA Great Lakes National Program Office
Anne Scofield, Great Lakes National Program Office (U.S. EPA), Chicago, IL
James Watkins and Lars Rudstam, Cornell University

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Figure 1. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-μm tows for Lake Superior. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 2. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Lake Huron. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 3. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Lake Michigan. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 4. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Lake Ontario. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 5. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Western Lake Erie. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 6. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Central Lake Erie. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 7. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Eastern Lake Erie. Length-weight coefficients used are listed in EPA SOP LG 403. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Figure 8. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO spring and summer survey mysid tows (whole water column, average of spring and summer shown) for each lake. Length-weight coefficients used are listed in EPA SOP LG 408. Data for 2008 Lake Ontario uncertain due to low sample size. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Last Updated
State of the Great Lakes 2022 Report

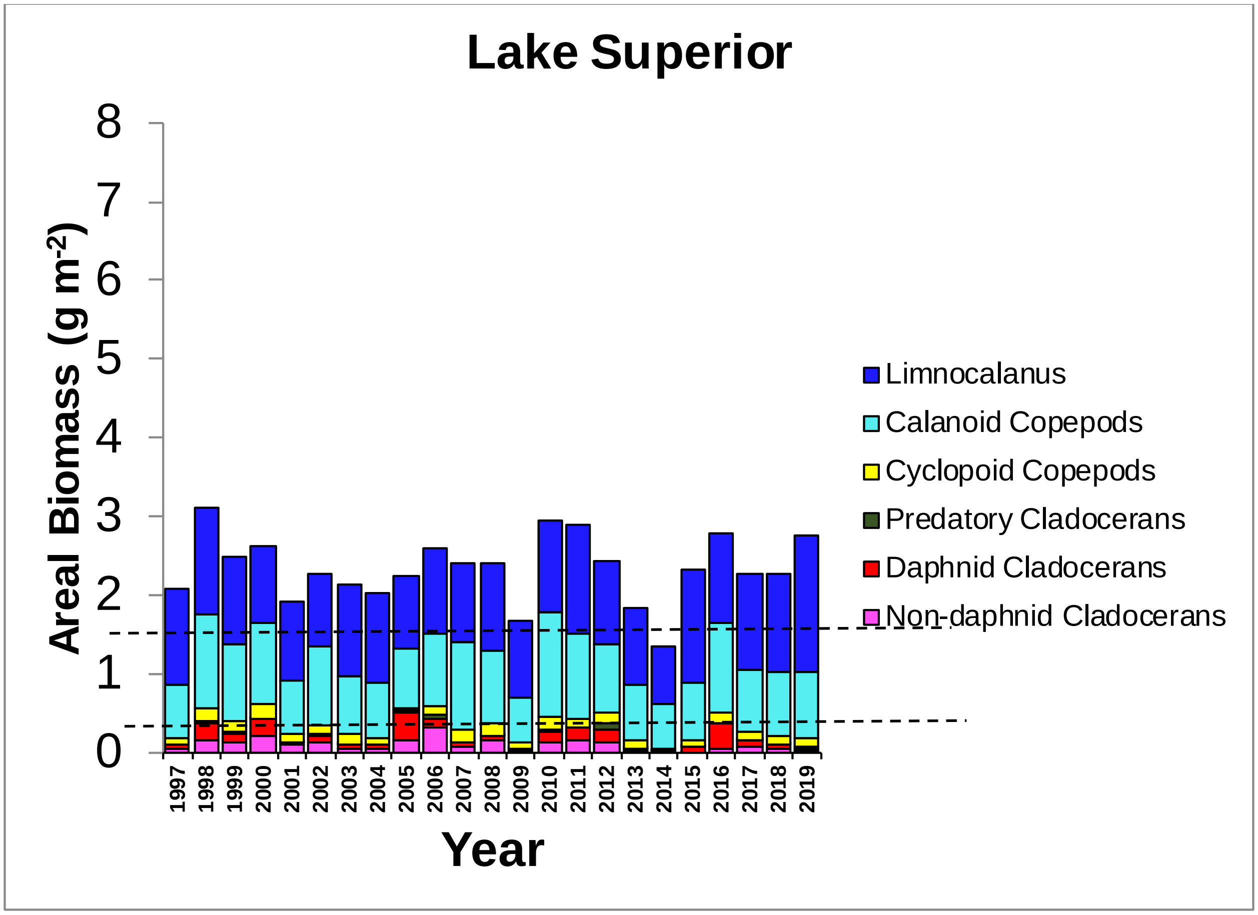


Figure 1. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Lake Superior. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines. Conditions are assessed as "Fair" when total biomass falls between the dashed lines. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University.

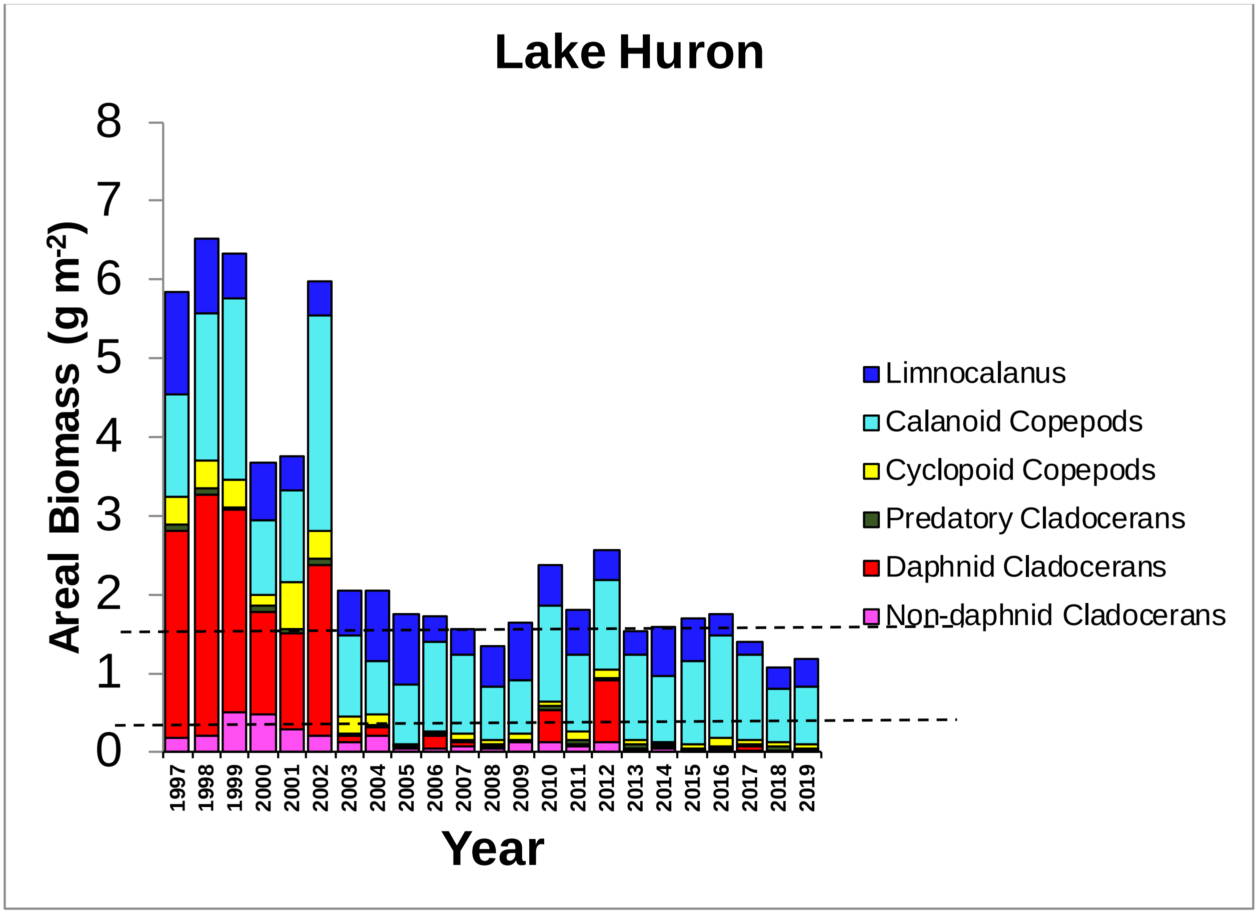
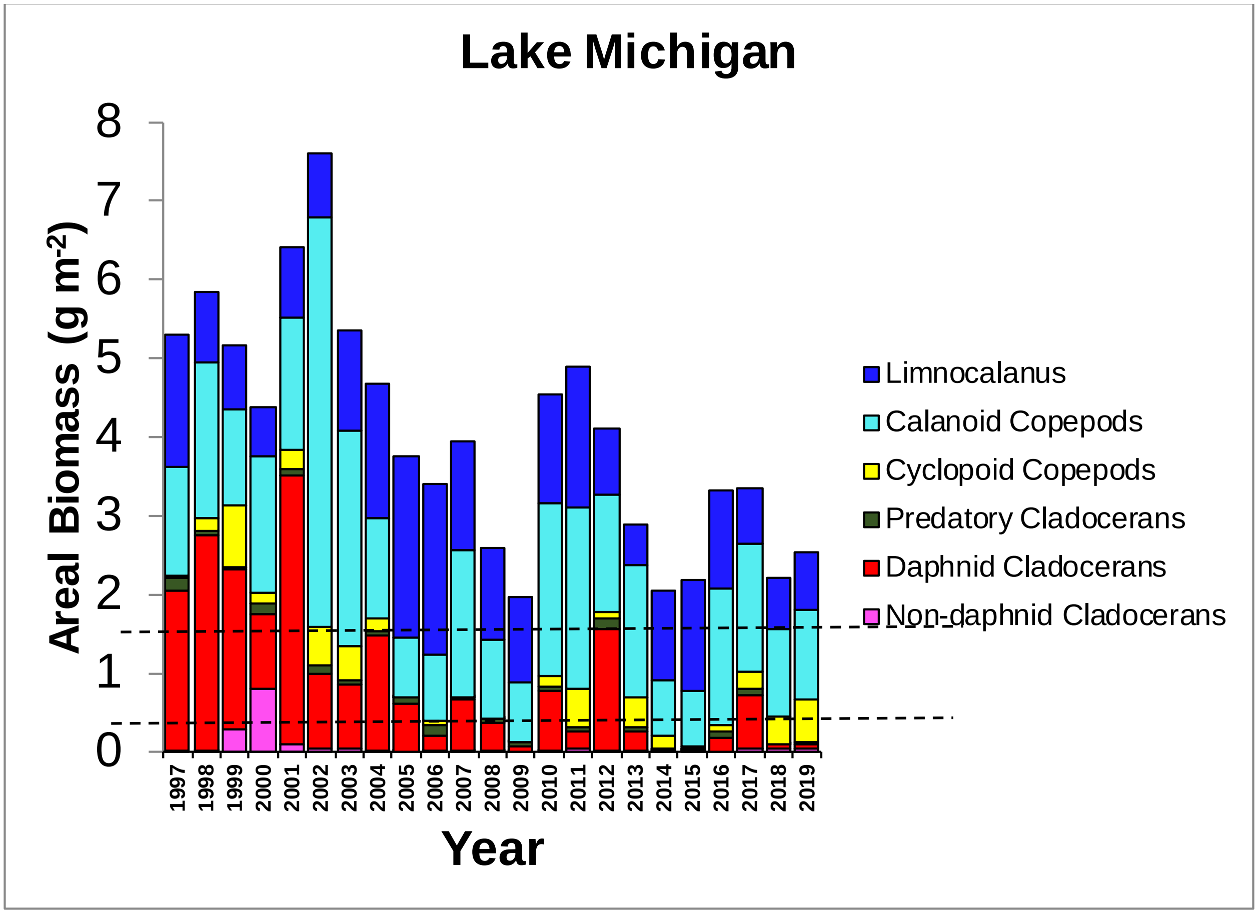


Figure 2. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Lake Huron. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines. Conditions are assessed as "Fair" when total biomass falls between the dashed lines. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University.



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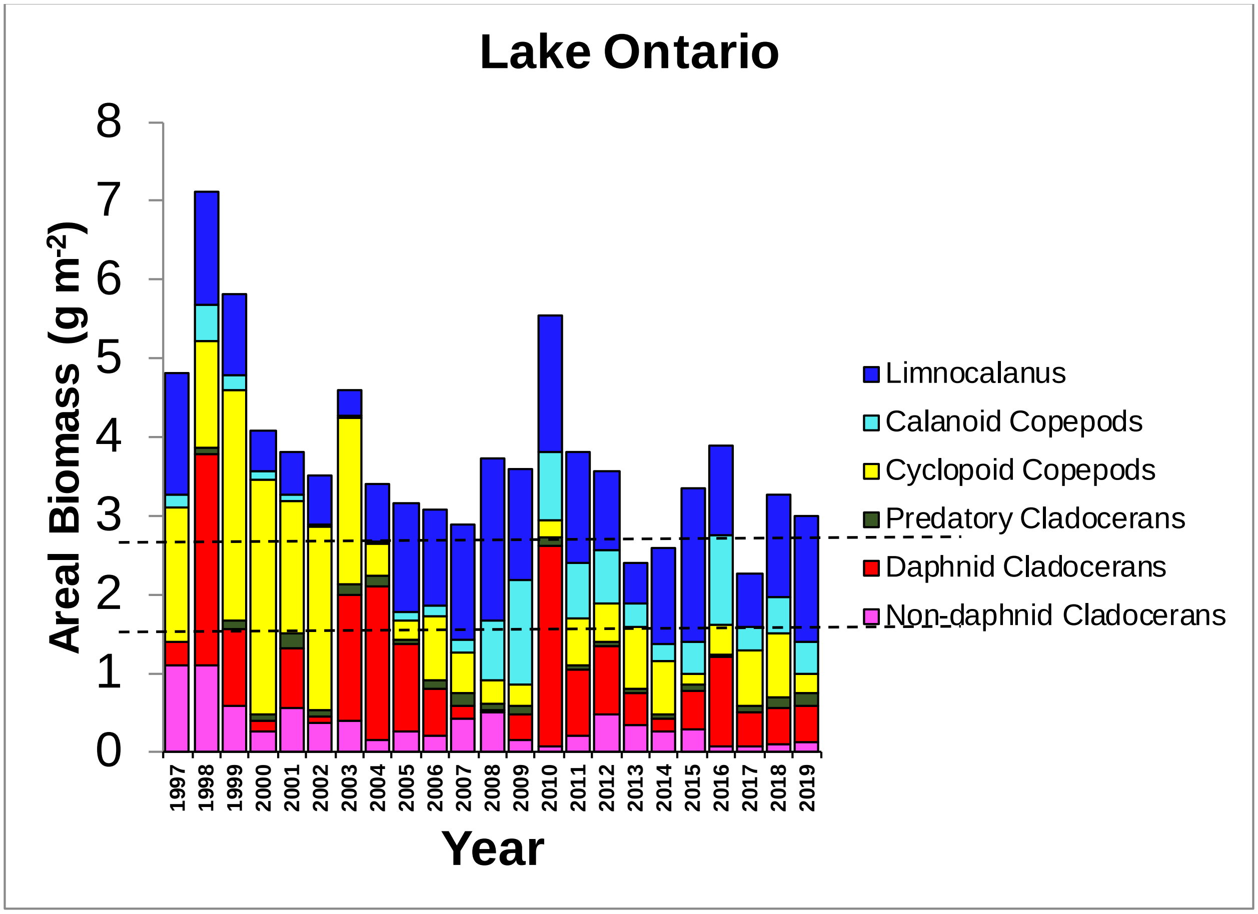

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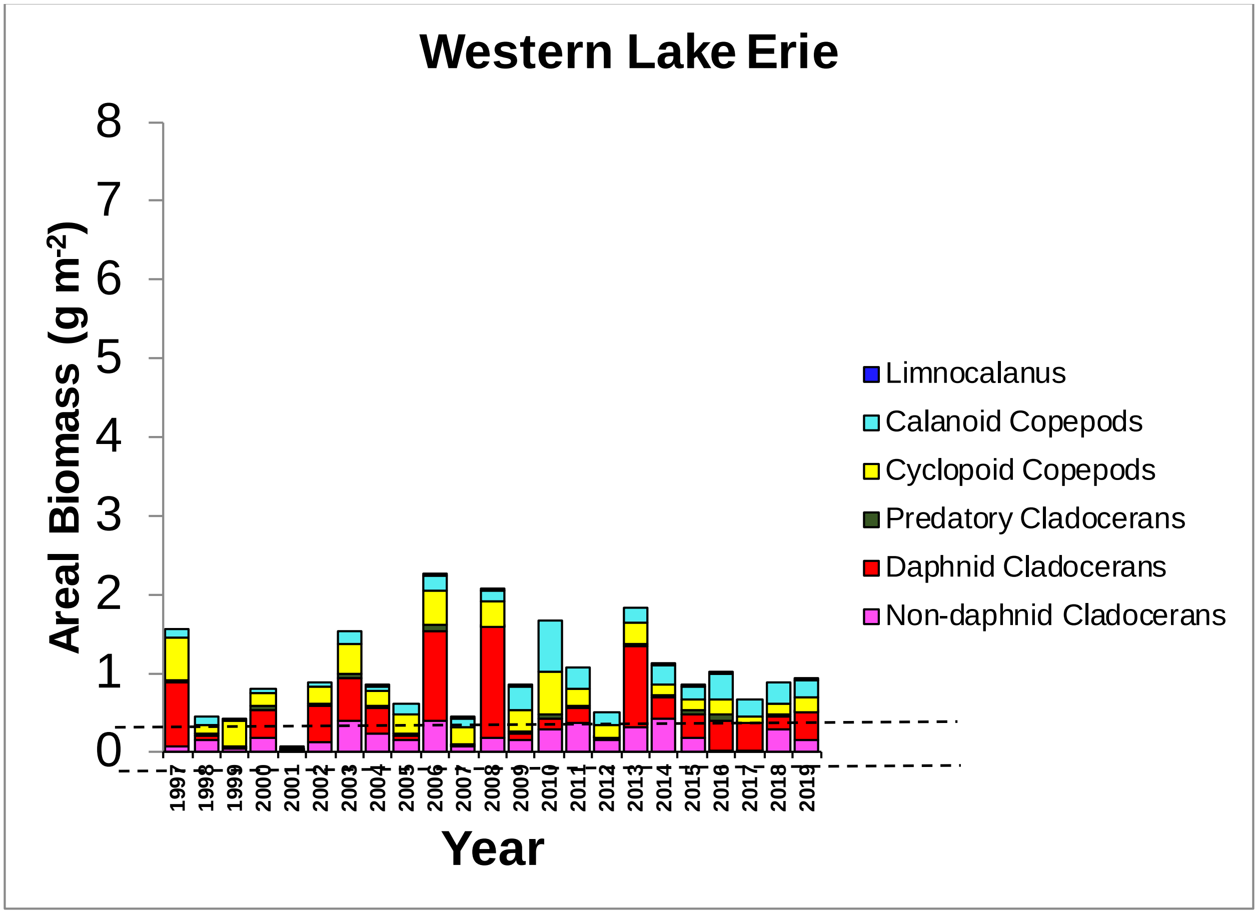


Figure 5. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Western Lake Erie. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines. Conditions are assessed as "Fair" when total biomass falls between the dashed lines. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University.

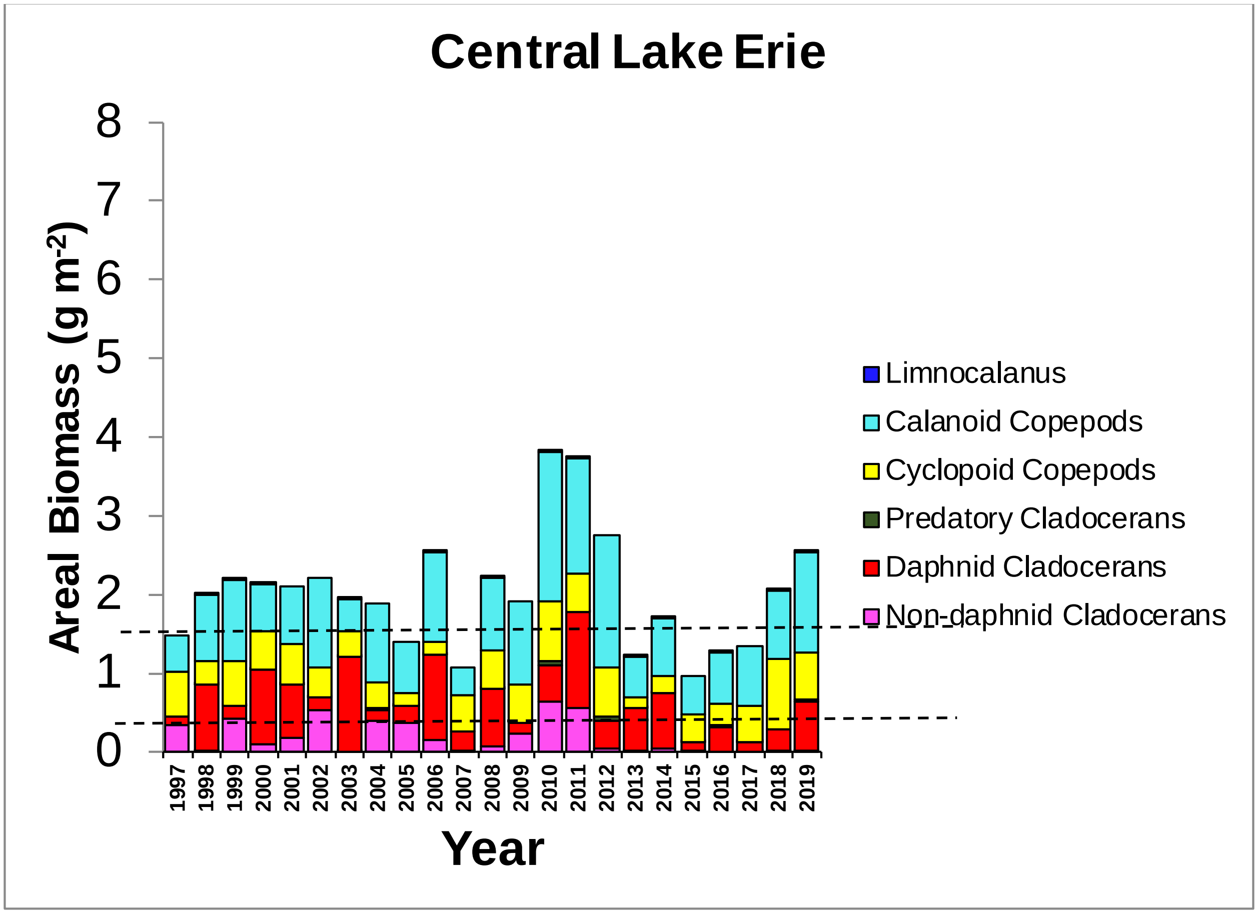
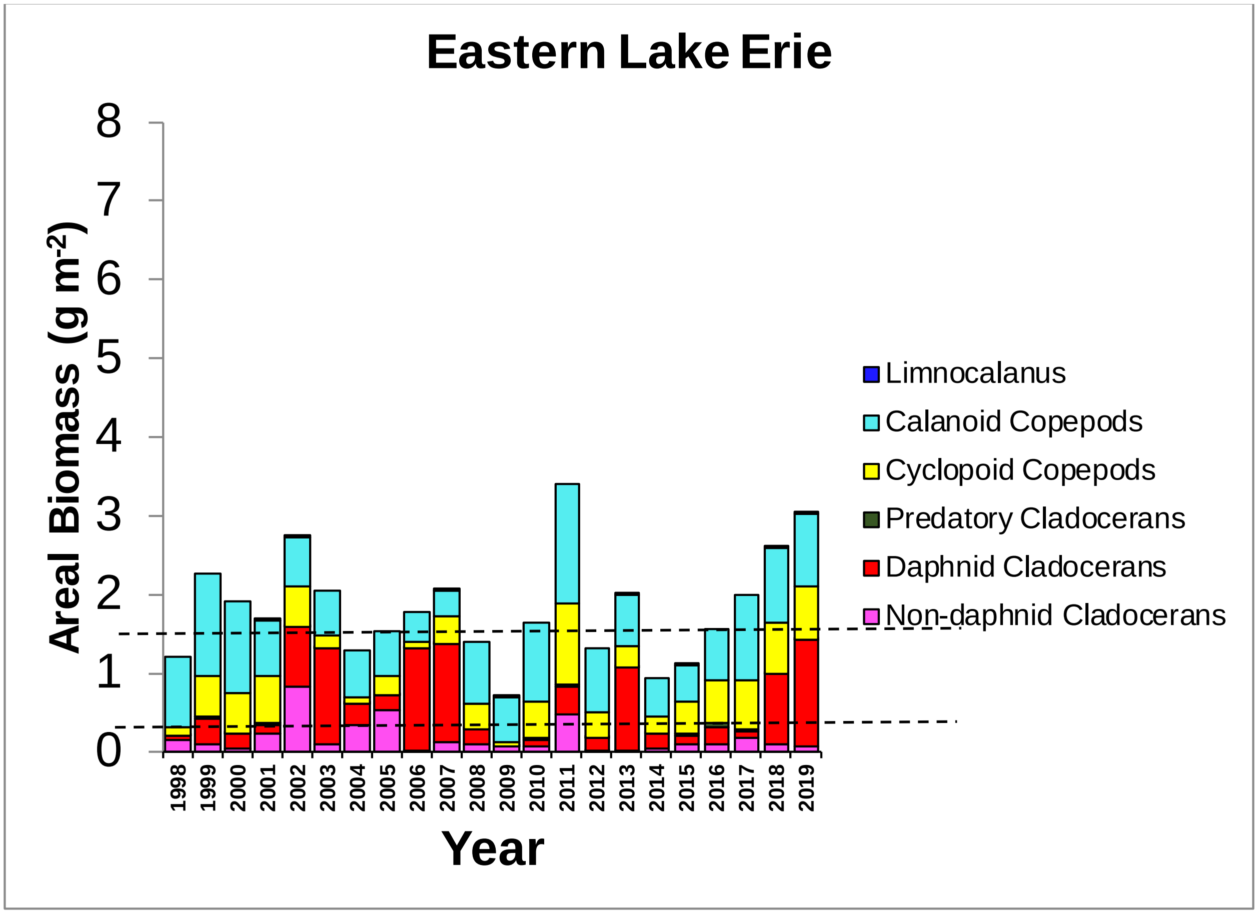


Figure 6. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (towed from 100 m depth or 2 m above bottom for shallower sites to the surface) 153-µm tows for Central Lake Erie. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure. Conditions are assessed as "Fair" when total biomass falls between the dashed lines. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University.



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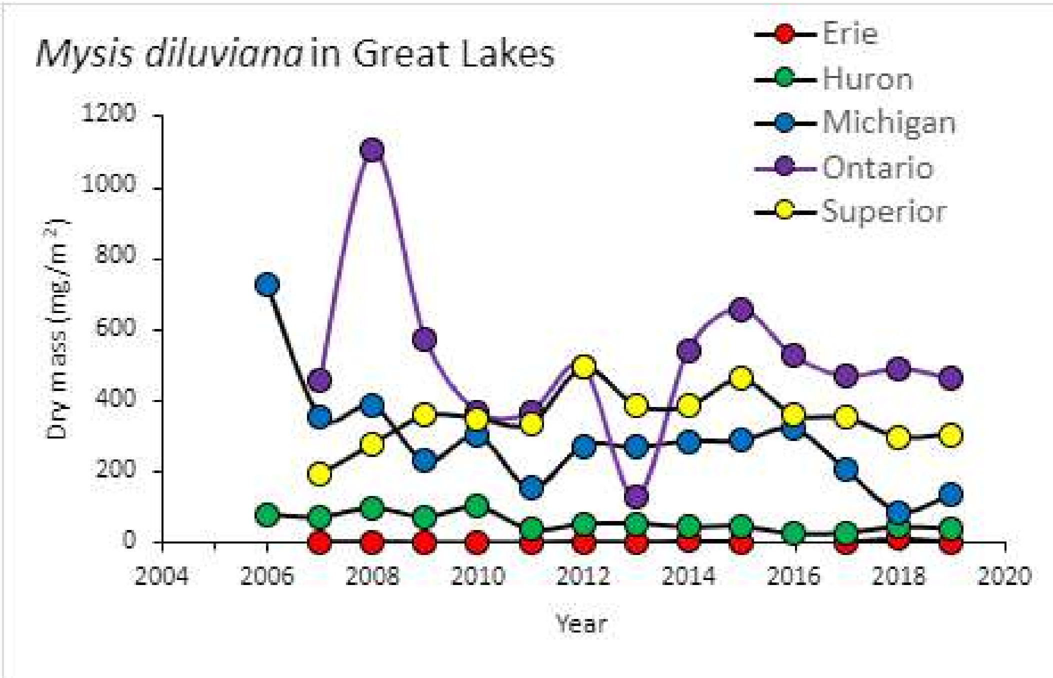

Figure 8. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO spring and summer survey mysid tows (whole water column, average of spring and summer shown) for each lake. Length-weight coefficients used are listed in EPA SOP LG 408. Data for 2008 Lake Ontario uncertain due to low sample size. Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University.

Sub-Indicator: Benthos Open Water

Overall Assessment

Status: Good

Trends:

10-Year Trend: Unchanging

Long-term Trend (1998-2019): Unchanging

Rationale: Based on the Oligochaete Trophic Index (OTI) scores, the status and trends in the trophic condition of the lakes are generally considered to be good and unchanging for long-term (1998 - 2019), 10-year (2010-2019), and short-term (2017-2019) time periods. Offshore sites in the Great Lakes are generally oligotrophic, but Lake Erie and some nearshore (<30m) stations in lakes Ontario, Huron, and Michigan are assessed as poor and the trends (both long-term and 10-year) are indicative of increased eutrophication. Overall, an increasing OTI score reflects an increase in trophic condition (more eutrophic).

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1998-2019): Unchanging

Rationale: All nearshore and offshore stations in Lake Superior were classified as oligotrophic, based on OTI scores, both long-term (1998-2019) and in recent years (2017-2019). No significant long-term or 10-year trends were observed in the trophic condition of the lake. The endpoint for this sub-indicator is to maintain oligotrophic conditions in the open waters of Lake Superior.

Lake Michigan

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1998-2019): Unchanging

Rationale: Since 1998, the majority (13 out of total 16) of stations in Lake Michigan had OTI scores below 0.6 indicating an oligotrophic condition. Of the nine nearshore stations, two were mesotrophic (0.6 < OTI < 1.0), and one was eutrophic (OTI scores > 1.0). Significant long-term trends of increasing eutrophication are evident at six nearshore stations, and trends of increasing oligotrophication were found at two offshore stations and one station in Green Bay. Overall, no significant long-term or 10-year trends were observed in the trophic condition of the lake. The endpoint for this sub-indicator is to maintain an oligotrophic state in the open waters of Lake Michigan.

Lake Huron

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1998-2019): Unchanging

Rationale: Almost all stations in Lake Huron are oligotrophic, except for three eutrophic nearshore stations. The trophic condition of the lake, as measured by OTI scores, has not changed significantly since 1998. The endpoint for this sub-indicator is to maintain an oligotrophic state in the open waters of Lake Huron.

Lake Erie

Status: Poor

10-Year Trend: Unchanging

Long-term Trend (1998-2019): Unchanging

Rationale: Almost all stations in Lake Erie are eutrophic, and two have a long-term trend of increasing OTI scores. Overall, no significant long-term or 10-year trends in trophic condition were observed. The endpoint for this sub-indicator is to maintain mesotrophic conditions in the open waters of the western and central basins of Lake Erie, and oligotrophic conditions in the eastern basin of Lake Erie.

Lake Ontario

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (1998-2019): Unchanging

Rationale: OTI scores indicate that all deep-water stations (>80 m) in both the western and eastern basins of Lake Ontario are oligotrophic, and all the nearshore stations are eutrophic. In the last three years, three eutrophic nearshore stations in the western basin showed long-term trends of increasing eutrophication. Overall, no significant decreasing trends were found in the trophic condition of the lake either since 1998 or in the last 10 years.

Status Assessment Definitions

Good: OTI score of less than 0.6 (oligotrophic conditions)

Fair: OTI score of 0.6 to 1.0 (mesotrophic conditions)

Poor: OTI score of greater than 1.0 (eutrophic conditions)

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

The desired trend is toward an oligochaete open water community indicative of oligotrophic conditions throughout the lakes, with the exception of the nearshore waters of all lakes and open waters of the western and central basins of Lake Erie, where mesotrophic conditions are desired.

Improving: a decrease in the OTI indicating declining eutrophication (i.e., increasing oligotrophication)

Unchanging: OTI score is not changing

Deteriorating: an increase in the OTI indicating increasing eutrophication (i.e., declining oligotrophication)

Undetermined: metrics do not indicate a clear overall trend, or data are not available to report on a trend

Endpoints and/or Targets

Maintain an oligotrophic state consistent with healthy aquatic ecosystems in the open waters of Lakes Superior, Michigan, Huron and Ontario; maintain mesotrophic conditions in the nearshore waters of all lakes and open waters of the western and central basins of Lake Erie, and oligotrophic conditions in the eastern basin of Lake Erie.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess trends in trophic conditions in the Great Lakes using oligochaete diversity, abundances, and the individual species responses to organic enrichment. We also report general trends in the full benthic community; although these supplementary data are not used directly in the indicator assessment, they provide useful context.

Ecosystem Objective

The Ecosystem Objective is that the benthic community in the Great Lakes should be comparable to unimpaired waters with similar depth and substrate.

This sub-indicator most closely aligns with General Objectives #5 and #6 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species" and "be free from nutrients that directly or indirectly enter the water as a result of human activity, in amounts that promote growth of algae and cyanobacteria that interfere with aquatic ecosystem health, or human use of the ecosystem".

This sub-indicator evaluates trophic conditions in the Great Lakes using oligochaete diversity, abundances, and the individual species responses to organic enrichment.

Measure

This sub-indicator (OTI) evaluates trophic conditions in the Great Lakes using oligochaete diversity, abundances, and the individual species responses to organic enrichment. In addition to the assessment of lake trophic status using OTI, we are providing summary data on status and trends in the whole benthic community and its role in food webs for additional context.

Calculation of the OTI

To evaluate trends in the trophic conditions of the Great Lakes based on status of benthic community, an OTI is used. The OTI was initially described by Mosley and Howmiller (1977) with subsequent modifications by Howmiller and Scott (1977), Milbrink (1983), and Lauritsen et al. (1985). This sub-indicator primarily follows Milbrink's formula (Burlakova et al. 2018a) with several modifications. Here we use the old division of Oligochaeta by Enchytraeidae, Lumbriculidae, Naididae and Tubificidae, comparable with historical data, despite the fact that only three orders are currently recognized (Enchytraeida, Lumbriculida and Tubificida) and former families of Naididae and Tubificidae in the order Tubificida are now combined in one family Naididae (Erseus et al., 2008). Therefore for practical reasons and clarity Tubificidae was retained for the analyses. Oligochaetes (tubificids and lumbriculids) are classified into four ecological classes relative to their tolerance to organic pollution, from 0 indicating intolerant of enrichment to 3 indicating tolerant of enrichment. The index ranges from 0 to 3; scores less than 0.6 indicate oligotrophic conditions; scores above 1 indicate eutrophic conditions; and scores between 0.6 and 1.0 indicate mesotrophic conditions. The index is calculated as:

$$c * [(1/2\sum n_0 + \sum n_1 + 2\sum n_2 + 3\sum n_3) / (\sum n_0 + \sum n_1 + \sum n_2 + \sum n_3)$$

where n_0, n_1, n_2, and n_3 indicate the abundances of organisms in each of the four trophic categories (see Table 1) and c is a density coefficient that scales the index to absolute densities (m^{-2}) of tubificids and lumbriculids. The c coefficient is as follows (Milbrink 1983):

$c=1$ if $n > 3,600$

$c=0.75$ if $1,200 < n < 3,600$

$c=0.5$ if $400 < n < 1,200$

$c=0.25$ if $130 < n < 400$

$c=0$ if $n < 130$

There are several parts of the OTI calculation that were interpreted as follows (Riseng et al. 2014):

- only lumbriculids and tubificids were used to calculate the index;
- all immature lumbriculids were classified as Stylodrilus heringianus;
- the c coefficient was estimated from abundances (n) of mature and immature lumbriculids and tubificids.

Milbrink (1983) assigned the tubificid *Tubifex tubifex* dual classifications depending on the dominance of *S. heringianus* or *Limnodrilus hoffmeisteri*. The dual classification was formalized as follows: if the ratio of abundances of n_0 oligochaetes to n_3 oligochaetes (*L. hoffmeisteri*) < 1 then *T. tubifex* is classified as a 3; if the ratio is >1 then *T. tubifex* is classified as a 0; however, if the ratio is close to one (0.75 to 1.25) then *T. tubifex* is a 3 if $c \geq 0.5$ and a 0 if $c < 0.5$; if *L. hoffmeisteri* density is zero and n_0 is relatively high and/or total density is low, then *T. tubifex* is 0, otherwise 3; and if the total density of oligochaetes is zero, then the index is zero.

Trophic classifications were obtained from literature for the Great Lakes and are shown in Table 1.

For each lake or sub-basin, a graph showing the OTI values on the y-axis and years on the x-axis is presented to illustrate the changes in species metrics over time (Figure 1). A map is used to show the major, within lake, spatial-temporal differences (Figures 2 and 3).

Data are sourced from the long-term benthic monitoring program of U.S. EPA Great Lakes National Program Office (GLNPO) which samples 57 stations annually within the five Great Lakes during summer (August). While this monitoring program started in 1997, data from 1997 are omitted from the status and trends assessments in this report due to unreconcilable discrepancies. The GLNPO benthic station ON64 was discontinued in 2016; prior to 2016 a total of 58 long-term monitoring stations were sampled and included in the OTI calculations. Less frequent but more spatially intensive sampling of all Great Lakes is now undertaken during Cooperative Science Monitoring Initiative (CSMI) field years to enhance lake-wide estimates of changes in benthic community diversity and abundance.

Ecological Condition

State of the Great Lakes Reporting uses the modified oligochaete-based trophic condition index (OTI, Milbrink 1983; Howmiller and Scott 1977) to assess trophic status of each station. Benthic invertebrate data for OTI calculation are collected annually at 57 permanent GLNPO sampling stations (Lakes Superior and Huron: 11 stations each, Michigan: 16, Erie: 10, and Ontario: 9 stations) located mostly offshore (Figure 2). The OTI is calculated based on known organic enrichment tolerances and abundances of aquatic annelid worms (Oligochaeta) (see the Measure section for more information). As organic pollution increases, oligochaete species with different physiological tolerances to water and sediment pollution replace each other and the community composition shifts. Overall, an increasing OTI means increasing eutrophication or increasing trophic conditions. The OTI scores range from 0 – 3: scores less than 0.6 (the lower blue line in Figure 1) indicate oligotrophic conditions; scores above 1 (the top red line in Figure 1) indicate eutrophic conditions; and scores between 0.6 and 1 suggest mesotrophic conditions. Scores approaching 3 indicate high densities of oligochaetes dominated by the pollution tolerant tubificids.

Because we expect nearshore and offshore stations to support different benthic communities, we make the distinction between offshore and nearshore zones when using the OTI to assess overall lake-wide trends in trophic conditions. The original GLNPO sampling program design established two components (offshore and nearshore) with different objectives: the offshore component is intended to assess the benthic invertebrate community as an indicator of basin-wide cumulative stressors, while the nearshore component serves as an indicator of more localized environmental stressors (Barbiero et al., 2018). For example, some of the benthic stations were chosen near tributary mouths that deliver organic matter to the lake. In general, Oligochaeta distributions exhibit depth-specific patterns, with species that are tolerant of organic enrichment found only at depths <70m. As a result, many nearshore stations will be classified as eutrophic. Because the offshore component of the monitoring program is designed to be used as an indicator of basin-wide cumulative stressors, the offshore data most directly address the goal of this indicator.

A consistent difference in trophic conditions among and within the Great Lakes was found during the study period (1998– 2019) (Figure 1). Trophic condition was significantly inversely related to station depth ($r = -0.61$, $P < 0.001$), with Lake Erie being the most eutrophic lake, followed in order of decreasing trophic condition by lakes Ontario, Michigan, Huron, and Superior. To assess the temporal trends in OTI scores at each station, we used linear regression, and trends were considered significant if $P < 0.05$. No significant lake- or basin-wide long-term trends of increasing trophic condition were found since 1998, but localized positive or negative trends in OTI scores over time were found at some stations in Lake Erie, Michigan, Huron, and Ontario (Figure 3).

All 11 stations in Lake Superior were oligotrophic based on OTI scores since 1998, and there were no significant long-term or 10-year trends in OTI scores either lake-wide scale or at specific stations (Figure 3), consistent with the 2019 report. Densities of major taxonomic groups, including *Diporeia*, an indicator of oligotrophic conditions and an important lipid-rich fish prey item, were stable in Lake Superior from 1998 (Figure 4). The Lake Superior total benthos biomass is the lowest of all Great Lakes, but *Diporeia* comprised over 50% of the biomass (Figure 5).

Almost all stations in northern and central Lake Michigan, as well as deep stations in the southern part of the lake (13 of 16 total stations) are oligotrophic (Figure 2). Of the 9 nearshore stations one in southeastern Michigan (MI48 near the Kalamazoo River outlet) was eutrophic and two stations (MI49 in Green Bay and MI46 near the Grand River outlet) were mesotrophic. Five nearshore stations and one offshore station (MI51, 106 m) had significant long-term trends of increasing eutrophication ($P < 0.03$) in 2017-2019, compared to four nearshore stations in the 2019 report. At the same time, two deep stations (MI18 and MI47, $P < 0.04$), along with one station in Green Bay (MI49, 44 m, $P < 0.001$), showed signs of increasing oligotrophication (Figure 3). Overall, no significant lake- or basin-wide long-term trends of increasing trophic condition were found in the lake since 1998. Along with Lake Ontario and Lake Erie, Lake Michigan exhibits the highest lake-wide density and biomass of *Dreissena* of all invaded Great Lakes (Figures 5, 6). The majority of Lake Michigan's non-dreissenid biomass is comprised by Oligochaeta (Figure 5). Nearshore *Diporeia* in Lake Michigan underwent large declines in early 2000s, followed by declines in offshore populations in the mid-2000s, but have remained relatively stable at depths >90 m since 2005. Despite the declines in *Diporeia*, high total benthic biomass (including *Dreissena*) in Lake Michigan can likely support abundant benthic fish populations and native fish that feed on benthos and/or bottom fish. The percent of Oligochaeta comprising the non-dreissenid community increased with the decline in *Diporeia* and Sphaeriidae that began in early 2000s (Figure 4).

Most of the stations in northern and central Lake Huron (8 of 11 total stations) are oligotrophic (Figure 2). Of the five nearshore stations, three were classified as eutrophic (Figure 2). One of these stations (HU96, near the outlet of Saugeen River in Ontario, Canada) was classified as mesotrophic and had an increasing eutrophication trend in the previous reporting period. The total density of Oligochaeta, including species tolerant of organic pollution, increased at this station 20-fold since the early 2000s, which resulted in an increase the OTI score. The other two eutrophic stations are located in the southern part of the lake and in Saginaw Bay (Figure 2). Since the 2019 reporting, one additional nearshore station in central basin (HU06) showed signs of eutrophication, while one offshore station in northern Huron (HU61) displayed a trend toward oligotrophication (Figure 3). Overall, no significant lake- or basin-wide long-term trends of increasing trophic condition were found in the lake since 1998. The Lake Huron benthic community, including *Diporeia*, has undergone similar changes as in Lake Michigan, but the lake-wide density and biomass of *Dreissena* is lower than in lakes Michigan and Ontario (Figures 5, 6). Currently, the majority of Lake Huron non-dreissenid biomass is comprised by Oligochaeta (Figure 5). The percent of Oligochaeta in non-dreissenid community increased with the decline in *Diporeia* and Sphaeriidae that began in late 1990s (Figure 4).

In Lake Erie all stations are shallower than 70m deep. Eight stations were classified as eutrophic and two stations that were eutrophic in the 2019 report (ER63 and ER91) were classified as mesotrophic in 2017-2019. No significant trends of increasing eutrophication were found lake-wide in the trophic condition during both the long-term and in the last 10 years. Only two stations showed significant long-term trends of increased OTI scores (ER15 and ER43, $P < 0.025$), compared to four stations in the 2019 report. One eastern nearshore station (ER 63) showed a decline in OTI score ($P = 0.004$, Figure 3), consistent with trends recorded in 2013-2016. Due to large declines in *Dreissena* in Lake Erie in the last decade caused by hypoxic events in the central and western basins, the lake-wide dreissenid density and biomass in these two basins are now the lowest of all invaded lakes, but the biomass in the eastern basin is still high (Karatayev et al., 2018, 2021a). Biomass of non-dreissenid benthos is the highest among all other lakes and is largely comprised by Oligochaeta in the eastern and central basins, and *Hexagenia* in the western basin. The largest density of Oligochaeta in the lake was found in 2000s. *Diporeia* has not been found in Lake Erie samples since the beginning of GLNPO monitoring in 1997, but abundant benthic communities can support large fish populations. Overall, due to shallow depths and high productivity, Lake Erie has higher benthic biomass and species richness than the other Great Lakes (Burlakova et al., 2018b).

All deep-water stations in Lake Ontario have remained oligotrophic throughout the 1998-2019 reporting period, and no significant trends in the trophic condition were found lake-wide both long-term and in the last 10 years. All the four nearshore stations are eutrophic (Figure 2). Three of these stations are located in the western basin and displayed a trend toward increasing eutrophication since 1998 (likely being affected in the southern shore by the outlet of the Niagara River, and on the northern shore by the Toronto metropolitan area) (Figure 3). Compared to the previous report, two stations changed status from mesotrophic to eutrophic in 2017-2019. However, deep stations, for the most part, remain oligotrophic and unchanging both the long-term and within the last 10 years. Lake Ontario has the largest biomass and the second largest density of *Dreissena* among the Great Lakes, and dreissenid populations are stable (Figures 5, 6). The replacement of *Diporeia* with quagga mussels and Oligochaeta began in Lake Ontario in the 1990s, which was earlier than in the other deep lakes, and very few *Diporeia* have been recorded in the samples since the late 2000s. As in the other lakes, both density and biomass of non-dreissenid benthos are now dominated by Oligochaeta.

Overall, benthic communities in lakes Ontario, Michigan and Huron have shifted from dominance by *Diporeia* to dreissenids and oligochaetes during the time series. Dreissenid populations continue to increase in deep regions of Lake Michigan, Huron and Ontario and decline in shallow areas of deep lakes as well as in Lake Erie, but non-dreissenid populations have exhibited relative stability over the past decade. Dreissenids were the largest contributor of biomass (>99%) in all lakes except Superior. Following the patterns of declining benthic diversity and density with depth, Lake Erie benthos is the most abundant and diverse, and supports the most productive Great Lakes fishery. In lakes where *Diporeia* no longer constitutes a significant part of benthic community (all lakes except Lake Superior), current benthic populations (including dreissenids) can still sustain substantial amount of benthivorous fish, transferring the energy from the lower to the upper trophic levels (Madenjian et al., 2010).

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Dreissenid Mussels – the relative abundance of non-native suspension feeding benthic species such as zebra and quagga mussels can dramatically change the structure of aquatic communities (both benthic and pelagic), affect ecosystem functioning, and alter lake trophic state. In addition to direct local effects (e.g. creating additional habitat and enhancing food supply for some benthic species), dreissenid mussels also affect other sub-indicators such as Nutrients in Lakes, Chlorophyll, Phytoplankton, Zooplankton, and *Diporeia*, and they alter the amount of available food for some profundal taxa. In addition, dreissenids together with other benthic taxa abundant in mussel aggregations serve as food for fish, including round goby, that in turn are preyed upon by many other fish species (e.g., lake sturgeon, walleye, salmon, smallmouth and largemouth bass, lake trout, whitefish and yellow perch). There are strong interactions between these sub-indicators that are not well understood and require further investigation.
- Nutrients in Lakes (open water) – as a natural and essential part of aquatic ecosystems, nutrients play an important role in supporting the production of aquatic plants and algae, which provide food and habitat for planktonic and benthic organisms at the base of the food chain. The addition of nutrients affects the structure, abundance, and population dynamics of the benthic community, changing the proportion of tolerant and intolerant species, but the magnitude of changes varies depending on the depth and lake trophic status. Since the OTI was designed to reflect community changes following organic enrichment, it is expected to co-vary with an increase in nutrients. Indeed, the OTI positively correlates with the amount of Total Phosphorus and Total Soluble Phosphorus measured in near-bottom waters (Burlakova et al.

2018a). On the other hand, dreissenids can alter the amount and turnover of nutrients in the water and sediments (Li et al. 2021).

- *Diporeia* (open water) – *Diporeia* is a benthic macroinvertebrate in the cold, deep-water habitats of all the Great Lakes (except Lake Erie, where it is extirpated). It is an indicator of oligotrophic conditions, and an important fish food item. Historically *Diporeia* had been a dominant benthic macroinvertebrate in profundal regions of all five of the Great Lakes (Cook and Johnson, 1974). Proliferation of dreissenid mussels coincided with significant declines in *Diporeia* in Lakes Ontario, Michigan, and Huron, but the nature of these interactions is still not well-understood. While the abundance of *Diporeia* is not included and not necessarily responsive to the OTI, a significant increase in organic enrichment may negatively affect *Diporeia*.
- Climate change – increasing water temperatures may affect survivorship and increase the developmental rate, the timing of development, spawning, and food availability for benthic taxa. As a result, there may be potential shifts in distribution and abundance of oligochaetes, affecting the performance of the OTI index, driven directly by water temperature. Both abiotic changes and biological responses in the lakes are complex, and changes in water chemistry, hydrology, etc. may also be important for the growth and survival of benthic organisms. Synergistic effects between climate and other anthropogenic variables and sub-indicators (i.e., establishment and spread of invasive species, nutrients) will likely exacerbate climate-induced changes in the oligochaete community.

This sub-indicator also links directly to the other sub-indicators in the Habitat and Species indicator.

Assessing Data Quality and Data Availability

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin		X*		
Data obtained from sources within the U.S. are comparable to those from Canada				X
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://www.epa.gov/great-lakes-monitoring/great-lakes-biology-monitoring-program		

Clarifying Notes:

*The annual benthic monitoring program design of U.S. EPA GLNPO samples 10-16 stations per lake, larger spatial coverage provided by CSMI surveys.

Data Limitations

The OTI index used in this assessment is based on only 27 out of total 66 species, all belonging to a single subclass, Oligochaeta. The performance of the OTI was recently evaluated against lake productivity and two more indices (an improved index, iOTI, based on a larger number of Oligochaeta species, and a modified trophic index, mTI, based on all benthic species) have been proposed for use (Burlakova et al., 2018a). In addition, the National Coastal Condition Assessment (NCCA) is considering improvements to the OTI for the nearshore zone where it fails to perform at a large number of sites due to absence of Oligochaeta. A better performing index or indices should be developed and applied to the 2025 sub-indicator report to improve classification of the trophic condition of the lakes and assessment of trends in benthic community.

The OTI was developed to characterize situations of organic enrichment due to anthropogenic eutrophication, which was an important issue addressed by the Great Lakes Water Quality Act in 1972. Currently, the Great Lakes open waters are undergoing a reverse trend of gradual oligotrophication due to the combined effect of nutrient mitigation and dreissenids (Evans et al., 2011). Therefore, excessive eutrophication remains an issue only in some nearshore areas and shallow bays (e.g., western Lake Erie, Saginaw Bay, Green Bay), but is no longer a concern at lake-wide scales or in profundal areas, limiting the utility of the OTI for this sub-indicator. In addition, the current index does not allow for assessment of the status and trends in the whole benthic community, which is a gap given the whole benthic community's role in food webs. For example, dreissenids are now an essential part of the benthic community and Great Lakes food webs, but they are not included in the OTI nor are other important (non-oligochaete) components of benthic community. Therefore, this sub-indicator would be improved by using an index or indices that account for the whole benthic community.

Two important factors should be considered while using the OTI to evaluate lake trophic conditions. First, due to species-specific patterns of Oligochaeta distribution with depth, most of the species tolerant of organic enrichment are found at depths <70 m (Burlakova et al., 2018b). Secondly, some of the benthic stations were deliberately positioned next to mouths of large tributaries that deliver large amounts of organic matter to the lake (Barbiero et al., 2018). As a result, many nearshore stations will be classified as eutrophic. The ratio of nearshore (≤70 m) to offshore stations varies by lake (100% in Lake Erie, 56% in Michigan, 45% in Huron, 44% in Ontario, and 18% in Superior) and may therefore additionally affect the overall OTI score and our estimates of trophic condition of the lake.

Additional Information

Benthic invertebrate biomonitoring has long been a tool of choice in assessing anthropogenic impacts on aquatic systems due to species' relative longevity, limited mobility, and wide range in tolerance to environmental stressors. Abundant, pollution-tolerant benthic species indicate degraded habitats. Increasing species diversity and decreasing abundance of pollution-tolerant species indicate return to healthy habitats. In addition, benthic secondary production plays a central role in supporting higher trophic level production, comprising over half of total consumption of fishes common to north-temperate lakes of North America (Vander Zanden and Vadeboncoeur, 2002). The benthic community serves as a good indicator of overall ecosystem health as it integrates water, sediment and habitat qualities. Changes in the benthic community closely reflect shifts in the overall productivity of the system.

The oligochaete sub-indicator used for the State of the Great Lakes report assesses trophic status of the lakes and may suggest pressures due to organic enrichment. Most of the stations that showed increasing eutrophication are located near large river mouths, suggesting that pollution abatement efforts in the upland watersheds could help to improve water quality and sediment conditions at these stations. Other pressures not accounted for in the OTI include invasive species, regional climate change, water level changes, and toxic or other contaminants. The tendency of OTI to decrease with depth (due to the lack of pollution-tolerant species residing deeper than 70 m) may affect lake-wide index values, depending on the ratio of deep to shallow stations sampled in each lake. Non-native species that strongly affect freshwater ecosystems (e.g., *Dreissena* spp.) can alter the composition and abundance of benthic communities, affecting behavior of benthic indices, and can modify the response of communities to trophic state.

There is an emerging realization of the importance of benthic processes and pathways within a whole-lake context (Vander Zanden and Vadeboncoeur 2002). A recent analysis of long-term dynamics of major trophic levels in the Laurentian Great Lakes revealed a far greater prevalence of bottom-up regulation since 1998, as a result of long-term declines in total phosphorus (TP) inputs and the more recent proliferation of nonindigenous dreissenid mussels (Bunnell et al. 2013). Filter feeding Ponto-Caspian bivalves *Dreissena polymorpha* and *D. rostriformis bugensis* are powerful ecosystem engineers that affect both abiotic (e.g., enhance water clarity and alter nutrient cycling) and biotic (e.g., reduce abundance of phytoplankton and microzooplankton, enhance benthic algae and macrophytes, induce changes in benthic community) components of the ecosystem (Karatayev et al. 1997, 2002; Higgins and Vander Zanden 2010; Burlakova et al., 2018b). Dreissenid tissues and shells now contain nearly as much phosphorus as the entire water columns of the impacted Great Lakes, and mussels have become a major agent of phosphorus cycling, offsetting or overshadowing the effects of external loading (Li et al. 2021). Filter-feeding activity, sediment deposition and habitat provided by dreissenids directly affect benthic macroinvertebrate community abundance and composition by promoting epifaunal predators, scavengers and collectors while replacing native filter feeders (e.g., Karatayev et al. 1997; 2002; Burlakova et al. 2012; Ward and Ricciardi 2007; Higgins and Vander Zanden 2010). However, most of the changes in benthic community structure following dreissenid invasion are described for the littoral zone rich in epifaunal species while changes in profundal infaunal community are poorly understood (Burlakova et al. 2014; Karatayev et al. 2015). After the *Dreissena* invasion, the abundance of non-dreissenid taxa (e.g., *Diporeia*, Sphaeriidae) declined in profundal habitats (Nalepa et al. 2007, 2009; reviewed in Karatayev et al. 2015) where quagga mussels compete for space and food resources with most of native invertebrates. This may be a result of system-wide (e.g. food interception effect, resulting in strong decline of spring phytoplankton blooms) vs. local *Dreissena* (e.g. enrichment of sediments with biodeposits) effects. The resulting effect of *Dreissena* on the oligochaete community (e.g., increasing amount of both tolerant and intolerant Oligochaeta in the presence of *Dreissena*, Burlakova et al., 2018b) may induce changes in the OTI that will not reflect the changes in the trophic status of the ecosystem. Therefore, more data on the effect of dreissenids on species composition and abundance of benthic invertebrates in profundal vs. nearshore zone are needed to fully understand dreissenid impacts on benthic communities.

The number of permanent stations visited by U.S. EPA GLNPO allows for a cost-effective benthic monitoring program each year. Less frequent (once every 5 years) but more spatially explicit sampling of all Great Lakes occurs during the Cooperative Science and Monitoring Initiative intensive field years to assess changes in benthic community diversity and abundance. These lake-wide samplings of historical stations and other stations of concern allow the diversity, density and biomass of benthic community to be monitored in each lake and enable the estimation of lake-wide population changes of invasive and native indicator species (e.g., dreissenids, *Hexagenia*, *Diporeia*; Mehler et al., 2020; Karatayev et al., in press; Burlakova et al., in press).

Values of the Benthos sub-indicator representative of the nearshore are very important, as nearshore benthos are more diverse and are also affected more quickly and more directly (than offshore benthos) by inputs from adjacent watersheds, including runoff of nutrients. However, different indicators and indices would need to be developed for the assessment of status and trends in the nearshore benthic community, since many of the nearshore substrates lack the Oligochaete species used in the calculation of the current benthic index (OTI) for State of the Great Lakes reporting. In addition, the current index does not allow for assessment of status and trends in the whole benthic community and its role in food webs.

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coefficient c but only mature specimens were used to calculate the number belonging to each ecological group of oligochaetes (see the description of index calculation in the Measure section).

Source: 1998-2019 U.S. EPA GLNPO benthic data.

Figure 2. Map of the Great Lakes showing the mean trophic condition at each sampling station calculated for 2017-2019. Trophic condition was based on the modified Oligochaete Trophic Index based on Milbrink (1983).

Source: 2017-2019 U.S. EPA GLNPO benthic data.

Figure 3. Maps of the Great Lakes showing stations with significant temporal trend in trophic condition between 1998 and 2019. Stations without significant changes in oligochaete trophic index with time ("no change", $P > 0.10$, linear regression), with significant trends ("increasing eutrophication" or "oligotrophication", $P < 0.05$) are indicated.

Source: 1998-2019 U.S. EPA GLNPO benthic data.

Figure 4. Trends in benthos areal density (bottom panels) and percent areal density (top panels) of major taxonomic groups in Great Lakes (1998–2019). These values exclude *Dreissena* spp. that was monitored only after 2003, and Bivalves include only Sphaeriidae. Error bars represent one standard error.

Source: 1998-2019 U.S. EPA GLNPO benthic data.

Figure 5. Absolute and percent areal biomass of benthos by major taxonomic group excluding *Dreissena* spp. (bottom and middle panels) and areal biomass of *Dreissena* spp. (top panels) at stations averaged across 2017-2019. Error bars represent one standard error. Bivalves in the lower graph include only Sphaeriidae.

Source: 2017-2019 U.S. EPA GLNPO benthic data.

Figure 6. Changes in density (2003–2019) of dreissenid mussels in Lakes Erie, Michigan, Ontario, and Huron. Dreissenid mussel populations are not established in Lake Superior. Error bars represent one standard error.

Source: 2003-2019 U.S. EPA GLNPO benthic data.

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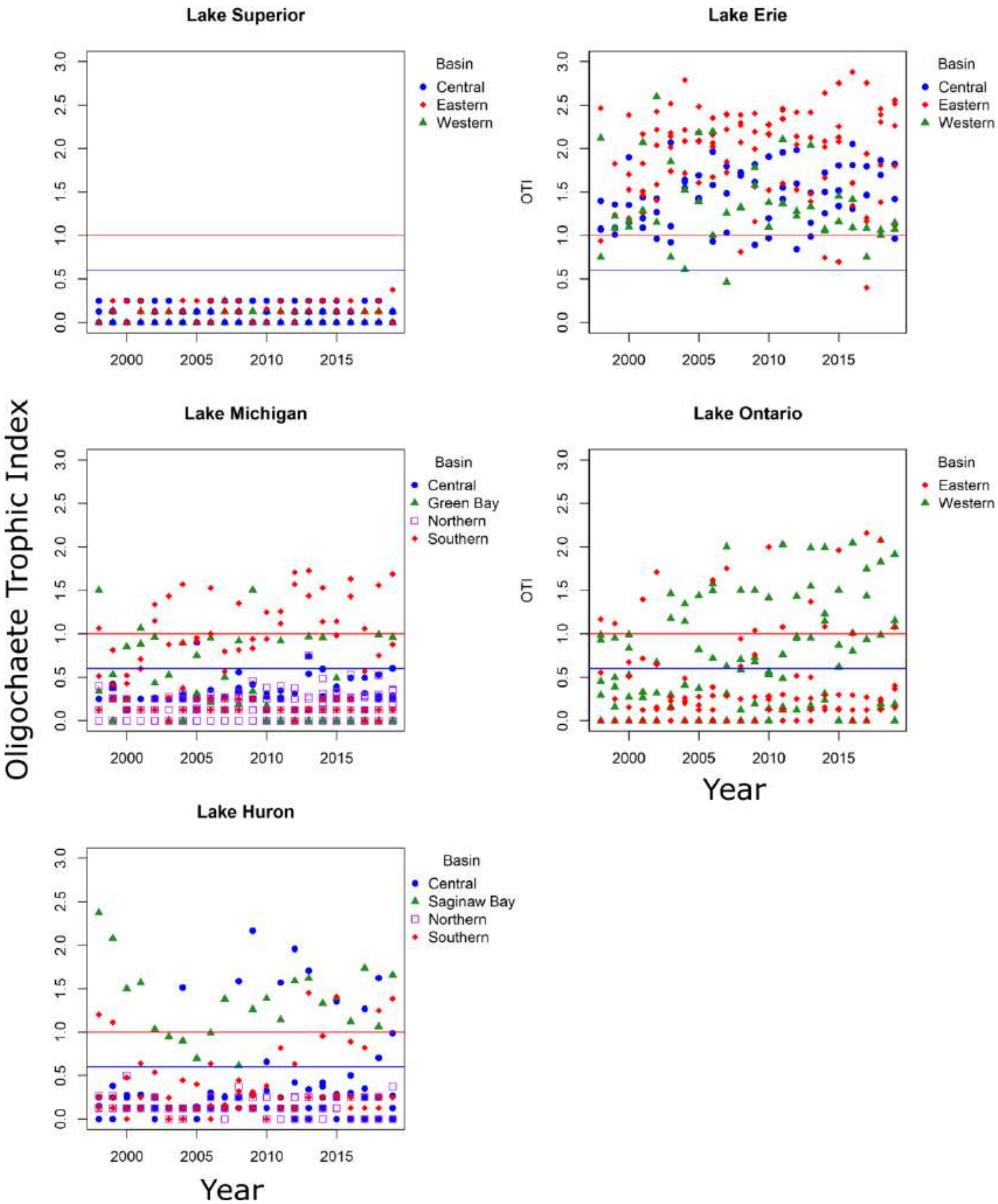
SPECCODE	GENUS	SPECIES	Trophic Class	Source	Comment
RHYCOCC	*Rhyacodrillus*	*coccineus*	0	Howmiller and Scott 1977	Same classification as Krieger 1984 & Lauritsen et al. 1985
TASAMER	*Tasserkidrilus*	*americanus*	0	Howmiller and Scott 1977	formerly *T. kessleri* in both Lauritsen et al. 1985 and Krieger 1984
LIMPROF	*Limnodrilus*	*profundicola*	0	Howmiller and Scott 1977	Same classification as Krieger 1984 & Lauritsen et al. 1985
RHYMONT	*Rhyacodrilus*	*montana*	0	Krieger 1984	Same classification as Lauritsen et al. 1985
RHYSP	*Rhyacodrilus*	*spp.*	0	Krieger 1984	Same classification as Lauritsen et al. 1985
SPINIKO	*Spirosperma*	*nikolskyi*	0	Krieger 1984	Same classification as Lauritsen et al. 1985
STYHERI	*Stylodrilus*	*heringianus*	0	Howmiller and Scott 1977	General agreement from all sources for this taxon
TASSUPE	*Tasserkidrilus*	*superiorensis*	0	Krieger 1984	Same classification as Lauritsen et al. 1985
AULAMER	*Aulodrilus*	*americanus*	1	Howmiller and Scott 1977	Classification based on Aulodrilus sp.
AULLIMN	*Aulodrilus*	*limnobius*	1	Milbrink 1983	
AULPIGU	*Aulodrilus*	*pigueti*	1	Milbrink 1983	
ILYTEMP	*Ilyodrilus*	*templetoni*	1	Krieger 1984	Same classification as Milbrink 1983 & Lauritsen et al. 1985
ISOFREY	*Isochaetides*	*freyi*	1	Krieger 1984	Same classification as Lauritsen et al. 1985
SPIFERO	*Spirosperma*	*ferox*	1	Howmiller and Scott 1977	Same classification as Krieger 1984 & Lauritsen et al. 1985
AULPLUR	*Aulodrilus*	*pluriseta*	2	Milbrink 1983	
LIMANGU	*Limnodrilus*	*angustipenis*	2	Howmiller and Scott 1977	
LIMCERV	*Limnodrilus*	*cervix*	2	Howmiller and Scott 1977	same as Milbrink 1983
LIMCECL	*Limnodrilus*	*cervix/ claparedeianus*	2	Howmiller and Scott 1977	same as Milbrink 1983
LIMCLAP	*Limnodrilus*	*claparedeianus*	2	Howmiller and Scott 1977	same as Milbrink 1983
LIMMAUM	*Limnodrilus*	*maumeensis*	2	Howmiller and Scott 1977	
LIMUDEK	*Limnodrilus*	*udekemianus*	2	Howmiller and Scott 1977	same as Milbrink 1983
POTBEDO	*Potamothrix*	*bedoti*	2	Milbrink 1983	
POTMOLD	*Potamothrix*	*moldaviensis*	2	Milbrink 1983	Same classification as Lauritsen et al. 1985
POTVEJD	*Potamothrix*	*vejdovskyi*	2	Milbrink 1983	Same classification as Lauritsen et al. 1985
QUIMULT	*Quistadrilus*	*multisetosus*	2	Howmiller and Scott 1977	
LIMHOFF	*Limnodrilus*	*hoffmeisteri*	3	Milbrink 1983	Differs from classification in Lauritsen et al. 1985
TUBTUBI	*Tubifex*	*tubifex*	0 or 3	Milbrink 1983	Depends on densities of LIMHOFF and STYHERI and total oligochaete density

Table 1. Trophic classifications for select mature lumbriculids and tubificids taken from Howmiller and Scott (1977), Milbrink (1983) with additions from Krieger (1984), and Lauritsen et al. (1985). If Milbrink classifications differed from Howmiller and Scott, Howmiller and Scott was used. Species are classified into four ecological classes relative to their tolerance to organic pollution, from 0 indicating intolerant of enrichment to 3 indicating tolerant of enrichment. Source: Riseng et al. 2014.



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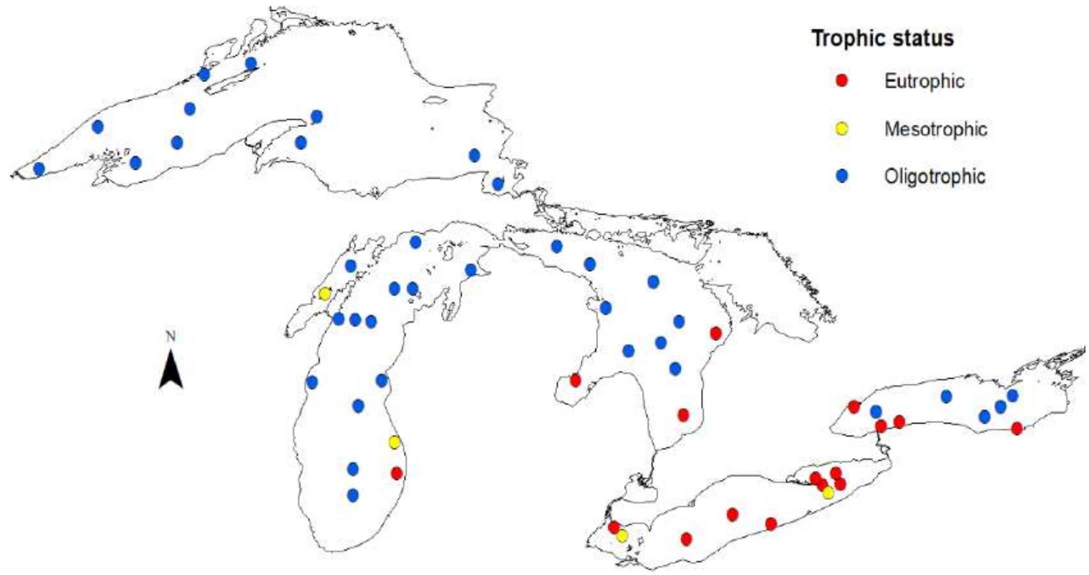

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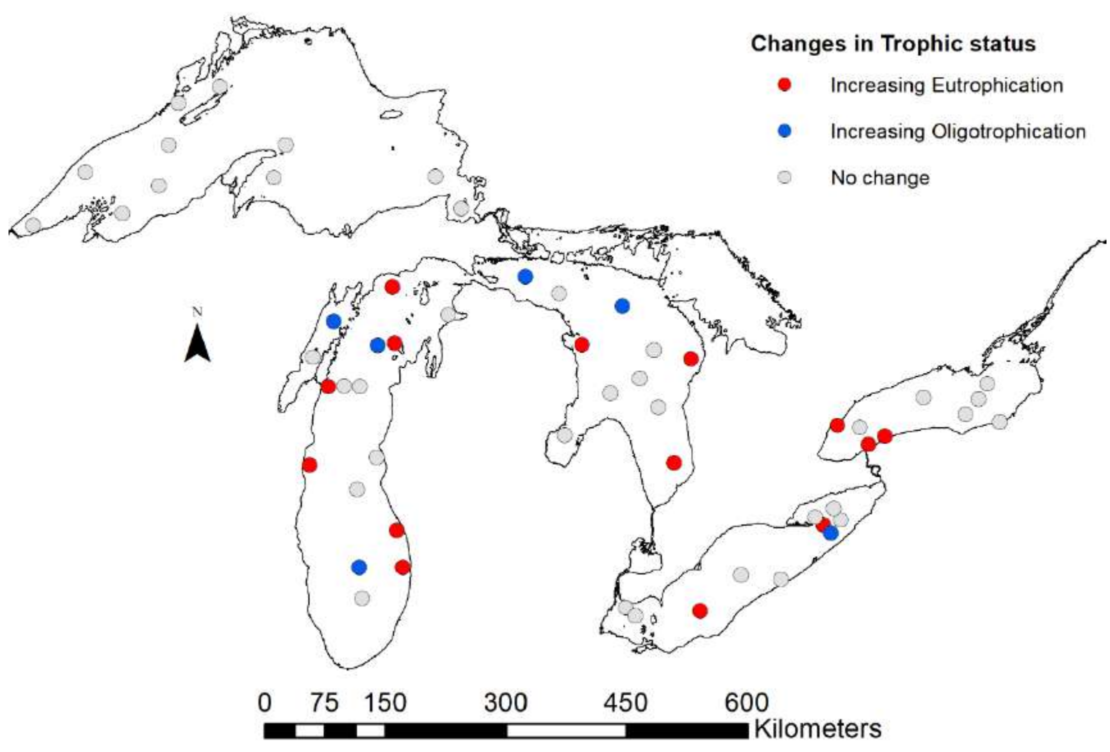

Figure 3. Maps of the Great Lakes showing stations with significant temporal trend in trophic condition between 1998 and 2019. Stations without significant changes in oligochaete trophic index with time ("no change", P > 0.10, linear regression), with significant trends ("increasing eutrophication" or "oligotrophication", P < 0.05) are indicated. Source: 1998-2019 U.S. EPA GLNPO benthic data.

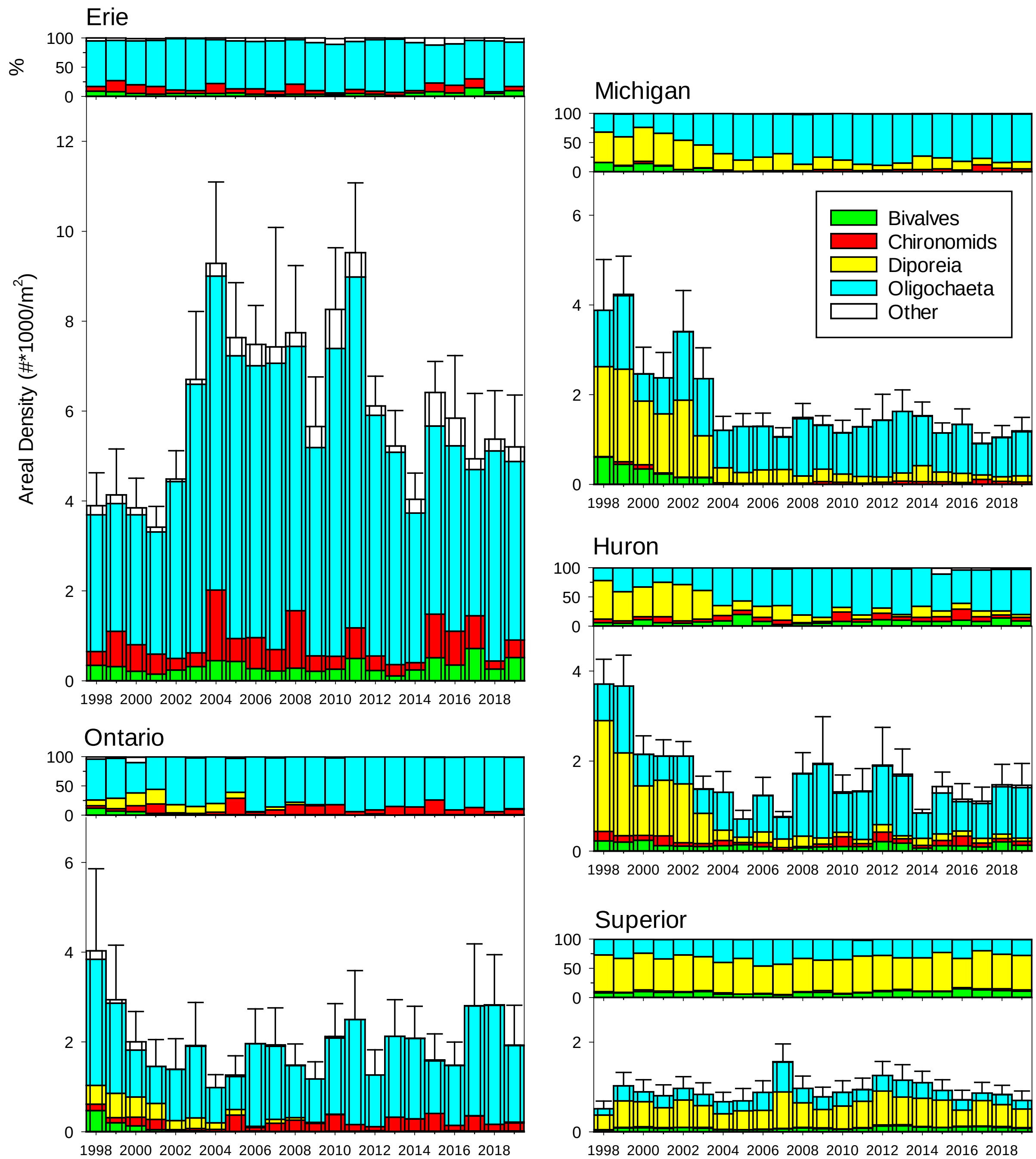

Figure 4. Trends in benthos areal density (bottom panels) and percent areal density (top panels) of major taxonomic groups in Great Lakes (1998–2019). These values exclude *Dreissena* spp. that was monitored only after 2003, and Bivalves include only Sphaeriidae. Error bars represent one standard error. Source: 1998-2019 U.S. EPA GLNPO benthic data.

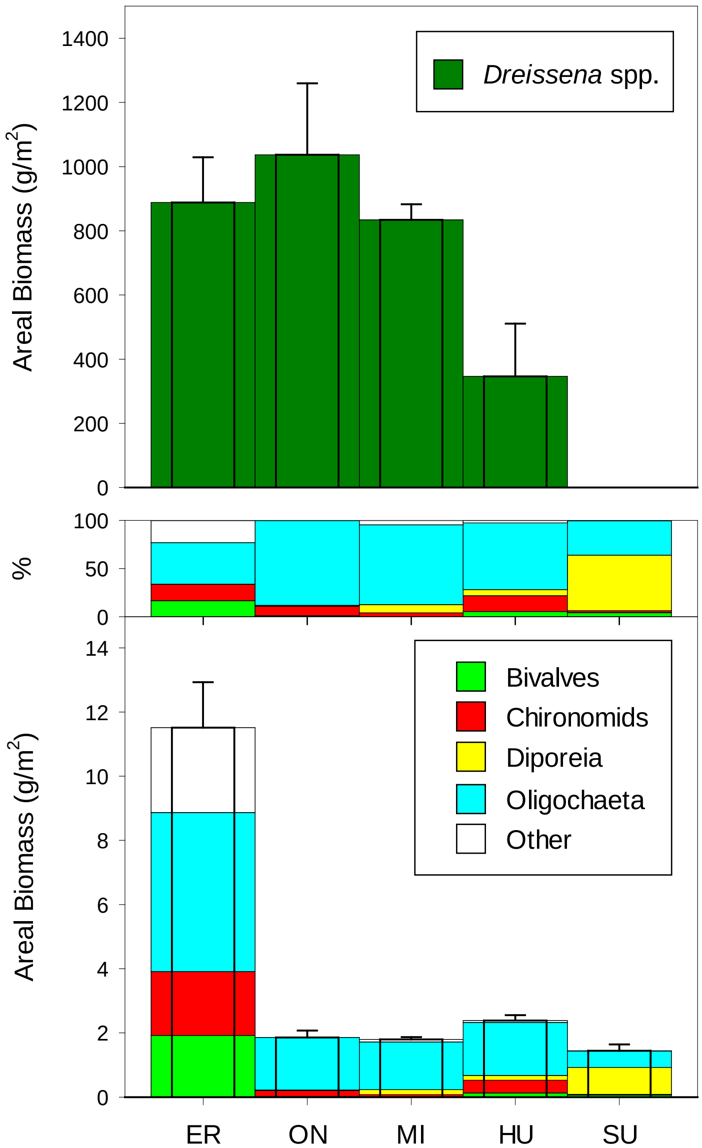

Figure 5. Absolute and percent areal biomass of benthos by major taxonomic group excluding *Dreissena* spp. (bottom and middle panels) and areal biomass of *Dreissena* spp. (top panels) at stations averaged across 2017-2019. Error bars represent one standard error. Bivalves in the lower graph include only Sphaeriidae. Source: 2017-2019 U.S. EPA GLNPO benthic data.

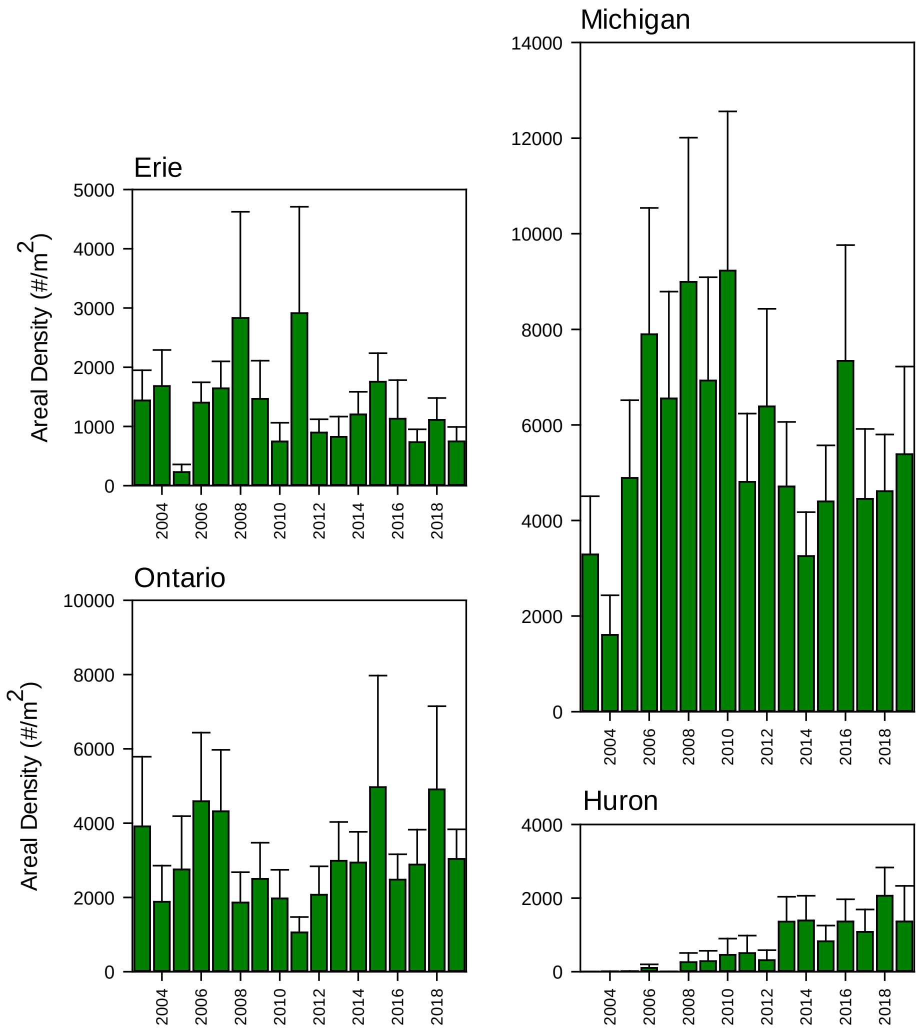

Figure 6. Changes in density (2003–2019) of dreissenid mussels in Lakes Erie, Michigan, Ontario, and Huron. Dreissenid mussel populations are not established in Lake Superior. Error bars represent one standard error. Source: 2003-2019 U.S. EPA GLNPO benthic data.

Sub-Indicator: *Diporeia*

Overall Assessment

Status: Poor

Trends:

10-Year Trend: Deteriorating

Long-term Trend (1972-2020): Deteriorating

Rationale: *Diporeia* spp. was once the dominant benthic invertebrate in the Great Lakes, but their populations experienced rapid declines starting in the 1990s. Since then, their declines have continued and densities remain low in Lakes Michigan, Huron, and Ontario. Abundances in Lake Superior are variable over time, with an overall trend of decline in the nearshore, but populations remain at good levels in the offshore and lake-wide. *Diporeia* is currently extremely rare in Lake Erie and is possibly close to extirpation. In all the lakes where *Diporeia* has declined, lower abundances first became apparent a few years after dreissenid mussels became established, though a causal link between *Dreissena* and the declines in *Diporeia* has been difficult to establish. The data presented here are primarily based on lake-wide surveys conducted over time by the US Environmental Protection Agency (EPA) and National Oceanic and Atmospheric Administration (NOAA), and the Canadian Department of Fisheries and Oceans (DFO). Since 2002, lake-wide benthic surveys have been a part of the Cooperative Science and Monitoring Initiative (CSMI), which occurs every 5 years for each lake on a rotating cycle. New to this reporting cycle are data from regional assessments and the Canadian Ontario Ministry of the Environment, Conservation and Parks (MECP) Nearshore Great Lakes Monitoring Network from 1992-2016 (https://data.ontario.ca/dataset/benthic-invertebrate-community-great-lakes-nearshore-areas).

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend (2006-2016): Unchanging

Long-term Trend (1973-2016): Unchanging

Rationale: Long-term monitoring and studies of distribution patterns indicate substantial temporal variability, with a trend of decreasing density at some U.S. nearshore stations between 1994-2016 (Figure 1). However, the population is not considered to be "Deteriorating" despite the observed declines because the density of *Diporeia* both nearshore (defined in this case as $\leq$100 m depth) and offshore remain above the level recommended by the 1978 Great Lakes Water Quality Agreement. Canadian nearshore monitoring in 1992, 1999, 2005, and 2011 measured consistently high *Diporeia* densities (ranging between 950-1885/m^2; MECP), which are comparable to densities shown from 1994-2003 (Figure 1). *Diporeia* are consistently found throughout the lake and remain the dominant benthic organism (Mehler et al. 2018). Further, the observed changes between 2006, 2011, and 2016 may be within the range of natural population variability for *Diporeia*, but more monitoring of the population is needed (Scharold and Corry 2019). The next lake-wide CSMI benthic survey is scheduled for 2021.

Lake Michigan

Status: Poor

10-Year Trend (2005-2015): Deteriorating

Long-term Trend (1981-2020): Deteriorating

Rationale: *Diporeia* abundances continue to decline in Lake Michigan. A lake-wide survey in 2015 indicated that *Diporeia* is still extremely rare at depths <90 m over the entire lake (Figure 2). At depths > 90 m, abundances were 58% lower compared to abundances found in 2005, and similar to 2010 levels (Figure 3). Recent annual surveys (2010-2020) conducted in just the southern region of Lake Michigan reveal *Diporeia* densities to be essentially absent <90 m and low and variable >90 m (Figures 4 and 5; Nalepa et al. 2014, 2020). No *Diporeia* were found in 2019 or 2020 at the one site (depth = 129 m) in southern Lake Michigan where they have been consistently found in the past. The next lake-wide CSMI benthic survey is scheduled for 2021.

Lake Huron (including St. Marys River)

Status: Poor

10-Year Trend (2007-2017): Deteriorating

Long-term Trend (1972-2017): Deteriorating

Rationale: *Diporeia* abundances continue to decline in Lake Huron. The most recent lake-wide survey occurred in 2017, and abundances showed further declines compared to a similar survey in 2012 (Figures 2, 3 and 6). Average density for the main basin in 2017 was 3/m^2 at depths 31-90 m and 161/m^2 at depths >90 m (Karatayev et al. 2020). Canadian nearshore monitoring within the main basin last reported *Diporeia* in 2002 (630/m^2), with no observations since in 2009 or 2015 (MECP). In 2017, the average density of *Diporeia* in the North Channel was 175/m^2, comparable to that observed at >90 m depths in the main basin, but densities at 51-90 m in Georgian Bay (2/m^2) were much lower (Karatayev et al. 2020). Canadian data showed moderate *Diporeia* densities in the nearshore of Georgian Bay (ranging between 158/m^2 in 1996 to 395/m^2 in 2015), declines in the nearshore of North Channel from 2563/m^2 in 1996 to 4/m^2 in 2011, and no *Diporeia* in St. Marys River with the exception of a few specimens collected at one site in 1999 (MECP). The next lake-wide CSMI benthic survey is scheduled for 2022.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend (2009-2019): Unchanging

Long-term Trend (1978-2019): Deteriorating

Rationale: Because of shallow, warm waters, *Diporeia* are naturally not present in the western basin and most of the central basin. *Diporeia* declined in the eastern basin beginning in the early 1990s (Dermott and Kerec 1997) and have not been found in that basin since 1998 according to one study (Barbiero et al. 2011). Canadian nearshore monitoring found a single specimen in 2010 in north central Lake Erie, but none before or since that time and no *Diporeia* have been found in the St. Clair River, Lake St. Clair, or the Detroit River (MECP). This trend is confirmed by no *Diporeia* found in the 2014 (Burlakova et al., 2017; Schloesser et al., 2017) or 2019 (Burlakova unpub. data) lake-wide surveys. The next lake-wide CSMI benthic survey is scheduled for 2024.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Poor

10-Year Trend (2008-2018): Unchanging

Long-term Trend (1994-2018): Deteriorating

Rationale: *Diporeia* abundances remain at low densities in Lake Ontario (Figures 2 and 3). The 2013 and 2018 lake-wide surveys in Lake Ontario produced only a single individual in each year (found at site depths >150 m) out of the 45-55 sites sampled (Nalepa and Baldridge 2016, Karatayev et al. 2021). Canadian nearshore surveys reported low densities of *Diporeia* (<30/m^2) at four sites across Lake Ontario in 1994, a few individuals at one site in 1997, and no *Diporeia* since then (MECP). Further, no *Diporeia* have been found in nearshore surveys of Hamilton Harbour, Bay of Quinte, or the St. Lawrence River (MECP). Additional benthic monitoring by USGS has identified a deep site (95 m) in western Lake Ontario with low but persistent numbers of *Diporeia* (B. Weidel, pers. comm.) Also, low numbers of *Diporeia* were found during EPA Long-term Monitoring surveys in 2018 (one individual at one station >90 m) and 2019 (31 total at two stations >90 m; GLNPO). The next lake-wide CSMI benthic survey is scheduled for 2023.

Status Assessment Definitions

Good: The following recommendation applies to Lake Superior only. Densities of *Diporeia* remain above 220-320/m^2 in nearshore waters (<100 m) and 30-160/m^2 in offshore waters (>100 m; 1978 GLWQA recommendations for Lake Superior; The Government of Canada and the Government of the United States of America 1978) and are found at sites well-distributed throughout the lake. For the other Great Lakes, the criteria are for *Diporeia* to be at moderate to high densities and also be the dominant non-dreissenid benthic organism.

Fair: Lake Superior: Densities of *Diporeia* remain above the GLWQA recommendations, but are only found at a few locations. Other Lakes: *Diporeia* are found in moderate to low densities.

Poor: Lake Superior: Densities of *Diporeia* are below the GLWQA recommendations. Other Lakes: *Diporeia* are found in low densities or are absent.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

Trend Assessment Definitions

Target values are provided to evaluate abundances on a historic basis. Trends over time provide a means to assess indicator direction. On a more direct basis, if target values are met, the system can be assumed to be healthy; if the values are not met there is health impairment. Causative agents of impairment are not addressed by the sub-indicator.

Improving: Increase in *Diporeia* densities and/or the number of stations with *Diporeia*.

Unchanging: Minor changes in *Diporeia* densities and the number of stations that have *Diporeia*.

Deteriorating: Decrease in *Diporeia* densities and/or the number of stations with *Diporeia*.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend

Endpoints and/or Targets

In Lake Superior, *Diporeia* should be maintained throughout the lake at abundances of >220-320/m^2 at depths <100 m and >30-160/m^2 at depths >100 m as per the 1978 GLWQA. In the other Great Lakes, for *Diporeia* to be the dominant non-dreissenid benthic organism and to be found at a majority of offshore locations.

Sub-Indicator Purpose

The purpose of this sub-indicator is to show the status and trends in *Diporeia* populations, and to infer the basic structure of cold-water benthic communities and the general health of the Great Lakes ecosystem.

Ecosystem Objective

The cold, deep-water regions of the Great Lakes should be maintained as a balanced, stable, and productive oligotrophic ecosystem with *Diporeia* as one of the key organisms in the food chain.

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the Waters of the Great Lakes should “support healthy and productive wetlands and other habitats to sustain resilient populations of native species.

Measure

This sub-indicator will measure the density (number/m^2) of *Diporeia* in the Great Lakes. The recent lake-wide trends presented here are based on extensive, lake-wide surveys (nearshore and offshore) that are conducted in each of the lakes every 5 years as a part of the CSMI cycle. Additional lake-wide surveys conducted by EPA, NOAA, and DFO provide longer frames of reference for *Diporeia* population dynamics. Also presented are data collected during the annual NOAA Great Lakes Environmental Research Laboratory benthic survey of the southern basin of Lake Michigan. New to this round is the inclusion of data from the MECP Nearshore Great Lakes Monitoring Network, which includes benthic data collected from the 1990s – 2016 in nearshore regions and connecting waterways in Canadian waters across the Great Lakes basin.

The U.S. EPA Great Lakes National Program Office (GLNPO) long-term monitoring program assesses abundances of *Diporeia* (and other benthos) in offshore regions of each of the lakes on an annual basis. In CSMI years, data from some of the GLNPO stations are included in with the lake-wide survey results. Otherwise, some GLNPO data are included anecdotally here (e.g., the few *Diporeia* observations in Lake Ontario), but are not included in the figures. GLNPO data generally follows similar trends within each lake and are accessible by request from the GLNPO data portal GLENDA.

Ecological Condition

This glacial-marine relic was once the most abundant benthic organism in cold waters greater than 30 m deep in each of the lakes. It was present, although less abundant in nearshore regions of the open lake basins, but it was naturally absent from shallow, warm bays, basins, and river mouths. *Diporeia* occurs in the upper few centimetres of bottom sediment and feed on algal material that freshly settles to the bottom from the water column (i.e., mostly diatoms). In turn, it is fed upon by most species of Great Lakes fish; including many forage fish species that themselves serve as prey for the larger piscivores such as trout and salmon. For example, sculpin feed almost exclusively upon *Diporeia*, and sculpin are eaten by lake trout. Also, lake whitefish, an important commercial species, feeds heavily on *Diporeia*. Thus, *Diporeia* was an important pathway by which energy was cycled through the ecosystem, and a key component in the food web of offshore regions.

To some degree, *Diporeia* abundances are affected by the amount of phytoplankton food settling to the bottom, and population trends may reflect the food conditions (Dermott 2001). Abundances can also vary somewhat relative to shifts in predation pressure from changing fish populations (Dermott 2001). In nearshore regions, this species is sensitive to local sources of pollution (Gossiaux et al. 1993; Maity et al. 2013), but because of varying conditions such as temperature fluctuations, substrate heterogeneity, and wave-induced turbulence, it can be difficult to assess population trends in this region.

Nearshore/offshore comparisons reveal that MECP nearshore data trends tend to match the offshore trends reported by the CSMI lake-wide surveys for most waterbodies (e.g., Lake Erie, Lake Huron main basin, Lake Ontario, and Lake Superior); however, the patterns observed in the two datasets diverge in North Channel and Georgian Bay. This could be due to the *Diporeia* populations in these regions exhibiting high levels of spatial and temporal variation. Also, the two surveys are often not conducted in the same years and have different sampling locations and numbers of stations. Lake Superior is the only lake to exhibit consistently higher *Diporeia* densities in the nearshore (here, Canadian and U.S. waters <100 m deep) than in the offshore (waters >100 m deep) (MECP; Mehler et al. 2018).

Diporeia populations have undergone dramatic declines with no signs of recovery in all the lakes except Lake Superior (Figures 1, 2 and 3). Based on the most recent surveys, *Diporeia* is present but continues to decline in Lakes Michigan and Huron, while it is nearing extirpation in Lake Erie and possibly Lake Ontario as well. The population in Lake Superior is highly variable and had exhibited declines in the nearshore zone since 1994 and 2003 (Figure 1). However, *Diporeia* is still found distributed throughout the lake, and densities remain relatively high. Declines in *Diporeia* within all lower Great Lakes were observed after zebra mussels (*Dreissena polymorpha*) or quagga mussel (*Dreissena rostriformis bugensis*) first became established (Dermott et al. 2005). It is worth noting that in some regions of Lake Ontario, *Diporeia* exhibited high population variability in the 1980s prior to the introduction of dreissenid mussels, as attributed to fish predation and/or reductions in productivity (Dermott 2001). Reasons for the negative response of *Diporeia* to these mussel species are not entirely clear and several hypotheses have been proposed (Watkins et al. 2007). One hypothesis is that dreissenid mussels are out-competing *Diporeia* for available food. That is, large mussel populations filter food material before it reaches the bottom, thereby decreasing amounts available to *Diporeia*. *Diporeia* declines in the early 1990s in nearshore Lake Ontario appear to correspond with grazing effects from mussels, but later declines in the offshore do not (Watkins et al. 2013). Edlund et al. (2021) suggested that reductions in preferred, more nutritious diatom prey in Lake Michigan exacerbated *Diporeia* declines. However, the reason for widespread reductions in *Diporeia* populations is more complex than a simple decline in food because *Diporeia* have completely disappeared from areas where food is still settling to the bottom and where there are no local populations of mussels. Also, individual *Diporeia* show no signs of starvation before or during population declines. Further, *Diporeia* and *Dreissena* coexist in some lakes outside of the Great Lakes (i.e., Finger Lakes in New York). Some empirical (Cave and Strychar 2015) and modeling (McKenna et al. 2017) studies suggest that the decline in *Diporeia* could be related to disease/parasites, but further work is needed in this area.

Linkages

Linkages of this sub-indicator to other sub-indicators include:

- Benthos (open water) – *Diporeia* is the dominant benthic species in Lake Superior deep-water habitats, where dreissenid mussels are absent. In the other Great Lakes, oligochaetes are now the most abundant non-dreissenid taxa.
- Prey fish – *Diporeia* is an important food source for several prey species. When abundant, *Diporeia* served as an important pathway to transfer energy from primary producers to higher trophic levels.
- Phytoplankton - The primary food source for *Diporeia* is phytoplankton that settles to the bottom of the lake.
- Impacts of Aquatic Invasive Species and Dreissenid Mussels – The rapid decline of *Diporeia* has coincided with the proliferation of invasive dreissenid mussels in multiple lakes (see Ecological Condition section above for more details).
- Toxic Chemicals in Sediment – *Diporeia* are a pollution-sensitive species and are absent or occur in low numbers in areas with elevated levels of contaminants (Nalepa and Thomas 1976).

This sub-indicator also links directly to the other sub-indicators in the Habitat and Species indicator, particularly lake trout, as lake trout are among the fish species that are energetically linked to *Diporeia*. Young lake trout feed on *Diporeia* directly, while adult lake trout feed on sculpin, and sculpin feed heavily on *Diporeia* (Hudson et al. 1995). Lake trout are a top predator in the deep-water habitat, and therefore assessments of both *Diporeia* and lake trout provide an evaluation of lower and upper trophic levels in the cold, deep-water habitat.

Diporeia may face mild to moderate impacts of changing climate trends. Being a cold-water stenotherm, *Diporeia* could potentially decline in the nearshore zones of Lake Superior as water temperatures increase, but a substantial cold-water refuge will remain in deeper waters. Climate-induced disruptions to preferred diatom food sources is another potential way in which *Diporeia* could be negatively affected.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: **Lake Michigan Data**: NOAA GLERL Technical Memo 164 (1994-2010) and NOAA GLERL Technical Memo 175 (2015). **Lake Huron Data:** NOAA GLERL Technical Memo 140 (1972, 2000-2003); NOAA GLERL Technical Memo 172 (2006-2012), **Lake Ontario Data:** Burlakova et al. Accepted 2021. *Ecology* Data Paper (1964-2018). Compilation of Canadian and U.S. data. https://esajournals.onlinelibrary.wiley.com/doi/10.1002/ecy.3528 **MECP Canadian Nearshore Data:** https://data.ontario.ca/dataset/benthic-invertebrate-community-great-lakes-nearshore-areas		

Data Limitations

This sub-indicator is of greatest value in assessing ecosystem health in the cold, open-water portions of the Great Lakes. It may also be useful when assessing long term trends within a specific lake region in the nearshore (< 30 m), but its value is questionable if widely applied to nearshore areas over all the lakes. Because this sub-indicator consists of only one taxon, it may not reliably diagnose causes of degraded ecosystem health. A number of lake-wide surveys and assessments of benthic invertebrate communities have been made over the past several decades in the Great Lakes and the current status of *Diporeia* populations is generally known, and an understanding of the changes related to the dreissenid mussel invasion is emerging. The available U.S. and Canadian data sources are comparable over time because both countries have performed whole-lake benthic surveys. In more recent years, the data sources are more complementary than comparable because they cover different depth zones and sampling locations. The benthic monitoring program conducted by MECP primarily covers nearshore areas (typically <3 km from shore) and DFO no longer conducts long-term offshore monitoring. However, the CSMI lake-wide benthic surveys include data from both U.S. and Canadian waters, with site depths ranging from ~10 m to >200 m, depending on the lake.

Additional Information

The historical dominance of *Diporeia* in cold, deep-water habitats in all of the Great Lakes provides a good basis for a basin-wide evaluation of ecosystem health.

The continuing decline of *Diporeia* has strong implications to the Great Lakes food web. As noted, many fish species rely on *Diporeia* as a major prey item, and the loss of *Diporeia* has impacted many of these species. Fish responses include changes in diet, movement to areas with more food, or a reduction in weight or energy content. Implications to fish populations include changes in distribution, abundance, growth, recruitment, and condition. Recent evidence suggests that fish are already being affected. Studies have shown that populations of lake whitefish, an important commercial species, have been affected, as well as fish species that serve as prey for salmon and lake trout such as alewife, sculpin, and bloater (Owens and Dittman 2003).

Because of the rapid rate at which *Diporeia* has declined in many areas, and its significance to the food web, documenting trends and reporting data needs to be completed in a timely manner. The population decline has a defined natural pattern, and studies of food web impacts should be spatially well coordinated. Also, studies to define the cause of the negative response of *Diporeia* to *Dreissena* should continue and build upon existing information. Potential areas of study are physiological and biochemical responses of *Diporeia* to *Dreissena*, and influence of potential pathogens, including bacteria and viruses. With a better understanding of why *Diporeia* populations have declined, one can better assess the potential for population recovery if dreissenid populations significantly decline.

Methods for estimating abundances of *Diporeia* are generally similar across the Great Lakes. Samples of bottom substrates are collected with a Ponar grab and contents are washed through a screen (or net mesh) of 0.5-mm openings (0.6-mm for the MECP Nearshore surveys). All *Diporeia* retained on the screen are immediately preserved, and later counted and identified. Densities are reported as numbers per square metre. Nalepa et al. (2009) provides additional details on sampling methods and laboratory analysis, which are largely consistent with the US EPA Standard Operating Procedure for Benthic Invertebrate Field Sampling Procedure

(SOP LG406, Revision 12, March 2018), and the US EPA Standard Operating Procedure for Benthic Invertebrate Laboratory Analysis (SOP LG407, Revision 09, April 2015). Given the decline and disappearance of *Diporeia* in nearshore regions, and very low abundances of *Diporeia* in offshore regions in each of the lakes except Lake Superior, the present monitoring programs are adequate to detect population changes.

Acknowledgments

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Figure 2. Mean densities (number per square metre) of the amphipod *Diporeia* spp. from sites at 31-90 m in lakes Michigan, Huron, and Ontario, 1995 – 2018. Data are from lake-wide surveys conducted mostly at 5-year intervals. Lake Michigan = triangles, long-dashed line (blue); Lake Huron = squares, short-dashed line (red); Lake Ontario = circles, solid line (black). The horizontal grey lines indicate the GLWQA recommendation threshold of 220-320/m^2 in nearshore water, for comparison. Data Sources: Great Lakes Environmental Research Lab, NOAA; Great Lakes Center, SUNY Buffalo; Dermott and Geminiuc 2003; Lozano et al. 2001; Watkins et al. 2007; Birkett et al. 2015; Nalepa et al. 2014, 2018, 2020; Karatayev et al. 2020 and 2021; Burlakova et al. 2021.

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Data Source: Great Lakes Environmental Research Lab, NOAA

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Small red dots indicate location of sampling sites.

Data Source: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2018, Karatayev et al. 2020.

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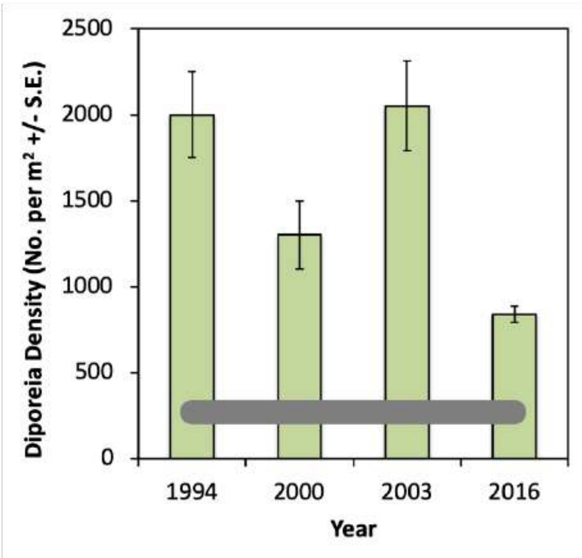

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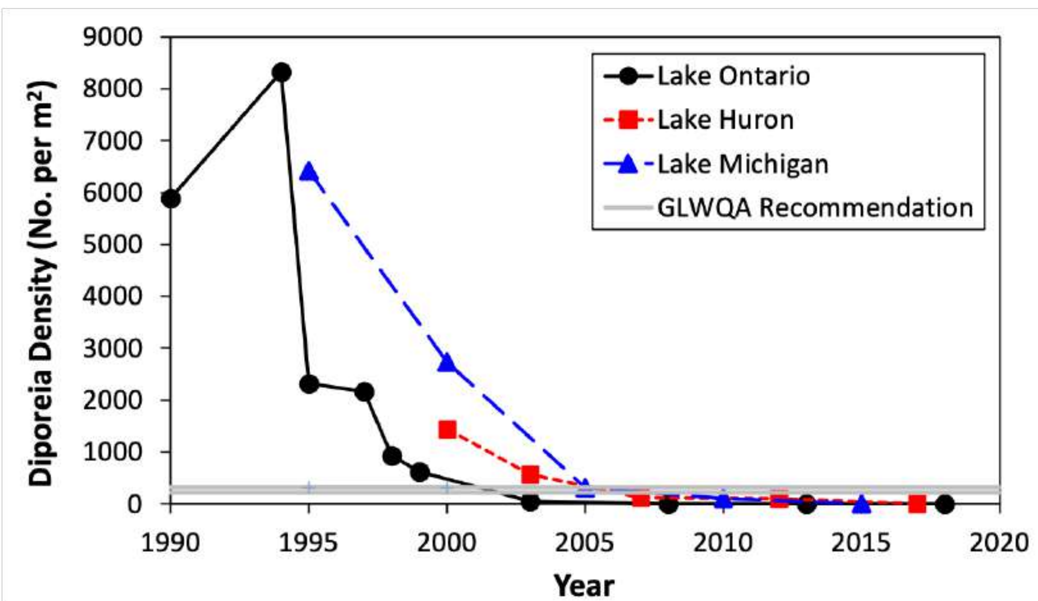

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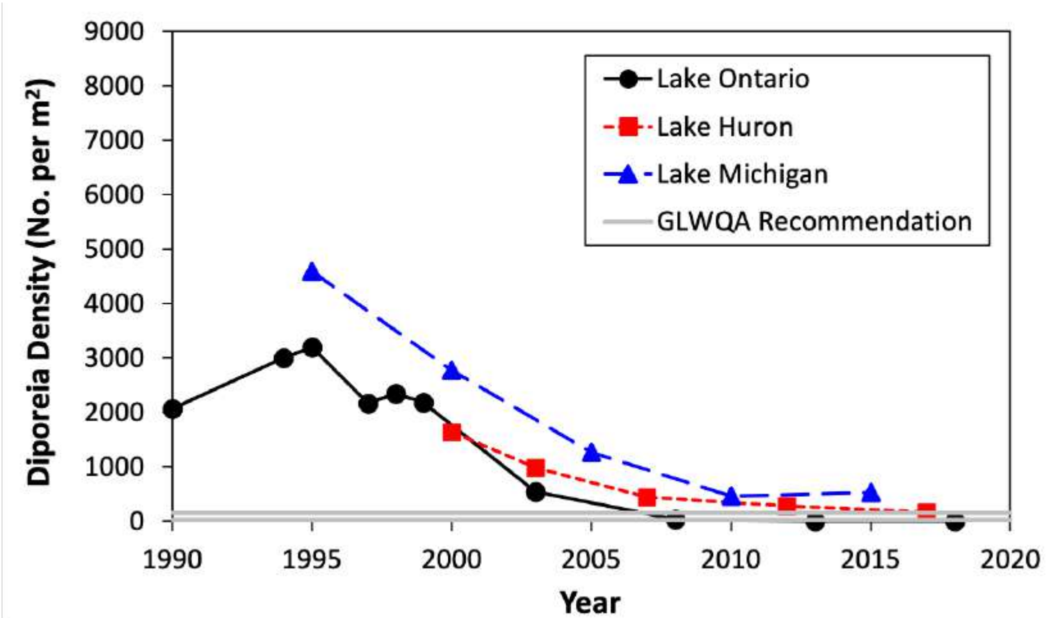

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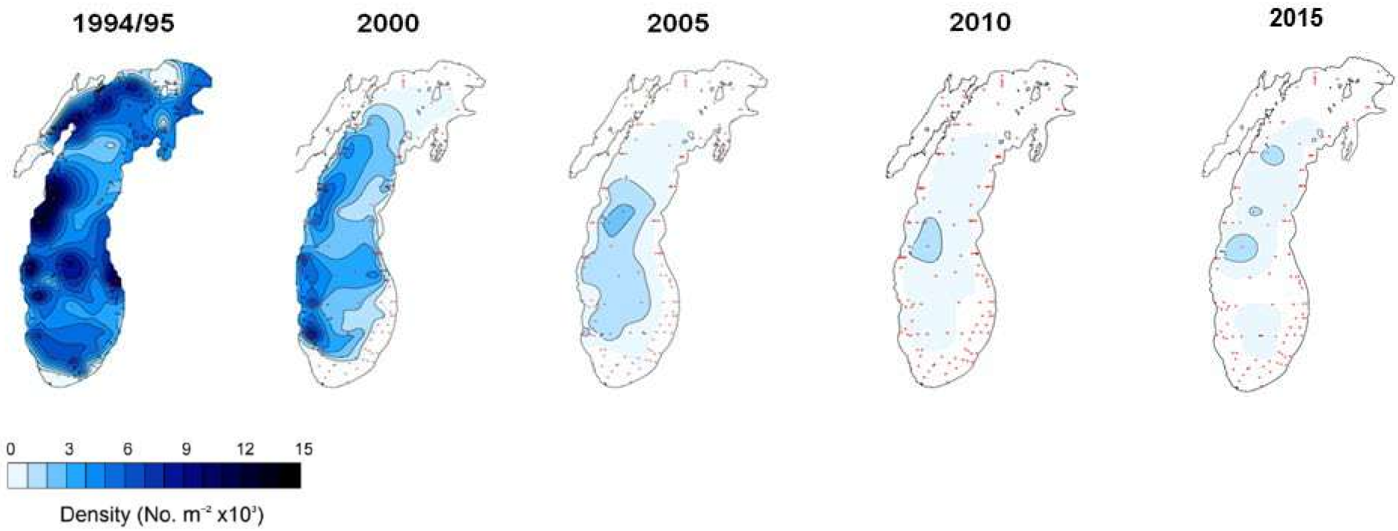

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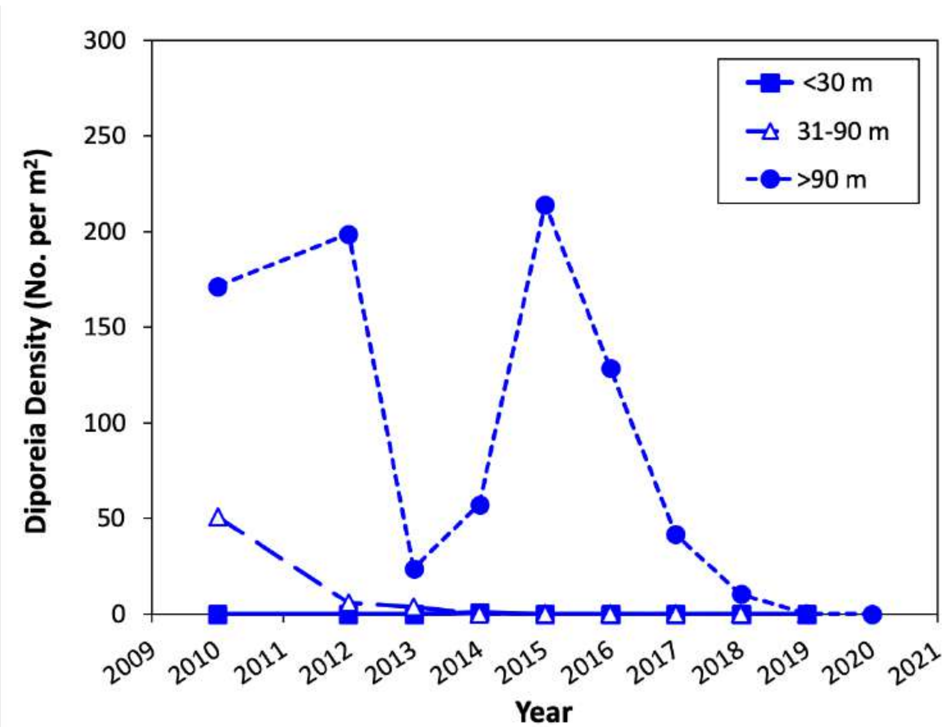

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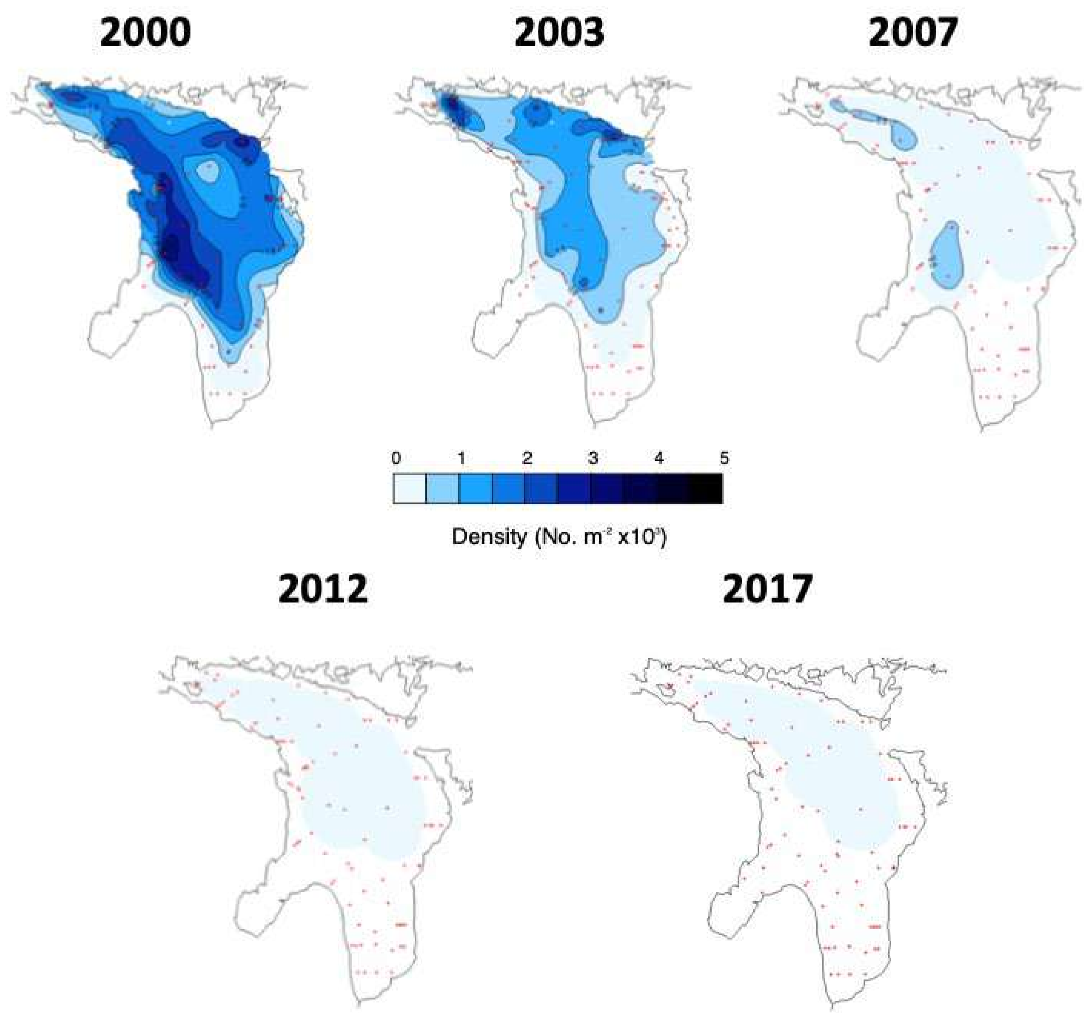

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Sub-Indicator: Lake Sturgeon

Overall Assessment

Status: Poor

Trends:

10-Year Trend: Improving

Long-term Trend (1980-2020): Undetermined

Rationale: There are remnant populations of Lake Sturgeon in each of the Great Lakes basins, but few of these populations are large. Progress continues as agencies learn more about population status in many tributaries and the Great Lakes proper. Confirmed observations and captures of Lake Sturgeon continue to increase in all lakes. Stocking is contributing to increased abundance in several areas. The trend for the overall and lake-by-lake assessments are improving over the last ten years based on increased observations, stocking, and habitat restoration efforts, however the number of self-sustaining populations remains poor compared to historical estimates. In many areas, habitat restoration is needed to attain full or significant recovery because spawning and rearing habitat has been destroyed or altered, or access has been blocked.

Lake-by-Lake Assessment

Lake Superior

Status: Poor

10-Year Trend: Unchanging

Long-term Trend (1980-2020): Improving

Rationale: Populations meet Lake Superior self-sustaining criteria in 2 of 19 historic spawning tributaries. Mature adults are known to spawn and produce evidence of successful reproduction (e.g., young-of-year or juveniles) in 11 tributaries. Adult abundance is improving where stocking has occurred as fish start to mature, but there are no spawning assessments on many tributaries. Juvenile abundance is unchanged over two lake-wide Index Survey years (2011 and 2016).

Lake Michigan

Status: Poor

10-Year Trend: Improving

Long-term Trend (2005-2020): Undetermined

Rationale: Wild populations of Lake Sturgeon spawn annually and recruitment can be detected in eight tributaries to Lake Michigan. Stocking appears to be reestablishing populations associated with several rivers, and ongoing supplementation continues to enhance recruitment for two wild populations. Sturgeon from stocking efforts are just beginning to mature and some have been observed returning to target rivers during the spawning season. Access to spawning and rearing habitat has increased with removal of several dams on some rivers, creation of spawning habitat on others, and continued passage of sturgeon around the lower two dams on the Menominee River. Catch rates of both wild and stocked juvenile Lake Sturgeon have increased in assessment gill nets in several areas over the past 10 years suggesting increasing recruitment.

Lake Huron (including St. Marys River)

Status: Poor

10-Year Trend: Improving

Long-term Trend (2000-2020): Undetermined

Rationale: Wild populations of Lake Sturgeon spawn annually and recruitment can be detected in eight tributaries to Lake Huron. Stocks of mixed sizes are consistently captured in the North Channel, Georgian Bay, St. Marys River, southern Lake Huron and Saginaw Bay. Lake Sturgeon stocking is taking place in the Saginaw River watershed, a tributary to Lake Huron. Reports of juvenile Lake Sturgeon captured in the Saginaw River and Bay are increasing along with juvenile catch per unit effort in the St. Marys River.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend: Improving

Long-term Trend (2000-2020): Undetermined

Rationale: Lake Sturgeon status remains poor as only three of the historic 15 tributaries contain self-sustaining populations. Catch per unit effort of juvenile Lake Sturgeon from fishery independent survey gear indicates an improving trend in the western basin of Lake Erie and in the North Channel of the St. Clair River (MDNR unpublished data). Many juvenile Lake Sturgeon captured in the western basin were recaptures from stocking events in the Maumee River. Spawning occurs in the Detroit and St. Clair rivers as well as Buffalo Harbor near the upper Niagara River.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Poor

10-Year Trend: Improving

Long-term Trend (2000-2020): Undetermined

Rationale: Lake-wide incidental catches since 1995 indicate a possible improvement in their status. Spawning occurs in the lower Niagara River, Trent River, and Black River. There are sizeable populations within the Ottawa and St. Lawrence River systems. Stocking for restoration began in 1995 in New York. From 2013 to 2020, Lake Sturgeon fingerlings were stocked in nine locations. In the same time period, young fish spawned by stocked populations were detected in Oneida Lake (Jackson et al. 2017) and the Oswegatchie River (NYSDEC and USFWS unpublished data). Spawning ready females have been captured in the Seneca River since 2013 and ripe males from previous stockings were captured in the Genesee River in 2016 (Dawn Dittman, USGS, personal communication).

Status Assessment Definitions

The status assessment is based on the number of historic Lake Sturgeon populations (Proceedings of the 2008 Great Lakes Lake Sturgeon Coordination Meeting (basin tables updated in 2013); http://www.fws.gov/midwest/sturgeon/index.htm). A self-sustaining population is defined as containing sexually mature adults spawning within a major watershed and evidence of recruitment. The number of self-sustaining populations may differ from the number identified in figures 3 – 7. The figures are meant to describe the "status" of Lake Sturgeon in certain areas, not necessarily the number of self-sustaining populations. For instance, in Lake Erie we suspect there were once 15 sites/tributaries with self-sustaining populations However, the maps also include the status of significant areas of Lake Sturgeon catches or other areas of interest (ex. Rondea Harbor, Pelee Island Shoals, etc.). The status and names of significant areas can be seen in Tables 1 – 5.

Note: The estimated number of historic Lake Sturgeon populations by lake was:

- Lake Superior: 19;
- Lake Michigan: 29;
- Lake Huron: 33;
- Lake Erie: 15; and
- Lake Ontario: 15

Good: Greater than 75% of all historic populations are self-sustaining.

Fair: Between 50-75% of all historic populations are self-sustaining.

Poor: Fewer than 50% of all historic populations are self-sustaining.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Number of self-sustaining populations of Lake Sturgeon has increased and/or model-generated estimates of abundance have increased and/or catch per unit effort of juvenile Lake Sturgeon from fishery independent surveys have increased.

Unchanging: Number of self-sustaining populations of Lake Sturgeon remains unchanged or model-generated estimates of abundance remain unchanged or catch per unit effort of juvenile Lake Sturgeon from fishery independent surveys remain unchanged.

Deteriorating: Number of self-sustaining populations of Lake Sturgeon has decreased and/or model-generated estimates of abundance have decreased and/or catch per unit effort of juvenile Lake Sturgeon from fishery independent surveys have decreased.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

Sufficient self-sustaining Lake Sturgeon populations which would allow state, provincial and federal agencies to remove this species from threatened or endangered lists.

Sub-Indicator Purpose

The purpose of this sub-indicator is to measure status and trends in population abundance of key life stages, distribution, habitat utilization, and recruitment of Lake Sturgeon in the Great Lakes and their connecting waterways and tributaries. Lake Sturgeon are representative of healthy fish communities in major habitats of the Great Lakes and support valuable fisheries in the Great Lakes and reflect ecosystem health through their roles in the aquatic food web.

Ecosystem Objective

Conserve, enhance, or rehabilitate self-sustaining populations of Lake Sturgeon where the species historically occurred and at a level that will permit all state, provincial and federal de-listings of classifications that derive from degraded or impaired populations (e.g., threatened, endangered or at-risk species).

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should “support healthy and productive wetlands and other habitats to sustain resilient populations of native species.”

Measure

This sub-indicator consists of standardized scoring of lake-specific adult abundance, juvenile abundance (recruitment), and number of self-sustaining populations for Lake Sturgeon. Data availability (quantity and quality) may limit complete spatial coverage of each lake and may only reflect area ranges of Lake Sturgeon stocks in each lake. Information from a specific area representing ideal habitats for Lake Sturgeon in that lake is considered appropriate for the purpose of this sub-indicator. The standard scoring of each fish species at each lake/location should be developed by fisheries experts in the inter-jurisdictional Lake Technical Committees of the Great Lakes Fishery Commission. The following hierarchy of data sources are used for calculating adult abundance and recruitment scores:

1. Number of Great Lakes self-sustaining Lake Sturgeon populations
2. Model-generated estimates of abundance for Lake Sturgeon under inter-jurisdictional fisheries management
3. Catch per unit effort of juvenile Lake Sturgeon from fishery-independent survey gears

Ecological Condition

Background

Lake Sturgeon (*Acipenser fulvescens*) were historically abundant in the Great Lakes with spawning populations using many of the major tributaries, connecting waters, and shoal areas across the basin. Prior to European settlement of the region, they were a dominant component of the nearshore benthivore fish community in each of the Great Lakes (Baldwin *et al.* 1979, Hay-Chmielewski and Whelan 1997, Haxton *et al.* 2014). In the mid- to late 1800s, they contributed significantly as a commercial species ranking among the five most abundant species in the commercial catch (Baldwin *et al.* 1979, Figure 1).

The decline of Lake Sturgeon populations in the Great Lakes was rapid and commensurate with habitat destruction, degraded water quality, and intensive fishing associated with settlement and development of the region. Sturgeon were initially considered a nuisance species of little value by European settlers, but by the mid-1800s, their value as a commercial species began to be recognized and a lucrative fishery developed. In less than 50 years, their abundance had declined sharply, and since 1900, they have remained a highly depleted species. Sturgeon remain extirpated from many tributaries and waters where they once spawned and flourished (Figures 2-7). They are considered rare, endangered, threatened, or of watch or special concern status by the various Great Lakes fisheries management agencies. Their harvest is currently prohibited or highly regulated in waters of the Great Lakes.

Status of Lake Sturgeon

Wild Lake Sturgeon populations remain throughout the Great Lakes basin though most persist at a small fraction of their historical abundance. The number of self-sustaining populations is less than half of what was historically estimated in each of the Great Lakes. In many systems, access to spawning habitat has been blocked and other habitats have been altered. However, there are remnant populations in each basin of the Great Lakes (Figures 3-7) and a few of these populations are large in number (thousands of fish). Genetic analysis has shown that Great Lakes populations are regionally structured and show significant diversity within and among lakes (DeHaan et *al*. 2006, Welsh et *al*. 2008).

Lake Superior

The Lake Sturgeon Rehabilitation Plan for Lake Superior (Auer 2003) serves as the guiding document for agency activities and defines criteria for self-sustaining populations in Lake Superior. The Lake Superior Fish Community Objectives (FCO) uses indicators to measure rehabilitation progress towards self-sustaining populations. Those indicators are 1) 1,500 mature adults (400 in spawning run) with a total adult population sex ratio around 1:1 (males:females) and spawning sex ratio around 4:1 or lower in a common tributary, and 2) 0.5 CPUE of ages 4-8 in the Lake Sturgeon Index Survey. These FCO indicators are in line with criteria of the Lake Superior Rehabilitation Plan and State of the Great Lakes sub-indicator reporting to measure rehabilitation progress.

There are 19 tributaries to Lake Superior and Lake Nipigon identified as supporting current or historic populations (Holey et *al*. 2000; Auer 2003; Pratt et *al*. 2016). Recent discussions within the Lake Superior Lake Sturgeon Work Group determined that two tributaries (Stokely Creek and Harmony River) were likely incorrectly identified during historic surveys and are not capable of supporting Lake Sturgeon populations. This reduces the number of spawning tributaries from 21 (as reported in the past) to 19. Populations in the Bad River, Wisconsin and Sturgeon River, Michigan meet Lake Superior Rehabilitation Plan criteria for self-sustaining populations (Auer 2003, Pratt et *al*. 2016). The Goulais, Pic, White, and Kaministiquia rivers in Ontario meet some rehabilitation criteria, but additional assessments are needed (Pratt et *al*. 2016).

Extant populations with mature adults in spawning runs are found in 11 Lake Superior tributaries (Figure 3; Table 1). In the St. Louis River, Minnesota/Wisconsin catch of adults increased since 2006 to a high of 330 fish below the Fond du Lac Dam, Minnesota but the sex ratio remains highly skewed towards males. The Bad River, Wisconsin population size was estimated at 1,426 males and 882 females in 2017 and annual spawning run estimates indicated a stable to increasing population trajectory (Schloesser and Quinlan 2019). In the Ontonagon River, Michigan a few (<5) fish have been captured or observed upriver near suspected spawning areas, which represent some of the first documented adults returning to spawning grounds from stockings that started in 1998 (stocking continues to present day). The Sturgeon River, Michigan spawning population size was estimated at 1,808 fish, and this population is the gamete source for stocking efforts in the Ontonagon River and Upper St. Louis River (Hayes and Caroffino 2012). In the Goulais River, Ontario, an estimate of 220 adult females was back-calculated from a juvenile abundance estimate in Goulais Bay, indicating it does not meet Lake Superior criteria for a self-sustaining population (manuscript in preparation; Pratt et *al.* 2014). There is no recent data from the Batchawana River, though the population is believed to be stable (Lacho et *al.* 2021). Spawning was documented in the Michipicoten River in 2016, but the population is thought to be small (Lacho et *al.* 2021). In the Pic and White rivers, Ontario, 321 Lake Sturgeon were captured and 95 were tagged in a movement study that observed presumed spawners moving upriver to the first insuperable barrier (Ecclestone et *al.* 2020). In the Nipigon River, Ontario, only 17 adult Lake Sturgeon were captured over three years of spring sampling between 2013 and 2015 and it is unknown if recruitment has occurred due to spawning in the river over the last decade (Barth et *al.* 2018; Lacho et *al.* 2021). In the Black Sturgeon River, Ontario mark-recapture analysis estimated approximately 90 spawners below the Camp 43 Dam in both 2003 and 2004 (Friday 2004). Lake Sturgeon in the Kaministiquia River, Ontario are year-round residents and in 2005 and 2006 genetic analysis of 769 larvae estimated the effective population size to be fewer than 100 breeders (Friday and Chase 2005, Welsh et *al.* 2015). Among all Lake Superior spawning tributaries, adult abundance is highly variable and ranges from extirpated to self-sustaining. There are no assessment activities on many spawning tributaries making it difficult to fully measure rehabilitation progress or population trends.

Juvenile Lake Sturgeon abundance (age 4-8) is measured through the Lake Sturgeon Index Survey for Lake Superior, which is conducted lake-wide on a five-year cycle (2011 and 2016) and annually at select locations (i.e., St. Louis River, Ontonagon River, Goulais River; Schloesser 2020). Mean catch per unit effort (CPUE) over the 2011 and 2016 surveys exceeded the recruitment target of 0.5 juveniles/net at the Bad, Ontonagon, Goulais, and Pic/White rivers. However, that threshold was derived using the same survey data we are evaluating here, so caution must be used if trying to determine rehabilitation progress. Catch rates of juvenile Lake Sturgeon were similar between the 2011 and 2016 Index Surveys at most locations. The exception was the Ontonagon River, where catches were lower in 2016, which corresponded to low stocking numbers and years when the Ontonagon stream side facility was not operating. Catch rates of juveniles in Goulais Bay were significantly higher than any other Lake Superior tributary. Juvenile abundance in Goulais Bay, Ontario was estimated at 4,977 (95% CI 3,295-7,517; Pratt et *al.* 2014). In the St. Louis River, catches of larval Lake Sturgeon have increased from ≤ 5 each year during 2011 to 2016, to a high of 1,028 in 2019. In general, catch rates of juvenile Lake Sturgeon in the lake-wide Index Surveys were unchanged over two sampling periods.

Spawning runs are absent, very small (<25 individuals), or undocumented in at least eight Lake Superior tributaries and only two populations meet Lake Superior Rehabilitation Plan criteria for self-sustaining populations. Lake Sturgeon abundance remains a small fraction of historical abundance. Haxton et *al.* (2014) used two methods to estimate historical Lake Superior adult abundance at 44,000 and 14,127 individuals, which they considered conservative. The 10-year trend is “unchanged” based on no change in the number of self-sustaining populations and similar CPUE in the 2011 and 2016 Index Surveys. The long-term trend is “improving” based on increasing adult and juvenile abundance where stocking has occurred. Adult spawning assessments are needed on many tributaries to better measure rehabilitation progress.

Lake Michigan

Remnant populations currently are known to spawn in waters of at least nine tributaries having unimpeded connections to Lake Michigan (Figure 4; Table 2; Schneeberger et *al.* 2005, Clapp et *al.* 2012, Hayes and Caroffino 2012, WDNR 2019). The Menominee River and the Peshtigo River continue to support the largest populations with annual spawning runs of several hundred adults (Elliott and Gunderman 2008, WDNR 2019, E. Baker, MDNR, personal communication). Six other rivers (Lower Fox, Oconto, Manistee, Muskegon, Grand, and Kalamazoo Rivers) continue to support smaller populations of >75 adults (Baker 2006, Elliott and Gunderman 2008, Hayes and Caroffino 2012, Harris et *al.* 2017, WDNR 2019, Partridge 2021, Tucker et al. 2021). Successful reproduction has been documented in all eight of these rivers, and age-0 juveniles were captured in many of these systems. Adult sturgeon were occasionally observed in the lower Manistique, Cedar, Milwaukee, Boardman, and St. Joseph Rivers during spawning times but spawning or recruitment has not been documented and abundance is considered low (< 25 adults, Hayes and Caroffino 2012, B. Eggold, WDNR, personal communication). A large self-sustaining population exists in the Lake Winnebago system upstream of the lower Fox River. This population spawns in the Wolf and Upper Fox Rivers and supports an active but highly regulated winter recreational spear fishery. The Lake Winnebago Lake Sturgeon stock is disconnected from the Lake Michigan population by a series of nine locks and dams on the Fox River. Out migration of Lake Sturgeon over these barriers occasionally occurs but upstream migration is completely blocked. The upper Menominee River also supports populations which are separated from each other and from the lower Menominee River population by several dams. The uppermost populations support a closely regulated fall hook and line fishery.

Continued monitoring and assessment has provided a growing collection of data on abundance, reproduction and recruitment for most of these Lake Michigan tributary populations (Caroffino et al. 2007, Harris *et al.* 2017, Lawrence *et al.* 2020, LRBOI 2020, Tucker et al. 2021, E. Baker, MDNR, unpublished data, J. Lorenz, Gun Lake Tribe, C. Jerome, Little River Band of Ottawa Indians, personal communication). These efforts suggest abundance of spawners in several rivers has increased over the last two decades. Additionally, catch rates of wild juveniles in assessment gill nets in Lake Michigan has increased over the past 10 years (Broadway *et al.* 2019, Ben Turschak, MDNR, unpublished data). This and other genetic based mixed stock analysis suggest that increased recruitment for many populations has been occurring for several years (Scribner *et al.* in review).

One area of continued management focus is minimizing mortality of age-0 sturgeon associated with sea lamprey treatments in high alkalinity rivers like the Muskegon and Manistee Rivers. Mortalities in these two rivers have been documented since 2013 and suggest that recruitment may have been reduced in these rivers during treatment years for decades (Smith 2014, LRBOI 2016, Pratt *et al.* 2020, LRBOI 2020). Mitigation measures including the capture and removal of juvenile sturgeon from these rivers during treatment periods have been implemented by a partnership of Tribal, State and Federal agencies since 2016.

The coordinated use of streamside facilities to culture sturgeon for reintroduction and supplementation stocking is an important component of Lake Sturgeon restoration in Lake Michigan. Since 2006, Michigan DNR, Wisconsin DNR, US Fish and Wildlife Service, Little River Band of Ottawa Indians, Gun Lake Tribe, and Riveredge Nature Center, along with other volunteer groups, have coordinated in the use of streamside facilities to rear and stock 4-6 month old fingerling sturgeon into six rivers (Milwaukee, Kewaunee, Cedar, Whitefish, Manistee and Kalamazoo) (Eggold *et al.* 2012). Methods follow the 2010 Genetic Guidelines for the Stocking of Lake Sturgeon in the Great Lakes Basin (Welsh et *al.* 2010) which include using appropriate donor sources, rearing fish in natal river water to facilitate imprinting, and maximizing genetic diversity. Since 2005, over 45,000 fall fingerling age-0 Lake Sturgeon have been stocked into Lake Michigan tributaries using these methods. Multiple year classes of stocked juveniles have been captured in assessment gillnets fished in adjacent Lake Michigan waters and catch rates have increased since stocking began (Troy Zorn, Michigan DNR, unpublished data; Tom Burzinski, Wisconsin DNR, unpublished data). Some of the earlieMost stocked fish are beginning to mature and have been observed returning to target rivers during the spawning season (B. Eggold, WDNR, personal communication, LRBOI 2020). The intent is to continue these stocking efforts for at least 20 years to establish founding populations of at least 750 mature adults in each river. Stocking also occurred in the upper Menominee River from 1994-2017 and in portions of the Winnebago system (WDNR 2019).

Habitat evaluations have been conducted in many of Lake Michigan's sturgeon tributaries (Daugherty et *al.* 2008) and have guided habitat restoration, dam removal, and fish passage projects. A fish elevator for upstream passage and downstream protection and bypass facilities have been used to pass Lake Sturgeon around the lower two dams on the Menominee River since 2015. Access to spawning and rearing habitat has increased with removal of several dams on some rivers and creation of spawning habitat on others.

Sturgeon populations in Lake Michigan continue to sustain themselves at a small fraction of their historical abundance. An optimistic estimate of the lake-wide adult abundance is less than 10,000 adult fish, well below 1% of the most conservative estimates of historic abundance (Hay-Chmielewski and Whelan 1997, Haxton *et al.* 2014). While at least eight of the extant populations appear to be self-sustaining, only the Winnebago, Menominee, and Peshtigo River populations are currently abundant enough to be considered healthy and secure. The improving 10-year trend assessment is based on increased abundance and recruitment (based on expert opinion), stocking, and improved river connectivity. Comprehensive data was not available to report a status over the long-term time period.

Lake Huron

Lake Sturgeon spawning has been identified in the Garden, Mississaugi and Spanish Rivers in the North Channel, and in the Moon, Musquash, and Nottawasaga Rivers in Georgian Bay (Figure 5; Table 3). During the spring of 2019, eggs and larvae were collected in the Moon, Musquash, and Garden rivers corroborating continued use of these rivers by spawning Lake Sturgeon (L. O'Connor, DFO, personal communication; L. Sumner, AOFRC, personal communication). Lake Sturgeon eggs and larvae were collected in the St. Marys River during the summer of 2018 and 2019 and spawning continues to occur at the mouth of the St. Clair River in southern Lake Huron evident by catches of sexually mature fish (Roseman *et al.* 2020). The spawning population at the mouth of the St. Clair River in southern Lake Huron contains one of the largest populations of Lake Sturgeon in the Great Lakes with a stock size of 20,184 (95% CI = 12,533 – 27,816) (USFWS, unpublished data). A mark-recapture assessment was initiated in the St. Marys River in 2019 as a result of increased capture of juvenile Lake Sturgeon in both a 2017 fish community survey and in Michigan DNR creel (N. Godby, MDNR, personal communication; O'Connor *et al.* 2019). A total of 35 Lake Sturgeon were captured, including five < 1000 mm. The population size of Lake Sturgeon in the St. Marys River was last estimated in 2007 at 505 (95% CI = 388 – 692; Bauman *et al.* 2011). Spawning surveys in the Mississaugi and Nottawasaga Rivers have consistently captured hundreds of Lake Sturgeon while over 50 fish are commonly captured during surveys in the Spanish River.

Lake Sturgeon stocking began in the Saginaw River watershed in the fall of 2017. A total of 3,317 fall fingerlings have been stocked into the Cass, Flint, Shiawassee, and Tittabawassee rivers. While increased catch of juvenile Lake Sturgeon is not yet evident in fishery independent surveys conducted in Saginaw Bay (A. Briggs, MDNR, personal communication), angler reports of juvenile Lake Sturgeon have increased the last two years. Since the last reporting period, two dams (Corunna and Shiatown) have been removed in the Shiawassee River reconnecting nearly 30 miles (48 kilometers) of unimpeded access for fish passage.

Stocks of Lake Sturgeon in Lake Huron are monitored by various resource management agencies along with the volunteer efforts of commercial fishers. To date the combined efforts of researchers in U.S. and Canadian waters have resulted in thousands of Lake Sturgeon tagged in Saginaw Bay, southern Lake Huron, St. Mary's River, Georgian Bay and the North Channel, with stocks of relatively mixed sizes being captured at each of these general locations. Tag recoveries, telemetry studies, and genetic collections indicate that Lake Sturgeon are moving within and between jurisdictional boundaries and between lake basins. There is currently no commercial or recreational harvest of Lake Sturgeon in Lake Huron. Regulation of subsistence harvest in Lake Huron varies by agency and is largely unknown.

In an effort to understand the migration patterns of Lake Sturgeon in southern Lake Huron and the St. Clair River, 126 adult Lake Sturgeon have been implanted with acoustic transmitters. Utilizing the Great Lakes Acoustic Telemetry Observation System (GLATOS) over 25 million detections have been documented since 2011, providing valuable information regarding the movements of adult Lake Sturgeon in Lake Huron and the St. Clair River system (Kessel *et al.* 2018; Colborne *et al.* 2019). Additional telemetry studies are planned for the St. Marys River.

The Lake Sturgeon population in Lake Huron continues to be well below estimated historical levels (Hay-Chmielewski and Whelan 1997, Haxton et *al.* 2014). Only two self-sustaining population exists in Michigan waters of Lake Huron (St. Marys and St. Clair Rivers), while four (St. Marys, upper St. Clair, Spanish, and Mississaugi Rivers) exist in Canadian waters. The improving 10-year trend assessment is based on the stocking program initiated in the Saginaw River watershed, an increase in catch per unit effort of juvenile Lake Sturgeon from fishery independent survey gear in the St. Marys River, and improved river connectivity. Comprehensive data was not available to report a status over the long-term time period. Barriers and habitat degradation in Michigan's and Ontario's tributaries to Lake Huron continue to be a major impediment to successful rehabilitation in Lake Huron.

Lake Erie

Spawning has been identified at eight locations in the connecting waters between Lakes Huron and Erie (Figure 6; Table 4; Caswell et *al.* 2004; Manny and Kennedy 2002; Fischer et *al.* 2018) and was also detected in Buffalo Harbor during the spring of 2017 (Neuenhoff et *al.* 2018). The St. Clair – Detroit River System contains the largest population of Lake Sturgeon with unimpaired access to the Great Lakes with stock sizes of 20,184 (95% CI = 12,533 – 27,816) in the upper St. Clair River/southern Lake Huron, 6,523 (95% CI = 5,720 – 7,327) in the lower St. Clair River, and 6,416 (95% CI = 4,065 – 8,767) in the Detroit River. The geometric mean population growth rate for all stocks exhibit stable populations and range from 1.00 – 1.16 (USFWS and MDNR, unpublished data). The population size of adult Lake Sturgeon in the upper Niagara River between 2012 – 2018 was estimated at 889 (95% CI = 611 – 1,352; Withers et *al.* 2019). Sidescan sonar imagery for a roughly seven square mile (18 square kilometers) section of Buffalo Harbor has been collected to develop a categorical habitat map intended to identify potential Lake Sturgeon spawning habitat.

Recreational angling for Lake Sturgeon continues within Michigan waters of Lake St. Clair and the St. Clair River. While angling effort has increased over time, there has been no detectable population-level effect on Lake Sturgeon in the system (Briggs et al. 2020). The North Channel of the St. Clair River, Detroit River (East of Fighting Island), and the western basin near the Detroit River mouth have been identified as nursery areas indicated by consistent catches of juvenile sturgeon in survey fishing gear (MDNR and USFWS, unpublished data). Lake Sturgeon stocking began in the Maumee River, OH during the fall of 2018. A total of 5,865 fall fingerlings have been stocked. The Ohio DNR and commercial fishermen in Ohio waters have recaptured six stocked fish based on PIT tag information. The upper Niagara River is a suspected nursery area based on reports from anglers and divers (C. Legard NYSDEC, personal communication). However, a dedicated Lake Sturgeon survey has not been conducted in the upper Niagara River to confirm these reports. In the central and eastern basins of Lake Erie, the detection of Lake Sturgeon is low with only occasional catches of sub-adult or adult Lake Sturgeon in commercial and research fishing nets.

In an effort to understand the migration patterns of Lake Sturgeon in the St. Clair – Detroit River System, nearly 300 adult Lake Sturgeon have been implanted with acoustic transmitters. Utilizing the Great Lakes Acoustic Telemetry Observation System (GLATOS) over 25 million detections have been documented since 2011, providing valuable information regarding the movements of adult Lake Sturgeon in this system as well as Lakes Huron and Erie (Hondorp et *al.* 2017; Kessel et *al.* 2018; Colborne et *al.* 2019). Additional juvenile Lake Sturgeon acoustic telemetry studies are taking place in the western basin to evaluate survival and movement of fish stocked in the Maumee River and movement of sub-adults near the Detroit River mouth. In Buffalo Harbor, 66 adult Lake Sturgeon were implanted with acoustic transmitters between 2014 – 2019. While some tagged Lake Sturgeon demonstrated large-scale movements that included traveling into western Lake Erie, the majority of individuals spent their time in the eastern basin.

The Lake Sturgeon population in Lake Erie continues to be well below historical levels. Self-sustaining populations are found in three (St. Clair, Detroit, and upper Niagara Rivers) of the 15 historic tributaries in Lake Erie. Research efforts will continue to focus on identifying rivers with suitable habitat for reintroduction efforts, identification of spawning locations, habitat requirements, and migration patterns. The improving 10-year trend assessment is based on an increase in catch per unit effort of juvenile Lake Sturgeon from fishery independent survey gear in the western basin and in the North Channel of the St. Clair River, and improved river connectivity. Comprehensive data was not available to report a status over the long-term time period.

Lake Ontario/ Upper St. Lawrence River

The numbers of mature Lake Sturgeon are not well quantified for most of the spawning areas surrounding Lake Ontario; however, data is available to assess Lake Sturgeon status (Figure 7; Table 5). Current mark-recapture population estimates of 6,500 (95% confidence interval of 6,000 to 7,000) mature and immature fish have been reported for the lower Niagara River (Beisinger et *al.* 2014). Also, numbers of Lake Sturgeon counted at or near the two artificial spawning beds constructed in the vicinity of Iroquois Dam in the upper St. Lawrence River ranged between 122 and 395 at the peak of spawning activity during 2008-2012 (NYSDEC 2013). Spawning populations also exist at Black River, NY (Klindt and Gordon 2014), and the Trent River, ON (A. Mathers, OMNR, personal. communication); however, these populations are small – likely in the 10s to 100s of fish.

Several actions have been taken to promote sturgeon recovery. Commercial harvest of sturgeon in Lake Ontario and upper St. Lawrence River was banned in 1976 in New York and in 1984 in Ontario. In addition, all recreational fishing has been closed since 1979. During the past decade, artificial spawning shoals for Lake Sturgeon have been created in the upper St. Lawrence River and their success has been evaluated showing egg deposition and emergence of larvae (NYSDEC 2013). The removal of the Hogansburg hydroelectric dam on the St. Regis River has opened up over 30 kilometers of potential habitat and released the population trapped above the impoundment (McKenna et *al.* 2015).

Between 1993 and 2020, NYSDEC in collaboration with U.S. FWS, have stocked approximately 175,000 (0 to 14,047 fish per year) Lake Sturgeon into the Lake Ontario system. Gametes for these efforts were collected in St. Lawrence River (below Moses-Saunders power dam since 1996). Stocking locations extend from the Genesee River east to Lake St. Francis tributaries. Research on sturgeon stocked in the lower Genesee River documented high level of survival and good growth suggesting these types of habitats are highly suitable for sturgeon and also that stocking has the potential to increase sturgeon abundance substantially (Dittman and Zollweg 2006). It is expected that spawning populations based on stocked fish will develop in the Genesee River, as well as the Oswego River, in the next few years (Chalupnicki et *al.* 2011). Cornell University researchers have captured wild produced fish from the 2011, 2012, and 2014 year classes in Oneida Lake (Jackson et *al.* 2017). Spawning is also suspected within Oneida/ Seneca/ Oswego River system at Fulton, Cayuga outlet, and Caughdenoy Dam. Spawning beds built in Montezuma National Wildlife Refuge attracted spawning adults in 2016.

Targeted surveys of Lake Sturgeon in the Trent River have occurred annually in the spring (2016-18), however only two sturgeon have been observed in this time. An acoustic tag was successfully implanted in one Lake Sturgeon in 2017; the same fish was detected again during the 2018 spring survey, back in the Trent River. When stationary receiver data are downloaded, more detail regarding seasonal movements of this fish may be revealed.

Research will continue assessing the Lake Sturgeon spawning shoals for aggregations of adults, egg deposition, and fry emergence. Monitoring of sturgeon catches and population age structure via agency fish community assessment programs will provide an index of population status in, and recruitment to, eastern Lake Ontario. Targeted surveys of sturgeon in the lower Niagara River are required to monitor this population. Efforts to stock sturgeon by agencies appear to be highly successful and monitoring of its effects should continue. Because sturgeon become sexually mature at an advanced age, a decade or more may be needed to observe responses to restoration efforts.

The Lake Sturgeon population in Lake Ontario/Upper St. Lawrence River continues to be well below historical levels. Self-sustaining populations are found in two (Lower Niagara and Upper St. Lawrence Rivers) of the estimated 15 tributaries with historic Lake Sturgeon populations in this system. The improving 10-year trend assessment is based on stocking programs and improved river connectivity. Comprehensive data was not available to report a status over the long-term time period.

Linkages

- Benthos – degradation of benthos and changes in the macroinvertebrate community likely affect Lake Sturgeon growth and habitat use.
- Aquatic Habitat Connectivity – loss of aquatic connectivity has contributed to the decline of Lake Sturgeon.
- Impact of Aquatic Invasive Species, Rate of Invasion of Aquatic Non-Native Species and Dreissenid Mussels – An additional concern for Lake Sturgeon in many of the Great Lakes is the ecosystem changes that have resulted from high densities of invasive species such as Dreissenid Mussels and round gobies and the presumed related exposure to Botulism Type E which has produced measurable die-offs of Lake Sturgeon in several years since 2001.
- Sea Lamprey – best management practices should be employed when controlling sea lamprey populations in tributaries where age-0 Lake Sturgeon may be present (O'Connor *et al*. 2017; Pratt *et al*. 2020). Sea lamprey wounding rates and potential impacts to juvenile and adult Lake Sturgeon should continue to be monitored (Dobiesz *et al*. 2018; Briggs *et al*. 2021).
- Surface Water Temperature and Precipitation Amounts – the affect of watershed impacts and climate trends on available habitat for all life stages is a continued research need. Predications from global climate models suggest that the amount of Lake Sturgeon habitat will decrease in the future as water temperature approaches thermal tolerance values near 28 – 30°C (Lyons and Stewart 2014).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	No			

Data Limitations

This is a relatively costly sub-indicator that requires coordination between federal, state, tribal, and provincial agencies. Stocking programs should last 20-25 years in order to ensure a sufficient level of genetic diversity in the donor population (Welsh et *al*. 2010). In order to obtain accurate measures of adult abundance, mark-recapture assessments need to take place over at least seven years (upper end of time interval between female spawning events) to ensure females have the opportunity to be captured on subsequent surveys. Variations in spawning periodicity of Lake Sturgeon and the effect that river flow rates have on spawning could affect annual results and complicate interpretation of long-term trends. More monitoring is needed to determine the current status of Great Lakes Lake Sturgeon populations, particularly the juvenile life stage (age 0-15). Considerable research is required to continue to determine the habitat preferences and location of this age group of Lake Sturgeon.

Additional Information

Lake Sturgeon is identified as an important species in Great Lakes Fishery Commission's Fish Community Goals and Objectives for each of the Great Lakes. Lake Superior has a Lake Sturgeon rehabilitation plan, and many of the Great Lakes states have Lake Sturgeon recovery or rehabilitation plans which call for increasing numbers of Lake Sturgeon beyond current levels.

The "mesotrophic" cool-water fish community is associated with more productive waters in nearshore areas. Mesotrophic communities, along with oligotrophic and eutrophic communities are found to varying degrees in all five of the Great Lakes. Lake Erie contains mostly mesotrophic habitat. The iconic Lake Sturgeon is the longest lived and largest of all Great Lake fishes, and an indicator of the connectivity of tributaries. Being co-evolved with the rest of the fish community and the natural ecosystem of the Great Lakes, Lake Sturgeon and other native species represent the natural biodiversity of the lakes. They have been subjected to the full slate of environmental effects resulted from human disruption of the Great Lakes including habitat loss, nutrient pollution, and persistent toxic pollutants. While restoration efforts like stocking can complicate interpretation of their status, the successes of these species are indicative of progress toward the goals of the Great Lakes Water Quality Agreement (GLWQA).

Research and development is needed to determine ways to selectively pass Lake Sturgeon around man-made barriers on rivers while not passing undesirable species. In addition, there are significant, legal, logistical, and financial hurdles to overcome in order to restore degraded spawning and nursery habitats in connecting waterways and tributaries to the Great Lakes. More monitoring is needed to determine the current status of Great Lakes Lake Sturgeon populations, particularly the juvenile life stage.

As monitoring programs and techniques are refined, the ability to detect Lake Sturgeon has likely increased making it difficult to determine whether changes observed are a result of increasing populations or reflect more efficient monitoring. Integration and continuation of long-term standardized monitoring programs will help effectively assess the status of Lake Sturgeon stocks in each lake into the future.

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Table 1. Lake Sturgeon population status in Lake Superior, 2022.

Map number	Population/River	Status
1	Pigeon River	Extirpated
2	Kaminsitiquia River	Extant
3	Wolf River	Extirpated
4	Black Sturgeon River	Extant
5	Nipigon River	Extant
6	Gravel River	Extirpated
7	Prairie River	Extirpated
8	Pic River	Extant
9	White River	Extant
10	Michipicoten River	Extant
11	Batchawana River	Extant
12	Chippewa River	Extirpated
13	Goulais River	Extant
14	Tahquamenon River	Extirpated
15	Sturgeon River	Extant
16	Ontonagon River	Extirpated-Reintroduced
17	Montreal River	Extirpated
18	Bad River	Extant
19a	St. Louis River - Lower	Extirpated-Reintroduced-Extant
19b	St. Louis River - Upper	Extirpated-Reintroduced

Table 2. Lake Sturgeon population status in Lake Michigan, 2022.

Map number	Population/River	Status
1a	Menominee River (below last dam)	Extant
1b	Menominee River (below Grand Rapids dam)	Extant
1c	Menominee River (below White Rapids dam)	Extant
1d	Pike River	Extant
1e	Menominee R. (below Sturgeon Falls)	Extirpated-Reintroduced
2	Cedar River	Unknown-Reintroduced
3	Escanaba River	Extirpated
4	Whitefish River	Extirpated-Reintroduced
5	Sturgeon River	Extirpated
6	Manistique River	Extant
7	Indian Lake	Extant
8	Millecoquins River	Unknown
9	Bear Creek	Unknown
10	Boardman River	Unknown
11a	Manistee River (below last dam)	Extant-Supplementation
11b	Manistee River (above last dam)	Unknown
12	Pere Marquette River	Unknown
13	Ludington Shoal	Unknown
14	White River	Unknown
15	Muskegon River	Extant
16	Grand River	Extant
17	Kalamazoo River	Extant-Supplementation
18	St. Joseph Shoal	Unknown
19a	St. Joseph River	Extant
19b	St. Joseph River (above dams)	Unknown
20	Chicago Reef complex	Extirpated
21	Root River	Extirpated
22	Milwaukee River	Extirpated-Reintroduced
23	Barr Creek	Extirpated
24	Sheboygan River	Extirpated
25	Manitowoc River	Extirpated
26	East/West Twin Rivers	Extirpated
27	Kewaunee River	Extirpated-Reintroduced
28	Sturgeon Bay area shoals	Extirpated
29a	Fox River (below last dam)	Extant
29b	Fox River (above last dam)	Unknown
30a	L. Winnebago- Upper Fox River	Extant-Supplementation
30b	L. Winnebago- Wolf River	Extant
30c	Wolf River- above Shawano dam	Extirpated-Reintroduced
31	Oconto River	Extant
32	Peshtigo River	Extant

Table 3. Lake Sturgeon population status in Lake Huron, 2022.

Map number	Population/River	Status
1	St. Marys River	Extant
2	Root River	Extirpated
3	Garden River	Extant
4	Echo River	Extant
5	Thessalon River	Extant
6a	Mississaugi R (N. Channel)	Extant
6b	Mississagi River (river pop'n)	Extant
7	Blind River	Extirpated
8	Serpent River	Extirpated
9	Spanish River	Extant
10	French River	Extant
11	Key River	Unknown
12	Magnetawan River	Extant
13	Naiscoot River	Extant
14	Seguin River	Extant
15	Moon River	Extant
16	Go Home River	Extirpated
17	Musquash River	Unknown
18	Severn River	Extant
19	Sturgeon River	Extirpated
20	Nottawasaga River	Extant
21	Manitou River	Unknown
22	Sauble River	Unknown
23	Saugeen River	Unknown
24	AuSable River	Extirpated
25	Blue Point	Unknown
26	Saginaw Bay	Extant
27a	Saginaw River	Extirpated-Reintroduced-Extant
27b	Tittabawassee River	Extirpated-Reintroduced
27c	Flint River	Extirpated-Reintroduced
27d	Shiawassee River	Extirpated-Reintroduced
27e	Cass River	Extirpated-Reintroduced
28	Rifle River	Unknown
29	Au Gres River	Unknown
30	AuSable River	Extant
31	Thunder Bay River	Extirpated
32	Otsego Lake	Extant-Reintroduced
33	Ocqueoc River	Unknown
34a	Cheboygan River	Extant
34b	Black Lake	Extant-Supplementation
34c	Black River	Extant-Supplementation
34d	Mullett Lake	Extant-Supplementation
34e	Burt Lake	Extant-Supplementation
35	Carp River	Extant
36	Munuscong Bay	Extant

Table 4. Lake Sturgeon population status in Lake Erie, 2022.

Map number	Population/River	Status
1	Upper Niagara River/Buffalo Harbor	Extant
2	Eastern Basin	Extant
3	Cattaraugus Creek	Extirpated
4	Walnut Creek, PA nearshore	Unknown
5	Conneaut, OH nearshore	Extirpated
6	Cuyahoga River	Extirpated
7	Sandusky River	Extirpated
8	Portage River	Unknown
9	Maumee River	Extirpated-Reintroduced
10	Raisin River	Extirpated
11	Huron River	Extirpated
12	Detroit River	Extant
13	Lake St. Clair	Extant
14	Lower St. Clair River	Extant
15	Upper St. Clair River	Extant
16	Pelee Island nearshore	Unknown
17	Point Pelee shoals	Unknown
18	Rondeau Harbor	Unknown
19	Clear Creek	Unknown
20	Long Point Bay	Unknown

Table 5. Lake Sturgeon population status in Lake Ontario and the St. Lawrence River, 2022.

Map number	Population/River	Status
1	Don River	Extirpated
2	Ganaraska River	Extirpated
3	Trent River	Extant
4	Salmon River	Extirpated
5	Napanee River	Extirpated
6	Amherst Island shoal	Unknown
7	Black River	Extant
8	Oswego River	Extirpated-Reintroduced
9a	Oneida Lake	Extirpated-Reintroduced
9b	Cayuga Lake	Extirpated-Reintroduced
10	Genesee River	Extirpated-Reintroduced
11	Lower Niagara River	Extant
12	St. Lawrence River - Thousand Islands	Extant
13a	Oswegatchie River mouth	Unknown
13b	Oswegatchie River	Extant-Supplementation
14	Black Lake	Extant-Supplementation
15	St. Lawrence River - Lake St. Lawrence	Extant
16	St. Lawrence River - Lake St. Francis	Extant-Supplementation
17	Grasse River	Extant
18	Raquette River	Extant-Supplementation
19a	St. Regis River	Extant
19b	St. Regis River - above Hogansburg dam	Extirpated-Reintroduced
20	Salmon River	Extirpated
21	Ottawa River mouth - Lac Des Deux Montagnes	Extant
22a	Ottawa River - Lac Dollard Des Ormeaux	Extant
22b	Ottawa River - Lac Deschenes	Extant
22c	Ottawa River - Lac Des Chats	Extant
22d	Ottawa River - Lac Du Rocher Fendu	Extant
22e	Ottawa River - Lac Coulonge	Extant
22f	Ottawa River - lower Allumette Lake	Extant
22g	Ottawa River - upper Allumette Lake	Extant
22h	Ottawa River - Holden lake	Extant
22i	Ottawa River - Lac La Cave	Extant
23	St. Lawrence River - Lac Saint Louis to the limits of freshwaters (downstream Quebec City)	Extant
24	Lake Champlain	Extant

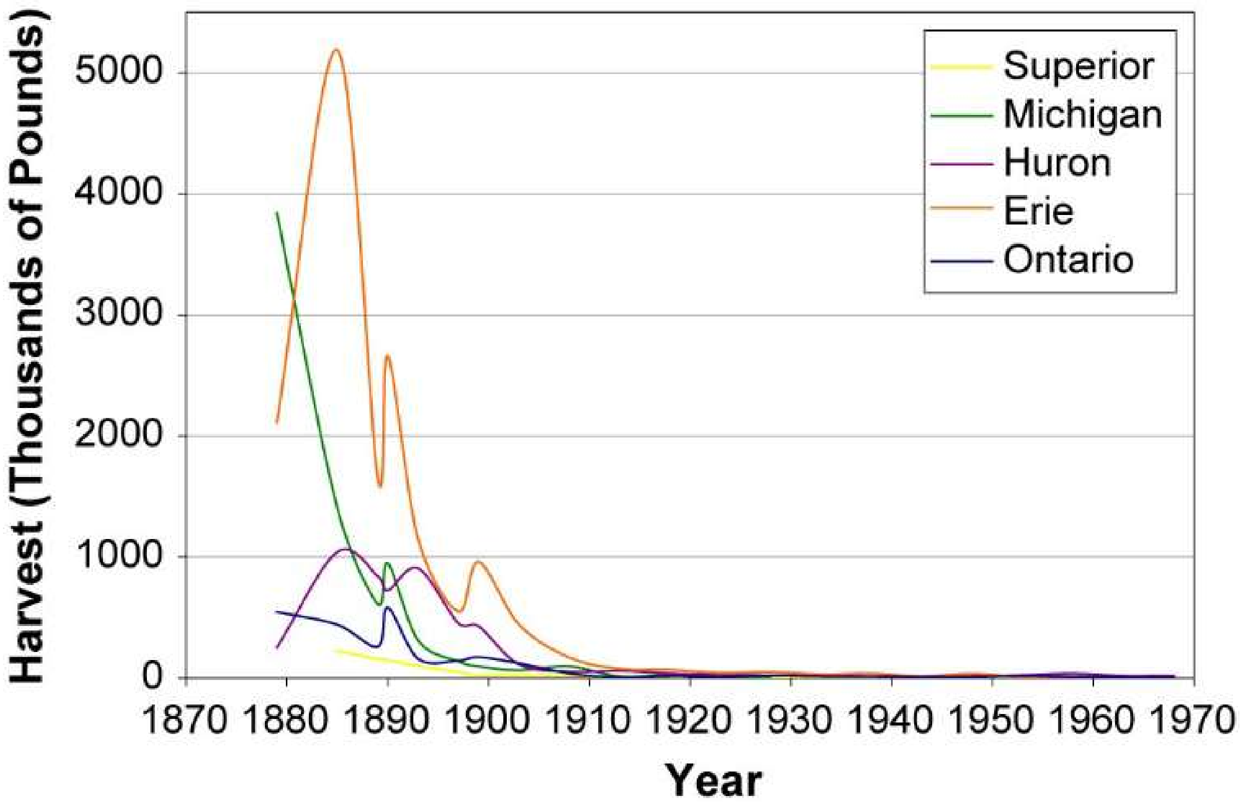

Figure 1. Historic Lake Sturgeon harvest from each of the Great Lakes. Source: Baldwin et *al*. 1979

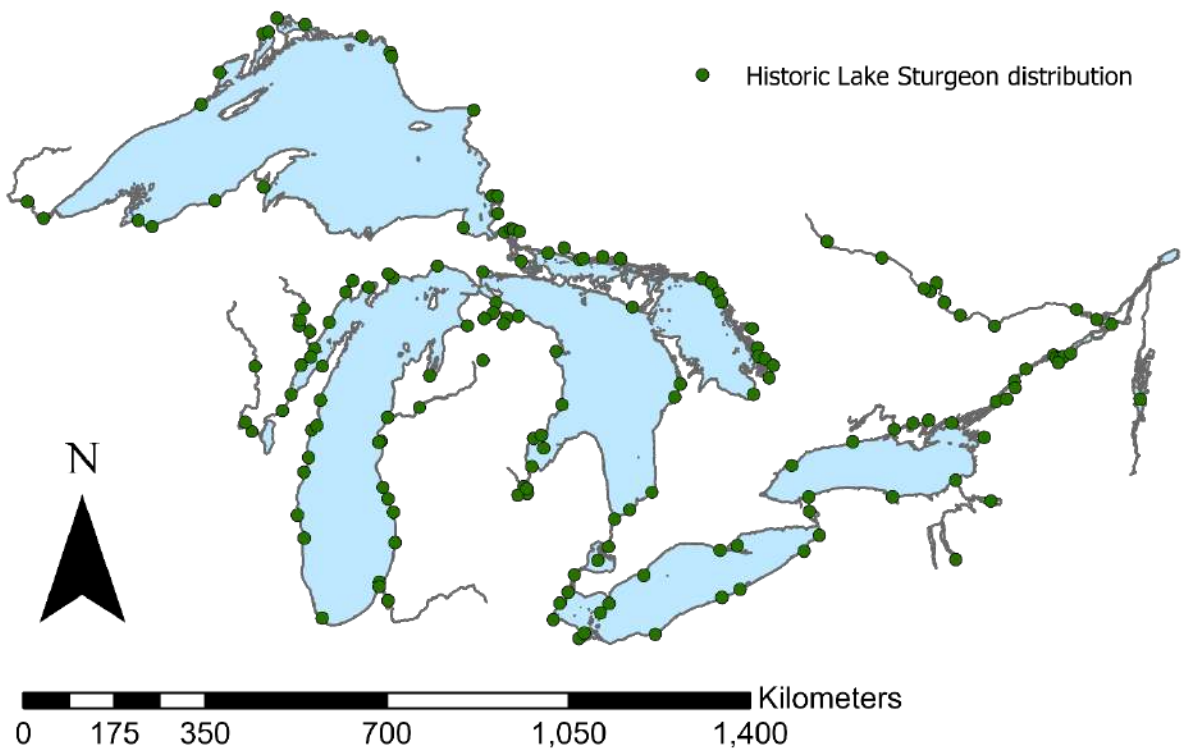

Figure 2. Historic distribution of Lake Sturgeon in the Great Lakes. Source: Lake Sturgeon Works Groups, Great Lakes Fishery Commission.

Lake Sturgeon Status

Lake Superior

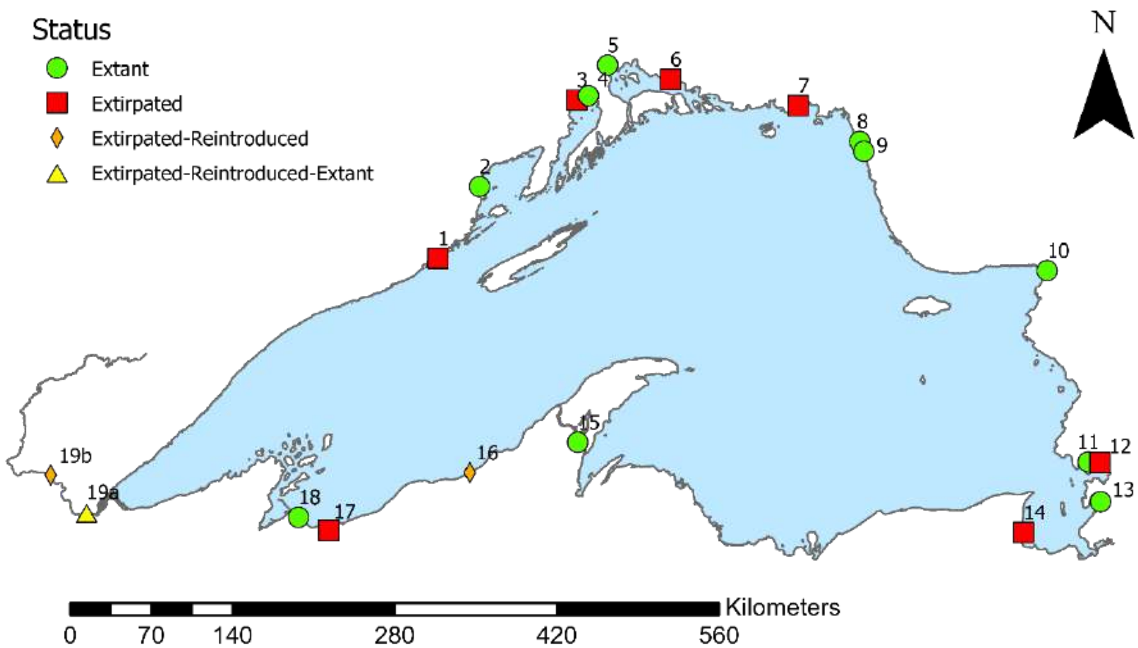

Figure 3. Lake Sturgeon population status in Lake Superior, 2022. Population status definitions are: Extirpated (no longer present); Extant (present and surviving; natural reproduction may or may not be known); Reintroduced (fish stocked into a system with an extirpated population); Supplementation (fish stocked into a system with an extant population); unknown (unknown). Source: Lake Superior Lake Sturgeon Work Group, Great Lakes Fishery Commission.

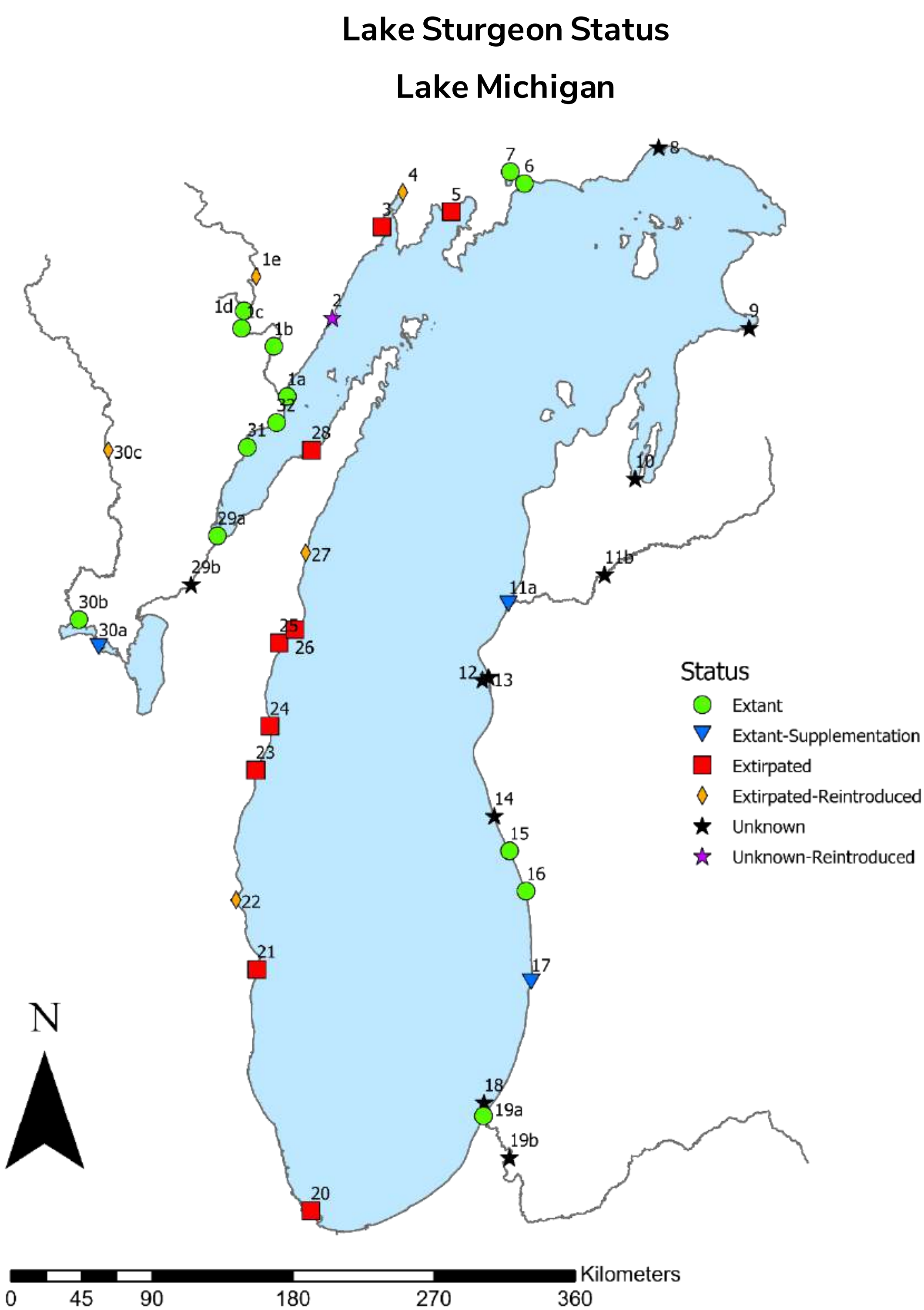

Figure 4. Lake Sturgeon population status in Lake Michigan, 2022. See figure 3 for population status definitions. Source: Lake Michigan Lake Sturgeon Task Group, Great Lakes Fishery Commission.

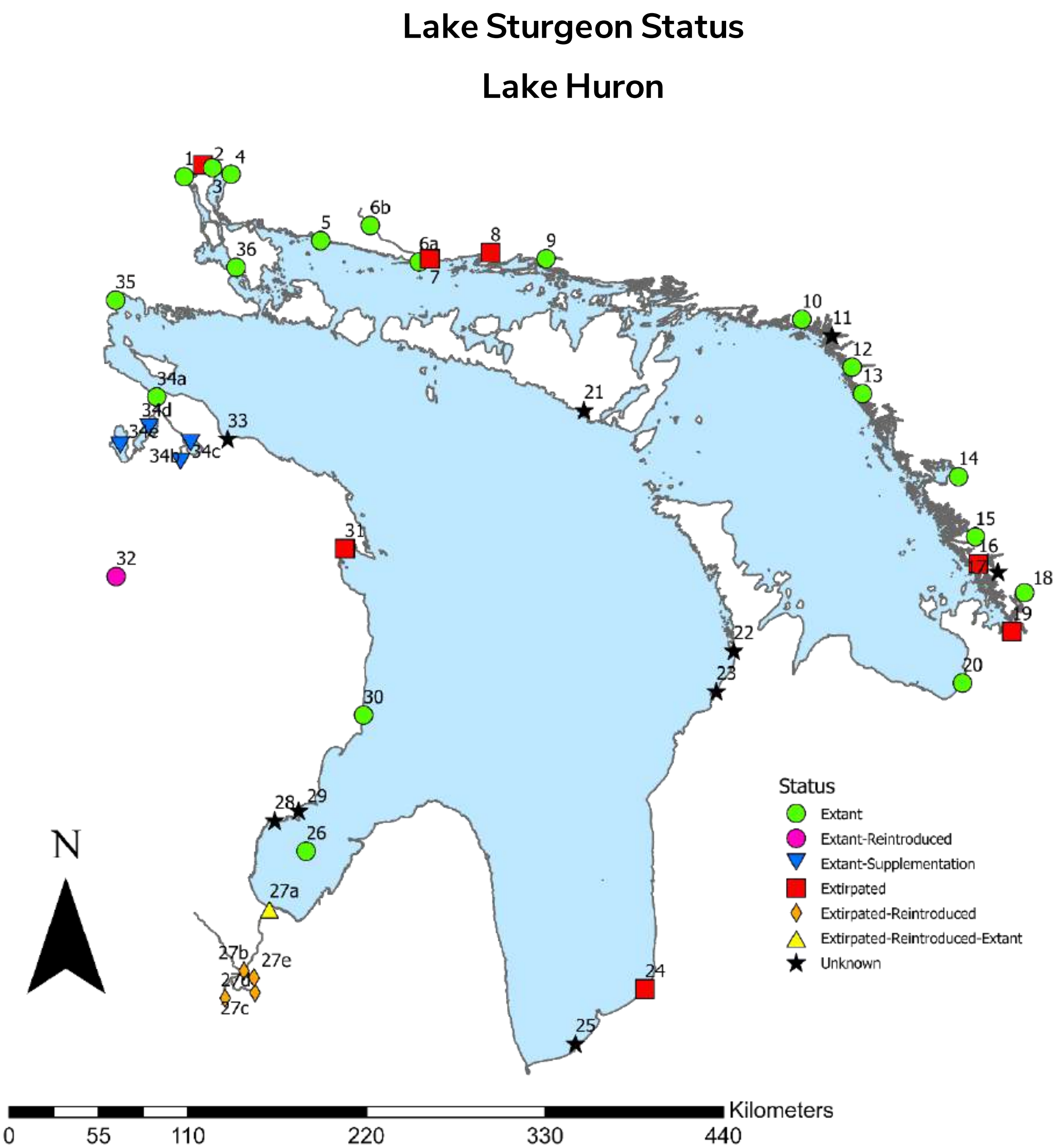

Figure 5. Lake Sturgeon population status in Lake Huron, 2022. See figure 3 for population status definitions. Source: Lake Huron Lake Sturgeon Task Group, Great Lakes Fishery Commission.

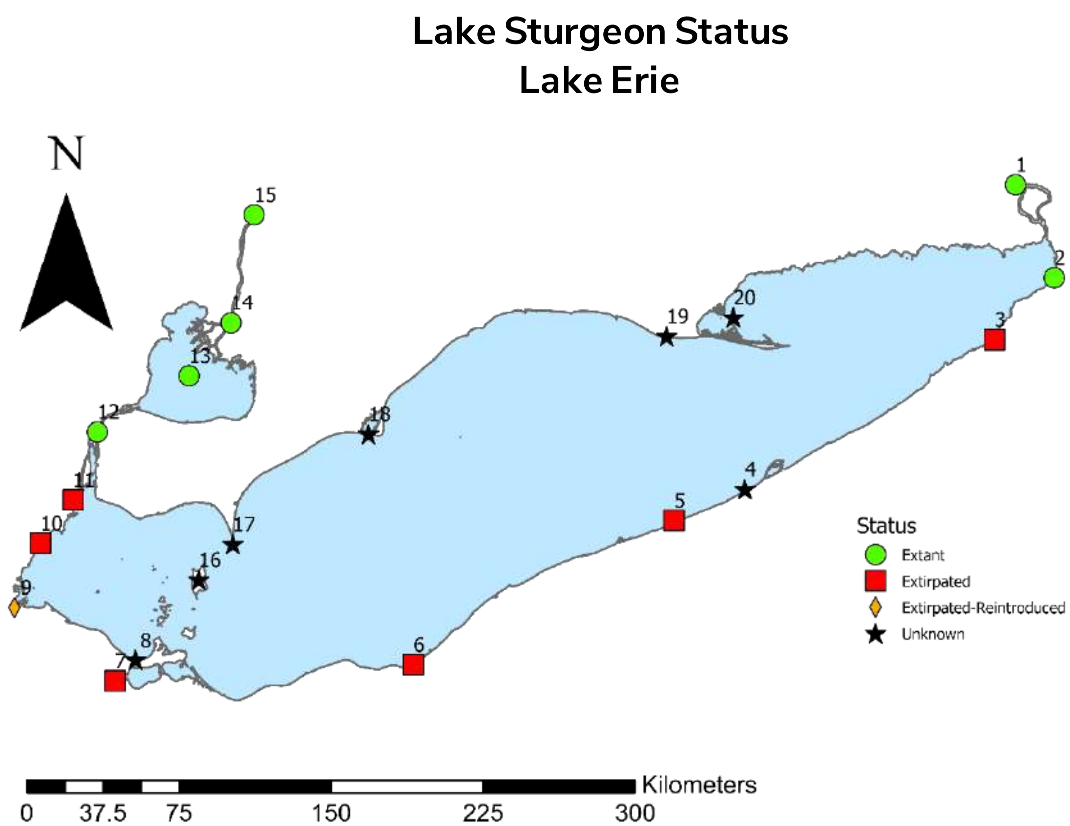

Figure 6. Lake Sturgeon population status in Lake Erie, 2022. See figure 3 for population status definitions. Source: Lake Erie Lake Sturgeon Working Group, Great Lakes Fishery Commission.

Lake Sturgeon Status
Lake Ontario and the St. Lawrence River

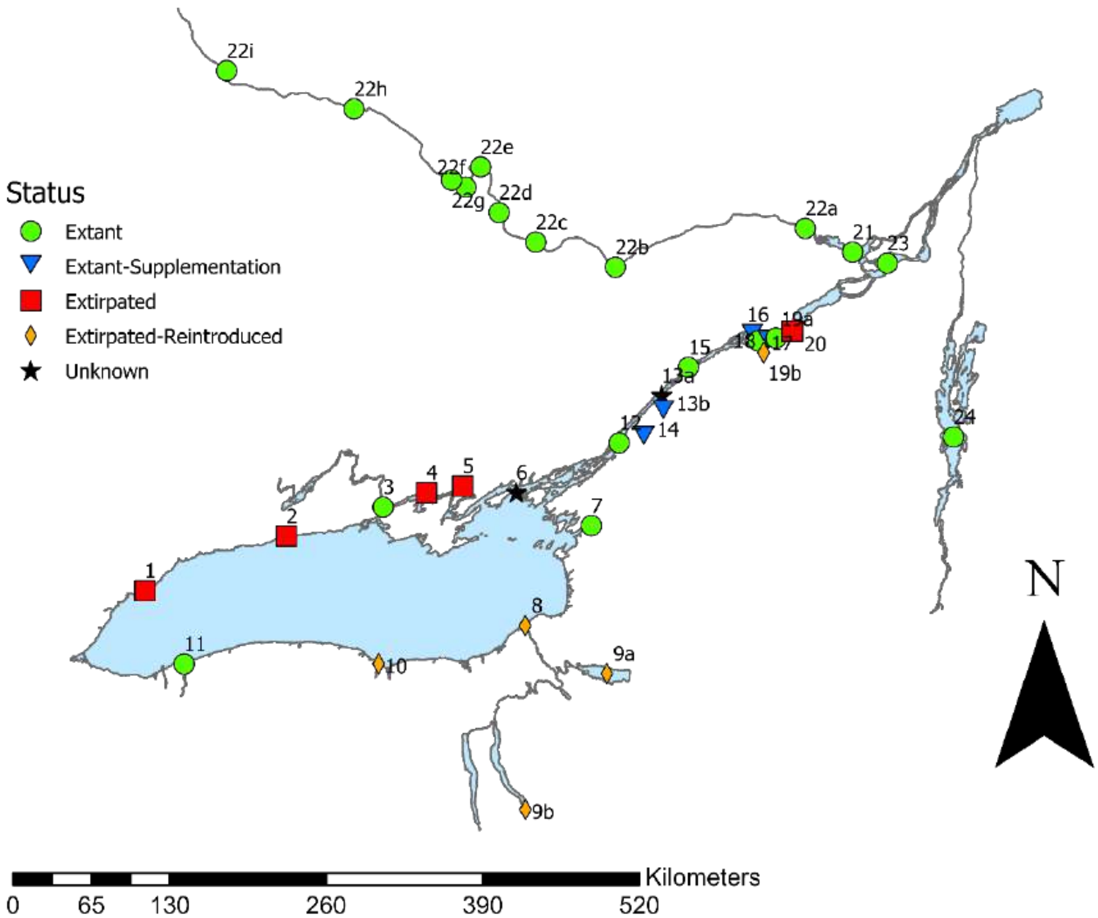

Figure 7. Lake Sturgeon population status in Lake Ontario and the St. Lawrence River, 2022. See figure 3 for population status definitions. Source: New York Lake Sturgeon Working Group, and Tim Haxton, OMNRF.

Sub-Indicator: Native Prey Fish Diversity

Overall Assessment

Status: Fair

10-Year Trend: (2011-2020): Undetermined
Long-term Trend (1973-2020): Undetermined

Rationale: Great Lakes prey fish diversity status remains 'Fair', but individual lake status varies from 'Good' in Lake Superior to 'Fair' in the remaining four lakes (Table 1). Lake Superior was the only lake where both prey fish diversity and the percent native species were categorized as 'Good'. Lake Erie prey fish status improved in this reporting cycle since the percent of native prey fishes increased to 'Fair' (Table 1). For the first time in this analysis, Lake Ontario's prey fish community diversity improved to 'Good', however since the percent native remained 'Poor' the overall status was 'Fair' (Table 1).

The long-term trend was 'Undetermined' because while six of the possible ten metrics (two per lake) were 'Unchanging', three were determined to be 'Improving', while one was 'Deteriorating' (Table 2). The 10-year trends were similarly diverse and resulted in an 'Undetermined' overall characterization (Table 2).

Prey fish biomass trends are not specifically graded in this indicator; however, these trends have generally declined or remained low, as oligotrophication continues in all main-lake habitats other than western Lake Erie, and native and stocked predator populations naturally have reduced prey fishes. Prey fish communities are driven by changing ecosystem conditions including productivity, fluctuating predator populations, climate, and non-native species. These factors are generally changing in similar directions across the region. However, because each lake is unique in its nutrient concentrations, morphometry, hydrology, and fish communities, the prey fish communities in each lake respond differently to ecosystem drivers (Figure 1).

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend (2011-2019): Unchanging

Long-term Trend (1978- 2019): Unchanging

Rationale: The Lake Superior prey fish community is categorized as 'Good', with individual metrics classified as 'Good' (proportion native species) and 'Good' (diversity). Lake Superior is the least anthropogenically-altered Great Lake and its prey fish community metrics reflect this more pristine state. Native species dominate a diverse prey fish community as measured by prey fish biomass and the Shannon Diversity Index (Tables 1 and 2, Figure 1). The percentage of the fish community biomass comprised of native species has averaged >80% over the long term and 82% over the past nine years (2011 – 2019). While Lake Superior prey fish metrics have been 'Unchanging', there are concerns among fish managers over population declines of coregonine species like Bloater (*Coregonus hoyi*), Cisco (*Coregonus artedi*), and Kiyi (*Coregonus kiyi*) which support significant commercial fisheries and are prey for Lake Trout (*Salvelinus namaycush*). Lake Superior prey fish data from 2020 were incomplete because COVID-19 precautions prevented sampling, and therefore community indices for that year were not calculated.

Lake Michigan

Status: Fair

10-Year Trend (2011-2020): Unchanging

Long-term Trend (1973-2020): Unchanging

Rationale: The Lake Michigan prey fish community is categorized as 'Fair', with individual metrics classified as 'Good' (proportion native species) and 'Fair' (diversity) (Table 1). This is identical to the status reported in 2019. When examined over the past ten years, trends in both the diversity index and the percent native species have improved since the previous assessment, shifting from 'Deteriorating' to 'Unchanging' and 'Unchanging' to 'Improving', respectively (Table 2). This in turn has improved the status of the overall 10-year trend to 'Unchanging'. While prey fish diversity has been 'Improving' over the longer time scale, the lack of directional change in the percent native metrics still results in an 'Unchanging' long-term trend (Table 2).

Lake Huron

Status: Fair

10-Year Trend (2011-2020): Unchanging

Long-term Trend (1976-2020): Undetermined

Rationale: The Lake Huron prey fish community is categorized as 'Fair', with the proportion of native species classifieds as 'Good' and diversity categorized as 'Fair' (Table 1). Over the 10-year period these metrics were unchanging; however, over the long term the percent native has 'Improved' while the diversity has 'Deteriorated,' resulting in an overall trend of 'Undetermined' (Table 2). The declining diversity trend in Lake Huron is driven primary by the shift from non-native Alewife (*Alosa pseudoharengus*) and Rainbow Smelt (*Osmerus mordax*) in the 1970s through 2000, to a community dominated by a single native species (Bloater). Long-term, consistent observations on prey fishes in the St Mary's River were not available for comparison.

Lake Erie

Status: Fair

10-Year Trend (2011-2020): Deteriorating

Long-term Trend (1990-2020): Unchanging

Rationale: The Lake Erie prey fish community status was classified as 'Fair, as both the percent native species and diversity metrics were classified as 'Fair' (Table 1). This is a change from the previous reporting period when the percent native species value fell in the 'Poor' category. Native species typically make up less than 50% of the Lake Erie prey fish community and while the proportion native has varied over time, it has not trended in a particular direction (Figure 1, Table 2). Lake Erie prey fish diversity values are among the highest in the Great Lakes, but they have declined slightly in the past 10 years (Figure 1, Table 2). Much of the changes in both metrics result from annual variability or declines in Emerald Shiners (*Notropis atherinoides*), Yellow Perch (*Perca flavescens*), Spottail Shiner (*Notropis hudsonius*) and Trout-perch (*Percopsis omiscomaycus*). The classifications for Lake Erie are based on lake-wide assessment trends and may not accurately reflect regional prey fish population status or trends. Lake Erie prey fish surveys results are presented as density, as opposed to biomass, which can result in greater variability in the metrics used in this report. Long-term, consistent observations for prey fishes in the St Clair-Detroit River or upper Niagara River were not available for comparison.

Lake Ontario

Status: Fair

10-Year Trend: Improving

Long-term Trend (1978 - 2020): Unchanging

Rationale: Lake Ontario prey fish percent native status was 'Poor', but the diversity index was 'Good', resulting in an overall status of 'Fair' (Table 1). Both the diversity index and proportion of native species were 'Improving' over the past 10 years (Table 2). These recent increases caused the long-term trend for these indicators to shift from previously 'Deteriorating' to now 'Unchanging'. Lake Ontario prey fish diversity is the lowest in the Great Lakes because Alewife continue to dominate the assemblage. However, as Alewife abundance has recently declined and Round Goby (*Neogobius melanostomus*) and Deepwater Sculpin (*Myoxocephalus thompsonii*) remain abundant, the community diversity improved. The increased proportion of Deepwater Sculpin also caused the proportion of native species trend to be 'Increasing'. Concerns over Alewife abundance declines have caused reductions in sport fish stockings. Prey fish data from 2020 was incomplete (COVID-19 precautions prevented sampling) and therefore community indices for that year were not calculated. Long-term, consistent observations on prey fishes in the lower Niagara River or St. Lawrence River were not available for comparison.

Status Assessment Definitions

Good: The average percent by biomass (or density) of native prey fish in the total prey fish catch is at or above 75% and the diversity index value is greater than or equal to 75% of the maximum diversity index value observed in the time series.

Fair: The average percent by biomass (or density) of native prey fish in the total prey fish catch is at or above 25% and the diversity index value is greater than or equal to 25% of the maximum diversity index value observed in the time series.

Poor: The average percent by biomass (or density) of native prey fish in the total prey fish catch is below 25% and the diversity index value is below 25 % of the maximum diversity index value observed in the time series.

Undetermined: Data are insufficient to assess the metrics.

If the two metrics status categories differ, the lake status will either be the average or conservatively be the lower of the two categories. For example, a status of 'good' for percent native and 'fair' for diversity will result in a status categorization of 'fair'. A status of 'good' and 'poor' results in an overall status of 'fair'.

Trend Assessment Definitions

Trends at 10-year and long-term timescales are evaluated for each metric using linear models with yearly observations. If trends are significant, positive, or negative, with a P-value of 0.10 or lower the trend is reported; otherwise the trend is considered Unchanging. If the trends of the two metrics differ the overall trend will conservatively be categorized as the 'lesser' of the two metrics (e.g. one metric Improving and the other Unchanging results in a classification of Unchanging). If individual metric trends are opposite (e.g. Improving and Deteriorating) the overall trend is Undetermined.

Improving: The slopes for year in both metrics is positive ($P < .10$).

Unchanging: The slope for year is insignificant, and one is positive OR both slopes are insignificant ($P < .10$).

Deteriorating: The slopes for year in both metrics are negative OR one slope is non-significant and one is negative ($P < .10$).

Undetermined: If data are not sufficient, or if the metric trends differ (e.g. 'Improving' and 'Deteriorating')

Endpoints and/or Targets

Lake-specific committees create Fish Community Objectives (FCOs) that identify how Great Lakes fisheries are managed according to the Joint Strategic Plan (Great Lakes Fishery Commission, 1981). Most of these objectives do not specify numerical targets or endpoints for prey fishes. Instead the objectives generally seek to 'maintain prey fish diversity' or 'maintain and restore native forage fish species' and often attempt to balance prey fishes (or forage fishes) with primary production or predator demand. The portions of these FCOs that relate to prey fishes are listed below.

Lake Superior: Fish Community Goal – "*To rehabilitate and maintain a diverse, healthy, and self-regulating fish community, dominated by indigenous species and supporting sustainable fisheries*". Additional principals note: "*Preservation of indigenous species is of the highest concern*" (Horns et al. 2003).

Lake Michigan: Planktivore Objective – "*Maintain a diversity of planktivore (prey) species at population levels matched to primary production and to predator demands. Expectations are for a lakewide planktivore biomass of 0.5 to 0.8 billion kg.*" (Eshenroder et al. 1995).

Lake Huron: Prey Objective – "*Maintain a diversity of prey species at population levels matched to primary production and to predator demands. Emphasis is placed on species diversity and self-regulation of the fish community*" (DesJardine et al. 1995).

Lake Erie: Prey Fish Objective – "*Maintain a diverse, abundant prey-fish community that is capable of sustaining abundant warm-, cool-, and cold-water predators and that contributes to ecosystem function and sustainable human use. The LEC recognizes that it cannot directly manage prey-fish populations, even though they are essential to support the Lake Erie basin fisheries. The LEC especially values native prey species but recognizes that naturalized prey species can be an important part of the prey-fish community, predator diets, and targeted fisheries. Status indicator: Prey-fish populations support predator condition and growth rates near the long-term average*" (Francis et al., 2020).

Lake Ontario: Offshore Pelagic Zone Objective- "*Increase prey-fish diversity – maintain and restore a diverse prey-fish community that includes Alewife, Cisco (formerly Lake Herring), Rainbow Smelt, Emerald Shiner, and Threespine Stickleback (Gasterosteus aculeatus). Status and trend indicators are 1) maintaining or increasing populations and increasing species diversity of the pelagic prey fish community including introduced species (Alewife, Rainbow Smelt) and selected native prey fish species (Threespine Stickleback, Emerald Shiner and Cisco); and 2) increasing spawning populations of Cisco (formerly Lake Herring) in the Bay of Quinte, Hamilton Harbor, and Chaumont Bay*" (Stewart et al., 2017). The introductory material also notes: "*The LOC will continue with programs to protect and restore native species with an emphasis on... Cisco (Lake Herring), Round Whitefish (Prosopium cylindraceum), Deepwater Sculpin and deepwater coregonines*", such as Bloater.

Sub-Indicator Purpose

This sub-indicator attempts to quantify the status and trends of Great Lake's prey fish communities according to their diversity and the percent of the community comprised of native species, which are common elements identified in the lake-specific FCOs. Prey fish abundance indices are also reported for context on the state of Great Lakes ecosystems.

Ecosystem Objective

Ecosystem objectives for prey fishes vary across lakes, but at the lake level generally seek to maintain diverse prey fish communities, that support predator populations, are in balance with primary productivity, and in many cases favor native prey fishes (Desjardine et al., 1995; Eshenroder et al., 1995; Francis et al., 2020; Horns et al., 2003; Stewart et al., 2017). This sub-indicator results also inform '*forage fish*' elements of the " *Joint strategic plan for management of all great lakes fisheries, as revised, 10 June 1997*" (Great Lakes Fishery Commission, 2007). In addition, this sub-indicator supports the Great Lakes Water Quality Agreement's Annex 4 as they consider setting the Substance Objective (phosphorus targets) by illustrating how phosphorus concentrations influence "*fisheries productivity requirements*" (United States of America and Canada, 2012). This sub-indicator supports General Objective #5 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species

Measure

The status and trends of prey fishes are based on the diversity and the percent of native species within the community. Community diversity is represented by the Shannon diversity index and the status of a period is a quantified against the maximum diversity value observed in the time series. Unfortunately, theoretical, or even widely agreed upon metrics or thresholds for what constitutes a prey fish community as 'good', 'fair', or 'poor' do not exist. Categorizing the state of prey fish communities is subjective and is influenced by an individual's perspective on the different ecosystem services the prey communities provide. For instance, while Lakes Erie and Ontario prey fish communities often exhibit low prey fish diversity or percent natives, these prey fish communities support the largest sport and commercial fisheries in the basin, providing provisioning and cultural ecosystems services to the region and nation. We recognize the metrics chosen to characterize prey fish status are imperfect and incomplete, but they serve as a starting point from which to understand differences among the lakes and time trends within a lake.

Prey fish abundance indices are also reported for context but are not a graded metric in this report because management actions for phosphorus reduction and predator abundance increases naturally reduce prey fish abundance. Energy properties of lake food webs require that prey fish biomass is negatively dependent on lake nutrient concentration and piscivorous fish abundance (Carpenter et al., 2001; Downing and Plante, 1993). Within the Great Lakes, resource management has, in most regions, successfully reduced phosphorus concentration below targets (Dove and Chapra, 2015) and restored piscivorous fish populations, both of which confound the utility of prey fish abundance as a graded metric.

Data used to calculate metrics are from bottom trawl surveys. These surveys have been generally conducted annually and consistently across time, sample across a wide range of available habitats and catch the most abundant prey fishes. Diversity and percent native metrics are based on biomass (kilograms per hectare) indices from bottom trawl catches, except for Lake Erie where data are represented by density indices (number per hectare). Data come from the USGS Great Lakes Science Center (GLSC), Ontario Ministry of Natural Resources, New York State Department of Environmental Conservation, Ohio Department of Natural Resources Division of Wildlife, and Pennsylvania Fish and Boat Commission. For Lakes Superior, Huron, Michigan and Ontario data are managed by the USGS GLSC. In Lake Erie data are managed and provided by the separate state, federal and provincial agencies. All data are provided at the whole-lake scale except for Lake Erie, which are provided at the basin scale and combined according to the proportional area of each of the three lake basins. Consistent annual or long-term data for prey fishes are not available for the connecting channel habitats.

Ecological Condition

Lake Superior, Status: Good, 10-year trend: Unchanging

Native species dominate the relatively diverse Lake Superior prey fish community (Figure 1). Rainbow Smelt are the only non-native species that substantially contributes to the Lake Superior bottom trawl catch (Vinson et al., 2020). Changes in Lake Superior prey fish metrics are primarily driven by fluctuations in coregonines (Bloater, Cisco, and Kiyi), Rainbow Smelt, and Lake Trout populations. Lake Trout are voracious predators of these prey species. These coregonine species are long-lived, up to 25 years, and exhibit high annual variability in survival to age-1, a.k.a., year class strength (Vinson et al., 2020). This year class strength variability leads to fluctuations in overall prey fish biomass, which can be readily observed across the time series (Figure 2). Declining biomass over the past twenty years has resulted from poor and variable survival of coregonines to age-1 and is thought to be related to climate change and the reduction in winter ice cover in particular which leads to warmer spring and summer water temperatures, but the specific mechanism reducing survival has not been identified.

Lake Michigan, Status: Fair, 10-year trend: Unchanging

Consistent with previous reports, the status of Lake Michigan prey fish is "Fair" (Table 1). While individual metric scores are identical to the State of the Great Lakes Report 2019 (diversity index = "Fair"; percent native = "Good"), we note that limitations on the number of transects sampled in 2020 (3 of 7) due to restrictions resulting from the COVID-19 pandemic may introduce some bias. The diversity index was 0.69 in 2020, the second lowest on record (Figure 1). If the diversity index excluded 2020, the current overall status (based only on 2018, 2019) for Lake Michigan prey fish would be "Good". Similarly, a historically high proportion of native species, 98%, was recorded in 2020. However, this estimate is in general agreement with a high proportion of native species in the bottom trawl from the previous five years (mean 2015-2019 = 81%). Further, a comparison of 10-year trends for each metric to the 2019 report seems to suggest improving conditions in Lake Michigan; the diversity index has shifted from "Deteriorating" to "Unchanging" and the proportion of native species is now "Improving", resulting in a shift in the overall 10-year trend status from "Deteriorating" to "Unchanging' (Table 2, Figure 1). Declining results may be in partial conflict with stated Lake Michigan fish community objectives to maintain prey fish biomass (Eshenroder et al., 1995). Prey fish biomass, based on the fall bottom trawl survey continues to remain at historical lows (Figure 2) and has been dominated by one native species, Bloater, since 2015. This represents a dramatic shift in prey fish composition, since from 2011 to 2014 Alewife were the dominant prey fish species in all but one year. Declines in Alewife biomass relative to other species since the mid-2010s may be the result of changes in catchability in the bottom trawl survey (Tingley et al. 2021). Beyond top-down controls of predators on Alewife populations and reductions in catchability of some species (e.g., Alewife and Bloater), reductions in nutrient loading and concentration, changes in climate, and the proliferation of dreissenid mussels may contribute to the long-term downward trend in estimated prey fish biomass in Lake Michigan (Dove and Chapra, 2015; Rowe et al., 2017; Warner and Lesht, 2015).

Lake Huron Status: Fair, 10-year trend: Unchanging

The Lake Huron prey fish community has undergone a dramatic transformation since the late 1970s. In the 1970s, non-native Alewife and Rainbow Smelt dominated the prey fish community (Riley and Adams 2010). Declines in those non-native species and lake-wide recovery of Bloater (a native coregonid) during the 1980s resulted in a steady increase in the proportion of the community comprised of native species (Figure 1). Prey fish biomass in Lake Huron began a steady decline in the mid-1990s and reached a historic low in 2008 (Figure 2). As prey fish biomass decreased, the proportion of native species in the community initially dropped, but then quickly increased with decline of Alewife populations in the early 2000s. Loss of Alewife from the prey fish community has been attributed to physical factors including cold winters, reduced mineral nutrients, proliferation of dreissenid mussels, and predation from naturally-reproduced Lake Trout and Pacific Salmon (Collingsworth et al., 2014; Dunlop and Riley, 2013; Kao et al., 2016). Prey fish biomass over the last decade had remained near historic lows, and the community is dominated by native Bloater, which during 2018-2020 accounted for over 85% of prey fish biomass sampled in bottom trawls. Accordingly, species diversity in 2019 and 2020 were the third and fourth lowest values observed in the time series, and the four lowest diversity index values (2014, 2015, 2019, 2020) all have occurred in the past 5 years (Figure 1).

Lake Erie Status: Fair, 10-year trend: Deteriorating

The Lake Erie prey fish community has experienced substantial changes in the past 10 years (Figure 1). The percent native metric has varied and the diversity index has declined to lower catches of Emerald Shiners, Yellow Perch, Spottail Shiner and Trout-perch (Forage Task Group, 2021). Dedines of these native species is most apparent in the central basin time series (Forage Task Group, 2021). Diversity is generally high, second only to Lake Superior among the five lakes, however this value has been declining slightly over the past 10 years (Figure 1, Table 2). Above-average reproductive success of Walleye has been observed over the past decade so that Lake Erie predator abundance is currently high and could partially explain some of the observed trends in Lake Erie prey fishes (Forage Task Group, 2021).

Lake Ontario, Status: Fair, 10-year trend: Improving

The Lake Ontario prey fish community dynamics and total biomass continue to change over time (Figures 1, Figure 2). Alewife have historically dominated the prey fish community (>90% by biomass), which caused for low values of diversity and the percent native metrics. The long term declines in Alewife biomass mirror the trends in spring phosphorus concentration (Dove and Chapra, 2015; Weidel et al., 2020), while variable and declining Alewife recruitment appears related to nutrient declines and climate variation (O'Gorman and Stewart, 1999). Both the diversity and percent native metrics are increasing (Table 1), as the proportional role of Alewife has declined and Deepwater Sculpin and Round Goby populations comprise more of the community (Weidel et al., 2020). Alewife biomass declines since 2013 are due to below average reproduction (cold springs and winters), lower nutrient concentrations, and increased abundance of wild-reproduced Chinook Salmon that prey heavily on Alewife (Bishop et al., 2020; Murry et al., 2010; Weidel et al., 2020). Since 2016, fishery managers have reduced stocking numbers in an attempt to maintain balance between predators and available prey (Great Lakes Fishery Commission Lake Ontario Committee, 2016). Lake Ontario management agencies continue to stock Bloater reared from Michigan gametes and broodstock in an effort to diversify the native pelagic prey fish community. Despite relatively low numbers of Bloater stocked, these fish have been documented in lake wide trawl surveys (Weidel et al., 2020).

Linkages

Prey fish communities are influenced by primary production and invertebrates as well as by higher trophic levels including stocked and naturally-reproduced piscivores and avian predators. Physical process driven by climate and land use such as water temperature, water clarity, geomorphology and spawning habitat quality also influence prey fish community composition and abundance by modifying their behavior, growth dynamics, and spawning success.

- Nutrient inputs and internally recycled nutrients are the primary driver of Great Lakes primary productivity and therefore are also a strong driver of prey fish biomass. Nutrient concentration is often the primary determinant of how many prey fish a lake supports while climate, habitat, species introductions and predators determine which species comprise the community. Climate also influences productivity at given nutrient levels by changing the volume or depth of the warm surface layers within lakes (Rowe et al., 2017; Warner and Lesht, 2015).
- Zooplankton are the primary food of pelagic prey fishes while benthic invertebrates are primary food of benthic or demersal prey fishes.
- *Diporeia* spp. are a genus of native, deepwater amphipods that were historically an important food source for Great Lakes prey fishes. As a detritivore, the species feeds on freshly settled organic material such as diatoms, linking those energy sources to prey fishes (Nalepa et al., 2006).
- Dreissenid mussels provide food for some prey fishes such as Round Goby and also act to increase water clarity which can shift prey fish behavior and intensify predator – prey interaction strength (O'Gorman et al., 2000).
- Lake Trout, Walleye (*Stizostedion vitreum*), and the other Great Lakes piscivores are a dominate driver of prey fish biomass and community composition.
- Changes in climate influence Great Lakes water temperature, ice coverage, and other aspects of hydrodynamics, all of which have strong influences on prey fishes.
- Water temperature drives prey fish energetics, behavior and habitat and food availability which in turn influence prey fish abundance and community dynamics.
- Ice cover influences the quality of egg incubation for fishes with overwintering egg stages and larval fish habitat.
- Water levels influence the availability of nearshore habitats which can affect prey fishes spawning and nursery habitat availability.

Assessing Data Quality

The data quality assessments below are based on the consensus expert opinions of the primary and contributing authors.

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable, and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin		X		
Data obtained from sources within the U.S. are comparable to those from Canada			X	
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report		X		
Data used in assessment are openly available and accessible	Yes*	Data can be found here: https://www.sciencebase.gov/catalog/item/57e185c8e4b0908250033a54 https://www.sciencebase.gov/catalog/item/58e2940fe4b09da67996a821		

***** USGS prey fish data from all lakes are available via links, however they may not always include the most recent year's data. Some Lake Erie prey fish data, from state and provincial agencies, is available upon request while open data access processes are under development.

Data Limitations

Evaluating this sub-indicator using bottom trawl data from across the basin helps to maintain consistency in our comparisons, but differences in trawl surveys may influence our results. Prey fish survey designs differ because the five lakes vary in size, bathymetry, habitat suitable for trawling, fish communities, survey history, and management information needs. For instance, across all five lakes bottom trawl surveys vary in: seasonal timing (April – October), annual effort (0.8 – 18 trawls per 1000 km^2 lake area), bottom trawl types, and reported statistics (biomass vs density, whole-lake vs basin-specific). Differences among surveys do not necessarily invalidate the results presented here but they are important to consider when interpreting results.

Time series of bottom trawl data have historically been used to infer relative changes of a prey fish over time in a lake, but they may not catch all species in equal proportion to their abundance in the environment. Pelagic species may be underestimated by bottom trawling relative to more demersal, or bottom oriented species (Yule et al., 2008). For example in Lake Superior, acoustic surveys estimated substantially greater biomass of pelagic Kiyi and Cisco, than bottom trawls estimates from the same year (Yule et al., 2013). Similarly, in Lake Ontario, acoustic and midwater trawl observations suggest Cisco can be a greater portion of the fish community than bottom trawls have indicated (Holden et al., 2017; Weidel et al., 2017). Alewife abundance relative to other fishes may also be biased by trawl survey type or seasonal timing. In Lake Michigan, fall-collected bottom trawl Alewife biomasses are usually lower than September acoustic estimates from the same year (Bunnell et al., 2017; Warner et al., 2017), and in Lake

Ontario Alewife biomass is much higher in spring than fall-based surveys (Weidel et al., 2017). Efforts to understand potential biases in surveys will be critical for improving cross-basin prey fish comparisons such as those presented in this report.

Acknowledgments

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Ohio Department of Natural Resources, Division of Wildlife
Pennsylvania Fish and Boat Commission
Michigan Department of Natural Resources
United States Fish and Wildlife Service

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Table 1. Status of Great Lakes prey fish communities in the current period (2018-2020) and previous reporting period (2015-2019) based on the percent native and the Shannon diversity index. For the percent native status, a categorization of good is for communities with 75% or more native species, while fair is above 25% and poor is below 25%. To attain as status of 'good' for the diversity index, the current period average indicator must be 75% or more of the maximum value observed in the time series. Similarly, the 'poor' status represents average values within the current period that are less than 25% of the maximum observed index value. If the metrics differ, they are averaged (good and poor yield fair) or if they are only more closely related the overall status is conservatively chosen as the lower metric (fair and poor yield poor). Metrics from 2020 were not available for Lake Ontario and Superior because precautions associated with COVID-19 pandemic prevented sampling therefore the current period represents 2018-2019.

Lake	Percent Native				Diversity Index				Overall
	Current	Previous	Max.	Status	Current	Previous	Max.	Status	Status
Superior	75	78	96	Good	1.7	1.9	2.2	Good	Good
Michigan	78	85	98	Good	1.2	1.2	1.7	Fair	Fair
Huron	77	70	90	Good	0.8	0.8	1.6	Fair	Fair
Erie	28	10	67	Fair	1.4	1.4	2.1	Fair	Fair
Ontario	11	5	15	Poor	0.7	0.4	0.8	Good	Fair

Table 2. Trend assessment for prey fish communities of the Great Lakes as determined by the community diversity index and proportion native species. A lake's metric was determined to be changing ('Improving' or 'Deteriorating') based on the slope (time) of a linear model with a p-value < 0.1. Asterisk denotes data from 2020 were not available for Lake Ontario and Superior because precautions associated with COVID-19 pandemic prevented sampling therefore the "10-year" trend represents data from 2011-2019.

Lake	Indicator	Whole Time Series		Last 10 Years*	
		Years	Trend	Years	Trend
Superior	Diversity Index	1978-2019	Improving	2011-2019	Unchanging
	Percent Native	1978-2019	Unchanging	2011-2019	Unchanging
Michigan	Diversity Index	1973-2020	Improving	2011-2020	Unchanging
	Percent Native	1973-2020	Unchanging	2011-2020	Improving
Huron	Diversity Index	1976-2020	Deteriorating	2011-2020	Unchanging
	Percent Native	1976-2020	Improving	2011-2020	Unchanging
Erie	Diversity Index	1990-2020	Unchanging	2011-2020	Deteriorating
	Percent Native	1990-2020	Unchanging	2011-2020	Unchanging
Ontario	Diversity Index	1978-2019	Unchanging	2011-2019	Improving
	Percent Native	1978-2019	Unchanging	2011-2019	Improving

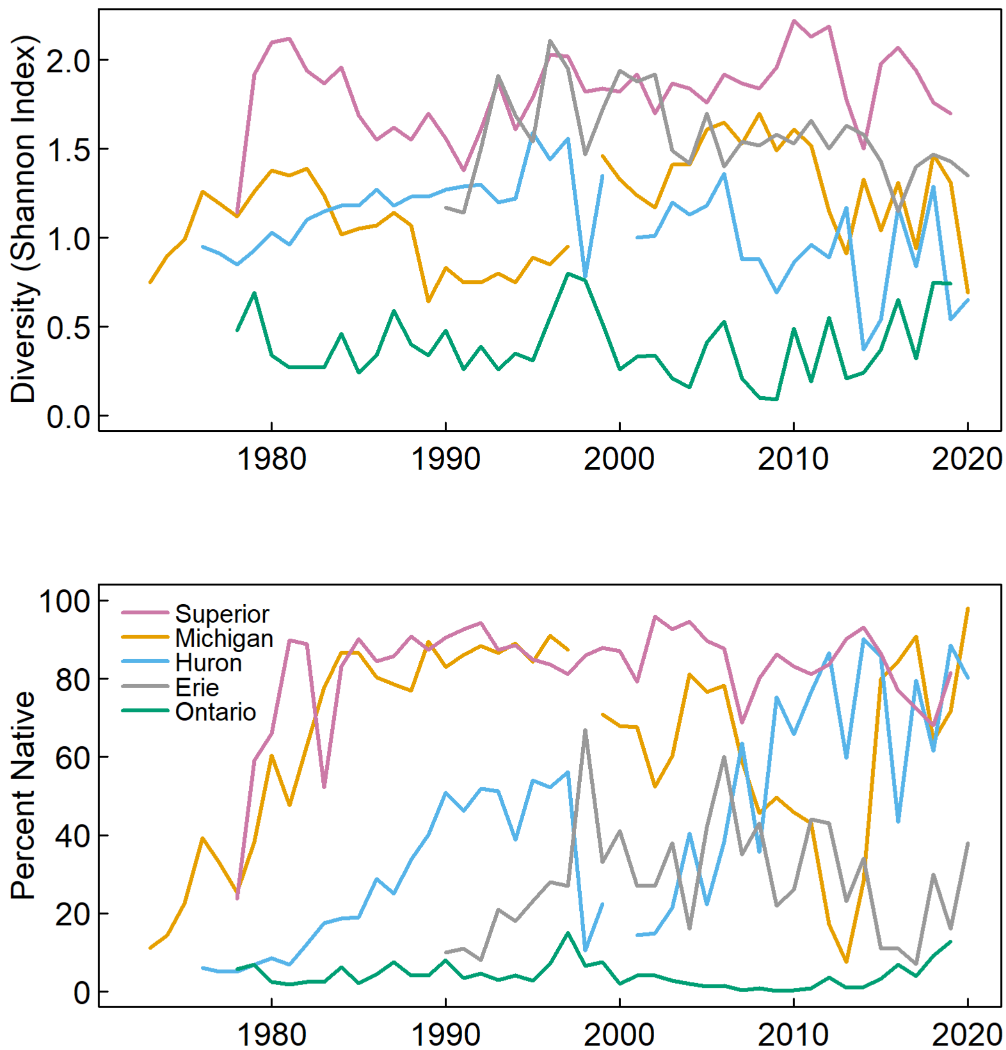

Figure 1. Shannon Diversity index values and proportion of native species of Great Lakes prey fish communities. Source: Data primarily derive from bottom trawl surveys conducted by U.S. federal and state and Canadian provincial agencies. Metrics from 2020 were not available for Lake Ontario and Superior because precautions associated with COVID-19 pandemic prevented sampling.

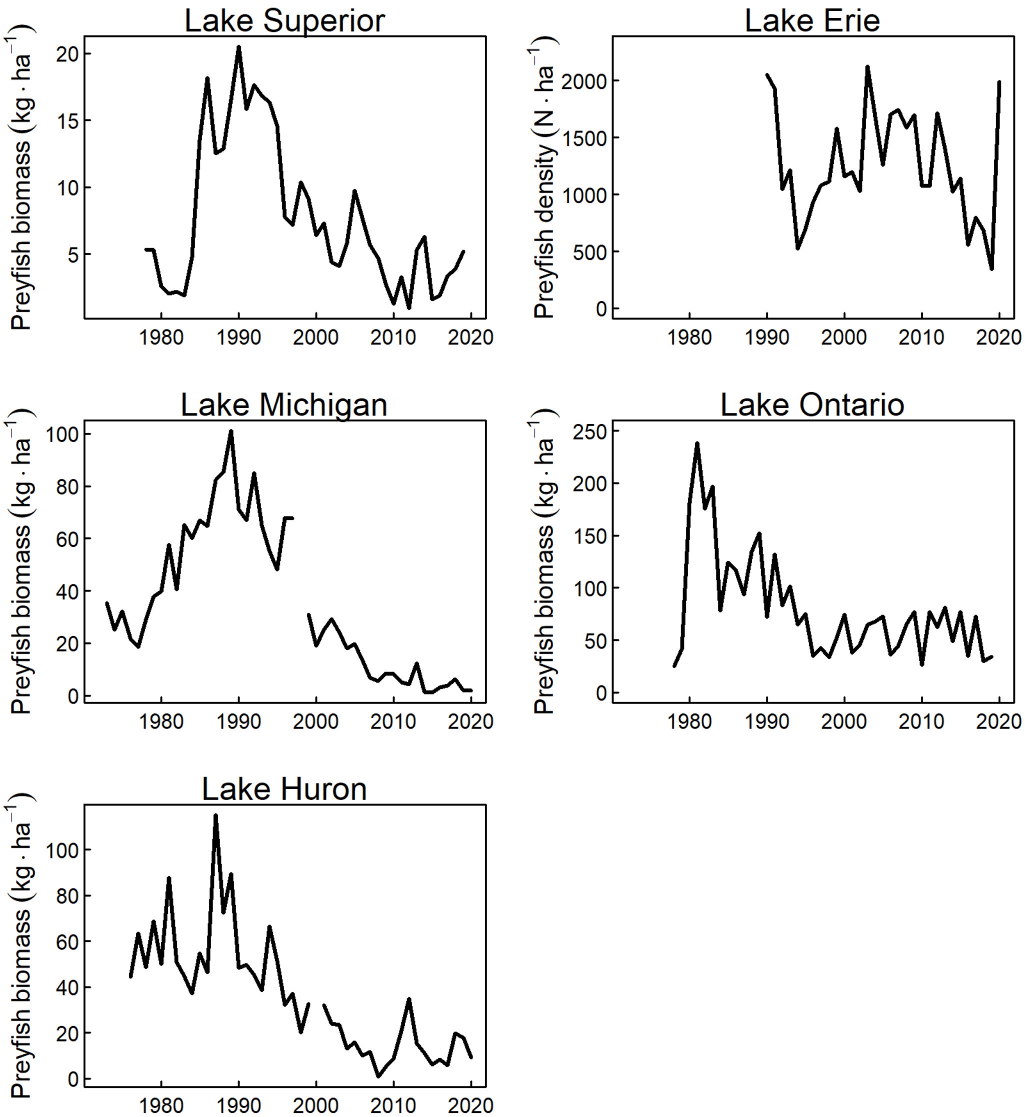

Figure 2. Prey fish biomass or density trends based on annual bottom trawl surveys. Note different scales. Source: Data primarily derive from bottom trawl surveys conducted by U.S. federal and state and Canadian provincial agencies. Metrics from 2020 were not available for Lake Ontario and Superior because precautions associated with COVID-19 pandemic prevented sampling.

Sub-Indicator: Lake Trout

Overall Assessment

Status: Fair

Trends:

10-Year Trend: Improving

Long-term Trend (1975-2020): Improving

Rationale: Sustained and ubiquitous natural reproduction is present only in lakes Superior and Huron, is minimal in lakes Michigan and Ontario, and absent in Lake Erie. Continued stocking, harvest regulation, and Sea Lamprey control are needed for continued restoration. High predator consumption and oligotrophication is reducing alewives to their lowest levels, especially in Lakes Michigan and Huron. The low number of alewives may be releasing Lake Trout from the hypothesized bottlenecks of thiamine deficiency and fry predation perceived to be caused by alewives, which may explain increases in wild recruitment. Lake trout, being opportunistic feeders, now consume round gobies, a high-density nearshore energy and nutrient subsidy, which has likely increased survival of both stocked and wild recruits. Reduced alewife densities are forcing managers to reduce lake trout stocking to preserve the residual forage base for other predators that support the sport fishery.

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Improving

Long-term Trend (1975-2020): Improving

Rationale: Since the 1970s, wild Lake Trout abundance has increased significantly in Lake Superior. Since 2010, abundance of wild lake trout has increased about 20%. Natural reproduction of both nearshore (lean) and offshore (siscowet) populations is widespread and supports nearly all populations across the system. Most stocking has been discontinued because rehabilitation targets have been achieved and wild fish comprise more than 90% of lake trout lakewide. In southeastern waters, stocking has ceased because of excessive commercial fishery exploitation. Sea Lamprey mortality continues to be the dominant mortality source but is managed below targets in most areas. In the most abundant populations, density-dependent declines in growth and recruitment have been observed, but overall lake-wide populations are healthy. Most agencies are committed to further restoration and conservation.

Lake Michigan

Status: Fair

10-Year Trend: Improving

Long-term Trend (1998-2020): Improving

Rationale: Wild fish continue to be observed in fisheries and fishery independent surveys but at low levels and mostly in the southern portion of the lake where mortality is low and adult stocks are more developed. Mortality from fishing and sea lamprey remains high in the northern area. Major concerns lie in the will of fisheries management agencies to stay the course of rehabilitation over concerns of angling public that favor non-native salmonines over lake trout in the face of a declining forage base, and the need to support fishery extraction

Lake Huron

Status: Fair

10-Year Trend: Improving

Long-term Trend (1977-2020): Improving

Rationale: Outside of the Lake Superior, lake trout recovery is the most pronounced in Lake Huron. Continuing increases in the abundance of wild lake trout, while the recruitment to fisheries and overall abundance of hatchery lake trout has continued to decline.

Lake Erie

Status: Fair

10-Year Trend: Improving

Long-term Trend (1992-2020): Improving

Rationale: Stocking levels over the past decade, combined with an emphasis on introducing strains less susceptible to Sea Lamprey attacks and mortality (Seneca and Lake Champlain strains), has increased adult stocks to levels near or above targets outlined in the rehabilitation plan. The adult stock is consistently comprised of over ten age groups. Lake trout are stocked in all basins in the lake. Sea Lamprey predation continues to be an issue, and there have been no significant levels of natural reproduction. Fishing mortality is low despite increased recreational angler effort in recent years. All agencies remain committed to further rehabilitation and conservation through the rehabilitation plan.

Lake Ontario

Status: Fair

10-Year Trend: Improving

Long-term Trend (1983- 2020): Improving

Rationale: Adult abundance is at values expected based on stocking and assessment history, conditions for natural reproduction are improving and the numbers of natural recruits have increased but remain low.

Status Assessment Definitions

Good: Relative abundance, harvest, or absolute abundance at or exceeding target levels. Wild fish making up 50% or more of population and increasing. Parental stock made up of many year classes beyond age of full cohort maturity.

Fair: Relative abundance, harvest, or absolute abundance below target levels. Wild fish making up between 15-49% of population. Parental stock made up of few year classes beyond age of full cohort maturity.

Poor: Relative abundance, harvest, or absolute abundance declining and below target levels. Wild fish making up less than 15% of the population. Parental stock made up of no or few age groups beyond age of full cohort maturity.

Undetermined: Data are not available or insufficient to assess conditions of the ecosystem components.

Trend Assessment Definitions

Improving: Relative or absolute abundance of wild Lake Trout, and wild and hatchery-reared adults are increasing, and the adult age composition is expanding.

Unchanging: Relative or absolute abundance of wild Lake Trout, and wild and hatchery-reared adults are stable, and the adult age composition is not expanding.

Deteriorating: Relative or absolute abundance of wild Lake Trout, and wild and hatchery-reared adults are decreasing, and the adult age composition is collasping.

Undetermined: Metrics do not indicate a clear overall trend or data are not available to report on a trend.

Adult = fish that are sexually mature, wild or stocked.

Endpoints and/or Targets

The goal of Lake Trout rehabilitation, as established by the Fish Community Objectives (FCOs) and Goals drafted by the Great Lakes Fishery Commission (GLFC), is self-sustaining, naturally reproducing populations that support fisheries. Appropriate quantitative measures of abundance, yield, or biomass should be established as reference values for self-sustaining populations of Lake Trout in the cold-water habitats of the Great Lakes. The sub-indicator target(s) for Lake Trout are based on the values provided in the Great Lakes Fishery Commission's rehabilitation plans for Lake Trout for each lake and/or for desired value(s) gained from analysis of the range and distribution of measures above compared to the ecosystem conditions.

Lake Superior

Achieve and maintain genetically diverse self-sustaining populations of Lake Trout that are similar to those found in the lake prior to 1940, with lean Lake Trout being the dominant form in nearshore waters, siscowet Lake Trout the dominant form in offshore waters, and humper Lake Trout a common form in eastern waters and around Isle Royale.

Lake Michigan

In targeted rehabilitation areas, re-establish genetically diverse populations of Lake Trout composed predominately of wild fish able to sustain fisheries. (Bronte et al. 2008).

Objective 1 – increase genetic diversity: increase the genetic diversity of Lake Trout by introducing morphotypes adapted to survive and reproduce in deep-water offshore habitats while continuing to stock a variety of shallow-water morphotypes.

Objective 2 – increase overall abundance: by 2014, increase Lake Trout population densities in targeted rehabilitation areas to levels observed in other Great Lakes locations where recruitment of wild fish to the adult population has occurred. To achieve this objective, catch-per-unit-effort (CPUE) in spring assessments should consistently exceed 25 Lake Trout/1,000 feet of graded mesh gillnet (2.5-6.0-inch mesh).

Objective 3 – increase adult abundance: by 2020, increase densities of spawning adult Lake Trout in targeted rehabilitation areas to that observed in other Great Lakes locations where recruitment of wild fish to the adult population has occurred. To achieve this objective, CPUE in fall assessments should consistently exceed 50 Lake Trout/1,000 feet of graded-mesh gillnet (4.5- 6.0-inch mesh).

Objective 4 – detect deposition of viable eggs: by 2021, detect a minimum density of 500 viable eggs/m^2 (eggs with thiamine concentrations >4 nmol/g) in targeted rehabilitation areas stocked prior to 2008. This milestone should be achieved by 2025 in areas newly stocked.

Objective 5 – build spawning populations: by 2024, spawning populations in targeted rehabilitation areas stocked prior to 2008 should be at least 25% female and contain 10 or more age groups older than age 7. These milestones should be achieved by 2032 in areas stocked after 2008.

Objective 6 – detect recruitment of wild fish: Consistent recruitment of wild Lake Trout in targeted rehabilitation areas should occur as follows: by 2022, detect age-1 fish in bottom trawls; by 2025, detect age-3 fish in spring graded-mesh-gillnet assessments; and, by 2028, detect sub-adults in gillnet assessments.

Objective 7 – achieve rehabilitation: By 2037, 75% or more of the Lake Trout in targeted rehabilitation areas should be age 10 or younger and of wild origin.

Interim targets were established by the Lake Michigan Committee in their strategic plan (Dexter et al. 2011) based on recommendations in Bronte et al. (2008). These are:

1. Increase the average catch-per-unit-effort (CPUE) to >25 lake trout/1000 feet of graded mesh gill net (2.5-6.0 inch) set overnight and lifted during spring stock assessments pursuant to the lakewide assessment in MM-3, WM-5, and at Julian's Reef by 2019. A reevaualtion is uderway currently.

2. Increase the abundance of adults to a minimum catch-per-effort of >50 fish/1000 ft of graded large-mesh (4.5-6.0 inch) gill net fished on spawning reefs in MM-3, WM-5, and at Julian's Reef by 2019.

3. Significant progress should be achieved towards attaining spawning populations that are at least 25% females and contain 10 or more age groups older than age-7 in first priority areas stocked prior to 2007. These milestones should be achieved by 2032 in areas stocked after 2008.

4. Detect a minimum density of 500 viable eggs/m^2 (eggs with thiamine concentrations of >4 nmol/g) in previously stocked first priority areas. This milestone should be achieved by 2025 in newly stocked areas.

Lake Huron

The goal of Lake Trout rehabilitation in Lake Huron is to restore self-sustaining populations that are capable of yielding 1.4 to 1.8 million kg by the year 2020 (Desjardine et al. 1995; Ebener, 1998; OMNR, 1996).

Lake Erie

Re-establish a genetically diverse, self-sustaining Lake Trout population in the eastern basin that provides ecological dominance within the cold-water community and produces a harvestable surplus. Interim goal – as demonstrations of feasibility, by 2030, establish a Lake Trout population in the eastern basin that produces yearling offspring at a measurable level.

Objective 1 – Increase overall Lake Trout Abundance: by 2022 achieve in all jurisdictions an average target CPE of 8 fish (all age groups combined) per 152.4 m of graded-mesh (38-152 mm) gillnet set overnight in summer. This is the minimum abundance recommended by Selgeby et al. (1995) for successful natural reproduction in the Great Lakes

Objective 2 – Maintain Adult Spawning-Stock Abundance: by 2024, achieve an abundance of adults (age 5+) that is equal to 25% (2 fish/lift) of the total CPUE defined in Objective 1. The adult population should be at least 25% female of a size >4,500 g and comprise at least 10 year-classes. This target CPUE is based on the maximum adult CPUE values in New York's waters of Lake Erie during the 1990s. CPUEs of spawning Lake Trout on reefs should be much higher, as adult fish are then aggregated.

Objective 3 – Maximize Reproductive Potential: by 2024, detect minimum egg densities of 25-500 eggs•m^{-2} in at least four different suitable spawning locations and of 1,000 eggs•m^{-2} in at least two high-quality locations.

Objective 4 – Demonstrate that Natural Recruitment is Possible: by 2030 and thereafter, achieve and maintain a consistent, measurable contribution of naturally produced age-1 Lake Trout (Markham et al. 2008).

Lake Ontario

Restore a self-sustaining population of Lake Trout in Lake Ontario .(Lantry et al. 2014)

Objective 1 – Increase abundance of stocked Lake Trout to a level allowing for significant natural reproduction defined as the catch of mature females > 4000g greater than 15 and 8 fish per kilometer of standard assessment gill net set for 24 hours in U.S and Canadian waters, respectively.

Objective 2 – Increase populations of wild Lake Trout across a range of groups (measureable increase in catches of wild juveniles and adults in assessment catches, with values exceeding those observed during 1994-2011)

Management Strategy 1: Stock 800,000 spring yearling equivalents per year in U.S. waters and 500,000 spring yearlings per year in Canadian waters.

Management Strategy 2: Minimize stocking and juvenile mortality by optimizing stage, size, and condition at stocking; stocking methods; stocking locations (measures: U.S.: adjusted catch rate of age-2 fish per 500,000 stocked > 200 fish per standard survey; Canada: adjusted catch rate of age-3 fish per standard gillnet per 500,000 stocked > 1.5 fish per standard gillnet set)

Management Strategy 3: Maintain high survival of older fish by controlling Sea Lamprey and fishing mortality measures: Yearly survival of adult fish >60%; maintain the Sea Lamprey wounding rate in fall gillnetting at <2 A1 wounds per 100 Lake Trout >432 mm total length; maintain annual harvest to <10,000 fish in U.S. waters and <5,000 fish in Canadian waters

Management Strategy 4: Emphasize stocking strains that show the best combination of low post-stocking, juvenile, and adult mortality.

Management Strategy 5: Emphasize stocking strains that are successfully producing a measureable level of wild recruits.

Management Strategy 6: Protect naturally produced fish

Management Strategy 7: Continue efforts to restore populations of native prey for Lake Trout.

The Lake Trout is a valued species that supports recreational and (where permitted) commercial fisheries, and harvest or yield reference values established for self-sustaining populations probably represent an attempt to fully utilize annual production; as a result, harvest or yield reference values for these populations can be taken as surrogates for production reference values.

Sub-Indicator Purpose

- To estimate the relative abundance of both stocked and wild (naturally reproduced) Lake Trout.
- To measure the success of rehabilitation through catch rates of wild Lake Trout
- To infer the control measures on fishing and Sea Lamprey predation by measuring the age structure and abundance of mature Lake Trout.
- To infer the basic structure of the cold-water predator community and the general health of the ecosystem

Ecosystem Objective

Maintain a balanced, stable, and productive ecosystem with self-sustaining Lake Trout populations as a major top predator in the cold-water regions of the Great Lakes.

Maintain self-sustaining populations by natural reproduction versus addition of stocked Lake Trout.

This sub-indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species."

Self-sustaining, naturally reproducing populations that support target yields to fisheries are the goal of the Lake Trout rehabilitation program. Target yields approximate historical levels of Lake Trout harvest or levels adjusted to accommodate stocked naturalized introduced predators such as Pacific salmon. Targets, most centered on desired harvest expectations, are set by Lake Committees of the Great Lakes Fishery Commission in Fish Community Objectives (Horns et al. 2003, Eshenroder et al. 1999, DesJardin et al. 1995, Ryan et al. 2003, Stewart et al. 1999), and are revised periodically. These targets are 1.8 million kg (4 million pounds) from Lake Superior, 1.1 million kg (2.5 million pounds) from Lake Michigan, 0.9 million kg (2.0 million pounds) from Lake Huron and 50 thousand kg (0.1 million pounds) from Lake Erie. Lake Ontario has no specific yield objective but has a population objective of 0.5 to 1.0 million adult fish that produce 100,000 yearling recruits annually through natural reproduction. The desired state will be for Lake Trout to serve as the primary top predator in Lake Superior and share this status with other native and established non-native predators in lakes Michigan, Huron, Erie and Ontario**.**

Measure

This sub-indicator will measure absolute abundance, relative abundance, harvest, and self-sustainability through natural reproduction of Lake Trout (wild and stocked) in cold-water habitats of the Great Lakes.
For each lake, a graph with Lake Trout metrics including measures of overall relative abundance and that of wild fishon the y-axis and year on the x-axis will be presented. Sampling method for each measure should be standardized at least within each lake, and if possible across lakes.

Ecological Condition

Trends in the relative abundance of stocked lean Lake Trout in Lakes Huron, Michigan, Erie and Ontario, and wild lean Lake Trout in Lakes Superior and Huron are displayed in Figure 1. Targets are set for most populations of lean Lake Trout as these are perceived to be biologically important to increase the probability of natural reproduction in Lakes Huron, Michigan, Erie and Ontario and to maintain wild populations in Lake Superior. Target values are measured and expressed as relative abundances of all or a portion of the population in multiagency gill net surveys that are standardized within each lake. These measures are superior to harvest objectives, which are harder to evaluate and represent desired states that cannot be easily tested for sustainability. Lake Trout abundance dramatically increased in all the Great Lakes after initiation of Sea Lamprey control, and after the stocking and harvest control of Lake Trout. Success in achieving population targets and ultimately self-sustaining, naturally reproducing populations has been mixed among the lakes. Stocking rates of Lake Trout in lakes Michigan and Ontario were reduced due to concern over predatory consumption of declining alewife populations; these reductions may hamper rehabilitation potential. In Lake Huron, the 2003 alewife population collapse led to major decline in lake trout growth and body condition (He and Bence 2007; He et al. 2008; Riley et al. 2008), but the decline did not continue. Lake Trout growth and body condition have stabilized at a healthy level as adult lake trout switched to feed on round goby (He et al 2015, 2016; Happel et al. 2018).

Since the last report, more wild recruitment is being seen in Lakes Michigan, Huron, and Ontario and Sea lamprey numbers have been reduced to target levels basin-wide. Sustained, modest recruitment has continued in southern Lake Michigan where populations are stable but below density targets. Parental stocks seem adequate in most areas with the exception of northern Lake Michigan due to high sea lamprey and fishing mortality In lakes Erie and Ontario, status and trends are the same as reported in 2019, and in Lake Ontario elevated levels of natural reproduction have continued during 2017-2020. Lake Superior saw a slight improvement in Lake Trout abundance since the last report. In southern Lake Huron, total abundance of Lake Trout declined (He 2019; Lenart et al. 2020), although management agencies did not reach a consensus on relative importance of changes in recruitment and mortality contributing to the apparent decline.

Desired states are populations that are self-sustaining through natural reproduction with minimal or no hatchery supplementation required, that support a sustainable harvest, and serve as a top predator. The resulting population size and sustainable yield compared to historical levels will likely be lower in most lakes since this apex trophic level is now shared by naturalized non-native predators that support a multi-billion dollar fishery.

Background

Historically Lake Trout were the keystone salmonine predator in most of the Great Lakes. Overfishing and predation by non-native Sea Lamprey, and to a limited extent other factors, destroyed nearshore lean populations and deep-water siscowet Lake Trout populations, but many populations survived in Lake Superior and a few lean Lake Trout populations survived in Lake Huron (Lawrie and Rahrer 1972, Berst and Spangler 1972, Wells and McLain 1972, Hartman 1972, Christie 1972). Rehabilitation efforts through stocking and controls on fisheries and Sea Lamprey have been ongoing since the early 1960s (Hansen et al. 1995, Eshenroder et al. 1995, Holey et al. 1995, Cornelius et al. 1995, Elrod et al. 1995). Lake trout require cold, well-oxygenated water, and nearshore and offshore rocky substrates for spawning in the fall. They are opportunistic omnivores and can live in variety of prey environments that range from plankton to fish. Although there has been some physical habitat loss in the Great Lakes, especially in Lake Ontario, habitat loss is not generally considered a significant impediment to restoration elsewhere in the Great Lakes.

Status of Lake Trout

Lake Superior

Wild lean Lake Trout populations recovered from collapse in the 1950s due to an aggressive rehabilitation program employing Sea Lamprey suppression, stocking of hatchery-reared fish, and fishery restrictions (Hansen et al. 1995; Bronte et al. 2003). Recovery began with the buildup of large populations of hatchery-reared Lake Trout, which were superseded by wild fish (Hansen et al. 1995; Wilberg et al. 2003). The transition to wild Lake Trout dominance began in the 1980s in Michigan waters and was subsequently followed in Wisconsin, then most recently in Minnesota. During the same period, central and western Ontario populations also recovered and have healthy wild Lake Trout populations supported by natural reproduction. In Michigan waters, abundance and recruitment of most Lake Trout populations are near historic high levels with some indications of density-dependent growth declines (Wilberg et al. 2003; Richards et al. 2004; Sitar et al. 2010). The latest progress in recovery was the cessation of most stocking in Minnesota waters. Although Lake Trout populations in most areas are healthy, there are some populations that are of concern due to excessive fishery exploitation. Little or no recovery has been observed in the Whitefish Bay region of southeastern Lake Superior due to excessive commercial fishery harvest hence Lake Trout stocking has been deferred in this area. In eastern Wisconsin and western Michigan waters, recovered Lake Trout stocks have been impacted by excessive fishery harvest. Recognizing this issue, restrictions on harvest have been implemented to reverse this trend.

Siscowet, a deep-water morphotype, is the most abundant form of Lake Trout in Lake Superior occupying deep water areas and have recovered from depressed levels in the 1940s (Bronte and Sitar 2008; Ebener et al. 2010). Recent harvest has been low, though emerging industrial interest in extracting omega-3 fatty acid from Siscowet may develop a demand. Sea Lamprey wounding rates on Siscowet are high, though the mortality inflicted may not be higher than that experienced by lean Lake Trout (Moody et al. 2010). Similar to lean Lake Trout, Siscowet are at high population levels and experiencing density-dependent declines in abundance.

Wild lean Lake Trout abundance has declined in some areas in recent years, but remains stable and on par with the prior State of the Great Lakes reporting period and has been trending upward since the 1970s. Fishing mortality has been controlled in most areas of Lake Superior through regulations. Despite continued Sea Lamprey management, wounding rates on Lake Trout in some areas have increased above target levels since 1995 (Pratt et al. 2016) but have been brought under control recently. In the near-term, further decline in Lake Trout abundance in some areas is expected due to density-dependent declines in growth and recruitment, but overall populations in Lake Superior are stable.

Lake Michigan

Stocking continues in all jurisdictions but has recently been reduced in nearshore waters of Wisconsin to address angler concerns about excessive alewife predation by Lake Trout. Lake Trout densities measured by spring assessment surveys remain below target in most areas and lakewide. More wild fish, especially in southern areas were recovered in assessment surveys in the last 10 years (Lake Trout Task Group 2018) than previously, which indicates that natural reproduction has increased but overall remains low. Illinois, Indiana, and southern Wisconsin waters with advanced age compositions and higher densities of Lake Trout show some evidence of sustained natural reproduction (Hanson et al. 2012; Lake Trout Task Group 2018). Northern Lake Michigan experiences high fishing mortality but Sea Lamprey mortality has been reduced in recent years. Sustained levels of excessive mortality have resulted in very low spawning stock biomass in the Northern Refuge. Increased stocking in the north has led to increases in overall densities but mature fish densities remain lower than desired. Lake Trout appear to be able to cope with the changing forage base with declining alewives by consuming more abundant round gobies. There is currently angler pressure to maintain or increase Chinook salmon stocking numbers, which has resulted in the stocking of fewer Lake Trout.

Recommendations to advance recovery include minimizing adult Lake Trout mortality from fishing and sea lamprey, focus hatchery production in refuge areas, restore a native forage base of coregonines and recast FCOs for desired population characteristics rather than harvest levels.

Lake Huron

Increases in wild recruitment and decreases in post-release survival of hatchery-reared fish in the last 10 years has led to the management decision to cease stocking in Canadian waters of the main basin, and in U.S. waters of the southern Lake Huron, and reduce the stocking by 50% in U.S. waters of northern Lake Huron. Adult biomass increased rapidly during the 1990s, peaked in 2007, and did not show a large lake-wide decline until 2014. Overall recruitment declined since the early 2000s, as the increases in wild recruitment did not fully offset declines in post-stocking survival of hatchery yearlings, even though total mortality has been controlled far below the 40% target (He et al. 2015; 2020).

In the main basin of Lake Huron, Sea Lamprey-induced mortality on Lake Trout was greatly reduced in the latter 1990s and early 2000s, and has been controlled at low levels since then. Lamprey wounding rates have declined in the last five years in both the North Channel and Georgian Bay, with recent rates approaching the target levels in both basins.

Lake-wide wild recruitment has continued since 2004, after the collapse of alewives (Riley et al. 2007; He et al. 2012, 2013; Johnson et al. 2015). Alewives are suspected to have adverse effects on reproduction via Thiamine Deficiency Syndrome and predation on Lake Trout eggs and fry (Fitzsimons et al. 2010). Recruitment of the wild 2010 year-class reached a new high level of abundance, although overall wild recruitment did not compensate for the declines in recruitment of hatchery-reared fish (He 2019; Lenart et al. 2020). Surveys indicate the relative abundance of wild adults has continued to increase since 2010 and now make up almost 50% of the assessment catch and fishery harvest. In Canadian waters of the main basin and North Channel, the proportion of wild Lake Trout now exceeds 75% of all fish sampled. The wild proportion of lake trout in Georgian Bay remains lower at 20 to 30%.

Lake Erie

Directed efforts to restore Lake Trout populations in Lake Erie began in 1982. Recruitment of stocked fish was good but their survival to adulthood was poor due to excessive Sea Lamprey predation. Adoption of the original Lake Trout rehabilitation plan in 1985 (Lake Trout Task Group 1985) brought higher annual stocking targets, Sea Lamprey control, and standardized assessment programs to monitor the population. Lake Trout responded quickly to the implementation of Sea Lamprey suppression and increased stocking, building a large population by 1990. However, these accomplishments were short-lived as stocking numbers were reduced in 1996 due to concerns about a shortage of forage fishes (Einhouse et al. 1999) while at the same time Sea Lamprey control was relaxed (Sullivan et al. 2003). Adult Lake Trout abundance was quickly reduced to low levels by 2000.

Overall Lake Trout abundance in Lake Erie has increased due to adoption of a revised rehabilitation plan (Markham et al. 2008) that increased stocking numbers back to their original level. Stocking has recently shifted to include all basins of the lake, including the western basin, and has come to focus on two strains that appear to be the most resistant to Sea Lamprey attacks and mortality (Seneca and Lake Champlain strains). Recruitment of stocked fish has declined since 2012 but adult (age 5+) abundance has increased and is at or near targets established in the rehabilitation plan. The adult population is consistently comprised of over ten age groups and abundance of age-10 and older Lake Trout is increasing. Sea Lamprey continue to be a major issue with adult abundance remaining well above targets despite consistent lampricide treatments, and this continues to suppress the adult Lake Trout population. Achievement of Lake Trout rehabilitation goals will continue to be hampered if Sea Lamprey abundance and wounding rates remain above target levels. Natural reproduction has yet to be found at significant detectable levels in Lake Erie. Fishing mortality remains low despite a recent increase in angler effort. An acoustic telemetry project was initiated in 2016 to determine locations and habitat used by Lake Trout in Lake Erie during spawning in order to gain a better understanding of the factors limiting natural reproduction. This research has successfully identified some areas and specific habitats being used by spawning Lake Trout and will guide the next phase of rehabilitation activities.

Lake Ontario

Prior to 1996, Lake Trout were monitored with a targeted binational lake-wide netting program that has continued in U.S. waters. Since 1996, Lake Trout targets in Canadian waters have been evaluated based on catches in the Kingston Basin only. In U.S. waters, the abundance of hatchery-reared adult Lake Trout was relatively high during 1986-1998, but declined by more than 30% in 1999 due to reduced stocking and declines in survival of stocked yearlings since the early 1990s (Elrod et al. 1995, Lantry et al. 2020). Adult abundance remained relatively stable during 1999-2004, but again declined by 54% in 2005 likely due to ongoing poor recruitment and increased mortality from Sea Lamprey predation. Enhanced control of Sea Lamprey and subsequent decreases in wounding on Lake Trout during 2008-2019 led to sharp recovery in adult Lake Trout numbers, which by 2010-2014 rose to levels similar to those observed during 1999-2004. Subsequent abundances trended downward during 2015-2017, in part due to stocking levels in U.S. waters that were at about 66% below normal in 2010 and absent in 2012. Abundance increased again in 2018 returning to levels similar to 2014 and has remained there through 2020 (Lantry et al. 2020). Within the Kingston Basin Lake Trout abundance has followed a similar trend of decline through the early 1990s and a recovery in stocks since 2005. Abundance within the Kingston Basin has been stable since 2013 albeit remains much lower than the 1990s and suggests a change in the ecosystem capacity to support Lake Trout in the post-dreissenid era.

Although the abundance of adults reached a peak in 1986, and evidence of natural reproduction was documented started in 1986, appearance of naturally recruited Lake Trout in assessment surveys occurred later, after the abundance of large adult females exceeded target levels in 1992 (Lantry et al. 2018). Despite widespread catches of small numbers of wild recruits nearly every year during 1993-2020, a failure to achieve self-sustaining stocks has been attributed to several factors: abundant population of Alewife in Lake Ontario, which may prey on Lake Trout fry and cause thiamine deficiency in adults and eggs; the absence of suitable alternative deepwater prey fishes; and the colonization of spawning reefs by invasive round gobies (Fitzsimons et al. 2003, Lantry et al. 2003, Schneider et al. 1997, Walsh et al. 2015) and dreisseinid mussels (Furgal et al. 2019). Recent predictions for restoration success have improved due to a number of changes: the recovery of the adult stock from the collapse experienced in 2005-2007; the reappearance of deepwater sculpin, which have steadily increased during 2002-2019 (Lantry et al. 2007, Weidel et al. 2019); the international efforts to re-establish cisco and bloater; and the addition of round gobies in Lake Trout diets (Dietrich et al. 2006; Rush et al. 2012). Signs of improving conditions for natural reproduction were substantiated during 2014-2017 where there was a mean wild contribution of 17% (11-31%) with assessment catches of wild age-1 and -2 Lake Trout rising sharply to a level nearly 10-fold greater than the 1994-2013 mean of 2% (2-8%) (Lantry et al. 2020). Small numbers of wild young-of-year Lake Trout were also observed in Kingston Basin trawl surveys during this period.

As encouraging as trends currently are, there is some cause for concern. The high levels of wild lake trout observed during 2014-2017 have not persisted through 2018-2020. The potential for Lake Trout diets to become more diverse with the inclusion of Round Goby and lake-wide declines in Alewife abundance (Weidel et al. 2019) has not fully played out as adult diets continue to be dominated by Alewife (Narwocki et al. 2020). Spawning habitat degradation appears to be ongoing as recent studies concentrating on historical habitat at Stony Island reef have demonstrated the potential for dreissenid mussel shell debris to completely infill interstitial habitat (Furgal et al. 2019). In addition, stocking reductions designed to decrease predator abundance and their impacts on declining Alewife stocks have reduced Lake Trout stocking targets from 1,250,000 (in each jurisdiction, Can. and U.S.) to 500,000 in 1993, to 400,000 for 2017 and to 320,000 for 2020. Some of the current and proposed research to address impediments includes using acoustic telemetry to describe habitat use and spawning aggregations, basin-scale and fine-scale spawning habitat characterization, investigating potential for spawning habitat remediation, assessment of egg and larval survival, and current and retrospective studies of Lake Trout reproductive health.

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Sea Lamprey – without sustained levels of Sea Lamprey control, Lake Trout stocking and building of parental stocks would not be possible
- Preyfish Populations and Communities – oligotrophication is making zooplanton communities more suitable for native coregonines and their restoration, than for non-native alewife that have sustained salmonine predators since the 1960s.
- *Diporeia* – Lake Trout are among the fish species that are energetically linked to *Diporeia*. Young Lake Trout feed on *Diporeia* directly, while juvenile Lake Trout feed on sculpin, and sculpin feed heavily on *Diporeia*.

This sub-indicator also links directly to the other sub-indicators in the Food Web category.
The rehabilitation of Lake Trout populations in the Great Lakes has linkages to Sea Lamprey, prey fish, and non-native species. Lake Trout stocking and building parental stocks would not be possible without sustained levels of Sea Lamprey control, as well as controls on fisheries. Non-indigenous alewives, while at lower levels now, still effect wild recruitment through predation on Lake Trout fry. Alewives also contain high levels of thiaminase that is thought to lower egg viability and fry survival in Lake Trout that consume mostly alewives. The lack of native pelagic and benthopelagic coregonines (whitefishes), lost to overfishing, habitat degradation and non-native invasions, is also hampering recovery as these lost species were conduits for offshore benthic and pelagic production (or the transfer of energy) to the nearshore environment and to Lake Trout as prey. Climate change is predicted to increase the warm thermal habitat of fishes in the water column with extended thermal stratification and reduced ice cover (Collingsworth et al. 2017). The effect on Lake Trout, which rely on the presence of cold thermal habitat, are bentho-pelagic in their foraging behavior and shown to be more thermally tolerant than previously realized (Sellers et al. 1998), is uncertain, however the lack of sustained ice cover during the egg incubation period will make more shallow shore oriented spawning habitat unsuitable for natural reproduction.

Traditional Ecological Knowledge (TEK), Citizen Science and other Bodies of Knowledge

None at this time but attempts to assess this are underway, especially in Ontario.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	No			

Data Limitations

The sub-indicator is of greatest value in assessing ecosystem health in the oligotrophic, open-water portions of Lake Superior, where native species at all trophic levels dominate the fish community. Because the sub-indicator includes only a single species, it may not reliably diagnose ecosystem health in other parts of a lake or in the other Great Lakes. Also, because Lake Trout abundance can be easily reduced by overfishing and Sea Lamprey predation, harvest restrictions designed to promote sustained use and enhanced Sea Lamprey control are required if the species is to be used as an indicator of ecosystem health. Annual interagency stock assessments in U.S. waters are fishery-independent, and measure changes in relative abundance, size and age structure, survival, and extent of natural reproduction but do not provide direct feedback to yield goals. Assessment activities in Canadian waters are not as spatially or temporally comprehensive or are based on sampling of commerical fishery catches, which may be less robust than assessments in U. S. waters. Yield goals alone may be inadequate to evaluate ecosystem health as they can vary dramatically dependent upon management induced quotas, user group preferences, community structure, and system productivity.

Additional Information

Self-sustainability of Lake Trout populations is measured in lakewide assessment programs carried out annually in each lake. The historical dominance of Lake Trout in oligotrophic waters in all of the Great Lakes provides a good basis for a basin-wide evaluation of ecosystem health. Maintaining or re-establishing historical levels of abundance, biomass, or production and re-establishing self-sustaining populations of Lake Trout throughout their native range in the basin will help ensure dominance in the ecosystem and the maintenance of a desirable aquatic community in oligotrophic, cold water habitats. The desired trend is increasing dominance of the indicator species to historical levels in cold water, oligotrophic habitats throughout the basin.

The oligotrophic fish community is associated with cold, clear, less productive, offshore waters. Oligotrophic communities, along with mesotrophic and eutrophic communities are found to varying degrees in all five of the Great Lakes with Lake Superior and more recently most parts of lakes Huron and Michigan, dominated by oligotrophic habitat.

The Lake Trout is the native top predator in deeper, open waters of all of the Great Lakes. Lake Trout is selected as an indicator because they represent a vital role in the original fish communities in the different habitats, they have value to the ecosystem and to fisheries, and they are the focus of fisheries management and restoration efforts. They have been subjected to the full range of other environmental effects as a result of human impacts on the Great Lakes including habitat loss, nutrient pollution, and toxic pollutants. While restoration efforts such as stocking can complicate interpretation of the status of native species, the successes of these species are indicative of progress toward the goals of the GLWQA.

Reporting frequency should be every five years, which is more appropriate give the rates of real change in population status of a long-lived species. Monitoring systems are in place, but in most lakes the measures do not directly relate to stated harvest objectives. Lake Trout population objectives need to be redefined as endpoints in units measured by the monitoring activities, as they are relevant to population characteristics required for restoration to proceed and should be incorporated into restoration guides and plans. The data time series presented are based on important population targets that can be measured with current assessment activities.

Objectives for Lakes Superior and Huron are harvest-based on historical yields or desired harvest states. They are divorced from “target” population states (CPUE, age composition) and fishery-independent survey data or population model output that better track rehabilitation progress toward desired targets. This lack of feedback makes evaluations difficult. Harvest levels alone are dictated by a variety of factors that are not necessarily tied to population abundance or trajectory.

Acknowledgments

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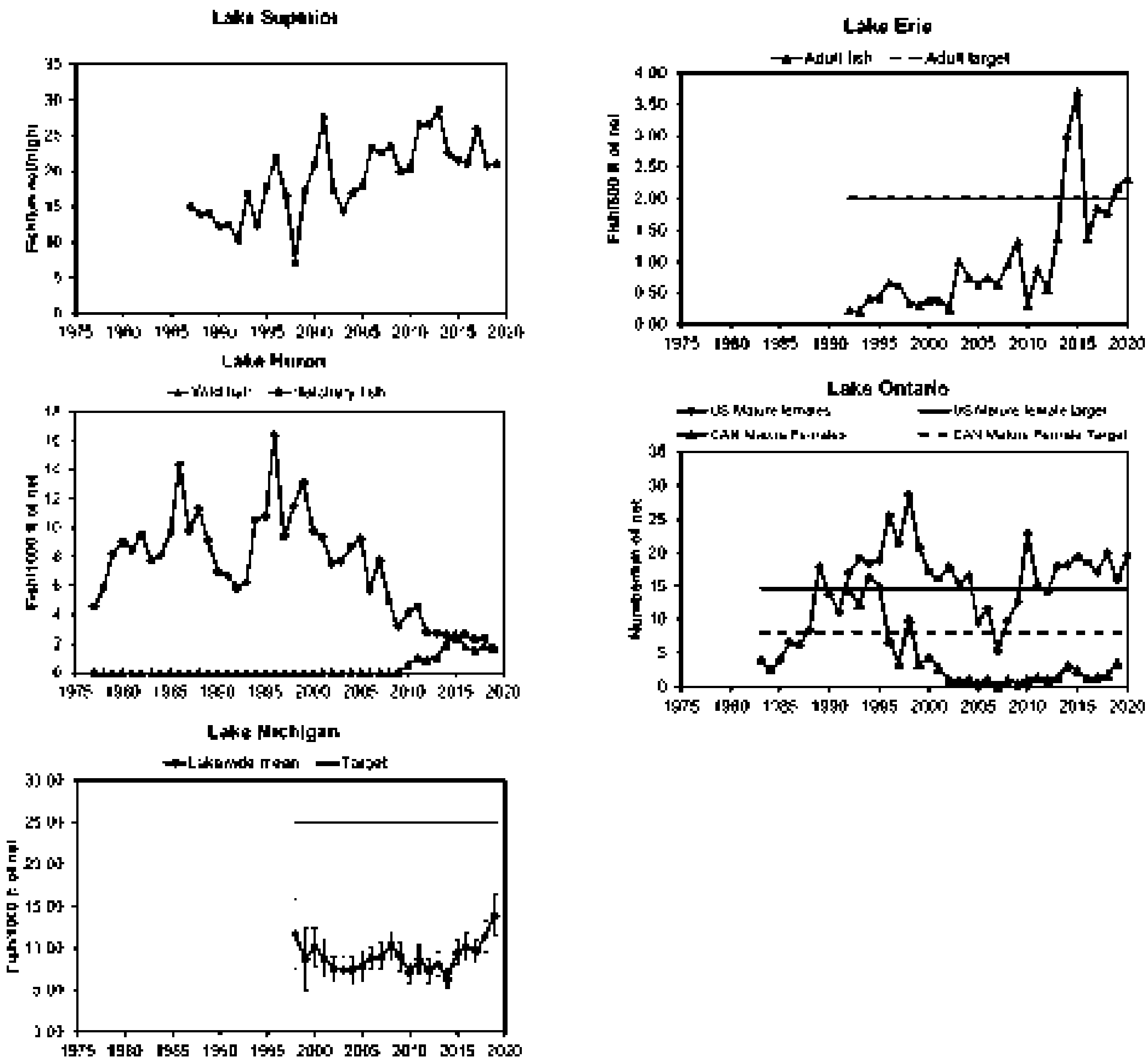
List of Figures

Figure 1. Relative abundance of stocked Lake Trout (and wild fish where provided) in the Great Lakes from 1975 - 2020. Due to Covid-19, many surveys were not conducted in 2020. The measurements and time frame reported vary from lake to lake, as shown on the vertical and horizontal scales, and comparisons among lakes may be misleading. Overall trends over time provide information on relative abundances for all or part of the population. No targets have been specified for Lakes Superior and Huron.

Source: Data sources are from biological assessments conducted cooperatively by state, federal, tribal and provincial agencies, and are largely contained in both non-peered reviewed and peer-reviewed reports to the Great Lakes Fishery Commission, Lake Committees, New York Department of Environmental Conservation, Ontario Ministry of Natural Resources, U.S. Fish and Wildlife Service and U.S. Geological Survey.

Last Updated
State of the Great Lakes 2022 Report



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Sub-Indicator: Walleye

Overall Assessment

Status: Good

Trends:

10-Year Trend: Unchanging

Long-term Trend (1975-2020): Unchanging

Rationale: Walleye populations across the Great Lakes are still quite variable, although their status was most often good. In several areas of the Great Lakes, including Lake Superior's Black Bay, the southern main basin of Lake Huron including Saginaw Bay, Lake Erie, Wisconsin's Green Bay, and Lake Ontario's Bay of Quinte, Walleye populations remain stable or are improving, supported by natural reproduction. However, in areas like the North Channel and Georgian Bay of Lake Huron, high exploitation rates and inconsistent recruitment has resulted in little to no improvement in the status of Walleye.

Lake-by-Lake Assessment

Lake Superior

Status: Fair

10-Year Trend: Improving

Long-term Trend (1980- 2020): Improving

Rationale: Walleye population assessment efforts have increased in parts of Lake Superior over the reporting period, although data gaps continue to exist. Index surveys in historically relevant areas such Black Bay, Nipigon Bay/River, ON and the St. Louis River MN/WI indicate that populations exhibit natural reproduction and have generally shown that Walleye populations have remained stable or improved over the time series. This includes increases in relative abundance (Black Bay), consistent recruitment and relatively low mortality (Black Bay, Nipigon Bay, and Chequamegon Bay). Efforts have been made throughout the lake to address management concerns for Walleye populations including stocking efforts in Wisconsin waters, limiting commercial and recreational harvest as well as improving nearshore and spawning habitat in tributaries.

Lake Michigan

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1985-2019): Unchanging (2020 data are not available for all jurisdictions due to staff shortages)

Rationale: Walleye populations are highly variable across Lake Michigan, as shown by harvest patterns for waters of Green Bay, which provide over 95% of the Lake Michigan's Walleye harvest. Estimates of angler harvest for the Wisconsin waters of Green Bay during 2010-2019 are higher than levels during the 1990s and 2000s, while angler harvests for the Michigan portion during 2010-2019 have been considerably lower than levels during the 1990s and 2000s. Walleye recruitment in southern Green Bay is entirely from natural reproduction, while that in northern Green Bay includes both wild and hatchery contributions. Walleye fisheries in the main basin of Lake Michigan are relatively small and with limited contributions of naturally-reproduced fish.

Lake Huron (including St. Marys River)

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1971- 2020): Unchanging

Rationale: The Michigan waters of Lake Huron remain dominated by the Saginaw Bay stock of Walleye which numbers 5.68 million age 2+ fish (in the spring of 2020). At least 37% of this stock typically migrates into the main basin of the lake during the open water months creating or substantially contributing to coastal fisheries. Some of these contributions likely also include some main basin Ontario fisheries. Predictions are for lower recruitment in 2021 and perhaps beyond because of high current stock density. The Saginaw Bay Walleye stock remains at recovery targets. A new recreational fishery management plan for Saginaw Bay is in development and new commercial legislation that may or may not affect Walleye is also under consideration. In Ontario waters, the dynamics of Walleye populations are basin distinct, with the southern main basin comprised primarily of transient fish originating from Saginaw Bay and western Lake Erie whereas the North Channel and Georgian Bay support local stocks associated with inflowing tributaries. Harvests and catch rates from the commercial fishery in the southern main basin have been stable over the past 10 years with variable catch rates and relatively high mortality rates. In the North Channel, commercial harvests remain relatively low but stable with slightly increasing but variable catch rates with most contributing stocks experiencing high mortality rates in both the commercial and recreational fisheries. In Georgian Bay Commercial harvest and catch rates remain depressed with no long-term change and with contributing stocks also experiencing high mortality rates in both the commercial and recreational fisheries. A recently completed technical report on the status of Walleye stocks in Ontario waters will contribute to the development of a Walleye Management Plan for Ontario waters of Lake Huron which is in progress.

Lake Erie

Status: Good

10-Year Trend: Improving

Long-term Trend (1975-2020): Unchanging

Rationale: Walleye abundance estimates in Lake Erie have fluctuated widely around the average of 53.1 million fish over the 40+ year time series, with no significant improving or deteriorating long-term trend (WTG 2021). In the last 10 years, the estimated Walleye abundance in Lake Erie has nearly tripled from 33.5 million fish in 2011 to 95.5 million fish in 2020. Three large year classes of Walleye drove this increase, as a strong 2015 year class was followed by record-setting levels of reproduction in 2018 and 2019. Walleye fisheries during this period were predominately supported by the 2015 cohort, with minor contributions from the 2010 and 2014 cohorts. The 2018 and 2019 cohorts should continue to increase the estimated abundance of Walleye in Lake Erie and support sustainable commercial and recreational fisheries for years to come.

Lake Ontario

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1992- 2019): Undetermined

Rationale: Following declines in juvenile and adult Walleye abundance in the 1990s, associated with reduced young-of-year (YOY) production in the mid-1990s, the Walleye population stabilized in the Bay of Quinte in Ontario and the New York waters of eastern Lake Ontario. Post-dreissenid mussel Walleye performance targets, identified in the Bay of Quinte Fisheries Management Plan (2010) are currently being met or exceeded. Recent YOY recruitment levels should keep the population at current or improved levels of abundance for the next several years.

Status Assessment Definitions

The Walleye is a highly-valued species that is usually heavily exploited by recreational fisheries and commercial fisheries (where permitted). Harvest or yield reference values established for self-sustaining populations are assumed to be proportional to annual production; as a result, harvest or yield reference values for these populations can be taken as surrogates for production reference values. Status reporting will be compared to the previous three-year reporting period, although it is recommended that a longer reporting period would allow for a more accurate trend for fishery harvest across time.

Good: Meets or exceeds attainment of harvest targets in all (most recent) reporting years

Fair: Met harvest target in one year and approached 50% of target levels in all years

Poor: Did not attain 50% of harvest target levels in all years

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components or harvest target not defined.

Trend Assessment Definitions

Improving: Walleye population estimates have been increasing; harvest rates have been increasing; and natural reproduction has consistently been better than average during 2011-2020. Survival trends are positive and disease outbreaks or occurrences are infrequent even though disease is often density-dependent so it is important to recognize that high population size is representative of a Good status but higher disease transmission is more likely. Growth rates are near historic averages with the understanding that density-dependent compensation occurs.

Unchanging: Populations of Walleye have not significantly increased or decreased; fisheries and catch rate trends remain mixed; and natural reproduction has been inconsistent on an annual basis during 2011-2020. Changes in growth rates, maturity, and fecundity have been occasionally noted and/or disease outbreaks or occurrences are present but not excessive.

Deteriorating: Populations of Walleye have decreased below target levels established for the lake or water body; reproduction, survival and/or growth has been significantly lower during 2007-2017; and there is evidence of a decline in fisheries and catch rate trends during 2011-2020. Disease or contaminant rates may have worsened in recent years.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

Appropriate quantitative measures of relative abundance, harvest, diversity, or biomass should be established as reference values for self-sustaining populations of Walleye in each lake. The sub-indicator target(s) for prey fish can be based on the values provided in the Great Lakes Fishery Commission (GLFC)'s Fish Community Goals and Objectives (FCGOs) and/or for desired value(s) gained from analysis of the range and distribution of measures above compared to the ecosystem conditions. Lakes Erie and Superior also have management and rehabilitation plans, respectively, concerning Walleye from which targets can also be obtained.

Lake Superior: Maintain, enhance and rehabilitate self-sustaining populations of Walleye and their habitat over their historical range (FCGO 2003).

The Lake Superior fish community will be managed to maintain, enhance, and rehabilitate habitat for, self-sustaining populations of Walleye in areas where the species historically maintained populations. Management strategies will be implemented to attempt to reach objectives specific to individual Walleye populations and habitats (Rehabilitation Plan, 2001). Rehabilitation targets for Walleye populations may include:

- 7 age-0 and age-1 Walleye/electrofishing hour/index station, and 8 ages classes are represented in assessment catches in the St. Marys River (Michigan) and Tahquamenon River
- The spawning population reaches 1,000 in the Pigeon River
- The spawning population reaches 7,000 in the Bad River
- The spawning population reaches 1,000 in the St. Marys River (Ontario)
- The spawning population reaches 500 in Goulais Bay
- The spawning population reaches 500-1,000 in Batchawana Bay
- The spawning population reaches 5,000 fish in Thunder Bay
- The population reaches either 22,000 adult fish or 41,000 fish over 356 mm in Nipigon Bay
- Catches in index gill nets reach 150 kg/km in Black Bay.

Lake Michigan: Expected annual yield: 0.1-0.2 million kg (100-200 metric tonnes) (FCGO 1995).

Harvest target sustainable levels of 200,000 – 400,000 pounds.

Lake Huron: Re-establish and/or maintain Walleye with populations capable of sustaining an annual harvest of 0.7 million kg (FCGO 1995)

Lake Erie: Manage the western, central and eastern basin ecosystems to provide sustainable harvests of valued fish species, including Walleye (FCGO 2020).

This is done within the construct of objectives stated or implied in the Lake Erie Committee's Walleye Management Plan (Kayle et al. 2015). These objectives are:

1. Maintain Walleye spawner biomass at levels greater than 20% of an unfished population (SSB20%) in the Total Allowable Catch (TAC) area
2. Maintain sport catch rates at or above 0.4 Walleye per angler hour
3. Maintain commercial harvest at or above 4 million pounds annually.

Additionally, the age and size structure of the fishery should be sufficient to:

- Promote migration of Walleye towards the eastern basin
- Provide diverse fishing opportunities to anglers

Lake Ontario: Fish-Community Objectives for Lake Ontario (Stewart et al., 2017) sanction action to "maintain, enhance and restore self-sustaining local populations of Walleye", as measured by "maintaining or increasing fisheries, populations, and recruitment of Walleye". Specific Walleye population performance targets are identified in the Bay of Quinte Fisheries Management Plan (2010). Performance targets for YOY recruitment, juvenile and adult abundance are currently being met or exceeded.

Sub-Indicator Purpose

The purpose of this indicator is to measure status and trends in Walleye population abundance and recruitment in various Great Lakes habitats; to infer the status of cool water predator communities; and to infer ecosystem health, particularly in moderately-productive (mesotrophic) areas of the Great Lakes and through their roles in the aquatic food web.

Ecosystem Objective

Protection, enhancement and restoration of historically important, mesotrophic habitats that support natural stocks of Walleye as the top fish predator. These habitats are necessary for a stable, balanced, and properly-functioning Great Lakes ecosystem.

This indicator best supports work towards General Objective #5 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the Waters of the Great Lakes should "support healthy and productive wetlands and other habitats to sustain resilient populations of native species."

Measure

Since Walleye function as apex predators in several of the Great Lakes ecosystems, population abundance and demographic changes in historical, cool water, mesotrophic habitats are important metrics for assessing Great Lakes health. As such, abundance and biological measures can be used to gauge ecosystem health and understand changes over time. Reviewing short-term (i.e., previous 10 years) and long-term (i.e., over the breadth of available data) changes to these measurements is a useful approach for describing the health of Great Lakes Walleye populations. Recognizing survey and financial constraints imposed on management agencies, metrics that describe regional (i.e., discrete stocks or spawning aggregations) status and trends of these Walleye populations may be more meaningful than lakewide indices in some circumstances. Likewise, while annual indices would be the most beneficial, information collected on an intermittent basis may be useful. Each agency makes decisions on the frequency of assessment based on a number of factors that take into account the level of the resources available (capacity) relative to the status of the stock and the social and economic pressure on the resource.

Abundance, spawner biomass, recruitment (i.e., natural reproduction), age/length at maturity, and fishery performance (effort, catch rate, yield) are useful metrics for describing Great Lakes ecosystem and fishery health. However, in the absence of absolute abundance and spawner biomass estimates for all lakes, relative measures from fishery-dependent (i.e., harvest) and fishery-independent (i.e., population assessments) surveys are suitable metrics for reporting on Walleye population health in the event population estimates are lacking.

Ecological Condition

The historical dominance of Walleye in mesotrophic habitats in the Great Lakes provides a good basis for a basin-wide evaluation of ecosystem health. Maintaining or re-establishing historical levels of relative abundance, biomass, or production of self-sustaining Walleye populations throughout their native range in the Great Lakes basin will help ensure the sustainability of this species in the ecosystem and the maintenance of a desirable and balanced aquatic community in cool water, mesotrophic habitats. Historical data can be used to develop status and trend information on Walleye populations. Commercial catch records for Walleye in the Great Lakes extend back to the late 1800s; recreational catch data and assessment fishing data supplement these commercial catch records in some areas in recent decades and sport fishing data are especially useful in areas where the commercial fishery for the species does not operate.

The "mesotrophic" cool-water fish community is associated with more productive waters in nearshore areas and embayments. Mesotrophic communities, along with oligotrophic and eutrophic communities are found to varying degrees in all five of the Great Lakes with more than half of Lake Erie represented by mesotrophic habitat.

The Walleye is the top predator in the cool nearshore and offshore waters of the Great Lakes and is selected as an indicator because they represent one of the original fish species in these different habitats. As a native apex predator, Walleye balance the fish community through top-down processes -adding value to the ecosystem and to fisheries, and they are the focus of fisheries management and restoration efforts. Being co-evolved with the rest of the fish community and the natural ecosystem of the Great Lakes, Walleye represent an important component to the natural biodiversity of the lakes. They have been subjected to the full slate of environmental effects resulting from human disruption of the Great Lakes ecosystem including habitat loss, nutrient pollution, exploitation, and water quality degradation, persistent toxic pollutants, and invasive species. While restoration efforts like stocking can complicate interpretation of their status, the successes of this species are indicative of progress toward the goals of the GLWQA. Walleye support large commercial and recreational fisheries throughout Lakes Erie and Huron; consequently, trends in harvest are useful for assessing ecosystem health. However, in Lakes Michigan, Ontario, and Superior, where Walleye are constrained to less abundant cool water habitats, harvest information may not be as reflective of ecosystem health as in Lakes Erie and Huron due to their limited spatial distribution. Rather, harvest trends may only reflect the ecosystem health of particular areas of suitable habitat in Lakes Michigan, Ontario or Superior because of the limited data available.

Lake Superior

Walleye populations in Lake Superior are limited to shallow embayments and lower gradient tributaries. As such, population assessments are conducted at appropriate scales. Thunder Bay (Kaministiquia River, ON) contains a small but healthy self-sustaining population, with evidence of consistent recruitment. In Black Bay, assessment work is showing an increase in the relative abundance of Walleye along with consistent recruitment and low mortality. Ongoing acoustic telemetry work suggests Walleye from Black Bay exhibit a range of movement patterns, including extended forays outside of Black Bay. In Nipigon Bay and Nipigon River, Walleye are low in abundance, but assessment work is showing signs of increasing density (high growth rates and low mortality). The St. Louis River (Minnesota/Wisconsin), supports one of the largest self-sustaining Walleye stocks in the Lake Superior basin, with an estimated population size of 50,000 adult Walleye (Olson et al, 2018). Populations in Chequamegon Bay, WI are comprised primarily of hatchery strain Walleye, while surveys indicate low levels of mortality. Due to limited assessment surveys, it is difficult to assess if population targets were met in other parts of the lake during the reporting period.

Lake Michigan

The trends for Walleye in Lake Michigan are largely driven by populations in the Wisconsin and Michigan waters of Green Bay, which constitute over 95% of the lake-wide harvest. A strong north-south productivity gradient in Green Bay has resulted in contrasting trends in Walleye abundance and harvests. For instance, estimates of angler harvest of Walleye from Wisconsin's portion of Green Bay during the 2010s were the highest since surveys began in 1986, while lowest harvests from Michigan waters of Green Bay occurred during the same decade. Walleye recruitment in southern Green Bay is entirely from natural reproduction, while that in northern Green Bay includes both wild and hatchery contributions. Young-of-year Walleye assessments in the Wisconsin waters of Green Bay show above average production in most years of the 2010's (Hogler and Surendonk 2019), but catch-at-age based estimates of age-1 abundance in Little Bay de Noc, the primary juvenile Walleye habitat in northern Green Bay, show relatively weak year classes during the 2010's despite significant stocking contributions (Zorn 2015; Zorn et al. *In prep*). Consistently low densities of zooplankton available to larval Walleye in Little Bay de Noc and Big Bay de Noc suggest prey availability as a potential limitation to strong year classes of Walleye (Zorn et al. 2020). Recent higher water level conditions in Lake Michigan may be positively impacting Walleye populations in Green Bay. Higher water levels are creating additional fish spawning and nursery habitat (i.e. emergent and submergent vegetation) along shorelines that may benefit inshore forage fish and potentially increase survival and recruitment of young-of-year Walleye. Walleye are very sparse in Wisconsin waters of Lake Michigan's main basin, with no natural reproduction and no stocking since 2007. Similarly, angler harvest estimates of Walleye from Michigan waters of the main basin have remained at relatively low levels compared to the 1990s, with harvest estimates of fewer than 1000 Walleye per year in 8 of 11 years during 2010-2020.

Lake Huron

Much of the trends of Walleye in Lake Huron are driven by the status and trends of the Saginaw Bay stock, the single largest population in Lake Huron and among the largest in the Great Lakes. The dynamics of the recovery of that stock is believed to be driven primarily by the collapse of the invasive Alewife (Fielder et al. 2007). Lake Huron's Alewife population declined dramatically in 2003. Now after 17 years of low abundance, it appears the forces behind its decline are on-going and resurgence seems increasingly unlikely. However, if Alewives were to become abundant again, it is expected that the Saginaw Bay stock of Walleye would decline and fall back into a depressed status and would likely require stocking to sustain. Changes in existing invasive species or new invasions remain the single largest threat to the status of the Saginaw Bay stock and probably all Walleye in Lake Huron. Walleye reproduction appears to remain dependent on tributary spawning but a reef restoration project (Coreyon Reef) in Saginaw Bay was completed in 2019 and is under evaluation to determine its efficacy in diversifying sources of recruitment. Two dam failures on the Tittabawassee River (which is believed to be the single largest source of Walleye reproduction for the population) occurred in spring 2020 and it is unclear how that may affect Walleye reproduction in years to come.

Spatially, Ontario's Walleye populations in Lake Huron are expansive and diverse. The mixed stock aggregation in the southern main basin is comprised of dynamic inputs from western Lake Erie and Saginaw Bay, and supports commercial harvests in the order of 100,000 kilograms annually. Stocks distributed throughout the North Channel and Georgian Bay are characterized by variable status and trends (UGLMU 2019) with only a few showing signs of improved recruitment and most found at depressed levels of abundance and experiencing high mortality rates. Walleye are harvested recreationally, commercially, and by Indigenous fisheries. The exact extent of the mixing of some of these reproductive sources is difficult to assess such that determining impacts of exploitation on localized stocks is difficult. Assessment netting provides trend information on localized reproductive stocks at specific spawning tributaries. Increased efforts to mark and tag spawning aggregations of Walleye (UGLMU 2019a) have provided insights into home ranges and stock mixing in nearshore and offshore waters but more work is required.

Overall the lake wide trend appears to be unchanging but for the North Channel and in particular Georgian Bay, populations remain depressed and unchanging. The overall status of the Lake Huron Walleye population and fisheries has to be characterized as Good in Michigan waters given the recovery of the Saginaw Bay stock but in Ontario waters, outside of the southern main basin, further improvement from current depressed abundances are desired. Generally, yield across all sources has not fully achieved the historic average or the Fish Community Objective of 0.7 million kilograms/year, but may be beginning to approach that for some parts of the lake.

Lake Erie

The annual Total Allowable Catch (TAC, or international fishery quota) set for the West and Central basins of Lake Erie has been continually increasing since 2013 (no TAC is set for the East Basin), resulting in increased Walleye harvest in both the sport and commercial fisheries. Since 2011, the commercial harvest has annually exceeded the 4 million pound management objective identified in the 2015-2019 Walleye Management Plan (Kayle et al. 2015), which was extended through 2024. In 2021, the spawning stock biomass was projected at 70.7 million kg, well above the 11.8 million kg limit reference point of 20% of the unfished spawner biomass (WTG 2021). Across Lake Erie, annual sport fishing effort remains below the long-term mean of nearly 5 million angler hours but has been trending upwards since 2011. Sport fishing harvest rates have generally increased with fluctuations since 2011, with a record of 0.81 fish/angler hour observed in both 2018 and 2019, nearly double the long-term mean of 0.45 fish/angler hour. Commercial gill net effort across the lake trended upwards from 2011-2016, then slightly decreased and settled at a level just below the long-term average of 18.6 thousand kilometers of gill net from 2018-2020. Lake-wide commercial harvest rates decreased from 2011-2015, when the trend in catch rates quickly rebounded to levels that approached the record 250.4 fish/km of gill net observed in 2006. Commercial harvest rates from 2017-2020 exceeded the long-term lake-wide average of 125.9 fish/km of gill net (WTG 2021).

Lake Erie Walleye fisheries remained largely dependent on the 2003 and 2007 cohorts, with some contributions from the 2010 and 2011 cohorts, until 2015. The mean age of Walleye harvested lake-wide by sport and commercial fisheries continued to rise through 2015 with the average age in the commercial harvest peaking at age-6 and the sport harvest peaking at age-7 in that year. Strong reproduction in 2015, 2018, and 2019 and subsequent recruitment has shifted the age structure of the harvest downward during recent years. In 2020, age-2 (2018 cohort), age-3 (2017 cohort), and age-5 (2015 cohort) Walleye comprised the greatest fractions of the commercial harvest and the harvest mean age dropped to age-4. The catch rate of the 2019 cohort in the Ohio-Ontario West Basin trawl index is the second highest on record (behind 2018) and this year class should further contribute to the fisheries in the coming years. Collectively, these recent strong year classes have led to increased abundance estimates, which in turn have led to increases in the lake's annual Total Allowable Catch (TAC) for commercial and recreational fisheries (WTG 2021) based upon the guidance established in the LEC walleye harvest strategy.

East basin Walleye stocks remain difficult to independently evaluate largely because migration of West and Central basin Walleye stocks masks trends of East Basin stocks when using traditional assessment approaches (Kayle et al. 2015). Research continues to examine the variability in the migration rates of fish among Lake Erie's basins to help address this issue. In the East Basin, fishing effort has more than doubled since 2011 to 674,000 angler hours in 2020, a time series high. Similarly, harvest rates in the sport fishery tripled between 2011 and 2019, dropping in 2020 but remaining more than 1.5 times higher than the long-term mean of 0.27 fish/angler hour. Commercial catch rates in the east end of the lake are at record or near-record levels and remain well above the long-term mean of 76.1 fish/km of gill net. Similar to West and Central basin fisheries, age-5 (2015 cohort) fish significantly contributed to the East Basin fishery in 2020 (WTG 2021). Like the other regions of the lake experiencing strong recruitment, the mean age of Walleye in the sport and commercial harvest has decreased from age-8 to ages 5 and 4, respectively.

Lake Ontario

The northeastern region provides most of Lake Ontario's warm and cool water fish habitats, including warm-water rivers, extensive embayments, and complex shorelines. These habitats provide the most potential for Walleye spawning, nursery and feeding habitat, and it is in northeastern Lake Ontario that the largest Walleye population, fishery, and assessment focus occurs. The Bay of Quinte provides the greatest quantity of Walleye spawning and nursery habitat in Lake Ontario. Walleye spawn in the four major rivers and along the shoreline of the Bay of Quinte. Walleye mark-recapture and acoustic telemetry studies indicate that young Walleye (less than 4 or 5 years of age) remain in the Bay of Quinte year-round while larger, older Walleye migrate to eastern Lake Ontario proper for the summer months. Annual bottom trawling (August) in the Bay of Quinte provides a long-term index of YOY abundance. Annual summer gill netting in both the Bay of Quinte and eastern Lake Ontario (Ontario and New York waters) provides excellent long-term abundance trends for juvenile and adult Walleye. Catches in eastern Lake Ontario are likely comprised of both migrating Bay of Quinte fish as well as Walleye produced in other nearby embayments, rivers and nearshore areas. The recreational fishery Walleye harvest averaged about 48,000 lb (21,772 kg) annually over the 2010-2019 decade (Data not available for all years). A small Walleye commercial fishery (less than 50,000 lb or 22,680 kg total annual quota) averaged about 26,000 lb (11,793 kg) harvest annually over the same time period.

Smaller, local Walleye populations exist in other areas of Lake Ontario. Some embayment areas support small but healthy and self-sustaining populations (e.g., Wellers Bay, West Lake) while other areas with degraded habitat require on-going rehabilitation efforts (e.g., Hamilton Harbour), including Walleye stocking. Stocking to restore Walleye populations in waters they formerly occupied serves to help diversify fish community trophic structure and to enhance recreational fishing.

Linkages

Walleye populations across the Great Lakes are affected by many other biotic (e.g., prey fish abundance) and abiotic (e.g., annual ice cover) variables.

Habitat and Species

- Aquatic Habitat Connectivity

A potential impediment to the continued health of Walleye populations in the Great Lakes is the connectivity between riverine spawning grounds and juvenile habitat. Often this phenomenon may be the result of human-induced alterations (e.g., dam construction) to the landscape.

- Phytoplankton, Zooplankton and Native Prey Fish Diversity

Phytoplankton, zooplankton, and prey fish collectively form the base of each Great Lakes' food web. Consequently, any changes in their population dynamics (e.g., seasonal abundance) and demographics (e.g., condition) may affect predatory fishes such as Walleye (including fish at larval, juvenile and adult stages), which are dependent on these as forage.

Nutrients and Algae

- Nutrients in Lakes
- Harmful Algal Blooms

Nutrient loading (e.g., phosphorous runoff) and their associated consequences, such as harmful algal blooms, can impact Walleye in some of the Great Lakes. For instance, harmful algal blooms in Lake Erie alter the foraging environment for Walleye and create areas of reduced dissolved oxygen that fish must navigate to find preferred habitat.

Invasive Species

- Impacts of Aquatic Invasive Species
- Dreissenid Mussels
- Sea Lamprey

Walleye across the Great Lakes are impacted by aquatic invasive species and the changes these species have brought about. For instance, establishment of Dreissenid mussels has caused shifts in nutrient cycling and water quality, which in turn alter phytoplankton, zooplankton, and prey fish abundance. Similarly, Sea Lamprey act as predators of large-bodied fishes such as Walleye, despite a successful control program administered by the Great Lakes Fishery Commission.

Watershed Impacts and Climate Trends

- Surface Water Temperature
- Ice Cover

A changing climate globally will likely increase water temperatures and reduce ice cover across the Great Lakes, which will alter the amount of preferred thermal habitat available to cool-water fishes such as Walleye and allow spawning to occur earlier in the year at sub-optimal conditions.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	LO, LE, LM	LH, LS		
Data obtained from sources within the U.S. are comparable to those from Canada	LE	LO, LS	LH	LM
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report*	LO	LH, LE, LS, LM		
Data used in assessment are openly available and accessible	Yes	See Information Sources section below for a list of who to contact to request data		

*Note that the COVID-19 pandemic affected data collection efforts across the Great Lakes basin in 2020, which may add additional uncertainty.

Data Limitations

Walleye abundance can be significantly reduced by overfishing; harvest restrictions designed to promote sustained use are required if the species is to be used as an indicator of ecosystem health. The Walleye sub-indicator cannot reliably diagnose causes of degraded ecosystem health. Target reference values have not been developed for all management objectives in the Great Lakes. The use of yield (especially in metric tonnes harvested) as a target reference value is problematic in that annual yield is not the most commonly assessed parameter across all Great Lakes. For instance, not all components of each lake's Walleye fisheries are assessed annually or on a regular basis (e.g., eastern basin recreational fisheries in Lake Ontario, recreational fisheries in Ontario waters of Lake Erie), which makes use of yield as an indicator potentially less useful compared to long-term fishery-independent surveys. On Lake Huron, Ontario lacks estimates of recreational Walleye harvest limiting full assessment and comparison of that source of extraction. Both Michigan and Ontario regularly use variable mesh gillnets as assessment gears but Ontario includes spring assessments which Michigan limits assessment to fall. Due to spatial limitations, Ontario cannot regularly assess all source populations which trace back to numerous river sources throughout Georgian Bay and North Channel where Michigan principally has only Saginaw Bay to monitor. Ontario and Michigan cooperate on a periodic assessment of the shared St. Marys River and their identical gear is used and pooled for a river-wide analysis. Despite the assessment differences, population level metrics derived from both Ontario and Michigan are comparable (e.g. mortality rates). In the Green Bay, fishery-independent assessment of Wisconsin Walleye populations employs spring and fall boat electrofishing, while Michigan conducts an experimental mesh gill net survey in fall. Regardless, both surveys provide comparable long-term indices of population abundance and structure.

Additional Information

Fishery yields (Figure 1) are appropriate indicators of Walleye health but only in a general sense. Yield was estimated for the recreational fisheries by multiplying the number of fish harvested by estimating the average size of fish harvested and extrapolating an estimated weight of harvested fish to the total number harvested. Fishery-dependent (i.e., effort and harvest) and fishery-independent (i.e., standard population surveys) assessments are lacking for some Walleye stocks in some years for all the studied areas. Moreover, measurement units are not standardized among fishery types (i.e., commercial fisheries are measured by mass while recreational fisheries are typically measured in numbers of fish), which means additional conversions are necessary which reduce accuracy. Also, "zero" values need to be differentiated from "missing" data in any figures. Therefore, trends in fishery yields across time (blocks of years) are probably better indicators than absolute values within any year, assuming that any introduced bias is relatively constant over time.

Many agencies have developed, or are developing, population estimates for many Great Lakes fishes. Walleye population estimates for selected areas (i.e., Lakes Erie and Huron) would probably be a better assessment of Walleye population health than harvest estimates, thus to the extent that it is possible, future efforts should focus on developing these capabilities.

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Information Sources

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Fishery harvest and population assessment data were obtained from the following sources:

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List of Figures

Figure 1. Walleye harvest, reported in metric tonnes, split into contributions from tribal, recreational and commercial fisheries in the five Great Lakes, 1975–2020. Fish community goals and objectives for Walleye are: Lake Superior, maintain, enhance, and rehabilitate self-sustaining populations over their historical range; Lake Michigan, maintain self-sustaining stocks, expected annual yield should be 100-200 metric tonnes; Lake Huron, reestablish and/or maintain walleye as the dominant cool-water predator over its traditional range with populations capable of sustaining a harvest of 700 metric tonnes; Lake Erie, maintain populations that support sustainable commercial and recreational fisheries; Lake Ontario, maintain, enhance, and restore self-sustaining local populations.

Source: Michigan Department of Natural Resources, Minnesota Department of Natural Resources, New York State Department of Environmental Conservation, Ontario Ministry of Northern Development, Mines, Natural Resources and Forestry, Ohio Department of Natural Resources, Pennsylvania Fish and Boat Commission, Wisconsin Department of Natural Resources

Last Updated

State of the Great Lakes 2022 Report

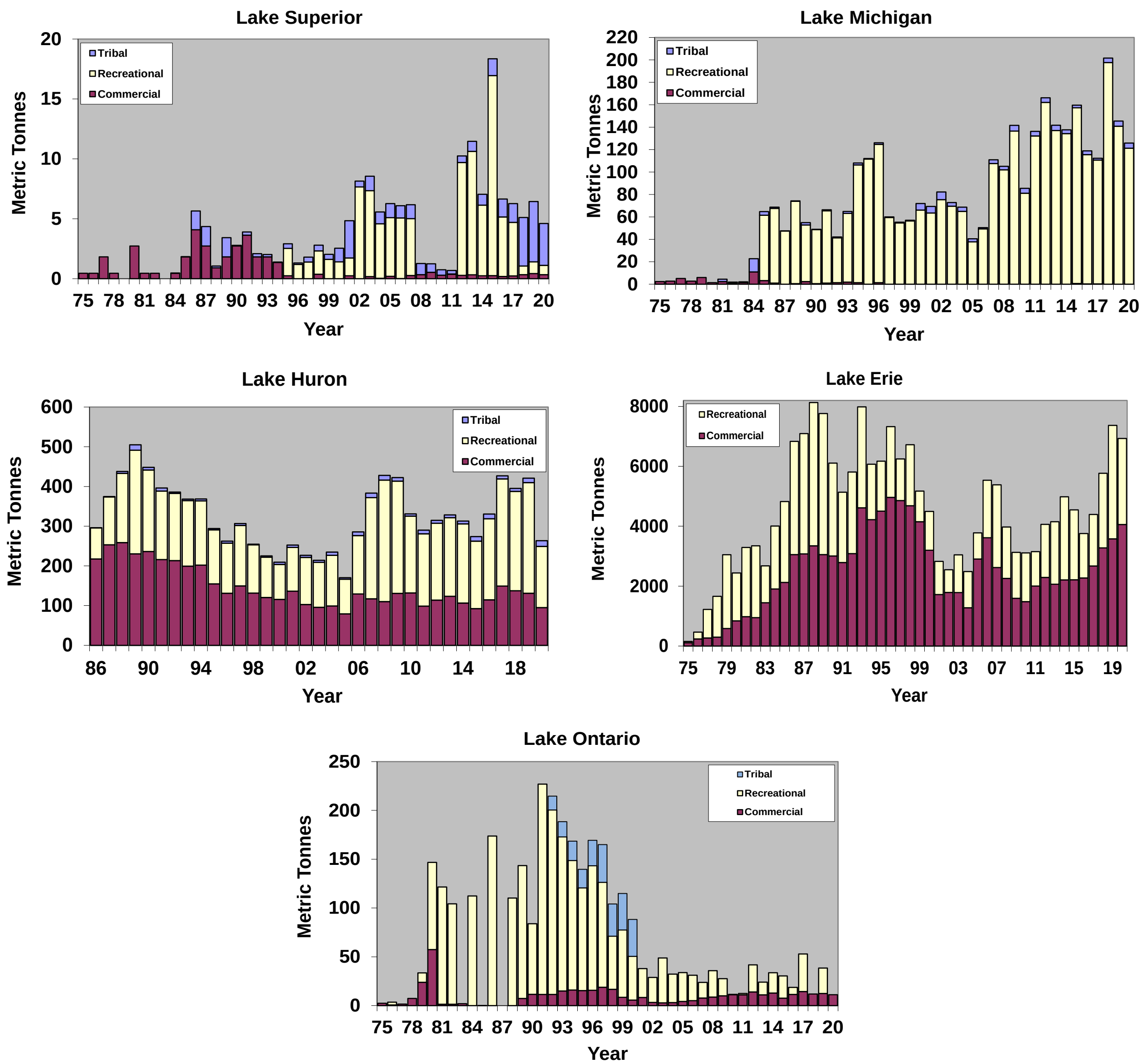

Figure 1. Walleye harvest, reported in metric tonnes, split into contributions from tribal, recreational and commercial fisheries in the five Great Lakes, 1975–2020. Fish community goals and objectives for Walleye are: Lake Superior, maintain, enhance, and rehabilitate self-sustaining populations over their historical range; Lake Michigan, maintain self-sustaining stocks, expected annual yield should be 100-200 metric tonnes; Lake Huron, reestablish and/or maintain walleye as the dominant cool-water predator over its traditional range with populations capable of sustaining a harvest of 700 metric tonnes; Lake Erie, maintain populations that support sustainable commercial and recreational fisheries; Lake Ontario, maintain, enhance, and restore self-sustaining local populations. Source: Michigan Department of Natural Resources, Minnesota Department of Natural Resources, New York State Department of Environmental Conservation, Ontario Ministry of Northern Development, Mines, Natural Resources and Forestry, Ohio Department of Natural Resources, Pennsylvania Fish and Boat Commission, Wisconsin Department of Natural Resources.

Sub-Indicator: Nutrients in Lakes

Overall Assessment

Status: Fair

Trends:

10-Year Trend: Unchanging
Long-term Trend (1970-2019): Deteriorating

Rationale: Phosphorus remains the growth-limiting nutrient in the Great Lakes. In the past, phosphorus concentrations were elevated throughout many of the lakes. Presently, the problems of excess phosphorus are confined primarily to some nearshore areas and parts of Lake Erie. In Lakes Michigan, Huron and Ontario, offshore total phosphorus concentrations are currently below objectives and may be too low, negatively impacting lake productivity (phytoplankton, zooplankton and fish production). In Lake Erie, objectives are frequently exceeded. Only in Lake Superior is the offshore objective being met and conditions acceptable. Nearshore, symptoms of nutrient enrichment persist in some locations.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1970-2019): Unchanging

Rationale: Objectives have consistently been met, and offshore total phosphorus concentrations are similar to historic values, indicating acceptable conditions

Lake Michigan

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (1983-2019): Deteriorating

Rationale: Offshore phosphorus concentrations are below objectives and have leveled off, with no signal of recovery. Low phosphorus concentrations may be negatively affecting lake productivity and nutrient conditions are therefore assessed as Fair and the long-term trend is assessed as deteriorating. In some nearshore areas, elevated phosphorus is observed and may be supporting nuisance algae growth. Data from Green Bay indicate eutrophic conditions.

Lake Huron

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (1970-2019): Deteriorating

Rationale: Offshore phosphorus concentrations have declined to values that are well below objectives and there is no indication of a recovery. Concentrations may be too low to support a healthy level of lake productivity based on the historic food web and nutrient conditions are therefore assessed as Fair and the long-term trend is assessed as deteriorating. In some nearshore areas, elevated nutrients may be contributing to nuisance algae growth.

Lake Erie

Status: Poor

10-Year Trend: Unchanging

Long-term Trend (1970-2019): Unchanging

Rationale: Total phosphorus objectives continue to be exceeded. Although high values are most frequently observed in the western basin, exceedances of objectives are observed in all three basins of Lake Erie in some years. Harmful algal blooms plague the western basin and parts of the central basin, and nuisance benthic algae have resurged in the eastern basin of Lake Erie. While there is some hint that the long-term trend in total phosphorus has improved, conditions in some years are similar to those observed in the 1970s. Due to highly variable conditions, statistical analysis indicates there is no trend over the long term or in the most recent 10 years.

Lake Ontario

Status: Fair

10-Year Trend: Unchanging

Long-term Trend (1970-2019): Deteriorating

Rationale: Offshore phosphorus concentrations have declined to values that may be approaching concentrations too low to support healthy offshore lake productivity based on the historic food web and there is no evidence of recovery. Due to the persistence of low phosphorus levels, nutrient conditions are assessed as Fair and the long-term trend is considered deteriorating. Certain nearshore areas are experiencing recurrent nuisance algae, possibly fueled by local phosphorus discharges or in-lake nutrient cycling

Status Assessment Definitions

Both nearshore and open water nutrient concentrations are considered in the assessment of nutrient status.

Good: The metrics show that the nutrient concentrations are meeting the ecosystem objectives and they are neither too high nor too low and should be considered in acceptable condition.

Fair: The metrics show that the nutrient concentrations are meeting the ecosystem objectives, but they are exhibiting minimally acceptable conditions and may be negatively impacting the food web.

Poor: *The metrics show that the nutrient concentrations are too high and are therefore not displaying minimally acceptable conditions and may be stimulating excessive and possibly toxic algal growth.*

Undetermined: *Data are not available or are insufficient to assess condition of the ecosystem components.*

Trend Assessment Definitions

Open water concentrations are of primary interest because they describe the main body of each lake and are sufficiently removed from nearshore variability, permitting temporal trends to be observed. Nearshore trends may be considered as the data allow.

Improving: *Nutrient concentrations are trending towards the ecosystem objectives, resulting in improving conditions.*

Unchanging: *No change in the nutrient concentrations.*

Deteriorating: *Nutrient concentrations are trending away from the ecosystem objectives, resulting in deteriorating conditions.*

Undetermined: Insufficient data exist for the determination of a trend.

Endpoints and/or Targets

The Annex 4 Interim Substance Objectives for spring total phosphorus (spring), and the resultant summer chlorophyll a (summer) concentration and the trophic state of the open waters of each lake are presented in Table 1. The offshore nutrient objectives represent expected conditions when tributary nutrient loadings targets are achieved. Both the Substance Objectives for Total Phosphorus Concentration in Open Waters and the Phosphorus Load Targets are Interim and are updated and revised as necessary.

There are currently no nitrogen endpoints, but the total nitrogen (N) to phosphorus (P) ratio is assessed. When nitrogen values are low relative to phosphorus concentrations, this can be associated with growth of harmful-blue green algae. Therefore, maintenance of spring conditions above the Redfield N:P ratio of 7.2 provides some protection against potentially harmful algal blooms.

Sub-Indicator Purpose

The purpose of this sub-indicator is:

- To assess nutrient concentrations in the Great Lakes;
- To assess progress in meeting GLWQA General Objective #6, Lake ecosystem Objectives and Substance Objectives for nutrient concentrations for the Waters of the Great Lakes;
- To infer progress in meeting nutrient loading targets and allocations;
- To support the evaluation of trophic status and food web dynamics in the Great Lakes; and
- To support assessment of the state of the nearshore waters for the nearshore framework

Ecosystem Objective

General Objective #6 of the 2012 Great Lakes Water Quality Protocol states that the Waters of the Great Lakes should "be free from nutrients that directly or indirectly enter the water as a result of human activity, in amounts that promote growth of algae and cyanobacteria that interfere with aquatic ecosystem health, or human use of the ecosystem."

Annex 4 of the 2012 GLWQA Protocol includes Lake Ecosystem Objectives to: maintain an oligotrophic state, relative algal biomass, and algal species consistent with healthy aquatic ecosystems, in the open waters of Lakes Superior, Michigan, Huron and Ontario (Lake Ecosystem Objective #5); maintain mesotrophic conditions in the open waters of the western and central basins of Lake Erie, and oligotrophic conditions in the eastern basin of Lake Erie (Lake Ecosystem Objective #6).

Interim Substance Objectives for Total Phosphorus concentrations in open waters are additionally established in Annex 4 for each of the Great Lakes. These interim objectives are shown in Table 1 and comprise objectives for both spring total phosphorus concentrations and summer chlorophyll a concentrations. The resultant nutrient (trophic) states corresponding to the objective concentrations are also displayed. Note that there are no objectives for nearshore nutrient concentrations.

There are no current ecosystem objectives for nitrogen. There is a requirement in Annex 4 to establish Substance Objectives for other nutrients, as required, to control the growth of nuisance and toxic algae to achieve Lake Ecosystem Objectives. As an interim measure, and as discussed in Dove and Chapra (2015), the Redfield ratio of 7.2 mgN/mgP is used as a benchmark to assess nitrogen levels; above this level, lakes would tend to be phosphorus limited, below this level, lakes would tend to be nitrogen limited, with nitrogen limitation favouring harmful cyanobacteria (a type of blue-green algae) in general. The goal would be to maintain ratios well above this level.

Measure

This sub-indicator reports on Total Phosphorus (TP) and other nutrients in the waters of the Great Lakes. The condition of the Great Lakes with respect to nutrients is determined using data collected by the federal agencies Environment and Climate Change Canada (ECCC) and the United States Environmental Protection Agency (USEPA) and data from additional nearshore locations, not well monitored by the federal agencies, are now included. The determination of the lakes' current status is based on total phosphorus in samples collected during recent spring (late March-May) or summer (generally July-August with some September data) seasons and considers data from all available stations. Data for the determination of long-term and recent trends are restricted to offshore stations using spring data (see Dove and Chapra, 2015) from the USEPA and ECCC only, to be able to provide a clearer signal of interannual changes.

Ecological Condition

Current Status

The most current Great Lakes total phosphorus concentrations are shown graphically in Figure 1. For the first time, TP data from nearshore monitoring programs have been included in this figure. The offshore data comprise surface samples for spring cruises from lakes Superior (2019), Ontario (2018), Huron (2017), Erie (2019) and from Georgian Bay (2017) from ECCC's Great Lakes Surveillance Program, as well as U.S. EPA data from spring of 2018 and 2019 in all of the Great Lakes (with the exception of Georgian Bay which they do not sample). Lake St Clair data are from a joint MECP/ECCC campaign in early June 2017. Green Bay data are from NewWater from late May 2017. MECP also contributed spring nearshore data from their Great Lakes Index program for Lake Superior and Ontario (2015), and Lakes Huron and Erie and Georgian Bay (2019). The USGS contributed Great Lakes data from their National Coastal Condition Assessment for the Great Lakes and their connecting channels for the time period 2014-2016, with most data collected in 2015.

The GLWQA interim substance objectives for total phosphorus (Table 1) are not exceeded in the offshore of the Great Lakes with the exception of Lake Erie. In Lake Erie, objectives have recently been exceeded at about 50% of stations in the central and eastern basins, and almost all stations in the western basin, as shown graphically in Figures 5, 6 and 7. High total phosphorus can fuel algal blooms, and blooms in the western basin of Lake Erie in particular have recently shown potentially harmful toxicity. In addition, western basin blooms may be playing a role in the development, severity and/or extent of hypoxic conditions in the central basin. For these reasons, conditions are considered Poor in Lake Erie.

In contrast to the elevated nutrients observed in Lake Erie, the data show that a majority of the offshore regions in Lakes Superior, Huron, Michigan and Ontario are below the objectives for total phosphorus and levels may indeed have fallen below those needed to sustain healthy offshore planktonic communities based on the historic food web. In some cases, phosphorus concentrations are less than one-half of objectives. In all cases except Lake Superior, concentrations are much lower in recent decades compared to historic values. For these reasons, Lake Superior status is considered Good and the remaining lakes are considered Fair. While offshore regions are nutrient deficient, elevated concentrations are observed in some nearshore regions of all the lakes. In particular, concentrations appear to be relatively high in Green Bay, the southern portion of Saginaw Bay, a portion of Lake St Clair, the western basin of Lake Erie, the southern shore of central Lake Erie and in nearshore regions along the northwest and southern shores of Lake Ontario (Figure 1). Local efforts may be required in these locations to reduce or abate sources and thereby reduce the pressure of elevated nutrients in the nearshore environment.

Temporal Trends

The long-term trends of total phosphorus are shown for the offshore of the Great Lakes in Figures 2 through 8. Statistical analysis of the data indicates long-term declining trends for all lakes. For Lake Superior, the decline is only observed for the very long (1970 – 2019) ECCC dataset and the rate of change is very slow. In lakes Michigan, Huron, Ontario and in Georgian Bay, the long-term trends show significant declines, with the most dramatic declines observed since the mid- to late-1990s. Due to the current low values and significant interannual variability, it is difficult to discern significant 10-year trends in any of the Great Lakes. Of all the Great Lakes, the greatest spatial and inter-annual variability is observed in Lake Erie. The central basin is shown in Figure 5; here, the trends can be interpreted to indicate high concentrations in the 1970s (in the range of 18 µgP/L) and lower concentrations in the 2000s (roughly 12 µgP/L). The variable nature of the data obscures any recent trends in all basins, resulting in an assessment of unchanging conditions for the most recent 10 year period for Lake Erie.

Trends of spring nitrogen levels are represented here as nitrate (NO_3). There are excellent, long-term records of nitrate available by both federal agencies, and nitrate represents more than 95% of the total oxidized nitrogen in the Great Lakes. Unlike phosphorus, concentrations of nitrate have increased over the long-term. Concentrations of nitrate are lowest in Lake Erie, the most productive of the lakes, where it is taken up by algae, phytoplankton and other consumers. There is some evidence that the growth of certain algal species may be fueled by nitrogen, but in general high nitrate is thought to be more protective against blue-green algae blooms, because these algae have a competitive advantage in their ability to use atmospheric nitrogen when nitrogen is low in the water. Because nitrate has increased over time and phosphorus has declined, it is therefore phosphorus, not nitrogen, which is increasingly limited in the Great Lakes (Dove and Chapra, 2015). Nutrient management is therefore most aptly targeted at phosphorus controls. Currently, all of the lakes are phosphorus limited, with the most extreme limitation occurring in the upper Great Lakes. The ecosystem objective to maintain ratios above the Redfield ratio of 7.2 is currently being met in all of the lakes, with Lake Erie showing greatest risk (Figure 9).

Inferred Nutrient Loadings

The existing GLWQA nutrient objectives for the Great Lakes offshore regions represent expected conditions when tributary nutrient loadings targets are achieved. Recent phosphorus loads are not available for the Great Lakes, with the exception of Lake Erie. The lack of sufficient and appropriate information upon which to directly compute loads remains a major shortcoming that needs to be addressed for much of the Great Lakes basin. Moreover, there is increasing evidence of algal imbalances in the lakes; that is, eutrophic (nutrient-rich) nearshore conditions may be persisting (or resurging) despite low offshore nutrient concentrations. In this way, the existing objectives may not be sufficient to protect all areas of the lakes. Insufficient monitoring of loadings, especially in the upper Great Lakes, combined with an incomplete understanding of changing in-lake nutrient dynamics, makes the previously-held models upon which nutrient budgets are understood insufficient for nutrient management currently and into the future.

Both the Substance Objectives for Total Phosphorus Concentration in Open Waters and the Phosphorus Load Targets are updated and revised, as necessary for the Great Lakes. Loadings targets have recently been adopted for Lake Erie; these call for a 40% reduction in annual total phosphorus loads to the western and central basins of Lake Erie and a 40% reduction in spring total and soluble reactive phosphorus loads from selected priority tributaries. These targets are intended to reduce loadings and thereby alleviate the issue of excessive phosphorus in Lake Erie. In the other Great Lakes, however, offshore phosphorus may be too far below objectives while certain nearshore areas may still receive excessive phosphorus loadings and suffer from the harmful effects of nutrient enrichment and excessive algal growth. This "feast and famine" dichotomy will require a different approach for nutrient management and environmental agencies are meanwhile faced with the challenge of effectively managing watershed nutrient sources.

Lake trophic status

A lake's trophic state describes its nutritional or growth status. Ranges of phosphorus, together with the response variables of chlorophyll a (an indicator of the amount of algae and phytoplankton in a sample) and Secchi disk depth (an indicator of water clarity) are used in combination to determine the trophic status. The objectives vary between each of the Great Lakes and for Lake Erie the objectives vary by basin. Collectively, the information shows that the open portions of Lakes Superior, Michigan and Huron are in the ultraoligotrophic range (i.e., very low in nutrients and below the objective of oligotrophy), Lake Ontario is in the oligotrophic range (i.e., nutrient poor and below the objective) and Lake Erie ranges from eutrophic in the west (nutrient rich and exceeding the objective) to mesotrophic in the central basin (exceeding the objective) and oligotrophic in the east (at or below objective). This indicates that the offshore regions of the Great Lakes are nutrient deficient with the exception of Lake Erie which suffers from elevated nutrient conditions.

Linkages

- Benthos – nutrient concentrations impact benthic community abundance and composition
- *Cladophora* – high nutrients in the nearshore favour the proliferation of nuisance benthic algae
- Dreissenid Mussels – Dreissenids influence the cycling of phosphorus, which may alter in-lake concentrations, their relationships with loads and may enhance the growth of *Cladophora*
- Harmful Algal Blooms –nutrient concentrations impact the development, timing and severity of harmful algal blooms

- Phytoplankton (open water) – nutrient concentrations impact phytoplankton community abundance and composition
- Water Quality in Tributaries – tributary nutrient concentrations impact nutrient concentrations in Great Lakes Waters
- Zooplankton – nutrient concentrations impact zooplankton community abundance and composition via the food web
- Wastewater treatment can reduce the nutrient loading to the lakes.
- Climate change – increased lake temperatures may be enhancing plankton and algal growth rates. Where abundant nutrients exist (e.g., nearshore regions), further loadings may exacerbate impairments under higher water temperatures.
- Precipitation Amounts – an increase in extreme events, could result in large inputs of nutrient-rich waters, potentially fueling nearshore algal blooms.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: ECCC; https://open.canada.ca/data/en/dataset/cfdafa0c-a644-47cc-ad54-460304facf2e USEPA; https://cdx.epa.gov/		

Data Limitations

The federal long-term monitoring records of total phosphorus that are shown here comprise high quality and spatially comprehensive datasets. The data are quality-assured and laboratories undergo rigorous testing and comparisons. Interagency field comparisons have been conducted and additional studies are ongoing. Additional nearshore data from Provincial, State and other agencies are now included; these are also high quality datasets and some but not all may have been tested for inter-comparability.

Nitrates are used in place of total nitrogen because there is a good long-term record for nitrates measured by both US and Canadian federal agencies and because nitrate alone can be used as an estimate of bioavailable nitrogen in the Great Lakes (Dove and Chapra, 2015). However, the USEPA has been measuring total nitrogen since 2014; Canada has total nitrogen data available through calculation since 1983 for Lake Ontario, 1989 in Lakes Superior and Huron and 1994 in Lake Erie.

Additional Information

Continued water quality monitoring at a basin-wide scale in Great Lakes waters is required for informed nutrient management, to track progress and update status and trend information. Ongoing efforts are required to maintain and improve the monitoring programs upon which this sub-indicator is successfully based.

The management of excessive nutrients is important in many nearshore areas of the Great Lakes, in particular in Lake Erie. Tracking the lake response to nutrient loading changes will help to assess if nutrient loading targets are resulting in improved conditions. Additional monitoring and/or improved monitoring coordination and data integration is warranted to improve our ability to detect meaningful change in such a dynamic system. For the other Great Lakes, loading targets revisions are being or will be considered by GLWQA Annex 4. Improved understanding of nutrient dynamics in the Great Lakes is necessary in order to improve the information basis upon which critical nutrient management decisions are made.

Integrating nutrient loading information to this sub-indicator will be a challenge without concerted efforts to improve load monitoring in the basin. Important work to coordinate, collect and manage such information has been initiated for Lake Erie but is largely lacking in all of the other Great Lakes on a lake-wide scale.

Acknowledgments

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Matthew Pawlowski, Great Lakes National Program Office, U.S. Environmental Protection Agency.

Information Sources

Dove, A. and Chapra, S.C. 2015. Long-term trends of nutrients and trophic response variables for the Great Lakes. Limnology and Oceanography. 60(2): 696-721. http://dx.doi.org/10.1002/lno.10055.

U.S. Environmental Protection Agency. 2021. National Coastal Condition Assessment: A Collaborative Survey of the Nation's Estuarine and Great Lakes Nearshore Waters. EPA 841-R-21-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC https://www.epa.gov/national-aquatic-resource-surveys/national-coastal-condition-assessment-2015-report

List of Tables

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show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed.

Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Figure 7. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in the western basin of Lake Erie (µg/L). The interim GLWQA TP objective of 15 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed.

Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Figure 8. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in Lake Ontario (µg/L). The interim GLWQA TP objective of 10 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed.

Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Figure 9. Trends of open lake, spring ratios of median NO3:TP for the Great Lakes. The Redfield ratio of 7.2 mgN/mgP is superimposed as an estimate of the level above which lakes would tend to be phosphorus limited. Phosphorus limitation is beneficial because nitrogen limitation would favor potentially toxic blue-green algae (cyanobacteria). After Dove and Chapra (2015).

Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Last Updated
State of the Great Lakes 2022 Report

Table 1. Interim Substance Objectives for Spring Total Phosphorus and Summer Chlorophyll a Concentrations, with resultant Trophic State

Basin	Total Phosphorus (µgP/L)	Chlorophyll a (µgChla/L)	Trophic state
Lake Superior	5	1.3	Oligotrophic
Lake Michigan	7	1.8	Oligotrophic
Lake Huron	5	1.3	Oligotrophic
Western Lake Erie	15	3.6	Mesotrophic
Central Lake Erie	10	2.6	Oligomesotrophic
Eastern Lake Erie	10	2.6	Oligomesotrophic
Lake Ontario	10	2.6	Oligomesotrophic

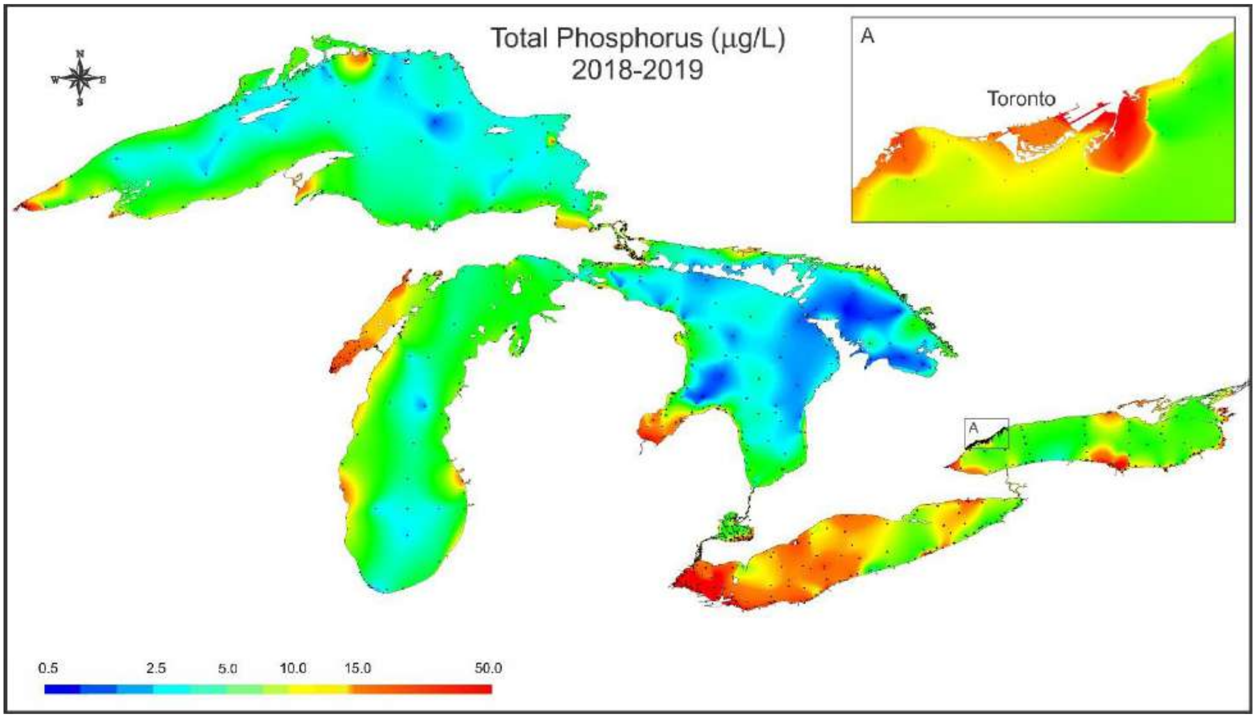

Figure 1. Spatial distribution of total phosphorus (µg/L) in the Great Lakes**.** Sampling stations are shown as black dots. Data sources: Environment and Climate Change Canada, United States Environmental Protection Agency, Ontario Ministry of Environment, Conservation and Parks, NewWater (Green Bay Metropolitan Sewerage District), United States Environmental Protection Agency. Offshore data are from 2018 and 2019; nearshore data span additional years, as described in Current Status.

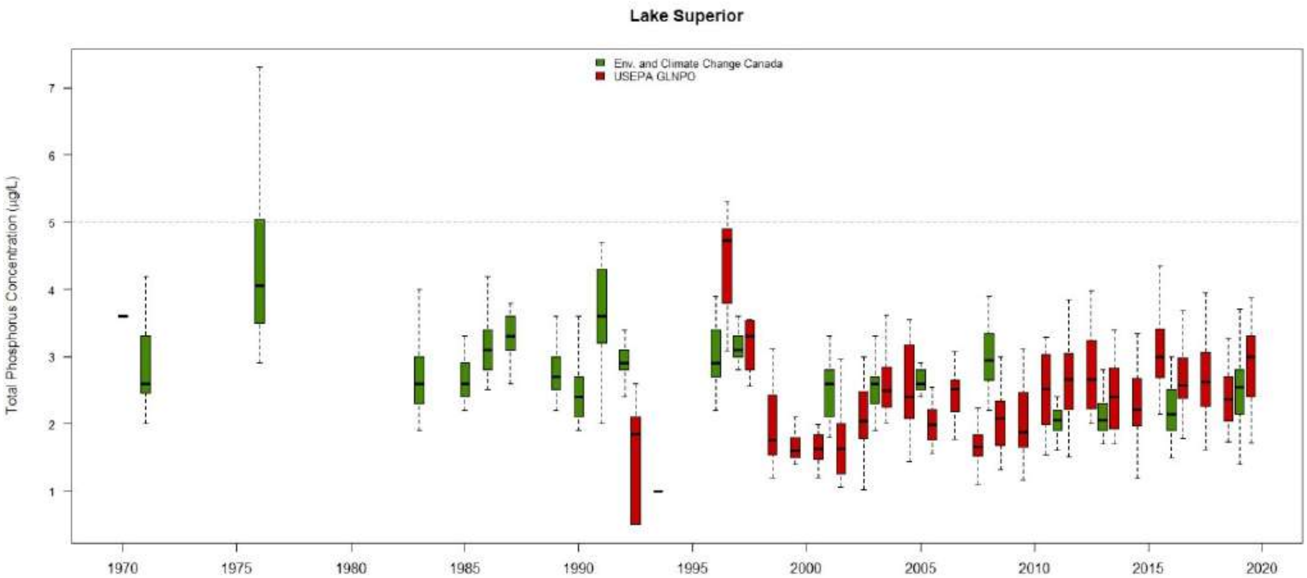

Figure 2. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in Lake Superior (µg/L). The interim GLWQA TP objective of 5 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

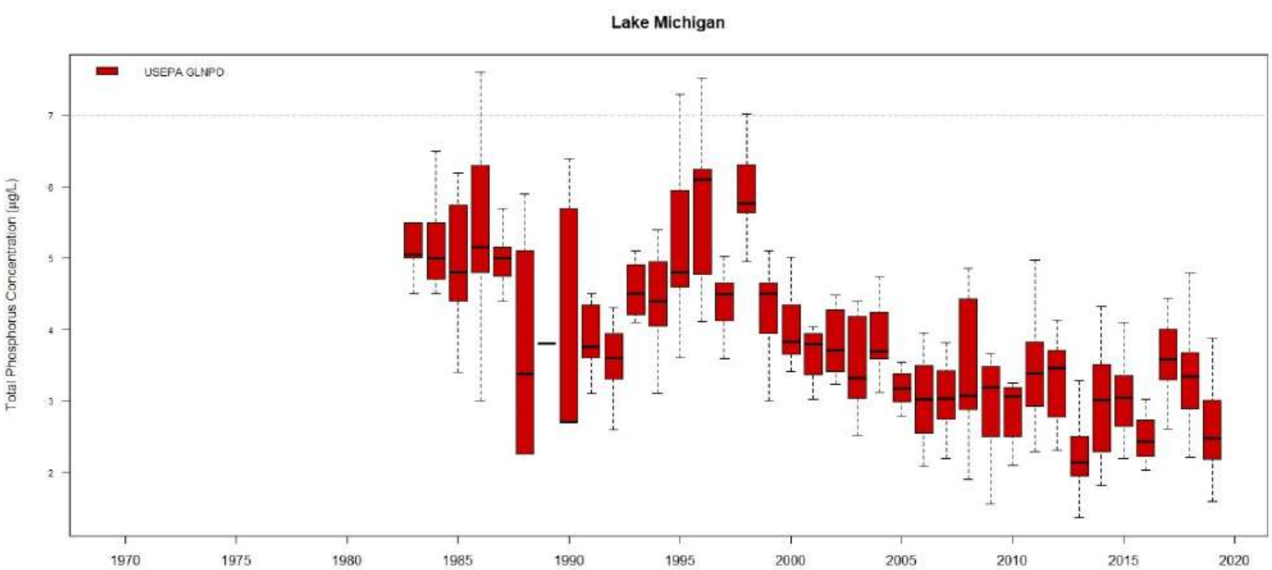

Figure 3. Federal (US) long-term record of offshore, spring (April - May) total phosphorus in Lake Michigan (µg/L). The interim GLWQA TP objective of 7 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Data source: U.S. Environmental Protection Agency.

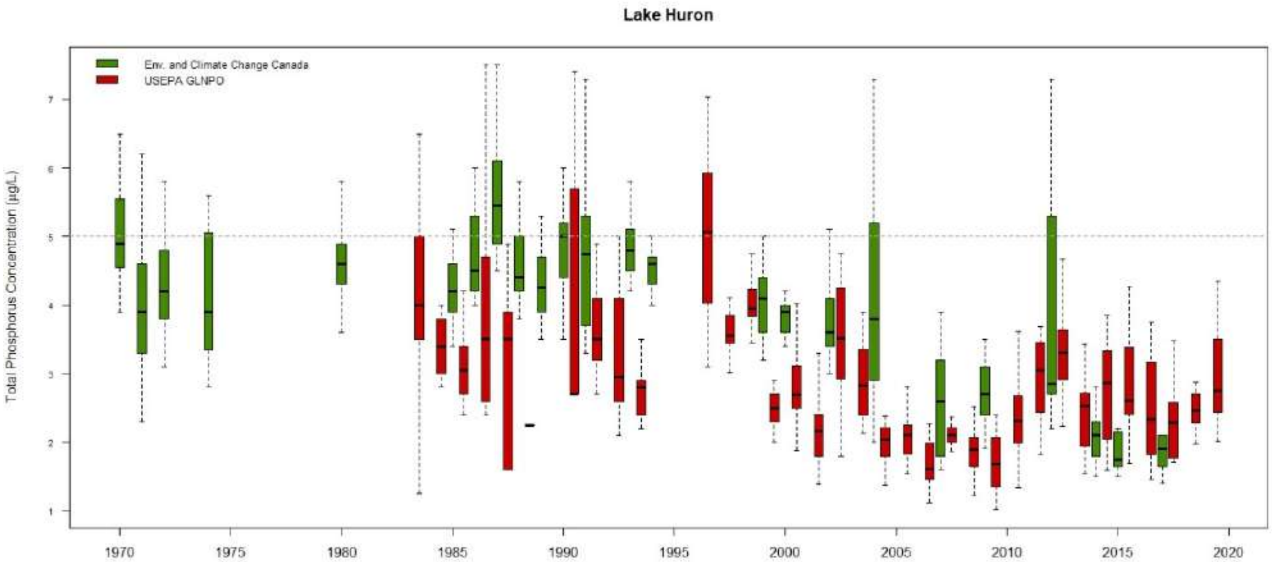

Figure 4. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in Lake Huron (µg/L). The interim GLWQA TP objective of 5 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Georgian Bay data are not shown but the temporal trends closely match those in Lake Huron.
Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

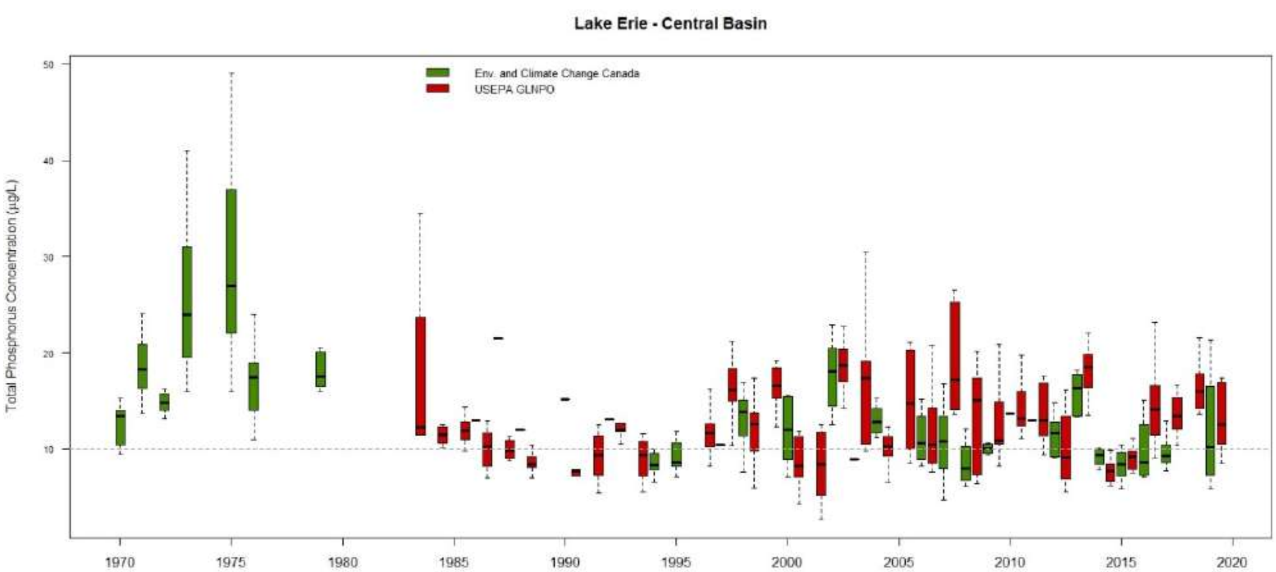

Figure 5. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in the central basin of Lake Erie (µg/L). The interim GLWQA TP objective of 10 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

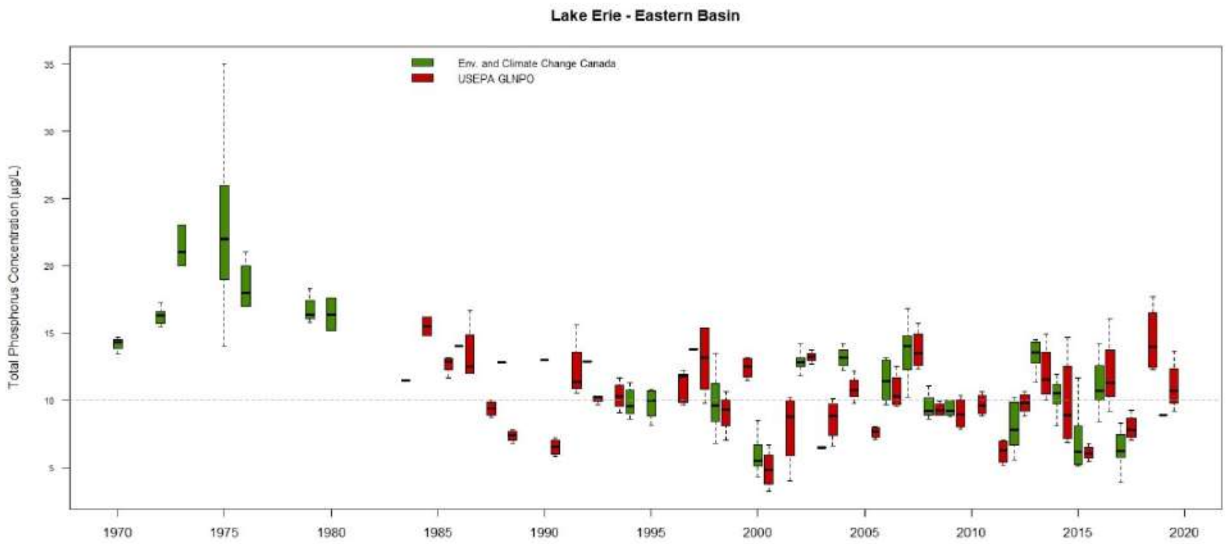

Figure 6. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in the eastern basin of Lake Erie (µg/L). The interim GLWQA TP objective of 10 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

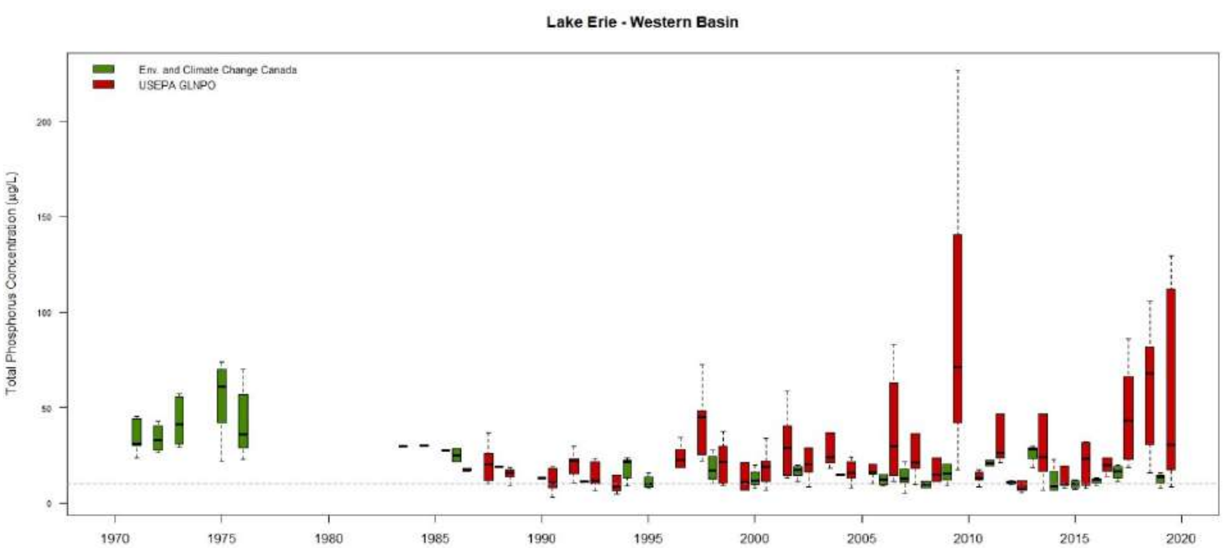

Figure 7. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in the western basin of Lake Erie (µg/L). The interim GLWQA TP objective of 15 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

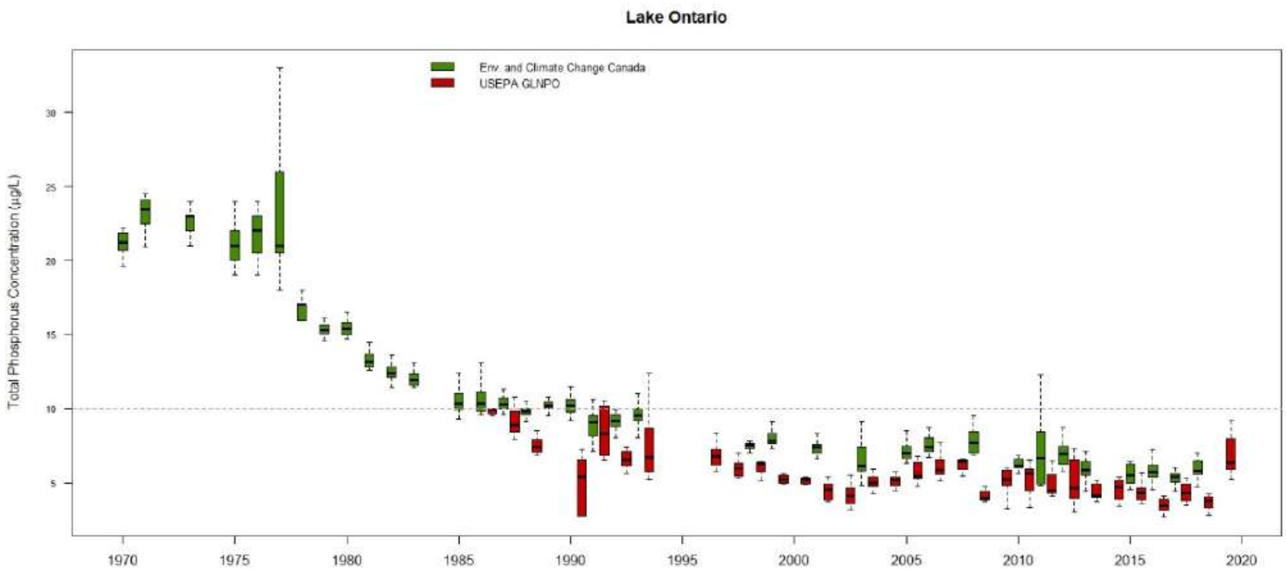

Figure 8. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in Lake Ontario (µg/L). The interim GLWQA TP objective of 10 µg/L is shown as the horizontal dashed line. Boxes show the median values and interquartile range. Outliers (upper and lower 5% of values) have been removed. Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

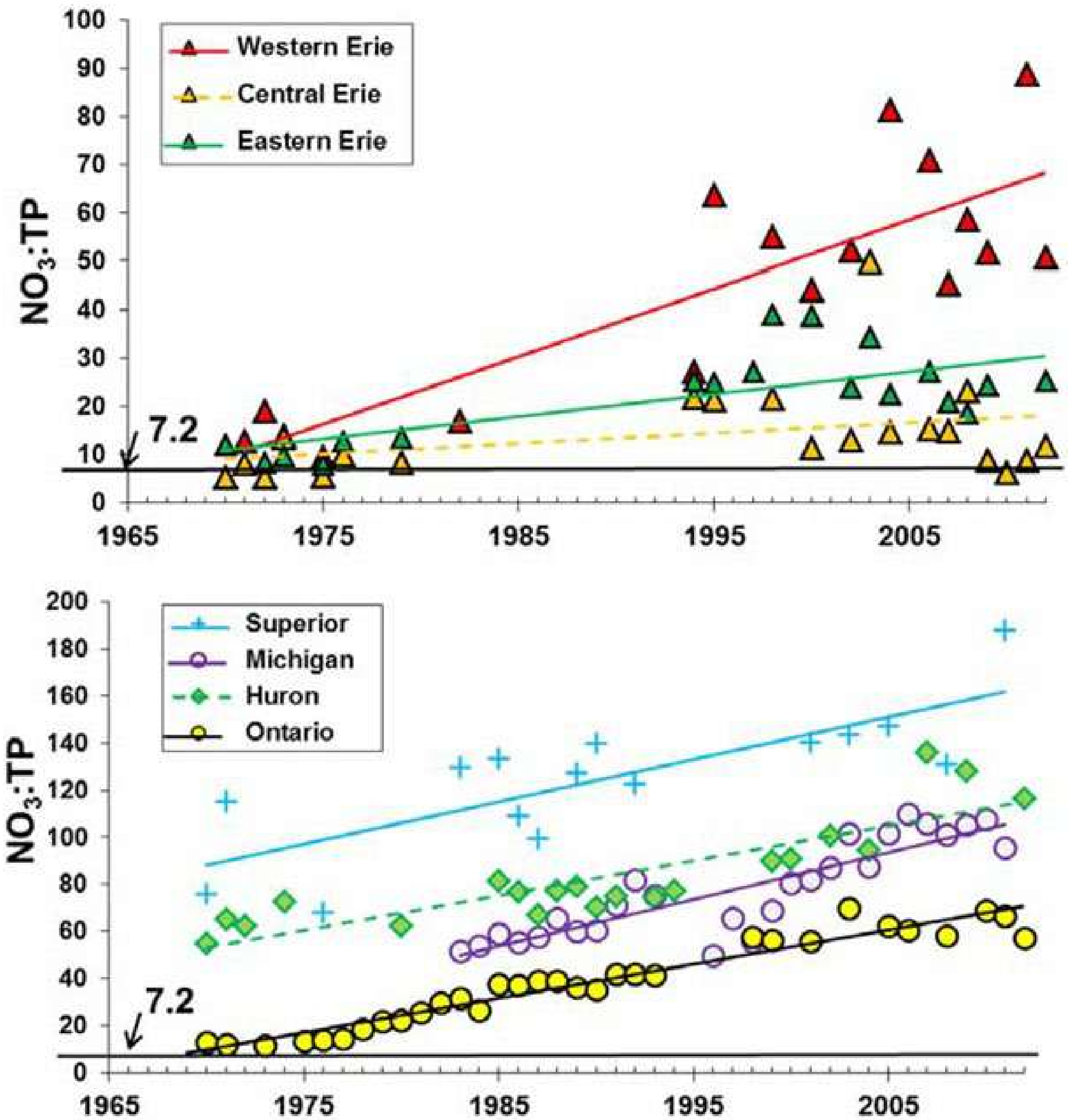

Figure 9. Trends of open lake, spring ratios of median NO3:TP for the Great Lakes. The Redfield ratio of 7.2 mgN/mgP is superimposed as an estimate of the level above which lakes would tend to be phosphorus limited. Phosphorus limitation is beneficial because nitrogen limitation would favor potentially toxic blue-green algae (cyanobacteria). After Dove and Chapra (2015). Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency.

Sub-Indicator: Harmful Algal Blooms – Nearshore and Embayments

Overall Assessment

Status: **Poor**

10-Year* Trend (Based on nearshore algal bloom percent coverage from 2012-2020): Improving

Long-term Trend: Not assessed

Rationale: Cyanobacteria harmful algal blooms (cHABs) are regularly observed in the western basin of Lake Erie, Green Bay (Lake Michigan), Saginaw Bay (Lake Huron), and Lake St. Clair. Algal bloom occurrences are very limited in the nearshore waters of Lake Ontario but have occurred notably within Hamilton Harbor and Bay of Quinte in Ontario. Episodic HABs have been reported in Lake Superior over the last decade, particularly in the western arm of the lake and in the Apostle Islands area, but for the most part these blooms are infrequent, short-lived, small in area, and localized to the nearshore region (Fig. 2). This sub-indicator utilizes satellite-derived bloom metrics (see "Measures" section) that consider the annual extent (by percent coverage) of cHABs in the nearshore waters of each of the five Great Lakes. For this report, "nearshore" is defined as areas of depth ≤16 m, which is based on the recommendation of ≤ 15 m in Binding et al. (2015) but was adjusted to 16 m due to the resolution of the available bathymetric data. Imagery from the Visible Infrared Imaging Radiometer Suite (VIIRS) satellite is used to identify probable cHAB occurrences in these nearshore areas when critical conditions are met for cHAB proliferation (i.e., surface water temperature > 20 °C and chlorophyll-a concentration > 18 mg/m^3; see Sayers et al. 2016, 2019). This metric does not indicate cHAB duration or concentrations of toxins but has been shown to accurately detect cHABs presence and extent within western Lake Erie, Saginaw Bay, and Green Bay; it therefore is expected to be reliable for cHABs detection in the nearshore areas throughout the lakes. This report does not consider the deeper, offshore areas of the Great Lakes because satellite imagery suggests blooms rarely occur in those areas. The period between 2012 and 2020 was the focus of the analyses in this report because these were the years in which VIIRS data was available for the Great Lakes. As such, no long-term trends are assessed in this report. Based on available data and the present approach, slightly more than 20% of the assessed nearshore area of the Great Lakes experiences cHABs and conditions are therefore rated as Poor (Fig. 1). This is driven largely by the algal blooms in western Lake Erie, which account for relatively large areas of the total Great Lakes nearshore, and to a lesser extent those in Green Bay and Saginaw Bay. However, based on data from 2012-2020 the extent of cHABs in the nearshore is declining significantly and the 10-year trend is therefore assessed as Improving.

* *Note: A 9-year trend is reported as only 2012-2020 VIIRS data were analyzed.*

Lake-by-Lake Status and Trend Assessment

Lake Superior

Status: Good

10-Year* Trend (Based on nearshore algal bloom percent coverage 2012-2020): Undetermined

Long-term Trend: Not assessed

Rationale: Localized, ephemeral blooms have been documented in several immediate nearshore areas of Lake Superior (e.g., the Duluth Superior Harbor, around the Apostle Islands since 2015, near Thunder Bay, ON in 2019, and near Terrace Bay, ON in 2020) since the beginning of the analysis period for this report (2012) (Fig. 4). Some blooms, such as a notable event in 2012, extended from the nearshore into the open waters of the western arm of Lake Superior near Duluth and the blooms in 2012 were detected in about 10% of the Lake Superior nearshore area. These blooms are most likely fed by localized nutrient sources from the watershed (Sterner et al., 2020). However, chlorophyll biomass continues to be at low levels in Lake Superior and observed blooms have been mostly non-toxic. The remote sensing methods used in this report are unable to evaluate many of the immediate nearshore areas due to spectral interference from land, which means that the bloom extents reported here may slightly underestimate true bloom extents. Despite this, average annual bloom extent in the nearshore area that is visible to the satellites has remained at about 2% of the nearshore area since 2018, which suggests that cHABs conditions are Good (Fig. 3). The VIIRS satellite data also indicate that nearshore bloom conditions are improving compared to the outlier years, such as 2012 (Fig. 3). However, reports of blooms in the nearshore areas which could not be assessed with the current methodology have increased in recent years. While these blooms are unlikely to extend across 5% or more of Lake Superior's nearshore area, the 10-year trend was assessed as Undetermined in this report due to these observations.

* Note: A 9-year trend is reported as only 2012-2020 VIIRS data were analyzed.

Lake Michigan

Status: Fair

10-Year* Trend (Based on nearshore Algal Bloom percent coverage 2012-2020): Unchanging

Long-term Trend: Not assessed

Rationale: Cyanobacteria blooms are reported in many of the river mouths along the western shore of Lake Michigan and eutrophic embayments such as Muskegon Bay and Green Bay (Fig. 6). The cHAB occurrence heat maps indicate cHABs are most prevalent on the southern end of Green Bay where the Fox River drains into the embayment (Fig. 7). In more severe bloom years (e.g., 2012, 2020), the cHABs extend further north along either the eastern or western shorelines of Green Bay. Blooms consistently occurred in 5-20% of the nearshore area of Lake Michigan during 2012-2020 and the average coverage between 2018 and 2020 was 10%. Conditions were therefore assessed as Fair (Fig. 5). Regression analysis based on the 2012-2020 data suggests that that the trend for the nearshore is Unchanging. The 10-year trend largely reflects conditions in Green Bay, where blooms are most common and extensive.

* Note: A 9-year trend is reported as only 2012-2020 VIIRS data were analyzed.

Lake Huron

Status: Fair

10-Year* Trend (Based on nearshore algal bloom percent coverage 2012-2020): Unchanging

Long-term Trend: Not assessed

Rationale: Lake Huron is generally oligotrophic throughout the offshore and in many nearshore regions, but experiences cHABs in some embayments, most notably Saginaw Bay in the summer months (Fig. 9). Between 2012-2020, cHABs generally occurred in 8-20% of the nearshore area in the lake but did exceed 20% of the nearshore area in 2014 (Fig. 8). During 2018-2020, the average coverage of cHABs in the nearshore was about 13%, so conditions in Lake Huron are therefore assessed as Fair and the trend is Unchanging due to the lack of a significant temporal trend from 2012-2020 (Fig. 8). The status of HABs in Lake Huron largely reflects conditions in Saginaw Bay, where cHABs are the most common and extensive (Fig. 10), especially in the southeastern section.

This nearshore cHABs trend assessment is consistent with Wynne et al. (2021), which noted an unchanging trend in cyanobacteria biomass from 2000 to 2019 for Saginaw Bay.

* Note: A 9-year trend is reported as only 2012-2020 VIIRS data were analyzed.

Lake Erie

Status: Poor

10-Year* Trend (Based on nearshore algal bloom percent coverage 2012-2020): Improving

Long-term Trend: Not assessed

Rationale: Lake Erie continues to experience cHABs, most notably in the western basin. Remote sensing heat maps (Fig. 13) indicate that since 2015, cHABs occur annually along the Ohio shoreline, in the mouth of Maumee River (Matson et al., 2020) and also extend halfway up the Michigan shoreline. Blooms also regularly occur in the central basin, primarily along the southern shore and occasionally occur in the eastern basin nearshore (Fig. 12). 2012 was a particularly severe cHABs year, with greater than 60% of the nearshore area experiencing cHABs according to the remote sensing metrics used. Since 2012, percent coverage has spanned between 30-55% of the nearshore area with the average coverage between 2018-2020 being 36%. The 2012-2020 trend is towards significantly decreasing areal coverage, so the status and trend are Poor and Improving, respectively (Fig. 11).

* Note: A 9-year trend is reported as only 2012-2020 VIIRS data were analyzed.

Lake Ontario

Status: Good

10-Year* Trend (Based on nearshore algal bloom percent coverage 2012-2020): Unchanging

Long-term Trend: Not assessed

Rationale: Annual algal bloom percent coverage for Lake Ontario from 2012-2020 derived from VIIRS satellite data was consistently very low, with little to no evidence of nearshore algal blooms over the time period. Sparse blooms are occasionally observed in the nearshore and cover small amounts of nearshore (Figs. 14, 15), with more notable events occurring along the far eastern shoreline (i.e., Hamilton Harbor and Bay of Quinte on the Canadian side). While the methods used in this report are unable to detect blooms that occur in some nearshore areas of the lake with known bloom occurrences due to spectral interference from the adjacent land (Fig. 15), these blooms would not account for more than 5% of the nearshore area if they were detected by the satellite. Average bloom coverage from 2018-2020 was 0.03% and conditions based on the assessed nearshore are therefore considered Good. There was no discernable trend in the amount of detectable bloom coverage from 2012-2020, so the 2012-2020 trend is considered Unchanging (Fig. 14).

* Note: A 9-year trend is reported as only 2012-2020 VIIRS data were analyzed.

Status Assessment Definitions

Status assessments were based on the average nearshore (areas ≤ 16 m deep) areal extent of suspected cHABs during 2018-2020 according to the VIIRS data. Bloom extent estimates were compared to the following thresholds to determine status:

Good: <5% of the nearshore area assessed (based on average extent for 2018-2020) experienced algal bloom conditions.

Fair: 5-20% of the nearshore area assessed (based on average extent for 2018-2020) experienced algal bloom conditions.

Poor: >20% of the nearshore area assessed (based on average extent for 2018-2020) experienced algal bloom conditions.

Undetermined: Data are not available or are insufficient to assess the condition of the ecosystem components.

See Measure section for a description of metrics.

Note that bloom extent estimates for each lake exclude a small (for Lakes Erie, Huron, and Michigan) to moderate (for Lakes Superior and Ontario) portion of nearshore areas along the shoreline due to limitations of this satellite approach in shallow water adjacent to land. The status assessments in this report reflect conditions in nearshore assessed areas. Conditions in the unassessed areas are discussed in the Ecological Condition section below.

Trend Assessment Definitions

Trend assessments were performed using regression analysis of average annual bloom extent and the following criteria:

Improving: HABs extent estimates show a significant ($p < 0.05$) decrease in average HAB extent over the time period in question.

Unchanging: HABs extent estimates show no significant change ($p > 0.05$) over the time period in question.

Deteriorating: HABs extent estimates show a significant ($p < 0.05$) increase in average HABs extent over the time period in question.

Undetermined: Data are not available or are not sufficient to report on a trend.

Note that 8-Year trends were made for each lake using VIIRS satellite data from 2012-2020.

Endpoints and/or Targets

The endpoint for this sub-indicator report is that all lakes should have minimal occurrences and extent of HABs. This target should ensure that cyanobacteria biomass remains at levels that do not produce concentrations of toxins that pose a threat to human health in the waters of the Great Lakes. The method used to estimate bloom extent in this report does not include *in situ* toxicity data. Toxicity data will likely be included in future reports.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess potential harm to human health, pets, and other organisms or ecosystems from harmful algal blooms (HABs).

Ecosystem Objective

Waters should be safe for drinking and recreational use and substantially free from toxic and/or high abundances of noxious cyanobacteria or algae that may harm humans, animals, or ecosystem health or have other significant adverse effects.

This sub-indicator best supports work towards General Objective #6 of the 2012 Great Lakes Water Quality Agreement, which states that the waters of the Great Lakes should "be free from nutrients that directly or indirectly enter the water as a result of human activity, in amounts that promote the growth of algae and cyanobacteria that interfere with aquatic ecosystem health, or human use of the ecosystem."

Measure

Algal blooms are highly dynamic in space and time, leading to severe limitations of *in situ* sampling for capturing whole lake conditions, identifying areas of potential concern, and documenting long-term trends. Bloom monitoring capabilities have been enhanced greatly by advancements in satellite remote sensing, offering the spatial coverage and temporal repeat visit times required for comprehensive, synoptic monitoring of bloom conditions across the Great Lakes. Multiple remote sensing algorithms are used to extract chlorophyll concentrations or cyanobacteria biomass to describe the spatial extent or intensity and severity of HAB conditions in the Great Lakes and the climatological record of satellite imagery allows interpretations of historical trends over basin-wide features. The maturity of science in algorithm and product development is such that several operational remotely sensed algal bloom products are now available for the Great Lakes and are distributed to the research and water-resource management stakeholder community (e.g., NOAA's HABTracker, Stumpf et al., 2016, and ECCC's EOLakeWatch, Binding et al. 2021). Acknowledging that there are a large number of algorithms and sensors available to the community, the focus here is on the processing streams of operational remote sensing products for algal blooms delivered by the Michigan Tech Research Institute (MTRI). The algorithm approaches are based on detecting key bio-optical properties of algae and cyanobacteria (e.g., absorption, backscatter, fluorescence) across the electromagnetic spectrum using semi-analytical inverse models developed by MTRI. MTRI generates annual estimates of remote sensing derived mean and maximum cHAB areal extents for western Lake Erie, Saginaw Bay, and Green Bay. MTRI also uses this approach to generate annual extents of surface cyanobacteria scums in the western basin of Lake Erie.

MTRI's approach for assessing cHAB extent in Green Bay, Saginaw Bay, and western Lake Erie is a modified Color Producing Agent Algorithm for HABs (MCH). Taken from Sayers et al. (2019), "the MCH approach was a modification of the Color Producing Agent Algorithm (CPA-A), a semi-analytical bio-optical chlorophyll-a retrieval algorithm which simultaneously optimizes estimated concentrations for all three optically active constituents (chlorophyll, non-algal particles, and colored dissolved organic matter, CDOM) using a hydro-optical model derived from extensive *in situ* measurements collected in western Lake Erie" (Shuchman et al., 2006, Shuchman et al., 2013). For HAB detection, the CPA-A is enhanced with empirical relationships between chlorophyll-a and environmental variables (e.g., water surface temperature) and is leveraged using the observed linear relationship between surface chlorophyll-a and phycocyanin pigment concentrations to estimate surface water HABs. In this approach, a threshold value of 18 mg/m^3 of chlorophyll-a was used to classify pixels as cHABs based on segmented regression analysis (Sayers et al. 2016).

For this report, these methods have been applied to all nearshore areas of the Great Lakes which were defined as all areas ≤ 16 m deep similar to the nearshore/offshore delineation of satellite-derived water clarity used in Binding et al. (2015). A cutoff of 16 m was used rather than the15 m in Binding et al. (2015) to delineate the nearshore due to limitations in the resolution of the bathymetric dataset used. The annual nearshore algal bloom extent estimates from 2012-2020 were generated for each lake using the MTRI CPA-A applied to VIIRS satellite data. For each year, cloud-free image pixels were processed with the CPA-A from the April-October period. A cyanobacteria bloom occurrence was recorded if a pixel had a chlorophyll-a concentration value ≥18 mg/m^3 and a surface temperature ≥20°C in at least one image for the April-October period. For each year, the percent of the total nearshore area that experienced at least one day of an algal bloom was computed as the assessment metric. Trends in the algal bloom occurrence metric from 2012-2020 were established for each lake and tested for statistical significance (i.e., linear regression). It should be noted that because this methodology does not include empirical relationships to better define blooms as harmful in areas outside of western Lake Erie, Green Bay, and Saginaw Bay, this metric is aimed at identifying cyanobacterial blooms of any composition which may or may not be harmful.

The cHAB status thresholds for nearshore algal bloom spatial extent were chosen as <5% of the lake nearshore area with algal bloom coverage representative of Good conditions, between 5-20% representative of Fair conditions and >20% as representative of Poor conditions. (Table 2.0). These ranges were based on a *priori* knowledge of bloom coverage and seasonal and episodic events from historical satellite data and on-water field sampling of HABs for each of the Great Lakes.

Ecological Condition

Background

Harmful cyanobacterial harmful algal blooms (cHABs) and/or algal blooms (HABs) are a global issue in eutrophic waters with high nutrient loadings. cHABs/HABs can be differentiated from 'non-harmful' (i.e., nuisance) blooms by their impacts on water quality and biota, generally associated with the production of toxins. Nuisance algal blooms (NABs) are a separate subclass of algal blooms whose impact on the ecosystem is generally associated with elevated levels of biomass and not with the production of toxins. cHABs/HABs and NABs can have detrimental impacts on ecosystem services provided by water bodies and negatively impact aesthetics or recreational use of water bodies. Prior to remediation in the late 1970s, cHABs/HABs and NABs were a major problem in many offshore and nearshore areas in the Great Lakes (e.g., Watson et al. 2008, 2009) and at that time, the risk of toxins had not been widely recognized and concerns focused on reduced aesthetics, taste and odour (T&O), food web structure, beach/intake/net fouling and economic impacts. Lake-wide remediation efforts initiated in the 1980s were mainly directed towards the reduction of point-source nutrient loading, and successfully mitigated many toxic and nuisance algal bloom impairments with progress largely gauged against the management reduction targets for Total Phosphorus (TP) and chlorophyll a (chl-a) (see Nutrients in Lakes sub-indicator report). While cHABs/HABs and NABs overlap in their causes and impacts, NABs (e.g., *Cladophora*) were not included in the estimates or analyses in this report. The MTRI analysis defines surface phytoplankton and does not capture benthic nuisance algae.

Most algal blooms in the Great Lakes are reported in the shallow, nearshore areas (Figure 2). These areas are most prone to shoreline development issues, a greater influx of nutrients, and to some extent, increased public awareness. Using the nearshore definition of areas 16 m or shallower, the size of nearshore zones varies from approximately 1-10% in Superior to 60-90% in Erie. Likewise, the area of influence of physical and climatic factors varies between lakes (e.g., runoff, erosion, thermal bar formation, upwelling/down-welling, alongshore/nearshore/offshore currents, circulation patterns, surface/groundwater inputs, lake level regulation, ice formation, etc). As a result, the nearshore zones are highly dynamic, and there is significant spatiotemporal variance in the biomass and composition of phytoplankton communities. Additionally, cHAB extents in the nearshore are somewhat underestimated using the present methods due to spectral contamination from nearby land and benthic reflectance in the nearshore areas closest to land. The amount of area where satellites could not observe bloom conditions was greater in Lakes Superior and Ontario, which have particularly narrow nearshore zones. For each lake, the spatial extents of cHABs reported in this approach represent the extents detected in areas visible to the satellite.

Current State of HABs in the Great Lakes

Lake Superior: Episodic HAB events have been reported in Lake Superior over the last decade, but for the most part blooms are infrequent, short-lived, small in area, and localized to the immediate nearshore region. Blooms have been shown to correspond with lake warming and extreme rainfall events (Sterner et al., 2020). A catastrophic flooding event in 2012 seeded a large areal bloom that extended into the open waters of the western arm of Lake Superior near Duluth. Another notable bloom event near Duluth in 2018 extended from the nearshore into the open water areas of the lake and as a result is not well represented in Fig. 3 due to the nearshore focus of this report. Blooms have also been observed around the Apostle Islands since 2015, Thunder Bay, ON in 2019, and Terrace Bay, ON in 2020 (Figure 1). Blooms in Lake Superior appear to be predominantly driven by climate change and localized nutrient sources from rivers and streams and are not known to be seeded from lakebed sources. The most common HAB-forming species in Lake Superior is typically *Dolichospermum lemmermannii*, a species that has not been shown to produce toxins yet in open water samples from Lake Superior. At Barker's Island, in the inner harbor of the Twin Ports (Duluth, MN and Superior WI), a sample collected from a short-lived bloom, with the duration on the order of hours contained the cyanotoxins microcystin (just above US EPA Swimming Advisory levels at 8.7 ug/L) and saxitoxin (at the level of detection, 0.022 ug/L). The sample contained two other species in addition to *Dolichospermum lemmermannii*, and these taxa could be the toxin producers (Wisconsin DNR, personal communication). The nearshore bloom events are typically wind-driven and last for hours. These observations are evidentiary from stakeholders in the region and from targeted in-lake monitoring programs.

The remote sensing bloom metrics used here offer the benefit of being able to look back in time and provide objective measures of bloom conditions throughout much of the nearshore not afforded by in-lake HAB monitoring efforts. However, there are some limitations related to the land adjacency effects (spectral interference) that pose challenges for measuring remotely sensed water parameters near land and which generally require nearshore corrections for bottom reflectance. Furthermore, blooms in Lake Superior have typically been documented below the 18 mg/m^3 chlorophyll threshold that was chosen for consistency across all of the Great Lakes. Therefore, the reported metrics likely underrepresent some of the reported nearshore and lower biomass bloom events that are known to have occurred in recent years. Despite limitations of this approach in some nearshore areas, cHABs are unlikely to be occurring in greater than 5% of the lake's nearshore area, and conditions were therefore assessed as Good. On the other hand, recognizing the limitations in the current approach combined with the increases in bloom reports in some nearshore areas, we classified the trend of cHABs in Lake Superior from 2012-2020 as Undetermined . Advances in remote sensing technology and further algorithm development for this region will improve our ability to monitor and assess the highly dynamic, low biomass blooms in the immediate nearshore areas in Lake Superior in the future.

Lake Michigan: In Lake Michigan, cHABs have been reported in some coastal regions and eutrophic embayments such as Green Bay, WI and in many of the drowned river mouths along the western shore. The average annual bloom spatial extent in Green Bay has remained stable over recent years, with no significant trend observed between 2012-2020 (Fig. 5). There has been substantial seasonal and inter-annual variability in bloom extent over this time period. Several years experienced particularly low bloom extent percent coverage (e.g., 2019) while other years (e.g., 2020) had much higher percent coverage of blooms. The cHAB occurrence heat maps reveal that the blooms are concentrated at the southern end of Green Bay where the Fox River drains into the Bay. In more severe bloom years (e.g., 2012, 2020), the cHABs extend further north along either the eastern or western shorelines (Fig. 7).

Lake Huron: Lake Huron is generally limited by low nutrients in most areas but experiences cHABs in some nearshore areas, most notably in Saginaw Bay and Sturgeon Bay (Georgian Bay). Cyanobacteria blooms were reported in the early 1980s (Bierman and Dolan, 1981), but toxicity was not reported until after the early 2000s (Brittain et al. 2000, Vanderploeg et al, 2001). Since 2012, cHAB extent accounted for approximately 8-22% of the nearshore area but regression analysis indicated that the short-term trend for HABs in Lake Huron's nearshore is Unchanging (Fig. 8). In Saginaw Bay, where blooms are most common and extensive, cHABs have been observed along most of the inner bay's coastline, with the blooms most frequently extending just east of the mouth of the Saginaw River (Fig. 10). With the exception of the heaviest bloom years, HABs are rarely observed in more than 40% of clear satellite images during the HAB growing season.

Lake Erie: Lake Erie continues to experience cyanobacteria blooms throughout the western basin and infrequently along the Ohio and Pennsylvania shorelines of the central basin (Fig. 12). Lake St. Clair was not included in this current analysis, but blooms in this location are referenced in McKay et al. (2020). The heat map time series (Figure 15) indicates that between 2012 and 2020, much of the western basin of Lake Erie was susceptible to HABs, with the waters near the Detroit River being the primary exception. Blooms are predominantly observed along the southern shore of Lake Erie, extending from Maumee Bay to Lakeside Marblehead, and even further into the central basin of Lake Erie in the more extreme bloom years. The blooms are most persistent near the mouth of the Maumee River, often being observed in more than 60% of the clear satellite images during the harmful algal bloom growing season. Since 2012, cHABs occurred on at least one occasion in 30% to 60% of Lake Erie's nearshore area. During the period from 2012-2022, however, bloom extent declined significantly at a rate of almost 3% of nearshore area per year, and the 10-year trend is therefore considered Improving.

Lake Ontario: Algal bloom occurrences are very limited in the nearshore waters of Lake Ontario but have occurred notably within Hamilton Harbor and Bay of Quinte in Ontario (McKindles et al., 2020). Annual (2012-2020) algal bloom percent coverage for nearshore Lake Ontario derived from the VIIRS satellite data often had little to no detectable algal blooms. As such, HABs conditions in Lake Ontario are classified as Good and the 2012-2020 trend is Unchanging as there is no significant change over this time period (Fig. 14). We recognize that relatively large portions of the nearshore could not be assessed in the central and eastern parts of the lake, including the aforementioned areas where blooms are known to occur. But, the blooms that occur in these areas would not account for more than 5% of the nearshore area of the lake if they could be detected by satellite using the current method. Therefore, the current assessment likely reflects recent conditions. As indicated for Lake Superior, future advancements in technology and analytical approaches will improve future assessments of cHAB conditions in Lake Ontario.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	x			
Data are from a known, reliable and respected generator of data and are traceable to original sources	x			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	x			
Data obtained from sources within the U.S. are comparable to those from Canada	x			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	x			
Data used in assessment are openly available and accessible	Yes	https://coastwatch.glerl.noaa.gov/		

Data Limitations

In the Great Lakes, HABs are monitored using a variety of *in situ* and remote methods. *In situ* monitoring for HABs in Lake Erie is generally widespread but monitoring in the other lakes (Michigan, Huron, Superior, Ontario) is sparse and largely reactive to observations and reports from local resource managers and the public. The remote sensing approach used in this report has the benefit of much higher spatial and temporal coverage than can be attained through traditional field sampling. However, satellite observations of algal blooms are limited by cloud cover and the spatial, temporal and spectral resolution of the satellite sensors. Satellites also typically detect only surface algal biomass, so imagery should be interpreted as not capturing the full vertical distribution of algal cells. While satellite imagery provides a good estimate of the extent of a surface bloom (e.g., elevated chlorophyll-a concentration), it is not yet possible to determine if the bloom is harmful (produces toxins or taste and odor compounds) based on the satellite imagery alone. Current satellite assets and algorithms are somewhat limited in their ability to detect algal blooms in the immediate nearshore environment due to spectral contamination from nearby land and benthic reflectance. Future work utilizing other satellite sensors, including the European Space Agency, Medium Resolution Imaging Spectrometer (MERIS) imager or the Sentinel Ocean and Land Colour Imager (OLCI) will have a higher spatial resolution and could provide better characterization of algal blooms but would still have limitations in the very nearshore regions in the Great Lakes. The approach used in this report has also not been specifically validated for use outside of Green Bay, Saginaw Bay, and western Lake Erie. However, the criteria used for identifying cHABs in these areas is likely accurate in the nearshore areas assessed here. Future reports may include both *in situ* and remotely sensed data to address these limitations.

Additional Information

Various assessment approaches have been developed to assess HABs in the Great Lakes. Future reporting on this sub-indicator will take into consideration these assessment approaches to ensure consistent binational messaging.

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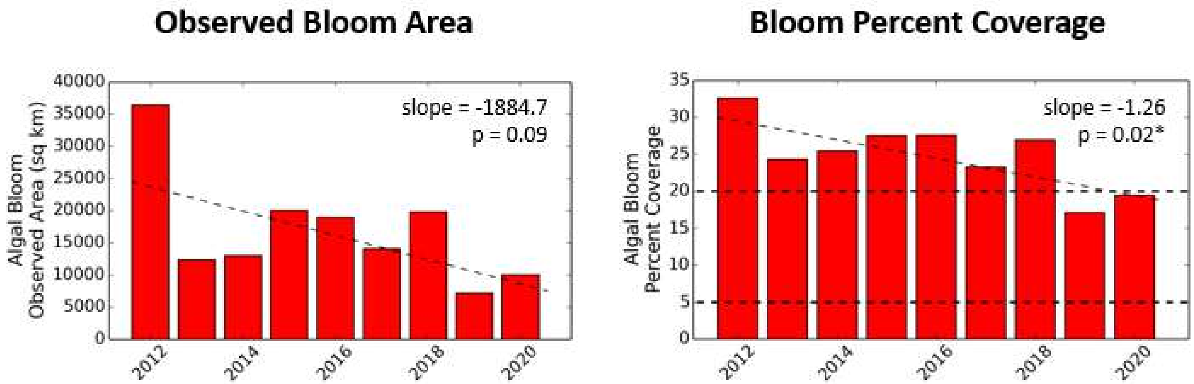
Last Updated
State of the Great Lakes 2022 Report



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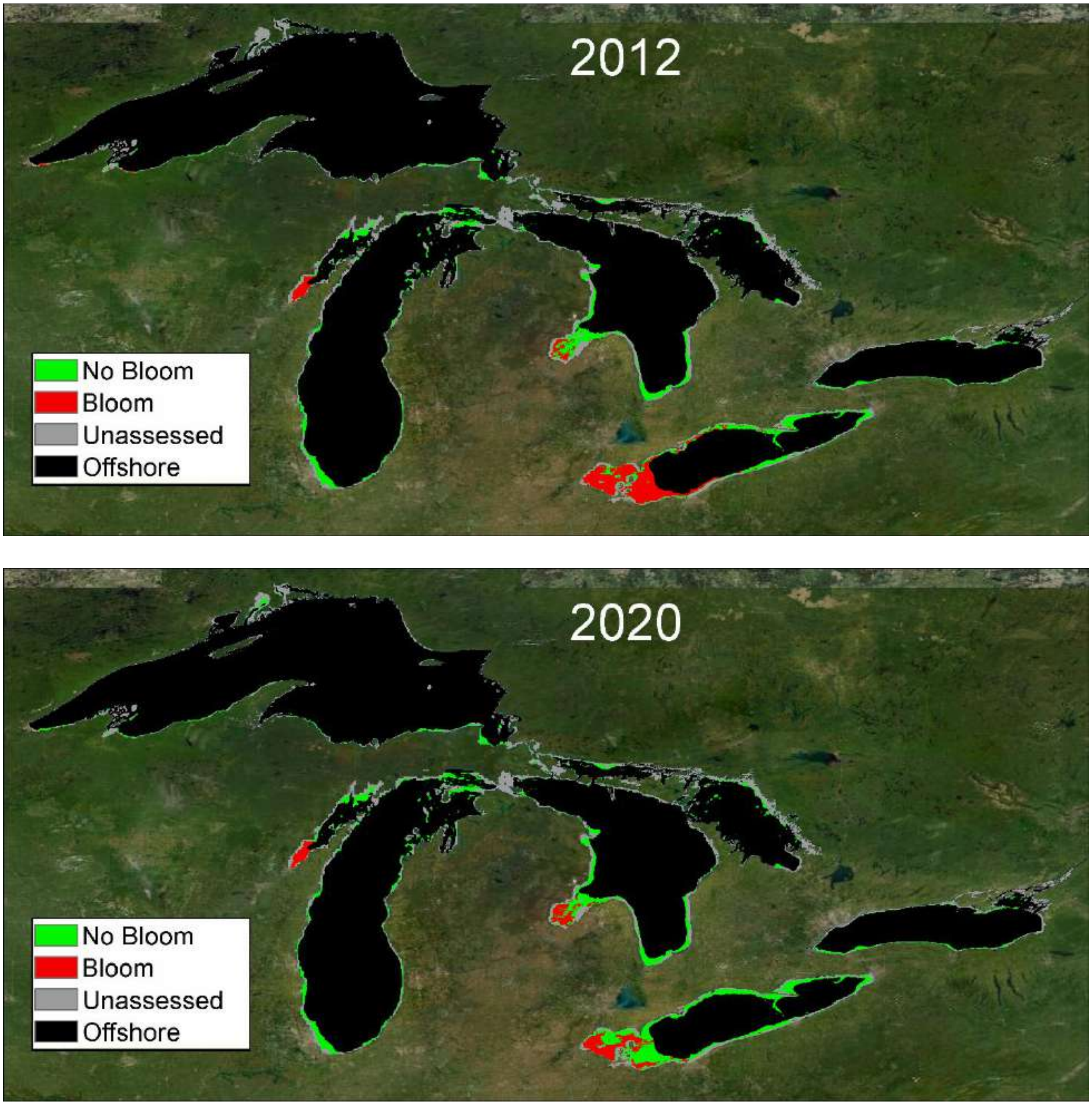

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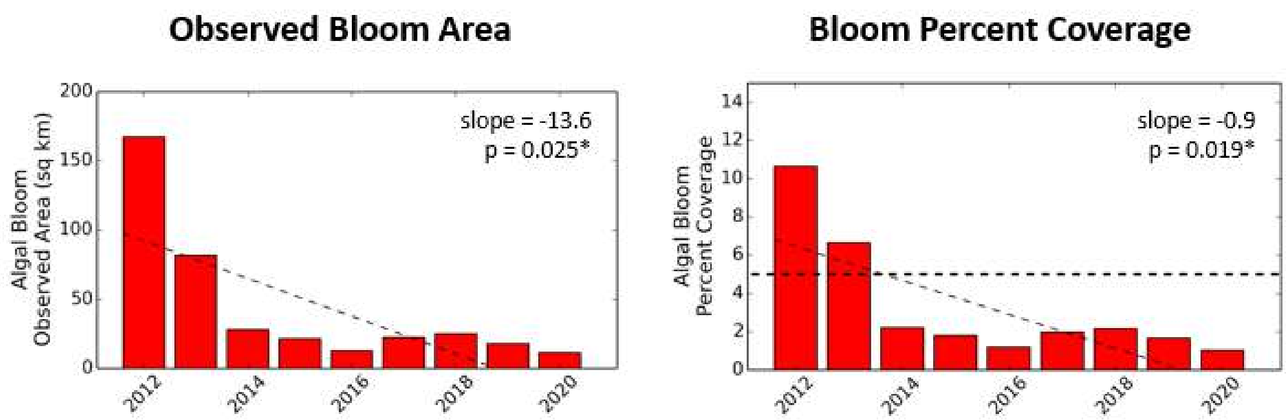

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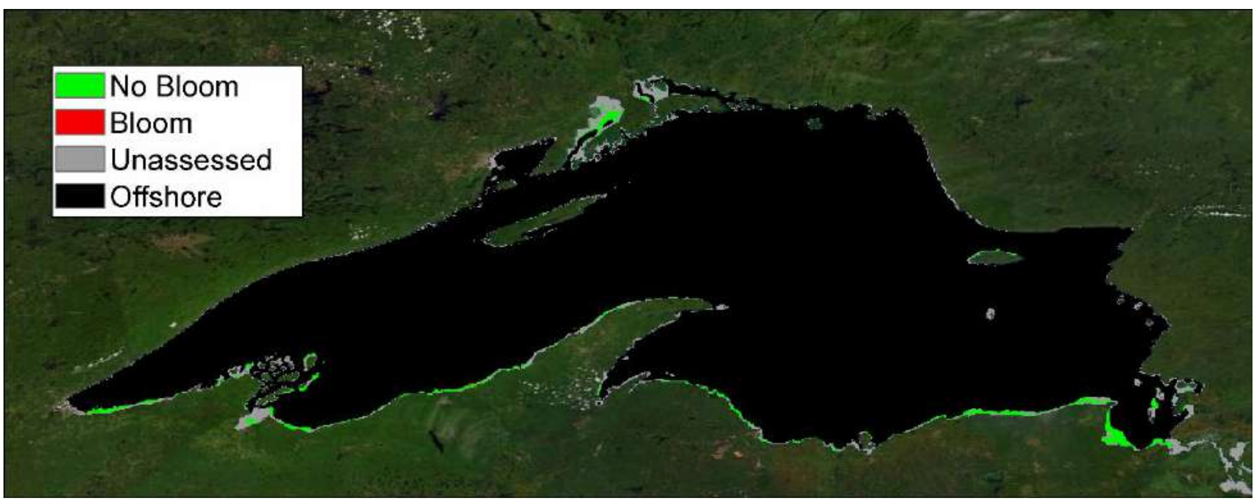

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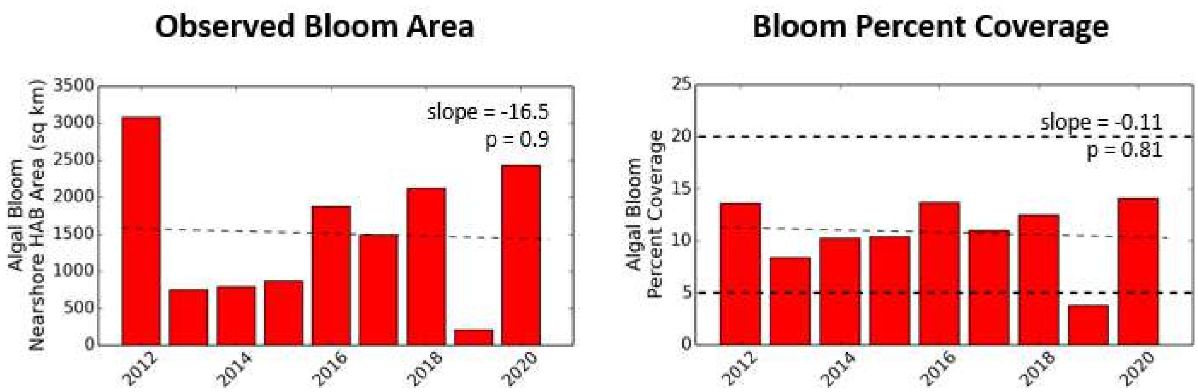

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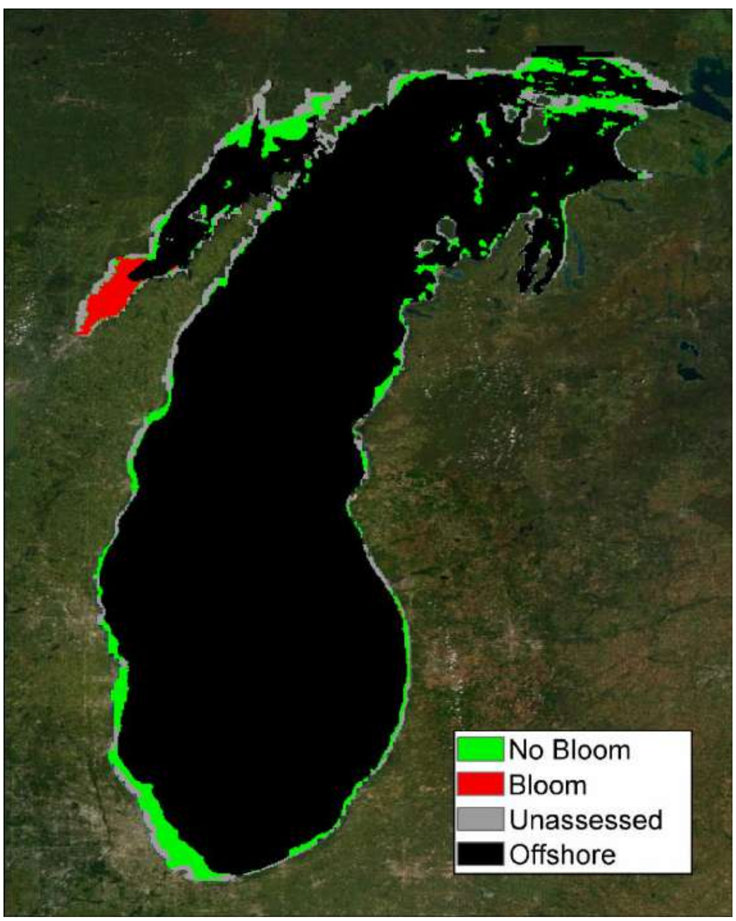

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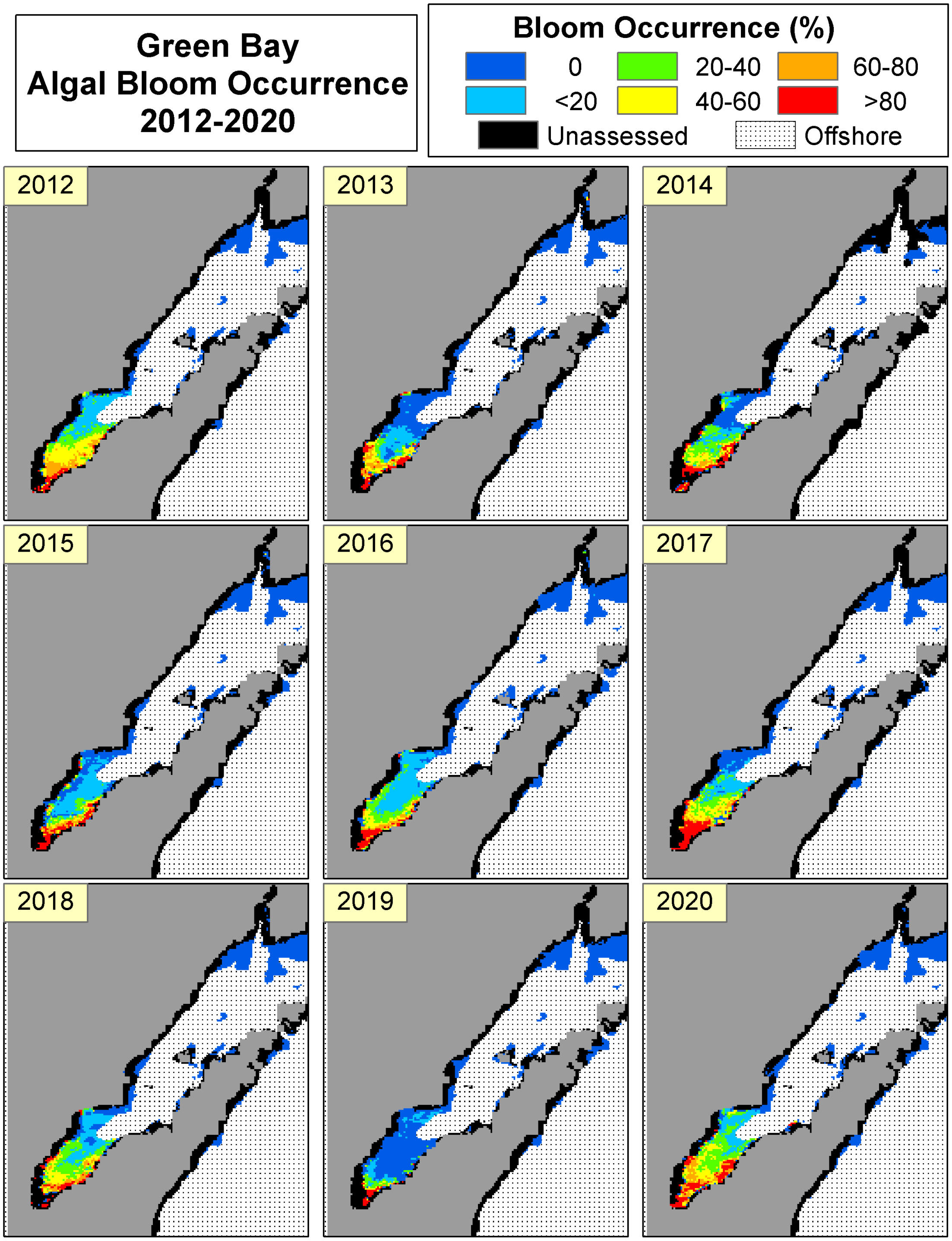

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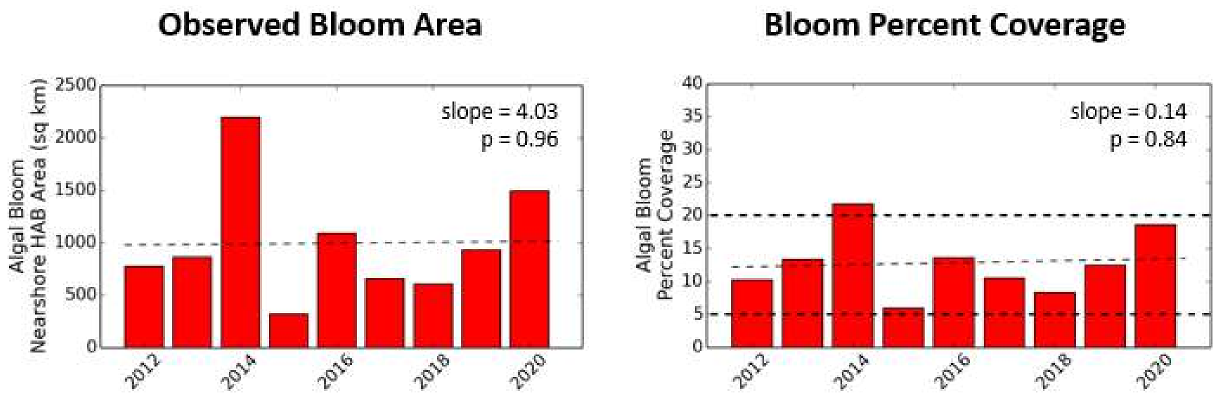

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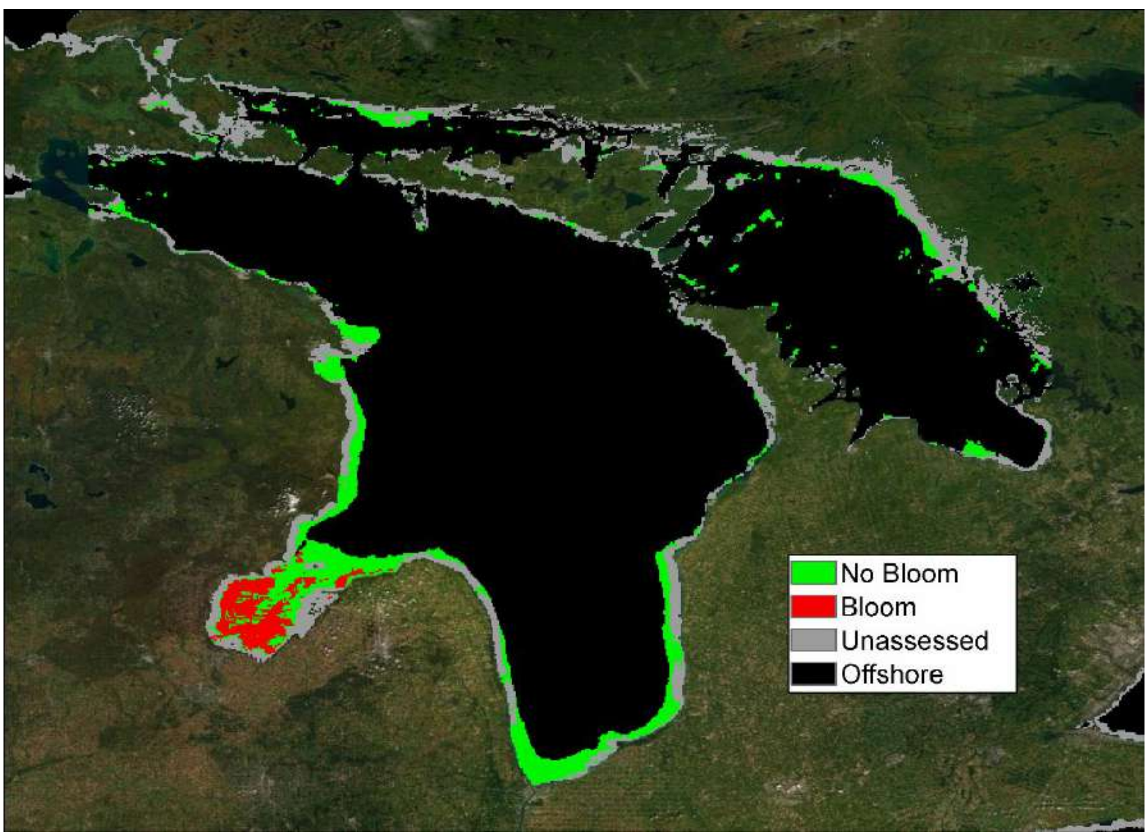

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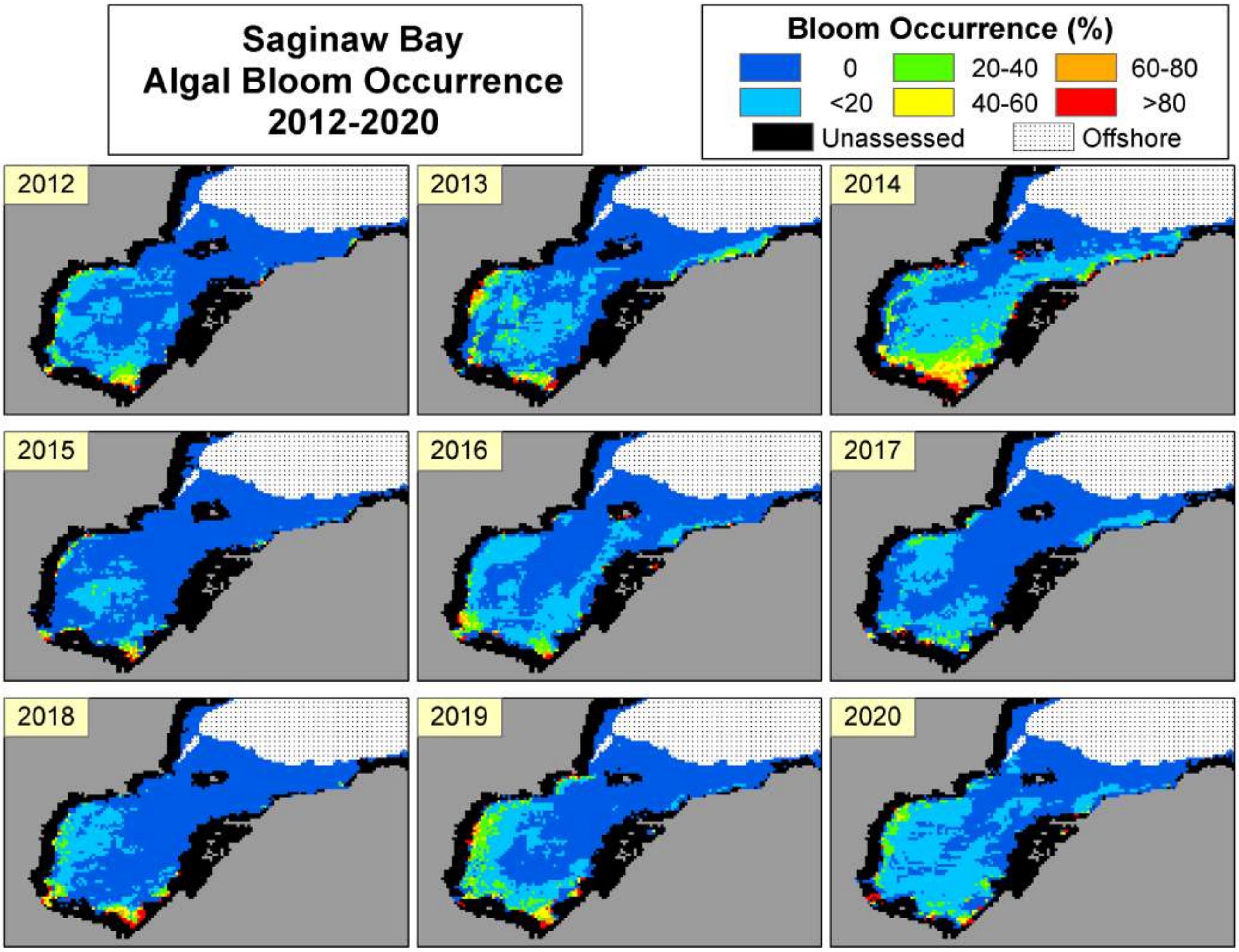

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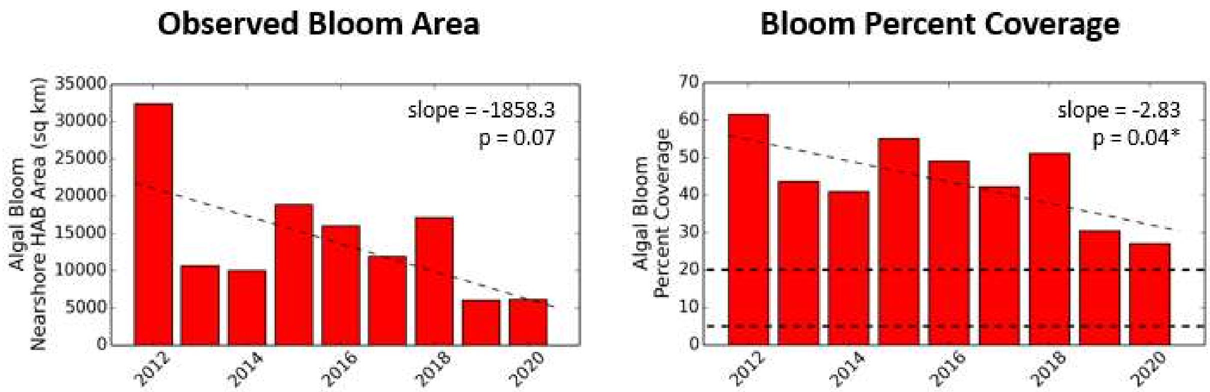

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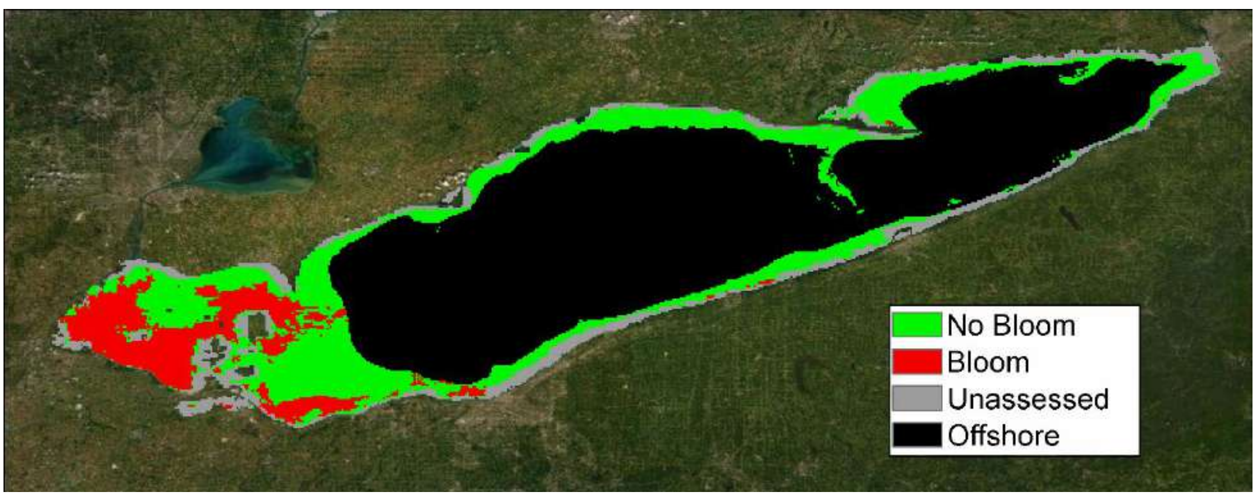

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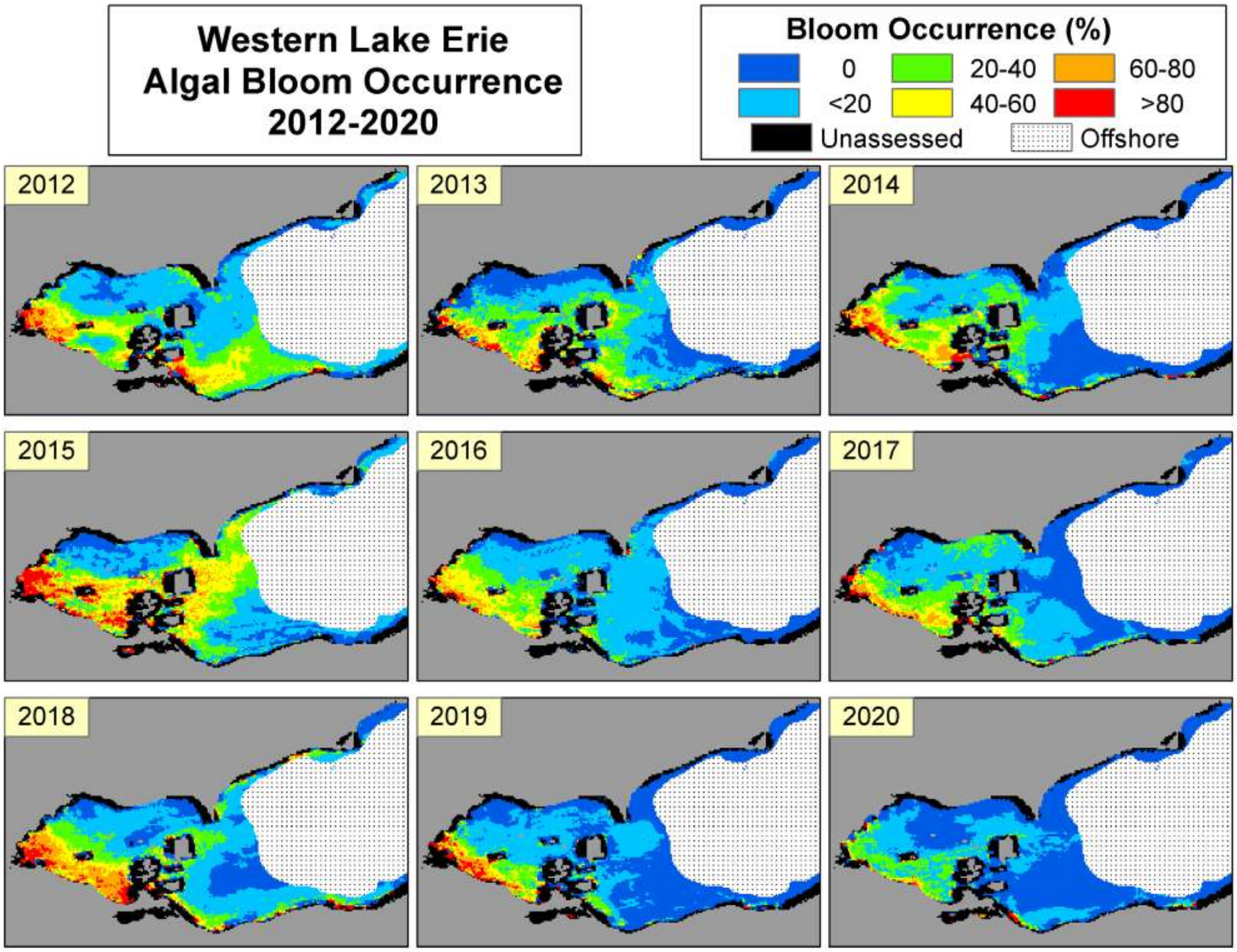

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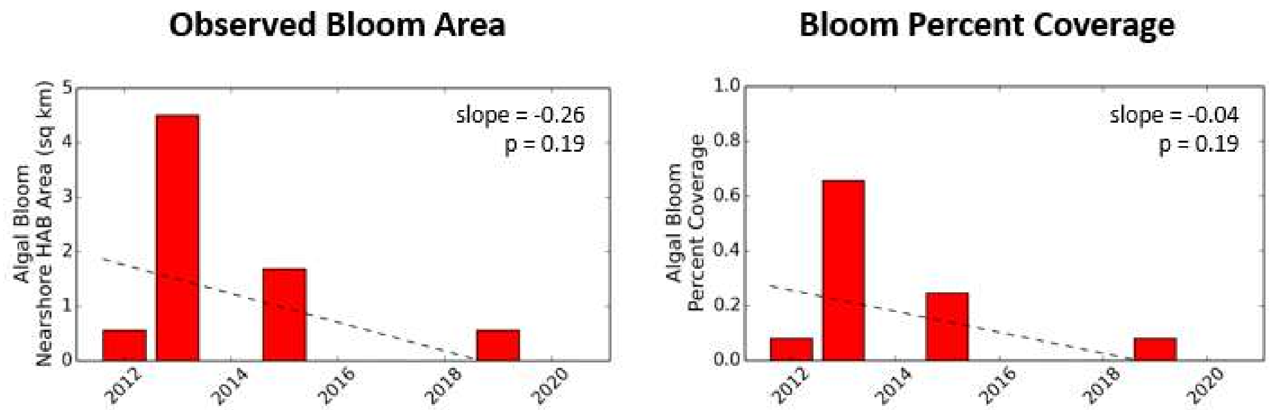

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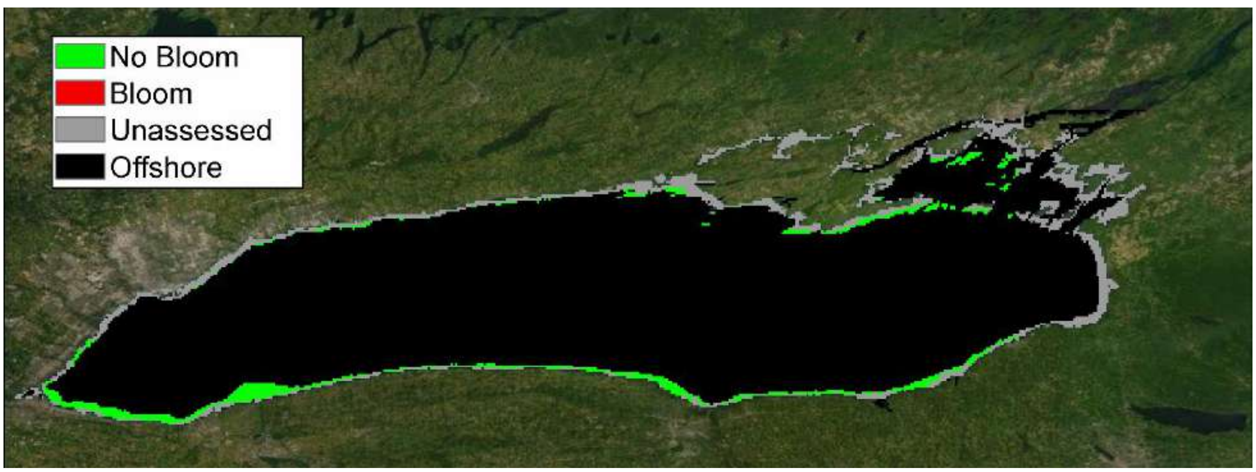

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Sub-Indicator: *Cladophora*

Note: The Overall and Lake-by-Lake assessment section at the top of this report is intended as a high-level summary of the status and trend of *Cladophora* in the Great Lakes. The definitions and date ranges for status and trend assessments are described immediately below this assessment section as are the endpoints from which these definitions are derived. Additional detail about the methods used to evaluate data are provided in the Measures section, and the Ecological Condition section provides a detailed discussion of the status and trend assessments including references to the figures and data as well as important context for these conclusions.

Overall Assessment

Status: Poor

Trends:

10-Year Trend: Undetermined

Long-term Trend (1995-2020): Undetermined

Rationale: *Cladophora* is distributed across broad regions of the littoral (nearshore) zones of Lakes Ontario and Michigan and the eastern basin of Lake Erie. Nuisance conditions in Lake Huron are limited to isolated locations. There is no evidence of algal fouling in Lake Superior. Temporal trends for all lakes are difficult to determine because of the lack of monitoring, with sufficient spatial and temporal scope, to assess trends in distribution or biomass, resulting in an "Undetermined" long-term trend. Empirical and anecdotal evidence suggests that recent biomass levels in Lakes Erie, Ontario, Huron and Michigan are comparable to those observed in the 1960s and 1970s, when conditions were considered problematic. Measurements made since 2004 indicate *Cladophora* biomass is unchanging in Lake Michigan.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Unchanging

Long-term Trend (based on anecdotal information): Unchanging

Rationale: Recent systematic data are not available for Lake Superior, however, fouling of shorelines by *Cladophora* has not historically been an issue in Lake Superior, nor have nuisance levels of *Cladophora* been observed growing in the nearshore, likely due to lower phosphorus loading and a lack of invasive dreissenid mussels in Lake Superior.

Lake Michigan

Status: Poor

10-Year Trend: Unchanging

Long-term Trend (2004-2020): Undetermined

Rationale: *Cladophora* grows at nuisance levels (tentatively set at >50 g dry weight (DW) DW m^{-2}) over large areas of the Lake Michigan nearshore zone. Long-term *Cladophora* monitoring has been conducted at two primary sites: offshore of Atwater Beach (north of Milwaukee, Wisconsin (WI)); and Good Harbor Bay (in the Sleeping Bear Dunes National Lakeshore region of northwest Lower Michigan). Maximum *Cladophora* biomass at both of these sites is highly variable from year-to-year. There is some evidence that biomass near Atwater Beach declined between 2006 and 2015, but it was high again in 2016, and monitoring since then has been insufficient to accurately determine trends. Biomass in Good Harbor Bay was consistently high from 2010 to 2017 and was slightly lower from 2018 to 2020. Since 2019, an additional site has been sampled south of Waukegan, Illinois (IL) and maximum standing crop in-lake biomass >50 g DW m^{-2} has been observed at this site. Prior to 2004 there was no formal *Cladophora* monitoring program in Lake Michigan. Based on earlier qualitative reports and time series of satellite imagery, *Cladophora* biomass was high in the 1970s, decreased in the 1980s after declines in phosphorus loads and concentrations in the lake, and resurged in the late 1990s coincident with the establishment of dreissenid mussels, staying relatively high since then.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend: Undetermined

Long-term Trend (2006-2020): Undetermined

Rationale: *Cladophora* biomass approaches nuisance levels in localized areas that coincide with points of nutrient input on the Canadian shores of the main Lake Huron basin. Field-based monitoring has been limited but suggests that *Cladophora* biomass is generally low and below nuisance levels. Nonetheless, there is periodic fouling of shorelines by mixtures of algae comprised of macroalgae (Charophytes and *Cladophora*) and diatom periphyton. At the sampling sites established on the United States (U.S.) side of Lake Huron since 2018, nuisance levels of growth outside the mouth of Thunder Bay have been recorded but these levels have not been recorded in Thunder Bay or Hammond Bay. *Cladophora* is not found at macroscopically visible levels in the nearshore of eastern Georgian Bay nor has it been reported to foul shorelines in Georgian Bay except in enclosed harbours.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend: Unchanging

Long-term Trend (1995-2020): Undetermined

Rationale: *Cladophora* remains broadly distributed along much of the north shore of the eastern basin of Lake Erie, as well as along offshore shoals with sufficient hard substrate for *Cladophora* attachment. Monitoring of 4 – 5 transects between Port Dover, Ontario (ON) and Port Colborne, ON since 2012 demonstrates that biomass is variable from year-to-year but remains at or above nuisance conditions at most sites sampled. Both offshore and local phosphorus supplies are implicated in this high growth. There is no indication of trends improving or worsening from 2007-2019 in Lake Erie as interannual variation is large. Prior to the initiation of monitoring in 2012, assessment of *Cladophora* in eastern Lake Erie was done on an ad-hoc basis from 1995 to 2010 as exploratory monitoring and research and in response to regional concerns. Surveys conducted in 1995 and 2002 on the north shore of the lake found *Cladophora* abundant and widely distributed over hard substrate in shallow water.

Nuisance levels of biomass were measured near the mouth of the Grand River in 2010. On the U.S. side of Lake Erie, sampling sites established since 2018 have recorded nuisance levels of growth at a site east of Erie, Pennsylvania (PA) but not at sites near Dunkirk, New York (NY).

Lake Ontario (including Niagara River and International Section of the St. Lawrence River)

Status: Poor

10-Year Trend: Undetermined

Long-term Trend (1998-2020): Undetermined

Rationale: *Cladophora* is widely distributed in Lake Ontario. Biomass routinely exceeds nuisance conditions where hard substrate dominates the nearshore lake bottom. Ad-hoc surveys and research studies from 2005 to 2013 indicate nuisance conditions both in the vicinity of point source inputs and in regions remote from any known sources. In U.S. nearshore areas, sites in the western portion of the lake have maximum standing crop in-lake biomass >50 g DW m^{-2} in both ad-hoc surveys and at a station near Olcott, NY that has been assessed since 2018 but not at sites further to the east (Fig. 1 and 4). Inter-annual variability is comparable to that observed in Lakes Erie and Michigan and the lack of a long-term monitoring record hinders assessment of trends.

Status Assessment Definitions

Good: maximum observed standing crop in-lake biomass <50g DW/m^2

Fair: maximum observed standing crop in-lake biomass >50g and < 75g DW/m^2

Poor: maximum observed standing crop in-lake biomass >75g DW/m^2

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

Note: See "Endpoints and Targets" section below for qualification of the above thresholds.

Trend Assessment Definitions

Improving: In-lake *Cladophora* biomass shows a statistically significant change toward more acceptable conditions

Unchanging: In-lake *Cladophora* biomass shows no statistically significant change

Deteriorating: In-lake *Cladophora* biomass shows a statistically significant change away from acceptable conditions

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend

Long Term trend date ranges are determined based on the dates of available data.

Endpoints and/or Targets

Previous observational and modeling studies in Lake Huron noted substantial reductions in quantity of sloughed biomass and shore fouling by *Cladophora* when the standing crop was reduced to ~ 50g DW/m^2 after improvements to local phosphorus loads (Canale and Auer 1982). While a definitive threshold for "nuisance" conditions does not yet exist, the 50 g DW/m^2 has often been used in this context. However, there may be a need to revisit this target because *Cladophora* now extends to greater depths and is not confined to locations near point sources, as it was when Canale and Auer conducted their studies on Lake Huron. Lower biomass spread over larger areas of the lakebed may still result in beach fouling, intake clogging, and other negative impacts. Alternatively, in areas with sparse rocky substrate, a standing crop of more than 50 g DW/m^2 on individual rocks may result in minimal impacts. Additional work is required to determine if this target needs to be revised, and if different targets may be appropriate for different regions.

Sub-Indicator Purpose

The purpose of this sub-indicator is to evaluate biomass of *Cladophora* in the Great Lakes. Data can be used to infer the availability of *Cladophora* to be transported to the lake shore by currents, wind and waves where it may foul beaches and clog water intakes, as well as potentially contribute to other negative impacts on ecosystem integrity such as avian botulism, habitat loss, and food web disruption.

Ecosystem Objective

Waters and beaches should be safe for recreational use and be free from nuisance algae which may: negatively impact drinking water infrastructure and beach use; pose a threat to human health by harbouring pathogens; and contribute to negative impacts on ecosystem health, such as avian botulism and hypoxia. This sub-indicator best supports work towards General Objective #6 of the 2012 Great Lakes Water Quality Agreement (GLWQA) which states that the waters of the Great Lakes should “be free from nutrients that directly or indirectly enter the water as a result of human activity, in amounts that promote growth of algae and cyanobacteria that interfere with aquatic ecosystem health, or human use of the ecosystem.” This sub-indicator also supports General Objective #2 of the 2012 GLWQA which states that the waters of the Great Lakes should “allow for swimming and other recreational use, unrestricted by environmental quality concerns.”

Measure

This sub-indicator reports the biomass of *Cladophora* as standing crop in grams dry weight (DW)/m^2. It may be argued that production of *Cladophora* biomass is a more relevant measure than the standing crop of *Cladophora* biomass, as production represents the amount of material that is ultimately available over the growing season. Although biomass may reflect production to some degree, instantaneous assessments of biomass reflect the balance between production and loss processes (i.e., sloughing and respiration), so variable rates of sloughing due to factors such as water temperature, light availability, and near-bottom shear stress can confound the relationship between biomass and production.

Quantitative sampling of *Cladophora*, typically by divers, is logistically challenging and method-sensitive, but ultimately more feasible than measures that integrate *Cladophora* production over time. Biomass is typically determined by harvesting of all algal filaments from a defined area of lakebed (often in triplicate), followed by removal of non-target organisms (benthic invertebrates) and debris. The cleaned sample is dried to a constant weight followed by gravimetric determination of dry mass. Methodological differences among agencies and investigators may be a source of uncertainty and should be standardized where possible.

Historically, *Cladophora* standing crop was determined on an ad-hoc basis, often in response to reports of local problematic conditions. Long-term monitoring of *Cladophora* is currently being conducted within the context of academic research programs at two locations in Lake Michigan (near Milwaukee, WI and near Sleeping Bear Dunes

National Lakeshore, MI). Annual monitoring of *Cladophora* at 5 locations in eastern Lake Erie has been conducted since 2012 and 3 locations on the north shore of Lake Ontario since 2017 by Environment and Climate Change Canada (ECCC). The U. S. Environmental Protection Agency (USEPA) and U.S. Geological Survey (USGS) initiated ongoing monitoring in the four lower lakes beginning in 2018 including 2 transects in each lake (northern and southern Lake Michigan, contrasting sites in northern Lake Huron, and sites along the southern shores of Lakes Erie and Ontario). Sentinel, long-term, monitoring transects has been selected by each sampling agency to be representative of broad scale trends in environmental conditions but also to reflect local influences; as such, the monitoring results should be considered in the context of light availability (i.e., depth and turbidity) and proximity to shoreline inputs (e.g., tributaries), amongst other factors. Attention needs to be given to the prevailing direction of alongshore currents, as nutrient and light conditions can vary over relatively small spatial scales (< 2km) depending on whether sites are up-current or down-current of any shoreline inputs. Variability in substrate type may also influence monitoring results given the requirement for hard and stable substrate in the inherently high energy environment were *Cladophora* is abundant.

Evaluation of long-term trends of *Cladophora* biomass is confounded by several factors: 1) Small-scale spatial variation is high, resulting in large confidence limits for biomass measurements at a given time and location. 2) Large scale variability makes it difficult to determine how well a small number of sites may represent lake-wide or regional conditions, and 3) there is significant seasonal variation in biomass, so that annual peaks and means may be affected by sampling dates, and sloughing events may confound the relationship between biomass and seasonal productivity.

Ecological Condition

Background

Algae occur naturally in freshwater systems. They are essential to the aquatic food web and healthy ecosystems. However, too much or too little algae can result in negative impacts to water quality, the ecosystem or human health.

Cladophora is a filamentous green alga that grows on hard substrates in all of the Great Lakes. The fouling of shorelines by rotting mats of filamentous algae (primarily *Cladophora*) in the summer months was a common phenomenon in the lower Great Lakes as far back as the mid-20th century (Taft and Kishler 1973). Generally attributed to excess phosphorus (P) pollution, these blooms elicited public outcry and were identified as an emerging issue under the 1978 GLWQA. Targeted research in the late 1970s generally concluded that P load reductions implemented under the GLWQA would contribute to a reduction of nuisance *Cladophora* growth (Auer and Canale 1982). A small number of documented (Canale and Auer 1982, Painter and Kamaitis 1987) and anecdotal reports (e.g., Painter and McCabe 1987) indicated some success of P s control programs in reducing *Cladophora* growth, but systematic monitoring was not conducted. By the mid-1990s, reports of shore fouling began to increase in Lake Erie (Howell 1998) and by the early 2000s, had also increased in Lakes Ontario (DeJong 2000, Malkin et al. 2008) and Michigan (Bootsma et al. 2005). A retrospective analysis of Landsat remote sensing data from 1975 through c. 2010 showed variable submerged aquatic vegetation (SAV) percent cover at focal areas in Lakes Michigan, Ontario, Erie, and Huron through 1990, followed by steep increases in percent cover at sites in Lakes Michigan and Ontario (Brooks et al 2015). Between 2013 and 2019, there has been interannual variability but no directional change in SAV percent cover at sites assessed using Landsat imagery (Shuchman et al 2021). With the recent resurgence of the nearshore algal problem in some areas and with other changes in the ecosystem (caused by invasive species including Dreissenid mussels as well as climate and land use change), the problem has become more complicated. A more detailed and considered history of *Cladophora* in the Great Lakes is provided in

Higgins et al. (2008) and Auer and Bootsma (2009).

The negative economic, aesthetic and recreational use impacts of excessive *Cladophora* growth and biomass are well documented and include the fouling of beaches and residential shorelines, clogging of municipal and industrial water intakes, unpleasant aesthetics, and potential pathogen accumulation associated with mats of decaying algae along the lake shore (Higgins et al., 2008, Peller et al. 2014, Ishii et al. 2006, Chun et al. 2017, Whitman et al. 2003). The ecological impacts of excessive *Cladophora* growth and biomass are not as well understood but may nonetheless be important. Expansive standing crops may represent a substantial (albeit perhaps temporary) nutrient sink over much of the growing season (Higgins et al. 2005). *Cladophora* is generally considered to be a poor food resource for grazers (Dodds and Gudder 1992), though it can be associated with higher quality epiphytes, such as diatoms that provide energy to upper trophic levels via invertebrates and the round goby (*Neogobius melanostomus*) (Tarsa 2021). Accumulation of attached or drifting mats can result in transient hypoxic conditions in shallow littoral regions (Gubelit and Berezina, 2010) and deeper depressions (Tyner 2013); this may have deleterious impacts on invertebrate communities (Berezina and Golubkov 2008) and create an environment conducive to the development of botulism and other pathogens, thus creating a risk for fish and wildlife (Chun et al. 2015).

Current conditions

Locations affected by excessive *Cladophora* biomass continue to be found across much of Lake Ontario and Lake Michigan, as well as eastern Lake Erie. In Lake Huron, reports of excessive biomass are generally restricted to isolated locations along the south-eastern shore. *Cladophora* is not currently considered an issue in Lake Superior where there are currently no long-term *Cladophora* monitoring sites (Fig. 1). A recent assessment of satellite imagery from 2008-2011 indicated that *Cladophora* and other submerged aquatic vegetation cover up to 40% of the nearshore lake bottom visible to satellites (Lake Huron - 15%, Lake Erie - 23%, Lake Michigan - 28%, Lake Ontario - 40 %; Brooks et al. 2015). A next step would be to better merge the areal coverage information with site specific assessments of biomass so that total in-lake biomass might be better tracked.

Lake Superior

Detailed study of the attached filamentous algae community in Lake Superior has not been comprehensively studied since 1969 – 1971 by Parker (1979). Localized investigations in the 1960s (Herbst 1969), 1970s (Gerloff and Fitzgerald (1976) and 1980s (Auer and Canale 1981, Jackson et al. 1990) identified *Cladophora* at locales adjacent to obvious point source inputs of phosphorus and/or warm water discharges. While quantitative information is generally lacking, these studies report the abundance of *Cladophora* was considered relatively low. Recent surveys in 2017 failed to detect any *Cladophora* at sites in Lake Superior (Ted Ozersky, University of Minnesota-Duluth, personal communication). It is assumed that the colder temperatures, low phosphorus levels and the lack of dreissenid mussels may limit the extent of *Cladophora* growth in Lake Superior waters.

Lake Michigan

In Lake Michigan, anecdotal evidence (primarily observations of accumulation on beaches and fouling of water intakes) indicates that *Cladophora* has been growing at nuisance levels since the mid-to late 1990s. Biomass has been monitored at the Atwater station, at a depth of 10-11 m, about 7 km north of Milwaukee Harbor since 2004. These measurements indicate that peak biomass varies from year-to-year ranging from a high of 283 g m^{-2} in 2016 to a low of 34 g m^{-2} in 2010 (excluding single low measurements in 2018 and 2020; Fig. 2), with most years exceeding the status assessment cutoffs for Fair (50 g DW/m^2) and Poor (75 g DW/m^2) condition, While most measurements since 2012 have been less than 100 g DW/m^2, peak biomass remains above this value in most years with multiple observations, and there continue to be problems with fouling of beaches and water intakes nearby. *Cladophora* biomass and phosphorus content have also been monitored on a shallow reef (10-11 m deep) in Good Harbor Bay, near Sleeping Bear Dunes National Lakeshore, since 2010. Similar to the Atwater station, peak biomass has been the status assessment cutoffs, indeed, above 100 g DW/m^2 in most years, and phosphorus content has frequently been above 0.6 µg/mg, which is considered the minimum threshold for *Cladophora* growth (Fig. 3). Of interest is the observation that *Cladophora* biomass at this location regularly exceeds that measured near Milwaukee, despite the absence of any major external nutrient inputs to the Good Harbor Bay area. Since 2019, an additional site has been sampled approximately 20 km south of Waukegan Harbor, IL where peak biomass >50 g DW m^{-2} has been observed (Fig 4 , MWA; Kelly et al. 2020a and b).

Lake Huron

Cladophora biomass can reach nuisance conditions in the vicinity of local nutrient inputs in isolated regions along the south-eastern shore of Lake Huron. Episodic fouling of beaches has occurred sporadically since 2004 although the degree of shore fouling is considerably less severe than that experienced in Lakes Michigan, Erie, and Ontario. In 2013 and 2014, limited measurements were made at a depth of 1 metre near Goderich ON (affected by a municipal waste water treatment plant (WWTP) discharge) and Kincardine ON (affected by a small inflowing agricultural drain). Biomass in 2013 and 2014 at Goderich was 46 g m^{-2} and 49 g m^{-2} respectively, and at Kincardine was 21 g m^{-2} and 33 g m^{-2} respectively. Similar observations of localized growth of *Cladophora* directly adjacent to nutrient discharge points have been observed over the coastline in recent years as in the past (Barton et al. 2013; Howell personal observations, 2003, 2005, 2006, 2010). The spatial extent of growth at these locations was limited. Over broader stretches of the eastern shoreline of this lake, *Cladophora* grows to depths of 20 m, although biomass rarely exceeds 10 – 20 g m^{-2} (Barton et al. 2013). A 2014 study by the Ontario Ministry of the Environment and Climate Change of 48 sites in eastern Georgian Bay found little *Cladophora* over the hard and mostly bare substrate surveyed (Fig. 1). Stefanoff et al. (2018) reported that dreissenid mussels were the strongest predictor of benthic algae and *Cladophora* percent cover at sites in eastern Lake Huron. *Cladophora* is part of a cosmopolitan assemblage of benthic macroalgae in Saginaw Bay (approximately 43°52'N 83°37'W) linked to episodic fouling of beaches with decaying organic matter (Francoeur et al 2014). While Saginaw Bay "muck" contains *Cladophora*, it has historically been recognized as a multi-species eutrophication problem (Francoeur et al 2014). Sampling sites established since 2018 have recorded nuisance levels of growth outside the mouth of Thunder Bay (Fig 4, HAL) but not in Thunder Bay or Hammond Bay (Fig 4, HTB and HHB; Kelly et al. 2020a and b).

Lake Erie

In Lake Erie, *Cladophora* has reached or exceeded nuisance levels since the mid-1990s, primarily along the northern shore of the eastern basin (Howell 1998). Biomass was measured infrequently between 1995 and 2012, with the exception of a significant effort in 2001 – 2002 (Higgins et al. 2005) comprising the most spatially comprehensive dataset for this lake. Since 2012, regular assessment of biomass has occurred at 4-5 transects in the vicinity of the Grand River, extending eastward to Port Colborne, ON by ECCC (points on the north shore of Lake Erie in Fig. 1). Inter-annual variation in peak biomass is high across these sites and displays no temporal trend (Fig. 5). *Cladophora* growth is promoted by high light and warm temperatures, therefore biomass tends to be higher in years characterized by low river discharge and warm, dry growing periods and lower in years with elevated river discharge and wet, cool growing periods (Depew et al. in prep). Additional samples collected at depths of 3 and 6 m at an offshore shoal (Nanticoke Shoal; ~ 7 km southwest of Port Dover, ON) in 2016 exceeded 50 g DW/m^2, indicating that even areas far removed from obvious nutrient inputs (i.e., wastewater outfalls, tributaries etc.) can experience high algal biomass. Valipour et al (2016) modeled upwelling of offshore waters and found significant Soluble Reactive Phosphorus (SRP) supplies to the nearshore in some years, potentially sustaining regional

Cladophora growth independent of local sources. On the southern shore of the eastern basin of Lake Erie, sampling sites established since 2018 have recorded nuisance levels of growth at a site about 4 km east of Erie, PA (Fig. 4, EEP) but not at sites near Dunkirk, NY (Fig 4, EDW and EDE; Kelly et al. 2020a and b).

Lake Ontario

It has been apparent for many years that portions of the shallow lakebed of Lake Ontario are widely and extensively colonized by *Cladophora* (Wilson et al. 2006; Malkin et al. 2008; Higgins et al. 2012). The general long-term trend of nuisance levels of *Cladophora* in the 1960s and 1970s, abatement in the 1980s and 1990s and resurgence in the past two decades is broadly similar to Lakes Erie and Michigan. Measurements of *Cladophora* have been made sporadically, over a wide range of locations, with no systematic monitoring over a long period of time. Surveys by the Ontario Ministry of the Environment, Conservation and Parks and ECCC in 2012-2013, 2015, and 2017-2019 identified general features of *Cladophora* in Lake Ontario: high surface coverage to the point of blanketing hard substrate; strong attenuation of biomass with depth but persisting cover to depths > 10-20 m; typical co-occurrence with high dreissenid mussel cover; and frequent co-occurrence with other filamentous green algae, notably *Spirogyra*. Biomass levels of >50 g m^{-2} have been observed at sites surveyed on the eastern, central and western shores of the main basin of the lake, however, there are few data on the occurrence of *Cladophora* in the eastern basin of the lake (Fig. 1; Fig. 4, OOL, OIR, OSI & OSN, Fig 6). High spatial and temporal variability in biomass levels make inferences about observed patterns challenging. The finding of Higgins et al. (2012) indicating higher *Cladophora* levels in areas of urbanized shoreline remains a central and significant hypothesis influencing the direction of recent studies (e.g., Auer 2014; Howell 2018), signaling the need for phosphorus management over the developed shoreline of the lake. However, additional study may be required as a clear association between spatial patterns of phosphorus loading and *Cladophora* biomass has not always been found (Howell and Dove 2017; Howell 2018).

Summary

The proximal drivers of *Cladophora* growth are reasonably well understood. Numerical models that are driven primarily by three variables – temperature, irradiance, and SRP concentration – perform moderately well in simulations of *Cladophora* growth (Higgins et al. 2006, Malkin et al. 2008, Tomlinson et al. 2010, Auer et al. 2010). However, there remains uncertainty about the processes that ultimately regulate these drivers. There is strong evidence that dreissenid mussels play an important role in Cladophora growth, due both to their ability to clear the water column (and hence increase *in situ* irradiance) by removing particulate material, and their recycling of particulate phosphorus, making dissolved phosphorus more available in the near-bottom layer where *Cladophora* grows (e.g., Ozersky et al. 2009, Martin 2010, Dayton et al. 2014). It is unclear at present if the enhanced phosphorus near the lakebed is derived from excretion of soluble nutrients as metabolic wastes (i.e., Conroy et al. 2005) or perhaps enhanced remineralization of non-edible algae and other detritus that accumulates within mussel beds. The role of dreissenids is highlighted by observations of increased production rates of *Cladophora* in the presence of mussels (e.g., Davies and Hecky 2005) and the presence of high *Cladophora* biomass even in regions where there are no major nutrient inputs (Wilson et al. 2006, Depew et al. 2011). This situation is unlike the 1960s and 1970s, when *Cladophora* was associated primarily with point sources of nutrients. For example, in 2012 *Cladophora* biomass in Good Harbor Bay (near Sleeping Bear Dunes) in Lake Michigan, where there are no major tributary sources of nutrients, peaked at 125 g DW m^{-2}, while peak biomass several kilometres north of Milwaukee Harbor, which is a major nutrient source, was 49 g DW m^{-2}. However, in other regions, (i.e., Lake Ontario) there is evidence that local nutrient inputs do indeed have an influence on *Cladophora* biomass (Higgins et al. 2012).

Biomass and Phosphorus Status as Indicators

Monitoring of standing crop biomass has been and remains the favored metric for assessing the status of *Cladophora*. Peak standing crops are usually achieved in mid-summer, although the exact timing varies between years and locations. Growth rates and loss processes (i.e., sloughing) are known to vary over short time periods (hours to days) in response to environmental conditions (i.e., wind and wave action, turbidity, nutrient supply, thermal regime). This variability in growth and loss rates, along with high benthic substrate heterogeneity, generally leads to significant spatial variability in attached biomass at a given point in time (e.g., Figs. 2 - 6). Therefore, comparisons of point-in-time measurements of biomass across spatial and temporal gradients may be misleading without appropriate consideration of environmental conditions and context.

Even with temporally extensive data series, evaluation of long-term trends is confounded by several factors: 1) small-scale spatial variation is high, resulting in large confidence limits for biomass measurements at a given time and location; 2) large-scale spatial variability makes it difficult to determine how well a small number of sites may represent lake-wide conditions, and so the selection of sentinel sites can be difficult; 3) significant variation in biomass within and among seasons, so that annual peaks and means may be affected by sampling dates; 4) from a fouling perspective, *Cladophora* production is a more relevant variable than *Cladophora* biomass, as production represents the amount of material that is ultimately available. Although biomass may reflect production to some degree, instantaneous biomass reflects the balance between production and loss (sloughing), and so variable rates of sloughing due to factors such as water temperature, light availability, and near-bottom shear stress can confound the relationship between biomass and production. When examining multi-year trends, it is tempting to compare peak biomass among years.

Specific challenges in monitoring *Cladophora* biomass that remain include: 1) determining the accuracy with which areal biomass can be determined with satellite imagery; and 2) development of sampling / measurement methods and approaches that are relatively simple while accounting for spatial and temporal variability.

The phosphorus content (or phosphorus status) of algal filaments has long been considered a useful metric for assessing the status of *Cladophora* and the potential for phosphorus management to be effective in controlling growth. Expressed most commonly as the proportion of dry weight (% DW; Q_P), Q_P is thought to provide a time-integrated measure of algal exposure to phosphorus that: a) removes uncertainty in phosphorus supply created by point in time measures of SRP from the overlying water column (which are frequently near or below the detection limit) and b) represents exposure and uptake of phosphorus by the alga in its physical habitat (i.e., at the lakebed). In general, values exceeding 1.6 mg g^{-1} (0.16 %) are considered phosphorus saturated, values between 1.6 and 0.6 mg g^{-1} (0.16 – 0.06 %) are phosphorus limited, and values below 0.06 % are critically limiting and insufficient to sustain net positive growth rates. Atwater and Good Harbor Bay sites in Lake Michigan generally fall into the phosphorus limited or critically phosphorus limited ranges, with a few exceptions (Figs. 2 and 3). Sites along the northern shore of Lake Erie vary widely in Q_P with both phosphorus saturated and phosphorus limited conditions implied (Fig. 5). Sentinel stie data for Lake Ontario show a pattern of generally increasing Q_P with increasing depth and decreasing light availability; deeper sites are generally phosphorus sufficient while shallower sites have higher biomass and lower Q_P (Fig. 6) However, phosphorus limitation cannot be inferred from Q_P alone for the simple reason that other factors can affect internal phosphorus levels. Q_P may be an inexact indicator of phosphorus availability, as phosphorus uptake rates have been shown to vary by several fold when Q_P is low (Auer and Canale, 1982). Furthermore, using Q_P to infer phosphorus availability and potential for *Cladophora* growth requires interpretation in the context of the growth environment (i.e., light, temperature and the degree of epiphyte colonization on *Cladophora*). In persistently turbid or deep waters, *Cladophora* growth will be light limited, and thus, unable to dilute internal phosphorus stores to minimum cellular quotas. In contrast, higher irradiance in very clear waters and/or shallow areas will lead to higher growth rates that may be sufficient to quickly deplete internal phosphorus stores. Research conducted in Lake Michigan in 2015 revealed that *Cladophora* biomass can vary by more than 10-fold within a distance of 10 km (Bravo et al 2019). In the same survey, *Cladophora* phosphorus content was found to vary more than 3-fold and *Cladophora* biomass was negatively correlated with Q_P (Bravo et al 2019). Q_P alone is not a good indicator of phosphorus availability and growth potential because Q_P can be expected to be low when growth conditions are good, due to dilution within *Cladophora* tissue (Bravo et al 2019). Similar observations were documented in eastern Lake Erie from 2012-2019, with Q_P increasing (and biomass decreasing) along a gradient toward the Grand River, which is a significant source of turbidity and nutrients to the Lake Erie nearshore (McCusker et al. in prep). Water quality monitoring conducted in 2013 at the mouth of the Welland Canal in Lake Ontario demonstrated the inverse relationship often seen between phosphorus and light in areas of the nearshore impacted by runoff (Howell and Dove 2017). These observations underscore the important role of light as a regulator of *Cladophora* growth and the importance of considering light climate when interpreting Q_P and biomass.

There is uncertainty about the degree to which high levels of biomass on the lakebed manifest as shore fouling. At the basin scale, shore fouling concerns align with elevated biomass levels on the lakebed, yet the specifics of fouling problems in an area may not. For example, Riley et al. (2015) found that structural development of beaches (i.e., breakwalls, jetties and piers) were important predictors of the degree of *Cladophora* fouling on Lake Michigan beaches and Barton et al. (2013) found that accumulation of algae on Lake Huron beaches was greatest where shoreline features intercepted nearshore currents. Despite these and other factors effecting *Cladophora* transport and accumulation, the presence of excessive standing crop biomass at a given area is likely to indicate the potential for shore fouling and other negative impacts.

Monitoring

Approaches for monitoring *Cladophora* were reviewed in a 2017 *Cladophora* report. These approaches include collection of grab samples at selected monitoring locations (Higgins et al. 2005), hydro-acoustic methods (Depew et al. 2009), and remote sensing (Schuchman et al. 2013). Recent studies suggest that *in situ* monitoring using time lapse imagery may also be a useful method for monitoring *Cladophora* biomass (Bootsma et al. 2015). Each of these approaches has advantages and disadvantages related to spatial coverage, quantitative accuracy and precision, technical difficulty, and cost. For example, remote imaging and acoustic survey methods offer potential to expand the geographic scope of assessment and subsume some of the variability in biomass induced by processes operating on the metre to sub-kilometre scale (i.e., substrate patchiness, degree of exposure, variation in light climate), however these methods suffer from precision and accuracy issues when estimating biomass. Even among quantitative studies, differences in protocols and approaches to collecting biomass may add additional uncertainty. Specific challenges that remain include: 1) Determining the accuracy with which areal biomass can be determined with satellite imagery; and 2) Development of sampling / measurement methods and approaches that are relatively simple while accounting for spatial and temporal variability.

The lack of an established and consistent framework for *Cladophora* monitoring has been cited as a major impediment to understanding the status and trends of *Cladophora* in the Great Lakes. In 2016, an expert workgroup in support of the GLWQA Annex 4 (nutrients), developed a research plan for Great Lakes *Cladophora* modeling, research and monitoring (Ciborowski 2016). The Parties, Canada and the US, are currently implementing monitoring plans including sentinel stie monitoring. Since 2012, sentinel site monitoring has been conducted at priority nearshore sites in Lake Erie by ECCC (McCusker et al 2019) and collaboratively with the Ontario Ministry of Environment, Conservation and Parks since 2017 in Lake Ontario (McCusker et al. 2021). Since 2018, the USGS has monitored 8-12 sites in the lower four Great Lakes (Nevers and Evans 2021).

There is a need to measure the extent, duration and severity of beach fouling in a consistent way across the lakes and to collectively assess the information. The areal extent of in-lake *Cladophora* beds and their biomass requires further development of assessment methods (e.g., remote sensing, broad spatial surveys, and alternative imaging techniques). Synoptic assessments of SAV were conducted in 2010 and 2012 via remotely sensed images throughout the nearshore of the Great Lakes (Shuchman et al., 2013). Such approaches, while limited in their ability to determine the quantity of algae present, can be used to address some of the challenges related to spatial variation in habitat and/or manual sampling provided that near shore waters are not overly turbid. Integration of habitat and quantitative biomass information is similarly one manner in which the European Union has begun to address status assessments for coastal waters under the Water Framework Directive (e.g., Scanlan et al. 2007).

Moving forward, it will be important to establish linkages between the extent and magnitude of in-lake biomass production and the degree of shore fouling. Significant challenges remain, particularly in terms of characterizing available habitat and improving the robustness of point-in-time sample estimates of algal biomass as representative of larger regional areas of the nearshore.

Currently, there are efforts to evaluate the effectiveness of autonomous under water vehicles (AUV) and remotely operated vehicle (ROV)-mounted optical sensors for determining *Cladophora* biomass and distribution (https://www.usgs.gov/centers/glsc/science/development-and-application-a-robot-assisted-computer-vision-system-map-great?qt-science_center_objects=0#qt-science_center_objects). While these will not be as synoptic as satellite imagery, they are likely to yield more quantitative data compared to satellite imagery and will allow for greater spatial coverage than conventional grab sampling methods.

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Nutrients in Lakes – *Cladophora* is sensitive to levels of soluble reactive phosphorus in the environment
- Benthos (open water) – benthos diversity and abundance may be correlated with the occurrence levels of *Cladophora* and connected by indirect mechanisms that are poorly understood.
- Dreissenid Mussels – *Cladophora* is significantly influenced by the state of water clarity and nutrients in the Great Lakes, which are influenced by dreissenid mussel populations.
- Water Quality in Tributaries – Nutrient loading from tributaries can have both an immediate and long-term effect on *Cladophora* growth. Likewise, tributary loads of suspended sediment and coloured dissolved organic material affect water clarity in the nearshore zone, which in turn affects light availability for *Cladophora* growth.
- Surface Water Temperature: If surface water temperatures continue to warm, the direct effect on *Cladophora* growth is uncertain. Considering that maximum summer surface temperatures in most of the lakes are already above the optimal growth temperatures for *Cladophora*, there may not be a direct positive effect of warming on growth and biomass. However, earlier spring warming can result in earlier maximum growth, alteration of seasonal patterns. Furthermore, available data indicates that dreissenid nutrient recycling rates will respond positively to increased temperatures, which will increase nutrient supply to *Cladophora* growing near dreissenids, and so rising temperatures may indirectly lead to higher *Cladophora* growth rates (Bootsma and Liao, 2014).

This sub-indicator also links directly to the other sub-indicators in the Harmful and Nuisance Algae indicator. Improved wastewater treatment and sustainable agriculture practices which result in decreased nutrient loadings to the Great Lakes may also result in decreases in *Cladophora* biomass.

Traditional Ecological Knowledge (TEK), Citizen Science and other Bodies of Knowledge

In Lake Michigan, the frequency of measurements of *Cladophora* biomass and nearshore water quality in Good Harbor Bay has been increased with the help of scuba-certified citizen scientists, with support from the National Parks Conservation Association (Zemansku Heis 2014, Tyner et al. *in prep*). This effort has resulted in a data set that is more amenable for calibrating and validating a *Cladophora* growth model (https://uwm.edu/glos/data/). However, it is important that protocols be in place to ensure the quality of data collected by citizen scientists. In Lake Erie, the Niagara Citizen Coastal Collaborative (NCCC) has undertaken beach surveys to assess the degree of shore fouling since 2018 and this effort is expanding to Lake Ontario (Ford 2021).

Assessing Data Quality

Authors' assessment of data characteristics and location of data sets

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin			X	
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: USGS 2018 data: https://www.sciencebase.gov/catalog/item/5f22beaf82cef313ed94abf8 USGS 2019 data: https://www.sciencebase.gov/catalog/item/5e3d8879e4b0edb47be258c7 ECCC data: https://open.canada.ca/data/en/dataset/4497ebe5-f45e-4b13-9e98-e9edd016fc66 University of Wisconsin-Milwaukee data: https://uwm.edu/glos/data/		

Data Limitations

A lack of long-term monitoring means that trends cannot be determined, and the current resurgence of *Cladophora* cannot be placed in historical context. Most studies of *Cladophora* in the Great Lakes are focused on specific sites with known algae problems and may therefore not be suitable for scaling up. Implementation of improved remote sensing and visualization techniques could permit reporting of regional or lake-wide biomass estimates.

Excessive growth of *Cladophora* was a management challenge in many parts of the Great Lakes in the 1960s and 1970s and was one of the hallmarks of eutrophication. In highly eutrophic regions, such as Lake Erie, Cladophora was relatively widespread, but in less impacted systems, such as Lake Huron, it was associated primarily with phosphorus point sources (Canale et al. 1982). Following the implementation of nutrient management programs that led to reduced phosphorus loading, *Cladophora* biomass declined in the 1980s and early 1990s. But in most of the Great Lakes, with the exception of Superior, there was a resurgence of *Cladophora* growth in the mid-1990s to 2000s (well underway by mid-1995 in eastern Lake Erie) (Higgins et al 2008, Auer et al. 2010, Brooks et al. 2015). The application of models that simulate *Cladophora* growth in response to light, temperature and dissolved phosphorus concentration (Tomlinson et al. 2010; Kuczynski et al. 2020), along with historical data for these variables, suggests that the resurgence of *Cladophora* is largely due to increased water clarity caused by the filter feeding of dreissenid mussels, with additional contributing factors being nutrient excretion by mussels (Auer et al. 2010; Bootsma and Liao 2014) and, in some areas, local inputs of allochthonous phosphorus (Auer et al. 2021). While management of nutrient loading remains an important tool for controlling *Cladophora* growth in the Great Lakes, the relationship between nutrient loading and *Cladophora* growth may have changed due to the alteration of internal nutrient dynamics by dreissenid mussels. Indeed, the apparent important role of light suggests that nutrient supply from dreissenids may meet algal nutrient requirements in most light environments, regardless of external nutrient loading rates. Hence there is a renewed need for time-series data and quantitative models to better understand the mechanisms that regulate *Cladophora* growth and how it may respond to specific nutrient management actions (Bootsma et al. 2015). These models will require an improved understanding of the physical and biogeochemical processes that drive nearshore nutrient dynamics.

Additional Information

Cladophora can be measured by both areal extent and by biomass. However, biomass is probably more useful, as percent coverage can remain relatively constant, even if biomass increases or decreases. It is anticipated that biomass of *Cladophora* will decline as nutrient concentrations decrease and algae become nutrient-stressed. Phosphorus control strategies have worked in the past for reducing *Cladophora* but the issue is now complicated by dreissenid-induced increases in water transparency and the conversion of particulate phosphorus to dissolved phosphorus at the lakebed-water interface. The trends provided by this sub-indicator may aid in identifying areas for targeted phosphorus reductions and may also provide guidance regarding any new targets for total phosphorus concentration in the nearshore zone. While the General Objective #6 of the GLWQA is relevant, it's important to recognize that, unlike in the past, the current nuisance algae problem is not necessarily a nutrient loading problem. For example, *Cladophora* is a problem in Lake Michigan, but total phosphorus concentrations in both the pelagic and nearshore are well below the GLWQA target of 7µg/L.

Cladophora often makes up the majority of nuisance benthic algae that fouls beaches and water intakes in the Great Lakes. However, *Cladophora* is often accompanied by other algae, especially diatoms that grow epiphytically on *Cladophora* filaments (Dodds and Gudder 1992). In some cases, nuisance benthic algae can also consist of other taxa, including the green algae *Chara*, *Spirogyra*, and *Oedogonium* (Barton et al. 2013, Higgins et al 2008, Francoeur et al. 2014). Hence, while the genus *Cladophora* is usually the dominant form of nuisance benthic algae in most regions of the Great Lakes, the term "*Cladophora*" often is used to refer to a conglomeration of benthic algae.

The issue of *Cladophora* in the Great Lakes merits sustained integrated research and monitoring because the symptoms of coastal impairment cannot be easily ignored given the proximity of the problem to recreational and industrial users. Given the apparent sensitivity of *Cladophora* to very low levels of SRP (Auer et al. 2010), the principal challenge is the better understanding of the relative contributions of nutrient supply from both lake-wide and local sources, as well as the internal processes that regulate phosphorus supply to *Cladophora* growth.

Following a robust binational science-based process and extensive public consultation, Canada and the U.S. have adopted phosphorus reduction targets (compared to a 2008 baseline) for the Western and Central basins of Lake Erie to address algal toxins and low–oxygen (hypoxic) areas (IJC 2016, Objectives and Targets Task Team 2015). For the Eastern Basin, a target has not been recommended to address nuisance algae (*Cladophora*) at this time (Objectives and Targets Task Team 2015). Nonetheless, it is important to note that targets have been recommended for the Western and Central Basins and these work in concert, not in isolation. Because all tributaries to Lake Erie, including the Detroit River and the Huron-Erie Corridor, contribute phosphorus loads to the Eastern Basin, the reductions needed to address algal blooms and hypoxia may lower the phosphorus concentrations in the Eastern Basin as well. This phosphorus load reduction may help address nuisance algal issues in the Eastern Basin, while maintaining enough nutrients to support the fisheries. Further work to identify potential targets that will minimize impacts from nuisance algae in the eastern basin of Lake Erie continues.

Evaluating the current status of *Cladophora* is a somewhat subjective exercise, based on measurements of biomass when and where available, the frequency and magnitude of accumulation on beaches, and the fouling of water intakes. From a management perspective, it would be ideal to designate a biomass target, which would be useful not only for the purpose of assigning a status, but also for developing management strategies with specific, quantitative objectives, the most obvious being nutrient loading targets. As discussed above, and in the commentary by Bootsma et al. (2015), a dry biomass of 50 g m^{-2}, which was suggested as a nuisance threshold for Lake Huron in the early 1980s (Canale and Auer 1982), may now be well above the level that leads to a "nuisance" and beneficial use impairment, because nuisance growth is no longer restricted to nearshore regions adjacent to point nutrient sources, and the depth range of *Cladophora* has increased due to greater water clarity. Other factors also confound the use of a single biomass target. In nearshore regions with sparse rocky substrate, biomass on rocks may exceed 50 g m^{-2}, but spatially averaged biomass may be well below that level, resulting in little accumulation on shore. Also, standing biomass may be a poor indicator of the actual amount of biomass available for accumulation on shorelines, because biomass is not necessarily correlated to production. A significant portion of *Cladophora* production may be lost to sloughing (Canale and Auer 1982), and in summers when sloughing rates are high (due to wave-induced turbulence or high temperatures), standing biomass may remain low while the availability of *Cladophora* for accumulation on beaches is high. While these conditions might suggest that the frequency and magnitude of accumulation on the shoreline is a more useful measure, shoreline accumulation can also be misleading, as shoreline accumulation is stochastic and subject to the vagaries of nearshore currents and waves. Reliable evaluations of the status of *Cladophora* will ideally depend on measurement of more than one variable, such as biomass. Additional measurements that will support evaluation, and lead to a better understanding of the factors and mechanisms that regulate *Cladophora* include tissue phosphorus content, water clarity (along with solar radiation), and growth rate. While direct measurement of growth rate is technically more challenging than measurement of biomass, it may be possible to use a proxy for growth rate, such as the ^{13}C:^{12}C ratio of *Cladophora*.

The designation of *Cladophora* as a nuisance is based primarily on its impact on shoreline conditions, which are the most visible to the public. As discussed above, there are a number of less obvious, and less well understood ways in which *Cladophora* affects nutrient and trophic dynamics (Turschak et al. 2014) and contaminant transfer (e.g., Lepak et al. 2015). These processes ultimately influence ecosystem integrity and beneficial uses, so a rigorous assessment of the status of *Cladophora* will require that these factors be increasingly understood.

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List of Figures

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Figure 2. *Cladophora* biomass (A) and tissue phosphorus content (B) at the Atwater location (Figure 1) in Lake Michigan (7.5 km north of Milwaukee Harbor, depth = 10-11 m). Vertical grey lines represent the standard deviation of triplicate measurements. Source: H Bootsma

Figure 3. *Cladophora* biomass (A) and tissue phosphorus content (B) on the Good Harbor Bay (Figure 1) reef, near Sleeping Bear Dunes National Lakeshore, northeastern Lake Michigan (depth = 10-11 m). Vertical lines represent the standard deviation of triplicate measurements. Source H. Bootsma.

Figure 4. Time series of benthic algal biomass, dominated by *Cladophora*, for sites in the lower 4 Great Lakes. Lake is indicated by the first letter of each abbreviation (M for Michigan, H for Huron, E for Erie, and O for Ontario), specifically sites are identified in Figure 1: Lake Michigan near Sleeping Bear (MSB), Michigan City, IN (MMC), and Waukegan, IL (MWA); Lake Huron near Hammond Bay (HHB), Thunder Bay (HTB), and Alpena, MI (HAL); Lake Erie near Erie, PA (EEP), west of Dunkirk, NY (EDW), and east of Dunkirk, NY (EDE); and in Lake Ontario near Olcott, NY (OOL), Irondequoit, NY (OIR) and near Stony Island, NY (OSI&OSN). Site OSI, east of Stony Island, was sampled in 2018 and site OSN, north of Stony Island, was sampled in 2019. Note y-axis on MMC panel has a different scale than the other panels.
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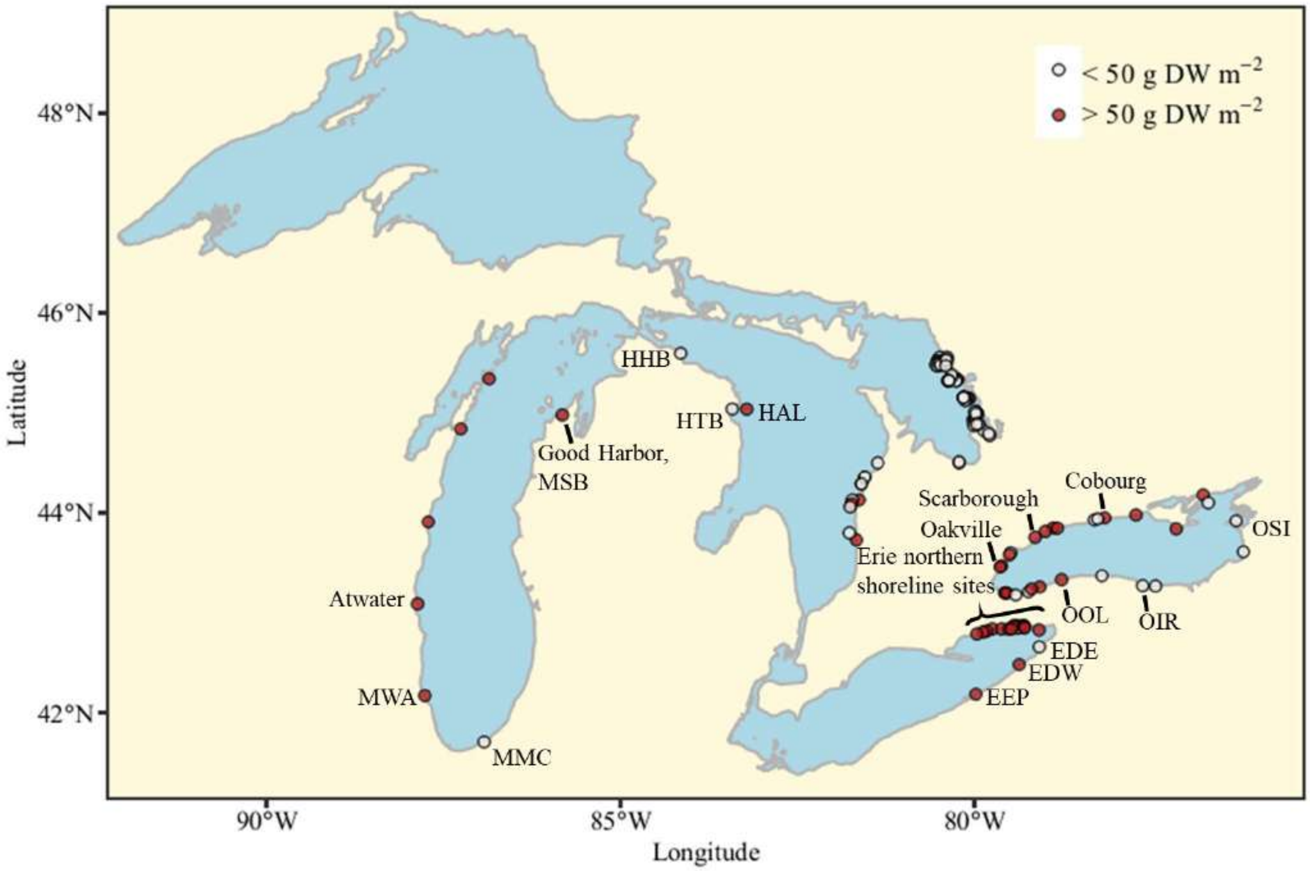

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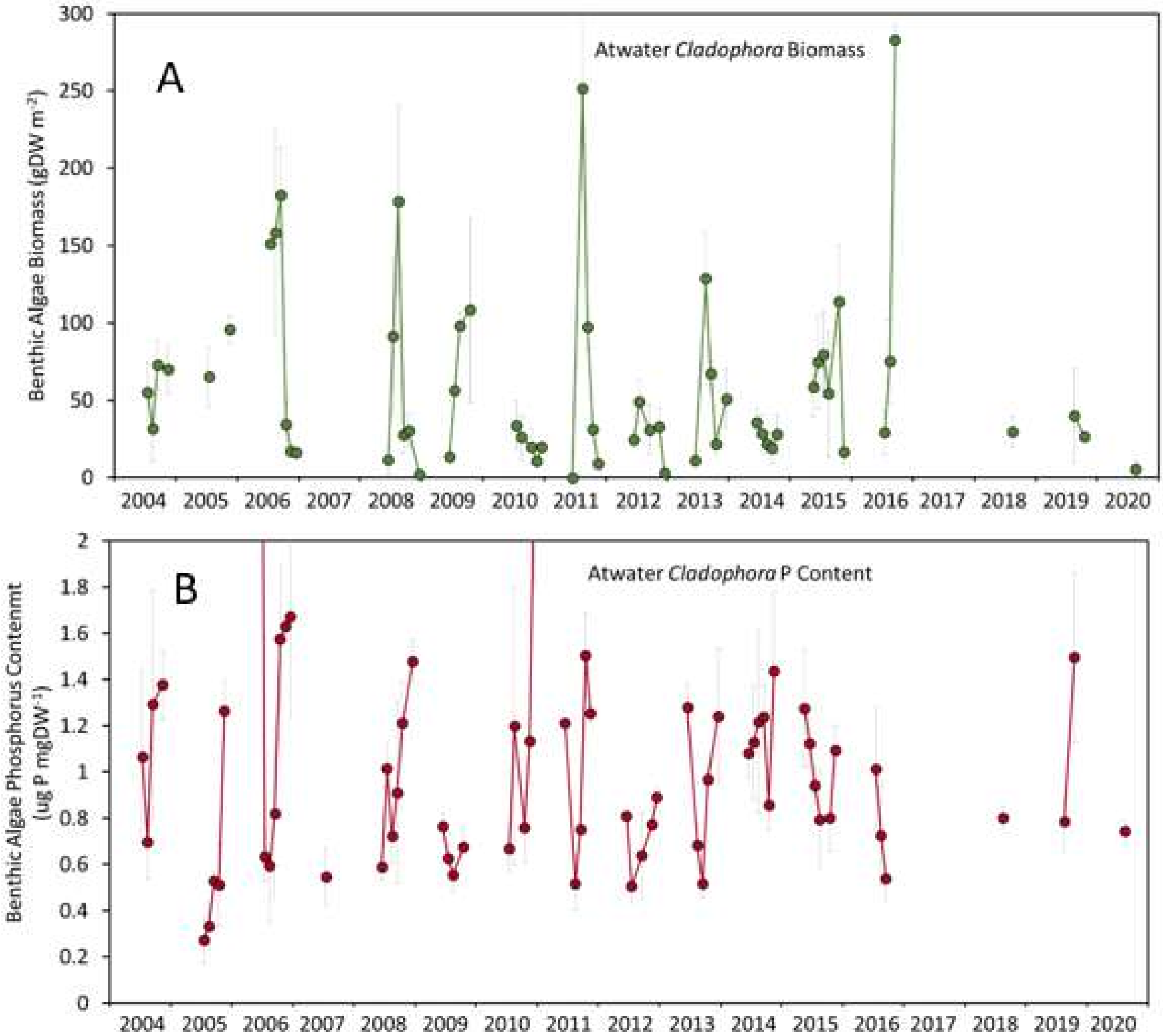

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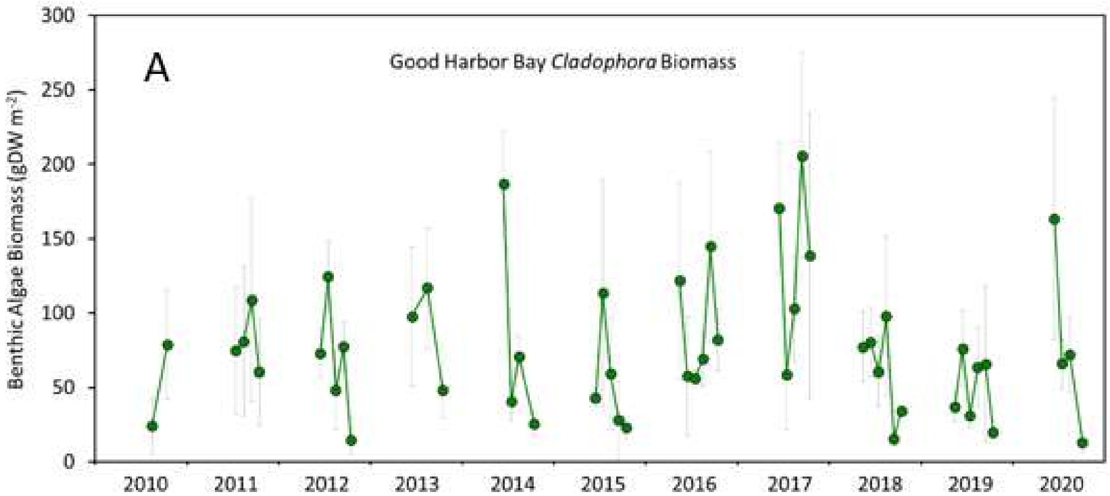

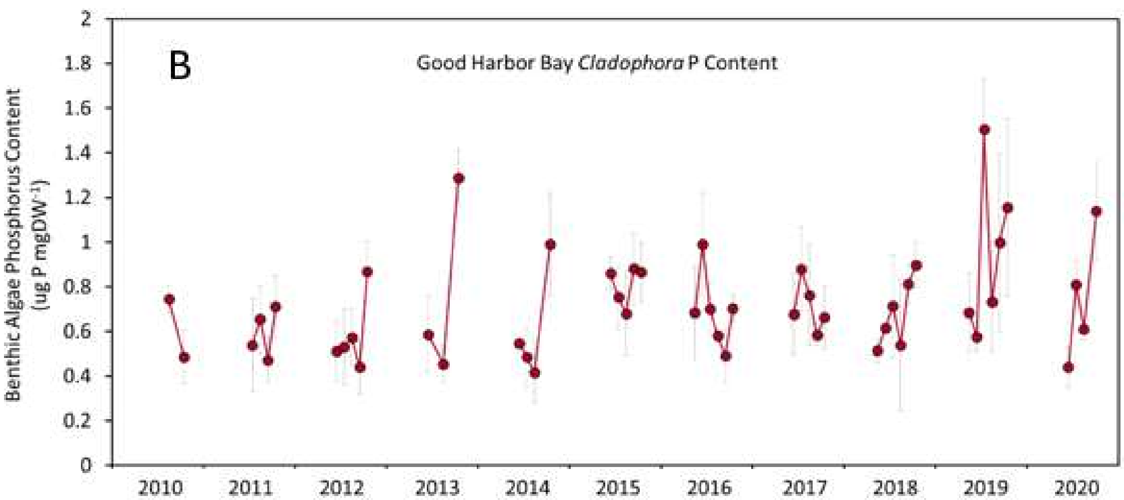

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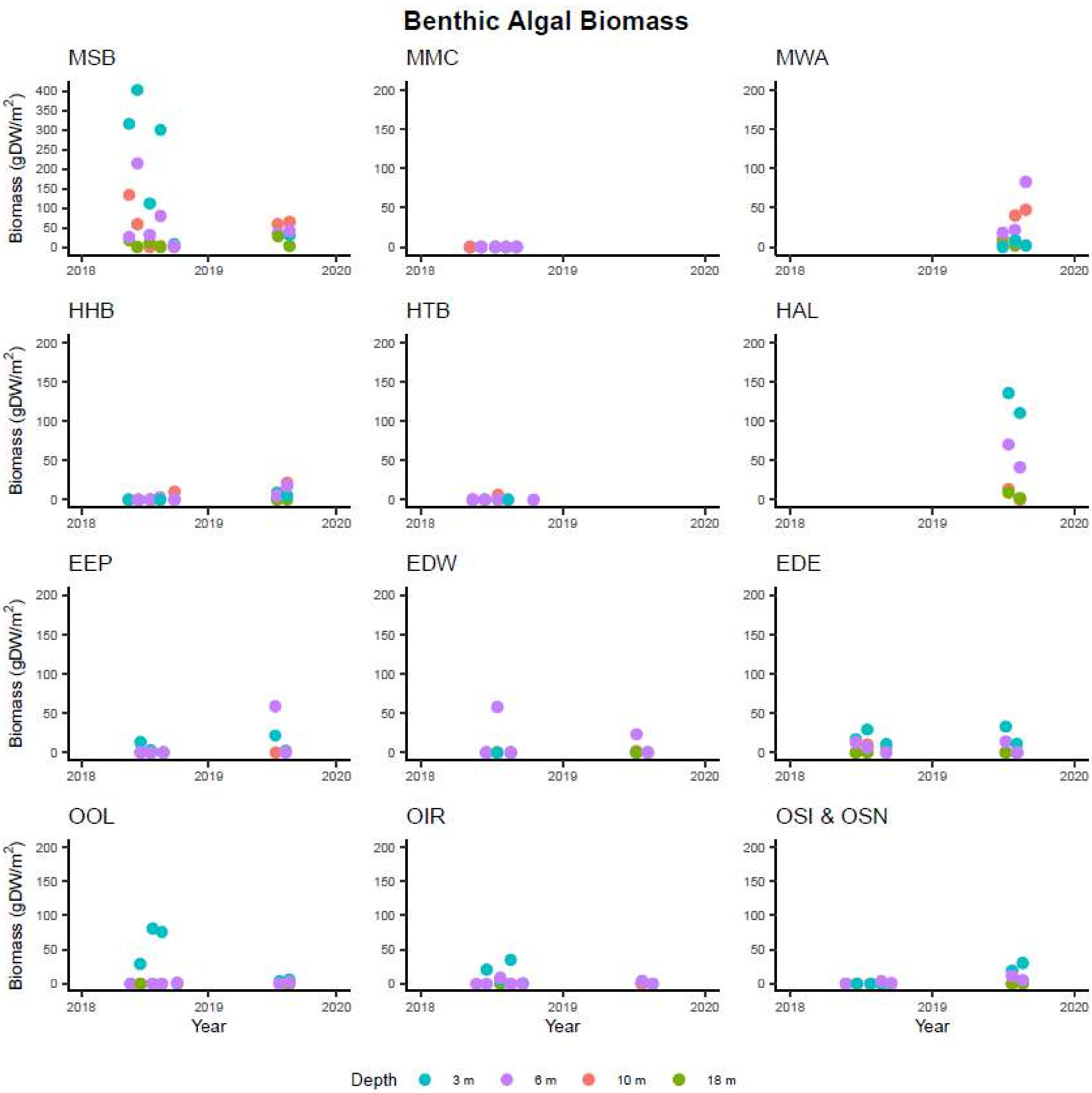

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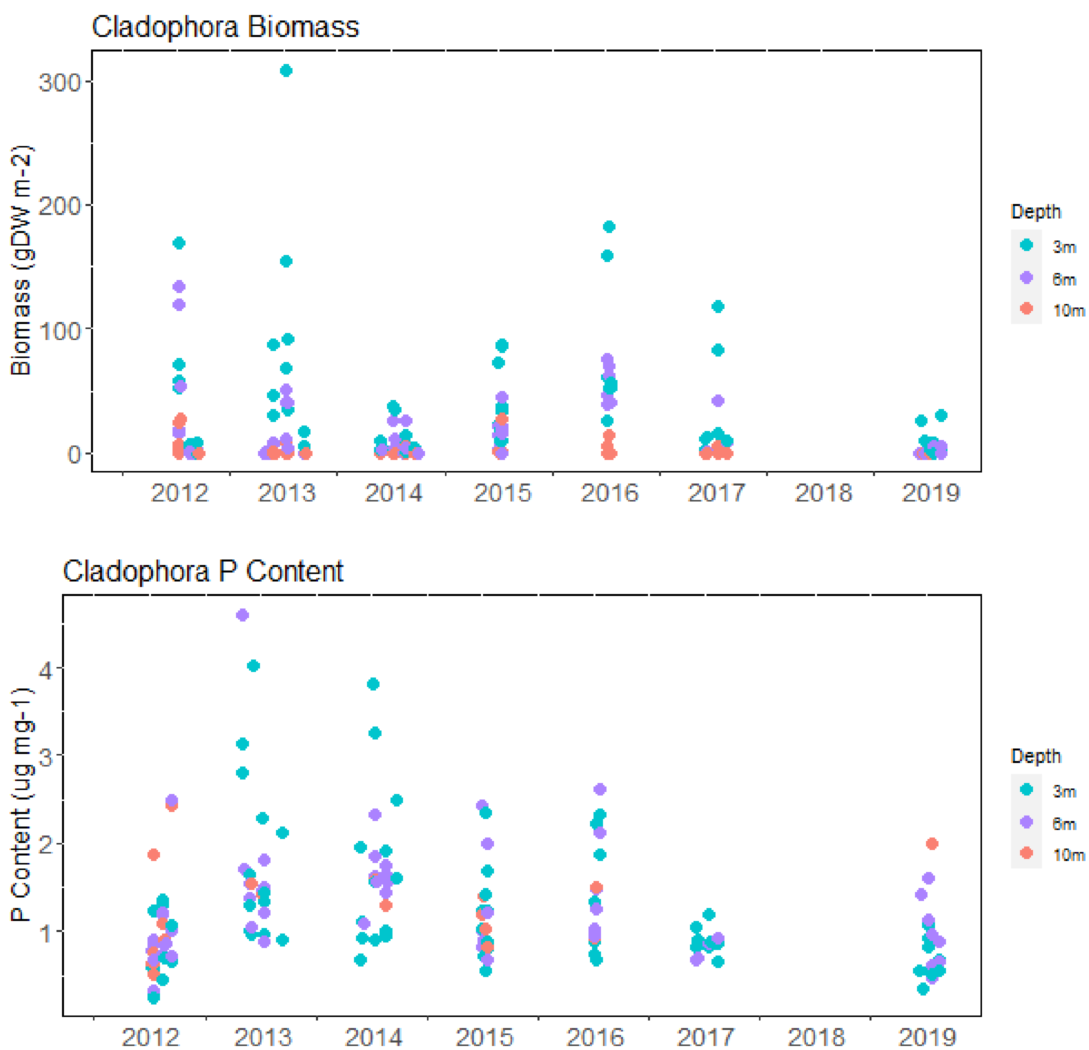

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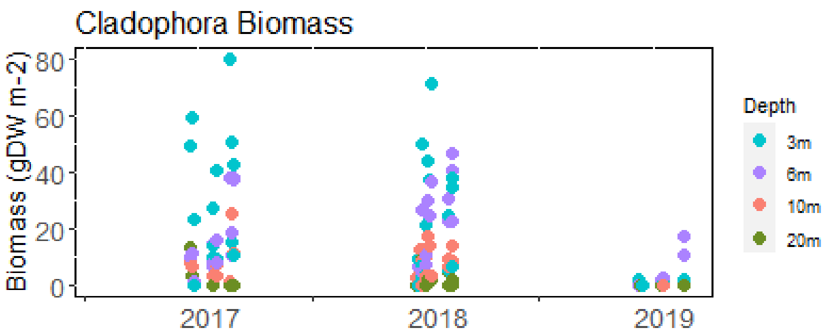

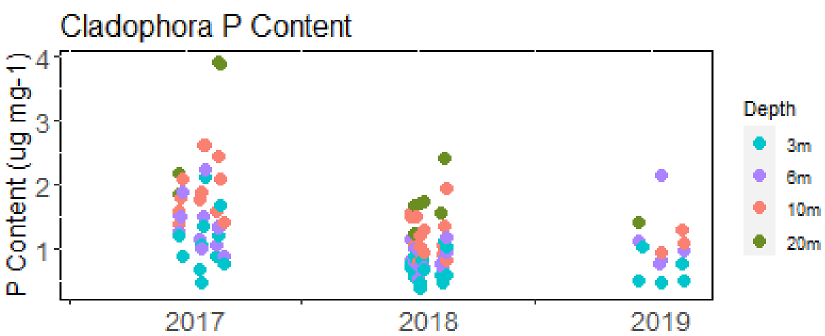

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Sub-Indicator: Rate of New Aquatic Non-indigenous Species (ANS) Establishment

Overall Assessment

Note: The Overall assessment is solely based on new establishments of aquatic non-indigenous species from outside the Great Lakes basin.

Status: Good

10-Year Trend: Unchanging

Long-term Trend (1991-2020): Improving

Rationale: In the past decade (2011-2020), four new aquatic nonindigenous species (ANS) have established over-wintering and reproducing populations in the Great Lakes (Figure 1). *Thermocyclops crassus* (2014) is currently assigned as an unknown vector while *Diaphanosoma fluviatile* (2015) and *Mesocyclops pehpeiensis* (2016) are assigned as 'hitchhikers with organisms in trade' as both are associated with aquatic plants and were previously established in the southern U.S. *Salmincola californiensis* (2014) is assigned as a hitchhiker with stocked fish. This results in a rate of 0.4 new species per year (Figure 2). This rate is significantly lower than the previous two decades and is also significantly lower than the pre-1950 (1837-1949) average rate (Figure 3). Given no new species associated with ballast water have become established since 2006, the long-term trend reflects a dramatic improvement likely associated with the implementation of the regulations for NOBOB (No Ballast on Board) vessels in 2006 and implementation of the joint inspection program to confirm all ships were managing ballast.

Lake-by-Lake Assessment

Note: The Lake-by-Lake assessments below include both establishments introduced from outside the Great Lakes AND inter-basin spread (new establishment to the particular lake basin from populations earlier established in other parts of the Great Lakes basin). These subcomponents – introductions from outside the Great Lakes and interbasin spread are provided separately.

Lake Superior

Status: Poor

(New Establishment of Species from Outside of the Great Lakes Basin: Good; Inter-basin Spread into the Lake Superior Basin: Poor)

10-Year Trend: Improving

Long-Term Trend (1991-2020): Improving

Rationale: No new species have become established in the Lake Superior basin from outside the Great Lakes basin since 2001; with the current decadal rate of new establishments (zero) significantly below the pre-1950 average. All new establishments in Lake Superior during the past decade have been due to spread from the lower lakes with 8 species becoming established in the Lake Superior basin (Figure 4). Within the last decade (2011-2020) the rate of establishment in Lake Superior (including spread from the lower lakes) has undergone a steady decline

(Figure 5); however, the rate of establishment (0.8 species per year) remains more than twice the pre-1950 average of 0.17 species per year (Figure 6).

Lake Michigan

Status: Fair

(New Establishment of Species from Outside of the Great Lakes Basin: Fair; Inter-basin Spread into the Lake Michigan Basin: Fair)

10-Year Trend: Improving

Long-term Trend (1991-2020): Improving

Rationale: No new species (from outside the Great Lakes basin) have become established since 2003.

However, three additional species have spread into Lake Michigan from the other Great Lakes within the last decade (Figure 7); which is not significantly different from the pre-1950 average (0.35 species per year). Within the last decade (2011-2020) the rate of establishment in Lake Michigan (including spread from the other lakes) has undergone a steady decline (Figure 8); however, the current rate (0.3 species per year) is not significantly different from the pre-1950 average rate (Figure 9).

Lake Huron (including St. Marys River)

Status: Poor

(New Establishment of Species from Outside of the Great Lakes Basin: Good; Inter-basin Spread into the Lake Huron Basin: Poor)

10-Year Trend: Undetermined

Long-term Trend (1991-2020): Undetermined

Rationale: No new species have become established in the Lake Huron basin from outside the Great Lakes basin in the last decade. However, spread to Lake Huron of species previously established in the other Great Lakes has resulted in the establishment of 8 additional species in the Lake Huron basin in the last decade (Figure 10). Within the last decade (2011-2020) the rate of establishment in Lake Huron (including spread from the other lakes) cannot be determined with sufficient confidence to report a trend (Figure 11). Likewise, while introductions for the current decade are below the previous decade (2001-2010), they are not significantly below the decade 1991-2000 leaving the overall trend uncertain. Despite lack of confidence in the trend, the overall rate of establishment in Lake Huron (due entirely to spread from the other Lakes) remains more than triple the pre-1950 average of 0.22 species per year (Figure 12).

Lake Erie (including the St. Clair and Detroit River Ecosystem)

Status: Fair

(New Establishment of Species from Outside of the Great Lakes Basin: Poor; Inter-basin Spread into the Lake Erie Basin: Fair)

10-Year Trend: Undetermined

Long-term Trend (1991-2020): Improving

Rationale: Three new species became established in the Lake Erie basin from outside the Great Lakes basin in the last decade (Figure 13), which is higher than the pre-1950 average rate. *Thermocyclops crassus* (2014) is currently assigned as an unknown vector while *Diaphanosoma fluviatile* (2015) and *Mesocyclops pehpeiensis* (2016) are assigned as 'hitchhikers with organisms in trade' as both are associated with aquatic plants and were previously established in the southern U.S. Two additional species expanded their ranges into Lake Erie from the other Great Lakes (lower than the pre-1950 average). Together (new introductions plus spread from the other Great Lakes), the rate of establishment in the Lake Erie basin (0.5 species per year) is not significantly different from the pre-1950 rate (0.58 species per year). Within the last decade (2011-2020) the rate of establishment in Lake Erie (including spread from the other lakes) cannot be determined with sufficient confidence to report a trend (Figure 14). However, comparing the current decadal rate of establishment (2011-2020 = 0.5 species per year) with the previous two decades shows significant improvement with each successive decade (Figure 15), primarily driven by lower rates of spread into Lake Erie from the other Great Lakes.

Lake Ontario (including Niagara River and Headwaters of the St. Lawrence River)

Status: Fair

(New Establishment of Species from Outside of the Great Lakes Basin: Good; Inter-basin Spread into the Lake Ontario Basin: Poor)

10-Year Trend: Unchanging

Long-term Trend (1991-2020): Improving

Rationale: One new species became established in the Lake Ontario basin from outside the Great Lakes basin in the last decade (Figure 16). *Salmincola californiensis* (2014) is assigned as a hitchhiker with stocked fish. The rate of new establishments (from outside the Great Lakes basin) of 0.1 species per year has now fallen well below the pre-1950 average of 0.37 species per year. Six additional species expanded their ranges into the Lake Ontario basin from the upper Great Lakes; the rate of spread into Lake Ontario (0.6 species per year for 2011-2020) remains more than double the pre-1950 rate (0.19 species per year). Within the last decade (2011-2020), the rate of establishment in Lake Ontario (including spread) has remained constant (Figure 17) at an overall rate of 0.7 species per year. Comparing the current decadal rate of establishment with the previous two decades shows significant improvement (Figure 18).

Status Assessment Definitions

Good: Current decadal establishment rate is significantly lower than the pre-1950 average establishment rate.

Fair: Current decadal establishment rate is not significantly different from the pre-1950 baseline establishment rate.

Poor: Current decadal establishment rate is significantly higher than the pre-1950 average establishment rate.

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Decreasing rate of establishment of Aquatic nonindigenous species from outside the basin

Unchanging: No change in rate of establishment of Aquatic nonindigenous species from outside the basin

Deteriorating: Increasing rate of establishment of Aquatic nonindigenous species from outside the basin

Undetermined: Data are unavailable or are insufficient to assess trends, i.e. confidence interval too wide.

Endpoints and/or Targets

The endpoint is a reduction in establishment of new aquatic non-indigenous species and in the current distribution and population size of established Aquatic Invasive Species (AIS). There is a breakpoint in the historic record at 1950, after which time the establishment rate accelerated rapidly (a shift from linear to logarithmic increase), thus the pre-1950 average is used as a benchmark to assess status in this sub-indicator report.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the rate of establishment as a measure of progress towards the prevention of establishment of new non-indigenous species. At the individual lake level, establishment includes the establishment (in the particular lake) of species previously established in the other Great Lakes, which have invaded the lake in question via spread from the other Great Lakes as well as the establishment of species coming from outside the Great Lakes basin.

Ecosystem Objective

The goal of the Great Lakes Water Quality Agreement is to restore and maintain the biological integrity of the Great Lakes Ecosystem. Fundamental to this goal is to prevent further introduction of aquatic invasive species.

This sub-indicator best supports work towards General Objective #7 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from the introduction and spread of aquatic invasive species and free from the introduction and spread of terrestrial invasive species that adversely impact the quality of the Waters of the Great Lakes."

Measure

Prior to the 2019 report, the cumulative number of established non-indigenous species was used as a single sub-indicator for invasive species impacts (identical in format to Figure 1). Beginning with the 2019 report, the sub-indicator report previously called the Impact of Aquatic Invasive Species was separated into two sub-indicator reports: Rate of New ANS Establishment (which considers new establishment from outside of the basin as well as interbasin spread) and Impact of Aquatic Invasive Species which considers the number of species present and their ecological and socioeconomic impacts. This approach is being maintained here with the impacts component being covered in a separate sub-indicator report. However, the SLOPE of Figure 1 (number of new species establishments per year) is the more relevant metric now that rate and impact are separated. Figure 1 is included because it (a) allows direct comparison to earlier reports and (b) shows the historical context for the breakdown of introductions by vector.

For comparison purposes, this sub-indicator still includes the cumulative number of non-indigenous species established in the Great Lakes basin since the mid-1800s. The operational definition of "established" in this report includes overwintering and reproduction as evidenced by at least two consecutive years of reports and reports of all life-history stages. Species with individual reports but no evidence of overwintering and/or reproduction (e.g., tropical and marine species such as flounder, alligator, Chinese mitten crab, and piranha) are introduced species, but not established. These introduced species are excluded from this analysis because reports are not believed to be comprehensive. Grass Carp (*Ctenopharyngodon idella*) has not yet been declared established by the Invasive Carp Regional Coordinating Committee, thus it is NOT included here despite otherwise meeting the definition applied to all the other taxa (overwintering, reproducing as evidenced by at least two consecutive years and report of all life-history stages).

At the whole basin scale, the new (as of 2019) measure for Rate of Establishment is the rate (species per year) of new aquatic non-indigenous species found in the Great Lakes. Species are included only after evidence of overwintering and reproduction are verified, but the original discovery dates are used for consistency. Decadal rates of establishment are calculated and used as a direct metric. Currently, no species have been successfully eradicated from the entire basin, but eradication rate should be considered as an accompanying measure when relevant.

Lake-by-lake assessments were NOT included in the final 2019 report due to difficulty with interpretation and wide confidence intervals obscuring trends. These are included in this report. Each Lake-basin is treated as a discrete unit and analyzed independently using the same method as employed for the overall basin. We find this is the more meaningful boundary from the perspective of the status of the particular Lake. Partitioning by vector (e.g. ballast water, deliberate release, canal, etc.) can only be done for the establishments coming from outside the Great Lakes – there is not currently consistent data attributing species movements between lakes to particular vectors. Individual vectors are thus not analyzed at the Lake-basin scale, however we do include the breakdown of which species represent new introductions from outside the Great Lakes versus which represent inter-basin spread (new introductions to the particular lake basin from populations previously in other parts of the Great Lakes basin). Where possible, the distinction between rate of establishment due to new introductions (managed via prevention strategies) and due to rate of inter-basin spread (managed through control strategies) is called out for each lake basin, though the lake assessment is based on the sum of these components.

This report uses the binational standardized data from the Great Lakes Aquatic Non-indigenous Species Information System (GLANSIS).

Ecological Condition

Background

The National Oceanic and Atmospheric Administration (NOAA) reports a total of 188 overwintering and reproducing Great Lakes aquatic nonindigenous species (ANS), plus 26 species native to some part of the basin which have expanded their ranges to other parts (source: GLANSIS, accessed December 2020). This data includes all verified reports from EDDMaps, iMapInvasives and other state-specific and institutional databases that are publicly available. Data for species expanding their ranges is not included in the basin-wide assessment (e.g., Rusty crayfish is indigenous to the Great Lakes because it is indigenous to Lake Erie) but ARE included in the individual lake assessments where relevant (e.g., Rusty crayfish is non-indigenous to Lake Superior). Spread within a lake is not included in this sub-indicator (e.g, rusty crayfish is considered native to Lake Erie and not included even though its native range may be limited to the western basin). While the GLANSIS database has a distinct US-bias at sub-watershed scales, every effort has been made to verify and include any earlier discovery dates reported for Canada in the peer-reviewed literature at the individual lake and whole-basin scales.

In the Great Lakes, transoceanic ships (including solid ballast, packing materials, ballast water and ballast residuals) have been the primary vector responsible for 42% of the total established ANS. Historically, deliberate introductions (stocking fish and agricultural/horticultural plants) have also been a significant vector (22%). Other notable vectors include hitchhikers with organisms in trade (12%), aquarium releases (4%), canals (4%), escapes from culture (4%), bait (1%) – despite best efforts to attribute ANS to particular sources, 10% of the species remain unattributed (unknown vector).

During the 1980s, the importance of ship ballast water as a vector for ANS introductions was recognized, prompting ballast management measures in the Great Lakes. In the wake of Eurasian ruffe and zebra mussel introductions, Canada introduced voluntary ballast exchange guidelines in 1989 for ships declaring "ballast on board" (BOB) following transoceanic voyages; this action followed recommendations by the Great Lakes Fishery Commission and the International Joint Commission. In 1990, the United States Congress passed the Non-indigenous Aquatic Nuisance Prevention and Control Act, producing the Great Lakes' first ballast exchange and management regulations in May of 1993. The National Invasive Species Act (NISA) followed in 1996. Following initiation of voluntary guidelines in 1989 and mandated regulations in 1993, the overall rate of Great Lakes invasion did not decline immediately (Grigorovich et al. 2003; Holeck et al. 2004; Ricciardi 2006). However, more than 90% of transoceanic ships that entered the Great Lakes during the 1990s declared "no ballast on board"

(NOBOB; Colautti et al. 2003; Grigorovich et al. 2003; Holeck et al. 2004) and were not required to exchange ballast, despite their tanks containing residual sediments and water that could be discharged in the Great Lakes. Residual water and sediment in these ships were found to contain several species previously unrecorded in the basin; such species could be discharged after the ship undergoes sequential ballasting operations as it travels between ports within the Great Lakes to offload and take on cargo (Duggan et al. 2005, Ricciardi and MacIsaac 2008). In June 2006, Canada implemented new regulations for the management of residuals contained within NOBOB tanks and requires the salinity of all incoming ballast water to be at least 30 parts per trillion (Government of Canada 2006). In the decade since, there have been no new ballast water ANS introductions (the last being *Hemimysis anomala*, collected in May 2006) despite a fairly steady number of NOBOB transits; in comparison there were 5 new ANS establishments attributed to this vector in the previous decade (2001-2010) and 12 in the decade prior to that (1991-2000). Multiple lines of evidence indicate that ballast water regulation is largely responsible for the rapid reduction in the rate of introduction after 2006 (Ricciardi et al 2021, Sturtevant et al 2019). No other equivalent period of time in the documented history of the Great Lakes basin since 1835 has had fewer new ANS introduced from outside the basin than the period of 2007–2020. However, there have been reports of new ANS in the Great Lakes which may have been introduced by ballast water for which population status is unknown (e.g. *Brachionus leydigii* (2016), *Paraleptastacus wilsoni* (2017) (Connolly 2016, Cangelosi et al. 2018, respectively). Canada and the U.S. EPA have recently introduced requirements for the use of ballast water treatment systems, in addition to ballast water exchange, in an effort to further prevent new ANS introductions by transoceanic shipping. The Canadian regulations also require the use of treatment systems for management of ballast water being moved within the Great Lakes basin (phasing in by 2030) to reduce inter-basin spread of ANS (Ballast Water Regulations (SOR/2021-120) Canada Gazette, Part 2, Volume 155, Number 13: Ballast Water Regulations).

Second to shipping, deliberate release, transfer, and escape have introduced ANS into the Great Lakes. Species introduced within the last decade (from outside the Great Lakes) have most likely been inadvertently introduced with stocked fish or plants. The fish parasite *Salmincola californiensis* (2014) is attributed as a hitchhiker with stocked fish. *Thermocyclops crassus* (introduced in 2014) is currently assigned as an unknown vector, but considered unlikely to have invaded via transoceanic ballast (given prior establishment in Lake Champlain) while *Diaphanosoma fluviatile* (introduced 2015) and *Mesocyclops pehpeiensis* (introduced 2016) are assigned as hitchhikers with aquatic plants. Of particular continuing concern are private sector activities related to aquaria, garden ponds, baitfish, and live food fish markets. These 'emerging' vectors are of concern not only for the organisms directly in trade, but also for hitchhikers, parasites and diseases that may be travelling with them.

Even with the successes of ballast management regulations in preventing new introductions from outside the Great

Lakes basin, the number of established non-indigenous species in each lake continues to increase as species spread from lake-to-lake. Currently 33% non-indigenous species established in the Great Lakes have spread to all 5 lakes – 67% of the ANS in the Great Lakes thus still pose a significant risk for additional spread. Over the historic record, Lake Superior has received 86% of its new invaders from the other 4 lakes, Lake Huron 91%, Lake Michigan 73%, Lake Erie 64%, and Lake Ontario 55%.

Inter-basin transport results in a time lag effect in which individual lakes continue to accumulate new species and have high overall establishment rates despite the decline in the rate of introductions to the whole Great Lakes basin and declines in introductions from outside the region to particular lakes. Lakes Superior, Huron and Ontario all have establishment rates (new introductions plus inter-basin spread) which remain above the respective pre-1950 average rates despite low rates of new introductions from outside the region with the remaining rate driven by inter-basin spread. Lake Michigan has returned to establishment rates approximately equal to the pre-1950 average (both new introductions and spread). The overall rate of establishment in Lake Erie (new introductions plus inter-basin spread) has also returned to a rate approximately equal to the pre-1950 average, although new introductions remain approximately 50% higher than pre-1950.

Ballast water movement between lakes within the Great Lakes basin, which was not regulated prior to 2021, may pose a relatively high risk of spreading ANS (Casas-Monroy et al. 2014, DFO 2019) between lake basins. However other vectors such as recreational boating, water gardening, etc. also are contributing to spread (Drake et al 2017). Species crossing basin boundaries in the last decade are not dominated by any particular taxa and were not entirely species originally introduced via ballast.

Evidence indicates that newly invading species may benefit from the presence of previously established species (Ricciardi 2001). For example, *Echinogammarus ischnus* (amphipod) have thrived in the presence of previously established zebra (*Dreissena polymorpha*) and quagga mussels (*Dreissena bugensis*) (Stewart et al 1998). In effect, dreissenids have set the stage to increase the number of successful invasions, particularly those of co-evolved species from the same home region (the Ponto-Caspian assemblage). This result may be a critical factor contributing to the continued spread of species across lakes within the Great Lakes system.

Until recently, most of the invasive carp (bighead, silver, black and grass carp) sold at fish markets within the Great Lakes basin were sold live. All eight Great Lakes states and the province of Ontario now have some restriction on the sale of live invasive carp. Enforcement of many private transactions, however, remains a challenge. The U.S. Fish and Wildlife Service published a final rule in March 2011, officially adding the bighead carp to the federal injurious wildlife list and codifying the Asian Carp Prevention and Control Act. Bighead, silver, and black carp are now listed as nuisance species under the Lacey Act, prohibiting interstate transport. There are currently numerous shortcomings in legal safeguards relating to commerce in non-native live fish in Great Lakes and Mississippi River states, as well as Quebec, and Ontario, as identified by Alexander (2003); though recent regulations are beginning to close gaps (CBSA Memorandum D19-8-5). These include: express and de facto exemptions for the aquarium pet trade; de facto exemptions for the live food fish trade; inability to proactively enforce import bans; lack of inspections at aquaculture facilities; allowing aquaculture in public waters; inadequate triploidy (sterilization) requirements; failure to regulate species of concern (e.g., invasive carp); regulation through “dirty lists” only (e.g., banning known nuisance species); and failure to regulate transportation. Silver and bighead carp escapees from southern United States fish farms have developed large populations in the middle and lower segments of the Illinois River, which connects the Mississippi River to Lake Michigan via the Chicago Sanitary and Ship Canal (CSSC). A prototype electric barrier on the CSSC was activated in April 2002 to block the transmigration of species between the Mississippi River system and the Great Lakes basin. The U.S. Army Corps of Engineers (partnered with the State of

Illinois) completed construction of the second and third permanent barriers in 2005 and 2011, respectively. Since 2009, environmental DNA (eDNA) surveillance has been used to complement the use of traditional monitoring and suppression tools. Between 2009 and 2010, DNA of both bighead and silver carp was detected past the electric barriers; however, only a single bighead carp was subsequently found (Lake Calumet, June 2010). No additional bighead or silver carp have been confirmed above the electric barrier. Triploid grass carp (sterile) have been reported in the Great Lakes since the early 1970s, diploid have occasionally been captured since ~2011. New evidence indicates that grass carp are spawning and recruiting in the Sandusky and Maumee Rivers (tributaries to western Lake Erie) (Wieringa et al 2017, Kocovsky et al 2019, Chapman et al 2020). However, the Invasive Carp Regional Coordinating Committee (ICRCC) has not yet declared grass carp as established in the Great Lakes. Grass carp, Bighead carp, Silver carp, and Black carp are thus all excluded from this analysis as they are not established in the Great Lakes.

Status

The total number of non-indigenous species introduced and established in the Great Lakes increased steadily from the 1830s to 2006, but has stabilized in the last decade (Figure 1). Although there have been 37 species established since the signing of the 1987 GLWQA, only four new species have become established since 2006. In the past decade (2011-2020), these four new aquatic nonindigenous species (ANS) have established overwintering and reproducing populations in the Great Lakes. *Thermocyclops crassus* (2014) is currently assigned as an unknown vector (possibly ballast direct from Europe or possibly with recreational boating from Lake Champlain where it was reported in the early 1990s) while *Diaphanosoma fluviatile* (2015) and *Mesocyclops pehpeiensis* (2016) are assigned as 'hitchhikers with organisms in trade' as both are associated with aquatic plants and were previously established in the southern U.S. *Salmincola californiensis* (2014) is assigned as a hitchhiker with stocked fish. This results in a rate of 0.4 new species per year (Figure 2). This rate is significantly lower than the previous two decades and is also significantly lower than the pre-1950 (1837-1949) average rate (0.8 new species per year) (Figure 3). The long-term trend reflects a dramatic improvement; given no new species associated with ballast water have been introduced since 2006 this improvement is likely associated with the implementation of the regulations for NOBOB (No Ballast on Board) vessels in 2006 and implementation of the joint inspection program to confirm all ships were managing ballast.

Figure 19 depicts the location of each new introduction (earliest report) to the Great Lakes (excluding early introductions that were already widespread at the time of discovery).

Lake-by-Lake Assessments were completed by treating each Lake basin as a discrete bounded unit. Establishments in a lake basin include BOTH the new introductions from outside the Great Lakes basin and spread of species into the Lake basin from the other Great Lakes as both affect the status of a lake basin. A distinction between the two types of establishment are maintained as they are addressed through separate management objectives (prevention versus control).

As noted above, only four new species in total have become established in the Great Lakes basin in the last decade. *Salmincola californiensis*, (a fish parasite) was first detected in Lake Ontario in 2014. *Thermocyclops crassus* (2014), *Diaphanosoma fluviatile* (2015), and *Mesocyclops pehpeiensis* (2016) are all warm-water zooplankton first detected in Lake Erie. See Table 1 for a full list of species that contributed to establishment of new species and inter-basin spread in the last decade.

Phragmites australis subsp. australis is not included in the lake-by-lake analyses as early records fail to distinguish native and invasive strains leading to uncertainty with regard to earliest dates at the lake-basin scale. This species was established in all basins before 2011, so does not substantively affect the analysis of the current decade.

In Lake Superior, eight (8) new non-indigenous species have become established in the last decade (2011-2020). These species are all secondary invasions from populations established first in the lower lakes that have spread to Lake Superior. No new species have become established in the Lake Superior basin from outside the Great Lakes basin since 2001. However, spread to Lake Superior of species previously established in the lower Great Lakes has resulted in the establishment of 8 additional species in the Lake Superior basin in the last decade (Figure 4). Within the last decade (2011-2020) the rate of establishment in Lake Superior (including spread from the lower lakes) has undergone a steady decline (Figure 5) and is once again approaching a rate (0.8 species per year) comparable to the late 1980s (which is significantly lower than peak of 3 species per year ~2004) but not yet returned to pre-1950s establishment rate (0.2 species per year). Separating the overall rate into components, the current decadal rate of new establishments from outside the Great Lakes (zero) is significantly below the pre-1950 average, but the rate of establishment due to spread from the lower lakes remains more than twice the pre-1950 average of 0.17 species per year (Figure 6).

In Lake Michigan, three species – *Hydrocharis morsus ranae* (2016), *Thermocyclops crassus* (2017), and *Diaphanosoma fluviatile* (2018) have spread into Lake Michigan from populations introduced to other portions of the Great Lakes (Figure 7); but no new species have become established directly from outside the Great Lakes basin since 2003. Within the last decade (2011-2020) the rate of establishment in Lake Michigan (including spread from the other lakes) has undergone a steady decline (Figure 8). The current rate of establishment (no species representing new introductions from outside the basin and 0.3 species per year spread from other portions of the Great Lakes) are not significantly different from the pre-1950 rates.

In Lake Huron, no new species have become established directly from outside the Great Lakes basin since 1994. However, spread to Lake Huron of species previously established in the other Great Lakes has resulted in the establishment of 8 additional species in the Lake Huron basin in the last decade (Figure 10). Within the last decade (2011-2020) the rate of establishment in Lake Huron (including spread from the other lakes) cannot be determined with sufficient confidence to report a trend (Figure 11). Likewise, while establishments for the current decade are below the previous decade (2001-2010), they are not significantly below the decade 1991-2000, leaving the overall trend uncertain. Separating the overall rate into components, the current decadal rate of new establishment from outside the Great Lakes basin (zero) is significantly below the pre-1950 average (0.05 species per year), but the rate of establishment due to spread from the lower lakes remains more than triple the pre-1950 average of 0.22 species per year (Figure 12).

In Lake Erie (including Lake St. Clair, Detroit River and St. Clair River), three new species were established directly from outside the Great Lakes basin in the last decade. *Thermocyclops crassus* (2014) is currently assigned as an unknown vector while *Diaphanosoma fluviatile* (2015) and *Mesocyclops pehpeiensis* (2016) are assigned as 'hitchhikers with organisms in trade' as both are associated with aquatic plants and were previously established in the southern U.S. Two additional species expanded their ranges into Lake Erie from the other Great Lakes including *Neoergasilus japonicus* (2011), and *Cirsium palustre* (2013) (Figure 13). Within the last decade (2011-2020) the rate of establishment in Lake Erie (including spread from the other lakes) cannot be determined with sufficient confidence to report a trend (Figure 14). However, comparing the current decadal rate of establishment (2011-2020 = 0.5 species per year) with the previous two decades shows significant improvement with each successive decade (Figure 15), primarily driven by lower rates of spread into Lake Erie from the other Great Lakes. The rate of establishment (0.3 species per year) due to new introductions to the Lake Erie basin from outside the Great Lakes basin remains greater than the pre-1950 rate (0.19 species per year), while the rate of establishment due to spread into the Lake Erie basin from the other Great Lakes (0.2 species per year) is significantly lower than the pre-1950 rate (0.39 species per year). Together, these two factors result in an overall rate of establishment in Lake Erie of 0.5 species per year which is not significantly different from the pre-1950 average of 0.58 species per year.

In Lake Ontario, only one new species became established directly from outside the Great Lakes basin in the last decade. *Salmincola californiensis* (2014) is assigned as a hitchhiker with stocked fish (Figure 19). Six additional species expanded their ranges into Lake Ontario from the upper Great Lakes: *Ictiobus cyprinellus* (2015), *Procambarus acutus acutus* (2017), *Neoergasilus japonicus* (2018), *Heteropsyllus nr. nunni* (2018), *Schizopera borutzkyi* (2018), and *Proterorhinus semilunaris* (2019). Within the last decade (2011-2020), the rate of establishment in Lake Ontario (including spread) has remained constant (Figure 20) at an overall rate of 0.7 species per year. Comparing the current decadal rate of establishment with the previous two decades shows significant improvement (Figure 21), primarily driven by lower rates of spread into Lake Ontario from the other Great Lakes coupled with a slightly earlier decrease in new establishments from outside the Great Lakes basin. The rate of establishment due to new introductions (from outside the Great Lakes basin) of 0.1 species per year has now fallen well below the pre-1950 average of 0.37 species per year, but the rate of inter-basin spread into Lake Ontario (0.6 species per year for 2011-2020) remains more than double the pre-1950 rate (0.19 species per year).

Linkages

- This sub-indicator also links directly to the other sub-indicators in the Invasive Species category, particularly Impacts of AIS, Sea Lamprey, and Dreissenid Mussels.
- Aquatic Habitat Connectivity – the potential for AIS to colonize new locations is increased with removal of dams. In contrast, ecological separation of the Great Lakes from the Mississippi River basin is being discussed as a way to limit transfer of aquatic nonindigenous species (ANS) and aquatic invasive species (AIS) between these basins.
- Surface Water Temperature – Higher temperatures may be related to the spread of some aquatic non-native species.
- Multi-stressors: Changes in water quality, global climate change, and land use also may make the Great Lakes more hospitable for the establishment of new aquatic nonindigenous species. Climate changes may also be facilitating the northward spread of both non-native species and the spread of native species into adjacent habitats to which they are not native (e.g., range expansion). Increasing lake temperatures associated with climate change will lead to increased potential for ANS introduced from warmer climates to establish overwintering populations (Adebayo et al. 2011; Mandrak 1989). The rate of establishment may increase if positive interactions involving established ANS or native species facilitate the establishment of new ANS.

Traditional Ecological Knowledge (TEK), Citizen Science and/or other Bodies of Knowledge

While this report is based only on verified reports, many of these reports were first made by observant citizens. Citizen scientists play a key role important role in helping to map the invasion front and the spread of species to new watersheds. Verified reports from EDDMaps, iMap Invasives, GLIFWC and other Citizen Science AIS monitoring initiatives are included in the GLANSIS database and in turn, included in this analysis.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	x			
Data are from a known, reliable and respected generator of data and are traceable to original sources	x			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	x			
Data obtained from sources within the U.S. are comparable to those from Canada		x		
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	x			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://www.glerl.noaa.gov/glansis/		

All raw data and specific lake basin lists are available through GLANSIS at https://www.glerl.noaa.gov/glansis/. Spreadsheets including data analyses are available from the lead author.

Data Limitations

All 'first dates' used in this report are dates of first discovery for the individual lake basin and Great Lakes basin scales as relevant to the particular analysis. Monitoring effort influences the temporal lag between actual time of introduction and date of first discovery. Furthermore, there is often a delay between the time in which the new ANS is discovered and the date at which a species is documented to be established (sufficient evidence gathered to prove the species is overwintering and reproducing requires a minimum of 2 years, often much longer). In some cases, the interval between likely introduction (with known opening of a canal, for example) and the date at which establishment is concluded can exceed 20 years, in others the interval is much shorter. Dates of 1st discovery are used for consistency and because they are the least subjective numbers available. In a handful of cases, date of first discovery may reflect a failed introduction that did not directly lead to the established population, but still indicates a vector was in place that eventually led to introduction and establishment.

This sub-indicator can be biased by sampling effort. It is possible to see a change in 'number of reported species' (what can be measured) that is lower than 'number of introductions' simply by not looking for them or conversely, an increased number of reported species may be an artefact of an investment in monitoring (they were introduced previously, but not reported).

More than 90% of the historic data in GLANSIS is from the U.S. This consideration does not impact the discovery dates used for the rate calculations at the Great Lakes basin scale nor at individual lake basin scales as every effort was made to obtain earliest records from Canada at the Lake-basin scale. However, lack of equivalent historic Canadian data may contribute to time lags.

Additional Information

Aquatic nonindigenous species have invaded the Great Lakes basin from regions around the globe. Increasing world trade and travel elevates the risk that additional species will continue to gain access to the Great Lakes. Existing connections between the Great Lakes watershed and systems outside the watershed, such as the Chicago Sanitary and Ship Canal, and growth of industries such as aquaculture, live food markets, and aquarium retail stores will also increase the risk that new ANS will be introduced. New vectors may arise as the face of industry in the region changes.

Studies suggest each of the Great Lakes may differ in vulnerability to introduction and establishment. Healthy, intact ecosystems tend to be less vulnerable while degraded ecosystems tend to be more susceptible. Previous establishment of non-indigenous species may facilitate the establishment of new ones.

This report draws attention to the various vectors of introduction including those related to shipping (i.e. ballast water and canals) and additional private sector activities not limited to aquaria and live fish markets. There exists variability in the distribution of these vectors between different lakes. The variation of vector distribution and magnitude also influences the vulnerability of a lake to new nonindigenous species.

Data on range expansion populations (those native or cryptogenic to a portion of the basin but introduced to other areas of the basin) is currently still lacking – GLANSIS tracks only 26 such species (mostly those that invaded the upper lakes via the Welland Canal). More monitoring data will be needed to assess potential expansion of these populations due to climate change. For example, rusty crayfish is considered native to Sandusky Bay, therefore it is not included in the calculations for the overall basin. It is, however, non-indigenous to 4 of the 5 individual lake basins and so it is included as a non-indigenous species for those 4 lakes. *Phalaris arundinacea* (originally a rare native found throughout the basin) is an unusual case in which a native species has been heavily influenced by genetic contamination by a European strain and the hybrid is behaving as an invasive; this species has been excluded from the analysis entirely at this time. *Ictiobus niger* and *Phenacobius mirabilis* are also excluded due to uncertainty related to native status. *Actinocyclus normanii fo. subsalsa* has recently been found in EPA cores dating as early as 1709, calling into question the native status for the lower lakes; we continue to include this species only in the analysis for Lake Superior, treating it as a range expansion into Lake Superior (1978) from potentially native populations in the lower lakes. *Mentha x gracilis*, previously treated as a separate species, has been confirmed as a hybrid of *M. spicata* and so is no longer accounted separately. *Phragmites australis subsp. australis* is included in the overall Great Lakes analysis with a confirmed introduction date of 1869 in Lake Erie. However we have been unable to confirm earliest dates for the other lake-basins as early records fail to distinguish native and invasive strains so we exclude this species from the individual lake analyses.

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List of Tables

Source: GLANSIS

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Figure 11. Running decadal establishment rate of aquatic non-indigenous species in Lake Huron for 2011-2020 with 95% confidence intervals (regardless of origin).

Source: GLANSIS

Figure 12. Decadal establishment rates of new aquatic non-indigenous species from outside of the Great Lakes basin (blue), and aquatic non-indigenous species spread from the other Great Lakes (red) for the period of record for Lake Huron. Horizontal solid lines represents the pre-1950 average rate of establishment.

Source: GLANSIS

Figure 13. Cumulative established aquatic non-indigenous species in Lake Erie including new introductions (blue) and inter-basin spread (red). 154 aquatic non-indigenous species in total have become established in Lake Erie as of 2020.

Source: GLANSIS

Figure 14. Running decadal establishment rate of aquatic non-indigenous species in Lake Erie for 2011-2020 with 95% confidence intervals (regardless of origin).

Source: GLANSIS

Figure 15. Decadal establishment rates of new aquatic non-indigenous species from outside of the Great Lakes basin (blue), and aquatic non-indigenous species spread from the other Great Lakes (red) for the period of record for Lake Erie. Horizontal solid lines represents the pre-1950 average rate of establishment.

Source: GLANSIS

Figure 16. Cumulative established aquatic non-indigenous species in Lake Ontario including new introductions (blue) and inter-basin spread (red). 131 aquatic non-indigenous species in total have become established in Lake Ontario as of 2020.

Source: GLANSIS

Figure 17. Running decadal establishment rate of aquatic non-indigenous species in Lake Ontario for 2011-2020 with 95% confidence intervals (regardless of origin).

Source: GLANSIS

Figure 18. Decadal establishment rates of new aquatic non-indigenous species from outside of the Great Lakes basin (blue), and aquatic non-indigenous species spread from the other Great Lakes (red) for the period of record for Lake Ontario. Horizontal solid lines represents the pre-1950 average rate of establishment.

Source: GLANSIS

Figure 19: Points of introduction of aquatic non-indigenous species to the Great Lakes basin.

Last Updated
State of the Great Lakes 2022 Report

Table 1. Species contributing to establishment and inter-basin spread in the last decade. Source: GLANSIS

Species	Date of 1st Introduction to Great Lakes basin	Likely vector of 1st Introduction to the Great Lakes Basin	Basin of 1st Introduction (or native range)	Basins spread to in 2011-2020
Faxonius immunis	Native		Michigan, Huron, Erie, Ontario	Superior
Esox niger	Native		Ontario	Huron
Procambarus acutus acutus	Native		Michigan, Erie	Ontario
Didymosphenia geminata	Native*		Superior, Michigan	Huron
Butomus umbellatus	1905	Planted	Erie	Superior
Najas minor	1932	Hitchhiker with organisms in trade	Erie	Huron
Cirsium palustre	1934	Unknown	Michigan	Erie
Lupinus polyphyllus	1959	Planted	Superior	Huron
Ictiobus cyprinellus	1962	Stocked	Michigan	Ontario
Nitokra hibernica	1972	Ballast	Ontario	Superior
Hydrocharis morsus-ranae	1972	Escaped cultivation	Ontario	Michigan, Huron
Glyceria maxima	1979	Planted	Michigan	Superior, Huron
Corbicula fluminea	1980	Unknown	Erie	Huron
Schizopera borutzkyi	1988	Ballast	Michigan	Superior, Ontario
Proterorhinus semilunaris	1990	Ballast	St. Clair	Ontario
Potamopyrgus antipodarum	1991	Ballast or Hitchhiker	Ontario	Huron
Neoergasilus japonicus	1994	Unknown	Huron	Erie, Ontario
Heteropsyllus nr. nunni	1996	Ballast	Michigan	Ontario
Hemimysis anomala	2006	Ballast	Ontario	Superior
Thermocyclops crassus	2014	Unknown (from Lake Champlain?)	Erie	Superior, Michigan
Salmincola californiensis	2014	Hitchhiker with hatchery fish	Ontario	
Diaphanosoma fluviatile	2015	Hitchhiker with ornamentals or recreational boats	Erie	Superior, Michigan
Mesocyclops pehpeiensis	2016	Hitchhiker with ornamental plants	Erie	

*There may be separate native and invasive strains of this species.

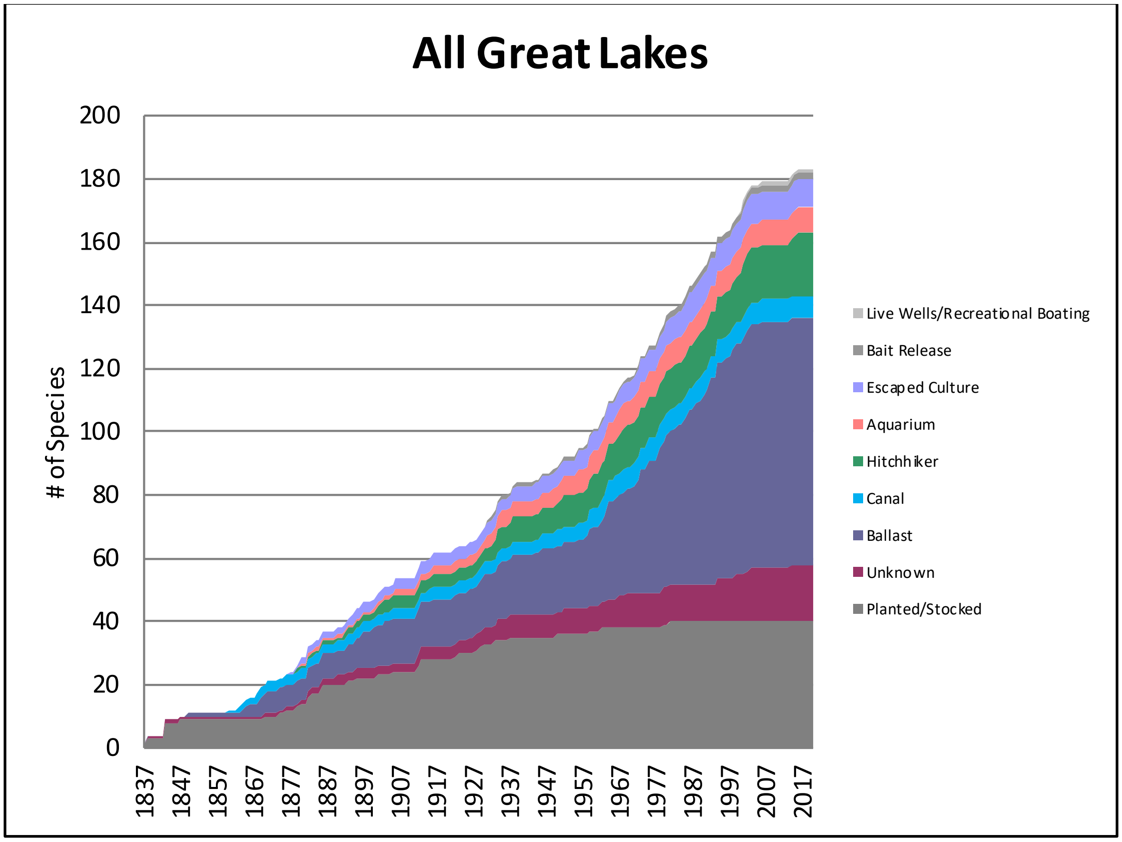


Figure 1. Cumulative discovery of established aquatic non-indigenous species to the Great Lakes basin by vector (1837-2020). Source: GLANSIS.

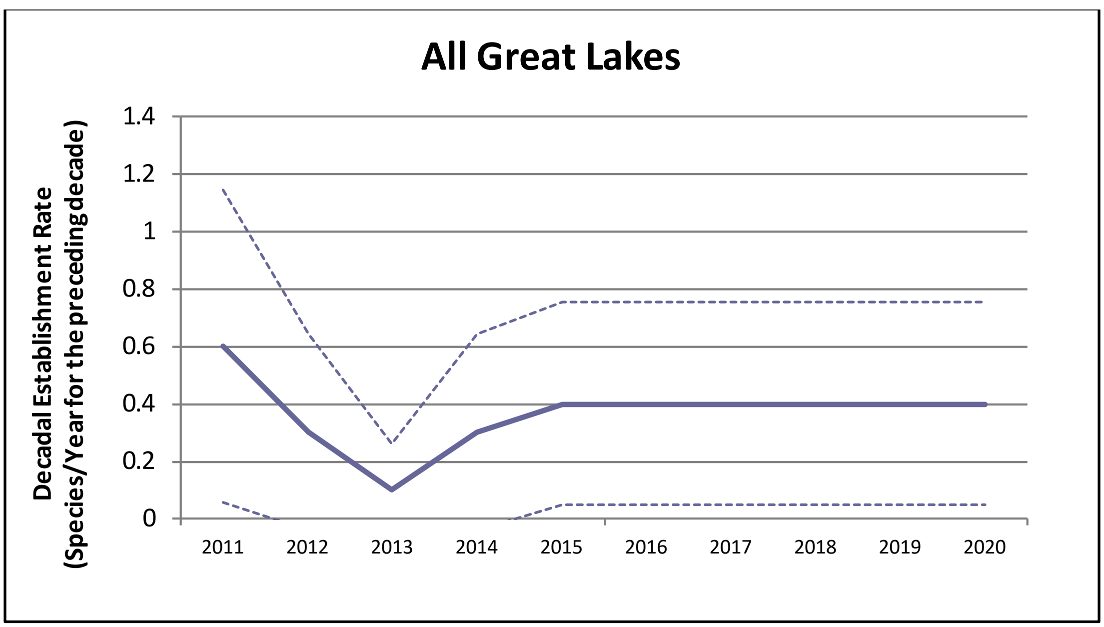


Figure 2. Running decadal establishment rate of aquatic non-indigenous species in the Great Lakes for 2011-2020 with 95% confidence intervals.Source: GLANSIS.

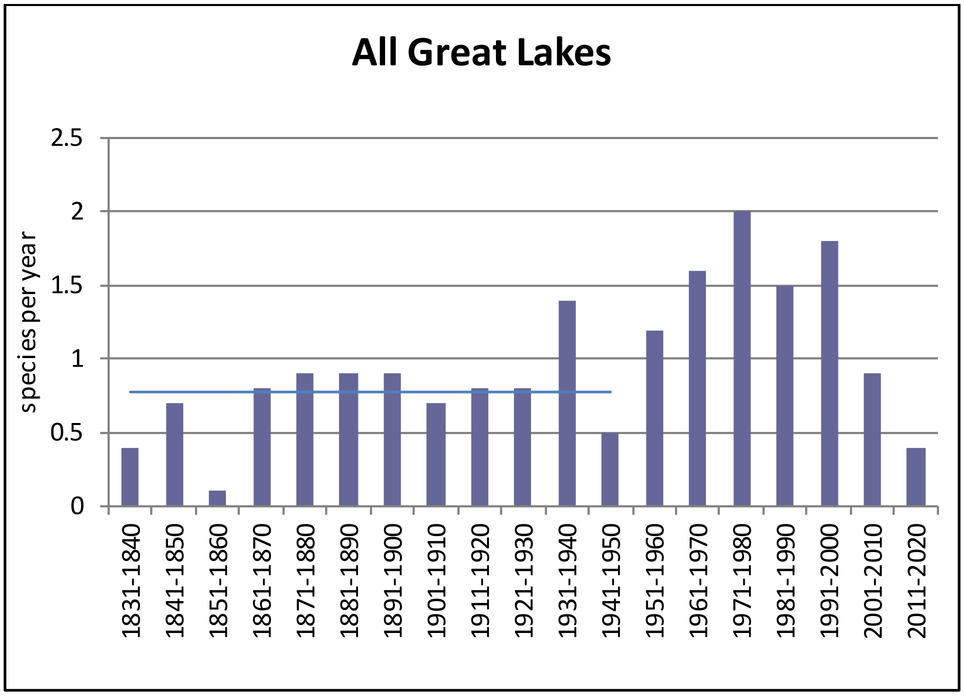


Figure 3. Decadal establishment rates of aquatic non-indigenous species in the Great Lakes for the period of record. Solid horizontal line indicates the pre-1950 average.

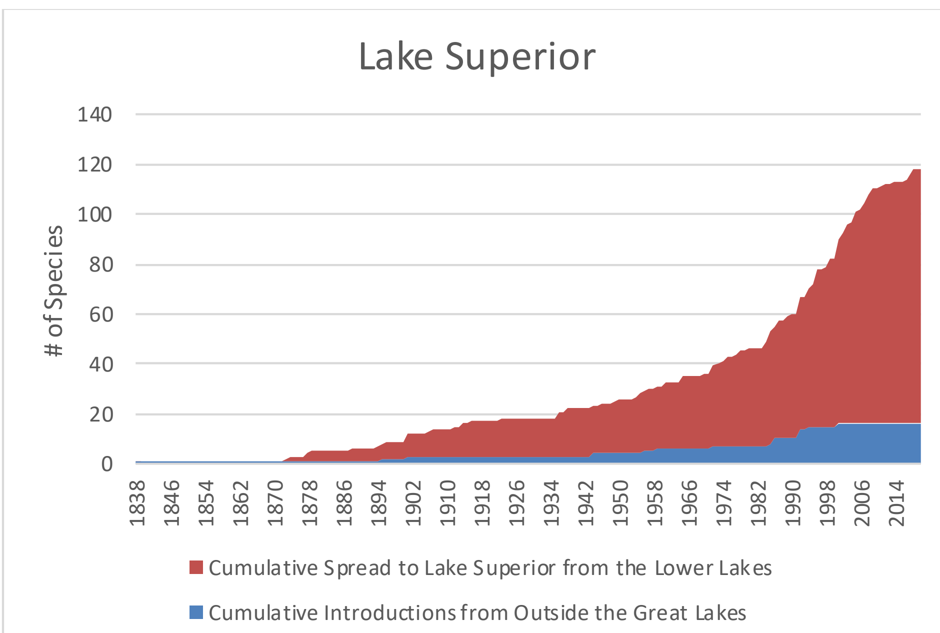


Figure 4. Cumulative established aquatic non-indigenous species in Lake Superior including new introductions (blue) and inter-basin spread (red). 118 aquatic non-indigenous species in total have become established in Lake Superior as of 2020. Source: GLANSIS.

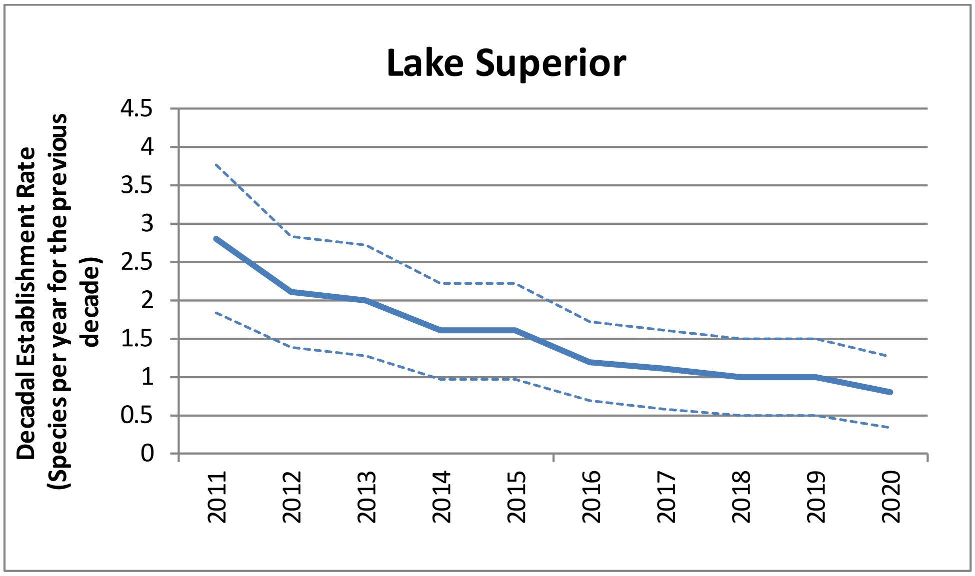


Figure 5. Running decadal establishment rate of aquatic non-indigenous species in Lake Superior for 2011-2020 with 95% confidence intervals (regardless of origin). Source: GLANSIS.

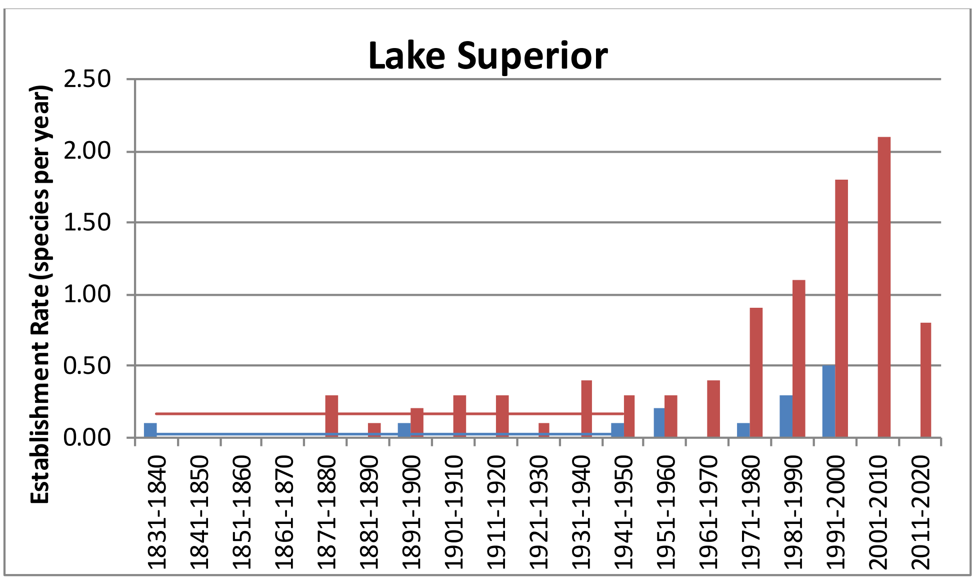


Figure 6. Decadal establishment rates of new aquatic non-indigenous species from outside of the Great Lakes basin (blue), and aquatic non-indigenous species spread from the other Great Lakes (red) for the period of record for Lake Superior. Horizontal solid lines represents the pre-1950 average rate of establishment. Source: GLANSIS.

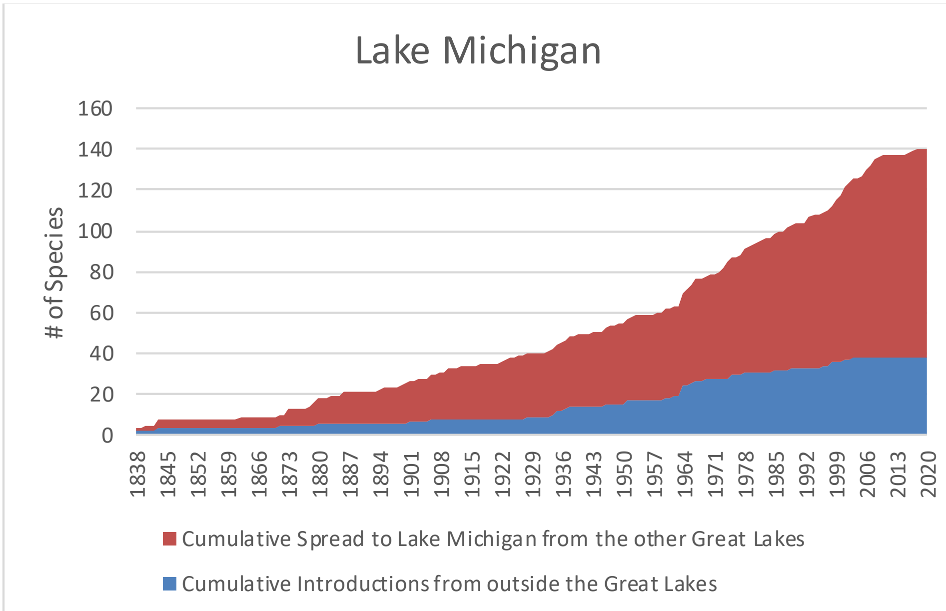


Figure 7. Cumulative established aquatic non-indigenous species in Lake Michigan including new introductions (blue) and inter-basin spread (red). 140 aquatic non-indigenous species in total have become established in Lake Michigan as of 2020. Source: GLANSIS.

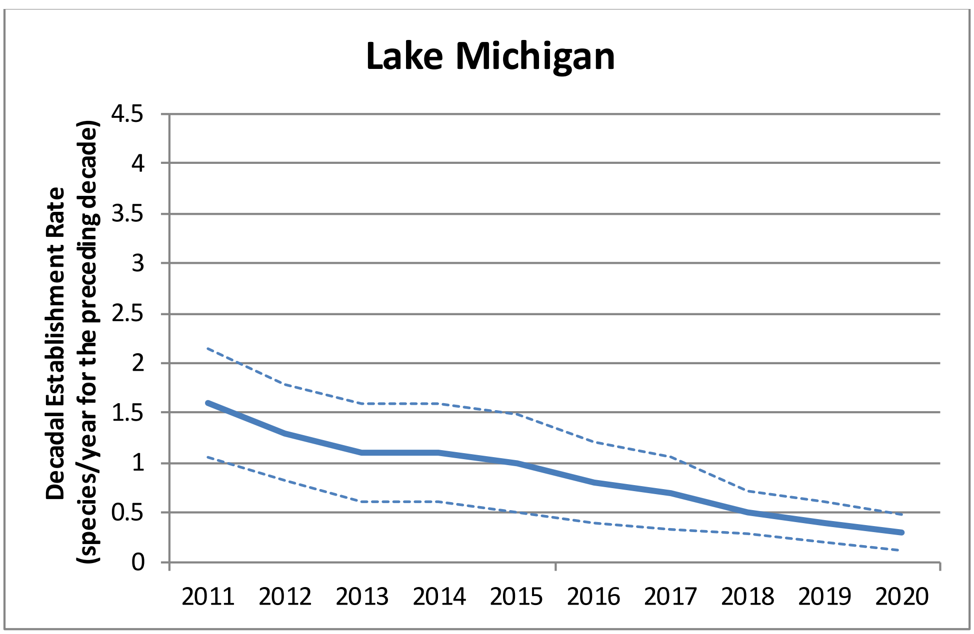


Figure 8. Running decadal establishment rate of aquatic non-indigenous species in Lake Michigan for 2011-2020 with 95% confidence intervals (regardless of origin). Source: GLANSIS.

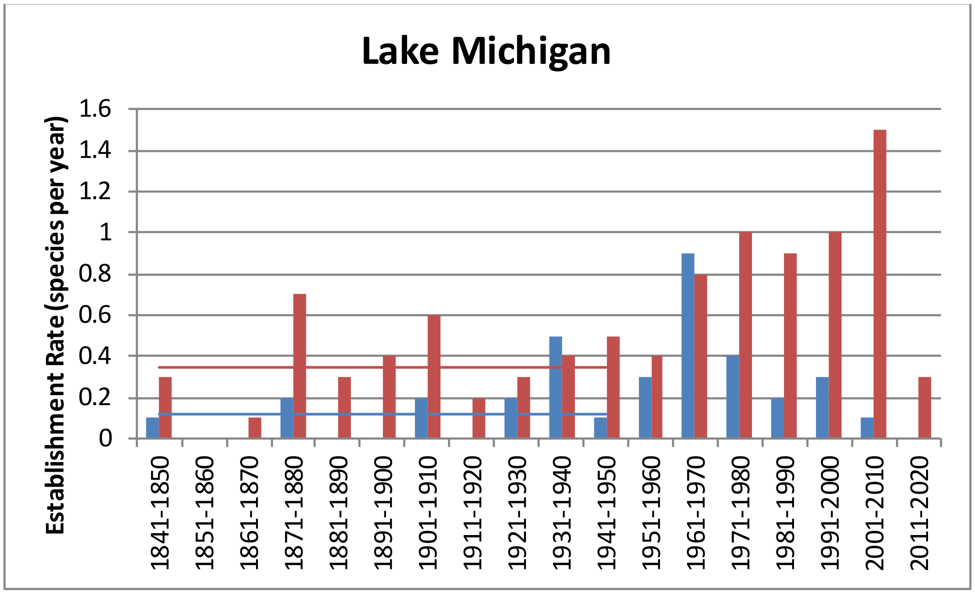


Figure 9. Decadal establishment rates of new aquatic non-indigenous species from outside of the Great Lakes basin (blue), and aquatic non-indigenous species spread from the other Great Lakes (red) for the period of record for Lake Michigan. Horizontal solid lines represents the pre-1950 average rate of establishment. Source: GLANSIS.

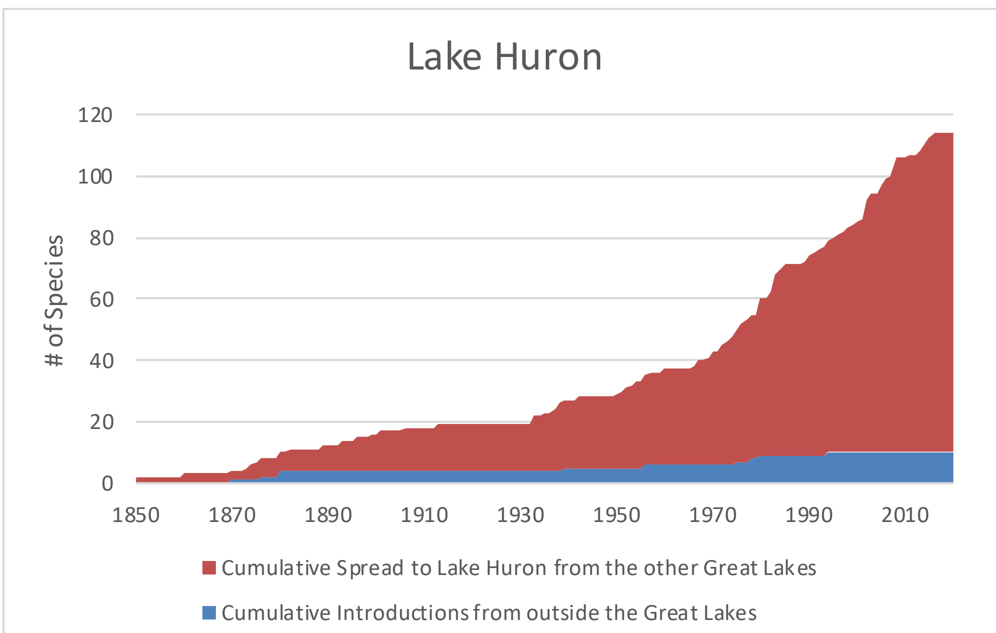


Figure 10. Cumulative established aquatic non-indigenous species in Lake Huron including new introductions (blue) and inter-basin spread (red). 114 aquatic non-indigenous species in total have become established in Lake Huron as of 2020. Source: GLANSIS.

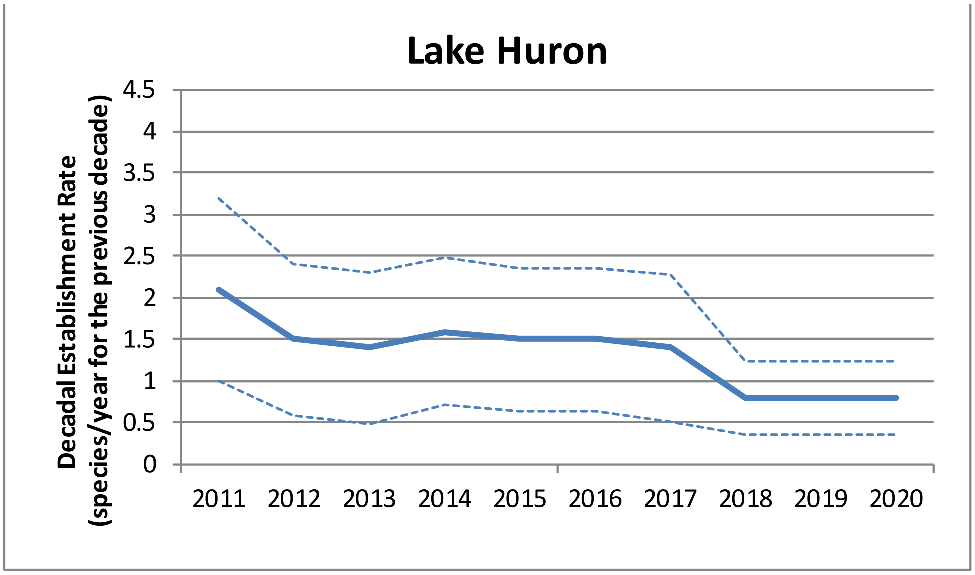
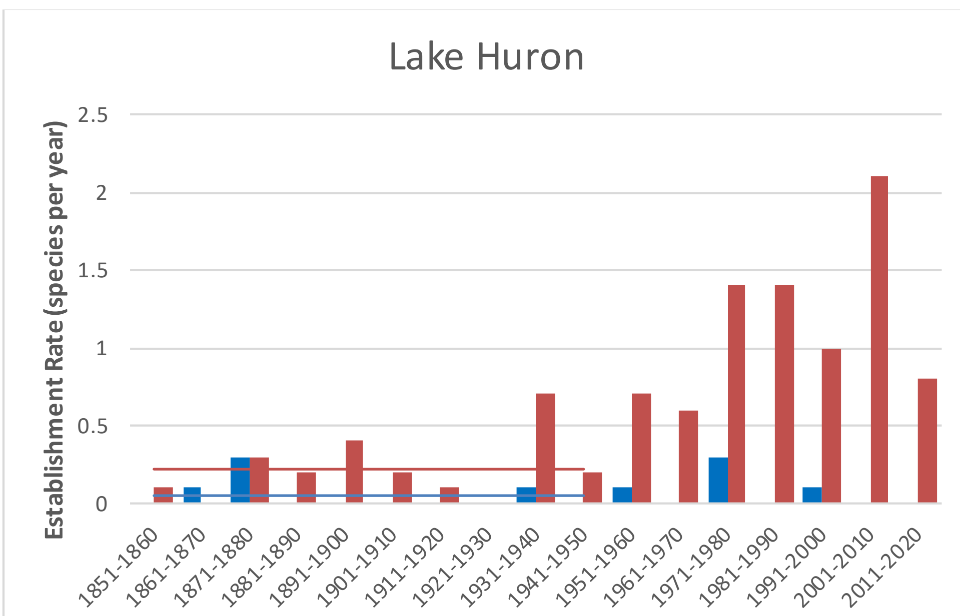


Figure 11. Running decadal establishment rate of aquatic non-indigenous species in Lake Huron for 2011-2020 with 95% confidence intervals (regardless of origin). Source: GLANSIS.



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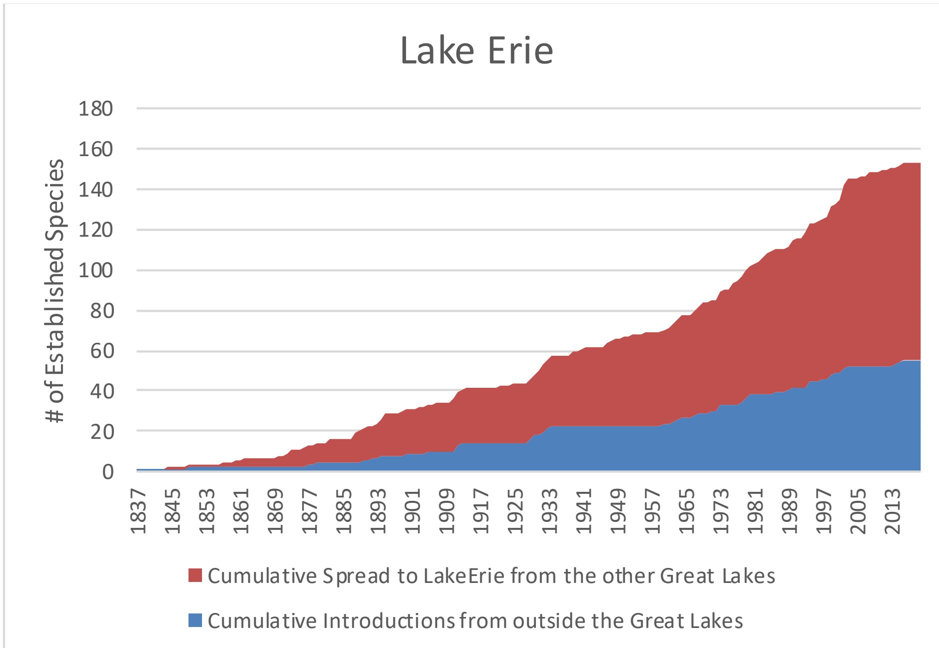


Figure 13. Cumulative established aquatic non-indigenous species in Lake Erie including new introductions (blue) and inter-basin spread (red). 153 aquatic non-indigenous species in total have become established in Lake Erie as of 2020. Source: GLANSIS.

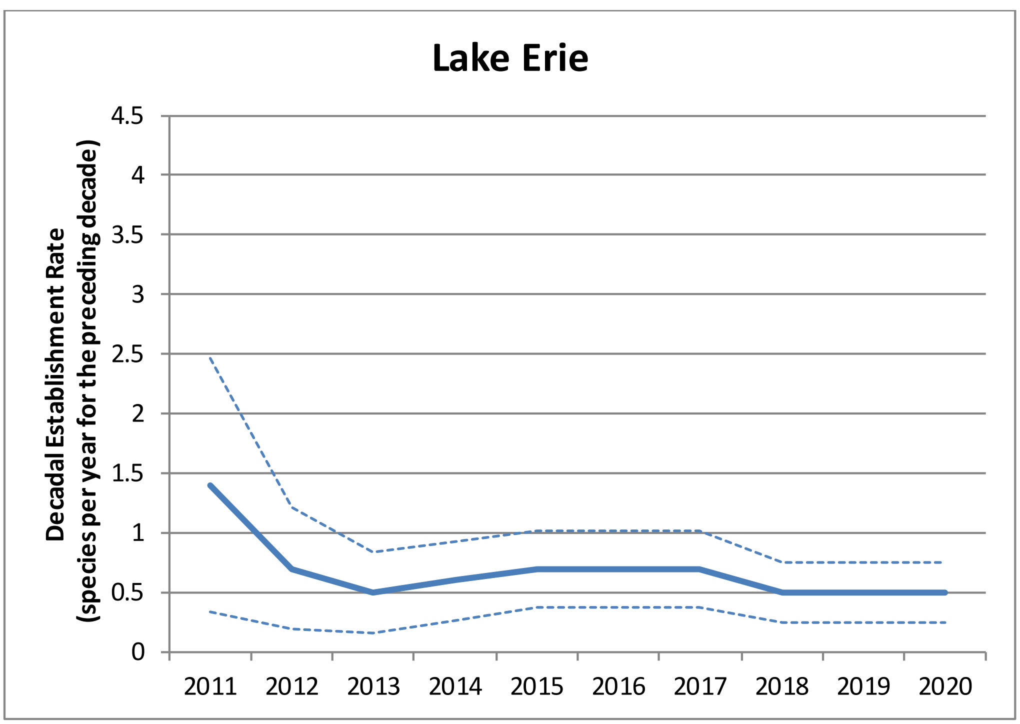
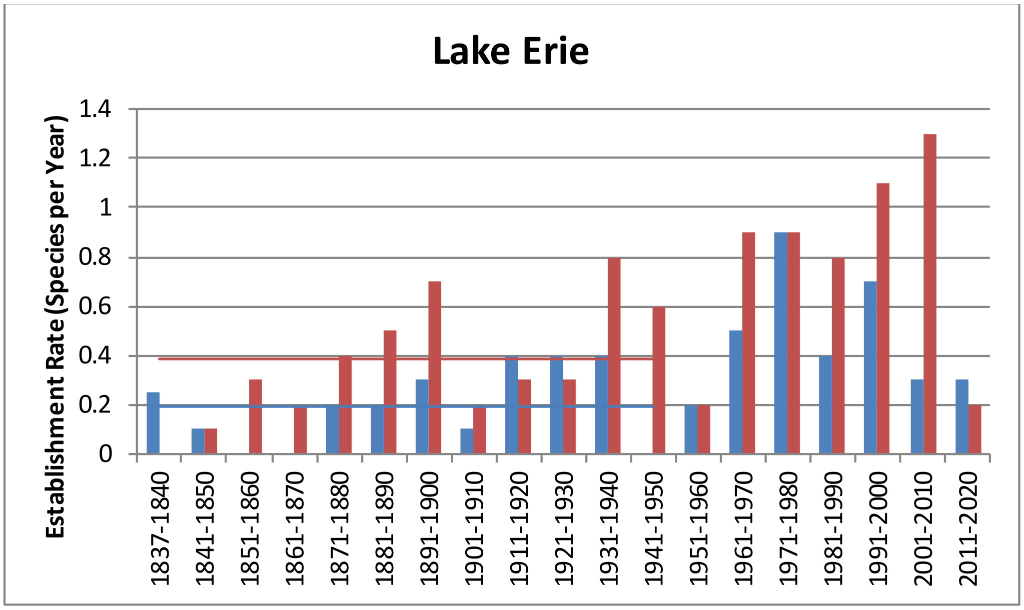


Figure 14. Running decadal establishment rate of aquatic non-indigenous species in Lake Erie for 2011-2020 with 95% confidence intervals (regardless of origin). Source: GLANSIS.



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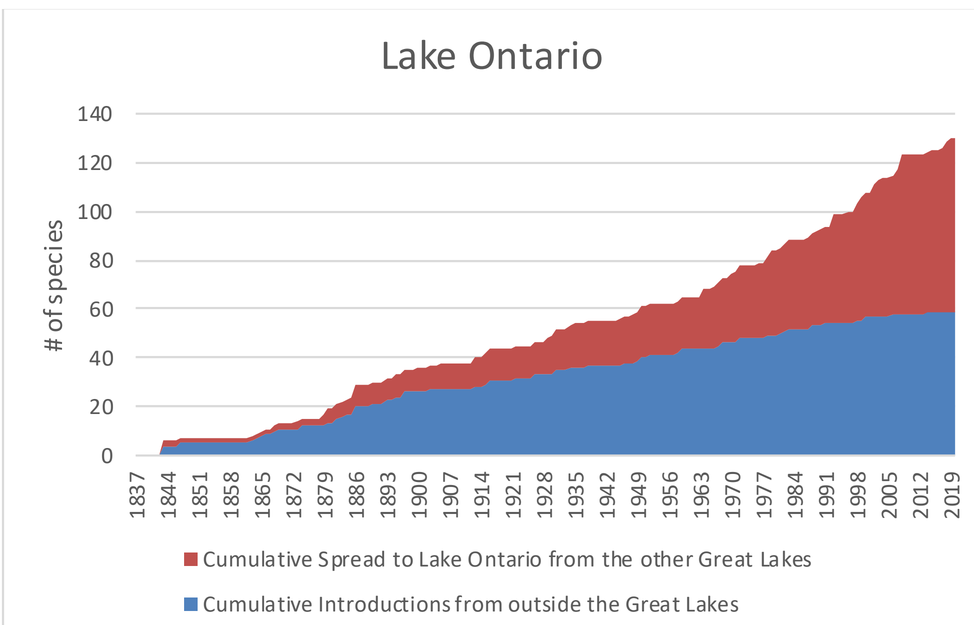
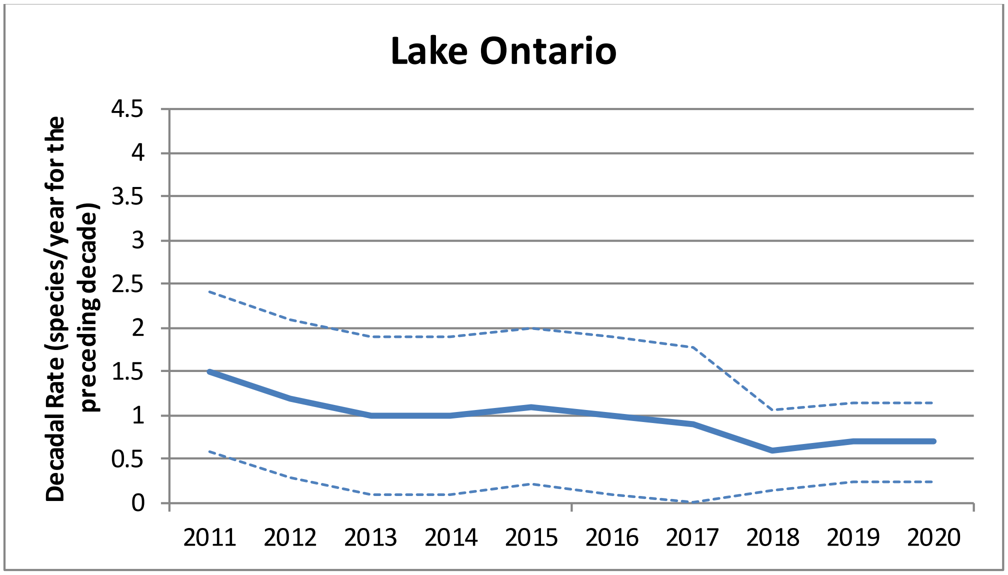


Figure 16. Cumulative established aquatic non-indigenous species in Lake Ontario including new introductions (blue) and inter-basin spread (red). 130 aquatic non-indigenous species in total have become established in Lake Ontario as of 2020. Source: GLANSIS.



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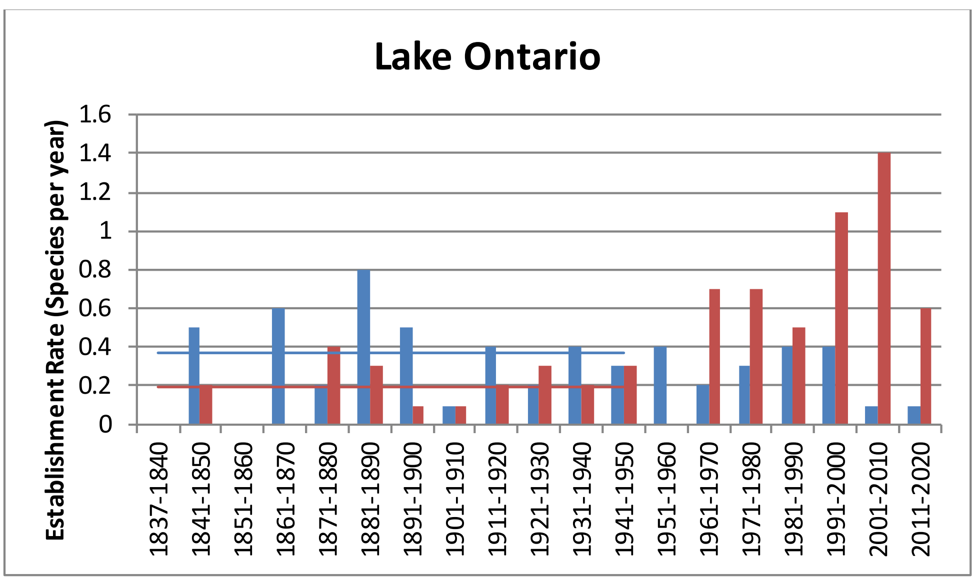

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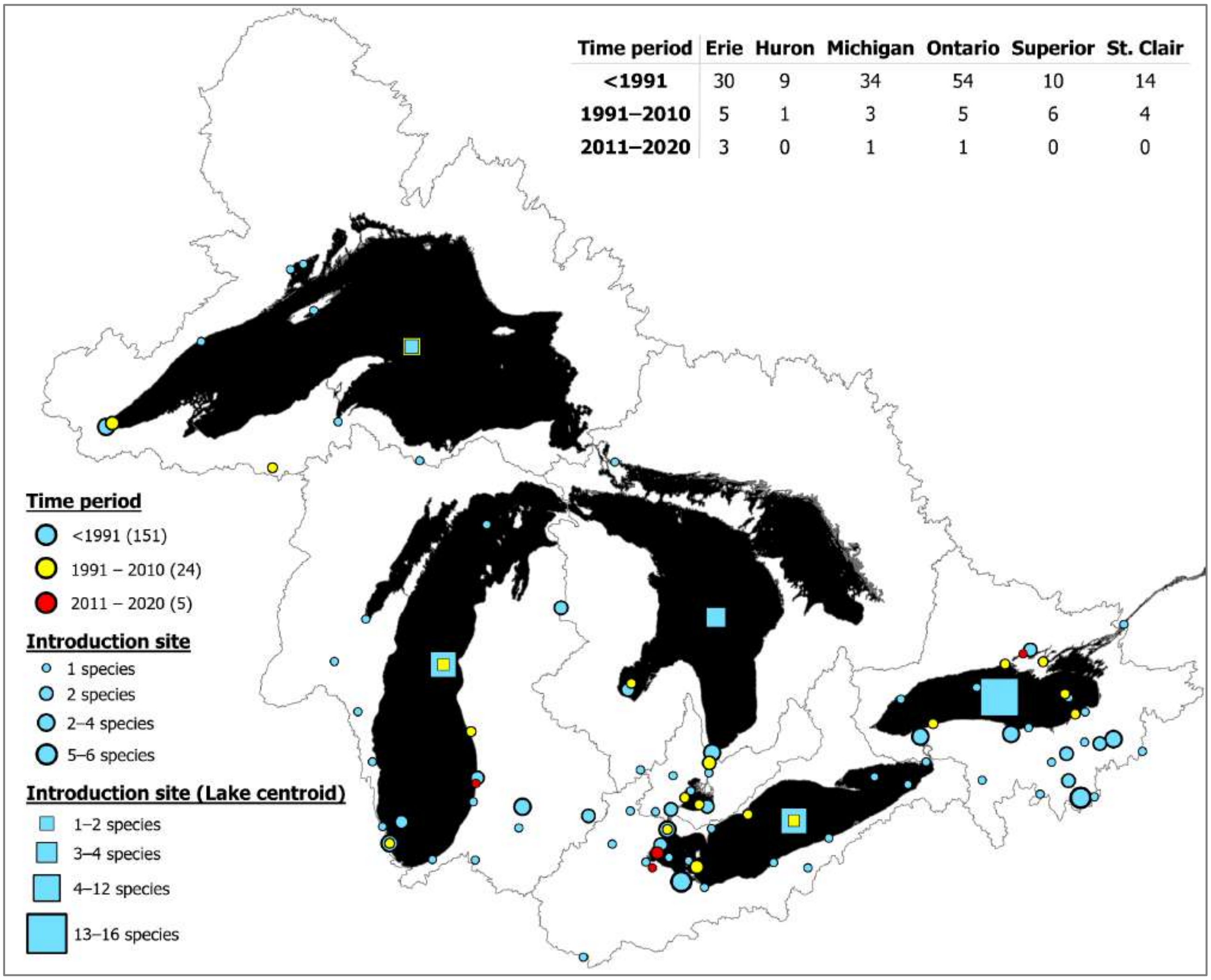

Figure 19: Points of introduction of aquatic non-indigenous species to the Great Lakes basin.

Sub-Indicator: Impacts of Aquatic Invasive Species (AIS)

Overall Assessment

Status: Poor

Trends:

10-Year Trend (2011-2020): Unchanging

Long-term Trend (1837-2020): Deteriorating

Rationale: The current Cumulative Impact Index for impacts of aquatic invasive species (AIS) in the Great Lakes overall (CII-GL) is 509, more than double the 1950 baseline of 194 for this metric. This Great Lakes-wide CII value derives in part from the growth in raw numbers of AIS in the Great Lakes ecosystem overall from 36 in 1950 to 64 in 2020. However, it also considers numbers of lake basins impacted by each species (which also continues to increase), as well as the magnitude of impacts and the range of types of impacts of each species. Notably, within the last decade 11 AIS already present in the Great Lakes ecosystem have spread to include established populations in new lake basins (increasing overall impact). Although the CII-GL value reflects continued increase in impacts basin-wide over time, annual increases in impacts have recently begun to decline with fewer new impacts added to the total each year.

Lake-by-Lake Assessment

The 2020 Lake Cumulative Impact Indices (LCII) for each of the five Great Lakes are presented below. The LCII value reflects the cumulative AIS impacts present in each of the five Great Lakes over time. It considers changes for each Great Lakes in raw numbers of established AIS, their geographic range within the lake (i.e., the number of sub-watersheds impacted), and the magnitude and range of types of the impacts of each established species to the native ecosystem. Note, this method parallels the method for the overall impact (above), but overall impact includes only interbasin spread (between lake-basins) while this section includes intrabasin spread (between watersheds within the lake).

Lake Superior

Status: Poor

10-Year Trend: Undetermined

Long-term Trend (1867- 2020): Deteriorating

Rationale: The LCII for Lake Superior (LCII-LS) is currently more than an order of magnitude higher than its 1950 baseline (221 versus 15, respectively), highlighting a strong increase in impact over the past 70 years. The raw number of AIS has increased in Lake Superior from 16 in 1950 to 49 in 2020 (almost tripling). Further, the impact of AIS to Lake Superior continues to increase as the AIS already colonizing Lake Superior expand their ranges within the basin. Within the last decade (2011-2020), 20 AIS have established populations in new watersheds within the Lake Superior basin, contributing to the increasing impact. Time-lags in reporting spread between individual watersheds within the Lake Superior basin and spread to Superior from the lower lakes create uncertainty in determining trends within the last decade.

Lake Michigan

Status: Poor

10-Year Trend: Undetermined

Long-term Trend (1838- 2020): Deteriorating

Rationale: The LCII for Lake Michigan (LCII-LM) is currently more than an order of magnitude higher than the 1950 baseline (302 vs. 29, respectively). The raw number of AIS has increased for Lake Michigan from 24 in 1950 to 63 in 2020 (almost tripling). Further, the impact of AIS to Lake Michigan continues to increase as the AIS already colonizing Lake Michigan expand their ranges within the basin. Within the last decade (2011-2020), 36 AIS have established populations in new watersheds within the Lake Michigan basin, increasing overall impact. However, time-lags in reporting spread between individual watersheds within the Lake Michigan basin as well as invasion from the other lake basins, create uncertainty in determining cumulative impact trends within the last decade.

Lake Huron (including St. Marys River)

Status: Poor

10-Year Trend: Undetermined

Long-term Trend (1870- 2020): Deteriorating

Rationale: The LCII for Lake Huron basin (LCII-LH) is currently more than an order of magnitude higher than the 1950 baseline (234 vs. 20, respectively). The raw number of AIS has increased in Lake Huron from 15 in 1950 to 52 in 2020 (more than tripling). Further, the impact of AIS to Lake Huron continues to increase as the AIS already colonizing Lake Huron expand their ranges within the basin, increasing overall impact. Within the last decade (2011-2020), 28 invasive species have established populations in new watersheds within the Lake Huron basin. Time-lags in reporting spread between individual watersheds within the Lake Huron basin as well as spread into Lake Huron from the other Lake basins, create uncertainty in determining cumulative impact trends within the last decade.

Lake Erie (including Lake St. Clair, and the St. Clair and Detroit Rivers)

Status: Poor

10-Year Trend: Deteriorating

Long-term Trend (1861- 2020): Deteriorating

Rationale: The LCII for the Lake Erie basin (LCII-LE) is currently an order of magnitude higher than the 1950 baseline (236 vs. 31, respectively). The raw number of AIS has increased in the Lake Erie basin from 27 in 1950 to 58 in 2020 (more than doubling). Further, the impact of AIS to Lake Erie continues to increase as the AIS already present in the basin expand their ranges within the basin. Within the last decade (2011-2020), 30 invasive species have established populations in new watersheds within the Lake Erie basin, increasing overall impact. Time lags in reporting spread among watersheds within the Lake Erie basin as well as spread into Lake Erie from the other Lakes creates uncertainty in determining trends within the last decade. However, significant steady increases in both numbers of AIS populations and their impacts within the last decade are sufficient to consider the LCII-LH trend as deteriorating.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Poor

10-Year Trend: Deteriorating

Long-term Trend (1843- 2020): Deteriorating

Rationale: The LCII for the Lake Ontario basin (LCII-LO) has increased almost sixfold since its 1950 baseline (311 vs. 55, respectively). The raw number of AIS has increased in the Lake Ontario basin by an order of magnitude from 30 in 1950 to 55 in 2020. The impact of AIS to Lake Ontario continues to increase as AIS already present in the basin expand their ranges. Within the last decade (2011-2020), 23 invasive species have established populations in new watersheds within the Lake Ontario basin. Time lags in reporting spread among watersheds within the Lake Ontario basin as well as spread into Lake Ontario from the other Lakes creates uncertainty in determining trends within the last decade. However, significant steady increases in both number of AIS and their impacts within the last decade are sufficient to consider the LCII-LO trend as deteriorating.

Status Assessment Definitions

Good: Existing aquatic invasive species are not impacting the Great Lakes ecosystem (or not significantly higher than 1950 levels).

Fair: Existing aquatic invasive species are having significant negative impact in the Great Lakes ecosystem.

Poor: Existing aquatic invasive species are negatively impacting the Great Lakes ecosystem. (>2x 1950 impact)

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Improving: Impacts of AIS are declining.

Unchanging: No significant changes in the impact of AIS.

Deteriorating: Impact of new or existing AIS is increasing (through introduction or spread).

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

The endpoint is no negative impacts to water quality and/or ecosystem health as a result of aquatic invasive species.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the impact of Aquatic Non-Indigenous Species (ANS) in the Laurentian Great Lakes. Aquatic invasive species (AIS) are the subset of ANS that have impact as individual species. ANS without evidence of impact are thus excluded from this analysis. In doing so, this sub-indicator assessment qualifies AIS as a SOGL-relevant pressure to the biological integrity of the Great Lakes ecosystem.

Ecosystem Objective

The goal of the Great Lakes Water Quality Agreement is to restore and maintain the biological integrity of the Great Lakes Ecosystem.

This sub-indicator best supports work towards General Objective #7 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from the introduction and spread of aquatic invasive species and free from the introduction and spread of terrestrial invasive species that adversely impact the quality of the Waters of the Great Lakes."

Measure

Prior to the 2019 report, the cumulative number of established non-indigenous species was used as a single sub-indicator for invasive species impacts. Beginning with the 2019 report, the sub-indicator report previously called the Impact of Aquatic Invasive Species was separated into two sub-indicator reports: Rate of New ANS Establishment (which considers new establishment from outside of the basin as well as interbasin spread) and Impact of Invasive Species which considers the number of species present and their ecological and socioeconomic impacts. This approach is being maintained here with the impacts component being covered in this sub-indicator report.

Impact determinations are a function of both 1) types and degree of AIS *degradation of* the native ecosystem and economy (a quality relevant to progress towards control/eradication goals) and 2) the *rate of AIS new introductions and spread*, into the Great Lakes and between lakes and sub-watersheds, respectively.

The steps to the ANS rating process include:

- **Determination of a semi-quantitative Species Impact Factor (SIF) for each ANS of interest -** The SIF is a one-time determination based on Sturtevant et *al*, 2014, and expressed on a scale of 0-72. Full details of each species assessment are available in the appendices to NOAA GLERL TM-161 et seq. The SIF considers up to 12 potential impact types (6 environmental and 6 socioeconomic) associated with a specific AIS; and potential impact intensity (little or no, moderate, and high) across impact types. A score of 0 (low or unknown reported impact in every category) means that ANS is not considered an AIS, and is excluded from the analysis. Scores 1-5 indicate only moderate impacts have been observed in any category. Scores 6-12 may result from a single strong impact OR from a combination of multiple moderate impacts. Scores higher than 12 indicate at least one highly significant impact. The theoretical maximum score of 72 suggests high impact across all categories. Benefits (positive impacts) are noted here, but do not influence the SIF.
- **Determination and incorporation of extent of geographic scope of AIS spread across lake basins (for the Overall Assessment) – For each year of the record (1838-2020),** the SIF is multiplied by the percentage of lakes affected by that AIS (i.e., from one-fifth if the species is in only one lake, to 100% if present in all 5 Great Lakes basins) to generate a Great Lakes-wide SIF (GL-SIF). GL-SIF therefore increases with interbasin spread.
- **Determination and incorporation of extent of spread within individual lakes (for the individual Lake Assessments) -** More fine-grained lake-specific Species Impact Factors (L-SIFs) also were calculated to help describe the state of each individual lake with respect to AIS alterations and scope of spread. First, within-lake watersheds were defined as HUC8s, which vary in number within each Great Lake from 10 for Lake Ontario to 33 for Lake Michigan. A L-SIF was determined for each year of the record (1838-2020) by multiplying the SIF by the percent of the HUC8s within a lake affected by that AIS. Thus, if the

species has spread to 2/3 of the HUC8s within a lake, the L-IF for that lake is 2/3 of its SIF. L-SIF thereby increases with intra-basin spread. Notably, use of the percentages of affected HUC8 to account for extent of in-lake spread in the L-CII weights nearshore spread heavier than open lake spread as the nearshore is divided into HUC8s more finely than the offshore zones (only 1-3 HUC8 watersheds comprise the offshore waters of each lake).

- **Cumulative and trend determinations**: Cumulative Impacts are assessed from the first date of each species detection within each lake (as noted and geographically assigned by GLANSIS). As such, the GL-CII reflects a summation of each SIF multiplied by number of lakes in which that species was detected across a time series beginning with the date of first detection for each AIS. The GL-CII trends are tracked over time. The trend could theoretically reflect a reduction in cumulative impact (if a high-impact AIS were somehow extirpated over time, for example), but GL-CII trends generally mean that the numbers and impacts of AIS in the Great Lakes are worsening. Such trends can be measured, and the statistical significance calculated. Lake-specific CIIs (L-CIIs) reflect similar cumulative and trend information but at a more geographically granular scale (HUC8s as opposed to whole lakes).

There is a relationship between the rate of non-indigenous species introduction at the individual lake scale as they appear in the Rate related sub-indicator report and the whole Great Lakes assessment included in this sub-indicator report, however this is a relationship, not a duplication. The rate report uses all non-indigenous species in its calculation. This report uses only the subset that are confirmed invasive (impactful). This impact indicator weights the contribution of each species by its impact factor. For example, in the rate report, a movement of zebra mussel from Michigan to Superior and a movement of *Thermocyclops crassus* from Erie to Superior have the same weight – one species moving into the lake. In this assessment of impact, the movement of zebra mussel from Michigan to Superior scores 55 and the movement of *Thermocyclops crassus* from Erie to Superior scores zero (*Thermocyclops crassus* impact is currently assessed at unknown/zero).

Ecological Condition

The overall economic impact of invasive species on the Great Lakes region—spanning direct operating costs, decreased productivity, and reduced demand within sport and commercial fishing, power generation, industrial facilities, tourism and recreation, water treatment, and households—is estimated at well over $100 million annually (Rosaen et al. 2012). This figure includes both costs of basinwide control efforts such as that of Great Lakes Fishery Commission's sea lamprey control program, with an annual budget of about $23 million, and costs of local responses, such as the $1,040-$26,000 cost per acre of Eurasian watermilfoil removal (Rosaen et al. 2012). Economic costs from Dreissenid mussel control and monitoring are estimated at $1.2 million annually per power plant, $1.97 million per 400 cubic yards removed from a paper plant intake line, and $480,000-$540,000 annually at a water treatment plant (Rosaen et al. 2012).

As noted previously, for SOGL reports prior to 2019, the total number of ANS was used as a proxy for overall introduction and impact. For subsequent SOGL reports (2019 and 2022), however, only ANS demonstrated to be invasive (AIS) per the NOAA assessment are considered in the impact sub-indicator (with rate of ANS establishment continuing as an independent sub-indicator). Specifically, this impact status assessment shows the cumulative impact of AIS as they spread across the lakes, weighted by the diversity and strength of known impact of the individual species.

Using this approach, at least 37% of the non-indigenous species found in the Great Lakes qualify as invasive (AIS) given their moderate to high environmental and/or socioeconomic impact as assessed using NOAA's method (Sturtevant et al 2014; Figure 1). Approximately 20% of the species are assessed as having little to no impact (green portion of the graph) – while not native, these species are not documented to be causing harm. There is some time-lag following introduction before short-term impacts of a newly established ANS may be realized. Further, environmental changes within the Great Lakes ecosystem caused by climate change and other pressures make long term impacts unknowable. However, the proportion of non-indigenous species which have significant impacts is relatively constant through time (some species which have been here >50 years still have insufficient information for scientists to properly assess their environmental impact). ANS whose potential impact is unknown due to lack of data are scored 'Unknown' (grey portion of the graph).

The year 1950 (prior to the opening of the St. Lawrence Seaway in 1959 and the inflection point at which the rate of ANS introduction began to increase logarithmically) is used as the benchmark for comparison of the status of the system. It is evident from these data that nearly every watershed in the region has experienced an increase in the number of AIS within its boundaries over time (Figure 2), which drives increases in AIS impact (Figure 4). The overall cumulative impact of AIS on the Great Lakes has more than doubled since 1950 by our assessment (Figure 3). This increase is the result of the invasion of 28 new AIS from beyond the Great Lakes taken together with movement between lake basins of the 36 AIS that had been introduced prior to 1950.

Within the last decade (2011-2020), the cumulative impact on the Great Lakes basin scale appears to be stabilizing with fewer AIS moving between lakes than in previous periods (slope of the CII graph is decreasing). Underlying this trend is the fact that only two AIS have entered the Great Lakes in the last decade: *Salmincola californiensis* (2014) and *Procambarus acutus acutus* (2017). Yet, the cumulative impact of AIS in the Great Lakes does continue to increase over that same time period as AIS have established populations in new lake basins. Specifically, 9 invasive species have established populations in new lake basins (increasing overall impact): *Butomus umbellatus*, *Cirsium palustre*, *Conium maculatum*, *Glyceria maxima*, *Salmincola californiensis*, *Epilobium hirsutum*, *Hydrocharis-morsus ranae*, *Potamogeton crispus*, *and Hemimysis anomala*. While spread is a critical driver in the changes, cumulative impact shows that the strength and range of types of impacts of particular species are also important. The top 10 AIS – Zebra mussel, Quagga mussel, Alewife, Sea lamprey, Round Goby, White perch, Eurasian watermilfoil, VHS, BKD and Fishhook waterflea – are responsible for approximately 50% of the overall cumulative impact).

The same basic method was used to assess the impact of AIS in US waters of each of the lake basins. At this scale, spread between watersheds (HUC8) also increases an impact score rather than only inter-lake spread. Because the number of watersheds within each lake-basin vary, the index is scaled to the number of watersheds.

At least 117 non-indigenous species are overwintering and reproducing in Lake Superior (Figure 5), with 49 of these (42%) exhibiting notable environmental or socioeconomic impact (49 AIS included in this lake assessment). 33 of these AIS entered Lake Superior after 1950 and 4 of those in the last decade. The lake-specific cumulative impact index for US waters of Lake Superior has increased more than 14-fold since 1950 (Figure 6). Within the last decade (2011-2020), the following 22 invasive species have established populations in new watersheds within the Lake Superior basin (increasing overall impact): *Alosa pseudoharengus*, *Bithynia tentaculata*, *Butomus umbellatus*, *Bythotrephes longimanus*, *Cirsium palustre*, *Dreissena polymorpha*, *Dreissena rostriformis bugensis*, *Dorosoma cepedianum*, *Faxonius immunis*, *Faxonius rusticus*, *Frangula alnus*, *Glyceria maxima*, *Gymnocephalus cernua*, *Hemimysis anomala*, *Iris pseudacorus*, *Lythrum salicaria*, *Myriophyllum spicatum*, *Neogobius melanostomus*, *Potamogeton crispus*, *Renibacterium salmoninarum*, *Solanum dulcamara*, *and Typha angustifolia*.

At least 140 non-indigenous species are overwintering and reproducing in Lake Michigan (Figure 7), with 63 of these (45%) exhibiting notable environmental or socioeconomic impact (63 AIS). Thirty-nine (39) of these AIS entered Lake Michigan after 1950 but just 1 in the last decade. The lake-specific cumulative impact index for Lake Michigan has increased more than 10-fold since 1950 (Figure 6). Within the last decade (2011-2020), the following 36 AIS have established populations in new watersheds within the Lake Michigan basin (increasing overall impact):

Alosa pseudoharengus, Bithynia tentaculata, Butomus umbellatus, Bythotrephes longimanus, Cabomba caroliniana, Cirsium palustre, Conium maculatum, Corbicula fluminea, Cyprinus carpio, Dreissena polymorpha, Dreissena rostriformis bugensis, Echinochloa crus-galli, Echinogammarus ischnus, Faxonius rusticus, Frangula alnus, Glyceria maxima, Hemimysis anomala, Hydrocharis morsus ranae, Iris pseudacorus, Lysimachia vulgaris, Lythrum salicaria, Morone americana, Myriophyllum spicatum, Najas minor, Neogobius melanostomus, Nitellopsis obtusa, Nymphoides peltata, Oncorhynchus gorbuscha, Persicaria maculosa, Potamogeton crispus, Potamopyrgus antipodarum, Procambarus clarkii, Salix fragilis, Solanum dulcamara, Typha angustifolia, and VHSv,

At least 114 non-indigenous species are overwintering and reproducing in Lake Huron (including the St. Marys River) (Figure 8), with 52 of these (46%) exhibiting notable environmental or socioeconomic impact (52 AIS). Thirty-eight (38) of these invasive species entered Lake Huron after 1950 and 6 of those in the last decade. The lake-specific cumulative impact index for US waters of Lake Huron has increased more than 10-fold since 1950 (Figure 6). Within the last decade (2011-2020), the following 28 AIS have established populations in new watersheds within the Lake Huron basin (increasing overall impact): *Bithynia tentaculata, Butomus umbelatus, Bythotrephes longimanus, Cirsium palustre, Corbicula fluminea, Cyprinus carpio, Didymosphenia geminata, Dreissena polymorpha, Dreissena rostriformis bugensis, Echinogammarus ischnus, Epilobium hirsutum, Faxonius rusticus, Frangula alnus, Glyceria maxima, Gymnocephalus cernuua, Hydrocharis morsus-ranae, Iris pseudacorus, Lythrum salicaria, Morone americana, Myriophyllum spicatum, Najas minor, Neogobius melanostomus, Nitellopsis obtusa, Osmerus mordax, Potamogeton crispus,Potamopyrgus antipodarum, Solanum dulcamara, and Typha angustifolia.*

At least 153 non-indigenous species are overwintering and reproducing in Lake Erie (including Lake St. Clair and the Detroit and St. Clair Rivers) (Figure 9), with 58 of these (38%) exhibiting notable environmental or socioeconomic impact (58 AIS). Thirty-one (31) of these invasive species entered Lake Erie after 1950 but only 1 of those in the last decade. The lake-specific cumulative impact index for US waters of Lake Erie has increased nearly 8-fold since 1950 (Figure 6). Within the last decade (2011-2020), the following 30 AIS have established populations in new watersheds within the Lake Erie basin (increasing overall impact): *Alosa pseudoharengus, Bithynia tentaculata, Cabomba caroliniana, Butomus umbellatus, Cirsium palustre, Conium maculatum, Corbicula fluminea, Cyprinus carpio, Dreissena polymorpha, Dreissena rostriformis bugensis, Echinogammarus ischnus, Epilobium hirsutum, Faxonius rusticus, Frangula alnus, Hydrocharis morsus-ranae, Iris pseudacorus, Lepomis microlophus, Lysimachia vulgaris, Morone americana, Myriophyllum spicatum, Najas minor, Neogobius melanostomus, Nitellopsis obtusa, Nymphoides peltata, Osmerus mordax, Petromyzon marinus, Potamogeton crispus, Procambarus clarkii, Solanum dulcamara, and Typha angustifolia.*

At least 130 non-indigenous species are overwintering and reproducing in Lake Ontario (including the Niagara River and international section at the headwaters of the St. Lawrence River) (Figure 10), with 55 of these (42%) exhibiting notable environmental or socioeconomic impact (55 AIS). Twenty-five (25) of these AIS entered Lake Ontario after 1950 but only 2 of those in the last decade. The lake-specific cumulative impact index for US waters of Lake Ontario has increased more than 5-fold since 1950 (Figure 6). Within the last decade (2011-2020), the following AIS have established populations in new watersheds within the Lake Ontario basin (increasing overall impact): *Alosa pseudoharengus, Bithynia tentaculata, Butomus umbellatus, Bythotrephes longimanus, Cabomba caroliniana, Cirsium palustre, Dreissena polymorpha, Echinogammarus ischnus, Hydrocharis morsus-ranae, Iris pseudacorus, Lythrum salicaria, Morone americana, Neogobius melanostomus, Nitellopsis obtusa, Potamogeton crispus, Potamopyrgus antipodarum, Procambarus acutus acutus, Ranavirus, Salmincola californiensis, Scardinius erythropthalmus, Solanum dulcamara, Trapa natans, and Typha angustifolia.*

Linkages

Linkages to other sub-indicators in the indicator suite include:

- Food Web category - AIS may exert significant direct and indirect pressures upon native species, including facilitation of parasitism, transmission of viral/bacterial infections, magnification of toxins, competition, food-web alteration, genetic introgression, degradation of water quality, and degradation of physical habitat. AIS have promoted the proliferation of native nuisance species, including cyanobacteria (Skubinna et al. 1995; Vanderploeg et al. 2001).
- Walleye – Walleye are among the native fish species impacted by AIS.

This sub-indicator also links directly to the other sub-indicators in the Invasive Species category, particularly Rate of New ANS Establishment in the Great Lakes, Sea Lamprey and Dreissenid Mussels. In particular, evidence indicates that the presence of one ANS may facilitate the establishment or population growth of another in a process termed "invasion meltdown" (Ricciardi 2001). For example, the Sea Lamprey (*Petromyzon marinus*) may have contributed to opening and maintaining enemy-free space that facilitated the Alewife's (*Alosa pseudoharengus*) invasion, and the *Dreissenid* mussel invasion may have facilitated the subsequent Round Goby (*Neogobius melanostomus*) and amphipod (*Echinogammarus ischnus*) invasion by providing a food source and/or habitat. Both the rate and impact of invasion may increase if positive interactions involving established ANS or native species facilitate the establishment of new ANS. Further, each new invader can interact in unpredictable ways with previously established invaders, potentially creating synergistic impacts (Ricciardi 2001, 2005). For example, recurring outbreaks of avian botulism in the lower Great Lakes are thought to result from the effects of Dreissenid Mussels and round gobies, in which the mussels create environmental conditions that promote the pathogenic bacterium and the gobies transfer bacterial toxin from the mussels to higher levels of the food web.

In addition, AIS may exert significant direct and indirect pressures upon native species including facilitation of parasitism, transmission of viral/bacterial infections, magnification of toxins, competition, food-web alteration, genetic introgression, degradation of water quality, and degradation of physical habitat. AIS have promoted the proliferation of native nuisance species, including green algae (*Cladophora*); cyanobacteria (Skubinna et al. 1995; Vanderploeg et al. 2001), and bacteria (botulism). Many invasive plants are capable of forming dense mats that may exclude fish from nearshore habitats. Colonization of lakebed areas by Dreissenid Mussels and the consequent filling of remaining interstitial spaces with pseudofeces and fine-grained sediments led to the exclusion of Lake Trout from some of their native spawning grounds (Redman et al 2017).

Finally, climate change may also be facilitating the northward migration of species as well as altering habitat in a way that favors some invaders over natives or alters their impacts.Specifically, increasing lake temperatures associated with climate change increases potential for ANS introduced from warmer climates to establish overwintering populations (Adebayo et al. 2011; Mandrak 1989). High-impact, warm-water species like Asian clam (*Corbicula fluminea*) appear to have spread downstream into the lower Great Lakes, possibly in conjunction with lake warming. Data on species (native or not) that may be expanding their range within the Great Lakes is currently still lacking; GLANSIS tracks only 24 such species (mostly those that invaded the upper lakes via the Welland Canal). More monitoring data will be needed to assess potential expansion of these populations due to climate change.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada			X	
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://www.glerl.noaa.gov/glansis		

Data Limitations

The state of scientific knowledge remains insufficient to determine the impact of nearly half (43%) of the overwintering and reproducing ANS and range-expanding species. By limiting this indicator to only AIS (those species with documented notable impact), the indicator is a conservative estimate of impact.

Significant effort has been made to conduct quality control with regards to the earliest date of each ANS detection for the Great Lakes basin as well as for individual lakes. A similar quality control effort is underway, but not yet complete, for checking data regarding the watershed (HUC 8) scale earliest dates. Data for spread of species at the watershed (HUC 8) scale is currently available only for the US. Thus, while data from the U.S. and Canada are comparable for the overall Great Lakes basin-wide assessment in this analysis, the indicators for the individual lake basins are necessarily biased toward the US watersheds. Nonetheless, this remains a conservative indicator of impact.

There is often a delay between the time of introduction for a new AIS and the time in which the new species is discovered. This temporal lag has implications for accuracy of the cumulative number of invasive species introduced to the Great Lakes, as well as to those for each lake basin and to HUC 8 watersheds. Monitoring coverage also influences the temporal lag. Time-lags for reporting spread between watersheds (HUC8) is likely longer than the time-lags for reporting movements to new lake basins.

Because impact factor is calculated as a constant for each species, this indicator fails to take into account 'naturalization' processes in which an AIS may evolve to become better adapted to the host system and/or the system (including humans in the case of socioeconomic impacts) evolve or adapt to mitigate impact.

Finally, the Cumulative Impact Index includes a component of spread at the watershed (HUC8) scale; it does not take into account population numbers/density within the watershed. Thus Cumulative Impact Index declines only if a species is eradicated at the HUC8 scale. Control efforts that use strategies of reducing population density short of eradication are thus not reflected in the CII and not captured by this sub-indicator. This indicator thus fails to capture successes of AIS population surpression and any associated benefits via programs such as sea lamprey control and biocontrol of purple loosestrife which do not have eradication as a realistic target.

This sub-indicator can be biased by sampling effort. That is, output is ambiguous as to whether an increase in 'number of reported AIS' or 'number of introductions' over time may be due to increased surveillance or actual increase in species numbers in the Great Lakes. However, the rate of new invasions has seemingly dropped since 2007 despite increased monitoring effort. This will need to be accounted for in discussions accompanying the sub-indicator.

Additional Information

This sub-indicator will aid in the assessment of the status of biotic communities, as aquatic invasive species alter both the structure and function of ecosystems, thereby compromising the biological integrity of these systems.

ANS have invaded the Great Lakes basin from regions around the globe. Increasing world trade and travel elevates the risk that additional species will continue to gain access to the Great Lakes. Existing connections between the Great Lakes watershed and systems outside the watershed, such as the Chicago Sanitary and Ship Canal, and growth of industries such as aquaculture, live food markets, and aquarium retail stores will also increase the risk that new ANS will be introduced. New vectors may arise as the face of industry in the region changes.

Acknowledgements

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Data Source: GLANSIS 2021

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Data Source: GLANSIS 2021

Last Updated

State of the Great Lakes 2022 Report

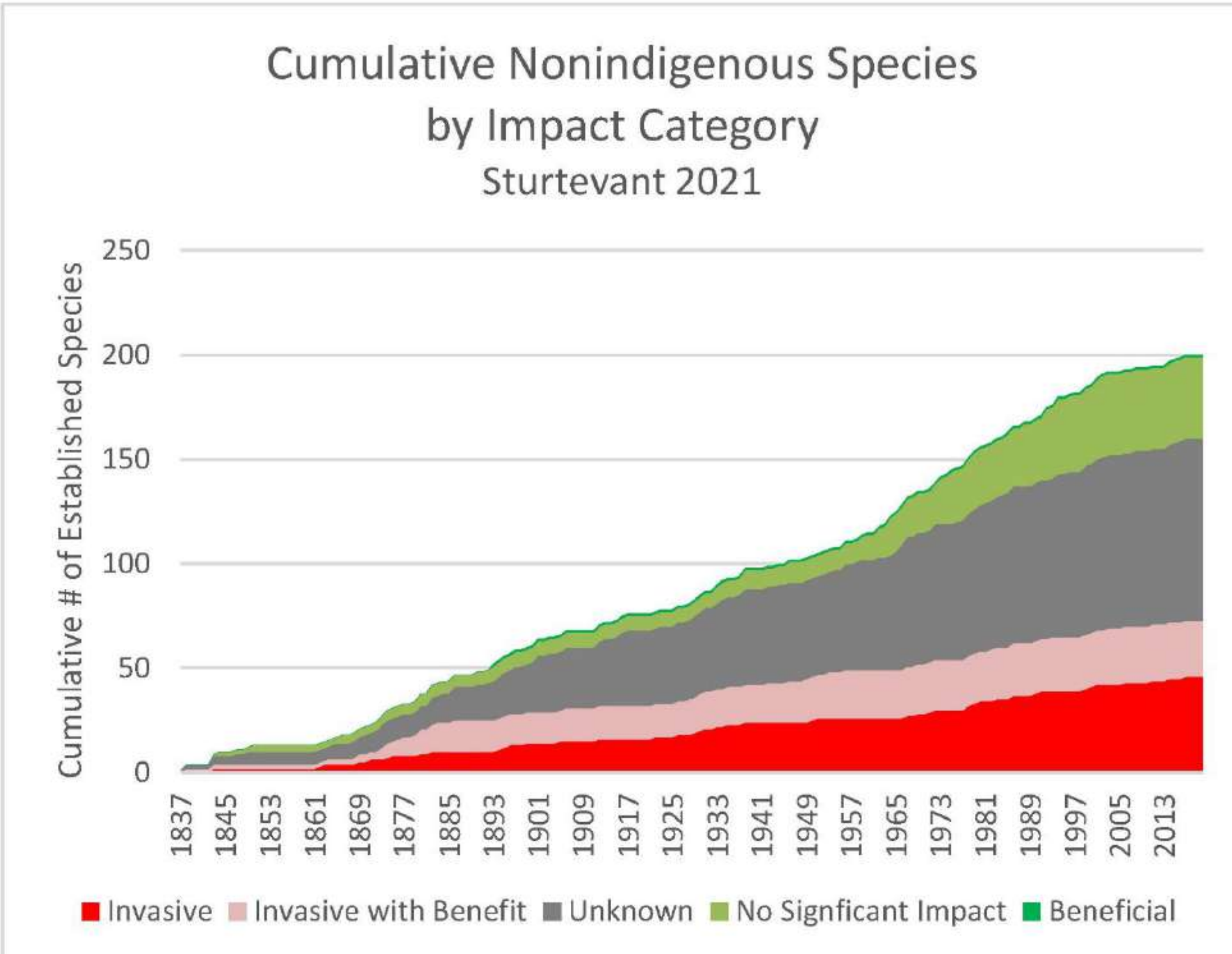

Figure 1. Cumulative non-indigenous species by impact category – Great Lakes basin.
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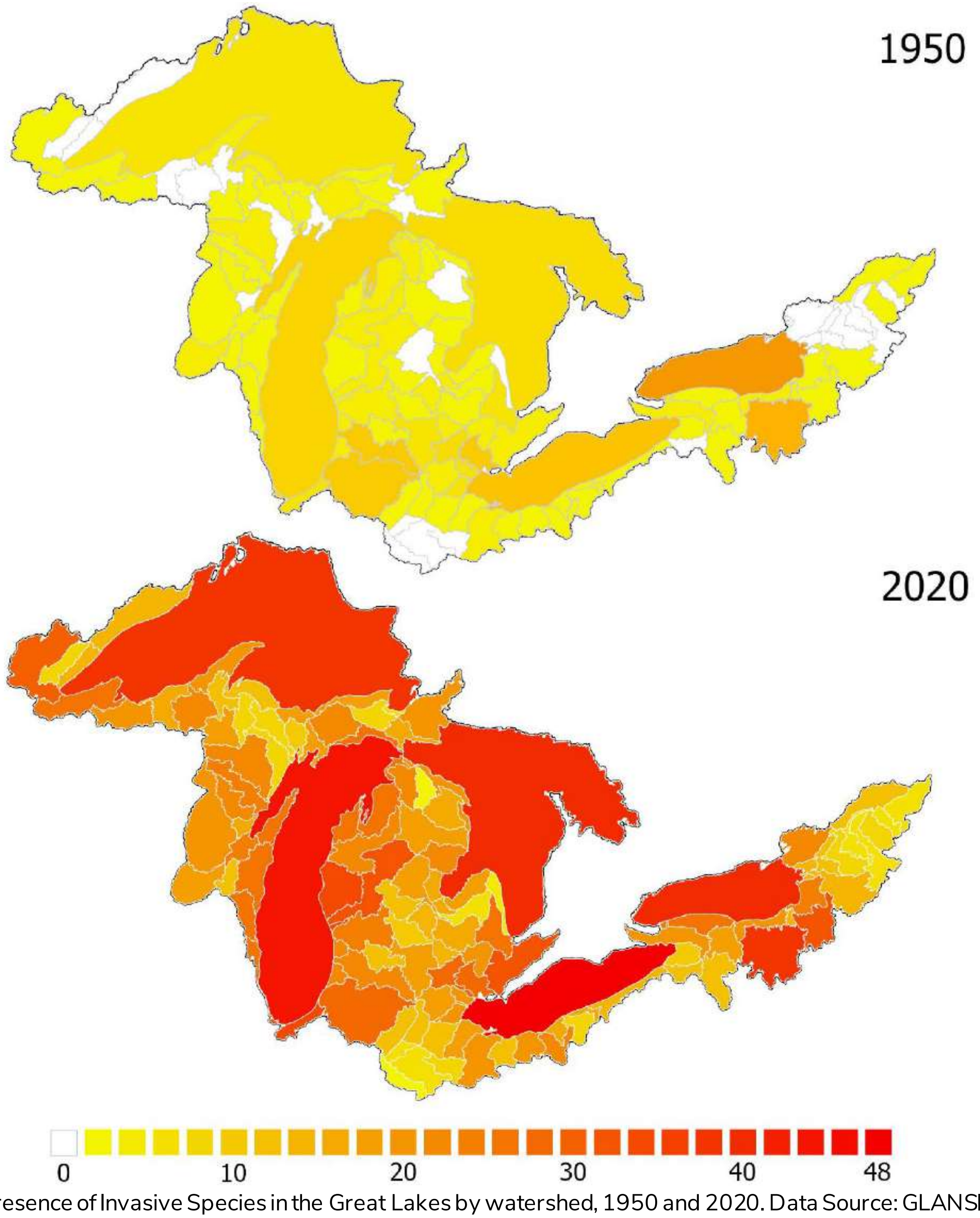


Figure 2. Presence of Invasive Species in the Great Lakes by watershed, 1950 and 2020. Data Source: GLANSIS 2021

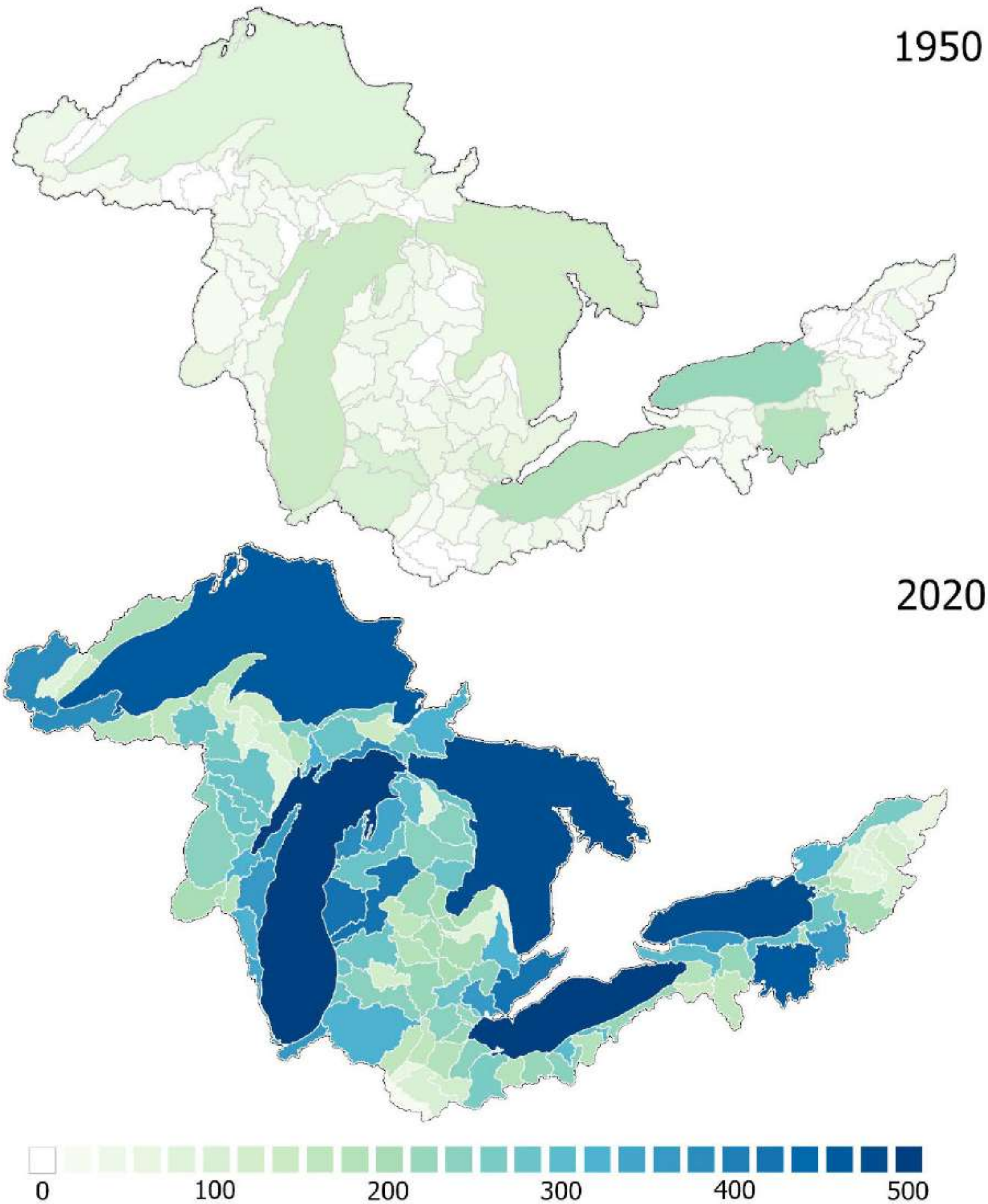
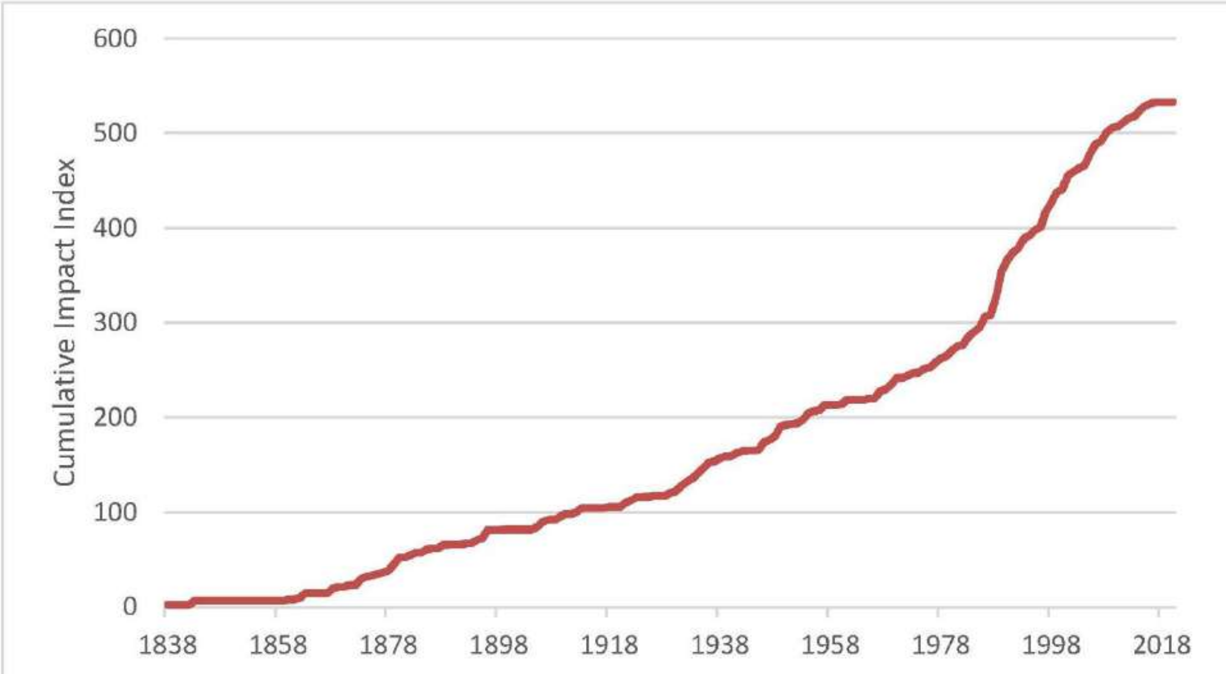


Figure 3. Cumulative Impact Index by watershed (HUC 8) for US waters of the Great Lakes Basin. Data Source: GLANSIS 2021



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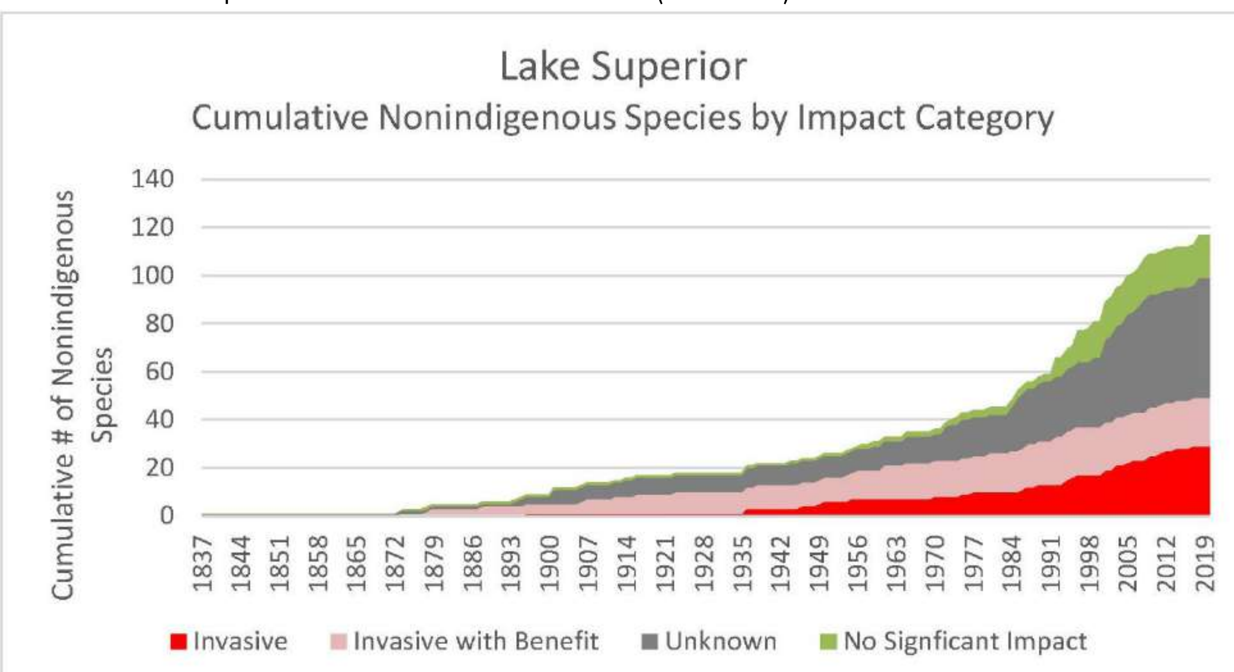


Figure 5. Cumulative non-indigenous species by impact category – Lake Superior basin.
Data Source: GLANSIS 2021

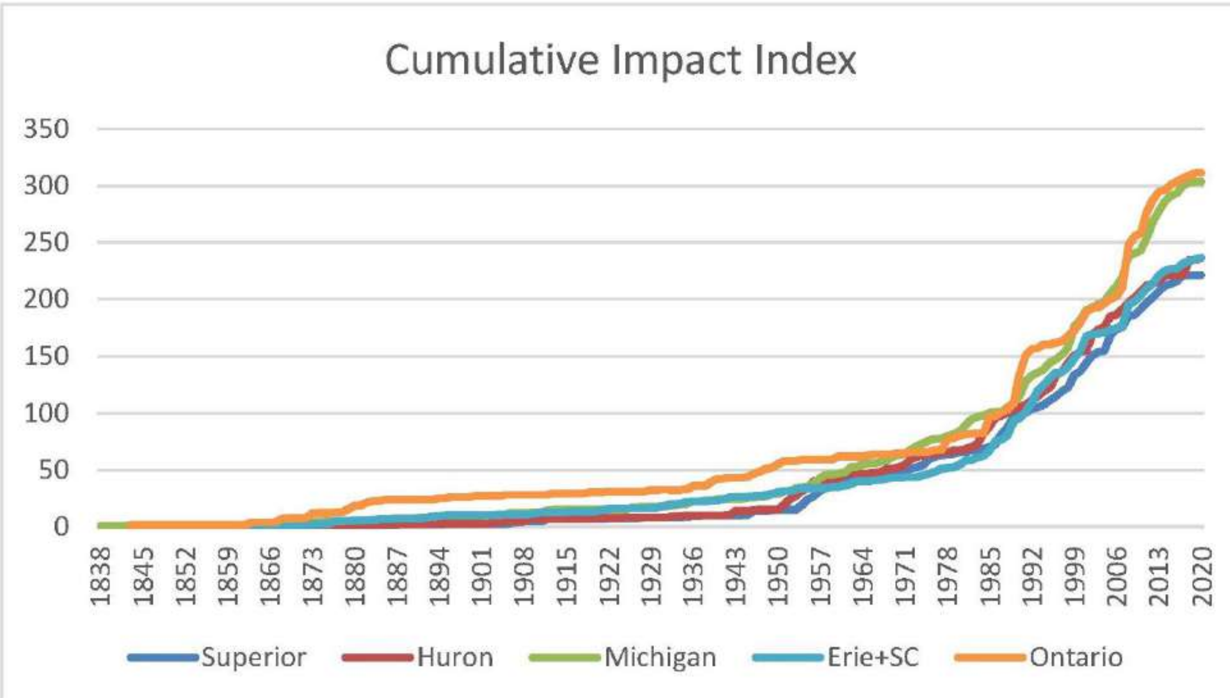
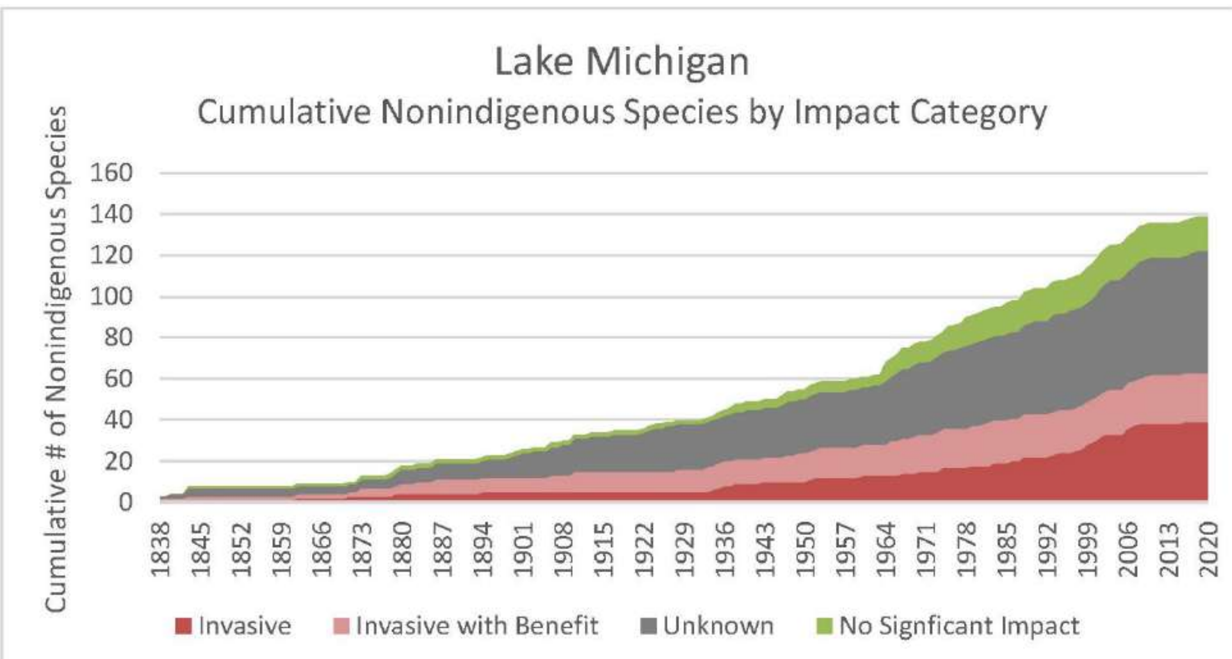


Figure 6. Cumulative impact index for watersheds of the Great Lakes basin (U.S.). Data Source: GLANSIS 2021



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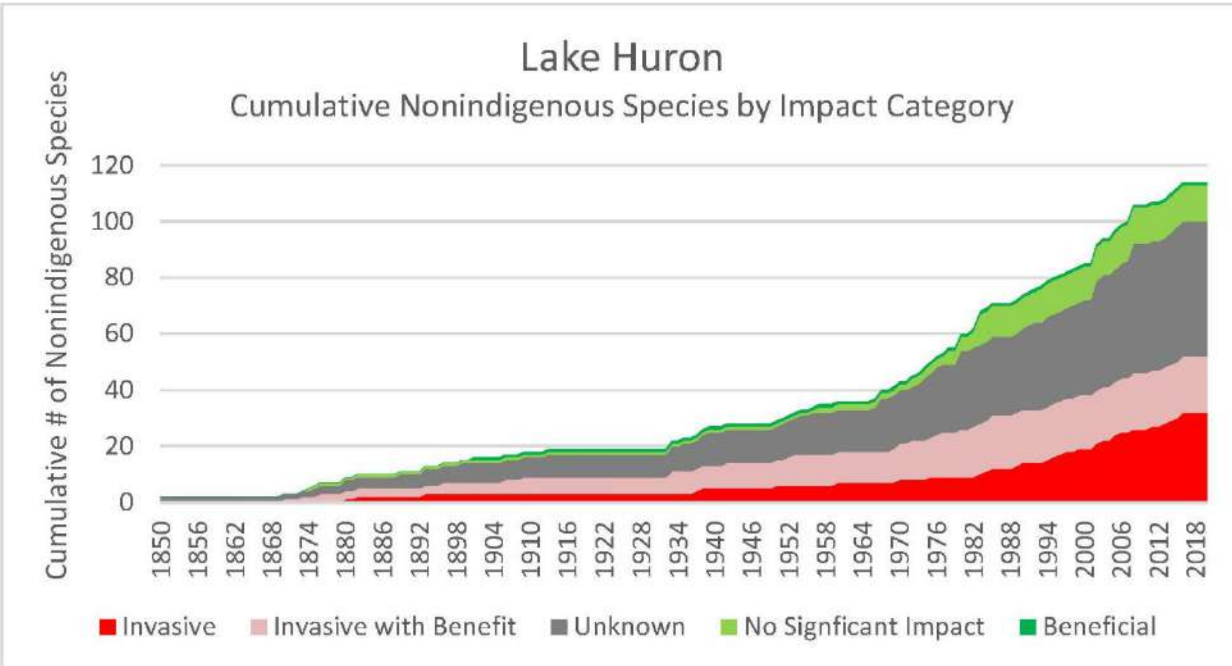
Data Source: GLANSIS 2021



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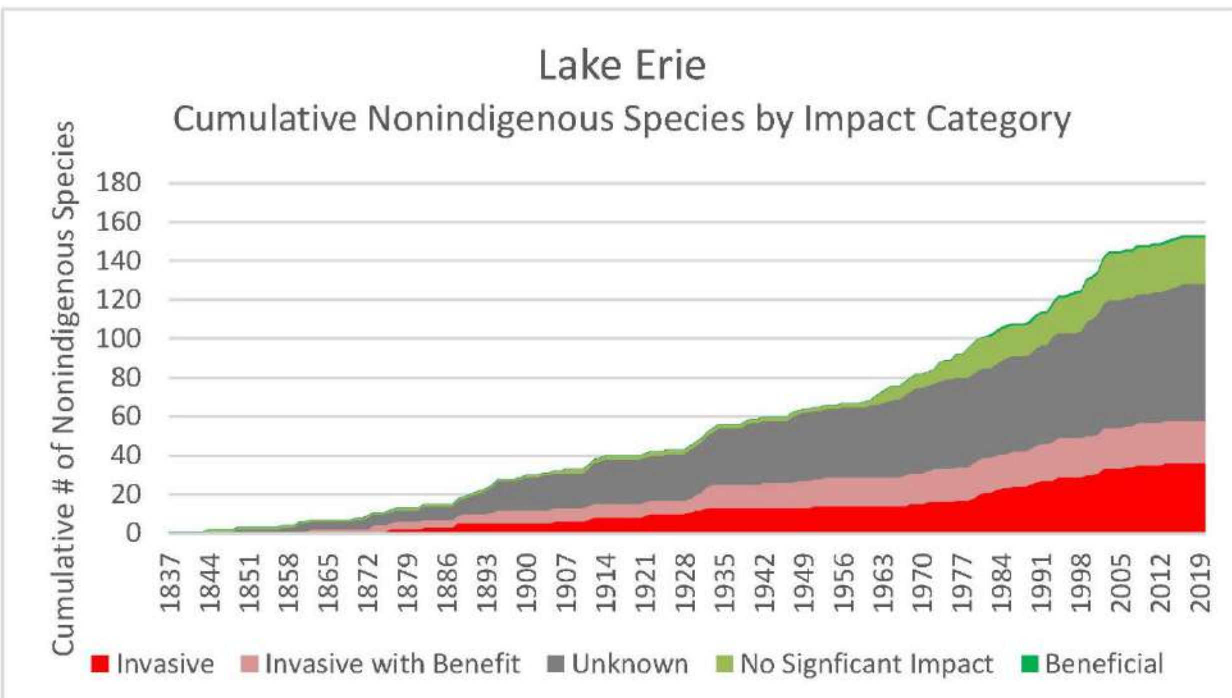
Data Source: GLANSIS 2021



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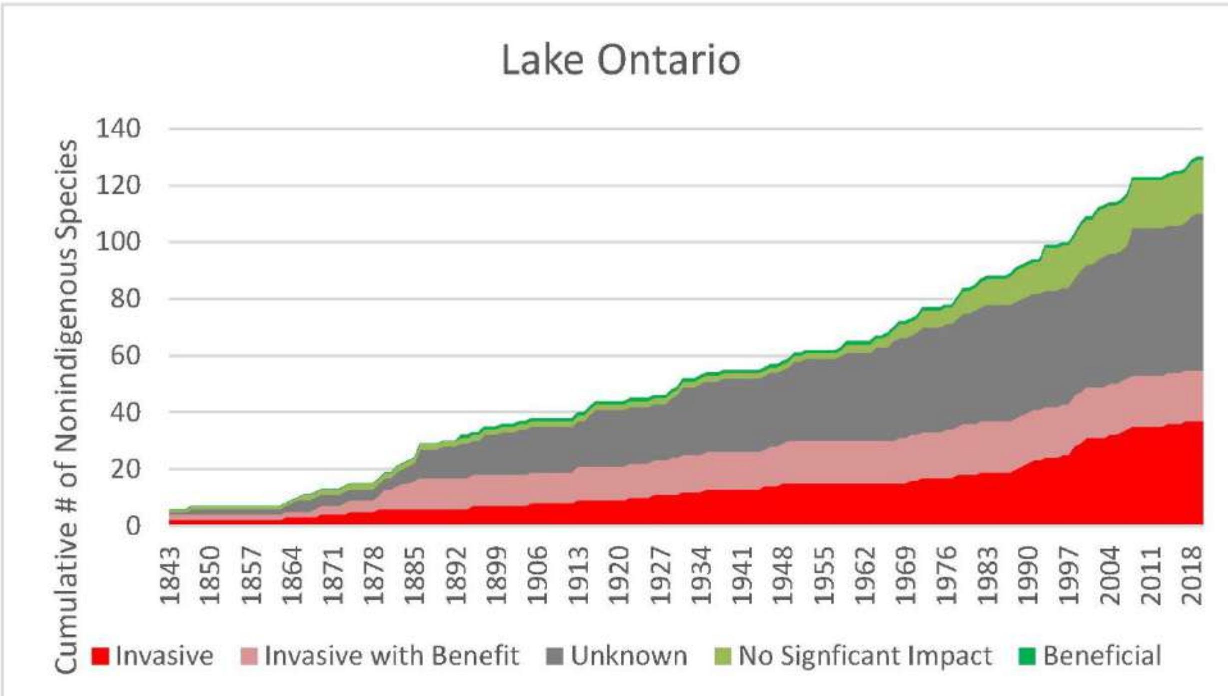

Figure 10. Cumulative non-indigenous species by impact category – Lake Ontario basin.
Data Source: GLANSIS 2021

Sub-Indicator: Sea Lamprey

Overall Assessment

Status: Fair

10-Year Trend: Improving

Long-term Trend (1985-2020): Improving

Rationale: Annual Sea Lamprey control activities in the Great Lakes have successfully suppressed Sea Lamprey populations from levels by about 90% since pre-control efforts. Currently, 3-year average adult Sea Lamprey indices are meeting targets in Lakes Michigan, Erie, and Ontario and are above targets in Lakes Superior and Huron. The basin-wide 10-year and long-term adult Sea Lamprey abundance trends are declining. More suppression is needed to bring Sea Lamprey populations to targets in all lakes.

Lake-by-Lake Assessment

Lake Superior

Status: Poor

10-Year Trend: Deteriorating

Long-term Trend (1986 – 2020): Deteriorating

Rationale: Adult Sea Lamprey index is above target and deteriorating.

Lake Michigan

Status: Good

10-Year Trend: Improving

Long-term Trend (1995-2020): Improving

Rationale: Adult Sea Lamprey index is meeting the target and improving.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend: Improving

Long-term Trend (1985-2020): Improving

Rationale: Adult Sea Lamprey index is above target and improving.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Good

10-Year Trend: Improving

Long-term Trend (1991-2020): Unchanging

Rationale: Adult Sea Lamprey index is meeting target and improving.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Good

10-Year Trend: Improving

Long-term Trend (1987-2020): Unchanging

Rationale: Adult Sea Lamprey index is meeting the target and improving.

Status Assessment Definitions

Good: Adult Sea Lamprey index is below target (see lake-specific targets below)

Fair: Adult Sea Lamprey index is above target, but the 10-year trend is improving

Poor: Adult Sea Lamprey index is above target and the 10-year trend is deteriorating

Trend Assessment Definitions

Improving: Adult Sea Lamprey index shows a change toward acceptable conditions

Unchanging: Adult Sea Lamprey index shows no change

Deteriorating: Adult Sea Lamprey index shows a change away from acceptable conditions

Endpoints and/or Targets

Endpoints for this sub-indicator refer to the adult Sea Lamprey index targets, which correspond to the fish community objectives for each lake. Index targets were agreed upon by the cooperating fisheries management agencies on each lake committee and represent index levels during periods where mortality caused by Sea Lampreys was tolerable and would allow attainment of critical fish community objectives including restoration and maintenance of native species populations and valuable fisheries. Index targets remain the same from year-to-year unless a new index stream is added and/or removed, or estimator methodology changes.

Lake Superior: Suppress Sea Lampreys to population levels that cause only insignificant mortality on adult Lake Trout. The adult Sea Lamprey index target in Lake Superior is 10,000 Sea Lampreys.

Lake Michigan: Suppress the Sea Lamprey to allow the achievement of other fish-community objectives. The adult Sea Lamprey index target in Lake Michigan is 32,000 Sea Lampreys.

Lake Huron: Reduce Sea Lamprey abundance to allow the achievement of other fish-community objectives. The adult Sea Lamprey index target in Lake Huron is 31,000 Sea Lampreys.

Lake Erie: The fish community objectives for Lake Erie does not include a specific Sea Lamprey objective, however it does acknowledge that effective Sea Lamprey control is needed to support the fish-community objectives for Lake Erie. The adult Sea Lamprey index target in Lake Erie is 3,300 Sea Lampreys.

Lake Ontario: Suppress abundance of Sea Lamprey to levels that will not impede achievement of objectives for Lake Trout and other fish. The adult Sea Lamprey index target in Lake Ontario is 14,000 Sea Lampreys.

Sub-Indicator Purpose

- To estimate and track the relative adult Sea Lamprey abundance for each lake.
- To monitor the damage caused by Sea Lamprey to the aquatic ecosystem.
- To monitor the success of Sea Lamprey control actions.

Ecosystem Objective

This sub-indicator supports Great Lakes Fishery Commission (GLFC) and fishery management agencies fish community objectives that were established under "A Joint Strategic Plan for the Management of Great Lakes Fisheries" (Great Lakes Fishery Commission - Joint Strategic Plan Committees (glfc.org)). Fish community objectives call for suppressing Sea Lamprey populations to levels that cause only insignificant mortality on fish to achieve objectives for Lake Trout and other members of the fish community.

This sub-indicator best supports work towards General Objective #7 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should "be free from the introduction and spread of aquatic invasive species and free from the introduction and spread of terrestrial invasive species that adversely impact the quality of the Waters of the Great Lakes."

Measure

Indices of adult Sea Lamprey abundance are currently calculated as the sum of the spawning run estimates for a subset of streams in a given lake basin (called index streams; Figure 1). The number of adult Sea Lampreys migrating into each index stream are estimated with traps using an adjusted pooled Petersen estimator (~90% of estimates) or are derived from a lake-specific, weighted least squares, two-way analysis of variance (ANOVA) with main effects only (~10% of estimates). More detail on the current methodology used to calculate adult Sea Lamprey indices can be found in Adams et al. (2021).

The GLFC assesses the status of sea lamprey populations in each lake (http://www.glfc.org/status.php) by comparing the 3-year average adult Sea Lamprey index to its lake-specific target (meeting or above target) and evaluating the 5-year linear trend in abundance (decreasing, increasing, or steady); 3-year averages and 5-year trends are used to address variability in the annual point estimates that may not be reflective of the actual population. 3-year average adult Sea Lamprey indices and 5-year trends are updated on an annual basis. For the purposes of this sub-indicator report 3-year average adult Sea Lamprey indices relative to targets and 10-year and long-term trends are used to assess the status of sea lamprey populations in each lake.

Before 2015, this indicator encompassed whole-lake adult Sea Lamprey abundances calculated as the sum of spawning run estimates for all Sea Lamprey-producing streams in a given basin. Abundances were obtained in streams with traps using a modified Schaefer estimator or extrapolation from previous trap capture efficiency estimates, and in streams without traps using a model that relates spawning run size to stream discharge, larval abundance, and year since last treatment (spawner model; Mullett et al., 2003). The majority of the abundances were obtained using the spawner model. In 2015, the GLFC changed its adult Sea Lamprey monitoring protocols by switching from the spawner model to an adult Sea Lamprey index on a subset of streams in a given basin. The change was made because of the high amount of uncertainty inherent to the spawner model. The index provides a means to track adult Sea Lamprey populations using best available data - actual population assessment data, and providing a better method to track adult Sea Lamprey populations and assess the impacts of the Sea Lamprey control program. In 2017, the GLFC made an additional change to its Sea Lamprey monitoring protocols by switching from a modified Schaefer estimator to an adjusted pooled Petersen estimator. The pooled Petersen estimator performs better than the modified Schaefer and other estimators in terms of accuracy and precision with large sample sizes and was more accurate than the modified Schaefer estimator with small sample sizes. See Adams et al. (2021) for more detail on these methodological changes. The current method to calculate the indices has been applied to historic data to promote comparisons across the time series.

Ecological Condition

The Sea Lamprey is a non-native species and a lethal parasite of many fish species in the Great Lakes (e.g. Bergstedt and Schneider 1988; Kitchell 1990), and has caused ecologic and economic tragedy in terms of its impact on the Great Lakes fish communities and ecosystem (Smith and Tibbles 1980). Before control, Sea Lampreys killed an estimated 103 million pounds (47 million kilograms) of fish per year with the average Sea Lamprey killing up to 40 pounds (18 kg) of fish during its parasitic stage. Sea Lampreys prefer trout, salmon, whitefish, and Lake Sturgeon but they also parasitize smaller fish like cisco, Walleye, and Yellow Perch (GLFC 2015). The first complete round of stream treatments with the lampricide TFM (as early as 1960 in Lake Superior) successfully suppressed Sea Lamprey populations to about 10% of pre-control abundances in all of the Great Lakes except Lake Erie, and subsequent lampricide treatments conducted on a regular basis across the Great Lakes have successfully maintained Sea Lamprey populations at this level in all lakes except Lake Erie. The Sea Lamprey, however, continues to be a significant source of mortality for many fish species (Bergstedt and Schneider 1988; Kitchell 1990) and its continued control is needed to restore and maintain the Great Lakes fish communities and ecosystem.

Indices of adult Sea Lamprey abundance relative to lake-specific targets are the primary performance indicators of the Sea Lamprey control program (Figure 2). Index estimates are calculated as the sum of the spawning run estimates for a subset of streams in a given lake basin. The numbers of adult Sea Lampreys migrating into each index stream are estimated with traps using mark/recapture methods. Index estimates are updated on an annual basis.

On all lakes except Huron and Michigan, index targets are the average index estimate in each lake during times when lake-wide Sea Lamprey wounding rates on Lake Trout were tolerable, that is, causing less than 5% annual mortality (or when Lake Trout wounding rates were less than or equal to five wounds per 100 fish). For Lake Huron, Lake Trout wounding rates have not been at tolerable levels for five consecutive years, so the index target is set at 25% of the average index estimate during the late 1980s. For Lake Michigan, Sea Lamprey index estimates are not available during times when Lake Trout wounding rates were tolerable, so the index target is set using index data from the late 1990s corrected for the higher than tolerable Lake Trout wounding rates. Index targets are only updated when an index stream is either added and/or removed from the estimation procedure or if estimator methodology changes.

Sea Lamprey wounding rates on Lake Trout have also been previously included as another measure of the abundance of Sea Lamprey in relation to their prey. However, wounding rates were not used directly to assess Sea Lamprey abundance in previous Sea Lamprey indicator reports. Lake Trout wounding rate trends do not always match Sea Lamprey abundance trends. Lake Trout wounding rates are dependent on Sea Lamprey abundance and abundances of ALL host fish. These relationships are hard to reconcile because of the lack of abundance data on hosts other than Lake Trout, which leads to inconsistencies between Sea Lamprey abundance and Lake Trout wounding rates (e.g., a Lake Trout wounding rate can increase in the presence of a steady Sea Lamprey population if the abundance of other host fish declines). However, Sea Lamprey wounding rates on Lake Trout for each lake along with their targets are graphically summarized in Figure 3 to show some of the impact Sea Lamprey have on Great Lakes fish, specifically Lake Trout.

Lake Superior

In Lake Superior, the 2020 adult Sea Lamprey index could not be calculated because of the COVID-19 pandemic. The 3-year average adult Sea Lamprey index is above target and has been deteriorating over the past 10 years. Sources of Sea Lampreys that are of concern include the Bad and Sturgeon rivers and lentic (estuaries, bays, and slower moving tributaries) populations in the Kaministiquia, Nipigon, Gravel, and Batchawana rivers where populations are sparsely distributed and lampricide treatments are less effective. Overall, lampricide control effort has increased since 2005 with additional tributary and lentic areas being treated. Intensive lampricide treatment effort was focused on Lake Superior during 2016 and 2019.

Lake Michigan

In Lake Michigan, the 2020 adult Sea Lamprey index could not be calculated because of the COVID-19 pandemic. The 3-year average adult Sea Lamprey index is meeting target and has been improving over the past 10 years. Sources of Sea Lampreys that are of concern include the Manistique River, other productive tributaries in the northern and eastern parts of the lake, and the St. Marys River (Lake Huron). Lampricide control effort has increased starting in 2006, and intensive lampricide treatment effort was focused on Lake Michigan during 2017. In addition, the Manistique River has been treated seven times since 2003. Reductions in Sea Lamprey abundance during the past ten years are likely a result of increased lampricide treatment efforts.

Lake Huron

In Lake Huron, the 2020 adult Sea Lamprey index was calculated. The 3-year average adult Sea Lamprey index is above target and has been improving over the past 10 years. Sources of Sea Lampreys that are of concern include the St. Marys River, other productive tributaries in the northern part of the lake (e.g. Garden and Mississagi rivers), and the Manistique River (Lake Michigan). Lampricide control effort has increased starting in 2006 with additional treatments. A large-scale effort to treat the North Channel area of Lake Huron (including the St. Marys River) occurred from 2010-2011 along with geographically expanded treatment in the northern parts of Lakes Huron and Michigan in 2012-2013 and 2014-2015. Application of this strategy successfully reduced larval Sea Lampreys in the St. Marys River to historically low levels.

Lake Erie

In Lake Erie, the 2020 adult Sea Lamprey index estimate was calculated, but is partially based on ANOVA modeling estimates due to the COVID-19 pandemic. The 3-year average adult Sea Lamprey index is meeting target and has been improving over the past 10 years. Sources of Sea Lampreys that are of concern include hard-to-treat tributaries (e.g. Cattaraugus Creek), tributaries with non-target species of concern (Conneaut Creek), and the St. Clair and Detroit River System. Lampricide control effort dramatically increased during 2008-2010 with the implementation of a large-scale treatment strategy where all known Sea Lamprey-producing tributaries to Lake Erie were treated in consecutive years. Increased control effort was also applied during 2013 with the treatment of twelve tributaries. Assessment and treatment strategies continue to be developed for the St. Clair and Detroit River System, which could be a significant contributor of Sea Lamprey to Lake Erie.

Lake Ontario

In Lake Ontario, the 2020 adult Sea Lamprey index estimate was calculated. The 3-year average adult Sea Lamprey index is meeting the target and has been improving over the past 10-years. A source of Sea Lampreys that is of concern is the Niagara River – the larval Sea Lamprey population is currently small, but could become an issue with improved habitat and water quality. Steady lampricide control effort on Lake Ontario has maintained the adult Sea Lamprey index at or near the target.

Linkages

Lake Trout; Walleye; and Lake Sturgeon

Sea Lampreys remain a significant source of mortality (basin-wide and/or locally) on many fish species of the Great Lakes including Atlantic, Chinook, and Coho Salmon, Burbot, ciscoes, Lake Sturgeon (threatened in some parts of the Great Lakes basin), Lake Trout, Lake Whitefish, Steelhead, Walleye, etc. Short lapses in Sea Lamprey control can result in rapid increases in Sea Lamprey abundance and the damage they inflict on fish. Continued stream and lentic area treatments are necessary to overcome the reproductive potential of the Sea Lamprey and to ensure the achievement of population management objectives for many different species, and to preserve functioning ecosystems.

Aquatic Habitat Connectivity; Water Quality

The potential for Sea Lampreys to colonize new locations is increased with improved aquatic habitat connectivity through the removal of dams and improved water quality. For this reason, improvements in habitat connectivity must be weighed with the costs of potential increases in Sea Lamprey habitat. The failure of the Manistique River Dam to block Sea Lampreys and the subsequent Sea Lamprey production from the river is an example of the linkages between Sea Lamprey and aquatic habitat connectivity. Additionally, as water quality improves, streams and lentic areas once inhospitable to Sea Lampreys may become viable spawning and nursery habitats. During the mid-2000s, a significant larval population requiring regular lampricide treatment was established for the first time in the estuary of the Kaministiquia River (Lake Superior) after a local paper mill began tertiary treatment of its effluent. The establishment of larval populations in the St. Marys, St. Clair, and Lower Niagara rivers followed concerted efforts to improve water quality, and with observations of successful reproduction by Lake Sturgeon, Lake Whitefish, and Brindled Madtom, evidence of Sea Lamprey reproduction in the Detroit River is likely inevitable.

Climate Change

Rising water temperatures in the Great Lakes have recently been associated with increasing size of adult Sea Lampreys (Kitchell et al. 2014). As water temperatures rise, Sea Lampreys may grow larger increasing metabolism and becoming more fecund (fertile), which may increase the number of Sea Lampreys and the damage they cause to host fish. Increased precipitation amounts can cause dam failures such as those experienced on the Tittabawassee River (Michigan) during 2020, allowing passage of sea Lamprey into new locations. Precipitation events can also move Sea Lamprey larvae downstream or into lentic areas where treatment is more difficult. See Lennox et al. (2020) for a recent review of the potential impacts of climate change on sea lamprey in the Great Lakes.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	No			

Data Limitations

- Adult Sea Lamprey trapping and index estimations, Lake Trout population assessments, and Lake Trout wounding data collections were all impacted by COVID-19 during 2020. Estimates may not be available, may be based on modeling instead of actual catches, or may be based on a smaller sample size or geographic sampling area than is typical. These limitations are noted above.
- There are no direct measures of Sea Lampreys during the period when they are parasitic on Great Lakes fish. Adult Sea Lamprey indices are used as a surrogate. Relating adult Sea Lamprey indices to the parasitic population assumes insignificant or at least constant mortality between the parasitic and adult life stages.
- Adult Sea Lamprey indices are limited to streams where it is possible to trap migrating adult Sea Lampreys and generate a mark/recapture population estimate. Consequently, the adult Sea Lamprey indices only use a subset of streams in a given lake basin.
- Direct mark/recapture data for parasitic or newly metamorphosed Sea Lampreys might provide better estimates of damage to other fishes, but these direct estimates may only be obtained with confidence when large numbers of individuals can be recaptured. To date, assumptions of mark/recapture methods, particularly the assumption of equal survival among marked and unmarked individuals, cannot be met and estimates of juvenile Sea Lamprey are highly uncertain.
- Sea Lamprey wounding rates on other important fish species could also inform the impacts of Sea Lamprey to the Great Lakes ecosystem, but are excluded from this sub-indicator for the sake of simplicity because:

1) Lake-wide wounding rates are currently only available for Lake Trout. Lake Trout are considered the most vulnerable and preferred species, thus making them a good indicator species.

2) The observations of wounding rates are hard to interpret because they are influenced by the abundance of fish in the suitable size range for Sea Lampreys and may vary depending on the mix of these fishes in an area.

3) Classification of Sea Lamprey wounds (i.e., marks, Type A or Type B) is subjective and may vary among individuals and agencies making the observation.

4) The relationship between an observed wound and the mortality caused by Sea Lampreys involves understanding the lethality of an attack. Experimental and observational data regarding the probability of trout and salmon surviving an attack is available. However, these experimental observations are limited and verification of lethality in the field will improve understanding of Sea Lamprey mortality.

Additional Information

Increases in lampricide treatments have reduced 3-year average adult Sea Lamprey indices to within target ranges in three of the five Great Lakes (Michigan, and Ontario). The effects of increased lampricide treatments are observed in index estimates beginning two years after they occur. Efforts to identify new/unidentified sources of Sea Lampreys also need to continue. In addition, research to better understand Sea Lamprey/host interactions, recruitment dynamics, population dynamics of Sea Lampreys that survive treatment, and refinement of and research into other control methods are all keys to achieving and maintaining adult Sea Lamprey indices at targets.

Acknowledgments

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List of Figures

Figure 1. Adult Sea Lamprey index streams.

Source: Great Lakes Fishery Commission

Figure 2. 3-year average adult Sea Lamprey indices plotted on Sea Lamprey spawning year. Horizontal lines represent the targets for each lake. Note the scale differences for each lake.

Source: Great Lakes Fishery Commission

Figure 3. 3-year average A1-A3 Sea Lamprey wounds per 100 Lake Trout > 532 mm (Superior, Huron, Michigan, and Erie) and 3-year average A1 Sea Lamprey wounds per 100 Lake Trout > 432 mm (Ontario) from standardized assessments. Horizontal lines represent the wounding rate target for each lake. Note the scale differences for each lake.

Source: Great Lakes Fishery Commission

Last Updated
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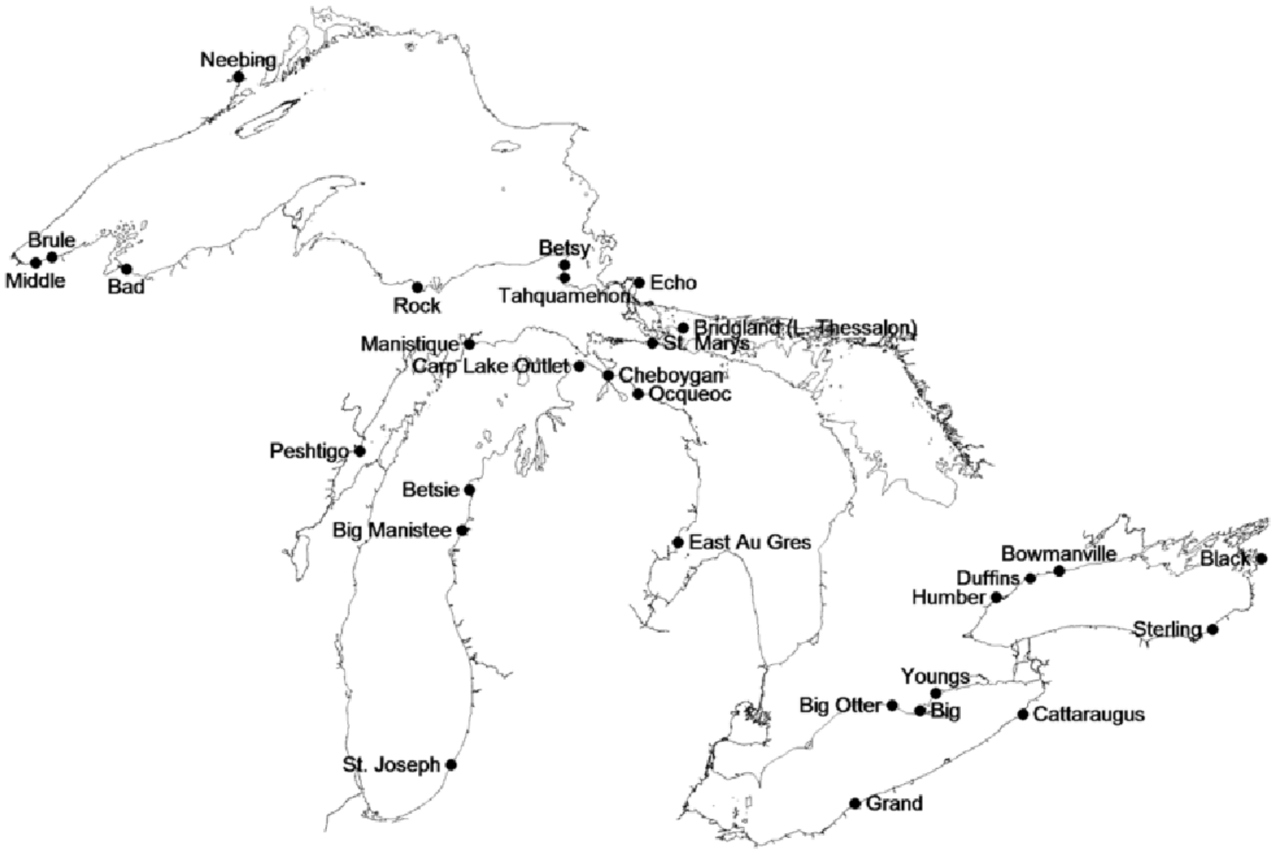
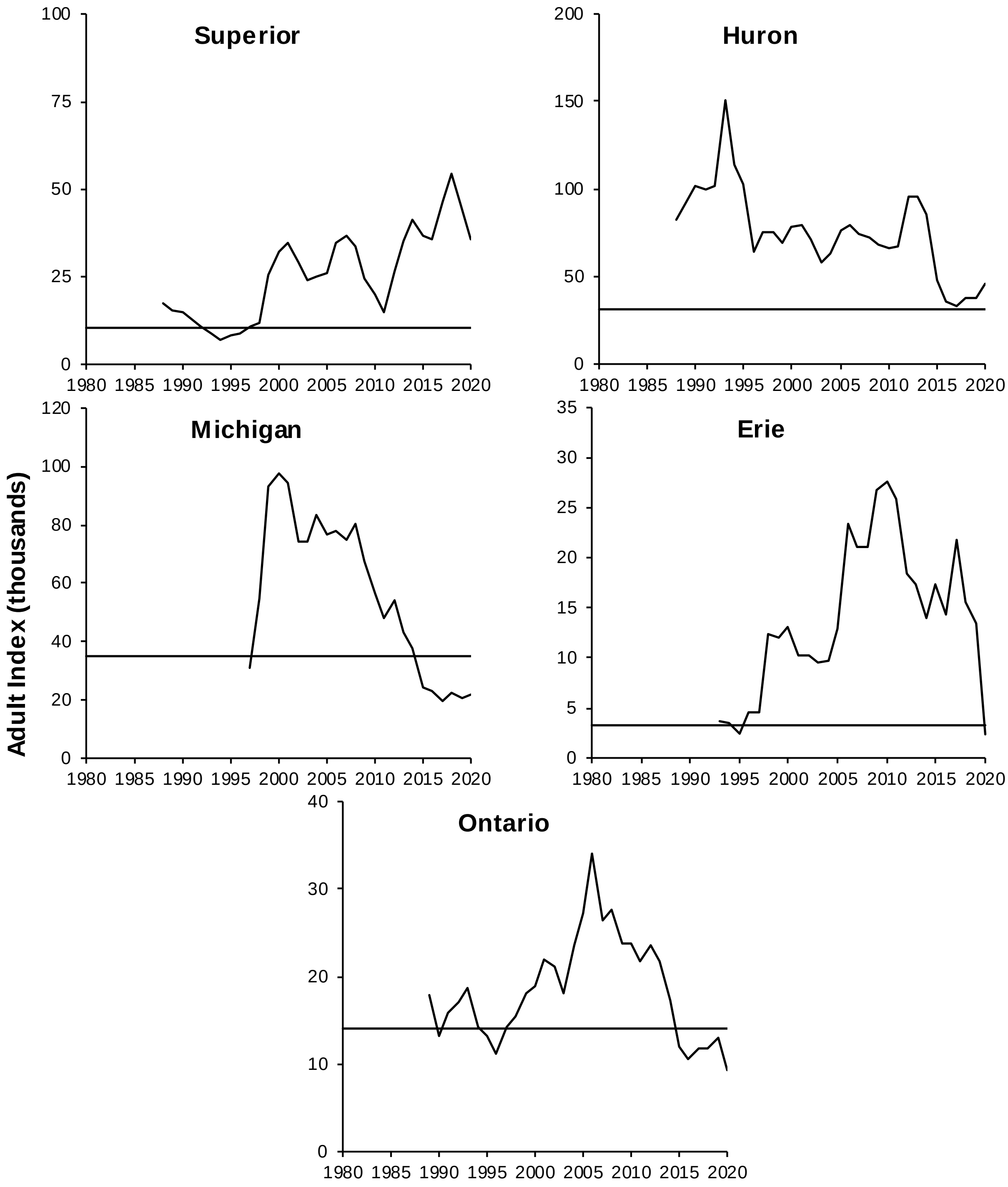


Figure 1. Streams selected for inclusion in the index of adult Sea Lamprey abundance. Source: Great Lakes Fishery Commission.



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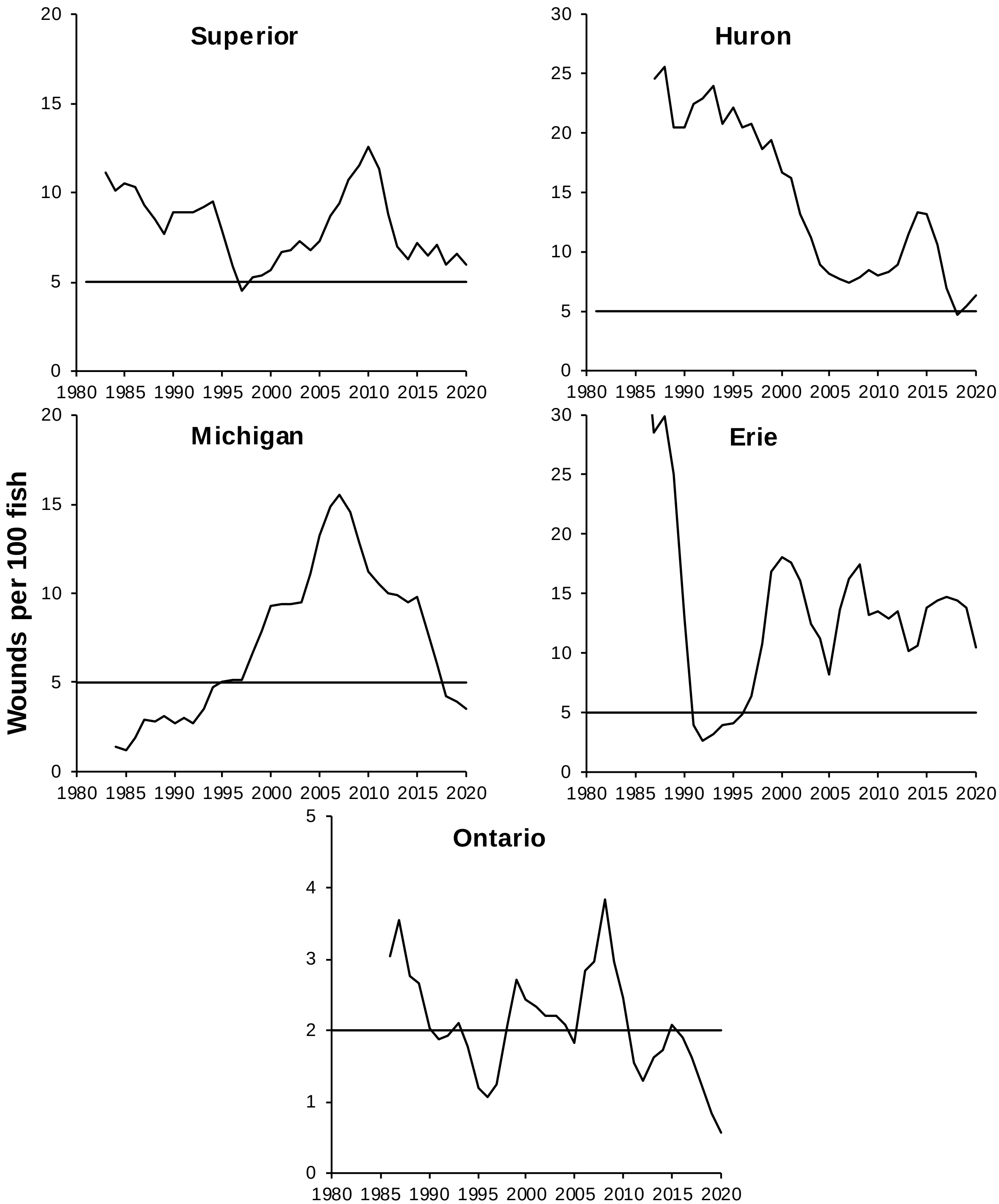

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Sub-Indicator: Dreissenid Mussels

Overall Assessment

Status: Poor

Trends:

10-Year Trend: Deteriorating

Long-term Trend (1989-2019): Deteriorating

Rationale: The status of dreissenid mussels (quagga mussel- *Dreissena rostriformis bugensis*, and zebra mussel- *D. polymorpha*) varies among water depths and lake regions. In general, dreissenid densities in Lakes Michigan, Huron, and Ontario appear to have stabilized or are decreasing at depths < 90 m. However, dreissenids remain the dominant component of the benthos and it is not clear if their impacts are lessening as a result of observed declines. The deep zone (>90 m) appears to be a continuing invasion front for quagga mussels; density and biomass have been gradually increasing, though at a slower population growth rate. In these three lakes, quagga mussels have displaced zebra mussels except in shallow, nearshore areas and bays. Dreissenid populations in eastern Lake Erie have been relatively stable at moderate levels for the past decade, very low in the central basin and declining in the western basin. Dreissenid populations in Lake Superior remain at low levels. The data presented here are primarily based on lake-wide surveys conducted over time by the US Environmental Protection Agency (EPA) and National Oceanic and Atmospheric Administration (NOAA), and the Canadian Department of Fisheries and Oceans (DFO). Since 2002, lake-wide benthic surveys have been a part of the Cooperative Science and Monitoring Initiative (CSMI), which occurs every 5 years for each lake on a rotating cycle. New to this reporting cycle are data from regional assessments and the Canadian Ontario Ministry of the Environment, Conservation and Parks (MECP) Nearshore Great Lakes Monitoring Network from 1992-2016 (https://data.ontario.ca/dataset/benthic-invertebrate-community-great-lakes-nearshore-areas).

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend (2009-2019): Unchanging

Long-term Trend (1989-2019): Unchanging

Rationale: Zebra mussels were first found in Duluth-Superior Harbor in 1989, and quagga mussels were subsequently found in the same area in 2005 (Grigorovich et al. 2008). Since then, the spread and population growth of both dreissenid species has been minimal. Both species are most abundant in the Duluth Harbor area or just outside the harbor in the immediate vicinity of nearshore Lake Superior. Canadian nearshore sampling reveals dreissenids to be largely absent (Figure 1), with the exception of a few specimens found at a site near Lake Superior Provincial Park in eastern Lake Superior in 1992 (MECP). Other discovered populations include in an Isle Royale bay in the northwest in 2009, Whitefish Bay in the east in 2002, the Apostle Islands, and a veliger from Nipigon Bay in the north (Trebitz et al. 2019). EPA GLNPO Long-term Monitoring surveys found one small quagga mussel in 2019 near Silver Bay in the western end (Burlakova unpub. data). Veligers are highly uncommon in the EPA GLNPO zooplankton surveys; only 16 occurrences were recorded for 2002-2017- one veliger was found in eastern Lake Superior in 2018, and none found in 2019 (J. Watkins pers. comm.). The broad spatial coverage but low densities of these occurrences indicate that Lake Superior is likely marginally habitable for dreissenids. Low calcium and other water quality characteristics pose a challenge to spawning success and veliger survival (Trebitz et al. 2019). The next lake-wide CSMI benthic survey is planned for 2021.

Lake Michigan

Status: Poor

10-Year Trend (2005-2015): Deteriorating

Long-term Trend (1994-2015): Deteriorating

Rationale: Results from the most recent lake-wide survey (2015) reveal depth-specific trends in dreissenid density and biomass. All mussels collected in 2015 were quagga mussels. While there were notable declines in density at depths 31-90 m between 2010 and 2015 (Figures 2-4), biomass held fairly steady (Figure 3). The overserved differences in abundance and biomass are a result of mean mussel size increasing. Dreissenid density and biomass continued to increase at depths > 90 m (Figures 2-4). Despite declines at sites in the 31-90 m interval, the quagga mussel population still well exceeds maximum densities previously reached by zebra mussels in that interval. The next lake-wide CSMI benthic survey is planned for 2021.

Lake Huron (including St. Marys River)

Status: Poor

10-Year Trend (2007-2017): Deteriorating

Long-term Trend (2000-2017): Deteriorating

Rationale: The most recent data available from a lake-wide survey of dreissenid populations in Lake Huron is from 2017. Between 2007 and 2012, dreissenid densities (all quagga) appeared to have stabilized at 31-90 m, but then increased slightly between 2012 and 2017 (from 1100/m^2 to 1980 /m^2; Figure 2, upper panel; Karatayev et al. 2020). From 2007 to 2017, decreased densities at 31-50 m were offset by increased densities at 51-90 m. Densities continue to increase at depths > 90 m (Figures 2 and 5), notably to the point that the deep-water dreissenid density in Lake Huron has caught up with that in Lakes Michigan and Ontario (Figure 2, lower panel). In Canadian nearshore waters of Lake Huron, dreissenids increased between 1991 and 2002 and have since held at moderate densities (~350-650/m^2; Figure 1; MECP). Dreissenids offshore in Georgian Bay are found at overall lower densities than in the main basin, but have followed similar depth-specific changes over time. In the Georgian Bay nearshore, dreissenid densities peaked in 2002 and then sharply declined (1230/m^2 in 2002 to 187/m^2 in 2015; Figure 1; MECP). While mussels were not present in North Channel at the sites sampled in 2007 and 2012, a few small quagga mussels and one zebra mussel appeared in the 2017 survey. No dreissenids have been found in the St. Marys River or North Channel in Canadian nearshore waters (Figure 1; MECP). The next lake-wide CSMI benthic survey is planned for 2022.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Fair

10-Year Trend (2011-2019): Unchanging

Long-term Trend (1992-2019): Improving

Rationale: The most recent data available is from a CSMI lake-wide survey of dreissenid populations in Lake Erie in 2019 (Karatayev et al. 2021a). Overall, lake-wide densities of dreissenids were much lower in 2019 compared to the peaks in 1993 (zebras) and 1998 (quaggas) and similar to densities observed in 2011. The three basins of Lake Erie exhibit different dreissenid mussel trends. In the western basin both species declined since 2004 and in 2019 reached minimum density and biomass (Figure 6). Canadian nearshore data from Lake Erie reflect this trend; peak density was recorded in 1991 (141,057/m^2) followed by declines through the 1990's and consistently low densities since (Figure 7; MECP). Nearshore surveys in the St. Clair-Detroit River Ecosystem show consistent low densities in the Detroit River and high variability over time in Lake St. Clair and the St. Clair River (Figure 7; MECP). In 2019, zebra mussels were dominant over quagga mussels in the western basin for the first time since the 1990's (72% of the combined dreissenid density). In contrast, zebra mussels are largely absent in the eastern basin (1% of all mussels) and are not abundant in the central basin (20% of all mussels). Dreissenid populations in the western basin are dominated by small mussels, which, along with overall low mussel densities, suggests periodic die-offs every 2-3 years most likely due to occasional hypoxia. In the central basin, dreissenid densities have declined 5-fold since their peak in 2002. Similar to the western basin, *Dreissena* population in the central basin is now largely dominated by small mussels, especially at depths >20 m which are subject to seasonal hypoxia. The eastern basin consistently supports the largest population of *Dreissena*, though there has been a strong decline since a peak in 2002 (Figure 6). The size structure of the eastern basin mussels indicates a lack of hypoxia events and consistent recruitment at depths < 40 m, while at deeper areas the population is dominated by mussels >16 mm suggesting lack of successful recruitment (Burlakova et al. 2017; Karatayev et al 2021a). The next lake-wide CSMI benthic survey is planned for 2024.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Poor

10-Year Trend (2008-2018): Deteriorating

Long-term Trend (1994-2018): Deteriorating

Rationale: According to the 2018 lake-wide survey of dreissenid populations in Lake Ontario, quagga mussel densities at 31-90 m were lower than in 2008 and very similar to 2013 (Figure 2; Nalepa and Baldridge 2016; Karatayev et al. 2021b). Densities at this depth interval appear to have peaked in 2003. Nearshore data along the Canadian shore of Lake Ontario also show a peak in dreissenid density in the early 2000's, followed by a decline (Figure 8; MECP). Despite declines in density, biomass at depths of 31-90 m has increased since 2008 as a result of an increase in average mussel length (Karatayev et al. 2021b). The population at > 90 m appears to be expanding, with continued, yet gradual increases observed since 1999 (Figure 2). In fact, the lake-wide dreissenid mussel density and biomass estimates in 2018 were the highest ever recorded (Karatayev et al. 2021b). No zebra mussels have been collected in a lake-wide survey since 2003 (Karatayev et al. 2021b) and nearshore monitoring has produced only a few zebra mussels in Lake Ontario since 2006, though zebra mussel densities were very high in the Bay of Quinte in 2009 and 2012 (MECP). Dreissenid densities in the St. Lawrence River have ranged from low to moderate since the early 1990's (Figure 8; MECP). The next lake-wide CSMI benthic survey is planned for 2023.

Status Assessment Definitions

Since there are no known methods to eliminate mussels, these status assessments are relative to historical population data collected in the Great Lakes.

Good: no or few mussels

Fair: moderate abundance

Poor: high abundance

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

Trend Assessment Definitions

Improving: change towards more acceptable conditions, i.e., a statistically significant decrease in dreissenid density/biomass

Unchanging: little or no change in density/biomass compared to previous data, with a preference for a time span of at least 10 years to indicate if the population is showing signs of stabilizing.

Deteriorating: change away from acceptable conditions, i.e., a statistically significant increase in dreissenid mussel density/biomass

Undetermined: Metrics do not indicate a clear overall trend, or sufficient data are not available to report on a trend

Endpoints and/or Targets

A quantitative endpoint has not been established. A proposed endpoint of zero dreissenids is unrealistic. A new reference point under consideration for future SOGL reporting cycles is when a dreissenid population becomes stable or varies within a given range, which is likely to be system-specific. A reference point such as this may change how we assess status and trends. Ultimately, identifying system-specific thresholds will better facilitate the modeling of dreissenid population dynamics and inputs to predictive ecosystem models. Such models are a necessary precursor to effective resource management.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the population status of the invading *Dreissena rostriformis bugensis* (quagga mussel) and *Dreissena polymorpha* (zebra mussel) in the Great Lakes.

Ecosystem Objective

Dreissenids have actively changed the integrity of Great Lakes ecosystems by increasing water clarity, altering nutrient and energy cycling, promoting nuisance levels of benthic algae, and negatively impacting some native species of invertebrates and fish. The intensive filtering activity of quagga mussels has been implicated in the reduction of the spring diatom bloom in Lake Michigan (Vanderploeg et al. 2010). Further, selective feeding by dreissenids on phytoplankton has been linked to changes in phytoplankton community composition, which may have consequences for harmful algal blooms (Tang et al. 2014). Such changes to ecosystem integrity create uncertainty in effective resource management. Thus, the sub-indicator addresses the objective of maintaining healthy and sustainable ecosystems.

This sub-indicator best supports work towards General Objective #7 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from the introduction and spread of aquatic invasive species and free from the introduction and spread of terrestrial invasive species that adversely impact the quality of the Waters of the Great Lakes."

Measure

This sub-indicator will report the density (number/m^2) and, when available, biomass and size distribution of dreissenid mussels in near to offshore benthic habitats of the Great Lakes. These metrics can be used to assess the status of mussels on both the population and individual levels. Annual sampling in environments with newly introduced populations and/or rapidly changing conditions is valuable. It is also important to sample in deeper regions of the lakes (>90 m), as this is the depth of continued population growth for quagga mussels.

The most recent lake-wide conditions reported for Lakes Michigan, Huron, Ontario, and Erie are based on Cooperative Science and Monitoring Initiative (CSMI) data collected in 2015, 2017, 2018, and 2019, respectively. Data are usually summarized by depth zone because dreissenid mussel dynamics are strongly influenced by depth. Offshore data collected by DFO in the 1990's provides reference data for the beginning of the dreissenid mussel invasion in Lake Ontario. Included for the first time in this subindicator are data from the MECP Nearshore Great Lakes Monitoring Network, which includes benthic data collected from the 1990's – 2016 in nearshore regions and connecting waterways in Canadian waters across the Great Lakes basin.

The U.S. EPA Great Lakes National Program Office (GLNPO) long-term monitoring program assesses density of *Dreissena* (and other benthos) in offshore regions of each of the lakes on an annual basis. In CSMI years, data from some of the GLNPO stations are included in with the lake-wide survey results. Otherwise, some GLNPO data are included anecdotally here (e.g., the few adult and veliger *Dreissena* observations in Lake Superior), but are not included in the figures. GLNPO data generally follows similar trends within each lake and are accessible by request from the GLNPO data portal GLENDA.

Ecological Condition

Dreissenid populations in the Great Lakes are presently in various stages of change. In many offshore regions, populations are increasing, but in most nearshore regions, populations seem to be declining or exhibit high levels of variability. While some year-to-year variability can be expected, a goal of this sub-indicator is to determine at what level of density/biomass populations become stable and at equilibrium with the surrounding environment. Such levels, along with associated degrees of uncertainty, can then be used in predictive models to better manage Great Lakes resources. The data for Lake Erie suggests that *Dreissena* may have reached a carrying capacity and are declining, but we need a longer data record for the other lakes to see where the *Dreissena* populations will land. Lake and depth-specific carrying capacities are expected. For example, food availability, temperature and oxygen regimes, and water movement (e.g., wave energy and currents) vary by depth and region, and will influence local *Dreissena* populations. These factors are already implicated in the success of quagga mussels over zebra mussels because quagga mussels have higher energetic efficiency and can persist in areas with lower food availability (Wilson et al. 2006) and tolerate colder temperatures. Predation by fish may also alter dreissenid populations (Rudstam and Gandino 2020). Lake whitefish and round goby are well documented predators on dreissenid mussels (Foley et al. 2017; Pothoven and Madenjian 2008). Round goby have been connected to localized population reductions in Onondaga Lake (Rudstam and Gandino 2020), but their consumptive effects were deemed to be low in Saginaw Bay (Foley et al. 2017). We do not have sufficient data on round goby in Western Lake Erie to draw solid conclusions about what role they may play in the observed dreissenid population declines.

Recent research connects the observed distribution of *Dreissena* in Lake Erie to hypoxia events (Karatayev et al. 2018; 2021a). Regular and long-lasting summer hypoxia in the hypolimnion of the central basin of Lake Erie may explain why the *Dreissena* population is all but absent at sites >20m. The western basin experiences infrequent and brief hypoxia events that may only affect the mussels every couple of years. However, these events are sufficient to cause mass mortality and thereby reduce *Dreissena* longevity (Karatayev et al., 2021a). In contrast, the eastern basin does not experience hypoxia and contains the most stable mussel populations in Lake Erie. As a result, the presence, density, and size structure of dreissenid populations can be good indicators of hypoxic conditions. *Dreissena* populations are exposed to hypoxia in other shallow, productive areas in the Great Lakes, such as Green Bay in Lake Michigan and Hamilton Harbour in Lake Ontario.

Biomass and size-distribution data can be used to assess population dynamics and predict the direction of populations over time. For example, a population with a large number of individuals and a size distribution skewed toward smaller individuals demonstrates high recruitment and possibly low survivability (or if survivability is not compromised then it may illustrate recent colonization). In contrast, populations showing a size-frequency distribution skewed towards larger individuals with fewer numbers suggests an aging population with relatively lower recruitment and greater survivability. Traditional population ecology suggests that stable populations move from a size-frequency distribution of low mean biomass towards one of higher mean biomass. As a population colonizes a new area, high resource availability promotes high recruitment. As resources are sequestered into the population, recruitment decreases with decreasing resource availability and mean biomass increases as fewer new (low biomass) individuals are added to the population and surviving members continue to grow.

Linkages

Linkages to other sub-indicators include:

- Benthos (open water) – the relative abundance of the benthic community other than dreissenids can be affected by dreissenids.
- *Cladophora* – *Dreissena* influences *Cladophora* by multiple main mechanisms: (1) *Dreissena* filtering increases water clarity, which allows more light to reach *Cladophora* and promotes growth at deeper depths; (2) local nutrient enrichment by mussel feces and pseudofeces; and (3) mussel beds provide additional substrate for algal attachment.
- *Diporeia* (open water) – *Diporeia* is an important component of the native benthic community that has been affected by dreissenids. *Dreissena* has replaced *Diporeia* as the dominant benthic organism in offshore habitats.
- Harmful Algal Blooms – although algal blooms are mainly caused by the input of dissolved reactive phosphorus (Scavia et al. 2014), the mechanisms of selective filtering and, to a lesser extent, nutrient excretion by dreissenids have the potential to increase frequency, distribution and severity of algal blooms and favour the predominance of toxin-producing cyanobacteria (Tang et al. 2014).
- Phytoplankton – the abundance and composition of phytoplankton has dramatically changed in areas of the Great Lakes where dreissenids have become abundant
- Nutrients in Lakes – A recent study estimated that dreissenid mussels have become a major agent of phosphorus cycling in all four of the invaded Great Lakes, and their tissues and shells now contain nearly as much phosphorus as the entire water columns above (Li et al. 2021). If the phosphorus dynamics in the Great Lakes are now regulated by dreissenid mussel populations, it is important to continue

monitoring dreissenid biomass and distributions and recognize that extreme changes in dreissenid populations in either direction will have wide-spread ecosystem consequences.

- Impacts of Aquatic Invasive Species – Dreissenid mussels are two of the most impactful invasive species to enter the Great Lakes. The presence of *Dreissena* may enable the success of other invasive species, particularly those from the Ponto-Caspian assemblage.

A change in the thermal regime of the lakes, as realized by changes in the timing of lake stratification and mixing events, could substantially impact dreissenids. Dreissenids have the greatest access to the phytoplankton community when the water column is fully mixed in the spring and fall. During stratification, the grazing potential of mussels is therefore reduced. If increased water surface temperatures in the spring and summer extend the stratified period, mussels may experience reduced food supplies. Further, increasing water temperatures in shallow bays could increase the probability of hypoxia, which would in turn reduce available suitable habitat.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: **Lake Michigan Data**: NOAA GLERL Technical Memo 164 (1994-2010) and NOAA GLERL Technical Memo 175 (2015). **Lake Huron Data:** NOAA GLERL Technical Memo 140 (1972, 2000-2003); NOAA GLERL Technical Memo 172 (2006-2012), **Lake Ontario Data:** Burlakova et al. Accepted 2021. *Ecology* Data Paper (1964-2018). Compilation of Canadian and U.S. data. https://esajournals.onlinelibrary.wiley.com/doi/10.1002/ecy.3528 **MECP Canadian Nearshore Data:** https://data.ontario.ca/dataset/benthic-invertebrate-community-great-lakes-nearshore-areas		

Data Limitations

Because of the rapid rate at which *Dreissena* populations have expanded in many areas, and because of the ability of dreissenids to cause ecosystem-wide changes, documenting trends and reporting data needs to be completed in a timely manner. Besides density, biomass should be routinely monitored and determined using common protocols (e.g., Nalepa et al. 2014). This allows comparisons across lakes and other food web components, and is most useful for predictive models. Since dreissenids are found on hard as well as on soft substrates, various sampling methods may be needed to truly assess population mass in a given lake or lake region. New, video-based methods are being developed that are showing promise for increasing sampling coverage and also reducing the lag time between collection and reporting (Karatayev et al. 2021a). The data presented here are entirely from Ponar samples collected in areas with soft substrate. Hard substrate areas in the nearshore are typically under-sampled and there is the potential for zebra mussels, which prefer hard substrate, to be underrepresented in these surveys. Also, these assessments are focused on the main basins of the lakes, but some data for connecting channels is available from the MECP Nearshore Great Lakes Monitoring Network. Physical factors in connecting channels such as substrate variability, current patterns, etc. can result in high interannual variations in population that make it more difficult to interpret temporal trends. The available U.S. and Canadian data sources are comparable over time because both countries have performed whole-lake benthic surveys. In more recent years, the data sources are more complementary than comparable because they cover different depth zones and sampling locations. The benthic monitoring program conducted by MECP primarily covers nearshore areas (typically <3 km from shore) and DFO no longer conducts long-term offshore monitoring. However, the CSMI lake-wide surveys include data from both U.S. and Canadian waters, with site depths ranging from ~10 m to >200 m, depending on the lake.

Additional Information

Dreissenid mussels may be responsible for adverse impacts to several other sub-indicators. Dreissenid mussels have directly or indirectly impaired native species and therefore have negatively impacted biological integrity. Further they have impaired several beneficial uses listed under Annex 1 of the Great Lakes Water Quality Agreement (GLWQA) including fish and wildlife consumption, and fish and wildlife populations. Aquatic invasive species, including dreissenid mussels, have been given a high priority in the renewed GLWQA.

In 2014, the Invasive Mussel Collaborative (http://invasivemusselcollaborative.net/) was formed to advance scientifically-sound technologies to control invasive mussels. The IMC includes representatives from Federal, Provincial, State, tribal, industry, not-for-profit, and academic organizations. The Collaborative aims to improve communication and coordination among researchers and resource managers. To this end, the IMC administers an email list and maintains a comprehensive website with access to resources, such as research publications, a decontamination guide, and management plans. In 2019, the IMC coordinated a demonstration control project on Good Harbor Reef in northern Lake Michigan to test the efficacy of covering mussel beds with benthic mats and injecting Zequanox® underneath the mats. Mussel mortality was greater than 90% one month after treatment and post-treatment monitoring will reveal the speed and degree of recolonization. Wide-spread removals remain unfeasible, but targeted efforts to control dreissenid mussels in high-value habitats are being explored.

Monitoring programs collect data on density, biomass, size-frequency distributions, and length-weight relationships for dreissenid mussels. Methods for estimating abundances of *Dreissena* are generally similar across the Great Lakes. Samples of bottom substrates are collected with a Ponar grab and contents are washed through a screen (or net mesh) of 0.5-mm openings (0.6-mm for the MECP Nearshore surveys). While abundances are the most common reporting measure of population status, biomass is more valuable for assessing ecological impacts and for input to predictive models. Biomass has been measured using a few different methods. One approach is to only measure the mussel soft tissue with the thought that this best represents the biologically active portion of the organism. Some protocols call for separating soft tissue from shell and directly determining soft tissue weight, while others determine the size frequency of the populations (shell length) and infer tissue biomass based upon a predetermined relationship between shell length and soft tissue weight (e.g., Nalepa et al. 2014). Tissue weight can be expressed as dry weight or ash free dry weight (as in Figure 2). Another approach is to measure the wet weight of the whole mussel (shell and tissue), which is consistent with the US EPA Standard Operating Procedure for Benthic Invertebrate Laboratory Analysis (SOP LG407, Revision 09, April 2015; e.g., Nalepa et al. 2020, Karatayev et al. 2020). It is important to pay attention to the metric used for biomass so that comparable values are used when combining data sources. Values may require conversion using available regressions (e.g, Nalepa et al. 2020 for Lake Michigan and Karatayev et al. 2020).

Ideally, the dreissenid population should be assessed in offshore lake regions every ~3-5 years and more frequently (every ~1-2 years) in areas with higher temporal variability such as nearshore regions including shallow bays and basins, and connecting channels. This level of sampling requires a considerable amount of resources and support. Presently, the main data sources are the lake-wide CSMI surveys conducted in each of the lakes every 5 years, the annual offshore EPA GLNPO benthic survey, the NOAA/GLERL annual southern Lake Michigan benthic survey, and the MECP Nearshore Great Lakes Monitoring Network surveys. Other programs assess dreissenid populations in specific locales as related to different program goals. On the whole, the current monitoring programs are adequate to detect population changes, though it would be beneficial to add more sampling of regions with hard substrates.

Acknowledgments

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Data Source: MECP Nearshore Great Lakes Monitoring Network.

Figure 2. Mean densities (number per square metre) of *Dreissena* from sites at 31-90 m (upper panel) and >90 m (lower panel) in Lakes Michigan, Huron, and Ontario. Data are from lake-wide surveys conducted mostly at 5-year intervals. Lake Michigan = blue triangles, long-dashed line; Lake Huron = red squares, short-dashed line; Lake Ontario = black circles, solid line. Note the difference in scale between the two figures.

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Data Sources: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2014, 2020

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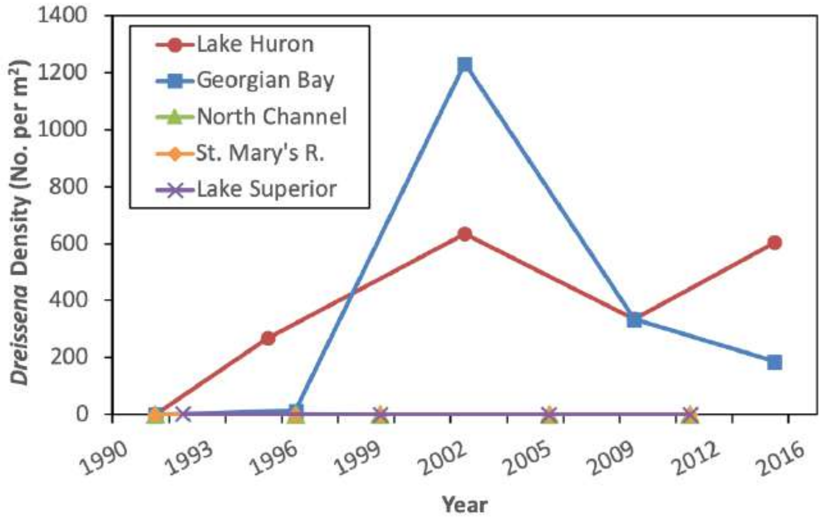
Last Updated
State of the Great Lakes 2022 Report



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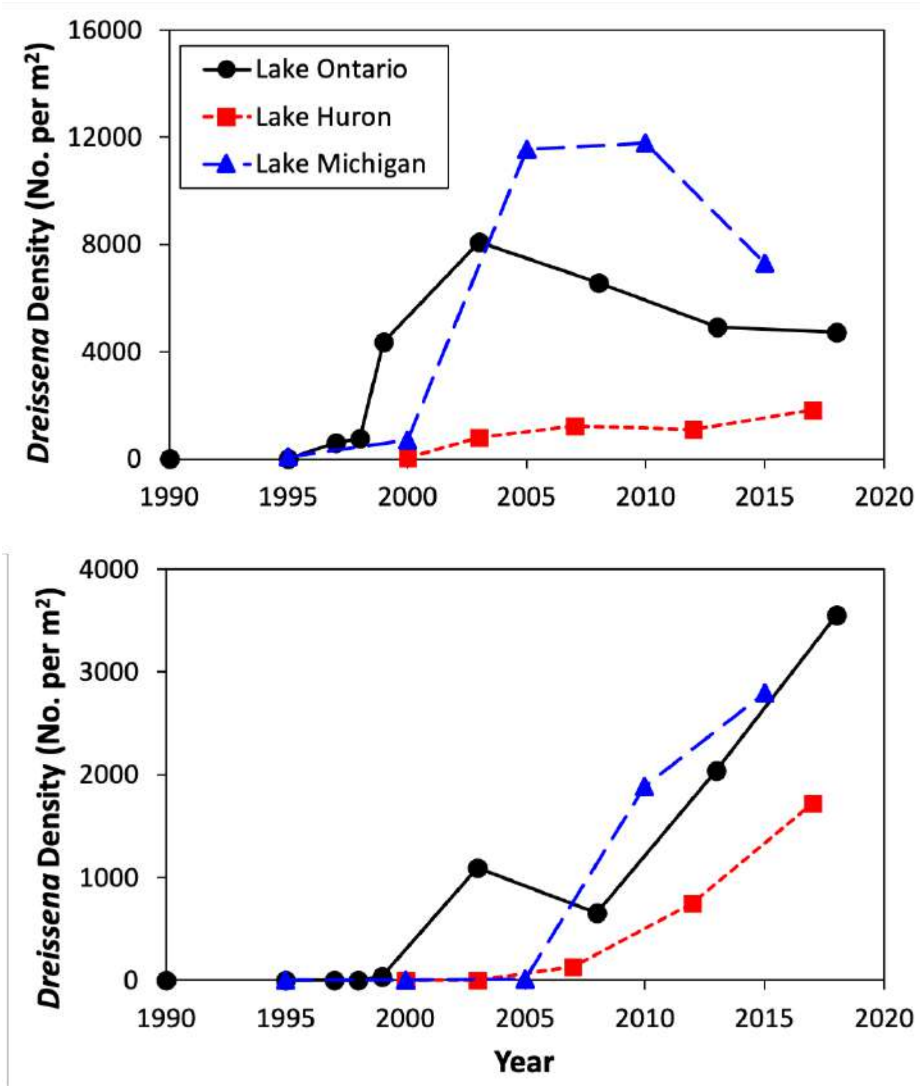

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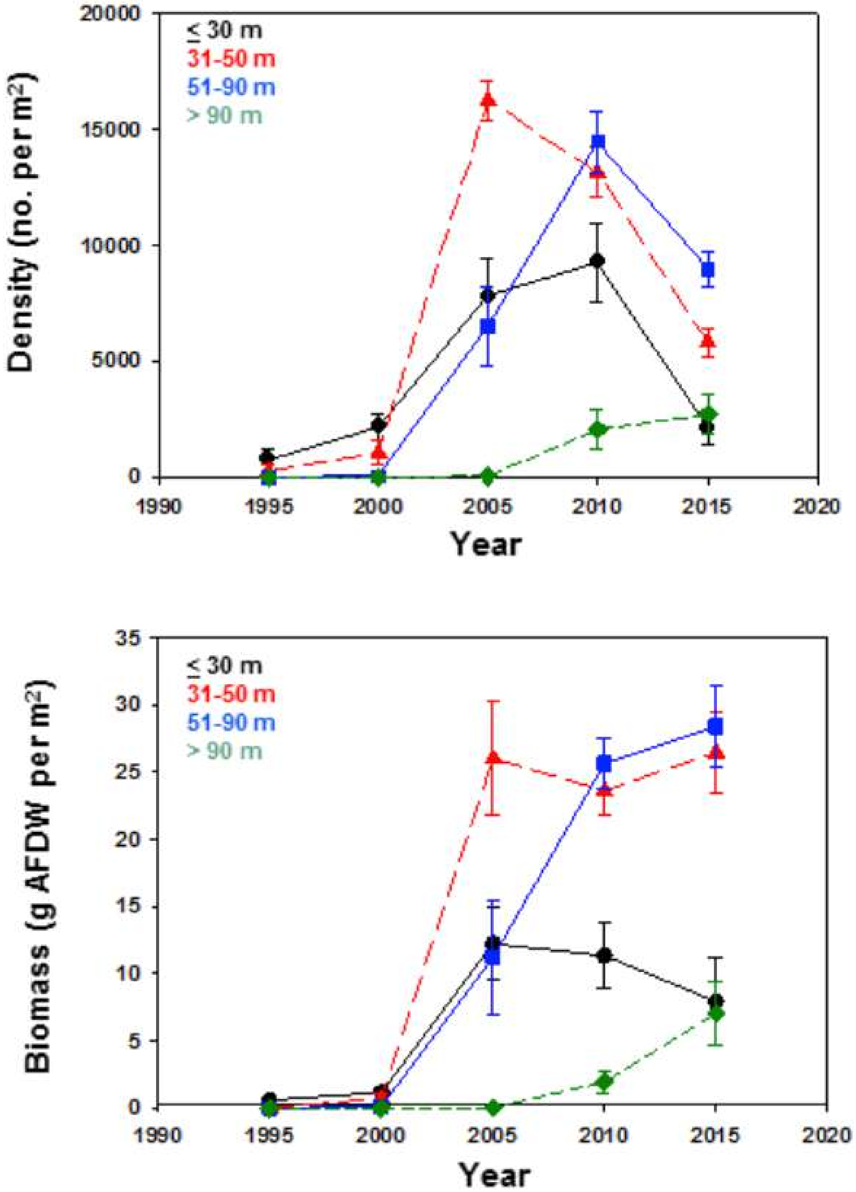

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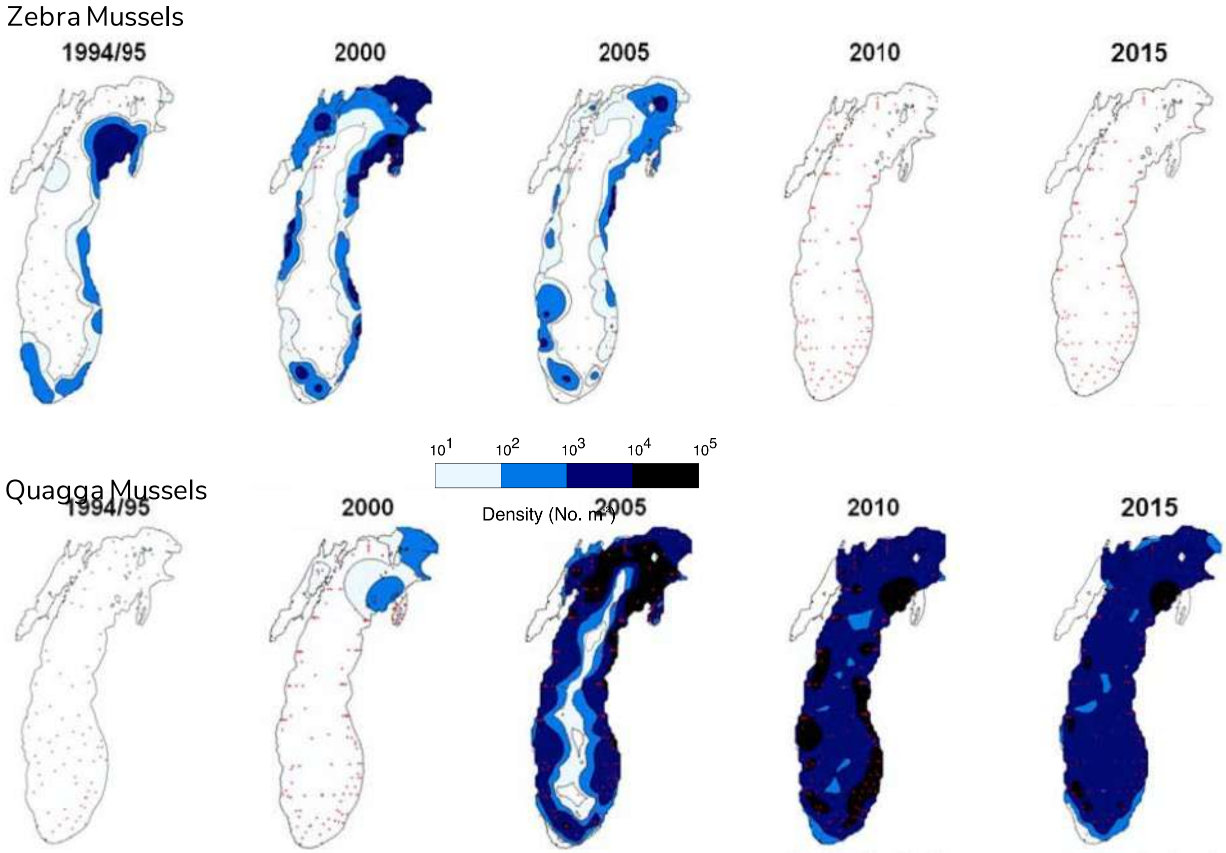

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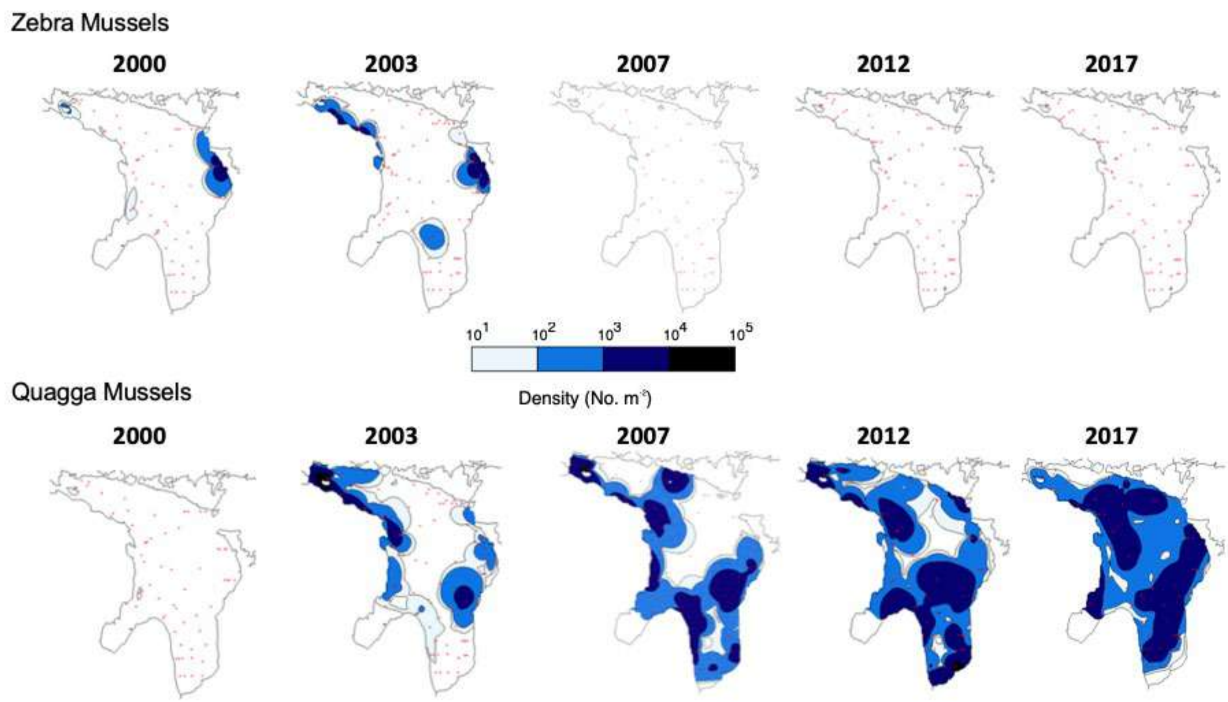

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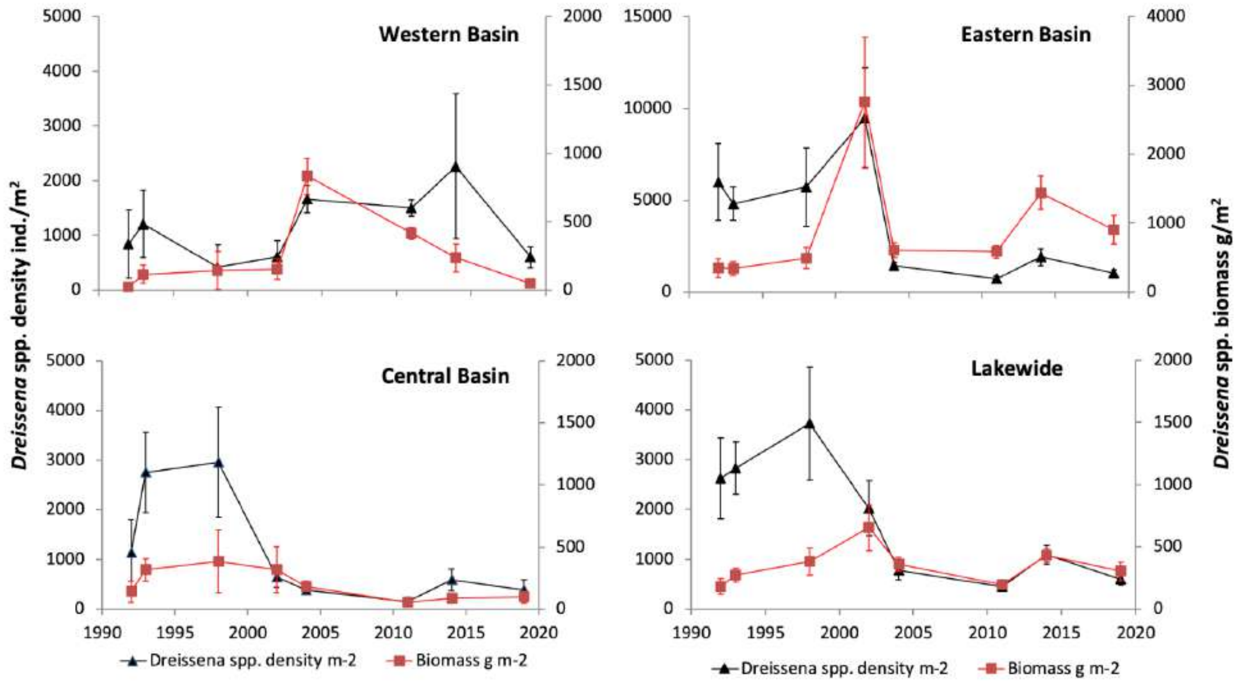

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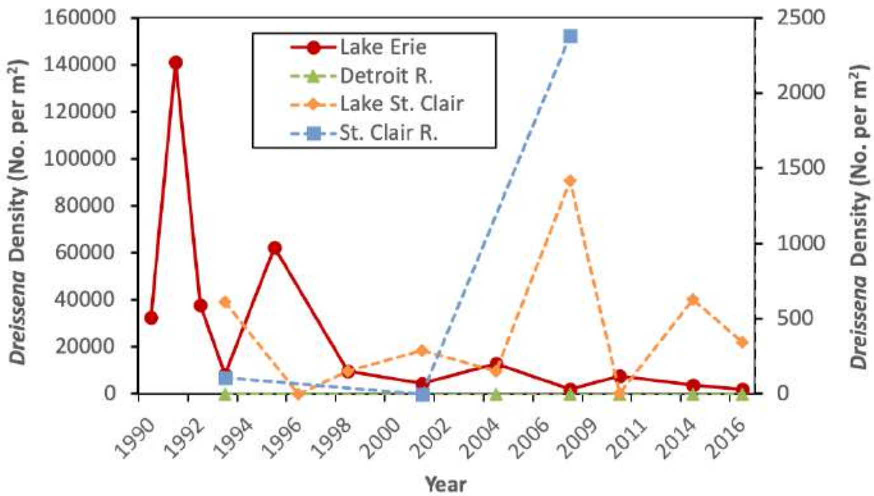

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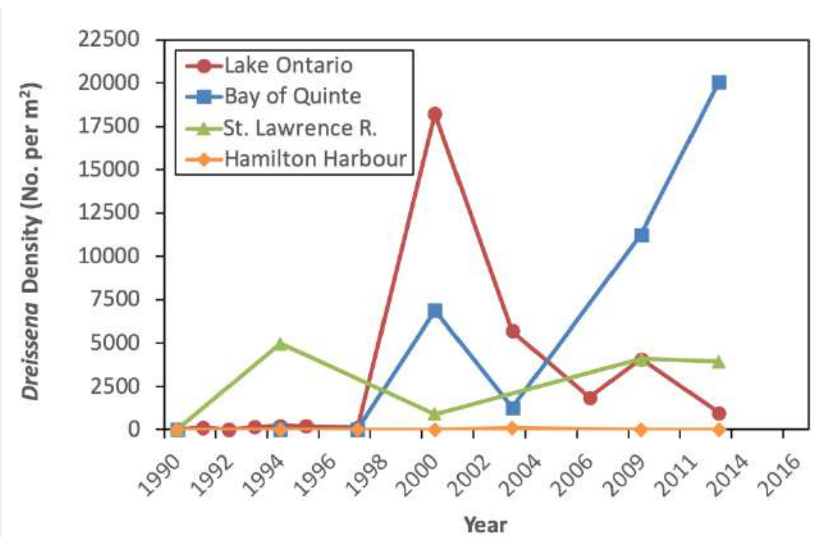

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Sub-Indicator: Terrestrial Invasive Species

Overall Assessment

Status: Undetermined

Trends:

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: This sub-indicator is being assessed as "Undetermined". Three new species have been included in this sub-indicator report for this cycle: mute swan, Japanese knotweed and common buckthorn. The new species are being added to this suite as they pose negative impacts on the health of the Great Lakes ecosystem. These species are replacing *Phragmites* and purple loosestrife which are already assessed in the Impacts of Aquatic Invasive Species and the Rate of New Aquatic Non-Indigenous Species Establishment in the Great Lakes sub-indicator reports. *Phragmites* and purple loosestrife have been removed from the Terrestrial Invasive Species sub-indicator due to concerns over double counting of the impacts of species within the Invasive Species indicator. The assessment methodology used in the past cycles' reporting was unable to be transferred to the assessment of the new species. Therefore, the species being added cannot be assessed in the same capacity as past reports and thus this cycles' report will not be comparable to past reports and lake-by-lake assessments are not made here. Efforts are underway to adapt the impact criteria and a scoring-based assessment used in the Impacts of Aquatic Invasive Species sub-indicator for use in future cycles.

Status Assessment Definitions

Good: TBD

Fair: TBD

Poor: TBD

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

Trend Assessment Definitions

Improving: TBD

Unchanging: TBD

Deteriorating: TBD

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend

Endpoints and/or Targets

TBD

Sub-Indicator Purpose

- To assess the presence, number, and distribution of six selected terrestrial invasive species (TIS) in the Laurentian Great Lakes watershed, and to understand the means by which these species persist within the ecosystem.
- Aid in the assessment of the status of biotic communities, as invasive species alter both the structure and function of ecosystems thereby compromising the biological integrity of these systems.
- This sub-indicator provides insight into the complex relationships between land and water that impact Great Lakes water quality.

Ecosystem Objective

To reduce and further prevent expansion of six selected terrestrial invasive species (Asian longhorned beetle, emerald ash borer, garlic mustard, common buckthorn, Japanese knotweed and mute swan) in the Great Lakes basin because of their negative impact on the biodiversity, habitat, chemical loads, nutrient cycling, and hydrogeology of terrestrial and aquatic ecosystems.

This sub-indicator best supports work towards General Objective #7 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from the introduction and spread of aquatic invasive species and free from the introduction and spread of terrestrial invasive species that adversely impact the quality of the Waters of the Great Lakes."

Measure

The status and trends for this sub-indicator are being assessed as "Undetermined" until new status and trend assessment definitions can be established. The distribution, impacts to water quality and potential control measures of six terrestrial invasive species — Asian longhorned beetle, emerald ash borer, mute swan, garlic mustard, common buckthorn and Japanese knotweed — are presented in this report. This report was supported by data from invasive species mapping databases, government reports, non-governmental agencies and peer-reviewed literature. In Canada, various groups and agencies provide supporting information on terrestrial invasive species distribution and impacts to water quality, such as Ontario's Invasive Species Awareness Program, Canadian Food Inspection Agency, and the Ontario Invasive Plant Council. Similar groups and agencies in the United Stated include the USDA Forest Service, USDA Animal and Plant Health Inspection Service (APHIS), US Fish and Wildlife Service, and state agencies. Additionally, citizen science systems such as EDDMapS and eBird are valuable sources of information for invasive species distribution. Other databases such as the Midwest Invasive Plant Network provides information on the regulation of invasive species within the Great Lakes basin US states.

Ecological Condition

The increased introduction of non-native terrestrial species in the Great Lakes basin has occurred as an unintended consequence of the global movement of people and goods. Some non-native species are considered invasive as they have the ability to significantly disturb the ecosystems in which they invade, However, not all non-native species are a threat to their introduced range. Since the 1800s, Canada has seen a dramatic increase in the number of invasive non-native plant species being introduced in the country (Canadian Food Inspection Agency, 2008; Figure 1).

The distribution, impact to water quality and potential control measure of six terrestrial invasive species (Asian longhorned beetle, emerald ash borer, mute swan, garlic mustard, Japanese knotweed, and common buckthorn) are presented in this report. When considering impacts to water quality by terrestrial invasive species, a key feature in the discussion are riparian areas. These areas act as buffers between the terrestrial and aquatic ecosystems and are often inundated with water, supporting moist soil, hydrophilic vegetation and recurring disturbance to the area (Ramey & Richardson, 2017). These six species were selected as they can be found in Great Lakes riparian areas and pose a potential threat to the health of the Great Lakes aquatic ecosystem. This means the species will tend toward a poor and deteriorating assessment, however, there are management actions currently in place and further actions that can be taken to mitigate and reduce their spread. It must be noted the available literature supporting impacts to water quality by terrestrial invasive species is lacking and these terrestrial invasive species may have further indirect or undocumented impacts which are not discussed here.

Asian Longhorned Beetle (ALB)

The Asian longhorned beetle (*Anoplophora glabripennis*; ALB; Figure 2a) is a wood boring beetle first discovered in the US Great Lakes basin in 1998 in Chicago, Illinois and the Canadian Great Lakes basin in 2003 in Vaughan, Ontario (Haack et al., 2010; Natural Resources Canada (NRCAN), 2020; U.S. Department of Agriculture Animal and Plant Health Inspection Service (USDA APHIS), 2020). The ALB is considered to be a forest pest which, in its larval form, tunnels into the sapwood and heartwood of hardwood trees (Haack et al., 2010; Meng et al., 2015). This tunneling impacts the structural integrity and vascular tissue, causing potential tree mortality and threatening the health of the forests in the Great Lakes basin (Haack et al., 2010). Healthy riparian forest ecosystems are critical in the protection of Great Lakes water quality. Linkages have been formed between forested landscapes and nutrient regimes, soil stabilization and the limiting of sediment-bound pollutants into receiving waters (Guiry et al., 2020; Turner & Rabalais, 2003). Further to this, not only do Great Lakes forests protect water quality, they provide food and habitat for a variety of native species (ex. beaver, moose, and migratory birds) which rely on a healthy Great Lakes ecosystem for their survival.

The ALB has no natural predators in North America, therefore they have posed a substantial threat to millions of trees across the country. In North America, extensive eradication measures have been introduced to ensure that the number and spread of ALB is limited, with treatment options including strict regulation of quarantine areas and tree removal (Haack et al., 2010). Trees are typically burned or fed through a wood chipper which kills larvae, pupae, or adult beetles in the bark (Ontario Ministry of Natural Resources and Forestry (OMNRF), 2020). In Ontario, the initial observations of ALB in Toronto and Vaughan were eradicated prior to two new populations being discovered in 2013 in Toronto and Mississauga. Susceptible and infested trees located in the two identified infestation sites were removed and the sites were monitored for 5 years. After recording no new infestations, ALB was considered eradicated in the Province and restrictions were appealed in 2020 (Canadian Food Inspection Agency, 2020). In the United States, the ALB population in the Great Lakes has been eradicated from Illinois since 2008 (USDA APHIS, 2020). However, while ALB populations have been eradicated in the Great Lakes basin, outside the Great Lakes basin quarantine and eradication efforts continue in nearby southern Ohio and New York to limit the spread of the ALB (USDA APHIS, 2020).

Emerald Ash Borer (EAB)

Emerald ash borer (*Agrilus planipennis*, EAB; Figure 2b), is a wood-boring beetle first discovered in North America in the Detroit-Windsor area in the early 2000s and is now found throughout the Great Lakes basin (Haack et al., 2015). These wood-boring pests are believed to have arrived in wood shipping containers and have been commonly observed feeding on various ash species across Ontario and all eight Great Lakes States (Haack et al., 2015). EAB infestations have resulted in the loss of millions of ash trees throughout the basin, with significant impacts observed to black ash wetland ecosystems in the western Great Lakes (Cianciolo et al., 2021). The loss of native ash trees, which are an important component of riparian habitats, can result in impacts to the food web, physical habitat and water quality of the Great Lakes ecosystem. As critical black ash riparian wetlands are lost to EAB infestation, higher water tables can be expected lead to increased soil erosion and flooding. Soil erosion into streams is particularly concerning as pollutants are bound to sediment, altering the stream's chemical properties and increasing nutrient loading (Hewlett et al., 2015). Riparian areas also play a critical role in regulating water temperature through canopy cover shading, thus increased ash tree mortality and the subsequent reduction in canopy cover from EAB infestations can pose implications to Great Lakes surface water temperatures (Nisbet et al., 2015). Another complex interaction within riparian areas is the transfer of energy between the aquatic and terrestrial ecosystem to support the aquatic food web. Riparian areas supply critical nutrients, such as nitrogen, to the aquatic ecosystem through leaf litter and runoff from the riparian soils. In addition, tree species within the ash genus (*Fraxinus*) are a preferred species for consumption by aquatic invertebrates (Kreutzweiser et al., 2018; Nisbet et al., 2015). Ash trees are known for producing nitrogen-rich leaf litter which have shown to be a major contributor of energy in aquatic ecosystems (Nisbet et al., 2015). A decline in nutrient and organic matter input due to ash mortality from EAB infestations can impact the aquatic microbial and invertebrate communities within the Great Lakes and its surrounding tributaries, and potentially have cascading effects on the remaining aquatic food web (Kreutzweiser et al., 2018; Nisbet et al., 2015). (Cianciolo et al., 2021; Kolka et al., 2018). Loss of ash trees also directly impacts traditional ways of life for Indigenous communities in the Great Lakes basin. One example is the Mohawk Community of Akwesasne, who rely on black and white ash for basket making. Traditional basket making is a critical component of the Akwesasne Mohawk way of life, connecting generations and acting as a medicine to promote healthy minds and peace (Saint Regis Mohawk Tribe, n.d.). As a response to this threat, the Saint Regis Mohawk Tribe established an EAB response plan that presents tools and strategies to respond and adapt to the presence of EAB, provides goals and objectives for the continuation of traditional Mohawk basketry, and summarizes options for preserving ash tree resources (Saint Regis Mohawk Tribe, n.d.)..

EAB has caused significant economic impact across Canada, with Canadian Forest Service scientists estimating that $2 billion CAD over a 30-year period will be required to remove and replace EAB infested stands in Canadian municipalities (Pegler, 2018). In the City of Toronto, on the northwestern shore of Lake Ontario, a cost of $71.2 million CAD was estimated to be required from 2013 to 2020 for EAB management (City of Toronto Parks, Forest and Recreation Division, 2012). Along with the high economic costs of EAB infestations, high mortality rates are also typical such that 8-10 years after the initial infestation, roughly 99% of ash trees in a woodlot are killed (NRCAN, 2021). In 2018, it was estimated that EAB infestations caused the decline and mortality of ash in approximately 4,580 hectares across southern Ontario (OMNRF, 2019). While EAB beetles are prominent in southern Ontario and are predicted to spread throughout the province, pheromone traps deployed in northwest Ontario districts detected no EAB beetles (OMNRF, 2020). However, EAB do still pose a potential threat to forests within the northern portion of the Great Lakes basin. For example, in October 2015, EAB populations were reported in Duluth, Minnesota (City of Duluth, Minnesota, 2016). The USDA Forest Service and Michigan State University maintain the binational Emerald Ash Borer Information Network website, which includes monthly updates on the confirmed locations of this species in the U.S. and Canada. The Emerald Ash Borer Information Network (http://www.emeraldashborer.info/about-eab.php) is a multinational effort to provide the latest information on EAB infestations within the US and Canada. The network provides information on EAB, tracks detections of EAB (Figure 3), and suggests preventative measures to reduce EAB spread (Emerald Ash Borer Information Network, n.d.).

Treatment, removal, and replacement of EAB infested stands is costly and intensive. Quarantine areas, strict regulation, education programs and removal of ash trees in infested areas are some important measures to limit the spread of EAB in the Great Lakes basin and beyond. Recent actions on the US side of the Great Lakes basin have seen the removal of EAB quarantine regulations in response to the challenges and their general ineffectiveness on stopping the spread of EAB (USDA APHIS, 2021). USDA is working to develop more effective non-regulatory management and control approaches such as the use of biological control agents (USDA APHIS, 2021). In Ontario, quarantine regulations are still in place and managed by the Canadian Food Inspection Agency (CFIA); EAB regulated areas can be found all along the lower Great Lakes with one regulated area present along Lake Superior in the City of Thunder Bay, Ontario (Canadian Food Inspection Agency (CFIA), 2019; Figure 4).

Mute Swan

The mute swan (*Cygnus olor;* Figure 2c) is an invasive species that was first introduced to North America for ornamental purposes in the late 1800s to early 1900s (Petrie & Francis, 2003). The initial appearance of this species within the Great Lakes basin was recorded in Charlevoix, Michigan in 1919 (Gehring et al., 2020). Since their introduction into the Great Lakes, mute swan populations have grown within the Ontario lower Great Lakes to a high of 4100 during the 2017 Mid-Summer Mute Swan Survey (Environment and Climate Change Canada (ECCC), 2019). Mute swans pose a multitude of ecological and social impacts within the Great Lakes basin. One of the more prominent impacts to the aquatic ecosystem is the degradation of submerged aquatic vegetation (SAV). Within wetland ecosystems, SAV is critical to the functioning of the biotic communities and abiotic systems, acting as critical refuge for aquatic invertebrates and fish while also providing forage material for aquatic mammals, fish, and waterfowl (Stafford et al., 2012). Beyond the biotic community, SAV also helps to reduce sediment transportation and turbidity by anchoring substrate and reducing the resuspension of sediment within the water column (Tang et al., 2020). Healthy wetland ecosystems with SAV can help reduce algal blooms, as well as alter nutrient and oxygen cycling within the waterbody (Guillaume et al., 2014; Tang et al., 2020). SAV is the main food source for the mute swan which can consume ~3.8 kg of SAV daily, with foraging practices further uprooting SAV (Stafford et al., 2012). With populations of invasive mute swan continuing to thrive within the lower Great Lakes, the success of SAV communities is threatened.

Some of the impacts from mute swans are limited to the terrestrial/anthropogenic environment, such as the increased potential for the spread of infectious diseases such as avian influenza virus and avian paramyxovirus serotype 1 to humans and agricultural species from mute swan hosts (Pedersen et al., 2013). Also, breeding mute swan pairs have been documented as territorial and aggressive to other native waterfowl, such as trumpeter swans and common loons, in an effort to compete for and protect their breeding grounds (Guillaume et al., 2014). There has also been evidence which suggests that the presence of invasive mute swans within the Great Lakes ecosystem threatens the genetic integrity of native swans such as the trumpeter swan through cross breeding (ECCC, 2019).

Using the citizen science platform eBird, which documents observations of various avian species across the globe, mute swan observations have been logged along the shores of every Great Lake. The binational mute swan population within the lower Great Lakes saw a 10-18% increase from 1980 to 2000, with a projected increase to 30,000 individuals by 2030 (Petrie & Francis, 2003). Within the U.S. portion of the Great Lakes basin, Mute swan management and control programs vary, with control in some states being managed by the USDA APHIS. Under Michigan's 2012 Mute Swan Management and Control Program Policy and Procedures, permits can be issued for the removal of mute swan eggs and nests along with the capture and humane euthanasia of mute swan individuals by permit holders or Michigan Department of Natural Resource staff (Michigan Department of Natural Resources - Wildlife Division, 2012). The New York Department of Environmental Conservation saw a decline in the upstate New York mute swan population along the southern shore of Lake Ontario to 39 individuals in 2014 due largely to the management practices put in place (egg-addling and removal of mature individuals) (New York State Department of Environmental Conservation, 2019). However, without any control measures in place, mute swan observation counts increased to 327 in the 2017 Midsummer Mute Swan Survey (New York State Department of

Environmental Conservation, 2019). On the Canadian side of the Great Lakes basin, mute swans are protected under the federal Migratory Birds Convention Act, however permits can be obtained for the removal of mute swan individuals or their eggs from Environment and Climate Change Canada's Canadian Wildlife Service (Environment and Climate Change Canada, 2019). To protect the productivity of coastal wetlands and its biotic communities, continued public education, management of invasive species such as mute swan and control measures are critical within the Great Lakes ecosystem.

Garlic Mustard

Garlic mustard (*Alliaria petiolata*; Figure 2d) is an aggressive herbaceous plant first discovered in the Great Lakes basin in 1868 in New York State and 1879 in the Province of Ontario (Anderson, 2012a; Meekins & McCarthy, 2001). Believed to be introduced for culinary or medicinal purposes, it can be found in a variety of ecosystems such as riverbanks, floodplains and hardwood forests (Anderson, 2012a; Meekins & McCarthy, 2001). Garlic mustard has a number of invasive traits allowing it to dominate an ecosystem, such as the general avoidance of garlic mustard by native herbivores, early phenology, and the release of allelopathic chemicals which suppress native plant growth by impacting beneficial mycorrhizal fungal communities in the soil (Haines et al., 2018). While the impacts of garlic mustard on native species varies, evidence has shown that the allelopathic effects of garlic mustard can impact mycorrhizal dependent species such as trout lily, starflower, Canada mayflower and seedlings of red maple and white ash (Haines et al., 2018). Further, two native species – the wood poppy and wood aster – have been designated as endangered and threatened, respectively, by the Committee on the Status of Endangered Wildlife in Canada in part due to the spread of garlic mustard. Impacts to the composition of forest ecosystems, specifically the impact to tree seedlings in invaded riparian forest ecosystems, may have implications for future riparian ecosystems. As garlic mustard suppresses the growth of new tree seedlings in riparian areas, the current canopy does not get replaced by native species. Therefore, aquatic ecosystems may become threatened by altered nutrient cycles due to limited leaf litter input and increased surface water temperatures as there is generally less sunlight obstruction from a lack of canopy. Also, garlic mustard bodes leaves with a high nutrient content causing the acceleration of native leaf litter decomposition in the terrestrial environment (Anderson, 2012a). A change in decomposition and composition of the forest floor can impact the habitat of ground-nesting birds, salamanders and other animals found on the forest floor (Anderson, 2012a).

Tracking the distribution of garlic mustard is an important step in eradication efforts as it highlights areas that require management. EDDMapS is one important platform that collects volunteered geographic information about garlic mustard observations in Canada and the United States. Data derived from EDDMapS indicates garlic mustard has been observed across the Province of Ontario and in all eight US Great Lakes States (EDDMapS, 2021; Figure 5). Over time it is predicted the range of garlic mustard will continue to expand across North America as it possesses a specific combination of traits, making it a successful competitor in a variety of ecosystems (Rodgers et al., 2008). Because garlic mustard can grow in numerous diverse ecosystems, unique management options are required for each site (The Nature Conservancy of Canada, 2007). Some potential control measures include prescribed burning, cutting, and application of glyphosate-based herbicides (Nuzzo, 1991). As garlic mustard is a biennial plant, control efforts may be more successful in certain stages of the species' biological cycle. Studies have shown management practices are more effective on garlic mustard when they occur in the second-year adult stage rather than the species' first-year rosette stage (Anderson, 2012a; Nuzzo, 1991; Pardini et al., 2009). Any control measure will need to be repeated each year until the existing seed bank is depleted (Anderson, 2012a). Biological control is another potential method of controlling garlic mustard populations. *Ceutorhynchus scrobicollis* Neresheimer & Wagner, a root-crown mining weevil, was approved for releases in Canada in 2018. This species has been shown to prey upon garlic mustard, resulting in a reduction in species growth, survival and seed production (Gerber et al., 2009).

Japanese Knotweed

Japanese knotweed (*Reynoutria japonica*; Figure 2e) was initially introduced in North America as a horticultural plant in the latter half of the 19th century (Anderson, 2012c). This highly invasive species can be found growing in a variety of ecosystems ranging from riparian and wetland areas to roadsides and fence lines (Anderson, 2012c). Its aggressive and competitive practices and extensive rhizome structures are part of what make Japanese knotweed a top invasive species (Anderson, 2012c). The spread of Japanese knotweed into riparian and wetland areas threatens the role of these ecosystems in protecting the water quality of the Great Lakes basin.

While there are gaps in the body of knowledge regarding the direct impacts that Japanese knotweed have to the Great Lakes water quality, this species has been found to both adversely impact and benefit biota within riparian areas (Lavoie, 2017). Literature suggests that the presence of Japanese knotweed has had beneficial effects on species of fungi and aquatic leaf shredders, however, is has also been documented to impact the success of certain species of birds, amphibians, soil bacteria, arthropods, and gastropods (Lavoie, 2017). Terrestrial invertebrates are a group of species adversely impacted by the spread of Japanese knotweed in riparian areas to varying degrees. Studied riparian areas invaded with Japanese knotweed have shown a lowered terrestrial invertebrate species abundance and diversity (Seeney et al., 2019). Terrestrial invertebrates in riparian ecosystems can be critical food sources for fish species in the neighboring waterbodies as well as facilitators of energy transfer between the terrestrial and aquatic food webs (Baxter et al., 2005; Seeney et al., 2019). Therefore, the reduced abundance and diversity of terrestrial invertebrates in riparian areas — due to Japanese knotweed invasion — potentially threatens the role species within this group play in the Great Lakes aquatic food web.

Japanese knotweed often reproduces via rhizomes in the soil or stem fragments (Anderson, 2012c; Weston et al., 2005). These rhizomes can become established in a new area by human transportation or transportation via water (Colleran et al., 2020; Weston et al., 2005). After initial establishment, the rhizome structure will expand to form dense stands of Japanese knotweed resembling monocultures (Colleran et al., 2020; Weston et al., 2005). These monocultures outcompete native vegetation, displacing critical bank stabilizing riparian vegetation (Colleran et al., 2020). The replacement of stabilizing native root structures with the weaker rhizome structures of Japanese knotweed within riparian areas, poses potential threats to the quality and health of the Great Lakes ecosystem from increased sediment loading via erosion carrying with it excess nutrients and potentially harmful chemicals (Colleran et al., 2020).

Climactic models produced in 2008 showed that 53% of Southern Ontario, including parts of the Lake Ontario, Lake Erie and Lake Huron basin, is climatically suitable for Japanese knotweed growth (Bourchier & Hezewijk, 2010). Data pulled from the Early Detection and Distribution Mapping System (EDDMapS) shows that Japanese knotweed has been observed in each individual basin within the Great Lakes, with limited spread in the northern Great Lakes likely due to adverse climactic conditions and fewer avenues for anthropogenic transportation (EDDMapS, 2021; Figure 6). In the Canadian Great Lakes basin, the majority of Japanese knotweed invasions have occurred in the past 20 to 30 years with the basins' northernmost populations of Thunder Bay and Sault Ste. Marie being confirmed in 2012 (Anderson, 2012c). Mapping the geographic distribution of Japanese knotweed invasions using herbarium records from 1900 to 2000 show a rapid increase in the species across North America. High concentrations were found in the US Northeast and Southeastern Ontario which are home to sections of the Great Lakes basin (Barney, 2006). By the year 2000, approximately 71% of counties in Northeast US states had recorded Japanese knotweed invasions, and across North America 577 US counties and 93 Canadian municipalities, had recorded invasions (Barney, 2006). The best approach to controlling and managing Japanese knotweed is to implement prevention and control actions before the species becomes established in an area. Japanese knotweed is a restricted invasive species under the Invasive Species Act, which prohibits the import, breeding, purchase or trade of Japanese knotweed. While control measures are not perfect and often do not lead to complete eradication of stands, success has been shown using glyphosate-based herbicides under various conditions including biannual foliar applications (Bashtanova et al., 2009; Jones et al., 2018). Current biological control measures within the Great Lakes basin and beyond involve the release of the knotweed psyllid *Aphalara itadori* (Grevstad et al., 2020). This biological control agent was first approved for use in Canada in 2014 with current research going towards maintaining stable and sustaining populations across the country (Grevstad et al., 2020). In the US, *Aphalara itadori* was approved for release in 2020 with the first population being released in the same year (Grevstad et al., 2020).

Common Buckthorn

Common buckthorn (*Rhamnus cathartica*, Figure 2f) is believed to have been initially introduced to North America in the 19th century for medicinal purposes, and populations have been observed in all the Great Lakes States and Ontario (Kurylo & Endress, 2012). In riparian areas, this invasive species has the potential to delay organic matter decomposition and impact populations of amphipods within water bodies. Amphipods or leaf shredders play a crucial role in the aquatic ecosystem and act as a food source for upper food web organisms. The energy provided by riparian leaf litter is converted into energy forms more readily available to other organisms within the aquatic food web by amphipods (Lewis, Freund, et al., 2017; Lewis, Piatt, et al., 2017). Studies have shown common buckthorn leaf litter decomposes at a higher rate than corresponding native species such as ash (Freund et al., 2013). This, in combination with common buckthorns delayed leaf fall, changes the availability of energy in the water body in areas where the riparian canopy is dominated by common buckthorn (Freund et al., 2013). While studied amphipod species appear to prefer common buckthorn leaf litter over that of native species, consumption of leaf litter deposited by this invasive species showed higher rates of mortality and lowered body mass within studied amphipod populations (Lewis, Freund, et al., 2017; Lewis, Piatt, et al., 2017).

Common buckthorn can be observed prolifically spreading across the Midwest US and southern Canada. In a Wisconsin-based study, common buckthorn accounted for 45% of the basal area and 81% of the relative density of the eight studied forest stands (Mascaro & Schnitzer, 2007). In 2010, the USDA Forest Service identified that the most abundant populations of common buckthorn in the Canadian Great Lakes basin were located in southern Ontario, and the most abundant populations on the U.S. side of the Great Lakes basin were found in southern Michigan and Wisconsin, and northern Illinois (Zouhar, 2011). As common buckthorn is such a prolific invasive species across North America, control measures are critical for reducing the spread and impact this species can have. Various tactics can be used, including pulling, cutting or girdling, mowing, and herbicide application with varying degrees of success (Anderson, 2012b; Pergams & Norton, 2006). A combination of methods involving the cutting or girdling of a single stem and the application of a herbicide such as Roundup has been shown to be successful (Pergams & Norton, 2006). However, to protect terrestrial and aquatic ecosystems from the impacts of terrestrial invasive species, management and control measures on established stands need to be coupled with measures to prevent new introductions, reduce the spread and future establishment of species. While cutting or girdling and herbicide application has been successful in removing currently established shrubs, some studies have shown the planting of native vegetation can assist in reducing the abundance of common buckthorn regrowth and reducing the likelihood of common buckthorn re-establishment (Wragg et al., 2021).

Linkages

Watershed Impacts and Climate Trends:

- Climate Trends – Impacts of terrestrial invasive species are limited by the range in which species have spread. Currently, distribution data from EDDMapS and eBird show limited spread of the selected invasive species in the northern Great Lakes. However, as the climate continues to change, the invasion range of these species could increase further into northern areas (Clements & DiTommaso, 2012). Warmer temperatures can also lead to extended lifespans of some species. For example, EAB has a lower survival rate in extreme cold temperatures, however as temperatures increase there is a potential for increased survival. With an extended lifespan, this invasive species can prey on a greater number of ash trees as it does not have to face extreme cold temperatures (Cuddington et al., 2018).

- Surface Water Temperature – Riparian canopy cover helps decrease the amount of sunlight directly reaching the surface waters (Moore, 2005; Nisbet et al., 2015). This coupled with a multitude of other factors helps moderate the water temperature of the Great Lakes and surrounding tributaries (Moore, 2005; Nisbet et al., 2015). However, EAB invasions are causing a decline in riparian canopy cover across the basin, and impacting the ability of the canopies to help maintain adequate surface water temperatures.
- Human Population – The growth of the human population in the Great Lakes basin means an increased demand for the development of adequate housing and transportation channels to supporting the population. Further development within the Great Lakes basin could support increased anthropogenic translocation of terrestrial invasive species across the basin.
- Forest Cover – ALB and EAB (to a greater extent) feed on the wood of trees across the Great Lakes basin, impacting their survival and decreasing the overall forest cover across the basin (Haack et al., 2010, 2015). However, it should be noted that the eradication of ALB distribution in the Great Lakes puts the continued impact to forest cover largely on EAB.

Habitat & Species – Coastal Wetland sub-indicators:

- The six terrestrial invasive species reported here have varying implications to the coastal wetland ecosystem. These include, but are not limited to, increased mortality of aquatic invertebrates from common buckthorn leaf litter, competition with coastal wetland birds by mute swans, degradation of coastal wetland SAV and the services they provides for fish and invertebrates by mute swan and the loss of critical black ash wetlands by EAB.

Nutrients & Algae:

- Healthy and functioning riparian areas play a key role in filtering out excess nutrients and pollutants from runoff before they enter the water body. The invasion of mono-culture forming and rhizome spreading Japanese knotweed reduces the functionality of riparian areas making them susceptible to increased erosion and nutrient loading into the waterbody (Colleran et al., 2020).

Traditional Ecological Knowledge (TEK), Citizen Science and Other Bodies of Knowledge

This report is supported by information collected from Early Detection and Distribution Mapping System (EDDDMaps) and the eBird database. EDDMapS is a North American wide database which provides verified, long-term and large scale distribution datasets for invasive species collected from volunteer observations and other monitoring or citizen science databases. As of June 2021, EDDMapS has over 5.6 million county-level reports. The eBird database is an international database which documents bird distribution, abundance, habitat and trends through citizen science observations. This database is used to document the distribution of mute swan. Citizens are encouraged to continue reporting their invasive species sightings in an effort to support the monitoring and reporting of terrestrial invasive species around the Great Lakes basin.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization				X*
Data are from a known, reliable and respected generator of data and are traceable to original sources				X*
Geographic coverage and scale of data are appropriate to the Great Lakes Basin				X*
Data obtained from sources within the U.S. are comparable to those from Canada				X*
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report				X*
Data used in assessment are openly available and accessible	No			

*As this sub-indicator for this SOGL reporting cycle is considered as "Undetermined", and no data were analyzed to conduct a formal assessment, these data fields in this table are all considered "Not Applicable".

Data Limitations

It is difficult to assess the impacts of terrestrial invasive species on Great Lakes water quality due to the specificity of the topic. Current research is limited in its discussion on direct impacts of terrestrial invasive species to the Great Lakes basin and the individual lake basins. Consequently, much of the impacts presented in this report are assumed or potential impacts supported with research which may have focused on geographical areas beyond the basin or the Great Lakes proper. Further to this, these impacts may only represent impacts to areas or regions within the basin and not the lake basin as a whole. Due to a lack of quality observation data within the set geographical bounds of the Great Lakes, determining distribution of the species chosen is difficult. While databases such as EDDMapS and eBird can and are being used, they come with their own challenges. There is a possibility of a disparity in observations across the basin as monitoring efforts are not uniform. Also, the locations of the observations are contingent upon the users who submit the data and the amount of resources expended in an area (a greater amount of resources will generally result in a greater amount of observations). The citizen science databases only reflect observations that were made and do not reflect treatment options that have been applied; for instance, a stand of Japanese knotweed may have been eradicated after the observation was submitted to EDDMapS. Also of note, additional mute swan data are available through the Great Lakes Marsh Monitoring Program, as well as Great Lakes Coastal Wetland Monitoring Program, however there was not enough capacity to analyze these datasets for this reporting cycle.

Additional Information

Globalization (i.e. the movement of people and goods) has led to a dramatic increase in the number of terrestrial non-native species that are transported between countries and across oceans. It is difficult to fully appreciate the impacts and distribution of terrestrial invasive species in the Great Lakes basin due to the extent of the region, the number of terrestrial invasive species and the differences in monitoring efforts across space and time. The management of invasive species is essential as they are one of the greatest threats to biodiversity in the Great Lakes region. Consequently, a greater amount of research is required to not only understand where terrestrial invasive species are located, but to also understand what impact terrestrial invasive species are having on different habitats and water quality.

It is understood that the number of species which may pose impacts to the Great Lakes ecosystem are vast and cannot all be represented here at this time. The six species presented in this report were chosen for inclusion as they are considered to have impacts to water quality, have accompanying distribution data and have been highlighted in past Great Lakes reporting such as Lakewide Action and Management Plans. Prior to the formal selection of these species for inclusion, a number of additional invasive species were considered. These species were not included due to a lack of distribution data and/or no known impacts to water quality, however, they are listed below (in no particular order) to ensure they are considered for inclusion in future reporting cycles.

Other TIS of Concern

- Dutch elm disease
- Invasive Earthworms
- Wild boar/Feral pig
- Spongy moth
- Swallow wort/Dog-strangling vine
- Invasive Honeysuckle
- Japanese silverberry
- Common baby's-breath
- Bird's-foot trefoil
- Spear thistle
- Creeping thistle
- Perennial sowthistle
- Blue sedge
- Hoary willowherb
- Oriental lady's thumb
- Common wormwood
- Japanese stiltgrass
- Hemlock Woolly Adelgid
- Spotted lanternfly

Future Terrestrial Invasive Species Assessment

The TIS sub-indicator is being assessed as Undetermined for this cycle. This is due in part to the removal of two invasive species previously included in this sub-indicator report, *Phragmites* and purple loosestrife, and the inclusion of three new species: mute swan, Japanese knotweed and common buckthorn. As well, there is currently not a formal process in place that sufficiently determines the status and trends for this sub-indicator. Efforts are currently underway to identify criteria which assesses the impacts and distribution of the selected invasive species to determine their status and trend. Options to improve the assessment procedure for this reporting are currently being explored, and will likely be implemented into the next reporting cycle.

One current possible method that is being explored to assess the status for this sub-indicator is the implementation of an assessment approach similar to the one that is overseen by the National Oceanic and Atmospheric Administration – Great Lakes Environmental Research Laboratory (NOAA-GLERL) to assess the impacts of Great Lakes aquatic invasive species. This approach is utilized in the Impacts of Aquatic Invasive Species Sub-indicator report, and assesses each species against criteria to determine the level of impact (either High, Moderate or Low impact). Draft modified criteria have been created which assess the impacts of terrestrial invasive species on water quality in the Great Lakes. These criteria score the impacts of select terrestrial invasive species on Great Lakes water quality, then determine an overall assessment for the sub-indicator using the individual species' scores. An appropriate approach to assess the trends for this sub-indicator are also currently being explored, and may be implemented next cycle. This approach may utilize citizen science supported datasets such as EDDMapS, which can be used to assess the temporal distribution of invasive species throughout the Great Lakes basin.

Other information which could be considered to support the assessment procedure for this sub-indicator in future reporting cycles include:

- TIS Hotspots – plot location of first occurrence for each TIS spatially using a publically available database such as EDDMapS and determine hotspots that need focused attention.
- Establish a base period for distribution data. This would involve identifying a period in time where a specific policy that aimed to curb the spread of terrestrial invasive species was implemented. The date of implementation could be used as the base period for this assessment.
- Examine well-studied areas that are revisited periodically (e.g. forest monitoring plots at a small scale or parks with monitoring programs). Study the change in the number of TIS within these smaller, but intensively sampled plots or regularly monitored areas. This approach helps reduce the data limitations related to time lags and effort.

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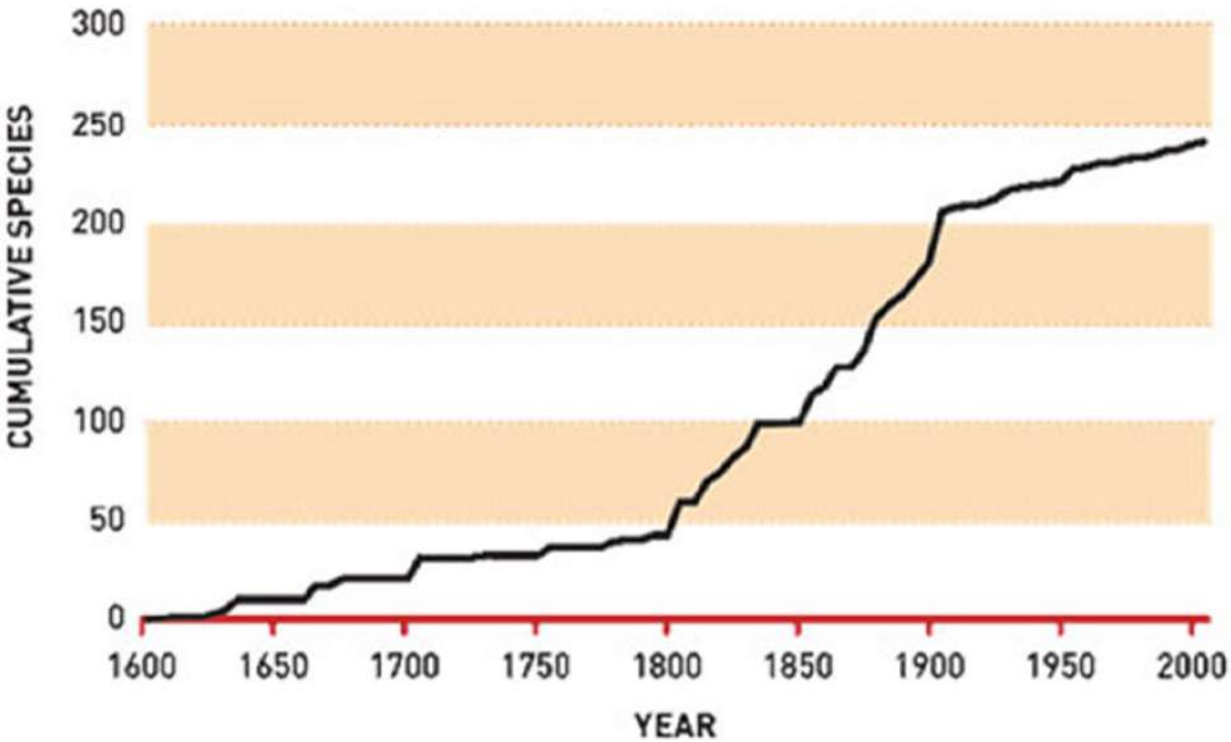
Figure 6. Range of Japanese knotweed in the US Great Lake States and the Province of Ontario by County.

Last Updated

State of the Great Lakes 2022 Report



Note: From "Invasive Alien Plants in Canada – Technical Report," by Canadian Food Inspection Agency, 2008, (https://epe.lac-bac.gc.ca/100/206/301/cfia-acia/2011-09-21/www.inspection.gc.ca/english/plaveg/invenv/techrpt/techrese.shtml)

Figure 1. Cumulative number of invasive non-native plant species introduced into Canada from 1600 to 2005 – Estimated.

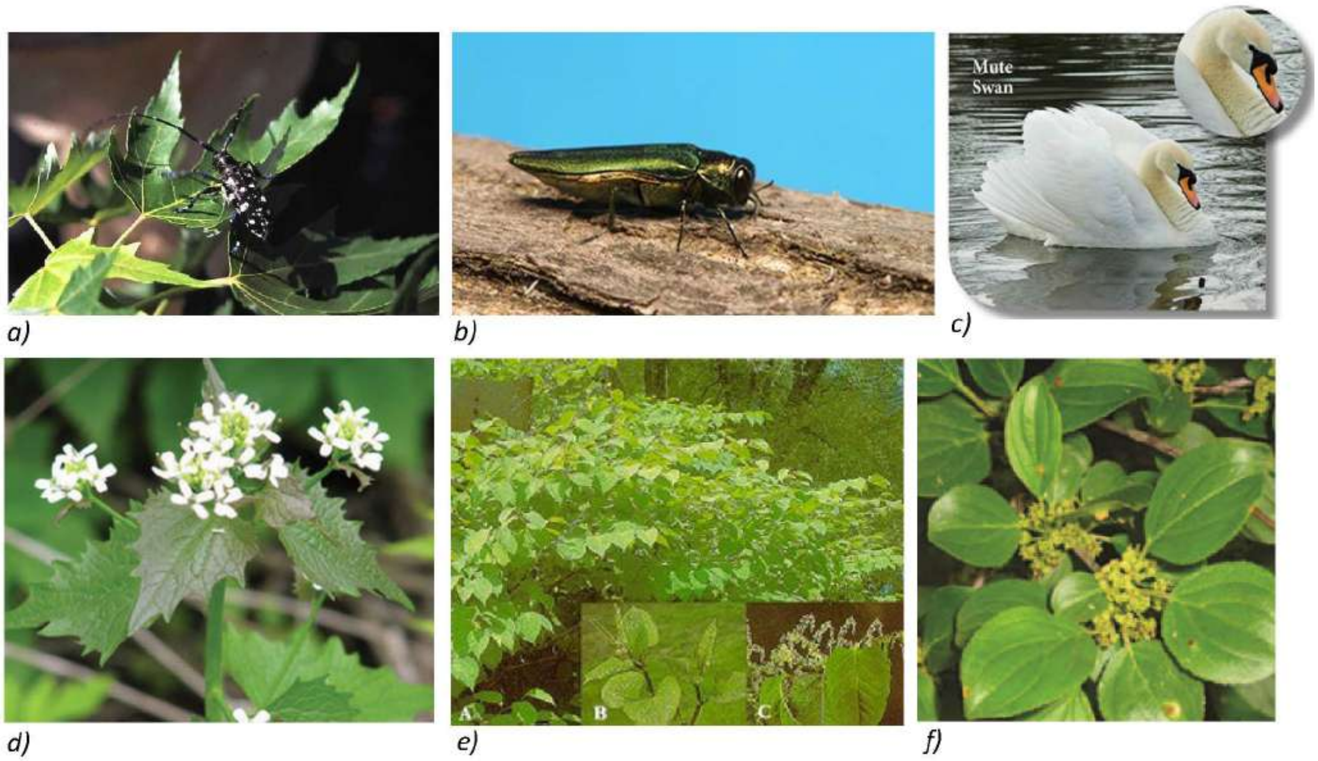
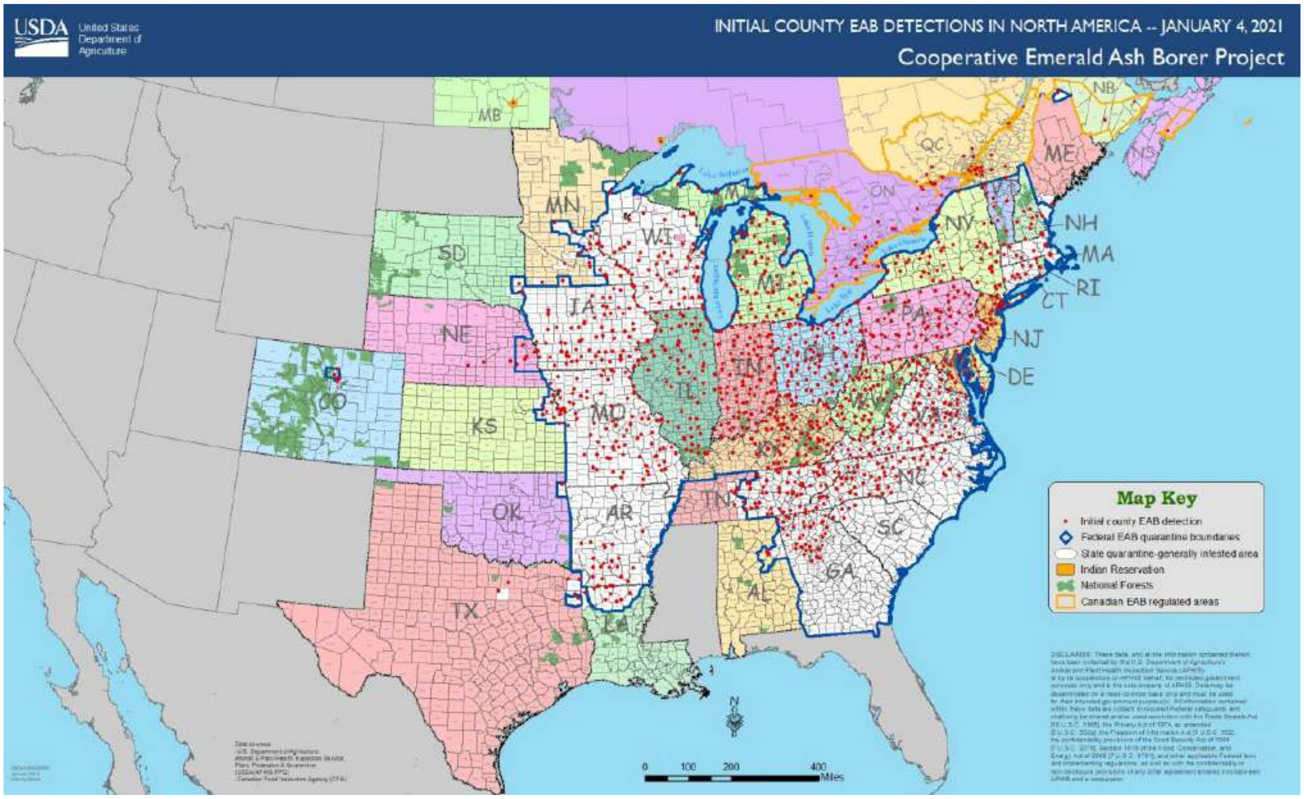


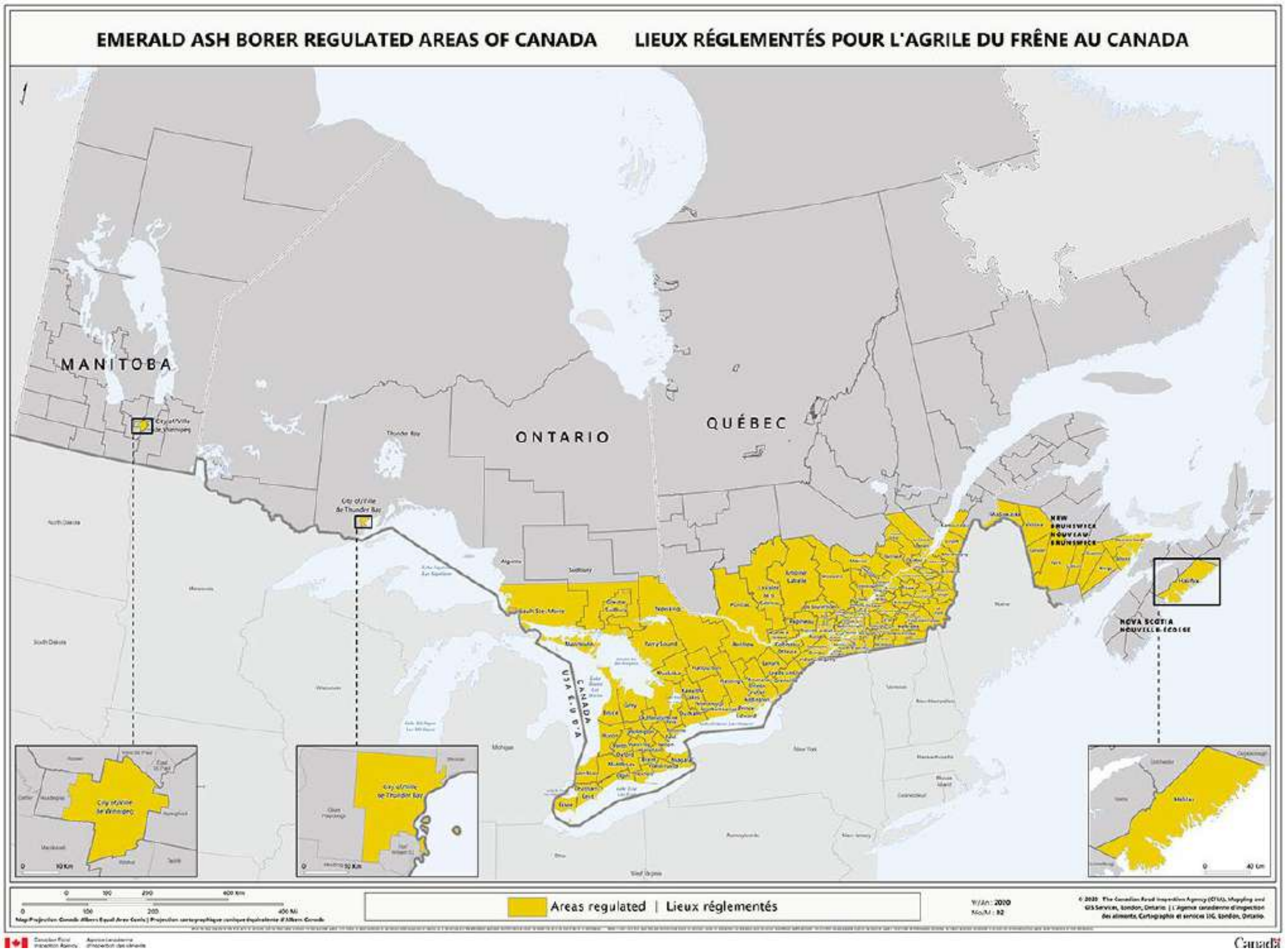
Figure 2. Photos of terrestrial invasive species included in this report, a) Asian longhorned beetle b) emerald ash borer c) mute swan d) garlic mustard e) Japanese knotweed f) common buckthorn.

Image sources: a) *Asian longhorned beetle* (2021), [Photograph]. Ontario Ministry of Northern Development, Mines, Natural Resources and Forestry. Asian long-horned beetle | ontario.ca; b) Pegler, S. (2018). *Fighting the emerald ash borer* [Photograph]. Natural Resources Canada. Fighting the emerald ash borer with science (nrcan.gc.ca); c) *Invasive species: mute swan* (2017) [Photograph]. Environment and Climate Change Canada. Invasive species: mute swan - Canada.ca; d) Mehrhoff, L. J. (2021). *First year plants produce a rosette of dark green, kidney shaped leaves* [Photograph]. University of Connecticut. Garlic mustard | ontario.ca; e) *Ontario Weeds: Japanese knotweed* (2021) [Photograph]. Ontario Ministry of Agriculture, Food and Rural Affairs. Ontario Weeds: Japanese knotweed (gov.on.ca); f) *Common buckthorn leaves and flowers* (2021) [Photograph]. Credit Valley Conservation Area. Common buckthorn | ontario.ca



Note: From “Emerald Ash Borer Information Network”, bu USDA APHIS, 2021 (Emerald Ash Borer | About Emerald Ash Borer (EAB))

Figure 3. Initial emerald ash borer detections in North America by county (January 2021).



Note: From "Areas regulated for the emerald ash borer", Canadian Food Inspection Agency, 2020, ([Areas regulated for the emerald ash borer - Canadian Food Inspection Agency (canada.ca)]())

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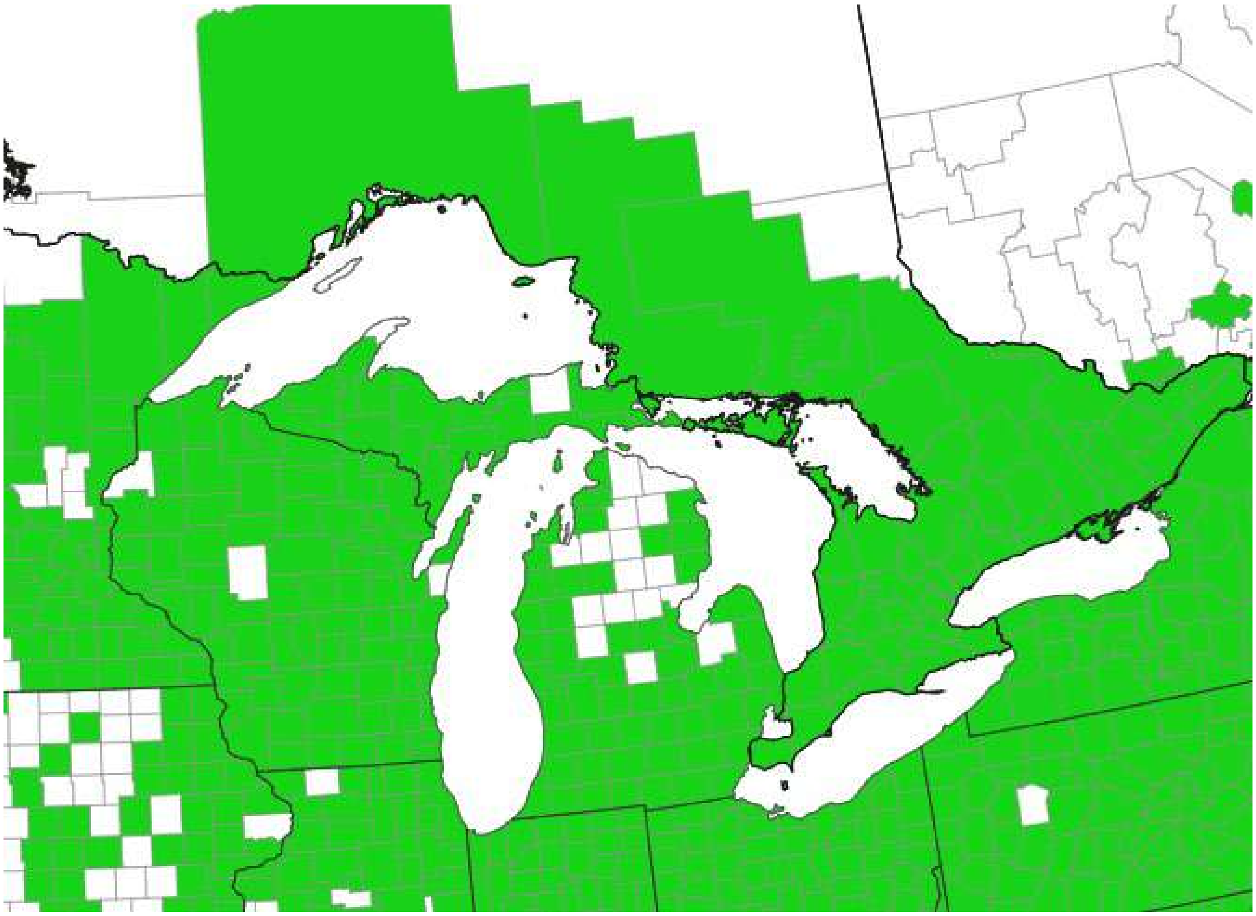
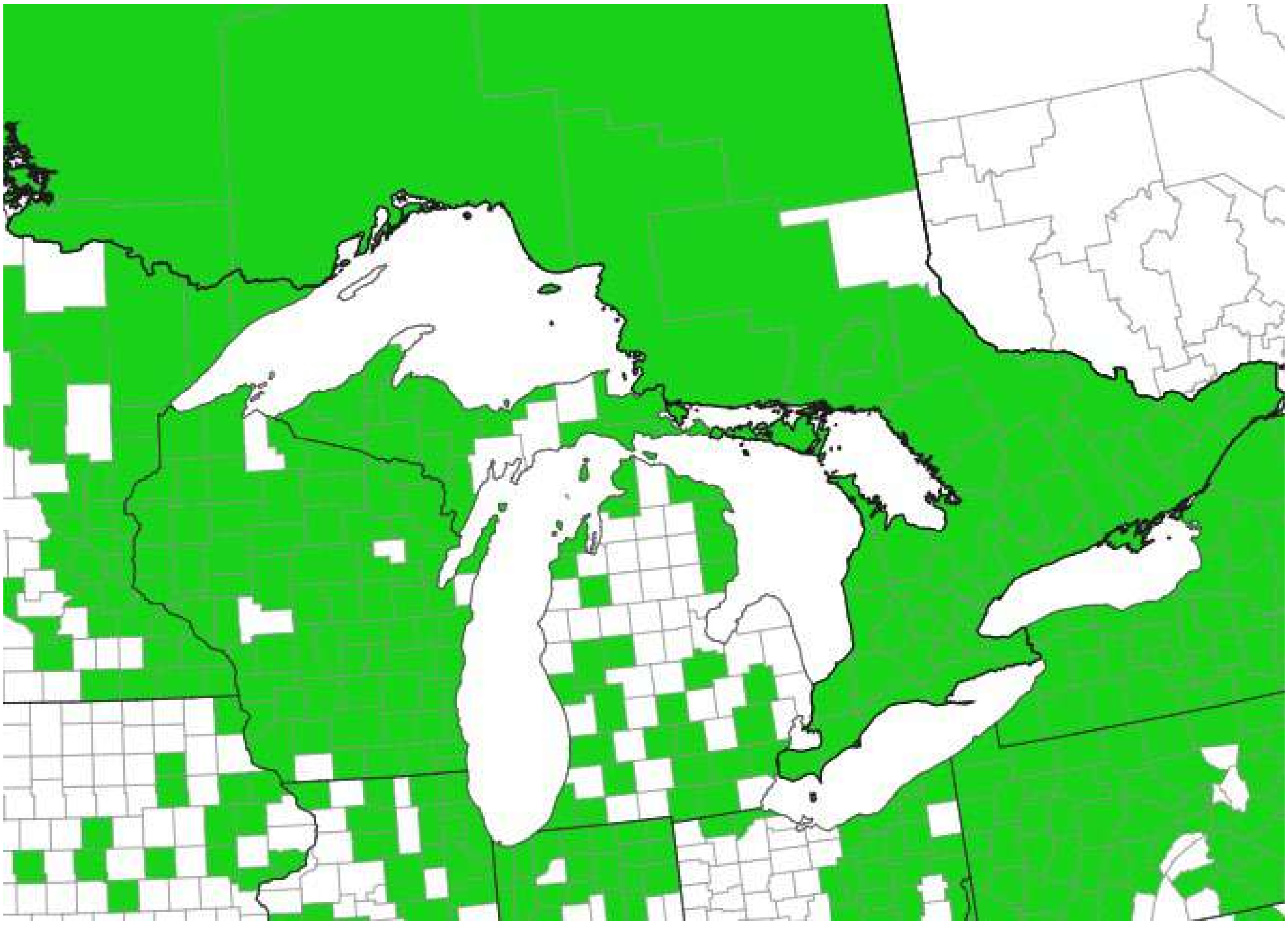


Note: EDDMapS. 2021. Early Detection & Distribution Mapping System. The University of Georgia - Center for Invasive Species and Ecosystem Health. Available online at http://www.eddmaps.org/; last accessed December 22, 2021.

Figure 5. Range of garlic mustard in the US Great Lake States and the Province of Ontario by County (U.S.) or Municipality (Canada). Green indicates the counties/municipalities where Garlic Mustard has been reported and verified species has been reported at least once, and white indicates counties/municipalities where it has not been reported/verified.



Note: EDDMapS. 2021. Early Detection & Distribution Mapping System. The University of Georgia - Center for Invasive Species and Ecosystem Health. Available online at http://www.eddmaps.org/; last accessed September 23, 2021.

Figure 6. Range of Japanese knotweed in the US Great Lake States and the Province of Ontario by County (U.S.) or Municipality (Canada). Green indicates the counties/municipalities where Japanese Knotweed has been reported at least once, and white indicates counties/municipalities where it has not been reported/verified.

Sub-Indicator: Groundwater Quality

Overall Assessment

Status: Good

Trends:

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: The overall status of groundwater quality in the Great Lakes Basin is assessed as "Good" (Figure 1). For the assessed fraction of the basin (84% of the total area), the groundwater quality is "Good" in 58% of the area, "Fair" in 41% of the area, and "Poor" in 1% of the area, resulting in an overall assessment of "Good". The portions of the basin that have insufficient data (16% percent of the total Basin area; e.g., the northern portion of the Lake Superior basin) are not assessed, and their indicator status is classified as "Undetermined" (see Basin-by-Basin Assessments below). The overall trend in groundwater quality in the basin is "Undetermined" primarily due to a lack of repeated sampling for most sites: most sites have only one sample result. However, increasing (upward) trends in chloride and nitrate concentrations in groundwater have been reported for various watersheds within the basin (see Ecological Condition section below).

The overall status of groundwater quality has changed from "Fair" in the previous report (2019) to "Good" in this report, which is attributed to the improved geospatial data coverage. Across the basin, the number of sites with available sample data and the spatial distribution of sites increased substantially for this assessment (670 data points in 2019 versus 6,554 in 2022). Although not all newly added samples were collected since the last report, the data were not available for the previous assessment.

Basin-by-Basin Assessment

Lake Superior Basin

Status: Good

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: As illustrated in Figure 2, of the 272 wells that were assessed, the groundwater quality is assessed as "Good" in 194 (71%), "Fair" in 60 (22%), and "Poor" in 18 (7%). For the assessed fraction of the lake basin area (52%), groundwater quality in 66% of the area is assessed as "Good", 34% as "Fair" and zero as "Poor." 48% of the basin area (mainly in the northern portion of the Lake basin) is classified as "Undetermined" due to insufficient data. Trend analysis was not completed in this assessment but is anticipated to be a component of future assessments.

The status of groundwater quality has changed from "Undetermined" in the 2019 report to "Good" in this report, which is attributed to the improved data coverage, especially in the southern portion of the Lake Superior basin where large data gaps existed previously. Although close to half of the basin area has insufficient data for an assessment, the total number of data points available has increased by more than 10 times since the last report (22 data points in 2019 versus 272 in 2022).

Lake Michigan Basin

Status: Good

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: As illustrated in Figure 3, of the 2,968 wells that were assessed, the groundwater quality is assessed as "Good" in 1,582 (53%), "Fair" in 735 (25%), and "Poor" in 651 (22%). For the entire lake basin area (100% assessed), groundwater quality in 77% of the area is assessed as "Good", 23% as "Fair" and zero as "Poor." Trend analysis was not part of this assessment but is anticipated to be a component of future assessments.

The status of groundwater quality has changed from "Fair" in the 2019 report to "Good" in this report, which is attributed to the improved data coverage across the lake basin. It was noted in the 2019 report that the "Fair" status should only be considered valid for the western portion of the Lake Michigan basin where sufficient data were available for an assessment. The total number of data points available has increased by more than 20 times since the last report (136 data points in 2019 versus 2,968 in 2022).

Lake Huron Basin

Status: Good

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: As illustrated in Figure 4, of the 1,308 wells that were assessed, the groundwater quality is assessed as "Good" in 739 (57%), "Fair" in 318 (24%), and "Poor" in 251 (19%). For the assessed fraction of the lake basin area (94%), groundwater quality in 62% of the area is assessed as "Good", 35% as "Fair" and 3% as "Poor." Trend analysis was not part of this assessment but is anticipated to be a component of future assessments.

Consistent with the 2019 report, the status of groundwater quality in the Lake Huron basin is assessed as "Good." It was noted in the 2019 report that the "Good" status should only be considered valid for the southeastern portion of the Lake Huron basin where data were available for an assessment. Although there are still some areas within the basin that have insufficient data for an assessment, the total number of data points available has increased by more than 10 times since the last report (77 data points in 2019 versus 1,308 in 2022).

Lake Erie Basin

Status: Good

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: As illustrated in Figure 5, of the 1,084 wells that were assessed, the groundwater quality is assessed as "Good" in 511 (47%), "Fair" in 318 (27%), and "Poor" in 251 (26%). For the assessed fraction of the lake basin area (97%), groundwater quality in 51% of the area is assessed as "Good", 48% as "Fair" and 1% as "Poor." Trend analysis was not part of this assessment but is anticipated to be a component of future assessments.

The status of groundwater quality has changed from "Fair" in the 2019 report to "Good" in this report, which is mainly attributed to the improved data coverage across the lake basin. It was noted in the 2019 report that the "Fair" status should only be considered valid within the areas of the basin where data were available for an assessment. The total number of data points available has increased by about six times since the last report (177 data points in 2019 versus 1,084 in 2022).

Lake Ontario Basin

Status: Fair

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: As illustrated in Figure 6, of the 922 wells that were assessed, the groundwater quality is assessed as "Good" in 340 (37%), "Fair" in 292 (32%), and "Poor" in 290 (31%). For the entire lake basin area (100% assessed), groundwater quality in 17% of the area is assessed as "Good", 83% as "Fair" and zero as "Poor." Trend analysis was not part of this assessment but is anticipated to be a component of future assessments.

Consistent with the 2019 report, the status of groundwater quality in the Lake Ontario basin is assessed as "Fair." The total number of data points available has increased almost four times since the last report (258 data points in 2019 versus 922 in 2022).

Status Assessment Definitions

Various standards and guidelines have been established by jurisdictions that have regulatory responsibilities in the Great Lakes basin (GLB) (Table 1). Under the Great Lakes Water Quality Agreement (GLWQA), ecosystem indicators of the GLB are required to be evaluated every three years for the status of water quality and aquatic ecosystem health. Therefore, the Canadian Council of Ministers of the Environment (CCME, 2012) water quality guideline criteria (i.e., 120 mg/L for chloride and 3 mg N/L for nitrate) for the protection of aquatic life were selected for this sub-indicator report. These criteria are also the most stringent ones among the applicable standards and guidelines as listed in Table 1. The following assessment criteria were initially applied to each monitoring location and then extrapolated to each Great Lake drainage basin (watershed).

Step 1 - Wells

For each monitoring location/well, the groundwater quality is assessed based on chloride (Cl^-) and nitrate (N/L NO_3^-) concentrations as being:

Good: less than or equal to (≤) 0.8 mg N/L NO3- AND ≤ 30 mg/L (Cl-)

Fair: greater than (>) 0.8 BUT less than (<) 3 mg N/L NO3- AND/OR >30 BUT < 120 mg/L (Cl-)

Poor: greater than or equal to (≥) 3 mg N/L NO_3^- AND/OR ≥120 mg/L (Cl^-)

These assessment criteria are shown graphically in Figure 7. In this approach, the distinction between "Fair" and "Poor" is based on the CCME 2012 water quality guideline criteria. The distinction between "Good" and "Fair" is based on concentrations equivalent to one quarter (25%) of the same guideline concentrations (i.e., 30 mg/L for chloride; 0.75, rounded up to 0.8 mg N/L for nitrate). Consistent with previous assessments, these "25% of guideline" criteria provide an interim, protective approach for this sub-indicator assessment, based on judgement rather than directly on established criteria. They may be modified in future if sufficient support for alternative criteria becomes available.

The monitoring wells used in this analysis were established for various purposes, sometimes without consideration of the regional context. However, for this sub-indicator, the objective is to provide a regional assessment of groundwater quality based on the selected parameters of interest. Statistical methods along with geographic information system (GIS) approaches could be used in future assessments to select monitoring locations that would provide sufficient information for a regional assessment for a portion of one or more of the lake drainage basins.

Step 2 - Tertiary Watersheds

Ideally, the areal (geographic) unit of observation for this sub-indicator will be the drainage basin (watershed) of each Great Lake, and also the entire Great Lakes basin. However, due to spatial gaps in available groundwater data, there were insufficient data to assess certain portions of most lake basins. Therefore, the indicator status for each tertiary watershed (TW) was evaluated prior to completing the assessment for their respective Great Lake basins based on the following criteria:

Good: If more than 50% of wells in a TW are assessed as "Good", THEN the TW is assessed as "Good".

Poor: If more than 50% of wells in a TW are assessed as "Poor", THEN the TW is assessed as "Poor".

Fair: If more than 50% of wells in a TW are assessed as "Fair", or none of the "Good", "Fair", or "Poor" is dominant (i.e., >50%), THEN the TW is assessed as "Fair".

Undetermined: TWs with fewer than two data points were not assessed and are shown as "Undetermined".

Table 2 presents a summary of the Step 2 results, by watershed and for the basin.

Step 3 - Drainage Basins

If more than 50% of the lake basin area is assessed, then an indicator status for the basin is determined. The overall groundwater quality for each of the Great Lake basin is assessed as follows:

Good: If more than 50% of the basin area are assessed as "Good", THEN the basin is assessed as "Good".

Poor: If more than 50% of the basin area are assessed as "Poor", THEN the basin is assessed as "Poor".

Fair: If more than 50% of the basin area are assessed as "Fair", or none of the "Good", "Fair", or "Poor" is dominant (i.e., >50%), THEN the basin is assessed as "Fair".

Undetermined: If greater than 50% of the basin area are unassessed due to insufficient data, THEN the basin is shown as "Undetermined".

The overall groundwater quality for the entire Great Lake basin is calculated on an areal basis using the same criteria described above for the individual lake basins. Table 3 presents a summary of the Step 3 results by watershed and for the basin.

Tables 2 and 3 and Figure 8 show the overall percentages of wells in each assessment category for each lake drainage basin and how the groundwater quality sub-indicator compares between basins. It should be noted that the TW assessment is based on the well assessment percentages, which could generate potential bias where the wells are clustered. Future assessments could explore and implement methods for a spatially unbiased assessment (e.g., Belitz et al, 2010).

As noted below, the data used for this sub-indicator will typically be derived from ongoing groundwater monitoring and surveillance programs. This will allow the use of the same data for trend assessment, as outlined below. However, this approach does not exclude the possibility of including some "one time" data for groundwater quality status assessments, particularly in areas where data gaps are found.

Trend Assessment Definitions

The following criteria are suggested for future trend assessment:

Improving: Concentrations of both nitrate and chloride are decreasing, or the decreasing trend is dominant between the two parameters (i.e., one parameter is decreasing and the other one is unchanging or increasing in a weaker pattern).

Unchanging: Concentrations of nitrate and chloride are not changing.

Deteriorating: Concentrations of both nitrate and chloride are increasing, or the increasing trend is dominant between the two parameters (i.e., one parameter is increasing and the other one is unchanging or decreasing in a weaker pattern).

Undetermined: Data are not available or are unclear to determine trends.

A statistical approach will be selected in a future reporting cycle to determine whether a concentration trajectory is decreasing, increasing, unchanging or undetermined. Future trend analysis would allow assessing whether there is a statistically significant change in the concentrations of either (or both) nitrate and chloride in a group of wells with "continuous" data within the Great Lakes basin (e.g., DeSimone et al. 2014), or a portion of it (lake watershed or sub-watershed). The time range and number of time steps or sequential analyses to be used to determine both long-term and short-term trends will be decided in future assessment reports. For this report, general trends at a national or Great Lakes scale that are reported in peer-reviewed articles and journals are referenced where appropriate.

A project initiated by the Ontario Ministry of the Environment, Conservation and Parks (MECP) and Environment and Climate Change Canada (ECCC) and completed by Kilgour & Associates Ltd. to explore a trend analysis methodology is described in the Additional Information Section of this report.

Endpoints and/or Targets

The main criterion for assessing the groundwater quality is the lowest concentration that is specified as a standard, guideline, or maximum level in one or more of the standards or guidelines shown in Table 1 for the protection of aquatic life. Under the GLWQA, the sub-indicator and indicators are used to report on water quality and aquatic ecosystem health and therefore guidelines for the protection of aquatic life will be used for assessment purposes. That being the case, the criterion for nitrate (NO_3^-) is 3 milligrams nitrogen per litre (mg N/L), and the criterion for chloride (Cl^-) is 120 mg/L.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess the general status of the quality of shallow groundwater in the Great Lakes basin, which is interactive with other components of the water cycle and has potential to impact the quality of the Great Lakes waters. Select chemical constituents of groundwater can be used to provide information about ecosystem health and potential risks to the waters of the Great Lakes basin.

Ecosystem Objective

This sub-indicator supports work towards General Objective #8 of the 2012 GLWQA, which states that the waters of the Great Lakes should be "free from the harmful impact of contaminated groundwater."

Measure

This sub-indicator utilizes available measurements of the dissolved concentrations of two relevant water-quality constituents in groundwater in the GLB: nitrate and chloride. The data are derived from groundwater samples analyzed as part of ongoing monitoring, surveillance, and other programs/projects as appropriate. Agencies conducting groundwater quality monitoring and surveillance programs in the GLB at the subcatchment to regional-scale (i.e., not site specific) include the U.S. Geological Survey (USGS) in coordination with State agencies, the MECP in coordination with various Conservation Authorities and municipalities, as well as Ontario Geological Survey (OGS).

It is important to note that for this sub-indicator, the available data are largely derived from existing monitoring networks, which were established in earlier programs, using various criteria; these networks were not established for the purposes of this sub-indicator. In this 2022 status assessment, existing data from USGS, MECP, OGS and municipal monitoring and surveillance programs for the period 2000-2020 were included. Well types include dedicated monitoring wells, private water supply wells, and public water supply wells in the GLB. For wells that have multiple samples for this period (2000-2020), the most recent sample concentrations were used.

Water-quality data for the U.S. states were primarily derived from a dataset compiled from three sources: the USGS National Water Information System (NWIS) (USGS, 2016), the U.S. Environmental Protection Agency (USEPA) Safe Drinking Water Information System (SDWIS) (USEPA, 2013), and numerous agencies and organizations at the state, regional, and local level. Most data are for sample dates from 2005 to 2013, and the most recent sample at each site was used. More details related to the dataset compilation methods are provided in the USGS ScienceBase data release (Erickson and Wilson, 2021). Water-quality data and well construction information are available in the data release (Erickson and Wilson, 2021). Groundwater quality data and well construction information were obtained for wells in Minnesota that are routinely sampled by the Minnesota Pollution Control Agency as part of its ambient groundwater monitoring program (Minnesota Pollution Control Agency, 2021). Groundwater quality data and well construction information were obtained for wells in Ohio that are routinely sampled by the Ohio Environmental Protection Agency as part of its ambient groundwater monitoring (Ohio Environmental Protection Agency, 2021).

The primary data source covering Southern Ontario is the MECP's Provincial Groundwater Monitoring Network (PGMN), which includes approximately 400 active long-term groundwater quality monitoring wells. The program, launched in 2000, is delivered in partnership with Ontario's Conservation Authorities and participating municipalities. Annual groundwater samples have been collected in fall since the establishment of this network.

As mentioned in the 2019 report, the OGS's ongoing Ambient Groundwater Geochemistry program released an extensive dataset (more than 2,300 samples have been collected from 2007-2014) for southern Ontario (Hamilton, 2015). Since then, the program has focused on sampling groundwater within the Lake Huron basin. In 2016, 196 wells were sampled in the Sudbury area (Dell et al., 2016). In 2017, 143 wells were sampled on Manitoulin Island

and along the adjacent north shore of Lake Huron (Dell et al., 2017), and in 2018, efforts were focused in the North Bay area, including a portion of the Lake Huron basin (data not available yet). The available Ambient Groundwater Geochemistry data were used for the status assessment to help address some of the spatial data gaps that were previously identified, particularly in the Lake Huron basin. However, there is no plan for future ongoing resampling of the same wells in this program that would support the trend assessment that is required for this sub-indicator.

Groundwater sampling data from some municipal supply and monitoring wells in the Greater Toronto Area compiled under the Oak Ridges Moraine Groundwater Program (ORMGP) were also used for the 2022 status assessment. In addition, some data were included from the northern portion of the Lake Huron basin and from the Lake Superior basin as compiled by Waters Environmental Geosciences Ltd. (2017). These data were from two sources: "background" wells for landfill monitoring and data from municipal water supply monitoring programs.

For subsequent status assessments, the number and location of the sampling points is expected to change over time as new wells are added, old ones are decommissioned, etc. Analyses of groundwater collected from shallow temporary drive-points might be included in future status assessments. Other available shallow groundwater data from monitoring, surveillance and research projects and programs can be included in the status assessment if appropriate for inclusion. Quality control and quality assurance measures will need to be considered for any additional data being considered. Although the use of such data would result in some inconsistency (in terms of data locations used for the assessments), their inclusion could potentially address information gaps in some areas of the GLB.

Monitoring data collected from known groundwater contamination investigations (for example, studies related to plume investigations) is excluded because such data are not representative of the regional-scale water quality that is being assessed with this sub-indicator. Point-source groundwater contamination issues would need to be dealt with on a site-by-site basis and summaries of such site-specific data could be reported in a special section of the sub-indicator report as contextual information. Point sources of contamination that are not included in this report may have groundwater quality impacts, but the purpose of this sub-indicator is to assess ambient groundwater quality at lake basin and Great Lakes basin scales. The work and resources needed to include an assessment of all known contaminated groundwater wells/sites in the Great Lakes basin would be significant. Work under the Groundwater Annex 8 of the GLWQA is considering including this type of work in future reporting.

Only "shallow" groundwater samples (collected from wells screened at depths less than 40 metres) are included in this assessment, given that shallow groundwater is the most interactive with the rest of the hydrologic system, including surface waters in the Great Lakes basin (Conant et al. 2016). Most of the shallow groundwater in the basin flows towards and will eventually discharge into the Great Lakes. This connection has many implications for water quality. Shallow groundwater tends to be "younger," or in other words, more recently recharged, and therefore it better reflects the groundwater quality impacts of recent activities in the recharge area (e.g., land use practices). That said, it should be understood that it can sometimes take years or decades for changes in land management practices to measurably impact the shallow groundwater (e.g., Zebarth et al. 2015).

The definition of "shallow groundwater" is arbitrary; for the purposes of this sub-indicator, shallow groundwater is defined as groundwater collected from wells screened less than 40 metres below ground surface. For multilevel monitoring wells, this would include only samples from the depth closest to the water table. A well screen is the perforated portion of a well that collects groundwater from a certain depth interval. For wells that had the depths of the screened interval defined in the database (491 of 6,554 wells), about 70% are potentially collecting groundwater from 20 metres or shallower (Figure 9). The 40-metre depth is a criterion that requires further research, and so the depth criterion for selection of "shallow groundwater" for this sub-indicator will be reviewed and may change over time. If the criterion is revised, the historical data will be re-assessed using the new criterion.

Ecological Condition

Background

Groundwater can become contaminated with various substances including nutrients, salts, metals, pesticides, pharmaceuticals and other contaminants. Groundwater plays an important role as a reservoir of water that, if contaminated, can become a continuous source of contamination to the Great Lakes. Chemical parameters, such as nitrate and chloride, can be used to assess groundwater quality and to provide information about ecosystem health and potential risk to Great Lakes water quality.

Nitrogen is an essential nutrient for plants and animals. It promotes rapid growth, increases seed and fruit production, and improves the quality of leaf and forage crops. Nitrogen exists in the environment in many forms as a part of the nitrogen cycle, with nitrate (NO_3^-) and ammonium (NH_4^+) being important inorganic species in aquatic systems. Nitrate concentrations in groundwater are often elevated in urban and agricultural areas (Dubrovsky et al. 2010; IJC, 2010). Chloride is believed to be mainly an urban contaminant as a result of de-icing road salt in addition to naturally occurring bedrock brines.

Elevated concentrations of nitrate in water have been shown to have detrimental effects on aquatic organisms and aquatic ecosystems (e.g., direct toxicity, or increased risk of algal blooms and eutrophication; CCME, 2012), and human health (Health Canada, 2013). Elevated concentrations of chloride in water have been shown to have detrimental effects on aquatic organisms and aquatic ecosystems (e.g., toxicity; CCME, 2012).

Nitrate and chloride are considered to be key indicator contaminants in groundwater for the following reasons:

- They are two of the most prevalent and widespread contaminants in groundwater that have been measured and reported in the GLB (and elsewhere);
- They both are derived from multiple contaminant sources in both rural (agricultural) and urban areas. Even though some geologic sources of these compounds exist in the environment, nitrate and chloride are considered as general indicators of anthropogenic impact to aquatic systems;
- As anions, they are both extremely mobile (soluble) in water, so they easily infiltrate through the soil profile and subsequently enters the groundwater system;
- They are stable contaminants that do not have much physical or chemical interaction with the material they flow through (although nitrate could be reduced or eliminated by denitrification in some subsurface environments), therefore may have a long-term adverse effect on the ecosystem; and
- Chloride is persistent in the subsurface. It is not subject to attenuation by processes such as biodegradation or sorption.

As noted in “Groundwater science relevant to the Great Lakes Water Quality Agreement” (Grannemann and Van Stempvoort, 2016): “The natural flux of groundwater to the Great Lakes and their tributaries can enhance water quality and water quantity and provide essential habitats for Great Lakes ecosystems. Groundwater can also be a transmitter (vector) of contaminants and excessive loads of nutrients, which are derived from both non-point sources and point sources, to the Great Lakes. In addition to the direct flux of groundwater that transports contaminants and nutrients to the Great Lakes, the flux of groundwater to streams flowing into the Great Lakes also must be considered because the ecology and habitats of streams are interconnected with ecology of the Great Lakes (for example, fish spawning and migration)."

Status Assessment

This sub-indicator regional-scale assessment was based on measurements (2000-2020) of the dissolved concentrations of nitrate and chloride in groundwater in the GLB, as part of ongoing monitoring of groundwater quality. For this assessment, the data were obtained from groundwater monitoring networks maintained by (1) the USGS and its partners, and (2) the MECP and its partners, and was supplemented by data from the OGS and some municipalities.

As illustrated in Figure 7, the definition of "Fair" is more inclusive than "Poor" or "Good", because "Fair" includes all cases where there is no majority of individual wells assessed as "Poor", "Fair", or "Good" (i.e., the central portion of this diagram, coloured in orange, where each of these three classifications is < 50%).

Although greatly improved from the previous assessments, the spatial distribution of data used in this assessment is still uneven (Figures 2-6), notably in the northern portion of the Lake Superior basin, the northern and eastern portions of the Lake Huron basin, and the southern portion of the Lake Erie basin.

There was a stronger tendency for groundwater quality to range from "Poor" to "Fair" in those portions of the basin that had more intense development, including urbanization (e.g., areas within the Michigan, Erie, and Ontario basins, Figures 3, 5, 6), and a tendency for groundwater quality to range from "Fair" to "Good" in the less developed areas (e.g., Superior and Huron basins, Figures 2, 4). For the subwatersheds of the Superior basin that have been assessed, the overall groundwater quality is "Good". However, only slightly more than half of the basin (by area) was assessed, due to lack of data in many areas. The unassessed subwatersheds tend to be north of the lake in areas of low populations and limited development pressures. Therefore, it is likely that the groundwater quality in at least some of these relatively pristine subwatersheds is also "Good".

There were statistically significant differences in nitrate and chloride concentration when wells were grouped by land cover and depth (Figure 10). For example, both median chloride concentration and median nitrate concentration were highest in the shallowest wells. Median chloride concentrations were highest in developed areas, and median nitrate concentrations were highest in agricultural areas. The differences between median concentrations in differing land covers and well depth ranges may reflect differences in the settings of the well sites that were outside the scope of this analysis (e.g., differences in nitrate loadings from surface, and differences in subsurface conditions such as permeability of geologic units and in intensity of microbial activity).

Although groundwater quality was assessed as "Good" for all lake basins except Lake Ontario, the distributions of chloride and nitrate concentrations were characteristically different among the basins and at various depths (Figure 11). The median chloride concentration was significantly lower in Lake Superior basin than in other basins, and was much higher in Lake Ontario and Erie basins than in Lake Michigan and Huron basins. At shallower depths (i.e., in wells less than 20 metres deep), the median chloride concentration was highest in Lake Erie basin, and at greater depths (i.e., in wells between 20 to 40 metres deep) it was highest in Lake Ontario basin. The median nitrate concentration was slightly lower in Lake Ontario basin and slightly higher in Lake Michigan basin than in other basins. However, the Lake Erie basin had a median nitrate concentration significantly higher than other basins in shallow groundwater (i.e., in wells less than 10 metres deep).

It is important to note that if only one of the two constituents that were combined for this sub-indicator (chloride and nitrate) was assessed individually, the results would be very different. For example, the "Poor" water quality assessment for the TW along the northeastern shoreline of Lake Huron was based on three sampling points that had elevated chloride but low nitrate. All three wells are located immediately adjacent to roads in relatively developed areas that were likely impacted by road salt. This example illustrates that different areas in the basin have different contaminant issues that may drive the overall assessment when combined into a multi-contaminant approach. However, for reasons discussed in previous sections, it is informative to analyze both contaminants together, in particular, as it provides a more representative assessment of ambient groundwater quality in the Great Lakes basin with the inclusion of these two contaminants from multiple sources. Consequently, the addition of other chemicals/constituents in the future would likely affect the assessments. This may require explanation when comparing updated results (that include additional constituents) to earlier sub-indicator reports.

Reported Trends of Chloride and Nitrate Concentrations in Groundwater in the Great Lakes Basin

Over the past several decades, various studies and status reports have provided information about trends of chloride and nitrate in surface water and groundwater in the Great Lakes basin. Studies indicated that groundwater concentrations of chloride and (or) nitrate in most Great Lakes basin area had increased, both in the United States (DeSimone et al., 2014; Figure 12) and in Canada (Sawyer, 2009). Ongoing monitoring of water quality in Ontario has shown that chloride concentrations have increased in lakes and streams over the past several decades (Ontario Ministry of the Environment and Climate Change, 2016).

In their effort to establish a methodology for trend analysis for this sub-indicator, Kilgour & Associates Ltd analyzed the test data from 24 municipal wells in Ontario that were sampled regularly between 1987 and 2017 as part of the MECP Drinking Water Surveillance Program. In their report, Kilgour & Associates Ltd (2018) reported nine different trends in the chloride and nitrogen data shown in Table 4. Many wells showed increasing chloride and nitrogen concentrations over time, but others exhibited decreasing trends. Though it is unknown how representative these results are with respect to the Great Lakes basin, the report provides valuable background that could support future trend analysis for this Groundwater Quality sub-indicator.

Chloride

Many studies throughout the United States side of the Great Lakes basin found that chloride levels in groundwater had increased on a decadal level (Bubeck et al.,1971; Kelly and Wilson, 2008). The USGS National Water-Quality Assessment (NAWQA) has evaluated various well networks across the United States for trends in chemical constituents. Well networks in the Lake Michigan, Huron and Erie basins all show an increasing trend in chloride (Lindsey et al. 2016, 2018).

It is generally reported that chloride concentrations were highest in shallow groundwater (DeSimone et al., 2014; Thomas 2000a; Mullaney et al., 2009), and reflected the use of deicing salt, water softeners, and many other anthropogenic sources of chloride in urban and suburban areas (Bubeck et al.,1971; Kelly and Wilson, 2008; Warner and Ayotte 2014; Rayne et al., 2019; Minnesota Groundwater Association, 2020). In contrast, recent studies by Curtis et al. (2018, 2019) suggested that natural upwelling of brines was the primary cause for the elevated chloride concentrations observed in discharge areas across Lower Michigan. An important limitation of the State of Michigan's groundwater chemistry database (WaterChem) that was used in these studies (Curtis et al., 2018, 2019) was that it does not include well depth information.

Findings on the Canadian side of the basin are similar. A decadal increase of chloride concentration in a watershed in Metropolitan Toronto was documented by Howard and Haynes (1993) and Perera et al. (2013). Their studies as well as Eyles and Meriano (2010) all indicated a connection between road salt impacted shallow groundwater and its discharge as springs and baseflow. The Ontario Ministry of the Environment (1998) reported that a high percentage of stream water quality monitoring stations along Lake Erie had increasing chloride concentrations "indicative of the significant amount of urbanization and development that has occurred in watersheds in southern Ontario since the early 1980s." In some locations in southern Ontario, natural elevated concentrations of chloride in groundwater are associated with shales (brines) (e.g., Singer et al., 2003).

It was reported that chloride concentration in groundwater exceeded background level for 100 – 1,000 times in some urban springs and shallow waters in southern Ontario (Howard and Beck, 1993). The increase of chloride concentrations was attributed to road salts, landfill leachates, agricultural fertilizers and saline bedrock waters (Howard and Beck, 1993; Bowen and Hinton, 1998; Williams et al., 2000). Chloride concentration increases in groundwater and the link to urban growth and associated land uses were also noted at some municipal wells across southern Ontario (Sawyer et al., 2009; South Georgian Bay-Lake Simcoe Source Protection Committee., 2015; Credit Valley Conservation Authority, 2012, 2015; Central Lake Ontario Conservation Authority, 2015).

Nitrate

There are limited studies regarding trends of groundwater nitrate concentration, especially on a regional or basin scale. The USGS NAWQA program found no trend or increasing trends in nitrate in the Lake Erie and Lake Michigan basins (Lindsey et al. 2016, 2018). Gardner et al (2020) observed no distinctive trends of groundwater nitrate concentration at three sites with different hydrogeologic and land-use settings in the northeastern Lake Erie basin. The study indicated that the combination of climatic conditions, land-use pressure, and hydrogeologic setting played an important role in the characteristics of groundwater nitrate time-series trends. Similar to chloride, increasing nitrate concentrations have been reported in some municipal supply wells in southern Ontario (Credit Valley Conservation Authority, 2015; Ontario Conservation Authority, 2015).

Some groundwater quality studies noted linkages between high levels of nitrate in groundwater and human sources, such as fertilizer, manure and septic systems (Hill, 1982; Sawyer, 2009; Thomas, 2000b). Others found the occurrence and concentration of nitrate in groundwater were directly proportional to well depth (i.e., higher in shallow dug or bored wells) and soil permeability (Goss et al, 1998). A water quality study of the glacial aquifer system in the northern United States found that nitrate above 10 mg N/L was also associated with oxidizing groundwater conditions (Erickson et al, 2019).

Linkages

Linkages to other Great Lakes sub-indicators include:

- Treated Drinking Water – The groundwater quality sub-indicator only incorporates information about chloride and nitrate in ambient groundwater (i.e., untreated), whereas assessment of the quality of drinking water takes into consideration a much broader range of parameters in both surface water and groundwater.
- Water Quality in Tributaries – This sub-indicator is based on the Water Quality Index (WQI), which is calculated using a total of eight parameters, including both chloride and nitrate concentrations in surface water.
- Coastal Wetlands: Extent and Composition – Groundwater seeps into coastal wetlands, and some coastal wetlands are groundwater-dependent; the health of these ecosystems is influenced by the quality of groundwater discharging to them.
- Nutrients in Lakes (open water) – Nutrients in groundwater that discharges to the lakes or to streams flowing into the lakes, may affect the nutrients in the lakes, especially in nearshore areas.
- Land Cover – to some extent, the pattern of groundwater quality status appears to be associated with land-use and development patterns. Poorer groundwater quality tends to be in shallow wells in areas of urban and agricultural land cover. Additional statistical analysis is warranted to confirm linkages in

future reports.

- Human Population – Similarly, poorer groundwater quality tends to be in areas that are more densely populated. Additional statistical analysis is warranted to confirm linkages in future reports.
- Baseflow due to Groundwater. Groundwater discharge is a main component of baseflow, and the quality of this groundwater affects the quality of streams.
- Water Levels – Changes in lake water levels could be linked to changes in groundwater quality, particularly in nearshore areas, and to changes in baseflow.
- Precipitation Amounts – Changes in precipitation amounts could be linked to changes in baseflow and groundwater quality, in part by changing the depth of the water table and by affecting shallow groundwater flow systems and groundwater – surface water interaction.

Future consideration of the above linkages may be useful in terms of demonstrating how regional groundwater quality patterns are related to surface water quality, habitats and various stressors.

Future consideration could include how changes in climate, including changes in precipitation amounts/timing and/or in surface water temperatures, might indirectly impact this sub-indicator. The nature of such impacts is currently unknown.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the United States are comparable to those from Canada		X		
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	If Yes, please include URL: MECP PGMN groundwater quality data: https://data.ontario.ca/dataset/provincial-groundwater-monitoring-network USGS data release: https://doi.org/10.5066/P9JT8PXS		

Data Limitations

The networks of monitoring wells that were used for this assessment differ on the United States and Canadian sides of the border. Specifically, the ages of the wells, their construction methods, and the criteria that were used for selecting these monitoring wells were different. Although the methods of chemical analyses used for the United States and Canadian data also differed, this is not likely to have had a substantial effect on the outcome of the assessment.

The well construction and water quality dataset were compiled from several sources to obtain information on well depths and water-quality data. Organizations provided data voluntarily, so data density varies across the study area. Nitrate and chloride are not monitored equally across the study area. Therefore, the degree of assessment of individual constituents is not equal across the study area. Results from assessed areas cannot be directly generalized to unassessed areas. We performed reasonable quality assurance during data compilation to ensure that wells were less than 40 metres deep and that samples were of raw (untreated and unblended) source water. Exhaustive quality assurance across the myriad data sources was, however, outside of the scope of the study.

This sub-indicator considers only two contaminants (nitrate and chloride) and is therefore not meant to capture all possible groundwater contamination issues or problems at a given location. Groundwater can be contaminated by many other substances including geologic source constituents (e.g., arsenic and manganese), human source constituents (e.g., petroleum compounds or organic pesticides), or other substances, such as pathogenic microorganisms. Also, the impact of some industrial sectors on groundwater quality is not well assessed by the two chemical constituents that have been selected (e.g., mining sector).

The groundwater quality assessment is based on the available samples/wells, and this limitation resulted in major spatial data gaps (e.g., Lake Superior basin), which are discussed above. There appear to be wells in most places where development has taken place; however, some data that could be useful may not be readily accessible (e.g., private monitoring networks).

Some zones in the Great Lakes basin may have shallow groundwater with high chloride concentrations that are controlled by hydrogeological/geochemical conditions. One approach to determining chloride source would be to look at chloride/bromide (Cl/Br) ratios, which holds some promise in terms of helping to distinguish between geologic versus anthropogenic chloride (e.g., derived from de-icing salt) (e.g., Thomas, 2000a; Panno et al. 2006; Katz et al., 2011). Zones with high chloride in groundwater from geologic sources might be particularly vulnerable to anthropogenic chloride contamination (i.e., even higher chloride concentrations), and in that sense could be still considered “Poor”. However, further research or literature review to see if such areas could support wetland or stream ecosystems that are adapted to higher chloride in the groundwater could be helpful. In future assessments, some analysis will be completed to determine how the datasets compare on the two sides of the international border (e.g., spacing of data, depth of wells, types of sampling and analytical methods) and original purpose of wells installed. Information about aquifers is not included in this assessment.

Additional Information

Potential for Future Trend Assessments

In February 2018, in collaboration with MECP, ECCC initiated a project to support future development of the required trend assessments for this sub-indicator. Specifically, ECCC contracted Kilgour & Associates Ltd. to undertake the following four tasks: (1) conduct a literature review to establish the most appropriate method or methods for determining significant trends of nitrate and chloride concentrations in groundwater within the Great Lakes basin; (2) conduct the first stages of determining appropriate and relevant trend methods such as testing for normalcy; (3) apply selected method to a representative dataset of well groundwater quality data; (4) review software used to apply the selected methods and discuss the limitations, common errors and results of their application. The results of this contract work are now available for consideration for future trend assessments.

Potential for Future Incorporation of Additional Data from other Sources to Address Gaps in Monitoring Networks

In February 2017, ECCC contracted Waters Environmental Geosciences Ltd. to explore potential additional sources of groundwater quality information in portions of the Lake Superior, Huron and Ontario basins in the Province of Ontario. Data were retrieved from 31 additional wells, including nine wells in the Lake Superior basin and 22 wells in the Lake Huron basin. Fourteen of these 31 wells were "background" wells from landfill monitoring programs, assumed to be unaffected by landfill impacts. The other seventeen were municipal water supply wells, which were clustered in six urban areas (including cities and small communities). Based on the status criteria for this sub-indicator, groundwater quality in the majority of the landfill monitoring background wells was "Good", whereas the majority of the municipal water supply wells had "Fair" groundwater quality. For the municipal supply wells, nitrate concentrations were low, but chloride was often between 30 and 120 mg/L, which caused the "Fair" classification. These additional data identified by Waters Environmental Geosciences Ltd. (2017) were included in the current status assessment. However, due to its sparse distribution and the fact that there was significant clustering of the municipal water supply wells in urban areas, large spatial information gaps remain regarding groundwater quality in the Lake Superior and Lake Huron basins.

For future consideration, Waters Environmental Geosciences Ltd. (2017) identified several potential data sources that could be used in the future to address these gaps. The key ones with data that would be updated in future (thus also useful for sub-indicator trend analyses) are additional "background" wells in landfill monitoring programs, and other monitoring wells used for Source Water Protection Programs and Municipal Groundwater Supply Systems.

Additional work to consider includes: (1) identification of additional groundwater quality data sources relevant to the study area; (2) capturing and aggregating the data identified by Waters Environmental Geosciences Ltd. (2017); (3) creating a map depicting the geographic locations of all data captured; (4) documenting the feasibility of identifying and capturing data from the approximately 4,000 landfill sites in southern Ontario.

Best Management Practices to Reduce Chloride and Nitrate

Ongoing development of Best Management Practices (BMPs), including research and development, promotion and implementation, will potentially result in reduction of chloride and nitrate concentrations in groundwater in the Great Lakes basin.

Various BMPs have been advanced to reduce fluxes of nitrogen (including nitrate) to groundwater from agricultural sources, such as avoiding excessive nitrogen loading (livestock manure, field applications of nitrogen for crops); proper storage of manure; timely applications of nitrogen for maximum crop uptake; and use of buffer strips in riparian zones. Some research has documented the successful implementation of such BMPs. For example, after a decade of implementing BMPs to reduce nitrate loading in Oxford County, Ontario, the average nitrate concentration in a municipal production aquifer dropped by about 50% (Rudolph, 2015).

BMPs have also been developed to mitigate the impact of de-icing salts on the environment. For example, Environment Canada (2004) published a "Code of Practice for the Environmental Management of Road Salts," which included a "Syntheses of Best Practices." In July of 2018, Conservation Ontario released "Good Practices Guidance for Winter Maintenance in Salt Vulnerable Areas", which includes guidance that currently focuses on protecting municipal drinking water sources that have high levels of sodium or chloride (Conservation Ontario, 2018). Also, the Regional Municipality of Waterloo in Ontario implemented several salt reduction BMPs in 2003-2004. Over the following 6-year period, a downward trend in chloride concentrations in shallow groundwater was documented in one municipal wellfield (Stone et al. 2010). In addition, some studies suggested optimizing water softener operation, especially in communities serviced by hard water, to reduce chloride contribution to shallow groundwater through septic systems (Strifling et al, 2017, Kyser and Doucette, 2018).

However, further adoption of these practices would be beneficial, being mindful that it can take many years for changes in land management practices to positively impact shallow groundwater quality.

Application to the Entire Great Lakes Basin

Provided that sufficient data are available (based on professional judgement), the same approach as used for the lake watersheds and sub-watersheds is used to provide a groundwater assessment for the entire Great Lakes basin. In this case, all data for all five lake drainage basins will be included in the assessment.

Main considerations for future assessments include:

- Improving the status assessment approach to resolve the potential bias associated with the spatially unevenly distributed data points.
- Determining the statistical approach to use in order to identify trends for the Great Lakes basin.
- Considering the inclusion of other constituents to assess groundwater quality in future reports. Although phosphate has been a major focus in current assessments of surface water quality in the GLB, phosphate was not used as a constituent for this sub-indicator. Phosphate is generally much less mobile in the subsurface compared to nitrate and chloride. However, phosphate can be mobile in groundwater under some conditions, so it may be appropriate to include phosphate in future assessments.
- Considering the inclusion and value of other chloride and nitrate surveillance data that are not part of ongoing monitoring programs in future reporting cycle to further address data gaps.

The "USGS Online Mapper" (https://www.usgs.gov/news/usgs-online-mapper-provides-decadal-look-groundwater-quality,) is an online interactive mapping tool that provides "summaries of decadal-scale changes in groundwater quality" across the United States, including areas in the Great Lakes basin (Lindsey et al. 2016, 2018). How the contaminated groundwater impacts and interacts with the water of the Great Lakes, in particular in the nearshore zone, requires a better understanding.

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Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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Figure 7. Criterion diagram for groundwater quality indicator status assessment.

Figure 8. Ternary diagram summarizing the individual well assessments for each lake basin (see Table 2). To improve clarity where symbols overlap, the symbols for two lakes are coloured blue.

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Figure 9. Cumulative percentage of wells screened above a given depth (metres below ground surface, mbgs). For example, for wells that had the depths of the screened interval defined in the database (491 of 6554 wells), about 50% are potentially collecting groundwater from 12 metres or shallower; 70% from 20 metres or shallower.

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Source: DeSimone et al. (2014)

Last Updated
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Table 1. Examples of standards and water quality objectives for some of the jurisdictions that are responsible for protecting human health and environmental water quality in the Great Lakes basin. Source: U.S. Environmental Protection Agency, Canadian Council of Ministers of the Environment, Health Canada, Ontario Ministry of the Environment, Conservation and Parks.

Jurisdiction	Standard / guideline for chloride (milligrams per litre)	Standard / guideline for nitrate (milligrams nitrogen per litre)
U.S. Environmental Protection Agency	230 (chronic criterion for aquatic life = criterion continuous concentration)	10 (maximum contaminant level goal for drinking water)
Canadian Council of Ministers of the Environment	120 (water quality guideline for the protection of aquatic life, freshwater concentration, long term)	3 (water quality guideline for the protection of aquatic life, freshwater concentration, long term)
Health Canada	≤ 250 (aesthetic objective)	10 (health based maximum acceptable concentration)
Ontario Ministry of the Environment, Conservation and Parks	250 (aesthetic objective level)	10 (maximum acceptable concentration in drinking water)

Table 2. Summary of individual well data assessments for each Great Lake basin and all basins. Source: Ontario Ministry of the Environment, Conservation and Parks, U.S. Geological Survey, Ontario Geological Survey and Oak Ridges Moraine Groundwater Program.

	Total Number of Wells	Assessed as "Good"		Assessed as "Fair"		Assessed as "Poor"	
		# Wells	%	# Wells	%	# Wells	%
Superior	272	194	71.3%	60	22.1%	18	6.6%
Michigan	2968	1582	53.3%	735	24.8%	651	21.9%
Huron	1308	739	56.5%	318	24.3%	251	19.2%
Erie	1084	511	47.1%	289	26.7%	284	26.2%
Ontario	922	340	36.9%	292	31.7%	290	31.5%
All Basins	**6554**	**3366**	**51.4%**	**1694**	**25.8%**	**1494**	**22.8%**

Table 3. Summary of areal data assessments by drainage basin for each Great Lake and all basins. Source: Ontario Ministry of the Environment, Conservation and Parks, U.S. Geological Survey, Ontario Geological Survey and Oak Ridges Moraine Groundwater Program.

	Total Number of Tertiary Watersheds	Number of Tertiary Watersheds Assessed	Areal Percentage of Lake Basin				Basin-wide Assessment
			Assessed as "Good"	Assessed as "Fair"	Assessed as "Poor"	Not Assessed ("Undetermined")	
Superior	25	17	66.0%	34.0%	0.0%	48.1%	GOOD
Michigan	33	33	76.7%	23.3%	0.0%	0.0%	GOOD
Huron	38	35	62.0%	35.4%	2.6%	5.7%	GOOD
Erie	31	30	51.5%	47.6%	0.9%	2.6%	GOOD
Ontario	23	23	17.3%	82.7%	0.0%	0.0%	FAIR
All Basins	**150**	**138**	**58.5%**	**40.6%**	**0.9%**	**14.6%**	**GOOD**

Table 4. Types of trends identified using MECP Drinking Water Surveillance Program data. Source: Kilgour & Associates Ltd. (2018) report

Type of trend	Number of wells with chloride data that followed this trend	Number of wells with nitrogen data that followed this trend
1. Linear increase	5	0
2. Increase that flattened out over time	4	1
3. Almost flat, very gradual increase	3	9
4. Flat period followed by an increase	1	2
5. Increase that leveled off, then resumed	3	0
6. Decrease that flattened out over time	0	2
7. Flat period followed by a decrease	4	4
8. Increase followed by a decrease	4	3
9. Decrease that leveled off, then resumed	0	3

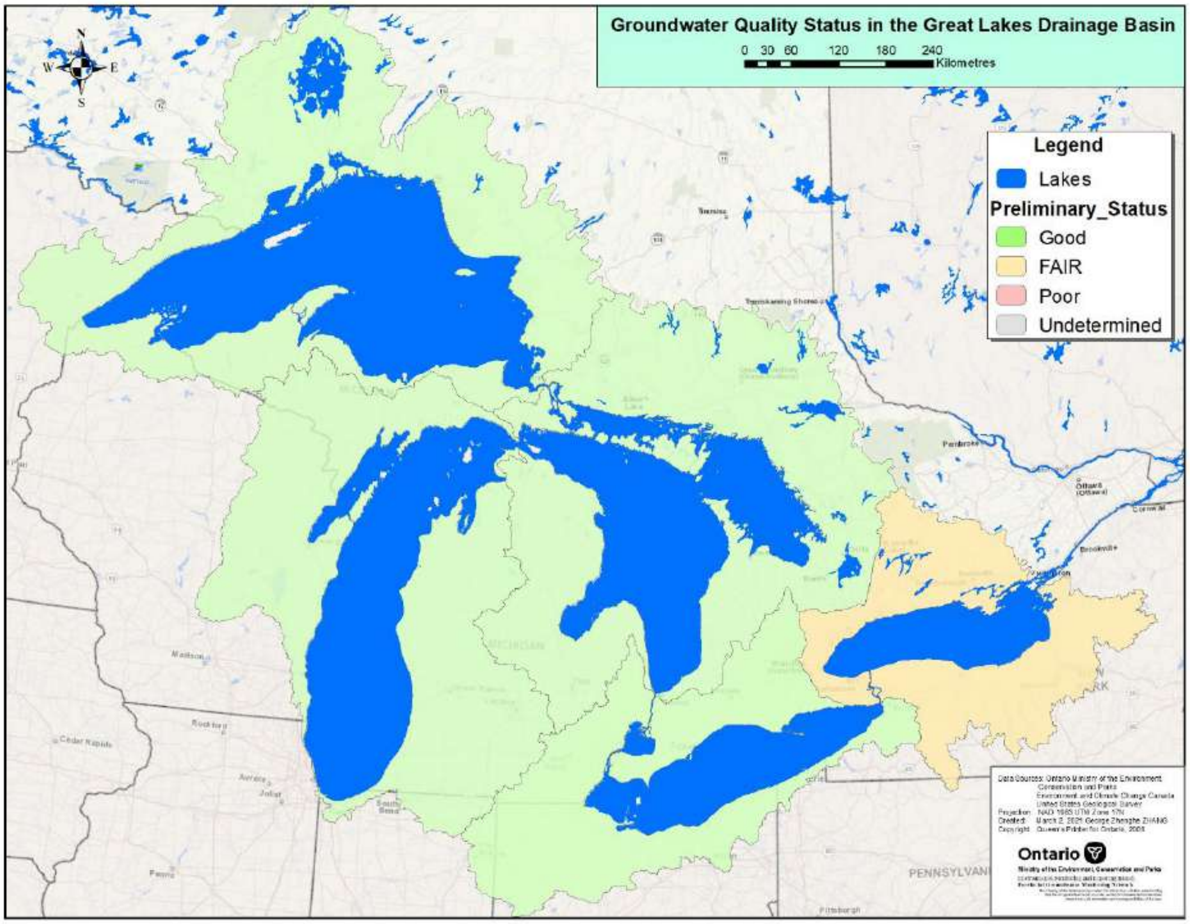


Figure 1. Groundwater quality status in the Great Lakes basin is based on nitrate and chloride concentrations in shallow groundwater (based on measurements from 2000-2020 for wells ≤ 40 m below ground). A total of 6,554 wells in the basin were included in the analysis. The groundwater quality was assessed as "Good" in 3,366 wells, "Fair" in 1,694 wells, and "Poor" in 1,494 wells. The shaded areas shown (see legend) indicate the status in the individual lake basins: fair in the Lake Ontario basin, good in the other four lake basins. See Figures 2 through 6 for details. Sources of data: Ontario Ministry of the Environment, Conservation and Parks, U.S. Geological Survey (Erickson and Wilson, 2021), Ontario Geological Survey, Oak Ridges Moraine Groundwater Program, and other Ontario sources (Waters Environmental Geosciences Ltd., 2017).

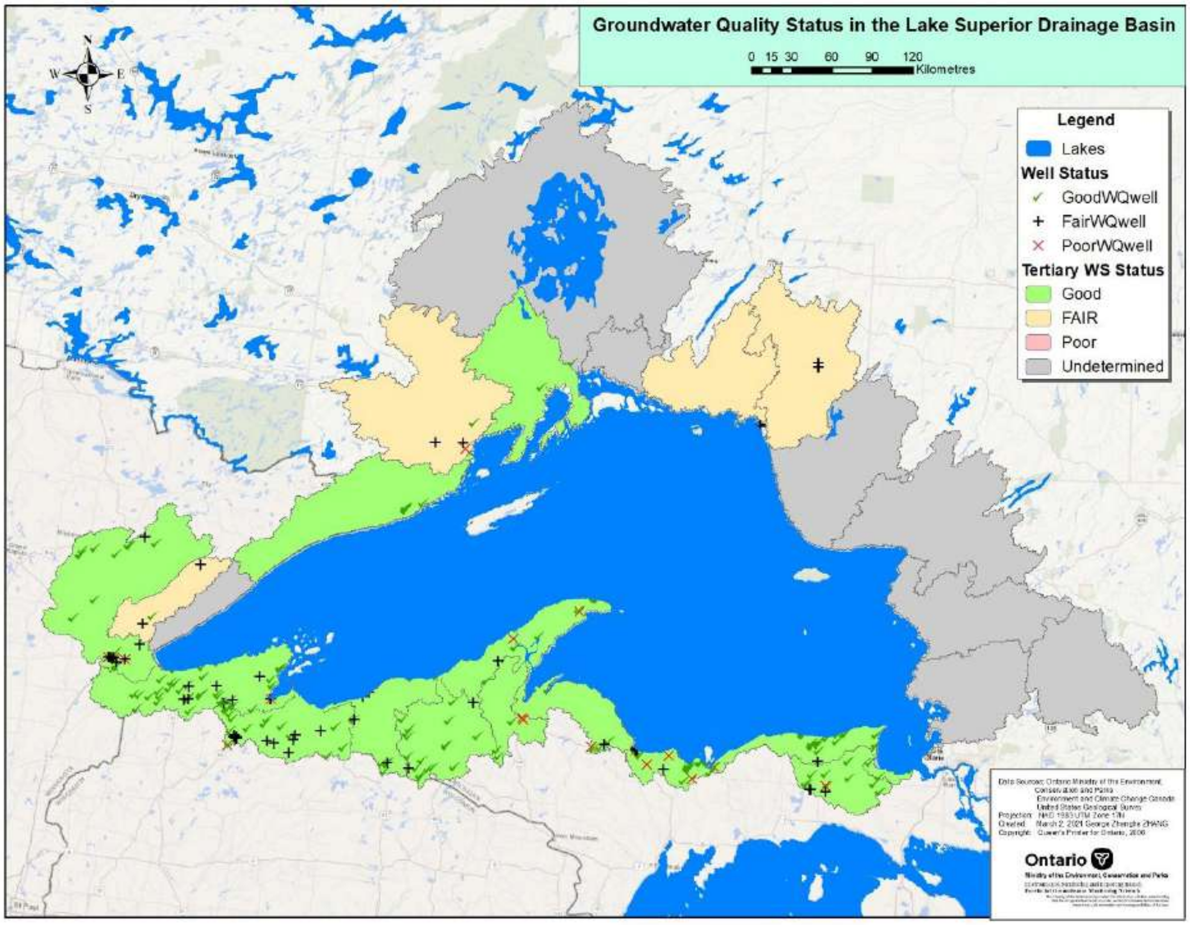


Figure 2. Assessment results for the groundwater quality sub-indicator for the Lake Superior basin (based on measurements from 2000-2020 for wells $\leq$ 40 m below ground). Symbols indicate the results for individual monitoring wells. Sources of data: U.S. Geological Survey (Erickson and Wilson, 2021), Ontario Ministry of the Environment, Conservation and Parks and other Ontario sources (Waters Environmental Geosciences Ltd., 2017).

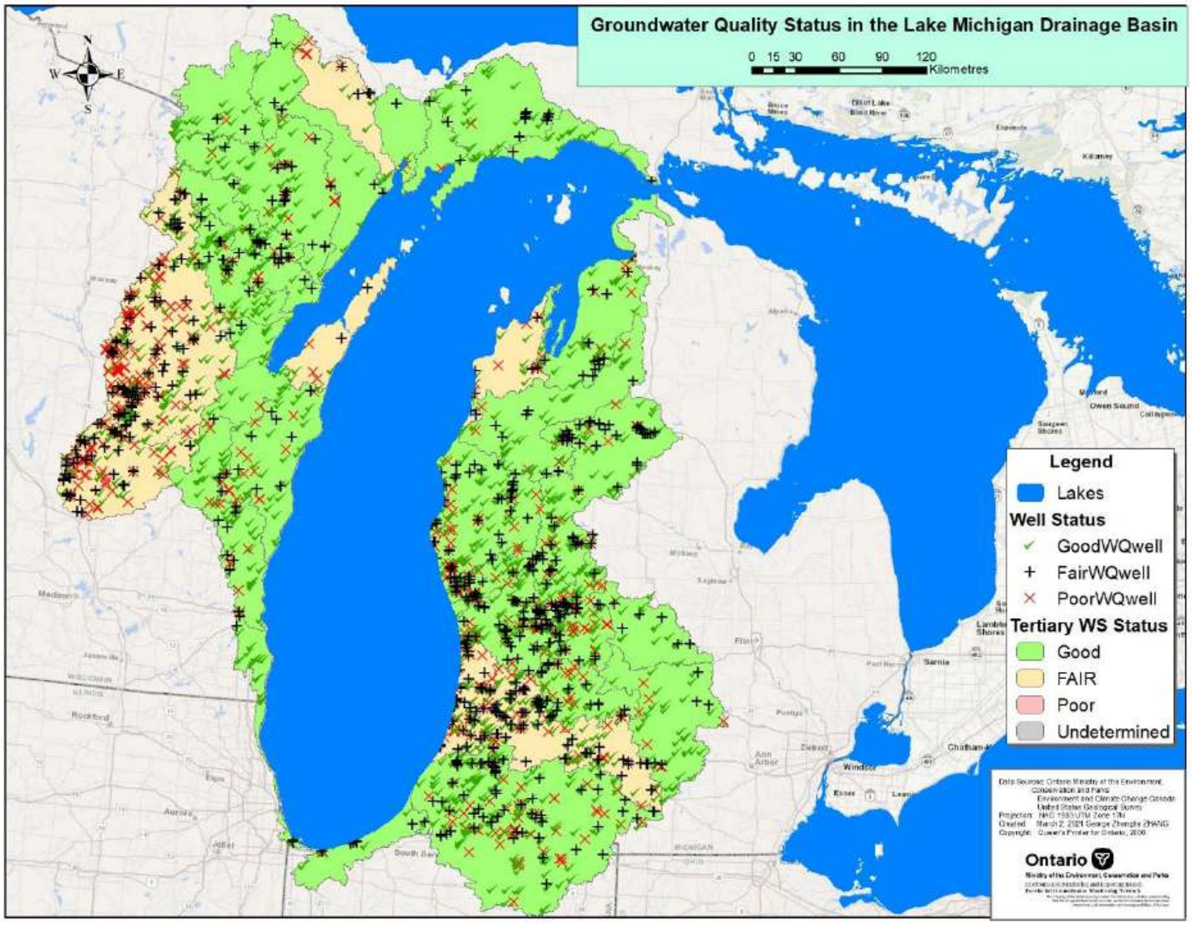


Figure 3. Assessment results for the groundwater quality sub-indicator for the Lake Michigan basin (based on measurements from 2000-2020 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Source of data: U.S. Geological Survey (Erickson and Wilson, 2021).

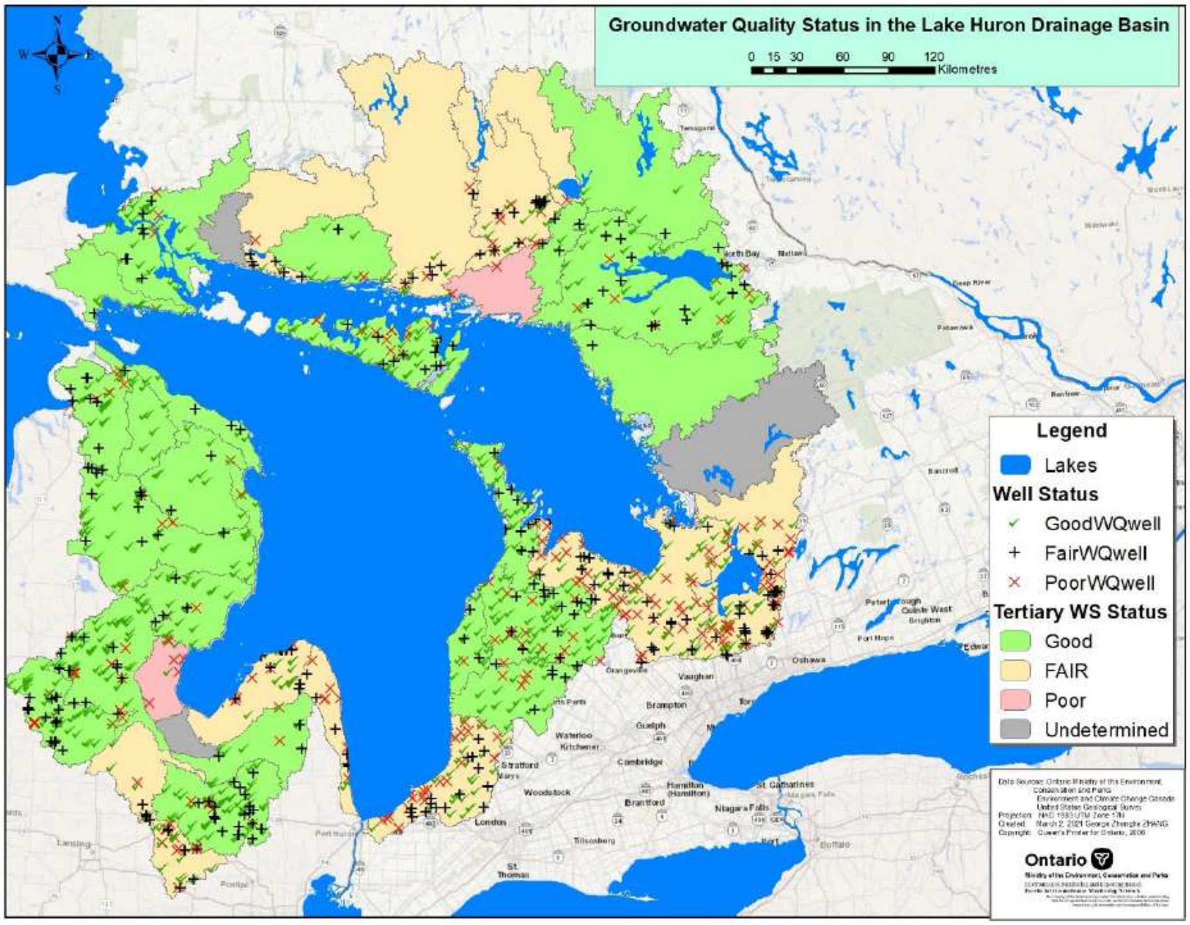


Figure 4. Assessment results for the groundwater quality sub-indicator for the Lake Huron basin (based on measurements from 2000-2020 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Sources of data: U.S. Geological Survey (Erickson and Wilson, 2021), Ontario Ministry of the Environment, Conservation and Parks, Ontario Geological Survey, other Ontario sources (Waters Environmental Geosciences Ltd., 2017).

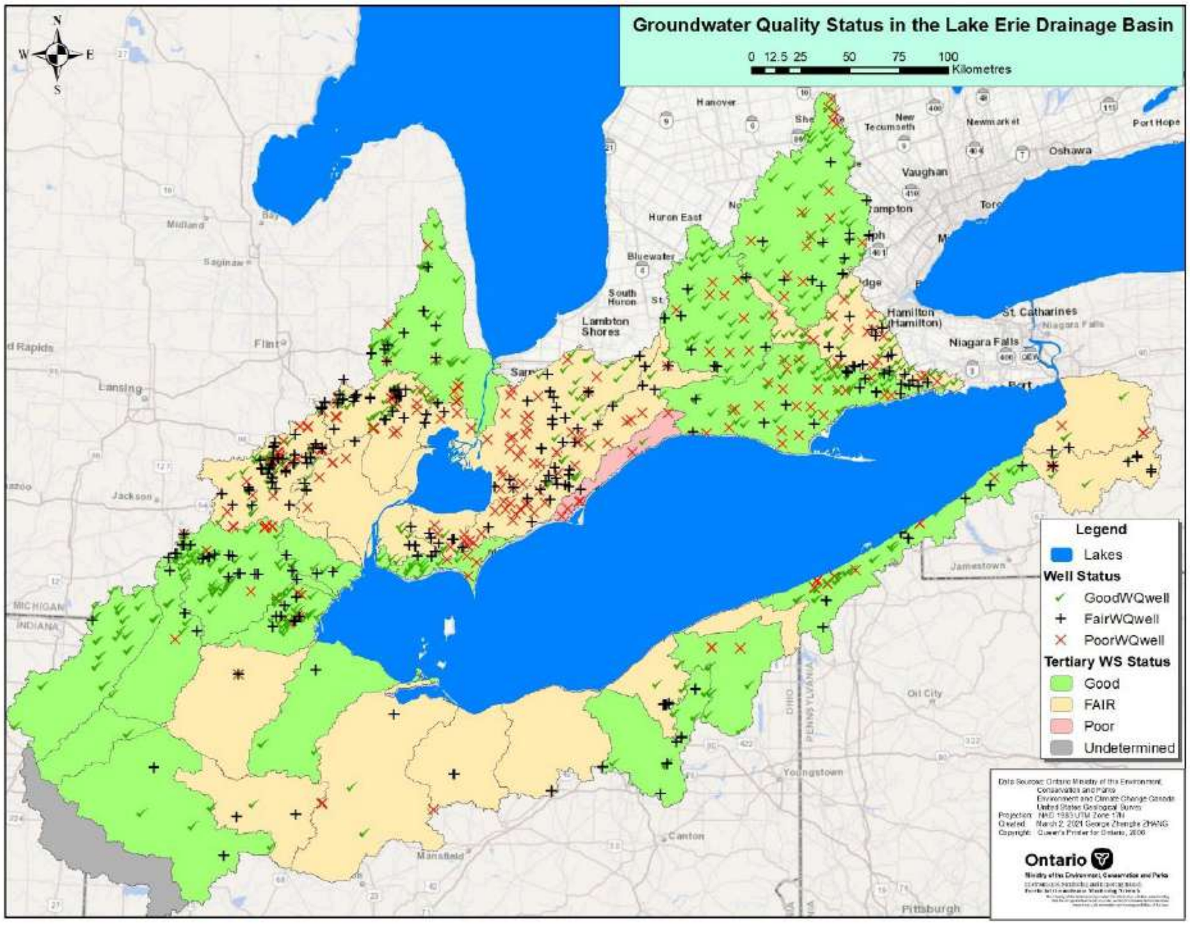


Figure 5. Assessment results for the groundwater quality sub-indicator for the Lake Erie basin (based on measurements from 2000-2020 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Sources of data: Ontario Ministry of the Environment, Conservation and Parks, Ontario Geological Survey and U.S. Geological Survey (Erickson and Wilson, 2021).

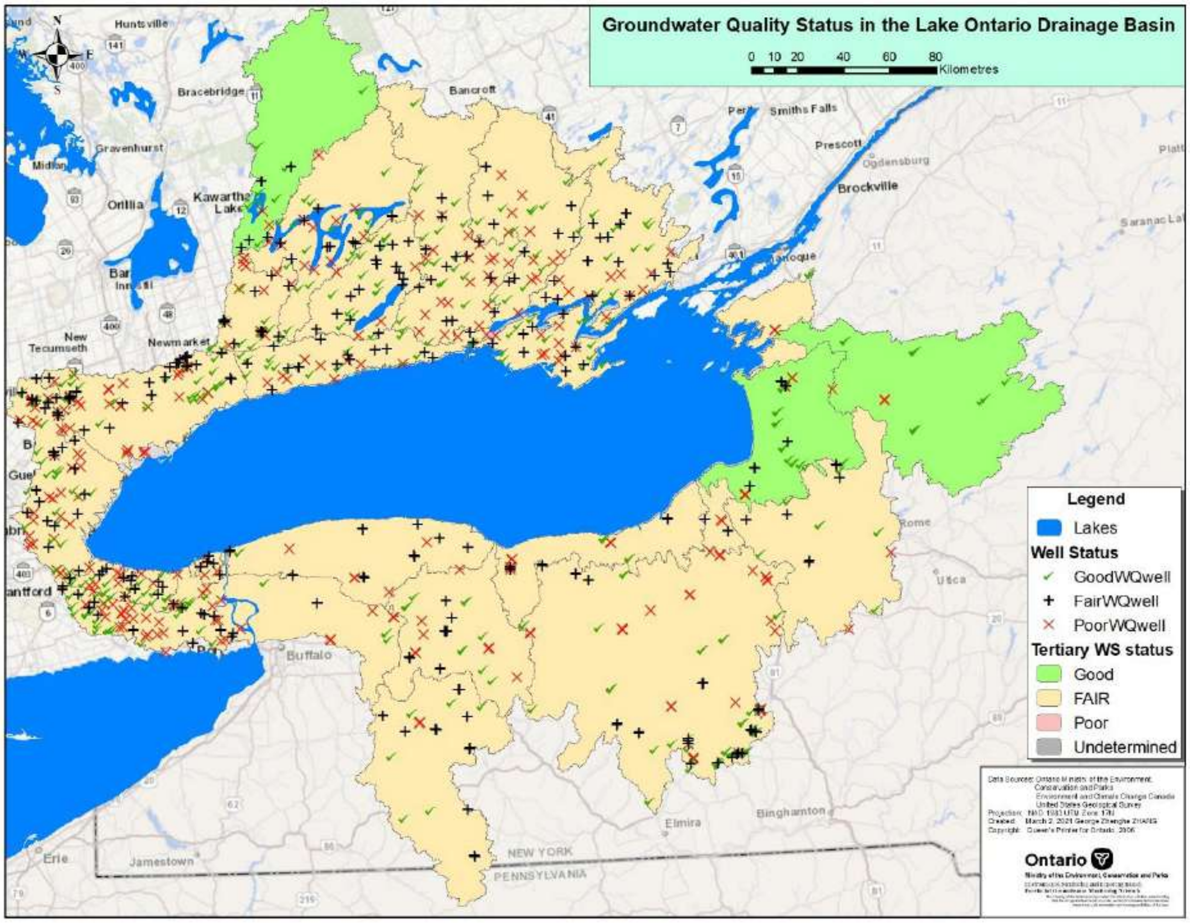


Figure 6. Assessment results for the groundwater quality sub-indicator for the Lake Ontario basin (based on measurements from 2000-2020 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Source of data: Ontario Ministry of the Environment, Conservation and Parks, U.S. Geological Survey (Erickson and Wilson, 2021), Ontario Geological Survey and Oak Ridges Moraine Groundwater Program.

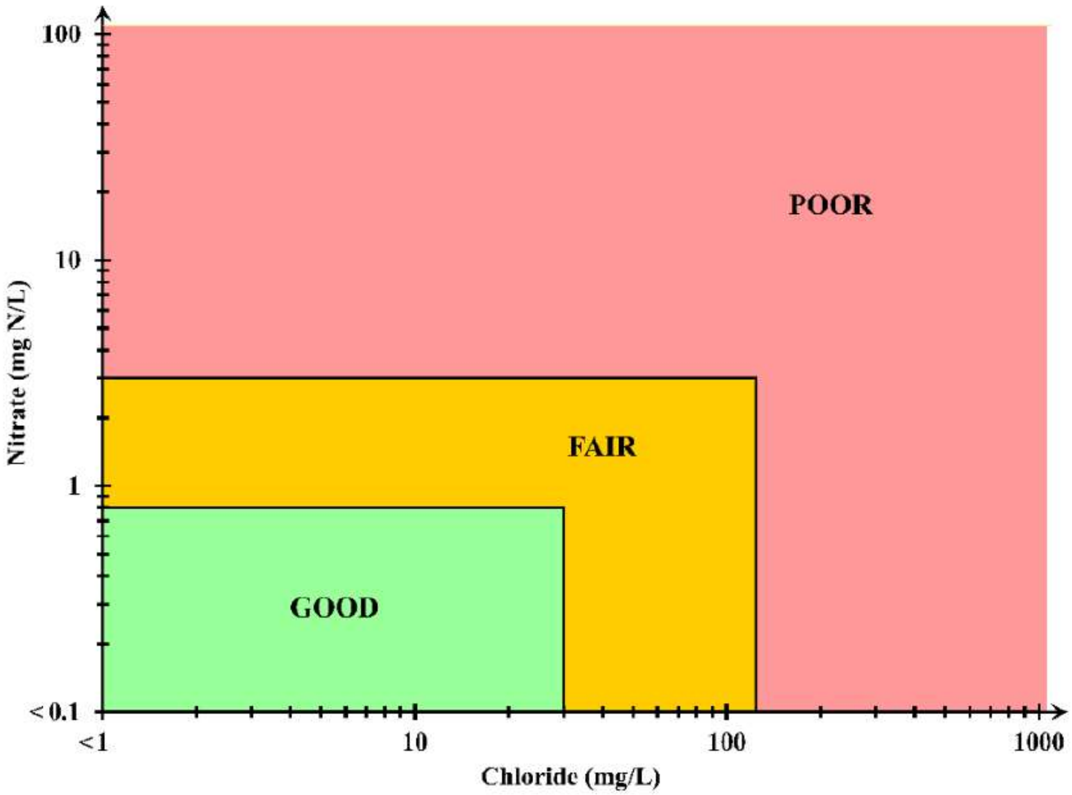
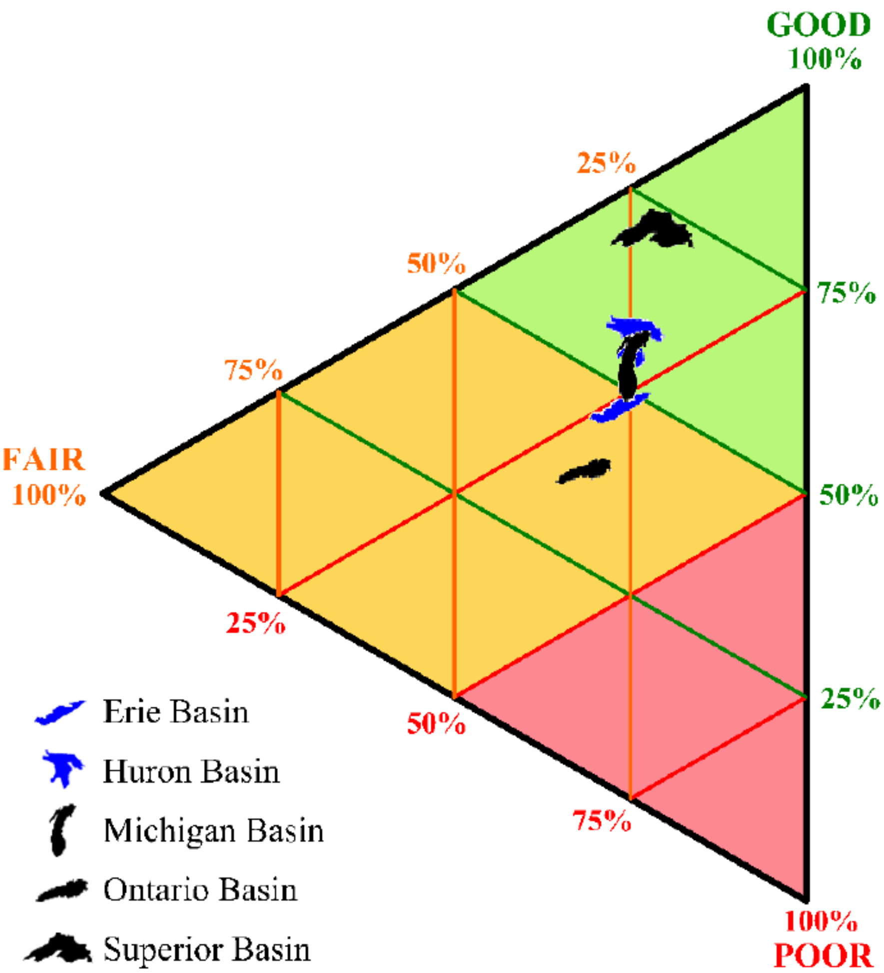


Figure 7. Criterion diagram for groundwater quality indicator status assessment. Source: Canadian Council of Ministers of the Environment.



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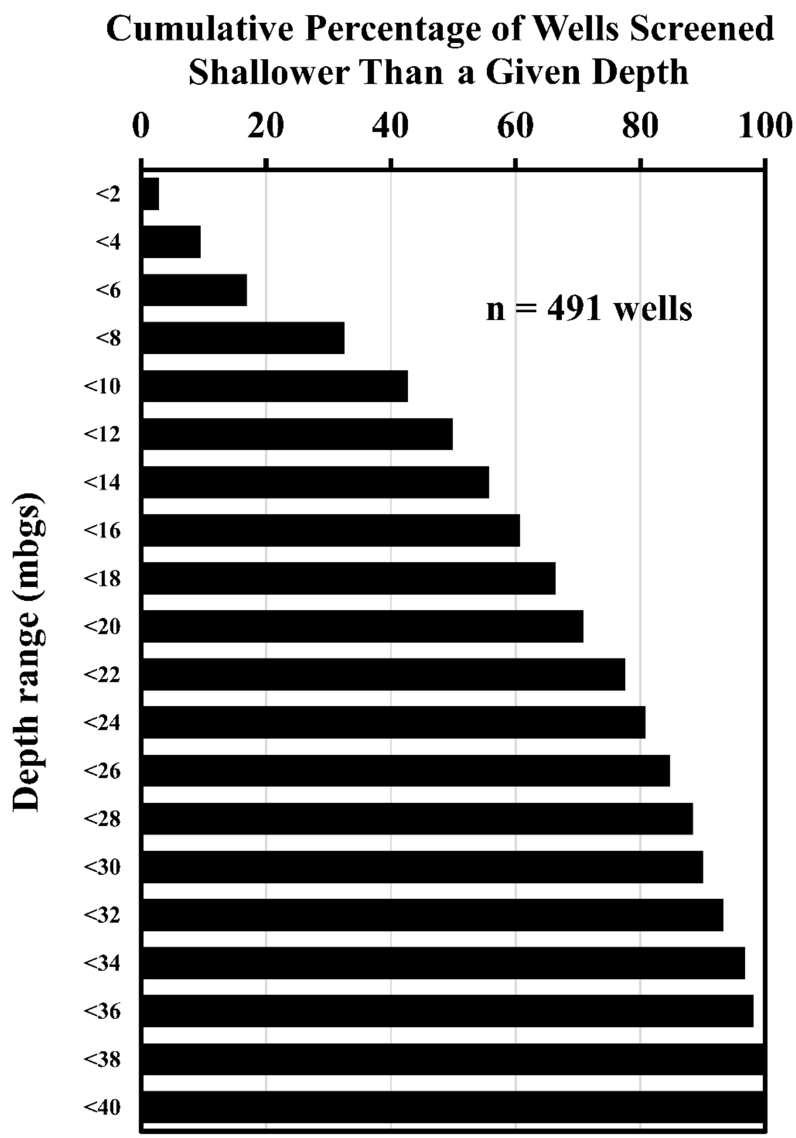

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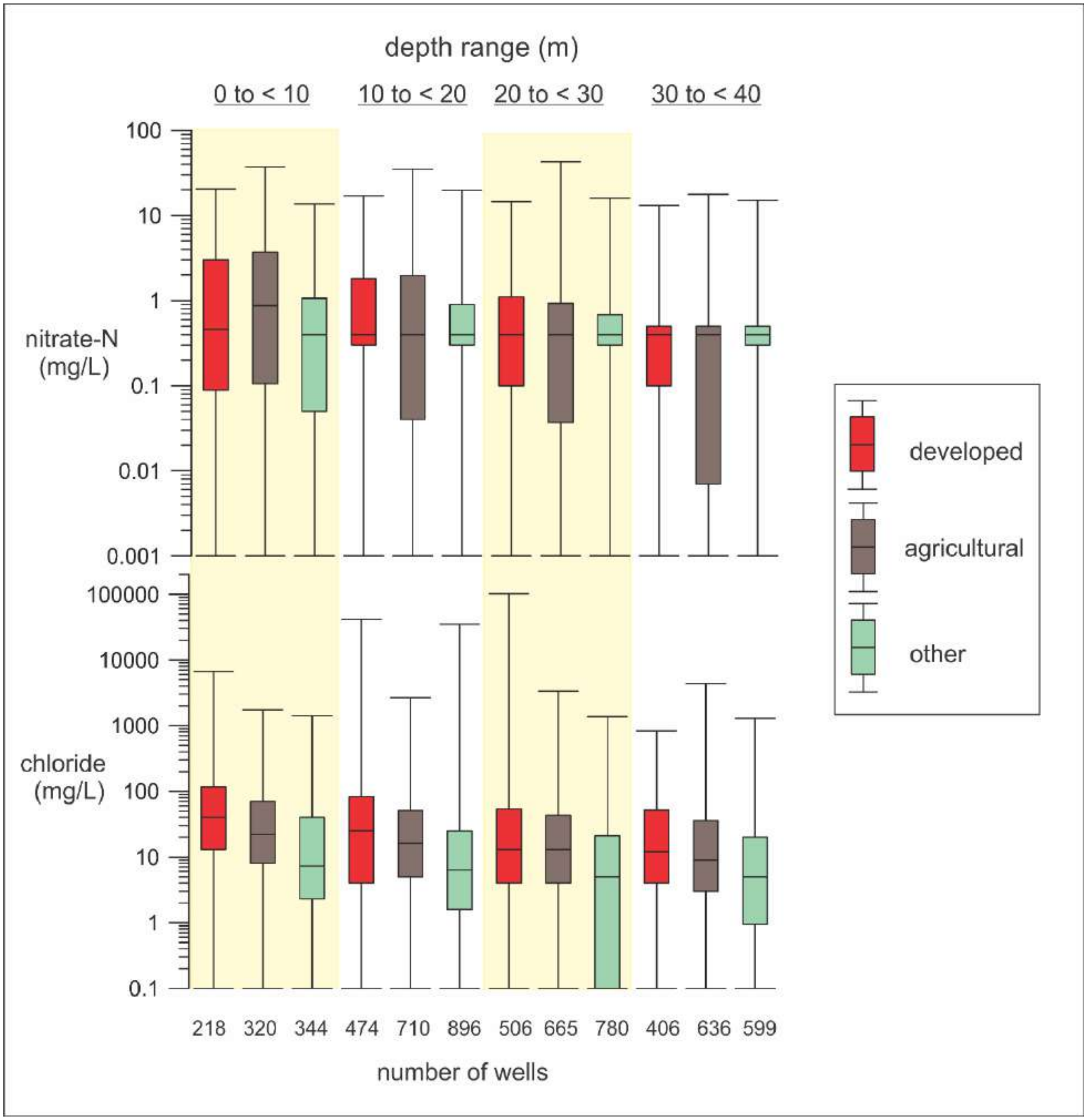

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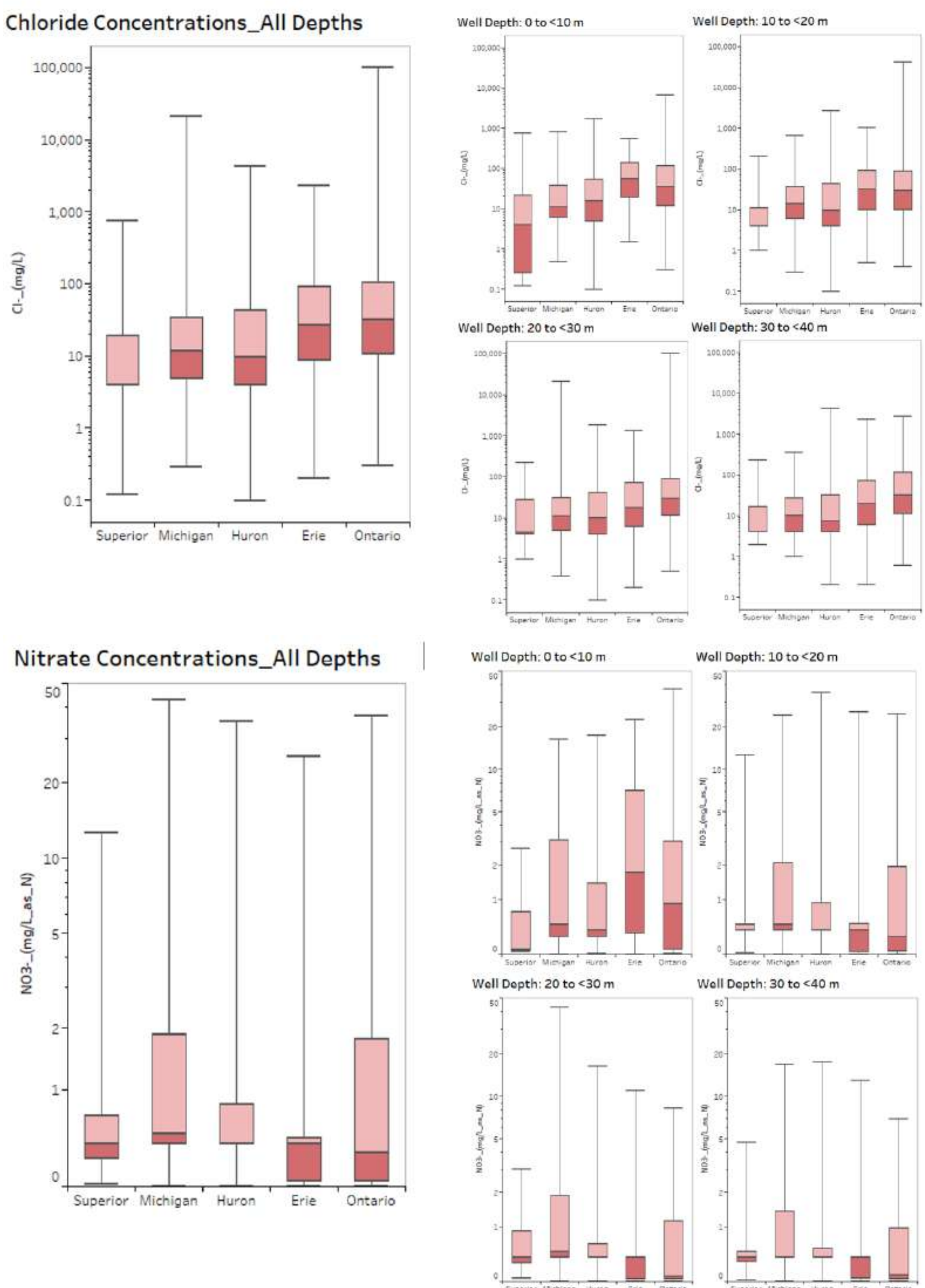

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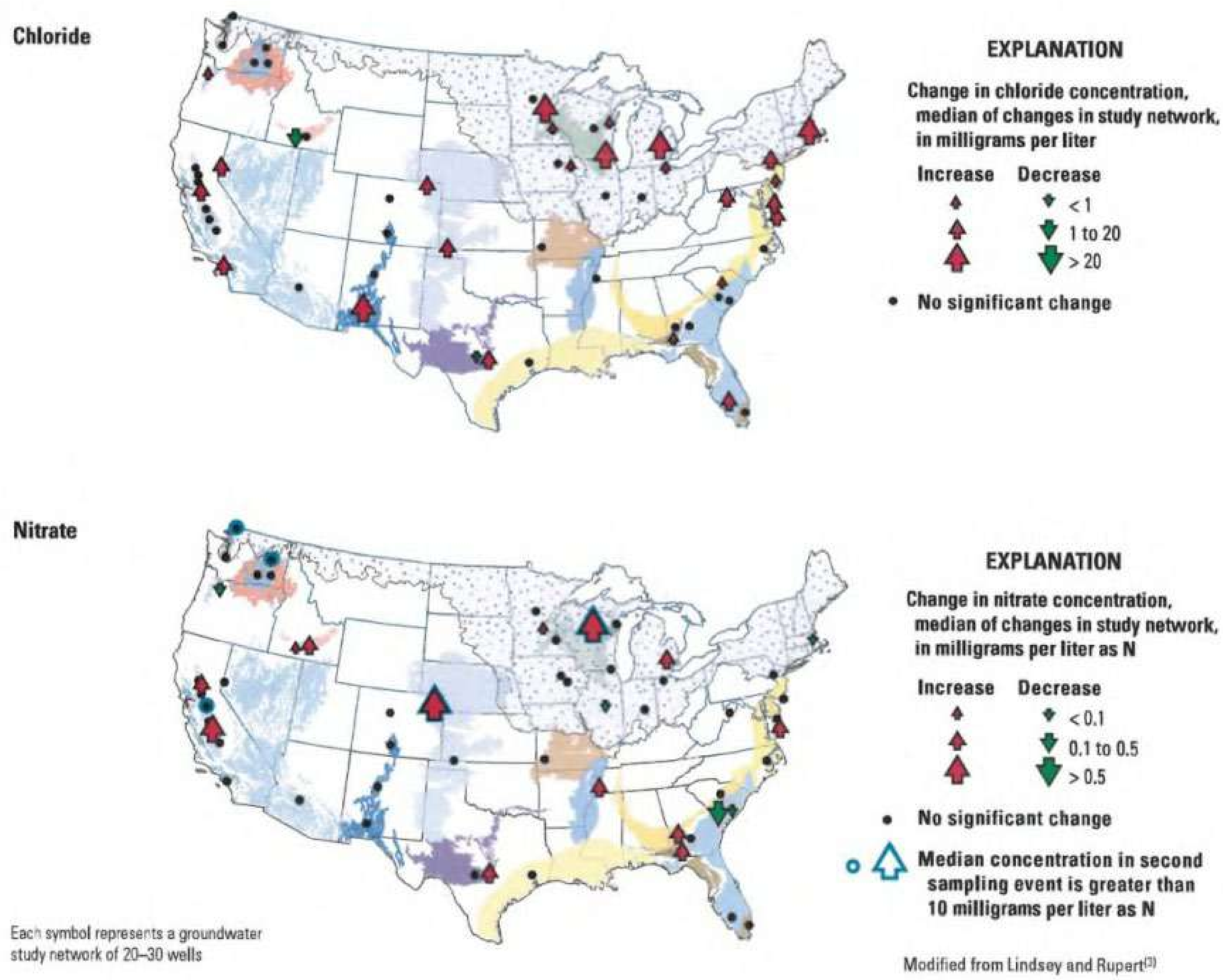

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Sub-Indicator: Forest Cover Riparian Zone

Overall Assessment

Status: Fair

Trends:

10-Year Trend (2006-2016 for U.S. and 2008-2018 for Canada): Unchanging

Long-term Trend: Undetermined

Rationale: Forested cover in the riparian zone of water bodies is high in the Lake Superior basin (96%), moderate in the Michigan, Huron, and Ontario basins (65 – 78%) and low in the Lake Erie basin (35%) based on satellite imagery. Trends in forested cover (2006 – 2016 in U.S. and 2008 – 2018 in Canada) in riparian zone are showing a small increase in the Lake Superior basin (+1.1%) and unchanging conditions in Michigan, Huron, Ontario, and Erie basins. The northern watersheds have much higher rates of forested riparian zones than watersheds in the south, where there is much greater development and agriculture.

Similarly, forested lands are a large percentage of total land area within the Lake Superior (85%) and Huron (62%) basins, a moderate percentage in the Lake Michigan (54%) and Ontario (48%) basins and low percentage in the Lake Erie basin (20%) based on satellite imagery. Trends in forest cover across the lake basins are very similar to the riparian zone assessments, showing small increases in the Lake Superior basin (1.2%) and unchanging conditions in the Michigan, Huron, Ontario, and Erie basins.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend (2006-2016 for U.S. and 2008-2018 for Canada): Improving

Long-term Trend: Undetermined

Rationale: Riparian zones in the Lake Superior basin have high forest cover overall (96%) and have been increasing on the U.S. side of this basin (+3.5%). The Lake Superior basin also has a high overall forest cover. These data suggest that there is unlikely to be long-term impairment of water quality due to forest cover change.

Lake Michigan

Status: Fair

10-Year Trend (2006-2016 for U.S. and 2008-2018 for Canada): Unchanging

Long-term Trend: Undetermined

Rationale: Riparian zones in the Lake Michigan basin have moderate forest cover overall (66%). Northern watersheds within this basin have high forest cover in riparian zones, while southern watersheds have reduced cover that may decrease water quality and ecosystem integrity. There is a similar pattern for forest cover in this basin, with high forest cover in the northern watersheds, while southern watersheds have low forest cover. These data suggest there is some potential in southerly watersheds to have impairments in water quality and ecosystem integrity due to forest cover change.

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend (2006-2016 for U.S. and 2008-2018 for Canada): Unchanging

Long-term Trend: Undetermined

Rationale: Riparian zones in the Lake Huron basin have moderate forest cover overall (78%). Northern watersheds within this basin have high forest cover in riparian zones, while southern watersheds have reduced cover that may decrease water quality and ecosystem integrity. There is a similar pattern for forest cover in this basin, with high forest cover in the northern watersheds, while southern watersheds have low forest cover. These data suggest there is some potential in southern watersheds to have impairments in water quality and ecosystem integrity due to forest cover change.

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend (2006-2016 for U.S. and 2008-2018 for Canada): Unchanging

Long-term Trend: Undetermined

Rationale: Riparian zones in the Lake Erie basin have low forest cover overall (35%). The trend (between 2006/08 and 2016/18) are unchanging on both the U.S. and Canada sides of the basin. This basin also has low forest cover, which has also remained unchanged on both sides of the basin. These data suggest that there is a large potential for water quality problems and risks to ecological integrity due to forest cover change.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Fair

10-Year Trend (2006-2016 for U.S. and 2008-2018 for Canada): Unchanging

Long-term Trend: Undetermined

Rationale: Riparian zones in the Lake Ontario basin have moderate forest cover overall (65%). Similarly, most watersheds in the Lake Ontario basin have moderate forest cover, which has remained unchanged over the 2006/08 to 2016/18 period on both sides of the basin. These data suggest there is a potential for water quality problems and risks to ecological integrity due to forest cover.

Status Assessment Definitions

Implications for water quality and quantity are difficult to establish, but the data provide insight on general trends in forest sustainability. Healthy, vigorous forests are crucial to basin ecosystem health. Interpreting the data with respect to forest health, however, will require additional assistance from forestry experts and stakeholders.

Suggested status for riparian zones:

Good: >80% forest cover in riparian zones

Fair: 50 – 80% forest cover in riparian zones

Poor: <50% forest cover in riparian zones

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

Trend Assessment Definitions

Forest cover within riparian zones

Improving: increasing % of forest cover within riparian zones > 1%.

Unchanging: no change in the % of forest cover within riparian zones ≤ ± 1%

Deteriorating: decreasing % of forest cover within riparian zones and changing if < - 1%

Undetermined: Data are not available or re insufficient to assess condition of the ecosystem components

Endpoints and/or Targets

Endpoints are not yet established. Establishing endpoints requires consensus on desired forest cover patterns.

Sub-Indicator Purpose

The purpose of this sub-indicator is to quantify forest cover in riparian zones in relation to its role in performing hydrologic functions, providing essential processes (e.g., evapotranspiration and nutrient transport), and protecting the physical integrity of the watershed (e.g., erosion control), all of which are necessary for supplying high quality water.

Ecosystem Objective

To have a forest composition and structure that reflects natural ecological diversity (i.e., under present climate conditions) of the region.

This sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from other substances, materials, or conditions that may negatively impact the chemical, physical, or biological integrity of the Waters of the Great Lakes."

Measure

This sub-indicator will measure, using remote sensing, the percent of forest cover within riparian zones of water-sheds by lake basin, over time.

For the purposes of this sub-indicator, the riparian zone is defined as a 30 metre buffer around all water bodies that has forest cover. Water bodies include water polygons, 2-sided rivers, arc-based streams and intermittent streams available where identified.

For the purposes of this sub-indicator, forest cover is defined as forested land or land regenerating to forested land that was disturbed as a result of forest harvest operations or natural disturbance (fire, insect, or blow down). Cover types were identified from Landsat satellite imagery. Forest cover includes areas classified as forests as well as treed/woody wetlands. With the higher resolution data now available on the Canadian side of the basin a decision was made to include shrubby lands (brush/alder, shrub/scrub) under forest for both jurisdictions given the functionally similar role to forest in riparian areas (e.g., canopy cover and erosion prevention). This change resulted in an increase in the amount of forest area. Efforts are made for the classified remote sensing data to be done in a consistent and comparable methodology between jurisdictions and across time intervals.

Ecological Condition

This sub-indicator includes the percent of forested lands within riparian zones by watershed, over time as the main component being assessed. The percent of forested lands within watershed by lake basin, over time is also included to support and provide context for the lake-by-lake and overall assessments.

Decades of research and monitoring have shown that water draining forested watersheds is of high quality, as measured by sediment yields, nutrient loadings, contaminant concentrations and temperatures. Increased forest coverage within a riparian zone decreases the amount of runoff and erosion (nutrient loadings, non-point source pollution and sedimentation) and increases the capacity of the ecosystem to store water. Riparian zones can also regulate and helping to maintain water temperatures. Forest cover also contribute to many other ecosystem services, including controlling soil erosion, increasing groundwater infiltration, stabilizing shorelines and mitigating storm run-off. Leaf litter and woody debris provide critical food and habitat for fish and other aquatic wildlife. Although there are different roles of non-forest vegetation in maintaining water quality and quantity, forest cover in riparian areas is a good representation of water protection.

In general, an increase in forest cover improves water quality. Ernst (2004), in a small survey of municipal water systems, showed that water treatment costs can be directly related to the degree of forest cover in the source watershed. The function she developed suggests that treatment costs are lowest at levels of forest cover above ~60%. Other studies have been less successful in discovering empirical relationships between forest cover and the economics of municipal water supplies.

Where watersheds have experienced large land-use changes due to agricultural activities or urban and suburban development, increased forest coverage within a riparian zone can mitigate many of the potentially harmful impacts on water bodies. Forested riparian zones can decrease the amount of surface runoff to water bodies (reducing erosion), mitigate nutrient loadings from fertilizer application and other non-point source pollutants and increases the capacity of the ecosystem to store water. Riparian zones are also important sources of energy and material to aquatic systems and help regulate water temperatures. Thus the amount of forest in riparian zones (30 metre buffer around all water bodies which includes water polygons, rivers, streams and intermittent streams where identified) within each lake basin is the component being used to assess the conditions within this sub-indicator. The status assessment is determined using the following criteria: Good = >80% forest cover in riparian zones; Fair = 50 – 80% forest cover in riparian zones; and Poor = <50% forest cover in riparian zones. For trends, a trend is considered unchanging if change is ≤ ± 1% and changing if > ± 1%. Overall forest cover in a lake basin is used as additional information to provide a larger context.

The riparian zone was assessed by creating a 30 metre buffer around all waterbodies and using it as a mask on the forest cover data layers. On a lake basin level (Figure 1), the Lake Superior basin has 96% of its riparian zones identified as forested, with moderate level of forest in riparian areas for Michigan (66%), Huron (78%), and Ontario

(65%). Only 35% of riparian zones in the Lake Erie basin is forested (Table 1). There is also substantial variation at the tertiary watershed level with each of the lake basins (Figure 2). The northern watersheds have much higher rates of forested riparian zones than watersheds in the south, where there is much greater development and agriculture.

Assessing trends in the forest cover within the riparian zone sub-indicator has proven difficult. Whereas the status of forest cover can be readily assessed through analysis of carefully checked and referenced satellite data, these data are usually available for single points in time. For this report, satellite imagery data was employed for the U.S. portions of the lake basins from 2006 to 2016 and for the Canadian portions of the basins from 2008 and 2018. Trend analysis showed that riparian forest is improving for Lake Superior basin (1.1%) and unchanging for Michigan, Huron, Erie and Ontario. These trends should be interpreted with some caution, particularly on the Canadian side of the basins given the dramatic changes in imagery type, resolution and partitioning of land classes. A longer record (>20 years) with consistency in imagery and methods is required in order to identify trends with any degree of reliability.

Patterns in forest cover within watersheds show similar findings to the forest cover in riparian areas. Figure 3 shows the tertiary watersheds draining into the Great Lakes and their level of forest cover. There is a strong North-South gradient evident in the degree of forest cover as would be expected given a similar gradient in population and agricultural activity. In the Lake Superior basin, 85% of the land area is forested (Table 2). In all the other basins, forests have been replaced by development and agriculture, leaving forest to occupy 54% (Michigan), 62% (Huron), 20% (Erie) and 48% (Ontario) of the basins (Table 2). However, it must be noted that within any given basin, there are watersheds with fair to good forest cover (Figure 4). Table 2 shows that in the all basins are showing unchanging conditions in forests within all basins, with the exception of the Lake Superior basin where there is a small increase (1.2%) in forest.

Linkages

The well-documented ability of forested lands to produce high quality water and in particular for forested riparian areas to protect water resources has linkages to many other sub-indicators. In particular, forest cover within riparian areas contribute directly to reducing nutrient and other non-point source pollutants, sediment loadings to the tributaries and lakes, and help to mitigate negative impacts of atmospheric deposition. Indirectly, the high quality water emanating for forested areas supports diverse aquatic communities. Climate change, through its effects on forest composition and function and on local hydrological processes is likely to affect the ability of forests to produce high quality water, although the magnitude and direction of these affects are not well known. For example, a decline in total annual runoff due to increased air temperatures and/or drier conditions in many Great Lakes basins may lead to increased concentrations of nutrients and contaminants in tributary waters. The timing of nutrient and contaminant release may also change if drier conditions lead to their retention within soils until flushed in episodic storm events. Also, changes in forest composition, due human activities (e.g., forest management) or natural vectors (e.g., emerald ash borer, an invasive species), may affect water quality and/or quantity.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found below		

Data URL

Canada (Ontario)

Geohub watershed layer (recently updated, extracted subset attached)
https://geohub.lio.gov.on.ca/datasets/mnrf::great-lakes-st-lawrence-basin-watersheds-glbw

Geohub rivers/streams layer or Ontario Hydro Network – Hydrographic Line –
https://geohub.lio.gov.on.ca/datasets/mnrf::ontario-hydro-network-ohn-hydrographic-line

Geohub lakes layer or Ontario Hydro Network – Waterbody – https://geohub.lio.gov.on.ca/datasets/mnrf::ontario-hydro-network-ohn-waterbody

Forest Resources Inventory Products – not currently available online part of management planning dataset - https://www.ontario.ca/page/forest-resources-inventory

Geohub Southern Ontario Land Resource Information System (SOLRIS) v2 (2009-11 data) -
https://geohub.lio.gov.on.ca/datasets/southern-ontario-land-resource-information-system-solris-2-0

Geohub Southern Ontario Land Resource Information System (SOLRIS) v3 (2011-15) –
https://geohub.lio.gov.on.ca/datasets/southern-ontario-land-resource-information-system-solris-3-0

Geohub provincial land cover 2002 (with 2008 updates) – https://geohub.lio.gov.on.ca/datasets/provincial-land-cover

United States

National Land Cover Dataset 2006 (NLCD 2006; [Fry et al. 2011]) and 2016 (NLCD 2016; [Yang et al 2018]), https://www.mrlc.gov/data?f%5B0%5D=category%3Aland%20cover&f%5B1%5D=year%3A2016

National Hydrography Dataset (NHD) (waterbodies, areas and flowlines) - shapefiles at: http://prd-tnm.s3-website-us-west-2.amazonaws.com/?prefix=StagedProducts/Hydrography/NHD/HU8/HighResolution/Shape

or as a geodatabase at: http://prd-tnm.s3-website-us-west-2.amazonaws.com/?prefix=StagedProducts/Hydrography/NHD/HU8/HighResolution/GDB/

Data Limitations

- Lack of consensus on the desired percentage of forested land in the Great Lakes Basin, for each lake basin or riparian zones (and the desired size of the riparian zone itself) makes it difficult to determine the specific implications.
- No historical 'range of variation' of forest cover data is available to establish trends to help assess potential changes to ecosystem function and diversity. E.g. what was the natural and historical context for the basins? Do some function well at 50% cover? Did some only ever have 40% cover? However, for the purposes of this sub-indicator, the suggested thresholds will be used based on best professional judgement.
- Data may not be available for all private lands in the Great Lakes Basin of Canada so a complete assessment of forests may not be available.

Additional Information

This sub-indicator is based upon the Montreal Process Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests, Third Edition, December 2007 specifically Criterion 4 (Conservation and Maintenance of Soil and Water Resources), Element 4.1 (Protective Function), Indicator 4.1.a.

Authors will consider rolling up the two measures reported into one overall assessment in the future. One option may be to use the ratio of Riparian: Total Forest Cover. Values closer to 1 would indicate forested area is preferential to riparian areas whereas low values would indicate that forested areas are not in riparian areas.

U.S. data are available from U.S.D.A. Forest Service, Forest Inventory and Analysis Database (USFS, FIADB; Burrill et al. 2018). Raw data available online at: https://www.fia.fs.fed.us/tools-data/default.asp. These are statewide data sets and therefore require geo-processing using GIS software to extract data relevant to the Great Lakes basin only. There are 7 integrated epochs of US land cover products between 2001 and 2016 (i.e., 2001, 2003, 2006, 2008, 2011, 2013, and 2016), with 2006 and 2016 used in this assessment. The Ontario Ministry of Natural Resources and Forestry (OMNRF) is the sole jurisdiction on the Canadian side of the Great Lakes with relevant terrestrial data. Summary inventory processing and geo-processing for basin-specific information is also required for Ontario data. Ontario forest resource inventory data is only available for forest management units of Central and Northern Ontario and must be combined with alternate remote sensing data interpreted for southern Ontario. Harvest data for southern Ontario are not collected as private land is not within the jurisdiction of the Ontario Ministry of Natural Resources and Forestry. The landcover data used for the 2008 assessment is based on 2002 imagery with 2004-2008 updates based on actual harvest, disturbance and renewal layers at 30 m resolution. The landcover data used for the 2018 assessment is a new and more detailed imagery at resolution of 5 m (northern Ontario) and 15 m (southern Ontario). The new data at higher resolution provides better differentiation in smaller features and moving forward will allow for better future assessment of change. For 2018 assessment, of the 24.4 million hectares assessed; 16.9 million ha compare landcover 2008 to 2018 FRI, 700 000 ha not updated, based on landcover 2008 (primarily Manitoulin Island) and 6.8 million ha updated based on 2009 vs. 2015 SOLRIS data. Also, two additional watersheds were added north of Lake Nipigon in Lake Superior Basin.

Estimating forest cover by remote sensing is widely used and generally reliable. However, many of the available datasets do not contain the long time series needed to adequately assess trends. Regular assembly of cross-border data sets are needed to measure changes in forest cover and to understand the drivers of change. Forest inventory data (e.g., USFS Forest Inventory and Analysis database) are also useful but Canada lacks an equivalent system. There also remains the challenge of integrating both forest inventory systems and remote sensing data across jurisdictions due to differences in goals and methodologies.

It is acknowledged that forest type and the age structure and composition of forests as a function of types and intensity of disturbance influence water quality and quantity. Although it may be desirable to expand the analysis to include these factors, devising a way to consistently compile and calculate indicators given the different sources of data remains a challenge. It is also recognized that a standard 30 m buffer may not be sufficient to protect water bodies and assessing different or variable buffer sizes might be more beneficial.

The following potential measures for future reporting could include:

- Forest roads and water crossings
- Area of disturbance (natural and anthropogenic disturbance)
- Forest age structure and composition
- Variable buffer width

Acknowledgments

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Information Sources

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Table 1. Percent of forest cover in riparian zones and percent change by basin for U.S. (2006 and 2016) and Canada (2008 and 2018) and combined U.S. and Canada Great Lakes region. Data was based on summing forest cover types in a 30 m buffer around all water bodies. Forest cover was identified from Landsat satellite imagery for U.S. and Canada (Ontario).

Sources: U.S. National Land Cover Database 2006 (Fry et al. 2006), 2016 (Yang et al. 2018) and Ontario Landcover 2002 (2004-2008 updates), FRI 2008, SOLRIS 2009 (OMNRF 2015, Forest Sustainability and Information Section, unpublished data) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

Table 2. Percentage of forest cover and percent change by lake basin for U.S. (2006 and 2016) and Canada (2008 and 2018) and combined U.S. and Canada Great Lakes region. Forest cover was identified from Landsat satellite imagery for U.S. and Canada (Ontario).

Sources: U.S. National Land Cover Database 2006 (Fry et al. 2006), 2016 (Yang et al. 2018) and and Ontario Landcover 2002 (2004-2008 updates), FRI 2008, SOLRIS 2009 (OMNRF 2015, Forest Sustainability and Information Section, unpublished data) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

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Figure 1. Percentage of forest cover within riparian zone (30 m buffer around water bodies) for tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed), treed/woody wetlands and shrub.

Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

Figure 2. Forest cover within riparian zone (30 m buffer around water bodies) rating for tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed), treed/woody wetlands and shrub.

Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

Figure 3. Percentage of forest cover in tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed), treed/woody wetlands and shrub.

Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

Figure 4. Forest cover rating in tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes.

Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

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U.S. Basin	Riparian Forest 2006 (ha)	Riparian Forest 2016 (ha)	% Change in Riparian Forest	Amount of Riparian Forest 2016	Class Value
Superior	175,791	181,864	3.5%	87.8%	Good
Michigan	297,540	299,914	0.8%	65.8%	Fair
Huron	103,744	104,794	1.0%	61.3%	Fair
Erie	107,862	107,569	-0.3%	38.4%	Poor
Ontario	112,925	113,651	0.6%	65.1%	Fair
Total:	797,862	807,792	1.2%	62.7%	Fair

Canada Basin	Riparian Forest 2008 (ha)	Riparian Forest 2018 (ha)	% Change in Riparian Forest	Amount of Riparian Forest 2018	Class Value
Superior	768,993	773,493	0.6%	98.6%	Good
Michigan					
Huron	720,239	726,571	0.9%	81.7%	Good
Erie	60,430	60,240	-0.3%	30.6%	Poor
Ontario	206,613	208,533	0.9%	65.3%	Fair
Total:	1,756,275	1,768,837	0.7%	80.8%	Fair

Great Lake Basin	Riparian Forest 2006/08 (ha)	Riparian Forest 2016/18 (ha)	% Change in Riparian Forest	Amount of Riparian Forest 2016/18	Class Value
Superior	944,784	955,357	1.1%	96.3%	Good
Michigan	297,540	299,914	0.8%	65.8%	Fair
Huron	823,983	831,365	0.9%	78.4%	Fair
Erie	168,292	167,809	-0.3%	35.2%	Poor
Ontario	319,538	322,184	0.8%	65.2%	Fair
Total:	2,554,137	2,576,629	0.9%	74.1%	Fair

Table 2. Percentage of forest cover and percent change by lake basin for U.S. (2006 and 2016) and Canada (2008 and 2018) and combined U.S. and Canada Great Lakes region. Forest cover was identified from Landsat satellite imagery for U.S. and Canada (Ontario). Sources: U.S. National Land Cover Database 2006 (Fry et al. 2006), 2016 (Yang et al. 2018) and Ontario Landcover 2002 (2004-2008 updates), FRI 2008, SOLRIS 2009 (OMNRF 2015, Forest Sustainability and Information Section, unpublished data) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

U.S. Basin	All Forest 2006 (ha)	All Forest 2016 (ha)	% Change in Forest	Amount of Forest 2016	Class Value
Superior	3,559,647	3,664,054	2.9%	90.9%	Good
Michigan	5,953,351	5,985,525	0.5%	54.0%	Fair
Huron	2,258,736	2,292,059	1.5%	56.7%	Fair
Erie	1,231,150	1,226,715	-0.4%	23.2%	Poor
Ontario	1,747,823	1,751,709	0.2%	52.9%	Fair
Total:	14,750,707	14,920,062	1.1%	53.7%	Fair

Canada Basin	All Forest 2008 (ha)	All Forest 2018 (ha)	% Change in Forest	Amount of Forest 2018	Class Value
Superior	8,347,140	8,387,781	0.5%	82.9%	Good
Michigan					
Huron	5,864,178	5,904,824	0.7%	64.7%	Good
Erie	326,759	326,020	-0.2%	13.8%	Poor
Ontario	1,204,526	1,213,688	0.8%	42.3%	Fair
Total:	15,742,602	15,832,314	0.6%	64.7%	Good

Great Lake Basin	All Forest 2006/08 (ha)	All Forest 2016/18 (ha)	% Change in Forest	Amount of Forest 2016/18	Class Value
Superior	11,906,787	12,051,835	1.2%	85.2%	Good
Michigan	5,953,351	5,985,525	0.5%	54.0%	Fair
Huron	8,122,914	8,196,883	0.9%	62.2%	Good
Erie	1,557,909	1,552,735	-0.3%	20.3%	Poor
Ontario	2,952,349	2,965,397	0.4%	48.0%	Fair
Total:	30,493,309	30,752,376	0.8%	58.9%	Fair

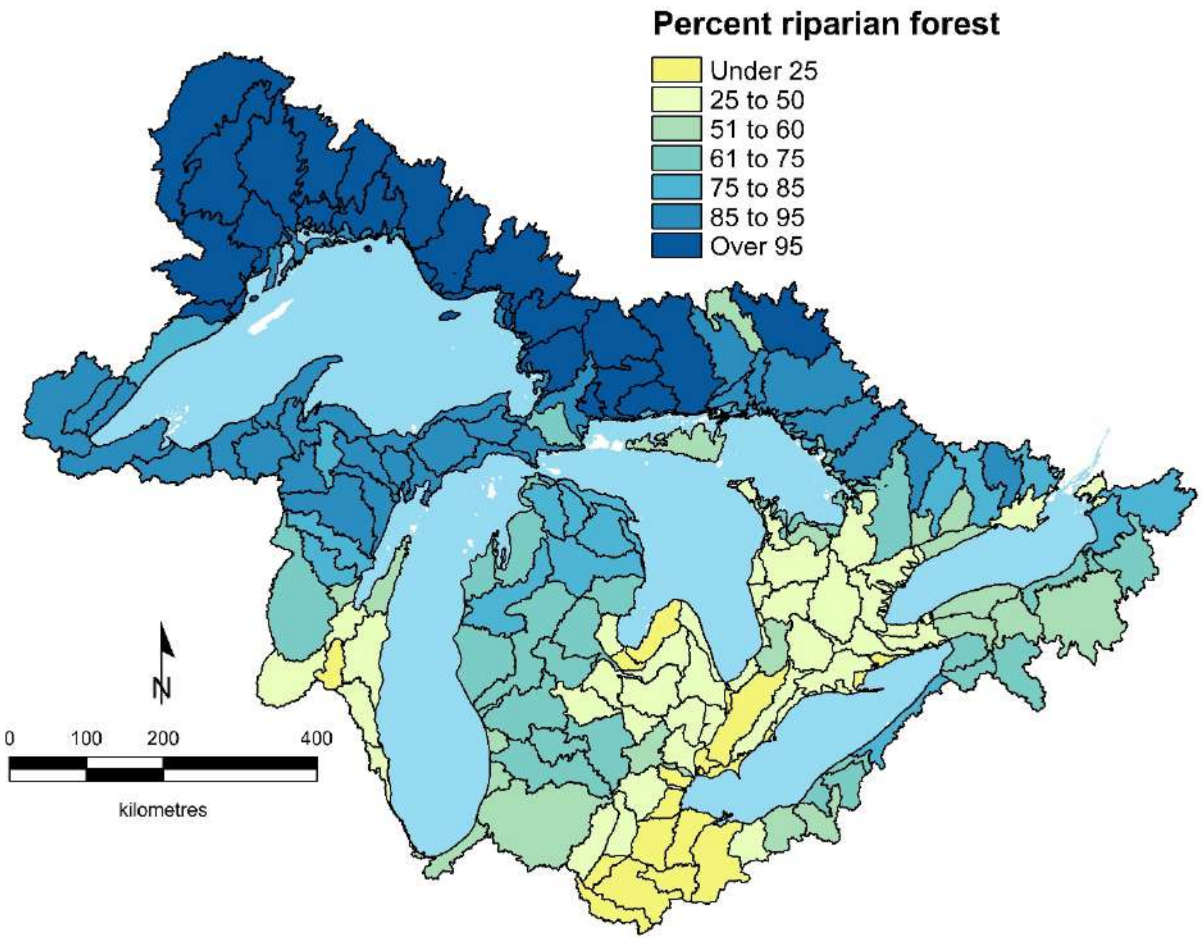

Figure 1. Percentage of forest cover within riparian zone (30 m buffer around water bodies) for tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed), treed/woody wetlands and shrub. Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

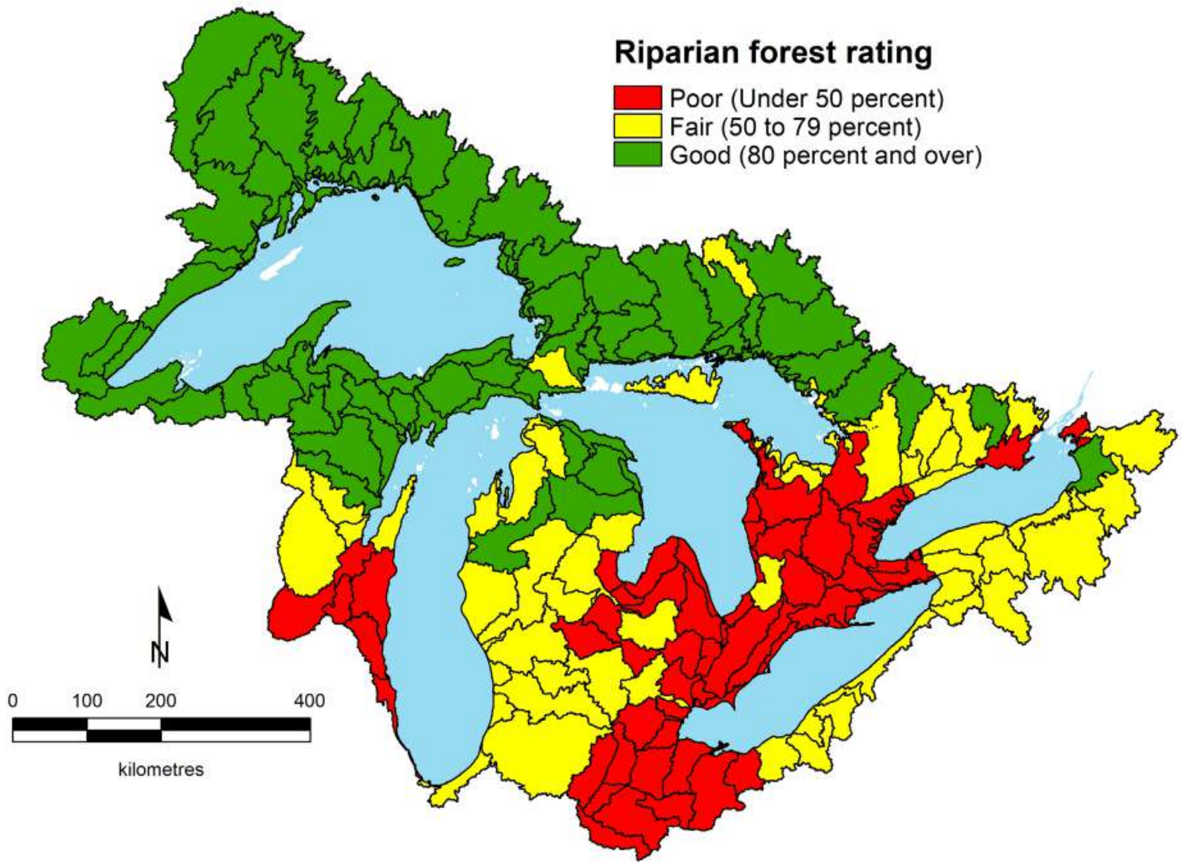

Figure 2. Forest cover within riparian zone (30 m buffer around water bodies) rating for tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed), treed/woody wetlands and shrub. Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

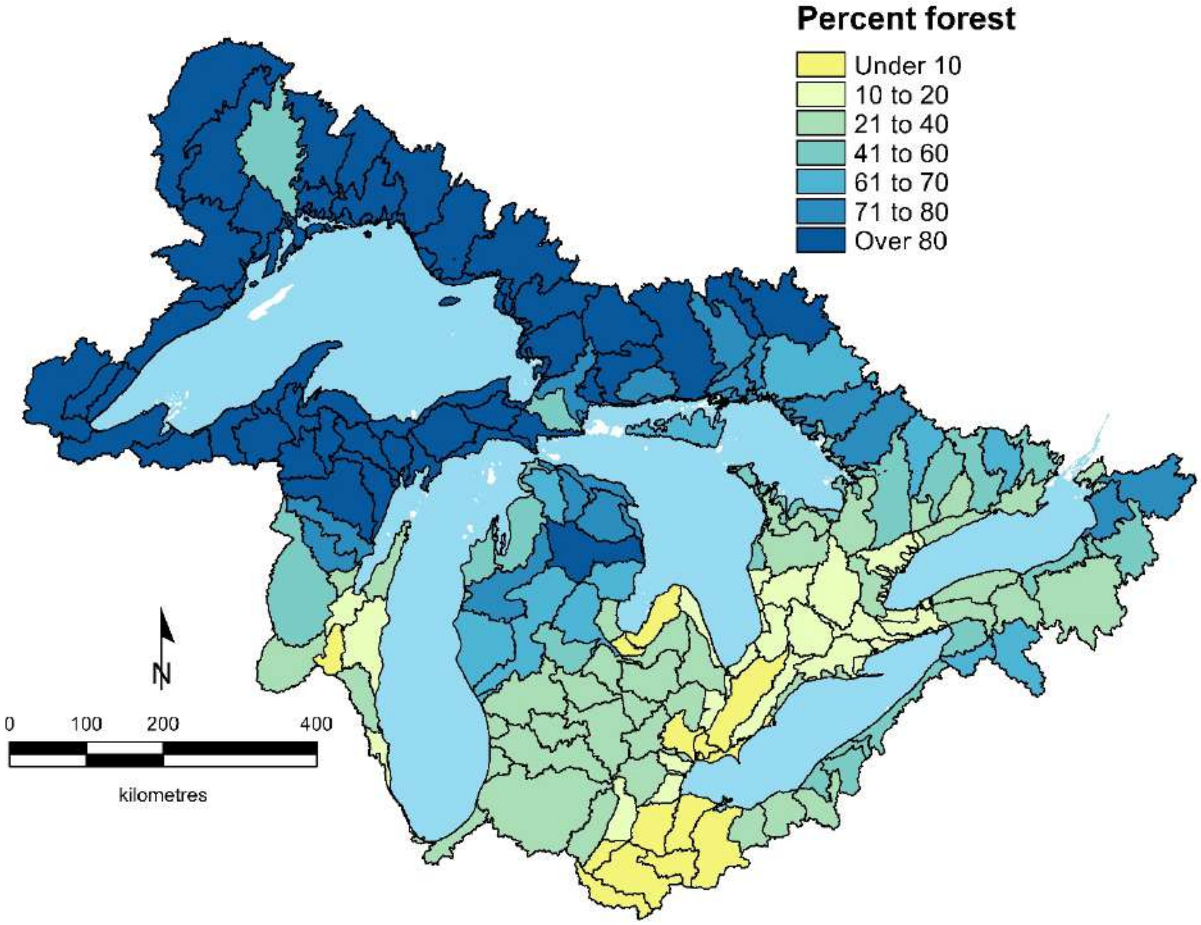

Figure 3. Percentage of forest cover in tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed), treed/woody wetlands and shrub. Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

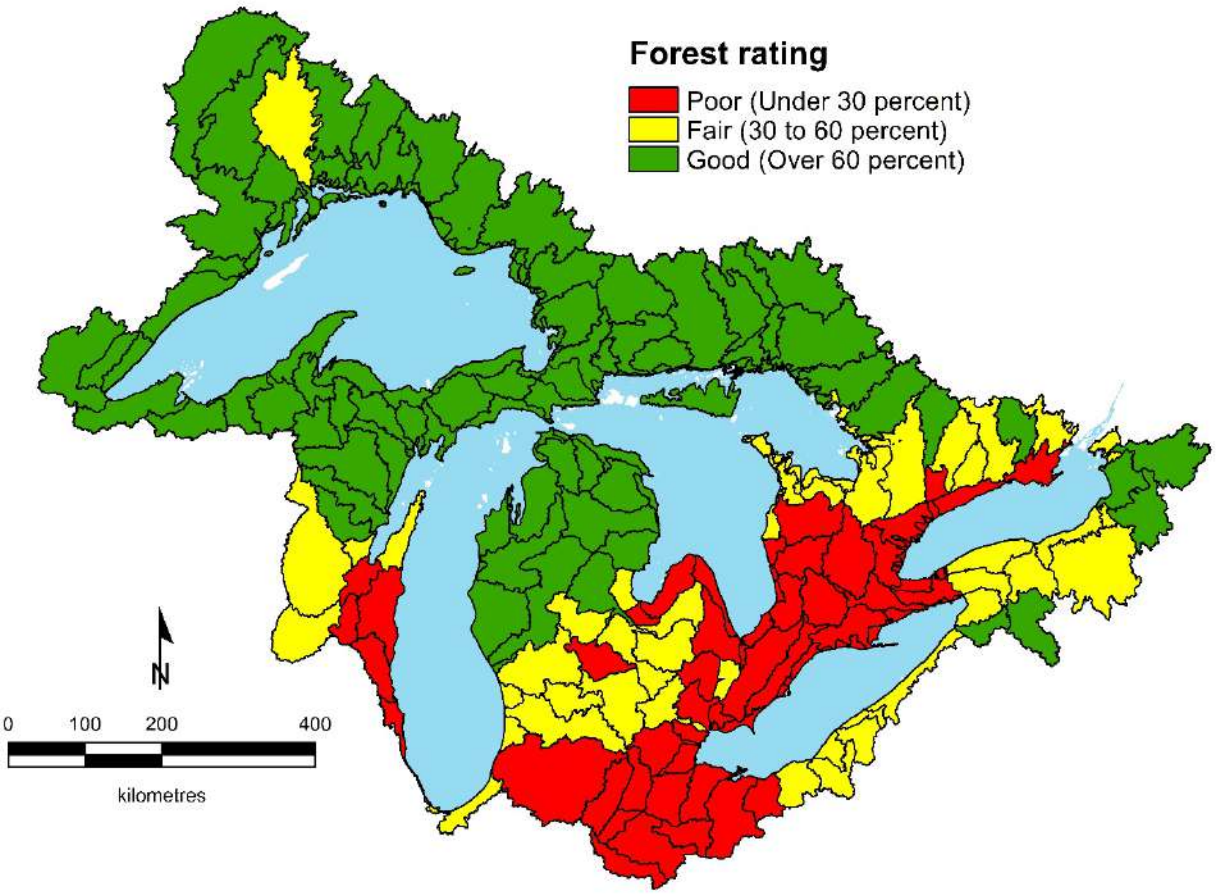

Figure 4. Forest cover rating in tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Source: U.S. National Land Cover Database NLCD 2016 (Yang et al. 2018) and FRI 2018 and SOLRIS 2015 (Forest Resources Inventories (OMNRF 2020, Forest Sustainability and Information Section, unpublished data).

Sub-Indicator: Land Cover

Overall Assessment

Status: Fair

Trends:

10-Year Trend: Not assessed

Long-term Trend (2000-2015): Unchanging

Rationale: Land cover in the Great Lakes basin was classified as approximately 8% developed, 26% agriculture, and 66% natural land cover. This indicates a medium risk of degraded water/habitat quality due to developed and agricultural land cover, and supports a status assessment of Fair (see Status Assessment Definitions and Table 3). From 2000 to 2015, there was an estimated net increase in developed land cover of 2,893 km^2 and estimated net decreases in forest land cover of 2,900 km^2 and in wetland land cover of 583 km^2. Changes in agriculture land cover included nearly offsetting effects of conversion of natural to agricultural land and conversion of agricultural to developed land. Despite these changes, the long-term trend is considered “unchanging” (see Trend Assessment Definitions and Table 14). A separate trend was not calculated for the 10-year period, as it is expected to be similar to the 15-year trend. An accuracy assessment completed for this analysis suggests that rates of land cover change are likely lower than reflected herein. Table 1 presents the land cover classes used in this assessment, Tables 2-3 and Figure 1 summarize information used in the status assessment, Tables 4-14 and Figure 2 summarize information used in the trend assessment, and Figures 3-8 present maps showing the distribution of land use/land cover basin-wide and for each lake watershed for 2015.

Lake-by-Lake Assessment

Lake Superior

Status: Good

10-Year Trend: Not assessed

Long-term Trend (2000-2015): Unchanging

Rationale: Land cover in the Lake Superior watershed in 2015 was classified as approximately 2% developed, 1% agriculture, and 97% natural land cover (Table 2 and Figure 1). This indicates a low risk of degraded water/habitat quality due to developed and agricultural land cover, and supports a status assessment of Good (see Status Assessment Definitions and Table 3). Approximately 76% of the land cover in the watershed is forest (Table 2 and Figure 1). From 2000 to 2015, there was an estimated net decrease in forest land cover of approximately 521 km^2. Developed land cover increased by an estimated 177 km^2, and agriculture land cover decreased by an estimated 14 km^2 (Table 4). The long-term trend is considered unchanging (Table 14 and Figure 2). An accuracy assessment completed for this analysis suggests that actual trends in land cover change during this period could be less significant (see Data Limitations).

Lake Michigan

Status: Fair

10-Year Trend: Not assessed

Long-term Trend (2000-2015): Unchanging

Rationale: Land cover in the Lake Michigan watershed in 2015 was classified as approximately 10% developed, 32% agriculture, and 58% natural land cover (Table 2 and Figure 1). This indicates a medium risk of degraded water/habitat quality due to developed and agricultural land cover, and supports a status assessment of Fair (see Status Assessment Definitions and Table 3). From 2000 to 2015, there was an estimated net decrease in forest land cover of approximately 353 km^2. Developed land cover increased by an estimated 528 km^2, and agriculture land cover decreased by an estimated 297 km^2 (Table 5). The long-term trend is considered unchanging (Table 14 and Figure 2). An accuracy assessment completed for this analysis suggests that actual trends in land cover change during this period could be less significant (see Data Limitations).

Lake Huron (including St. Marys River)

Status: Fair

10-Year Trend: Not assessed

Long-term Trend (2000-2015): Unchanging

Rationale: Land cover in the Lake Huron watershed in 2015 was classified as approximately 6% developed, 22% agriculture, and 72% natural land cover (Table 2 and Figure 1). This indicates a low/medium risk of degraded water/habitat quality due to developed land, a medium risk of degraded water/habitat quality due to agricultural land, and supports a status assessment of Fair (see Status Assessment Definitions and Table 3). Approximately 52% of the land cover in the watershed is forest. From 2000 to 2015, there was an estimated net decrease in forest land cover of approximately 1,064 km^2 and an estimated net decrease in wetland cover of 217 km^2. Developed land cover increased by an estimated 714 km^2. Agriculture land cover increased by an estimated 386 km^2, reflecting an estimated net conversion of natural to agriculture land cover of 627 km^2 and a net conversion of agriculture to developed land cover of 241 km^2 (Table 6). The long-term trend is considered unchanging (Table 14 and Figure 2). An accuracy assessment completed for this analysis suggests that actual trends in land cover change during this period could be less significant (see Data Limitations).

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend: Not assessed

Long-term Trend (2000-2015): Deteriorating

Rationale: Land cover in the Lake Erie watershed in 2015 was classified as approximately 18% developed, 61% agriculture, and 21% natural land cover (Table 2 and Figure 1). This indicates a medium risk of degraded water/habitat quality due to developed land, a high risk of degraded water/habitat quality due to agricultural land, and supports a status assessment of Poor (see Status Assessment Definitions and Table 3). From 2000 to 2015, there was an estimated net decrease in forest land cover of approximately 388 km^2 and an estimated net decrease in wetland cover of 179 km^2. Developed land cover increased by an estimated 762 km^2. Agriculture land cover decreased by an estimated 224 km^2, reflecting an estimated net conversion of natural to agriculture land cover of 284 km^2 and a net conversion of agriculture to developed land cover of 508 km^2 (Table 7). The long-term trend is considered deteriorating based on the rate of net increase in developed land cover and rate of net loss of forest and wetland to developed or agriculture land cover (Table 14 and Figure 2). An accuracy assessment completed for this analysis suggests that actual trends in land cover change during this period could be less significant (see Data Limitations).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Fair

10-Year Trend: Not assessed

Long-term Trend (2000-2015): Deteriorating

Rationale: Land cover in the watershed in 2015 was classified as approximately 12% developed, 34% agriculture, and 54% natural land cover (Table 2 and Figure 1). This indicates a medium risk of degraded water/habitat quality due to developed and agricultural land cover, and supports a status assessment of Fair (see Status Assessment Definitions and Table 3). From 2000 to 2015, there was an estimated net decrease in forest land cover of approximately 574 km^2 and an estimated net decrease in wetland cover of 184 km^2. Developed land cover increased by an estimated 712 km^2. Agriculture land cover increased by an estimated 34 km^2, reflecting an estimated net conversion of natural to agriculture land cover of 422 km^2 and a net conversion of agriculture to developed land cover of 388 km^2 (Table 8). The long-term trend is considered deteriorating based on the rate of net increase in developed land cover and rate of net loss of forest and wetland to developed or agriculture land cover (Table 14 and Figure 2). An accuracy assessment completed for this analysis suggests that actual trends in land cover change during this period could be less significant (see Data Limitations).

Status Assessment Definitions

Status assessment definitions are based on provisional thresholds representing degrees of risk of degradation of water/habitat quality of receiving waters. For the purposes of this sub-indicator, high risk of degradation corresponds to greater than 27% land cover classified as developed or greater than 50% land cover classified as agriculture. Moderate risk of degradation corresponds to between 6% and 27% (inclusive) land cover classified as developed or between 20% and 50% (inclusive) land cover classified as agriculture. Relatively low risk of degradation corresponds to less than 6% land cover classified as developed and less than 20% land cover classified as agriculture. See Ecological Condition and Additional Information sections for further discussion.

The status assessment is based on a combination of the risk levels from both developed and agriculture land cover.

Good: Risk of water/habitat quality degradation is low for both developed and agriculture land cover proportions (where less than 6% land cover is classified as developed and less than 20% land cover is classified as agriculture).

Fair: One or both land cover classes correspond to medium risk of water/habitat quality degradation, and neither land class corresponds to a high risk of water/habitat quality degradation (where between 6% and 27% land cover is classified as developed, or between 20% and 50% land cover is classified as agriculture).

Poor: One or both land cover classes correspond to high risk of water/habitat quality degradation (where more than 27% land cover is classified as developed or more than 50% land cover is classified as agriculture).

Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions

Trend assessment definitions were revised from previous reports to reflect the nature of land cover change as a potential stressor on the Great Lakes ecosystem. Previous trend assessments focused on absolute change in land cover area by class as a proportion of the watershed area (e.g., change in area of developed land cover as a proportion of total area of watershed). The revised approach measures trends based on relative change in area of each land cover class from a baseline (e.g., change in developed land cover from 2000 to 2015 as a proportion of developed land cover in 2000). The revised approach focuses on the rate of increase or decrease in a land cover class and is better aligned with the concept of a stressor.

The trend assessment reflects the understanding that net increases in developed and agriculture land cover and net losses in natural land cover (forest and wetland) represent an increased risk of degradation of water/habitat quality. The thresholds used in the trend assessment correspond to an average rate of net change (increase or decrease) of 1% by land cover class over a 10-year period, or an average of 0.1%/year. The long-term trend was characterized as "deteriorating" when the analysis estimated both an increase of ≥0.1%/year in developed and/or agriculture land cover and a decrease of ≥0.1%/year in forest and/or wetland land cover. For long-term trend assessment, a baseline year of 2000 was used, representing the earliest year for which reliable and consistent data across the U.S. and Canadian areas of the Great Lakes basin are available.

Land use change analysis is subject to significant uncertainty, and rates of actual land cover conversion among the classes considered in the trend assessment are likely lower than those based on measured change (see Data Limitations). Therefore, the effective threshold being applied in this sub-indicator is likely lower than 0.1%/year. However, this threshold is considered a reasonable indicator of an actual trend, particularly as a measure of trends in land cover conversion from natural or agricultural to developed land cover. Estimates of less than 10 km^2 change for any land cover class are not reported in the lake-by-lake summaries, as this is considered within the range of error for the change estimation process.

Improving: Net decrease in developed or agriculture land cover of ≥0.1%/year or net increase in forest or combined wetland (forest wetland and wetland) land cover of ≥0.1%/year and no class-specific trend assessment of "deteriorating."

Unchanging: No land cover class-specific trend assessment of "improving" or "deteriorating."

Deteriorating: Net increase in developed or agriculture land cover of ≥0.1%/year and net decrease in forest or combined wetland (forest wetland and wetland) land cover of ≥0.1%/year.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

In the absence of established targets that reflect relationships between land cover and ecosystem health at a watershed scale, provisional operational targets have been used for this sub-indicator. Provisional targets for developed and agricultural land cover were established in previous reports and were used for this assessment(see Status Assessment Definitions). Trend assessment thresholds reflect the target of stability and/or reduction in land cover types associated with increased stress on ecosystems and stability or increase in land cover types that protect or contribute to improved ecosystem health. The trend thresholds reflect a target of no net increase and/or net decrease in developed and agriculture land cover and no net loss and/or net increase in natural land cover.

Sub-Indicator Purpose

- Assess the status of natural land cover within the Great Lakes Basin
- Inform inferences about the major proximate causes of changes and trends in other biological communities, physical habitat, and water quality indicators that are more direct indicators of the health of the Great Lakes ecosystem

Ecosystem Objective

This sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should "be free from other substances, materials, or conditions that may negatively impact the chemical, physical, or biological integrity of the Waters of the Great Lakes."

Measure

This sub-indicator measures estimated areal extent (km^2) of different land cover types and change in the areal extent of land cover types over time. Land cover classification data, including change in land cover classification at different points in time, is analyzed at a 30x30-m resolution and reported at a lake-by-lake watershed scale. Two land cover datasets were used in the assessment of land cover status and change for this sub-indicator. The National Land Cover Database (NLCD) (Dewitz 2019) was used to evaluate land cover for the Great Lakes basin in the U.S., and land use data developed by Agriculture and Agri-Food Canada (AAFC) (AAFC 2021a) were used to evaluate land cover for the Great Lakes basin in Canada. The NLCD dataset extends back to 2001, and the most recent update reflects land cover conditions in 2016. AAFC updates land use maps every 5 years. The most recent product includes time series data from years 2000 to 2015 (AAFC 2021b).

Land cover classes were harmonized as described in Table 1 to be comparable on both Canada and U.S. sides of the basin. Land cover status is reported based on the most recent years included in both the AAFC and NLCD datasets: 2015 and 2016, respectively. Land cover trends are evaluated by comparing change in the harmonized land classification based on the 30x30-m pixels in each dataset between the years of 2000 and 2015 (AAFC) and 2001 and 2016 (NLCD). This change was analyzed for each dataset separately and the results were combined to calculate basin-wide and lake-specific statistics and trends.

NCLD data were collected from satellite imagery captured during the nominal year identified in the dataset and, to some extent, the preceding and succeeding years (USEPA 2021). Therefore, NCLD data for nominal years are considered to represent conditions for the nominal year +/- one year. Given this and rates of land use change at a watershed scale, status and trend data from the two sources are considered temporally comparable despite the nominal one-year offset. For simplicity, the nominal date for the land cover assessment is described as 2015 and the nominal start and end dates for the land cover trend assessment are described as 2000 and 2015, respectively. An accuracy assessment was performed to help interpret land cover change results. See Data Limitations for a more complete description of the methods used for this sub-indicator.

Ecological Condition

Changes in land cover at a watershed scale provides an indication of potential stressors on water quality and associated ecological conditions. When interpreted with other information (see Linkages section), land cover change can provide an indication of whether water quality and ecological stressors are increasing, remaining stable, or abating.

Research supports the understanding that increased developed land cover is associated with degradation of water quality and ecological condition due to increased impervious surface cover and likelihood of point source pollution (Bartsch et al. 2015, Cuffney et al. 2010, Morse et al. 2003, Paul and Meyer 2001, Thomas et al. 2018). Research also supports the understanding that increased agricultural land cover is associated with degraded water quality and ecological condition, though this is sensitive to regional conditions, crop types and management practices (Bosch et al. 2014, Michalak et al. 2013, Pearce and Yates 2020, Wang et al. 1997). Research suggests a positive relationship between natural land cover and water quality and ecological condition as a result of nutrient uptake into biomass and runoff storage and filtration (Pearce and Yates 2020, Price 2011). The influence of natural land cover on water quality and associated ecological conditions are greatest near shorelines and riparian zones.

Land cover in 2015 for the Great Lakes basin and each lake-specific watershed is shown in Figures 3-8. General variation in land cover and spatial distribution of land cover classes across the Great Lakes basin is similar to that observed in the previous report. The predominant land cover in the Lake Superior watershed is forest (Figure 4). The Lake Michigan and Lake Ontario watersheds include a relatively even distribution of agriculture and forest land cover, with significant pockets of developed land cover near urban centers (Figures 5 and 8). Land cover in the Lake Huron watershed varies geographically, with predominantly forest cover in the northern area of the watershed, agriculture land cover in the southeastern area, and a mix of agriculture, forest and developed land cover in the southwestern area (Figure 6). Land cover in the Lake Erie watershed is predominantly agriculture with significant developed land cover in urban areas (Figure 7).

The variation in land cover among and within the Great Lakes watersheds reflects a combination of factors, including variation in climatic and soil conditions, suitability for agricultural production, population growth and migration, and economic driving forces affecting land use (Pijanowski and Robinson, 2011). The extent and distribution of land cover across the Great Lakes basin suggests that Lake Erie and to a lesser extent Lakes Michigan, Huron, and Ontario, are most susceptible to water quality and ecological impacts associated with agricultural land use. All of the Great Lakes are susceptible, to different degrees, to water quality and ecological impacts associated with urban development.

Analysis of the harmonized spatial data across the entire Great Lakes basin showed an estimated net conversion of 2,775 km^2 from natural land cover to developed or agriculture land cover over the 15-year period from 2000 to 2015 (Table 9). The majority of this was associated with forest loss, with an estimated net conversion of 1,838 km^2 of forest land cover to developed or agriculture land cover (Table 10). An estimated net 613 km^2 of wetland land cover, including forest and other wetland land cover types, was converted to developed or agriculture land cover (Table 11). Estimated net change in agriculture land cover was relatively low on a basin-wide scale (115 km^2, or <1% of land area) (Table 13). However, this number obscures spatially distributed trends across the Great Lakes basin associated with conversion of natural to agriculture land cover in some areas and an offsetting conversion (on a watershed scale) of agriculture to developed land cover. The analysis also indicated net conversion of land cover among natural land cover types. While some of this change among natural land cover types likely reflects natural ecosystem progression or other factors, some may also be associated with uncertainty in the use of Landsat data for assessing change among natural land cover classes (AAFC 2015, Wickham et al. 2017).

On an absolute basis, the estimated net increase in developed land cover in the Lake Erie watershed was highest among the Great Lakes during the 15-year period (762 km^2). The net estimated increase in developed land cover was on a similar scale for Lake Huron and Lake Ontario (714 km^2 and 712 km^2, respectively). The average rate of increase in developed land cover exceeded 0.1%/year (1% over a 10-year period) in all lake watersheds (Tables 12 and 14). It is estimated that the Lake Huron watershed experienced a net increase in agriculture land cover of 386 km^2. The analysis estimated that net agriculture land cover decreased most significantly in the Lake Michigan and Lake Erie watersheds (297 km^2 and 224 km^2, respectively) (Table 13). On an absolute basis, the greatest estimated losses in forest and wetland land cover occurred in the Lake Huron watershed (1,064 km^2 and 216 km^2, respectively). The rate of conversion of forest to developed or agriculture land cover exceeded 0.1%/year in the Lake Erie and Lake Ontario watersheds (Tables 10 and 14). The rate of conversion of wetland to developed or agriculture land cover exceeded 0.1%/year in these same two watersheds (Tables 11 and 14).

Rates of land use change provide an indicator of risk of degradation to water quality and ecological conditions in the Great Lakes basin. Increased developed land cover in all Great Lakes watersheds indicates increased risk of degradation, and while sources of potential degradation (e.g., contaminated runoff) would be localized in urban areas, effects could be lake-wide. The extent and spatial distribution of conversion of natural land to agriculture land cover and the extent and spatial distribution of losses in forest and wetland land cover indicate potential stressors on the Great Lakes ecosystem. Overall, the trend assessment for the 2000 to 2015 time period suggests an increase in developed land at the expense of agricultural lands, forests, and wetlands.

Other Spatial Scales

Land cover is an indirect predictor of water quality and associated ecological condition. The effect of land cover on water quality and ecological condition is subject to a wide array of factors including, for example, spatial arrangement of land cover, terrestrial ecosystem conditions, and near-stream and instream abiotic factors (Bartsch et al., 2015, King et al. 2005, Pearce and Yates 2020, Wickham et al. 2014). Among these factors, consideration of spatial relationships among land cover, tributaries, and coastal areas could be used to supplement and help interpret watershed-scale land cover data. King et al. (2005) and Wickham et al. (2014) describe methods for accounting for spatial distribution of land cover that could be considered for future enhancements to the sub-indicator.

Linkages

Land cover change from natural land cover types to developed and agriculture land cover can result in increased stressors on the Great Lakes ecosystem. Potential implications of land cover change can be interpreted by considering the following other sub-indicators used to assess the state of the Great Lakes ecosystem (please refer to individual sub-indicator reports for more information):

- Coastal wetlands: extent and composition – land cover change in the vicinity of coastal wetlands can disrupt coastal and nearshore processes, flow and littoral circulatory patterns, ecosystem connectivity, and nearshore and coastal habitat structure
- Nutrients in lakes – watershed scale change in agriculture and developed land cover near coastal areas and tributaries provide indications of change in nutrient loading potential in tributaries and lakes
- Forest cover – the forest cover sub-indicator uses a similar approach with a more explicit focus on forest cover in riparian zones and linkages between forest land cover and stream water quality and habitat

- Human population – human population growth is one driver of developed land cover; assessing relationships between population and land cover change can provide insights into trends in spatial patterns of development and associated implications of the effects of population growth on water quality and habitat degradation

Climate change can affect and be affected by land use and land cover (USGCRP 2018). Changing climate conditions affect transition of land cover among natural land cover types in response to changing precipitation patterns, temperature conditions, solar radiation, and related impacts (Fei et al. 2017, Kulmatiski and Beard 2013, Nemani et al. 2003, Woodall et al. 2018). Changing climate conditions also affect agricultural production choices and irrigation water availability and socio-economic factors that affect agricultural land use, population demographics, and land development extent and patterns (Bowling et al. 2020, Lambin et al. 2001). In turn, large-scale changes in land cover, land use practices, and patterns of development contribute to feedback loops by affecting global circulation patterns, surface reflectivity, greenhouse gas emissions, and carbon sequestration (Pielke et al. 2011, Sleeter et al. 2018).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report		X		
Data used in assessment are openly available and accessible	Yes	NLCD: https://www.mrlc.gov/ AAFC: https://open.canada.ca/data/en/dataset/fa84a70f-03ad-4946-b0f8-a3b481dd5248		

Data Limitations

Land cover status and trends were analyzed using NLCD and AAFC land use and land cover datasets. NLCD datasets classify land cover and land use at a 30x30-m resolution for the entire conterminous United States. A consistent set of NLCD data are available for the years 2001, 2006, 2011 and 2016 (Jin et al. 2019). Land cover classes used in NLCD are at the approximate Anderson et al. (1976) Level II thematic detail (Homer et al. 2004). AAFC land use maps classify land use and land cover for all areas of Canada south of 60°N at a 30x30-m resolution for the years 1990, 2000, 2005, 2010, and 2015 (AAFC 2021b). Land use classes used in the AAFC dataset follow the protocol of the Intergovernmental Panel on Climate Change (IPCC) (IPCC 2003). Combined, the NLCD and AAFC land cover and land use datasets cover the entire area of the Great Lakes basin. Previous versions of the sub-indicator report relied on Southern Ontario Land Resource Information System (SOLRIS) data, which excluded northern portions of the Lake Superior, Lake Huron, and Lake Ontario watersheds.

Dataset integration required translation of land use/land cover classes from the two datasets into a single, harmonized set of land cover classes (Table 1). The resulting set of land cover classes generally aligns with classes used in previous reports with the exception that a new class, "forest wetland" was created to support analysis across the NLCD and AAFC datasets. NLCD defines "woody wetlands" as areas where forest or shrubland vegetation accounts for greater than 20 percent of vegetative cover and the soil or substrate is periodically saturated with or covered with water. AAFC defines "forest wetland" as wetland areas with forest canopy cover >40% (AAFC 2021b). To harmonize the land use classifications, pixels classified in NLCD as "woody wetland" were subdivided into "forest wetland" and "wetland" classes using NLCD tree canopy data, as described in Table 1. Tree canopy data are not available in the NLCD 2001 dataset. Pixels classified as "woody wetland" in 2001 were reclassified as "forest wetland" for this analysis unless the harmonized 2016 class for the pixel was "wetland" or "water," in which case the pixels were classified "wetland." AAFC pixels classified as "forest regenerating after fire <20 years" were reclassified as "forest wetland" if the pixel was previously or subsequently classified "forest wetland" in the time series.

Dataset harmonization required use of land cover classes comparable to Level I thematic detail versus the more detailed land cover classes (comparable to Level II) available in the individual datasets. For example, NLCD includes four sub-classes and AAFC includes six sub-classes corresponding to developed land cover (see Table 1). For this study, the single developed land cover class was used for simplicity. Different types of developed land cover can affect ecological condition differently. The use of harmonized Level I classes results in loss of information relative to use of Level II classes. This affects the descriptiveness but not the overall conclusions of the land cover status assessment as currently defined. This data limitation does not affect the overall trend assessment conclusions.

In general, the use of harmonized Level II land cover classes combined with other data would help improve land cover status and trend assessment. Use of harmonized Level II land cover classes would improve upon the current approach. However, even the Level II land use/land cover classes available in existing data products do not capture important differences in land quality. Urban design, agricultural practices, and forest management practices affect ecological outcomes but are not adequately captured in the necessarily simplified classes used in land use/land cover products created using remote sensing. Additional information (e.g., impervious surfaces, tree canopy cover) could be combined with land use/land cover class data to help address this data limitation.

Land cover data are updated at different intervals and at frequencies that do not correspond to the State of the Great Lakes three-year reporting cycle. This results in a lack of temporal concordance among different land cover products and confounds analyses of both land cover status and trends across the Great Lakes basin. NLCD land cover data for the year 2019 and AAFC land use data for the year 2020 are expected to be available for the next State of the Great Lakes report.

Developing any land cover and land use dataset and evaluating change in land cover using data collected and classified for different periods is subject to error. Accuracy assessment is used to characterize uncertainty in land cover analyses (Homer et al. 2020, IPCC 2003, Oloffson et al. 2014). Overall accuracy of NLCD 2001 and 2016 land cover products was estimated at 89.2% and 90.6%, respectively, for the aggregated (Level I) land cover classes (Wickham et al. 2021). Overall accuracy of AAFC 2010 land use data was estimated at 93% (AAFC 2015). Accuracy assessment of AAFC 2015 data has not been published, but consistent improvements in the accuracy of AAFC land use data products over time suggest that the accuracy of the 2015 data should be similar or improved. Higher levels of misclassification in data products with published accuracy assessments was generally associated with natural and agriculture land cover (AAFC 2015, Wickham et al. 2021).

Less than 2% of the area in the Great Lakes basin was mapped as grassland in the NLCD and AAFC datasets. Historically, grassland has been a difficult land cover class to map, and what is mapped as grassland is often a mix of developed and agriculture land use with low herbaceous vegetation (AAFC 2015, Gray et al. 2013, Wickham et al. 2017, Wickham et al. 2021). As a result, land cover change from grassland to agriculture and grassland to developed detected in the type of analysis used for this sub-indicator are subject to significant uncertainty. Therefore, land cover changes from and to the grassland land cover class were not used in the trend assessment.

In an accuracy assessment of land cover change based on NLCD data for the 2001-2016 change period, Wickham et al. (2021) estimated accuracies of forest land cover loss from 2011 to 2016 of ~75%, accuracies for developed land cover gain of 51% (user's accuracy) and 64% (producer's accuracy), accuracies for other natural land cover gain or loss between 27% and 80%, and accuracies for agricultural gain of less than 50%. A streamlined accuracy assessment was conducted for this sub-indicator report to assess uncertainty in the long-term land cover trend assessment (2000 to 2015). The accuracy assessment focused on the following land cover changes: agriculture to developed, forest to developed, forest to agriculture, wetland to agriculture, and wetland to developed. Accuracies on the order of 50% were estimated for this analysis for land cover change in both the AAFC and NLCD datasets.

Relationships between land cover change and water quality and ecological outcomes are highly complex, particularly at the watershed scale of this sub-indicator. The utility of this sub-indicator could be enhanced by supplementing the existing approach with validated methods that more explicitly examine these relationships. For example methods used to estimate the effects of observed land cover change on nutrient export (see, e.g., Wickham et al. 2008 and USDA 2018) could be used to supplement the sub-indicator trend assessment.

Additional Information

As natural lands become converted to agricultural or urban uses, ecosystem goods and services provided by those lands, such as timber, water storage and purification, wildlife habitat, carbon storage, recreation, and aesthetic beauty, are changed. Direct consequences for the Great Lakes ecosystem can include increased runoff and associated increased inputs of sediment, nutrients, and contaminants to inland waters and the Great Lakes (Bartsch et al. 2015, Cuffney et al. 2010, Michalak et al. 2013, Pearce and Yates 2020, Seilheimer et al. 2013, Thomas et al. 2020, Wolter et al. 2006). High rates of land conversion place stress on natural ecosystems and are typically associated with inefficient land uses, such as those resulting from urban sprawl. Spatial patterns of land conversion affect wildlife habitat and associated wildlife populations and communities. Fragmentation of natural or semi-natural lands can create migration barriers or inhospitable habitats for wildlife and interfere with other ecological processes. This is a particular concern under changing climate conditions. Forest interior breeding birds in the Great Lakes and other ecoregions have higher breeding success in relatively unfragmented landscapes than fragmented landscapes (Robinson et al. 1995). The size and number of natural habitat patches has a significant influence on a variety of wildlife populations, including populations in the Great Lakes region (Saunders et al., 2002). Finally, small ownership parcels found in fragmented landscapes complicate management and cooperation at landscape and watershed scales (Pijanowski and Robinson 2011).

Possible future enhancements to the land cover sub-indicator include the following:

- Development of thresholds for assessing land cover status based on natural land cover types based on research demonstrating linkages between natural land cover, water quality, and ecological condition
- Development of thresholds for assessing land cover trends based on research demonstrating linkages between rate of conversion among land cover classes and stressors on water quality and ecological condition
- Incorporation of additional information regarding land cover quality (e.g., impervious surfaces, tree canopy cover) to better align and allow for more detailed, harmonized land cover classes and/or support more robust land cover status and trend assessment
- Incorporation of more formalized accuracy assessment to quantify uncertainty in land cover change and trend analysis
- Incorporation of more explicit spatial relationships between land cover data, inland waters, and the Great Lakes to identify land cover changes occurring in proximity to water resources that represent greater risk of degradation to the Great Lakes ecosystem
- Incorporation of supplemental analyses and methods to more explicitly examine relationships between observed land cover change and water quality and ecological outcomes of interest

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Table 1. Harmonized land cover classification system used for 2022 State of the Great Lakes Land Cover Sub-indicator. Refer to Jin et al. (2019) and AAFC (2021b) for land cover class definitions. Forest wetland class was created to facilitate harmonization of land cover classes between datasets and required reclassification of NLCD Woody Wetlands (90) and AAFC Forest Regenerating after Fire <20 years (49) land cover classes. See Measure section.

Harmonized Level I Land Cover Class	**NLCD Level II Land Use/Land Cover Class (Land Use/Land Cover Code)**	**AAFC Level II Land Use/Land Cover Class (Land Use/Land Cover Code)**
Developed	• Developed, Open Space (21) • Developed, Low Intensity (22) • Developed, Medium Intensity (23) • Developed, High Intensity (24)	• Settlement (21) • High Reflectance Settlement (22) • Settlement Forest (24) • Roads (25) • Vegetated Settlement (28) • Very High Reflectance Settlement (29)
Agriculture	• Hay/Pasture (81) • Cultivated Crops (82)	• Cropland (51) • Annual Cropland (52)
Forest	• Deciduous Forest (41) • Evergreen Forest (42) • Mixed Forest (43)	• Forest (41) • Forest Regenerating after Harvest <20 years (43) • Forest Regenerating after Fire <20 years (49) that was previously or subsequently Forest (41)
Grassland	• Shrub/Scrub (52) • Herbaceous (71)	• Grassland Managed (61) • Grassland Unmanaged (62)
Forest Wetland	• Woody Wetlands (90) where 2016 NLCD Tree Canopy cover > 40%	• Forest Wetland (42) • Forest Wetland Regenerating after Harvest <20 years (44) • Forest Regenerating after Fire <20 years (49) that was previously or subsequently Forest Wetland (42)
Wetland	• Woody Wetlands (90) where 2016 NLCD Tree Canopy cover ≤ 40% • Emergent Herbaceous Wetlands (95)	• Wetland (71)
Barren	• Perennial Snow/Ice (12) • Barren Land (31)	• Other land (91)
Water	• Open Water (11)	• Water (31)

Table 2. Estimates of land cover area by land cover class in km^2 based on analysis of 2015 AAFC and 2016 NLCD datasets. Values rounded to the nearest 10 km^2.

Watershed/Basin	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Total Land
Lake Superior	2,370	1,480	96,370	1,890	19,080	5,610	250	127,040
Lake Michigan	11,430	35,970	33,420	3,990	22,120	5,910	460	113,310
Lake Huron	7,080	27,000	65,020	2,540	18,240	4,180	240	124,310
Lake Erie	13,320	46,150	10,860	390	3,660	1,300	160	75,850
Lake Ontario	7,830	21,460	23,010	700	7,520	2,320	90	62,920
Great Lakes Basin	42,010	132,060	228,690	9,520	70,630	19,320	1,210	503,440

Table 3. Status assessment based on risk of degraded water/habitat quality. Percent land cover is calculated based on total land area. Area covered by water (e.g., streams, ponds, lakes) is excluded from the calculation. Refer to Status Assessment Definitions section for rationale for cover-specific and overall assessment. See Data Limitations section for discussion of uncertainty in estimated numbers.

Watershed/Basin	Land Cover: **Developed**		Land Cover: **Agriculture**		Overall Status Assessment
	Percent Watershed Area	Risk of Water/Habitat Quality Degradation	Percent Watershed Area	Risk of Water/Habitat Quality Degradation	
Lake Superior	1.9%	Low	1.2%	Low	Good
Lake Michigan	10.1%	Med	31.7%	Med	Fair
Lake Huron	5.7%	Low	21.7%	Med	Fair
Lake Erie	17.6%	Med	60.8%	High	Poor
Lake Ontario	12.4%	Med	34.1%	Med	Fair
Great Lakes Basin	8.3%	Med	26.2%	Med	Fair

Table 4. Estimated net change in area by land cover class from 2000 to 2015 in Lake Superior watershed. Negative numbers represent net losses and positive numbers represent net gains in land cover class. Values displayed to the nearest 1 km^2; as a result, component values shown may not sum to total in "sources of net change" section. See Data Limitations section for discussion of uncertainty in estimated numbers.

Description of Change	Estimated Net Change in Area (km^2) 2000 to 2015: Lake Superior Watershed								
	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Water	Total
Area 2000	2,191	1,492	96,894	1,507	18,989	5,701	250	96,424	223,449
Area 2015	2,368	1,478	96,373	1,887	19,079	5,607	252	96,404	223,449
Total change	177	-14	-521	380	90	-94	2	-20	0
Change as %2000	8.1%	-0.9%	-0.5%	25.2%	0.5%	-1.6%	0.7%	0.0%	0.0%
Annual change	0.54%	-0.06%	-0.04%	1.68%	0.03%	-0.11%	0.05%	0.00%	0.00%
Sources of Net Change									
From Developed	0	-6	-156	-5	-6	-1	-3	0	-177
From Agriculture	6	0	5	10	-2	-5	0	0	14
From Forest	156	-5	0	371	-1	1	3	-3	521
From Grassland	5	-10	-371	0	-1	1	0	-6	-380
From Forest Wetland	6	2	1	1	0	-98	0	-2	-90
From Wetland	1	5	-1	-1	98	0	-1	-8	94
From Barren	3	0	-3	0	0	1	0	-1	-2
From Water	0	0	3	6	2	8	1	0	20
Total change	177	-14	-521	380	90	-94	2	-20	0
Net Conversion of Natural Land to Developed or Agriculture Land Cover									
Net area converted	---	---	-151	5	-8	-7	---	---	---
% Net area converted	---	---	-0.2%	0.3%	0.0%	-0.1%	---	---	---
Annual change	---	---	-0.01%	0.02%	0.00%	-0.01%	---	---	---

Table 5. Estimated net change in area by land cover class from 2000 to 2015 in Lake Michigan watershed. Negative numbers represent net losses and positive numbers represent net gains in land cover class. Values displayed to the nearest 1 km^2; as a result, component values shown may not sum to total in "sources of net change" section. See Data Limitations section for discussion of uncertainty in estimated numbers.

Description of Change	Estimated Net Change in Area (km^2) 2000 to 2015: Lake Michigan Watershed								
	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Water	Total
Area 2000	10,899	36,272	33,776	3,890	21,949	6,082	492	61,204	174,564
Area 2015	11,427	35,975	33,423	3,991	22,120	5,913	463	61,252	174,564
Total change	528	-297	-353	101	171	-170	-28	48	0
Change as %2000	4.8%	-0.8%	-1.0%	2.6%	0.8%	-2.8%	-5.8%	0.1%	0.0%
Annual change	0.32%	-0.05%	-0.07%	0.17%	0.05%	-0.19%	-0.38%	0.01%	0.00%
Sources of Net Change									
From Developed	0	-355	-73	-55	-24	-8	-9	-4	-528
From Agriculture	355	0	13	-128	-9	49	6	11	297
From Forest	73	-13	0	285	-2	5	2	2	353
From Grassland	55	128	-285	0	0	3	0	-1	-101
From Forest Wetland	24	9	2	0	0	-203	0	-3	-171
From Wetland	8	-49	-5	-3	203	0	-1	17	170
From Barren	9	-6	-2	0	0	1	0	27	28
From Water	4	-11	-2	1	3	-17	-27	0	-48
Total change	528	-297	-353	101	171	-170	-28	48	0
Net Conversion of Natural Land to Developed or Agriculture Land Cover									
Net area converted	---	---	-60	-182	-33	41	---	---	---
% Net area converted	---	---	-0.2%	-4.7%	-0.2%	0.7%	---	---	---
Annual change	---	---	-0.01%	-0.31%	-0.01%	0.04%	---	---	---

Table 6. Estimated net change in area by land cover class from 2000 to 2015 in Lake Huron watershed. Negative numbers represent net losses and positive numbers represent net gains in land cover class. Values displayed to the nearest 1 km²; as a result, component values shown may not sum to total in "sources of net change" section. See Data Limitations section for discussion of uncertainty in estimated numbers.

Description of Change	Estimated Net Change in Area (km²) 2000 to 2015: Lake Huron Watershed								
	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Water	Total
Area 2000	6,361	26,617	66,089	2,351	18,294	4,346	274	68,786	193,118
Area 2015	7,075	27,003	65,024	2,543	18,245	4,179	244	68,805	193,118
Total change	714	386	-1,064	192	-49	-168	-30	18	0
Change as %2000	11.2%	1.4%	-1.6%	8.2%	-0.3%	-3.9%	-10.9%	0.0%	0.0%
Annual change	0.75%	0.10%	-0.11%	0.54%	-0.02%	-0.26%	-0.73%	0.00%	0.00%
Sources of Net Change									
From Developed	0	-241	-413	-11	-27	-12	-9	-1	-714
From Agriculture	241	0	-369	-75	-63	-124	1	4	-386
From Forest	413	369	0	279	0	2	1	0	1,064
From Grassland	11	75	-279	0	0	1	0	0	-192
From Forest Wetland	27	63	0	0	0	-40	0	-2	49
From Wetland	12	124	-2	-1	40	0	-2	-4	168
From Barren	9	-1	-1	0	0	2	0	21	30
From Water	1	-4	0	0	2	4	-21	0	-18
Total change	714	386	-1064	192	-49	-168	-30	18	0
Net Conversion of Natural Land to Developed or Agriculture Land Cover									
Net area converted	---	---	-782	-86	-90	-136	---	---	---
% Net area converted	---	---	-1.2%	-3.6%	-0.5%	-3.1%	---	---	---
Annual change	---	---	-0.08%	-0.24%	-0.03%	-0.21%	---	---	---

Table 7. Estimated net change in area by land cover class from 2000 to 2015 in Lake Erie watershed. Negative numbers represent net losses and positive numbers represent net gains in land cover class. Values displayed to the nearest 1 km^2; as a result, component values shown may not sum to total in "sources of net change" section. See Data Limitations section for discussion of uncertainty in estimated numbers.

Description of Change	Estimated Net Change in Area (km^2) 2000 to 2015: Lake Erie Watershed								
	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Water	Total
Area 2000	12,553	46,369	11,250	376	3,727	1,420	167	27,779	103,641
Area 2015	13,315	46,146	10,862	394	3,664	1,303	162	27,793	103,641
Total change	762	-224	-388	18	-62	-117	-4	15	0
Change as %2000	6.1%	-0.5%	-3.4%	4.9%	-1.7%	-8.2%	-2.6%	0.1%	0.0%
Annual change	0.40%	-0.03%	-0.23%	0.32%	-0.11%	-0.55%	-0.18%	0.00%	0.00%
Sources of Net Change									
From Developed	0	-508	-188	-26	-20	-7	-9	-5	-762
From Agriculture	508	0	-146	-10	-49	-99	8	12	224
From Forest	188	146	0	52	0	1	0	1	388
From Grassland	26	10	-52	0	0	0	0	-2	-18
From Forest Wetland	20	49	0	0	0	-6	0	-1	62
From Wetland	7	99	-1	0	6	0	0	7	117
From Barren	9	-8	0	0	0	0	0	3	4
From Water	5	-12	-1	2	1	-7	-3	0	-15
Total change	762	-224	-388	18	-62	-117	-4	15	0
Net Conversion of Natural Land to Developed or Agriculture Land Cover									
Net area converted	---	---	-334	-36	-69	-106	---	---	---
% Net area converted	---	---	-3.0%	-9.5%	-1.9%	-7.4%	---	---	---
Annual change	---	---	-0.20%	-0.64%	-0.12%	-0.50%	---	---	---

Table 8. Estimated net change in area by land cover class from 2000 to 2015 in Lake Ontario watershed. Negative numbers represent net losses and positive numbers represent net gains in land cover class. Values displayed to the nearest 1 km^2; as a result, component values shown may not sum to total in "sources of net change" section. See Data Limitations section for discussion of uncertainty in estimated numbers.

Description of Change	Estimated Net Change in Area (km^2) 2000 to 2015: Lake Ontario Watershed								
	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Water	Total
Area 2000	7,115	21,423	23,581	659	7,580	2,446	88	22,306	85,199
Area 2015	7,827	21,457	23,007	705	7,518	2,323	86	22,277	85,199
Total change	712	34	-574	46	-61	-123	-2	-30	0
Change as %2000	10.0%	0.2%	-2.4%	6.9%	-0.8%	-5.0%	-2.7%	-0.1%	0.0%
Annual change	0.67%	0.01%	-0.16%	0.46%	-0.05%	-0.34%	-0.18%	-0.01%	0.00%
Sources of Net Change									
From Developed	0	-388	-272	-9	-27	-9	-5	-2	-712
From Agriculture	388	0	-239	-17	-56	-113	2	2	-34
From Forest	272	239	0	65	-1	0	0	-2	574
From Grassland	9	17	-65	0	-2	0	0	-4	-46
From Forest Wetland	27	56	1	2	0	-24	0	-1	61
From Wetland	9	113	0	0	24	0	0	-22	123
From Barren	5	-2	0	0	0	0	0	-1	2
From Water	2	-2	2	4	1	22	1	0	30
Total change	712	34	-574	46	-61	-123	-2	-30	0
Net Conversion of Natural Land to Developed or Agriculture Land Cover									
Net area converted	---	---	-512	-26	-84	-121	---	---	---
% Net area converted	---	---	-2.2%	-3.9%	-1.1%	-5.0%	---	---	---
Annual change	---	---	-0.14%	-0.26%	-0.07%	-0.33%	---	---	---

Table 9. Estimated net change in area by land cover class from 2000 to 2015 in Great Lakes basin. Negative numbers represent net losses and positive numbers represent net gains in land cover class. Values displayed to the nearest 1 km^2; as a result, component values shown may not sum to total in "sources of net change" section. See Data Limitations section for discussion of uncertainty in estimated numbers.

Description of Change	Estimated Net Change in Area (km^2) 2000 to 2015: Great Lakes Basin								
	Developed	Agriculture	Forest	Grassland	Forest Wetland	Wetland	Barren	Water	Total
Area 2000	39,120	132,173	231,589	8,783	70,538	19,995	1,271	276,499	779,970
Area 2015	42,013	132,058	228,689	9,520	70,627	19,324	1,208	276,531	779,970
Total change	2,893	-115	-2,900	737	88	-671	-63	32	0
Change as %2000	7.4%	-0.1%	-1.3%	8.4%	0.1%	-3.4%	-5.0%	0.0%	0.0%
Annual change	0.49%	-0.01%	-0.08%	0.56%	0.01%	-0.22%	-0.33%	0.00%	0.00%
Sources of Net Change									
From Developed	0	-1,498	-1,102	-105	-104	-37	-35	-12	-2,893
From Agriculture	1,498	0	-736	-219	-181	-293	17	29	115
From Forest	1,102	736	0	1,052	-4	10	6	-2	2,900
From Grassland	105	219	-1,052	0	-3	6	1	-13	-737
From Forest Wetland	104	181	4	3	0	-372	1	-9	-88
From Wetland	37	293	-10	-6	372	0	-4	-11	671
From Barren	35	-17	-6	-1	-1	4	0	50	63
From Water	12	-29	2	13	9	11	-50	0	-32
Total change	2,893	-115	-2,900	737	88	-671	-63	32	0
Net Conversion of Natural Land to Developed or Agriculture Land Cover									
Net area converted	---	---	-1,838	-324	-284	-329	---	---	---
% Net area converted	---	---	-0.8%	-3.7%	-0.4%	-1.6%	---	---	---
Annual change	---	---	-0.05%	-0.25%	-0.03%	-0.11%	---	---	---

Table 10. Estimated net change in forest land cover from 2000 to 2015, including total net change and net conversion to/from developed and agriculture land cover types. Negative numbers represent net losses and positive numbers represent net gains in forest land cover. See Data Limitations section for discussion of uncertainty in estimated numbers.

Watershed/Basin	Forest Land Cover				Change (2000 - 2015)			
	2000		2015		Total		Conversion to Developed or Agriculture	
	Area (km2)	% Land Area	Area (km2)	% Land Area	Area (km2)	Annual Rate	Area (km2)	Annual Rate
Lake Superior	96,894	76.3%	96,373	75.9%	-521	-0.04%	-151	-0.01%
Lake Michigan	33,776	29.8%	33,423	29.5%	-353	-0.07%	-60	-0.01%
Lake Huron	66,089	53.2%	65,024	52.3%	-1,064	-0.11%	-782	-0.08%
Lake Erie	11,250	14.8%	10,862	14.3%	-388	-0.23%	-334	-0.20%
Lake Ontario	23,581	37.5%	23,007	36.6%	-574	-0.16%	-512	-0.14%
Great Lakes Basin	231,589	46.0%	228,689	45.4%	-2,900	-0.08%	-1,838	-0.05%

Table 11. Estimated net change in forest wetland and wetland land cover (combined) from 2000 to 2015, including total net change and net conversion to/from developed and agriculture land cover types. Negative numbers represent net losses and positive numbers represent net gains in forest wetland and wetland land cover. See Data Limitations section for discussion of uncertainty in estimated numbers.

Watershed/Basin	Forest Wetland and Wetland Land Cover				Change (2000 - 2015)			
	2000		2015		Total		Conversion to Developed or Agriculture	
	Area (km2)	% Land Area	Area (km2)	% Land Area	Area (km2)	Annual Rate	Area (km2)	Annual Rate
Lake Superior	24,690	19.4%	24,686	19.4%	-4	0.00%	-15	0.00%
Lake Michigan	28,031	24.7%	28,032	24.7%	2	0.00%	8	0.00%
Lake Huron	22,640	18.2%	22,424	18.0%	-216	-0.06%	-227	-0.07%
Lake Erie	5,147	6.8%	4,967	6.5%	-179	-0.23%	-175	-0.23%
Lake Ontario	10,026	15.9%	9,841	15.6%	-185	-0.12%	-205	-0.14%
Great Lakes Basin	90,534	18.0%	89,951	17.9%	-583	-0.04%	-613	-0.05%

Table 12. Estimated net change in developed land cover from 2000 to 2015. Positive numbers represent a net increase in developed land cover. See Data Limitations section for discussion of uncertainty in estimated numbers.

Watershed/Basin	Developed Land Cover				Change (2000 - 2015)	
	2000		2015		Total	
	Area (km2)	% Land Area	Area (km2)	% Land Area	Area (km2)	Annual Rate
Lake Superior	2,191	1.7%	2,368	1.9%	177	0.54%
Lake Michigan	10,899	9.6%	11,427	10.1%	528	0.32%
Lake Huron	6,361	5.1%	7,075	5.7%	714	0.75%
Lake Erie	12,553	16.5%	13,315	17.6%	762	0.40%
Lake Ontario	7,115	11.3%	7,827	12.4%	712	0.67%
Great Lakes Basin	39,120	7.8%	42,013	8.3%	2,893	0.49%

Table 13. Estimated net change in agriculture land cover from 2000 to 2015, including total net change and net conversion to/from developed land cover. Negative numbers represent net losses and positive numbers represent net gains in agriculture land cover. See Data Limitations section for discussion of uncertainty in estimated numbers.

Watershed/Basin	Agriculture Land Cover				Change (2000 - 2015)			
	2000		2015		Total		Conversion to Developed	
	Area (km2)	% Land Area	Area (km2)	% Land Area	Area (km2)	Annual Rate	Area (km2)	Annual Rate
Lake Superior	1,492	1.2%	1,478	1.2%	-14	-0.06%	-6	-0.03%
Lake Michigan	36,272	32.0%	35,975	31.7%	-297	-0.05%	-355	-0.07%
Lake Huron	26,617	21.4%	27,003	21.7%	386	0.10%	-241	-0.06%
Lake Erie	46,369	61.1%	46,146	60.8%	-224	-0.03%	-508	-0.07%
Lake Ontario	21,423	34.1%	21,457	34.1%	34	0.01%	-388	-0.12%
Great Lakes Basin	132,173	26.3%	132,058	26.2%	-115	-0.01%	-1,498	-0.08%

Table 14. Trend assessment based on average annual change in land cover from 2000 to 2015. Trend assessment is based on average annual change of ±0.1%, or a 10-year average change of ±1%, where positive change for developed and agriculture land cover (increase) is defined as deteriorating condition and negative change in natural land cover to developed or agriculture land cover (loss of natural land cover) is defined as deteriorating condition. Refer to Trend Assessment definitions section. Annual change in agriculture land cover is based on total change in agriculture (Table 13, column 7), which includes estimated net conversion of agriculture land cover to and from all land cover classes.

Watershed/Basin	Annual Change in Land Cover		Annual Change from Natural to Developed or Agriculture Land Cover		Trend Assessment
	Developed	Agriculture	Forest	Wetland	
Lake Superior	0.54%	-0.06%	-0.01%	0.00%	Unchanging
Lake Michigan	0.32%	-0.05%	-0.01%	0.00%	Unchanging
Lake Huron	0.75%	0.10%	-0.08%	-0.07%	Unchanging
Lake Erie	0.40%	-0.03%	-0.20%	-0.23%	Deteriorating
Lake Ontario	0.67%	0.01%	-0.14%	-0.14%	Deteriorating
Great Lakes Basin	0.49%	-0.01%	-0.05%	-0.05%	Unchanging

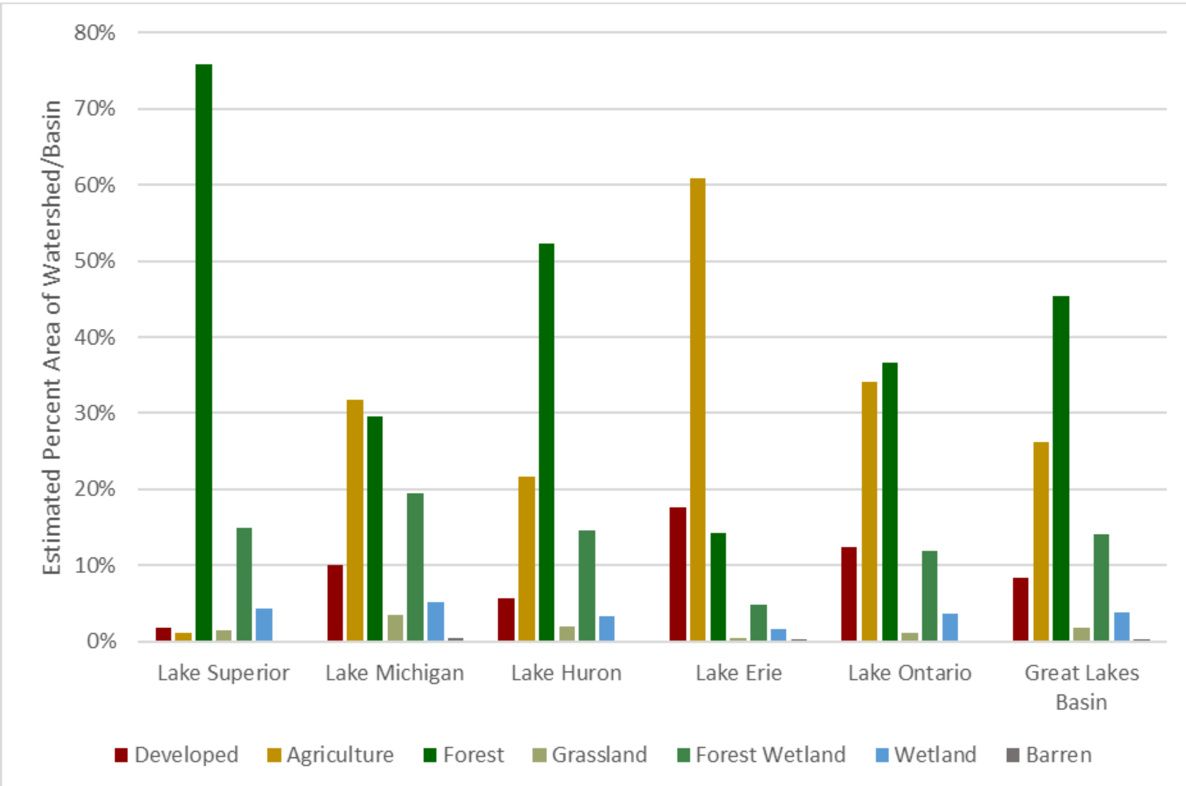

Figure 1. Estimates of land use/land cover area by land cover class across the Great Lakes basin and watersheds, 2015. Source: AAFC (2021a) and Dewitz (2019).

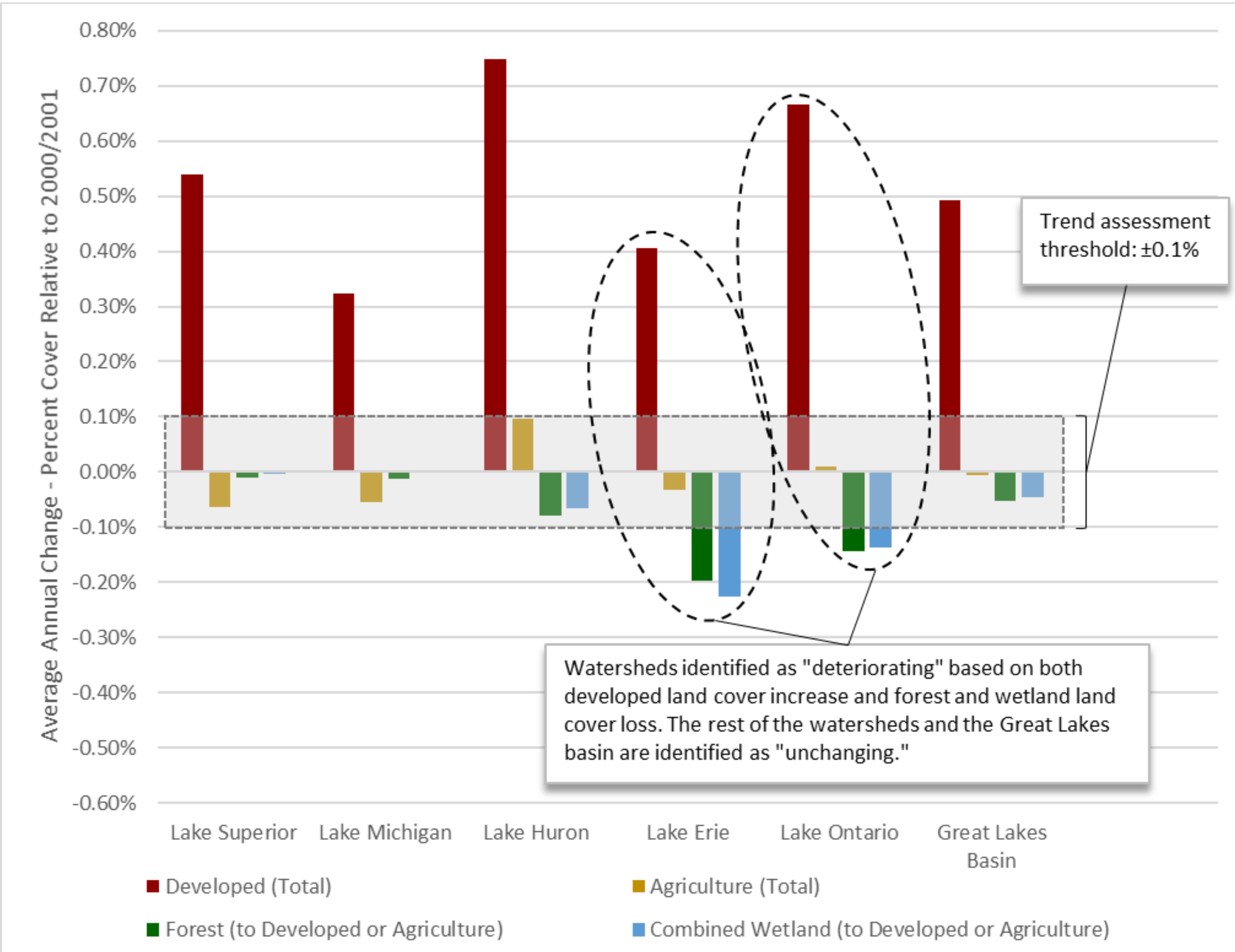

Figure 2. Trend assessment based on estimated average annual net change in developed, agriculture, forest, and combined wetland land cover from 2000 to 2015. Trend assessment is based on average annual change of ±0.1%, or a 10-year average change of ±1%. "Deteriorating" conditions were determined for the Lake Erie and Lake Ontario watersheds based on net estimated increases in developed land cover of ≥0.1%/year and net estimated decreases in forest and combined wetland (forest wetland and wetland) land cover of ≥0.1%/year in each watershed.

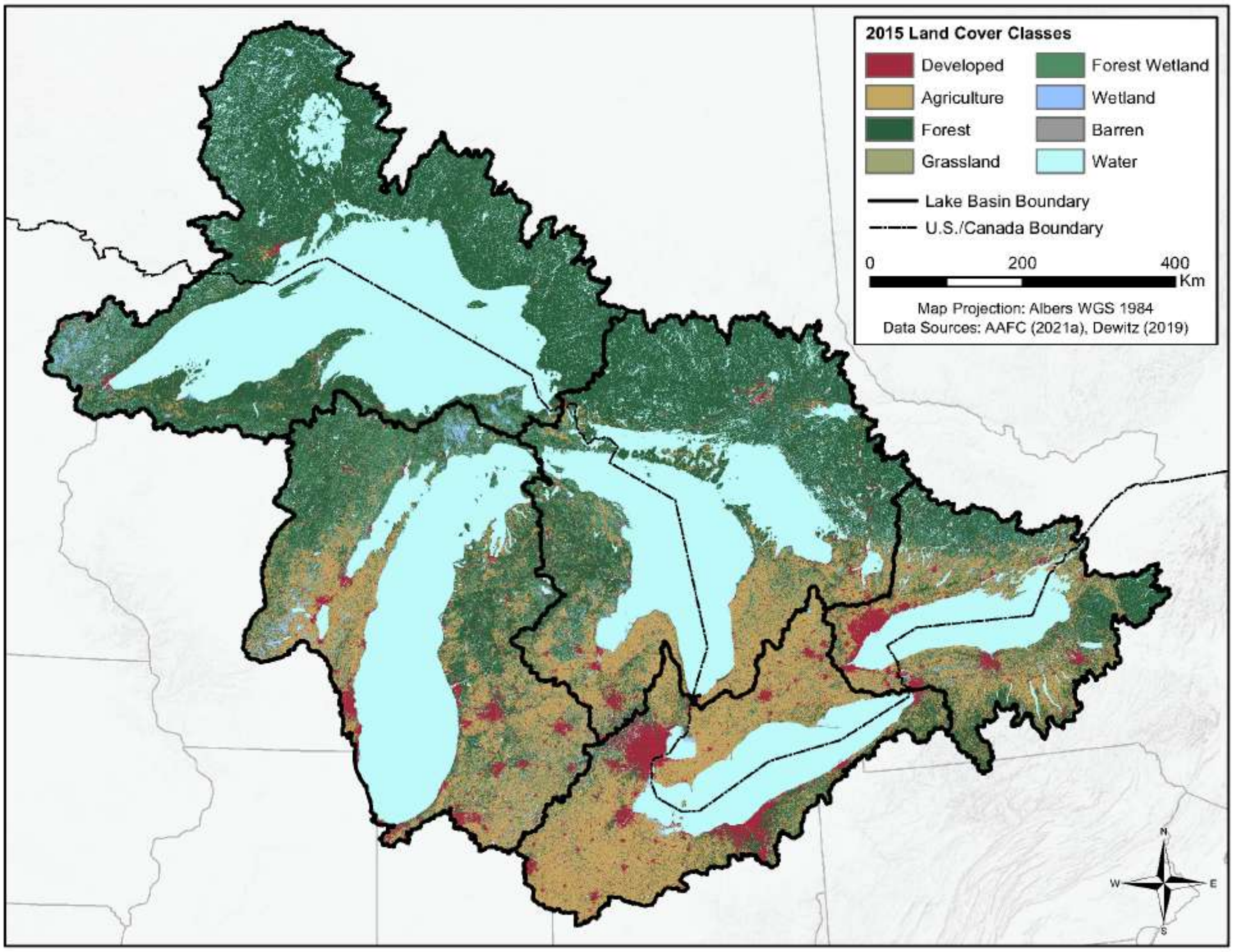

Figure 3. Distribution of land use/land cover across the Great Lakes basin, 2015. Sources: AAFC (2021a) and Dewitz (2019).

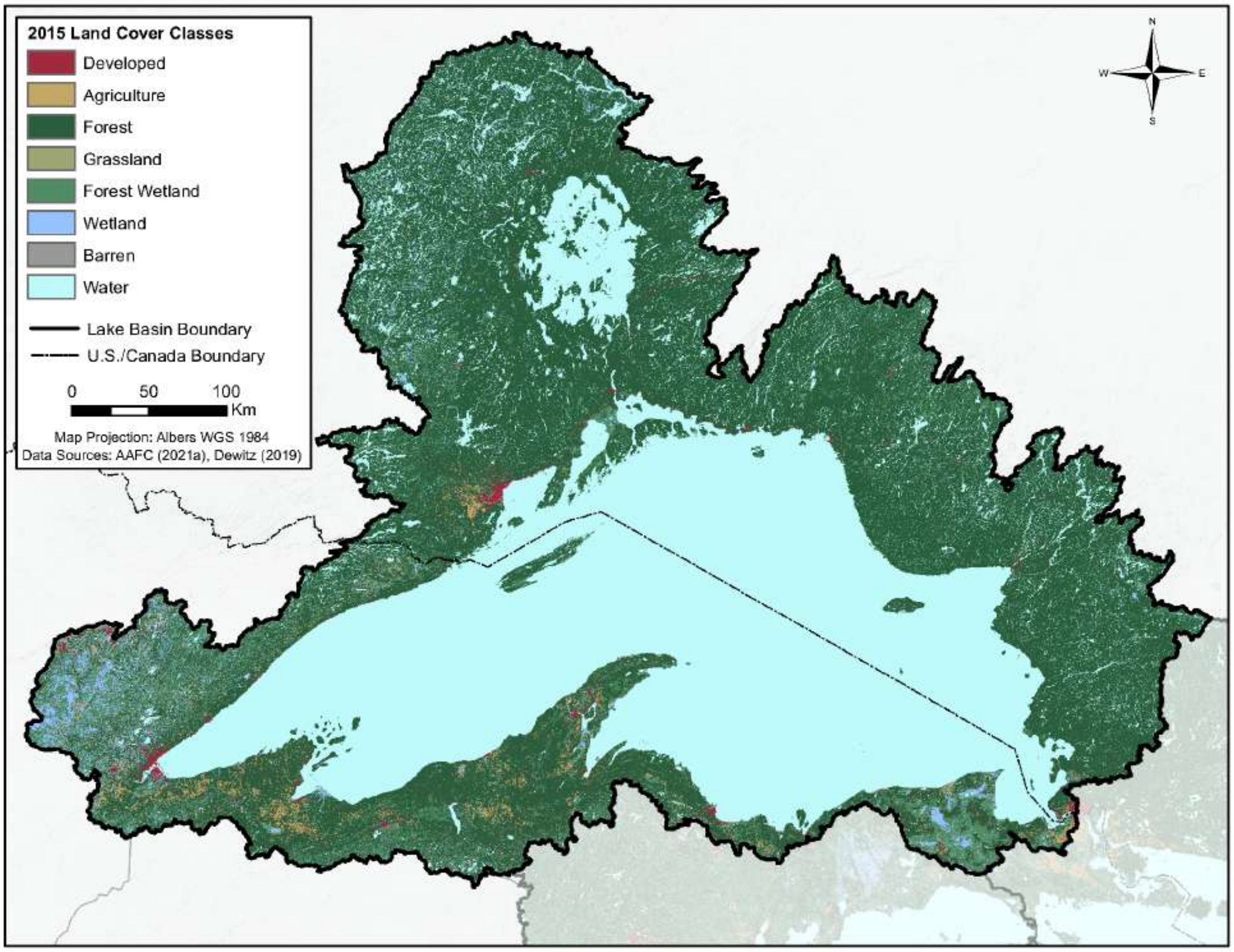

Figure 4. Distribution of land use/land cover across the Lake Superior watershed, 2015. Sources: AAFC (2021a) and Dewitz (2019).

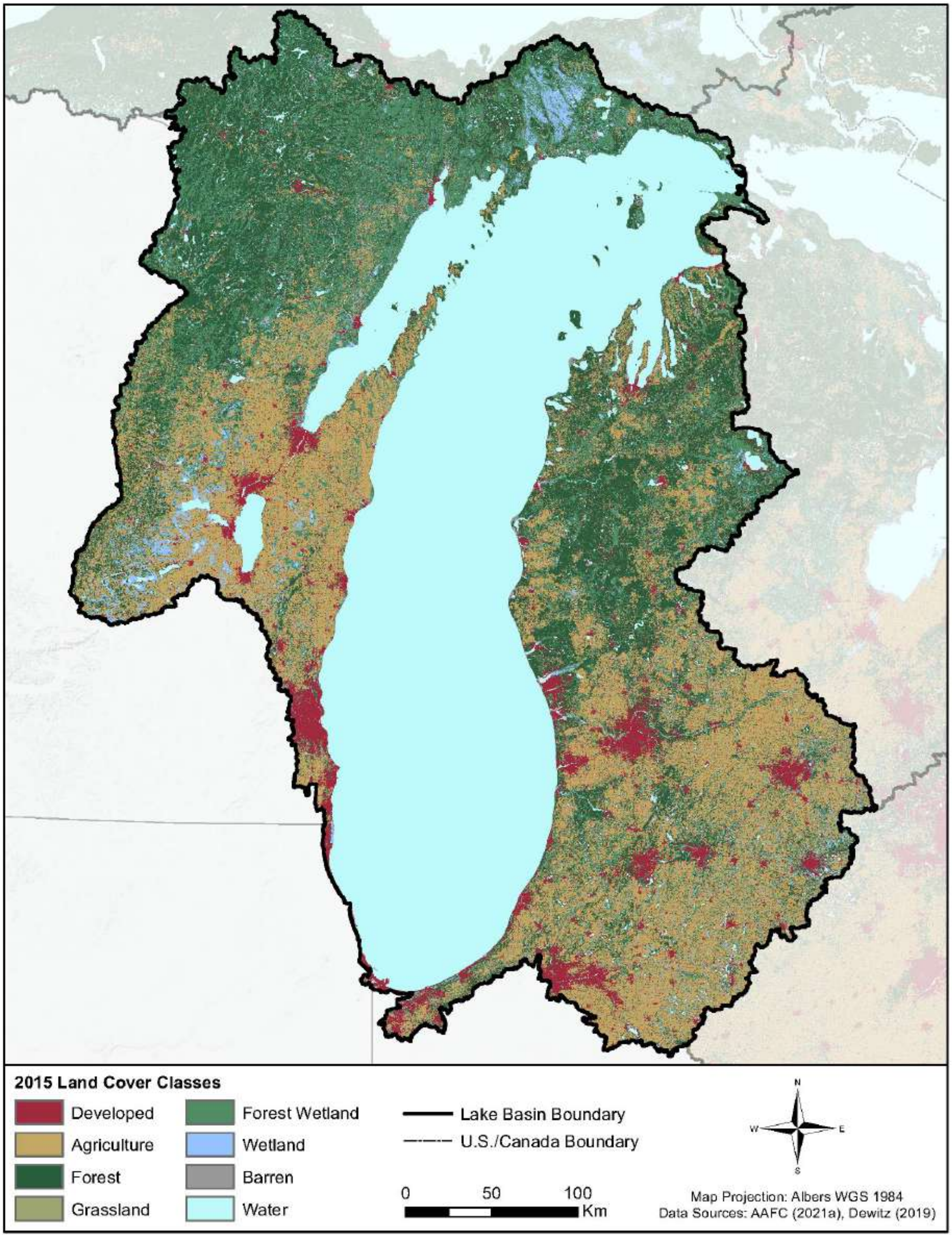

Figure 5. Distribution of land use/land cover across the Lake Michigan watershed, 2015. Sources: AAFC (2021a) and Dewitz (2019).

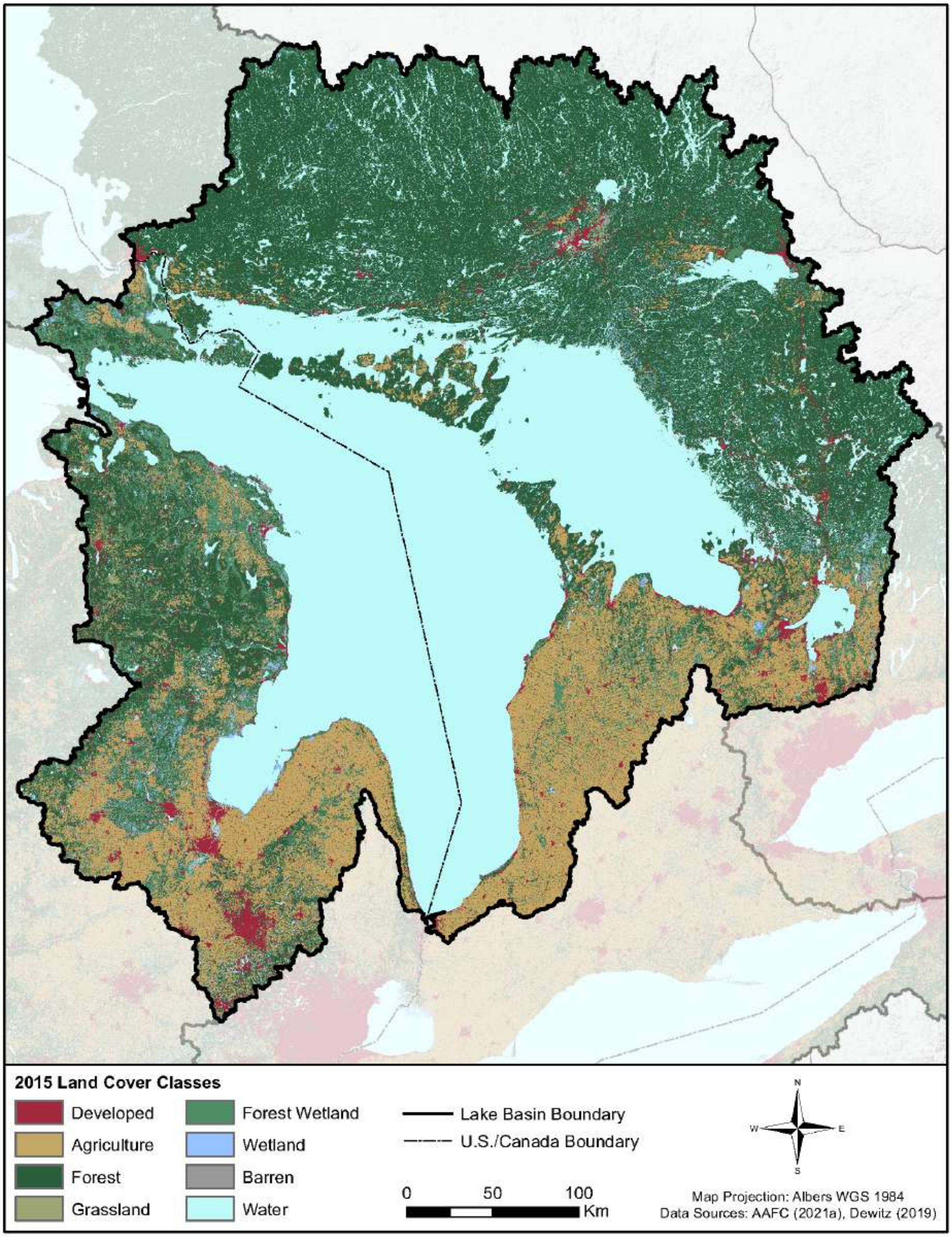

Figure 6. Distribution of land use/land cover across the Lake Huron basin, 2015. Sources: AAFC (2021a) and Dewitz (2019).

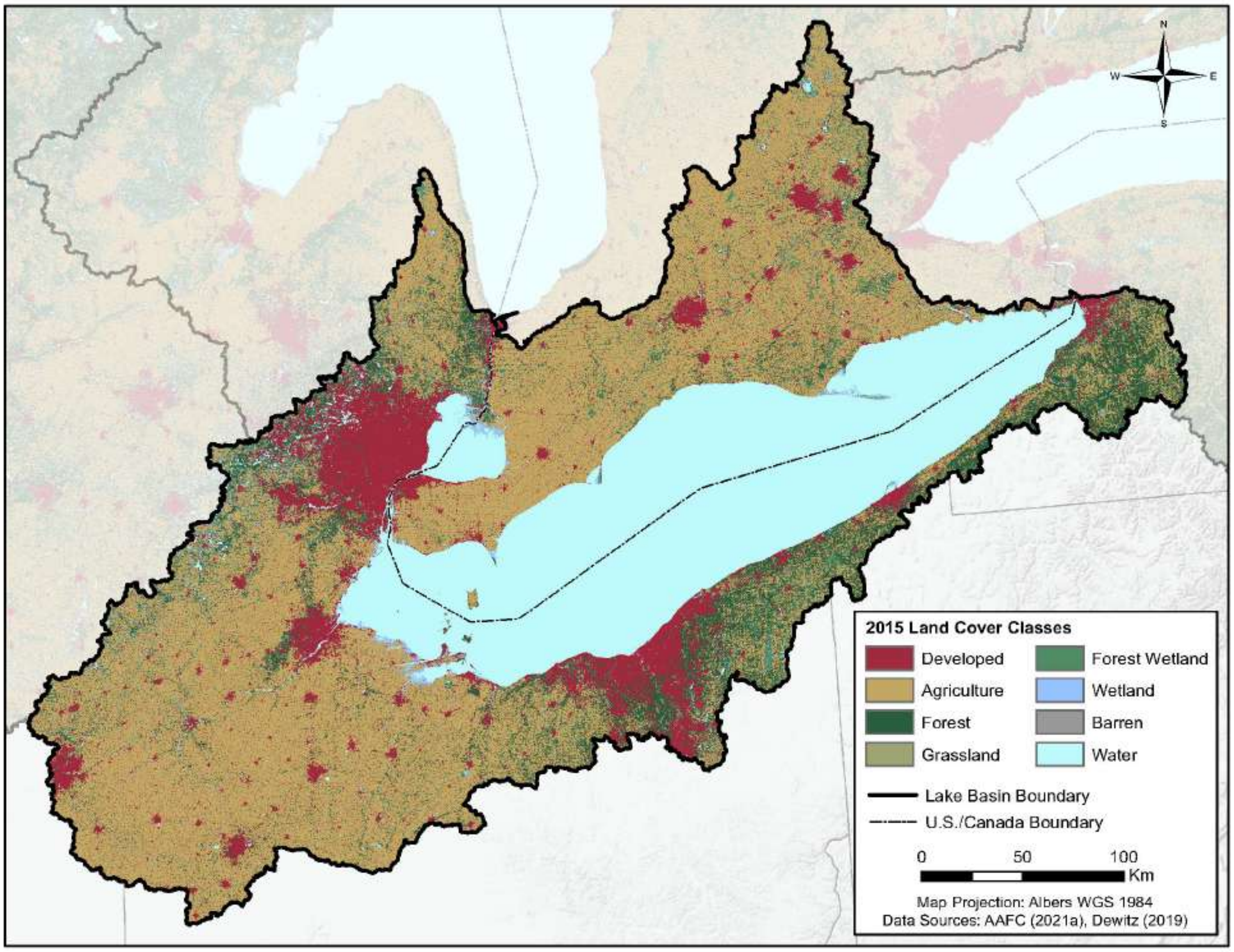

Figure 7. Distribution of land use/land cover across the Lake Erie basin, 2015. Sources: AAFC (2021a) and Dewitz (2016).

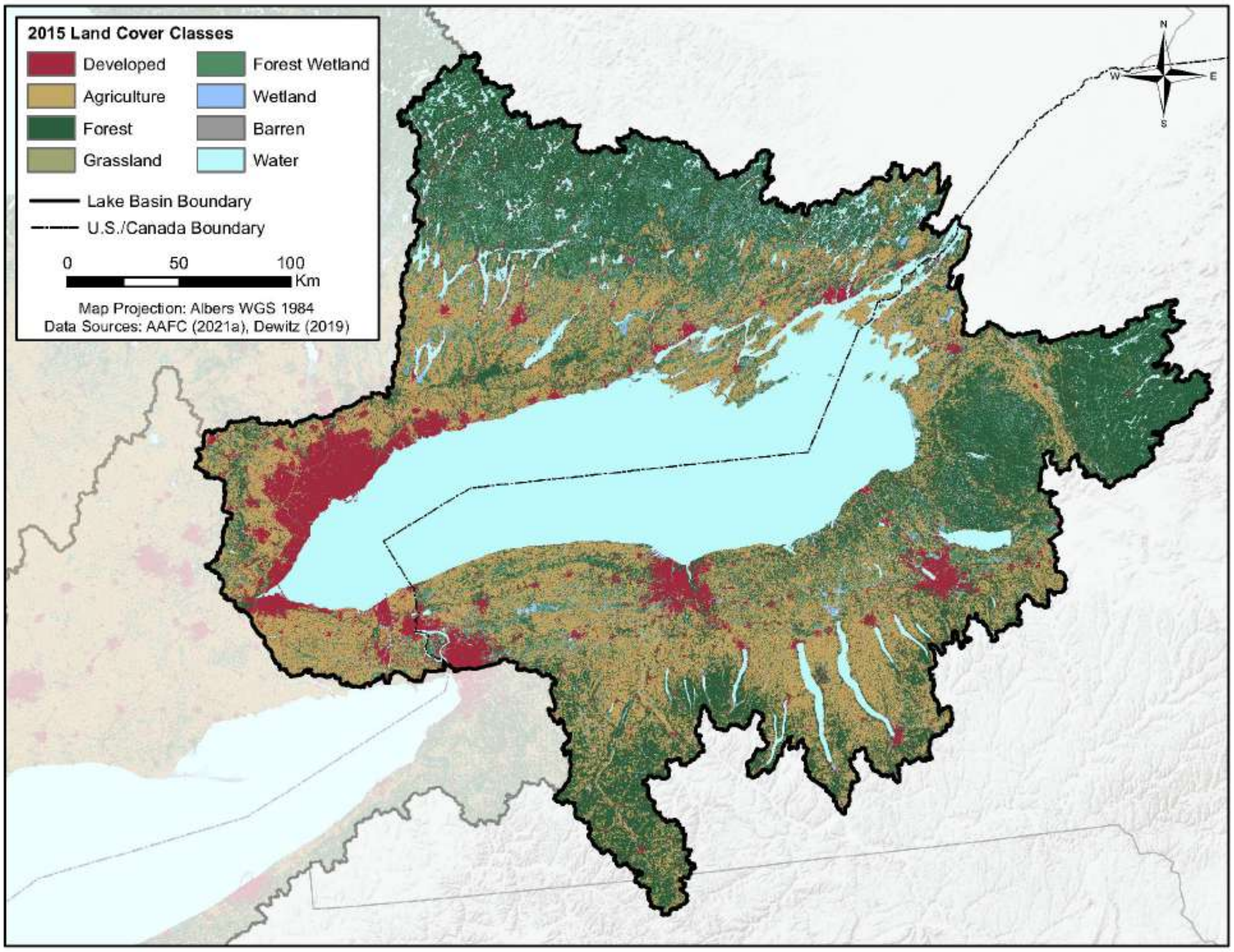

Figure 8. Distribution of land use/land cover across the Lake Ontario basin, 2015. Sources: AAFC (2021a) and Dewitz (2019).

Sub-Indicator: Hardened Shorelines

Overall Assessment

Status: *Fair (based on United States and Canadian shoreline data currently available for evaluation)*

Trends:

10-Year Trend: *Deteriorating (based on United States and Canadian shoreline data currently available for evaluation)*

Long-term Trend (2009 - 2021): *Deteriorating (based on United States and Canadian shoreline data currently available for evaluation)*

Rationale: A comprehensive, binational status and trends assessment of the hardened shorelines of the entire Great Lakes is not possible at this time. However, using the data that is currently available it is possible to provide an assessment of the status and trend for the hardened shorelines sub-indicator provided above in *italics*. At the time of reporting all shoreline data except for Canada's Lake Superior were available and incorporated into the 2022 assessment and reporting, see Additional Information section. Based on the currently available data (Figure 1) the overall status for the Great Lakes is currently classed as *Fair*, with 23.9% of the shoreline in the combined moderately and highly protected shoreline classifications (Table 3). It is anticipated that this percentage will change once the Canadian Lake Superior data is made available in 2023, however, the status classification is likely to remain the same, see Status Assessment Definitions section. The overall trend assessment is currently listed as *Deteriorating* based on the net increase in the percentage of hardened shorelines in the highly protected or moderately protected categories between 2009 to 2021 for each of the lake assessments (Table 4).

Lake-by-Lake Assessment

Lake Superior

Status: *Good (based only on United States shoreline evaluation)*

10-Year Trend: Undetermined

Long-term Trend (2009-2021): Undetermined

Rationale: A comprehensive, binational status and trends assessment of the hardened shorelines of Lake Superior is not possible at this time due to incomplete data availability, see Additional Information section. Based on US data only, greater than 90% of the shorelines of Lake Superior have minor or no shoreline protection, making the status Good. The Canadian shoreline of Lake Superior is also highly undeveloped, so the addition of Canadian data in the future is unlikely to change this assessment (Tables 3 and 4, Figure 3).

Since the Last Report: Lake Superior was not assessed for hardened shorelines in the 2011 report. Data is still incomplete on the Canadian side for Lake Superior at the time of draft submission, see Additional Information section.

Lake Michigan

Status: Good

10-Year Trend: Deteriorating

Long-term Trend (2009-2021): Deteriorating

Rationale: The status for Lake Michigan still remains Good from 2009 to 2021, but there has been an increase in the percentage of hardened shorelines in the combined moderately and highly protected shoreline classifications from 11.5% in 2009 to 16.6% in 2021 (Tables 2, 3, and 4), resulting in a Deteriorating trend.

Since the Last Report: This is the first attempt since the 2009 State of the Great Lakes hardened shorelines sub-indicator report (Tables 1 and 2) to quantify the amount of hardened shoreline in Lake Michigan. It appears that has been an increased amount of hardened shorelines in the combined moderately and highly protected shoreline classifications by over 5% (Tables 3 and 4, Figure 4).

Lake Huron (including St. Marys River)

Status: Good

10-Year Trend: Deteriorating

Long-term Trend (2009-2021): Deteriorating

Rationale: The status for Lake Huron and the St. Marys River remains Good from 2009 to 2021, but there has been an increase in the amount of hardened shorelines in the combined moderately and highly protected shoreline classifications from 2.7% in 2009 to 11.7% in 2021 (Tables 2, 3, and 4), resulting in a Deteriorating trend

Since the Last Report: This is the first attempt since the 2009 State of the Great Lakes hardened shorelines sub-indicator report (Tables 1 and 2) to quantify the amount of hardened shoreline in Lake Huron and the St. Marys River. It appears that has been an increased percentage of hardened shorelines in the combined moderately and highly protected shoreline classifications by 9% (Tables 3 and 4, Figure 5).

Lake Erie (including St. Clair-Detroit River Ecosystem)

Status: Poor

10-Year Trend: Deteriorating

Long-term Trend (2009-2021): Deteriorating

Rationale: The status for Lake Erie and its connecting channels still remains Poor from 2009 to 2021, and there has been an increase in the percentage of hardened shorelines in the moderately and highly protected shoreline classifications from 39.0% in 2009 to 56.9% in 2021 (Tables 2, 3, and 4), resulting in a Deteriorating trend.

Since the Last Report: This is the first attempt since the 2009 State of the Great Lakes hardened shorelines sub-indicator report (Tables 1 and 2) to quantify the amount of hardened shoreline in Lake Erie (including St. Clair-Detroit River Ecosystem). It appears that has been an increased amount of hardened shorelines in the combined moderately and highly protected shoreline classifications by over 15% (Tables 3 and 4, Figure 6).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Status: Poor

10-Year Trend: Deteriorating

Long-term Trend (2009-2021): Deteriorating

Rationale: While Lake Ontario was considered Fair in the 2009 State of the Great Lakes hardened shorelines sub-indicator report at 21.0% for the combined moderately and highly protected shoreline classification (Tables 1 and 2), the status for Lake Ontario and its connecting channels was classed Poor in 2019 and again in 2021. There has been an increased amount of hardened shorelines in the moderately and highly protected shoreline classifications now at 40.6% (Tables 4), resulting in a Deteriorating trend.

Since the Last Report: Since the 2019 State of the Great Lakes hardened shorelines sub-indicator report and 2009 report, increased armoring has occurred in Lake Ontario and connecting channels (Tables 3 and 4, Figure 7).

Status Assessment Definitions

Good: >80% of the shoreline reaches have minor to no protection (20% or less of the shoreline is moderately or highly protected)

Fair: 70-80% of the shoreline reaches have minor to no protection (20-30% of the shoreline is moderately or highly protected)

Poor: < 70% of the shoreline reaches have minor to no protection (30% or more of the shoreline is moderately or highly protected)

Undetermined: Data are not available or are insufficient to assess status condition of the ecosystem components.

Trend Assessment Definitions

Improving: Net decrease or no net increase in the percentage of hardened shorelines in the highly protected or moderately protected categories.

Unchanging: No change in the amount percentage of hardened shorelines in the highly protected or moderately protected categories.

Deteriorating: Net increase in the percentage of hardened shorelines in the highly protected or moderately protected categories.

Undetermined: Metrics do not indicate a clear overall trend or data are not available or are insufficient to report on an overall trend.

The defined parameters are intended to support an assessment of relative change over time and represents an initial suggestion for establishing preferred conditions. However, further discussion and refinement of the categories is required to reflect improved understanding of shoreline hardening and ecosystem impacts. These status and trend definitions apply to both the individual lake assessments and the overall assessment. The lake assessments are based on the available shoreline data for each lake, and the overall assessment is based on the available shoreline data for all of the Great Lakes. The Ecological Condition section below outlines some of the challenges with attempting to define reference conditions for hardened shorelines.

Endpoints and/or Targets

No net increase in the amount of hardened shoreline along any of the Great Lakes or connecting channels.

Sub-Indicator Purpose

- To assess the amount of shoreline altered by the construction of artificial shore structures, such as sheet piling, revetment, rip rap, and other erosion control and shore protection structures.
- To infer the potential harm to natural coastal processes, aquatic-dependent life, water quality, and shoreline habitat from conditions created by artificial shore structures.

Ecosystem Objective

Shoreline conditions should support natural and healthy coastal ecosystem habitats for aquatic and terrestrial plant and animal life, including rare and threatened species.

This sub-indicator best supports work towards both General Objective #5 and #9 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should, "support healthy and productive wetlands and other habitats to sustain resilient populations of native species" and "be free from other substances, materials, or conditions that may negatively impact the chemical, physical, or biological integrity of the Waters of the Great Lakes". Replacing natural shorelines with hardened and armoring structures alters natural coastal processes and the habitats created by these processes. Examples of the negative impacts from shoreline armoring include the erosion of coastal wetlands, increased wave energy reflected off of sheet piles leading to rapid rates of change to natural shorelines, and providing areas for the spreading of invasive species like *Dreissenid* mussels.

Measure

How to measure the linear distance of shorelines, often referred to as the coastal paradox, has often challenged coastal scientists and introduced various levels of uncertainty with any type of shoreline calculations. Factors that influence these calculations include the starting point of reach delineations, segment length (e.g. 100 meters or 1 kilometer), sinuosity, and the natural dynamics (erosion, deposition, water level fluctuations) associated with shorelines over time. In the past State of the Great Lakes Hardened Shorelines sub-indicator reports the use of one kilometer reaches have been used to quantify the amount of hardened shorelines present in the Great Lakes. However, without knowing the exact starting point of these reach delineations of various datasets for each of the Great Lakes there can be a shift in the classification between two different datasets for the same shoreline. To account for the positional changes of dynamic shorelines the use of a one kilometer by one kilometer Great Lakes shoreline reference grid was generated along the entirely the Great Lakes shoreline. This grid serves as a way to measure the linear shoreline distance within each grid cell and is not tied to any specific shoreline classification delineation method or dataset (Figure 8). This allows for the comparison of shorelines from different time periods, different datasets, and is agnostic of shoreline segmentation lengths. To maintain continuity with past reporting, the hardened shoreline percentage break values and classification categories (Shantz 2011) have been adopted and applied to the grid cell values. To determine the hardened shoreline classification category for each of the Great Lakes shoreline reference grid cells, the sum of hardened classified shorelines length within a grid cell is divided by the total length of shoreline within the grid cell to determine the percentage of hardened shorelines. Hardened shorelines, or shoreline armoring, as classified in this report include any placement of material used to armor the shoreline including rip rap, sheet pilings, bulkheads, revetments, and other ad hoc materials along the shore to offer protection from waves and water level changes. Below are the hardened shoreline classification categories:

- Highly protected (70-100% hardened)

- Moderately protected (40-70% hardened)
- Minor protection (15-40% hardened)
- No protection (< 15% hardened)

While the 2019 U.S. Great Lakes Hardened Shorelines Classification (NOAA 2019) does include artificial coastal perpendicular structures such as jetties, groins, breakwaters, piers, and docks, these features were not included in the hardened shorelines calculations. Additionally, islands were also excluded from the analysis, see Data Limitations.

Ecological Condition

Various ecological conditions are influenced by shoreline hardening including changes in coastal aquatic and terrestrial habitats, changes in littoral sediment transport, reductions in ecosystem services and changes in surface and ground water interactions (Gittman et al. 2016). The degree of negative impact to aquatic and terrestrial life and alterations to sediment transport rates in the coastal zone will vary depending on the structure type, design and condition including changes in water levels. Some types of hardened shoreline when not properly sited or engineered without consideration for the surrounding environment may lead to even more severe impacts.

Armoring of the shoreline is often performed to protect property from the impacts of waves and help to slow erosion, but do not necessarily protect against erosion from overland runoff or insufficient structure set back requirements, both of which could be addressed through other strategic management practices. These armoring structures can additionally be of concern if they are not properly installed and maintained over time. Proper planning, engineering, and design criteria for site specific placement are needed, but often armoring placement is rushed in response to rapid changes that are being seen in the Great Lakes. While some types of armoring can provide habitat for wildlife (see USACE ERDC 2012 and Gittman et al. 2016), many structures in the Great Lakes need retrofits or replacement to provide this type of habitat and require greater coordination and cost sharing. Working towards a blend of grey and green shoreline protection structures must be considered to address the increase in hardened shorelines that is currently being observed in the Great Lakes.

Since the hardened shorelines sub-indicator report in 2019, an interagency effort was completed to generate an updated baseline shoreline dataset for the United States portions of the remaining Great Lakes. In order to obtain a complete basin wide assessment of changes to the hardened shorelines, an effort is underway to digitize and create a baseline dataset for the remaining Canadian portion of the Great Lakes, please see the Additional Information section. The combined data from the Canadian and United States sides of Lake Erie and Lake Ontario and their respective connecting channels now allows for a more recent and complete picture of hardened shorelines within the lower portion of the Great Lakes. Additionally, the imagery used for the hardened shorelines analysis was collected just prior to the recent period of high water levels (~2017 to present) in all of the Great Lakes. We are seeing with the recent increase in water levels in all of the Great Lakes there has been a quick response by armoring of the shorelines to protect property, houses, and other critical coastal infrastructure.

Currently there is limited documentation present in the Great Lakes on addressing recommended shoreline hardening and armoring constraints and understanding long-term impacts at the basin wide and lake wide scales. Based on the hardened shoreline analysis of each lake basin for this report, Lake Huron, Lake Michigan and Lake Superior are classified as being in Good status while Lake Erie and Lake Ontario are classified as being Poor status based on a descriptive point of reference using the baseline Great Lakes (previously known as SOLEC) estimates of the extent and intensity of shoreline hardening (Shantz 2011). The challenge is defining an appropriate target value regarding shoreline hardening and understanding that each lake, and its connecting channels, are different and unique based on the ecosystem services provided within each basin. The current assessment categories only provide a general estimate of the extent and intensity of shoreline hardening and do not reflect an assessment of the ecosystem status and its sensitivity or impacts due to hardened shorelines in each lake.

Ecological Conditions in Lake Superior

Based on the United States data alone, Lake Superior currently has a status of Good at 5.3% (combined Highly and Moderately protected percentage). The shoreline armoring that has occurred in Lake Superior is primarily associated with the more urban and marine transportation settings including the Twin Ports of Duluth, Minnesota and Superior, Wisconsin; the Soo Locks near Sault Ste. Marie; and the various smaller harbors (Figure 3). However, with the recent increase in water levels there has been increased demand for shoreline structure permits. In Minnesota there has been a 1700% increase in permits from 2014-2020 with only two being issued in 2014 and 36 in 2020 (Perello et al. 2020). Prior to 2014, from 2010-2013, there were no permits applied for in Minnesota (Perello et al. 2020). Most of the new shoreline structures include breakwater, rip rap, seawalls, and gabion basket retaining walls. On the Wisconsin Lake Superior shoreline, a similar story has happened between 2010 and 2019 with a 1633% increase in shoreline structure permits (Perello et al. 2020). Most of the shoreline structures being placed in Wisconsin include bulkheads and rip rap. It will be interesting to see how the coastal ecosystem responds to this sudden increase in shoreline structures due to high water levels and how it adjusts accordingly when water levels do start to drop again. Similar in Michigan there have been reports on increased permitting for shoreline structures, but comprehensive numbers were not available for this report.

Ecological Conditions in Lake Michigan

Based on the hardened shorelines classification data for Lake Michigan its status is currently Good at 16.6% (combined Highly and Moderately protected percentage), however, it is less than four percent from changing to Fair status (20-30% of the shoreline is moderately or highly protected). Much of the hardened shoreline in Lake Michigan occurs in the southwest extending from Milwaukee, Wisconsin southward towards the Chicago metro area and also in the vicinity of Green Bay, Wisconsin (Figure 4). There are also smaller areas of hardened shoreline development associated with the various harbors along the eastern side of Michigan. While the hardened shoreline analysis completed for this report did not report on shoreline perpendicular structures, there are numerous structures present along the southwestern Lake Michigan shoreline (NOAA 2019). These shoreline perpendicular structures have been placed to help capture sediment moving southward along the shoreline in the hope of establishing pocket beaches which can have multiple benefits, including human recreation, open coastal space buffers and habitat restoration/protection. The recent period of high water levels has also caused significant changes to occur, impacting the remaining natural areas in southwestern Lake Michigan. The interplay between hardened shoreline structures and natural coastal ecosystems is of particular interest along Illinois Beach State Park where significant erosion has occurred south of the North Point Marina which is protected by rip rap and breakwaters (Figure 9). Project work is currently underway towards protecting the remaining panne wetlands present along this section of shoreline in Illinois.

Ecological Conditions in Lake Huron (including St. Marys River)

Based on the United States and Canadian data, Lake Huron currently has a status of Good at 11.7% (combined Highly and Moderately protected percentage). Much of the hardened shorelines located in Lake Huron occur within Saginaw Bay; near Tawas City, Michigan; along the St. Marys River; the southeastern portion of Georgian Bay in the Thornbury Basin; and the southern end of Lake Huron by Port Huron, Michigan, and Sarnia, Ontario (Figure 5). Much of the Canadian side of Lake Huron is classified as natural since little to no shoreline armoring has been applied due to the geologic setting where there are numerous coastal wetlands and inlets located between bedrock headlands.

When comparing the current status at 11.7% with the 2009/2011 status value of 2.7% (Table 4) there appears to be a 9% increase in the amount of hardened shoreline within Lake Huron and the St. Marys River Ecosystem. However, the comparison is not as straight forward given the resolution of the datasets used for each time period and the method by which the amount of hardened shoreline was analyzed and quantified, see Measure, Data Limitations, and Additional Information sections.

Ecological Conditions in Lake Erie (including St. Clair-Detroit River Ecosystem)

Based on the hardened shorelines classification data for Lake Erie and the St. Clair-Detroit Ecosystem its status is currently Poor at 56.9% (combined Highly and Moderately protected percentage). When comparing the current status at 56.9% with the 2009/2011 status value of 39% (Table 4) there has been a significant increase in the amount of hardened shoreline within Lake Erie and the St. Clair-Detroit River Ecosystem. However, the comparison is not as straight forward given the resolution of the datasets used for each time period and the method by which the amount of hardened shoreline was analyzed and quantified, see Measure, Data Limitations, and Additional Information sections. It does appear that the density of armoring along the shoreline is increasing when comparing values in Table 2 with Table 3. However, since the overall length of categorized shoreline decreased due to the refined shoreline delineation, there is uncertainty as to whether the identified change represents a true increase or a difference in dataset methodologies. Much of the highly armored shorelines are associated within the connecting channels of the St. Clair-Detroit River Ecosystem and the more urban and marine transportation settings of western, southern, and eastern Lake Erie (Figure 6). The amount of highly and moderately protected shorelines in Lake Erie and its connecting channels is of particular concern especially when trying to reconnect the lake with its coastal ecosystems. Given the shallow setting of Lake Erie and east to west orientation on the landscape it is prone to severe seiche impact events that can alter water levels several feet in the matter of hours. This can lead to coastal inundation and erosion which has resulted in the amount of shoreline armoring efforts present within this geography. Finding alternative "softer" protection solutions like nature-based shoreline protection or developing resilient shorelines through the Systems Approach to Geomorphic Engineering community of practice (SAGE 2021) are potential ways to work towards improving hardened shoreline trends in Lake Erie and the St. Clair-Detroit Ecosystem.

Ecological Conditions in Lake Ontario (including Niagara River and International section of the St. Lawrence River)

Based on the hardened shorelines classification data for Lake Ontario, the Niagara River, and International section of the St. Lawrence River its status is currently Poor at 40.6% (combined Highly and Moderately protected percentage). When comparing the current status at 40.6% with the 2009/2011 status value of 21% (Table 4) there has been a significant increase in the amount of hardened shoreline within Lake Ontario and its associated connecting channels. However, the comparison is not as straight forward given the resolution of the datasets used for each time period and the method by which the amount of hardened shoreline was analyzed and quantified, see Measure, Data Limitations, and Additional Information sections. It does appear that the density of armoring along the shoreline is increasing when comparing values in Table 2 with Table 3. However, since the overall length of categorized shoreline decreased due to the refined shoreline delineation, there is uncertainty as to whether the identified change represents a true increase or a difference in dataset methodologies. Much of the highly hardened areas appear in western Lake Ontario from Burlington, Ontario northeast towards Toronto and its surrounding coastal communities with some intermixing of highly protected and minor protection along Lake Ontario's southern shoreline in New York (Figure 7). After the flooding that occurred in Lake Ontario in 2017 several homeowners and communities implemented shoreline armoring to protect properties from future coastal inundation.

Linkages

The hardened shoreline sub-indicator can be directly linked to other sub-indicators currently used to assess the Great Lakes basin. Those sub-indicators/indicators are:

Coastal Wetlands sub-indicators **–** the placement of hardened shorelines have a direct impact to coastal wetlands across the Great Lakes including modifications to fish spawning and feeding habitats, changes in coastal processes, and alter shorebird habitats based on physical modification to the shorelines. These changes from natural to hardened shorelines can also weaken aquatic habitat connectivity.

Watersheds Impacts and Climate Trends sub-indicators **–** the hardening of shorelines can alter the coastal land water interface significantly where large scale anthropogenic shoreline modifications have occurred. This can lead to further erosion of unprotected lands and properties, and changes in local coastal landcover types which may influence coastal microclimates. The armoring of shorelines can also diminish littoral drift, impacting river mouths and regional sediment management.

Cladophora sub-indicator (Nutrients and Algae) **–** the placement of nearshore large rip rap, boulders, and similar coarse materials used to harden the shoreline may allow for the development of Cladophora in areas that may not have been supportive of Cladophora growth (e.g. soft substrates) in the past by providing a solid substrate upon which the algae can attach.

Dreissenid Mussels sub-indicator (Invasive Species) **–** similar to Cladophora, the placement of nearshore large rip rap, boulders, and similar coarse materials used to armor the shoreline may allow for zebra mussels to inhabit areas that may not have supported zebra mussels in the past by providing a solid substrate upon which the mussels can attach.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin		X		
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	USA-Y CAN-Y	United States Great Lakes Shoreline Data: https://coast.noaa.gov/digitalcoast/data/hardened-shorelines.html *Canadian data is available per request and is currently being worked into the Open Data Catalogue, see Information Sources*		

Data Limitations

1. The geographic scale for the updated information covers all of Lake Ontario (including the Niagara River and International section of the St. Lawrence River), Lake Erie (including St. Clair-Detroit River Ecosystem), Lake Michigan and Lake Huron (including St. Marys River). Lake Superior is only fully covered on the United States side and will be developed and classified at a later time on the Canadian side.
2. All island shorelines were removed prior to performing the data analysis. The inclusion of islands in the future is something that should be discussed by the State of the Great Lakes reporting team in forthcoming report updates by determining which islands should or should not be included in future calculations.
3. The dependence on orthophotography and other imagery sources, while high in resolution, are only valid by their production date. It is expected that all imagery sets currently used will be updated on a five-year interval and classified accordingly to a well documented shoreline classification schema.
4. While the use of a one kilometer by one kilometer Great Lakes shoreline reference grid is useful to quantify the hardened shoreline classification (see Measure section), this grid will need to be updated to include additional cells, or removal of cells, for future sub-indicator report calculations. The grid was generated using the Great Lakes and St. Lawrence Albers projection (EPSG: 3175) and will be shared with the State of the Great Lakes reporting team for use in forthcoming report updates.
5. The procedure for identifying hardened shorelines was applied consistently on both the Canadian and U.S. shorelines. However, the identification and interpretation of hardened shorelines was influenced by the imagery availability and resolution which varied greatly along certain areas of the Canadian shoreline. The specific age of input imagery used for individual shoreline reaches has been attributed to the hardened shoreline vector dataset.
6. All perpendicular shoreline structures, including jetties and groins (groynes), were removed to help with ensuring that shoreline lengths were not over calculated, however, this did leave some parts of the hardened shoreline incomplete and with small gaps. Similarly, small inlets and slips were also removed leaving small gaps.
7. There continues to be introduced uncertainty with regards to shoreline length and differences due to variations in water levels at the time of imagery collection and classification. Since the sub-indicator is based on a relative difference in the percent of shoreline within various categories, it is still possible to make some comparisons, especially with the use of the Great Lakes shoreline reference grid (Data Limitations #5 and Measure section).

Additional Information

In 2019 a new vector shoreline of the United States Great Lakes Shorelines was released (NOAA 2019). This dataset was previously mentioned in the prior hardened shorelines sub-indicator report, developed and used in this report, and can serve as a baseline moving forward. The dataset consists of shoreline segments classified as either artificial or natural, along with structure type and condition. NOAA's Office for Coastal Management, in partnership with the US Army Corps of Engineers, contracted with Tetra Tech in the development of a higher resolution, applicable to a mapping scale of 1:2,000. The data were created by digitizing shoreline using National Agriculture Imagery Program imagery from 2014 through 2017 and comparing it with oblique imagery, lidar, and other ancillary datasets. The classification of the shorelines is based upon the Flood and Erosion Prediction System classification schema (Stewart 2002, Baird 2005, and AECOM 2012) with some modifications to account for additional shoreline classification needs. The classification schema and methodology used to create this dataset are well documented and efforts to create new or update existing datasets should ensure that classification methodologies are similar to these efforts to ensure continuity in reporting in the future.

The 2019 dataset allows for comparison on a lake-by-lake basis and also allows for a break down of shorelines by state and county to further understand local characterization of the shorelines within the United States. The dataset also further classifies what types of structures are in place (jetty, rip rap, sheet piling) and general condition (good, moderate, poor). This level of detail could help further the understanding on what types of impacts to shore habitat are associated with the different types of artificial shore structures. Oblique imagery and lidar have also been collected for several years now along the Great Lakes shoreline by federal, provincial and state agencies. These data could further be utilized together with the hardened shorelines data to understand the dynamics and connections between hardened and natural shorelines. These data should be considered in future reporting efforts, where appropriate, to help develop new metrics that further explain shoreline dynamics across the Great Lakes.

The Canadian shoreline hardening data was completed primarily using orthophotography. Environment and Climate Change Canada acquired these images through the Land Information Ontario. Land Information Ontario coordinates public and private sector organizations to collect high resolution aerial imagery for Ontario through a partnership funding model. This aerial project was part of a five-year plan (2013-2017) to acquire 20 cm resolution, leaf-off imagery across the province. ECCC acquired three sets of orthophotography: South Central Ontario Orthophotography (SCOOP) 2013; Digital Raster Acquisition Project Eastern Ontario (DRAPE) 2014; and Southwestern Ontario Orthophotography (SWOOP) 2015.

These Canadian orthophotography sets were used to digitize line segments as 'natural' or 'hardened', and classify each line segment based on the exposure categories of 'lake', 'sheltered', or 'connecting channel'. For example, classify line segments within embayments and tributaries as having 'sheltered' exposure and line segments exposed to lake wave energy as having 'lake' exposure. Shoreline hardening is one of several measures being completed under the Canadian Baseline Coastal Habitat Survey. The current schedule for completion of each lake is as follows: Lake Erie (2020); Lake Ontario (2021); Lake Huron (2022); and Lake Superior (2023). A completed dataset of hardened shoreline for the entire Canadian Great Lakes is expected to be completed in 2023.

Acknowledgments

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Figure 7. Map of Lake Ontario, including Niagara River and International section of the St. Lawrence River, hardened shoreline protection status classification. Note islands were not included in the hardened shorelines analysis. Sources: Environment and Climate Change Canada (2020 *in prep*), National Oceanic and Atmospheric Administration (2019). Basemap: Open Street Map.

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Figure 9. Images of Illinois Beach State Park. Image on left is from 2008, image in middle is from 2018, and image on right is bathymetry elevation surface difference between 2018 and 2008 with red showing areas of erosion and blue showing areas of deposition in the littoral zone. Notice the erosion nick points downstream of the hardened shorelines. Sources: Google Earth imagery and bathymetric surfaces developed from Joint Airborne Lidar Bathymetry Technical Center of Expertise (JALBTCX) topobathy lidar. Credit: NOAA Office for Coastal Management.

Last Updated

State of the Great Lakes 2022 Report

Table 1. Original Baseline Great Lakes/SOGL hardened shoreline classification used in the 2011 State of the Great Lakes Hardened Shorelines sub-indicator assessment based on information provided in 2009 SOGL indicator report. Source: National Oceanic and Atmospheric Administration (1997).

Lake / Connecting Channel	Highly Protected (%) (>70% protected)	Moderately Protected (%) (40-70% protected)	Minor Protection (%) (15-40% protected)	No Protection (%) (<15% protected)	Non-structural Protection (%)	Unclassified (%)	Total Shoreline (km)
Lake Superior	3.1	1.1	3	89.4	0.03	3.4	5080
St. Marys River	2.9	1.6	7.5	81.3	1.6	5.1	707
Lake Michigan	8.6	2.9	30.3	57.5	0.1	0.5	2713
Lake Huron	1.5	1	4.5	91.6	1.1	0.3	6366
St. Clair River	69.3	24.9	2.1	3.6	0	0	100
Lake St. Clair	11.3	25.8	11.8	50.7	0.2	0.1	629
Detroit River	47.2	22.6	8	22.2	0	0	244
Lake Erie	20.4	11.3	16.9	49.1	1.9	0.4	1608
Niagara River	44.3	8.8	16.7	29.3	0	0.9	184
Lake Ontario	10.2	6.3	18.6	57.2	0	6.2	1772
St. Lawrence River	12.6	9.3	17.2	54.7	0	6.2	2571

Table 2. Modified 2009 Baseline Great Lakes/SOGL hardened shoreline classification used in the 2011 State of the Great Lakes Hardened Shorelines sub-indicator assessment with connecting channels added in with their respective lake as reported in the Lake-by-Lake Assessment. Source: National Oceanic and Atmospheric Administration (1997), modified (2021).

Lake / Connecting Channel	Highly Protected (%) (>70% protected)	Moderately Protected (%) (40-70% protected)	Minor Protection (%) (15-40% protected)	No Protection (%) (<15% protected)	Non-structural Protection (%)	Unclassified (%)	Total Shoreline (km)
Lake Superior	3.1	1.1	3	89.4	0	3.4	5080
Lake Michigan	8.6	2.9	30.3	57.5	0.1	0.5	2713
Lake Huron (including St. Marys River)	1.6	1.1	4.8	90.6	1.1	0.8	7073
Lake Erie (including St. Clair-Detroit River Ecosystem)	22.6	16.4	14.2	45.2	1.2	0.3	2581
Lake Ontario (including Niagara River and International section of the St. Lawrence River)	12.9	8.1	17.7	54.6	0	6	4527

Table 3. Updated 2021 Great Lakes hardened shoreline classification values. Sources: Environment and Climate Change Canada (2020 *in prep*), National Oceanic and Atmospheric Administration (2019), Zuzek Inc. (2018). Note: Lake Superior is incomplete at this time on the Canadian side.

Lake / Connecting Channel	Highly Protected (%) (>70% protected)	Moderately Protected (%) (40-70% protected)	Minor Protection (%) (15-40% protected)	No Protection (%) (<15% protected)	Non-structural Protection (%)	Unclassified (%)	Total Shoreline (km)
Lake Superior*	2.6%	2.7%	3.9%	90.8%	0.0%	0.0%	1588
Lake Michigan	11.0%	5.6%	5.8%	77.6%	0.0%	0.0%	2353
Lake Huron (including St. Marys River)	6.9%	4.8%	6.3%	82.0%	0.0%	0.0%	4324
Lake Erie (including St. Clair-Detroit River Ecosystem)	46.0%	10.9%	8.3%	34.7%	0.0%	0.0%	1633
Lake Ontario (including Niagara River and International section of the St. Lawrence River)	24.9%	15.8%	16.6%	42.7%	0.0%	0.1%	2224
TOTAL	**16.3%**	**7.6%**	**8.1%**	**68.0%**	**0.0%**	**0.0%**	**12,122**

* Hardened shoreline delineation and classification incomplete on Canadian side.

Table 4. Comparison between Modified 2009 Baseline Great Lakes/SOGL hardened shoreline classification and newer shoreline classification values. Sources: 2009 Baseline - National Oceanic and Atmospheric Administration (1997), modified (2021), 2021 Update - Environment and Climate Change Canada (2020 *in* prep), National Oceanic and Atmospheric Administration (2019), Zuzek Inc. (2018). Note: Lake Superior and Lake Huron are incomplete at this time on the Canadian side and islands are also not included.

Lake / Connecting Channel	2009/2011 Combined Highly and Moderately Protected %	2021 Combined Highly and Moderately Protected %
Lake Superior*	4.2%	5.3%
Lake Michigan	11.5%	16.6%
Lake Huron (including St. Marys River)	2.7%	11.7%
Lake Erie (including St. Clair-Detroit River Ecosystem)	39.0%	56.9%
Lake Ontario (including Niagara River and International section of the St. Lawrence River)	21.0%	40.6%

* Hardened shoreline delineation and classification incomplete on Canadian side.

Figure 1. Map of Great Lakes hardened shoreline classification datasets used for Lake-by-Lake Assessment as of November 2021. Note that classification work in Lake Superior is still in the process of being developed on the Canadian side of the Great Lakes and islands were not included in the hardened shorelines analysis. Sources: Environment and Climate Change Canada (2020 *in prep*), National Oceanic and Atmospheric Administration (2019), Zuzek Inc. (2018). Basemap: Open Street Map.

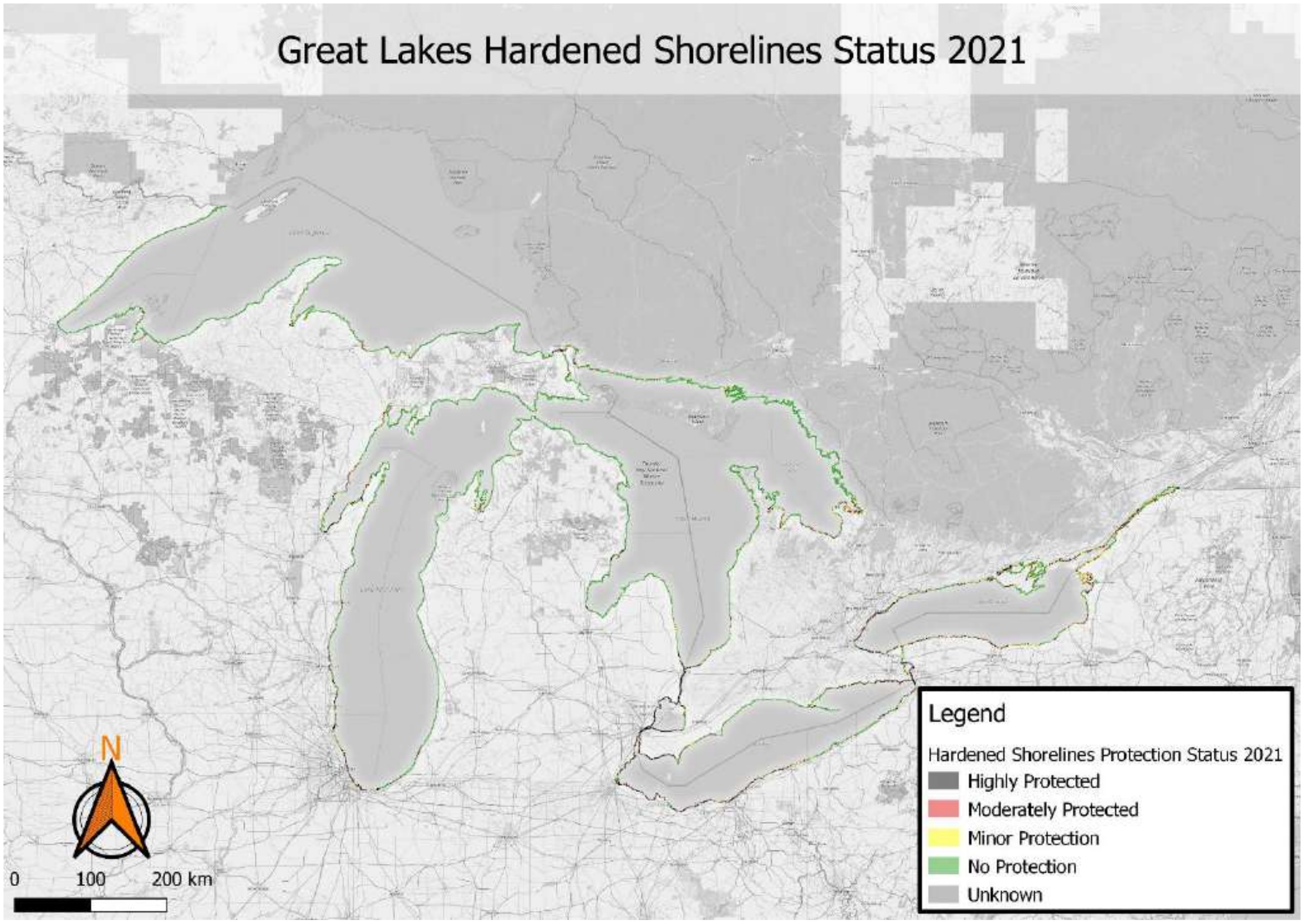


Figure 2. Map of Great Lakes hardened shoreline protection status classification. Note that classification work for Lake Superior is still in the process of being developed on the Canadian side of the Great Lakes and islands were not included in the hardened shorelines analysis. Sources: Environment and Climate Change Canada (2020 *in prep*), National Oceanic and Atmospheric Administration (2019), Zuzek Inc. (2018). Basemap: Open Street Map.

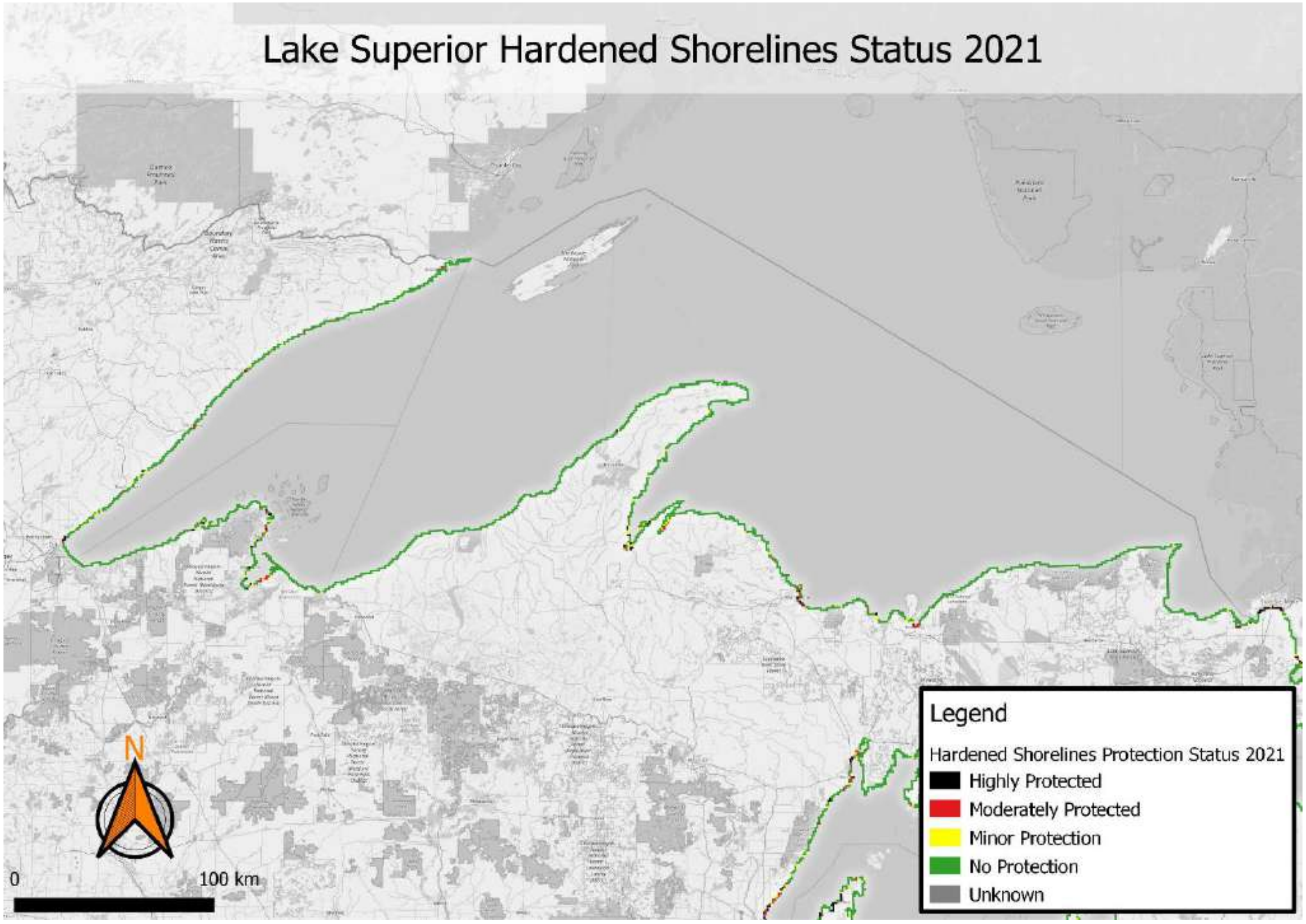


Figure 3. Map of Lake Superior hardened shoreline protection status classification. Note Lake Superior is still in the process of being developed on the Canadian side of the Great Lakes and islands were not included in the hardened shorelines analysis. Sources: National Oceanic and Atmospheric Administration (2019). Basemap: Open Street Map.

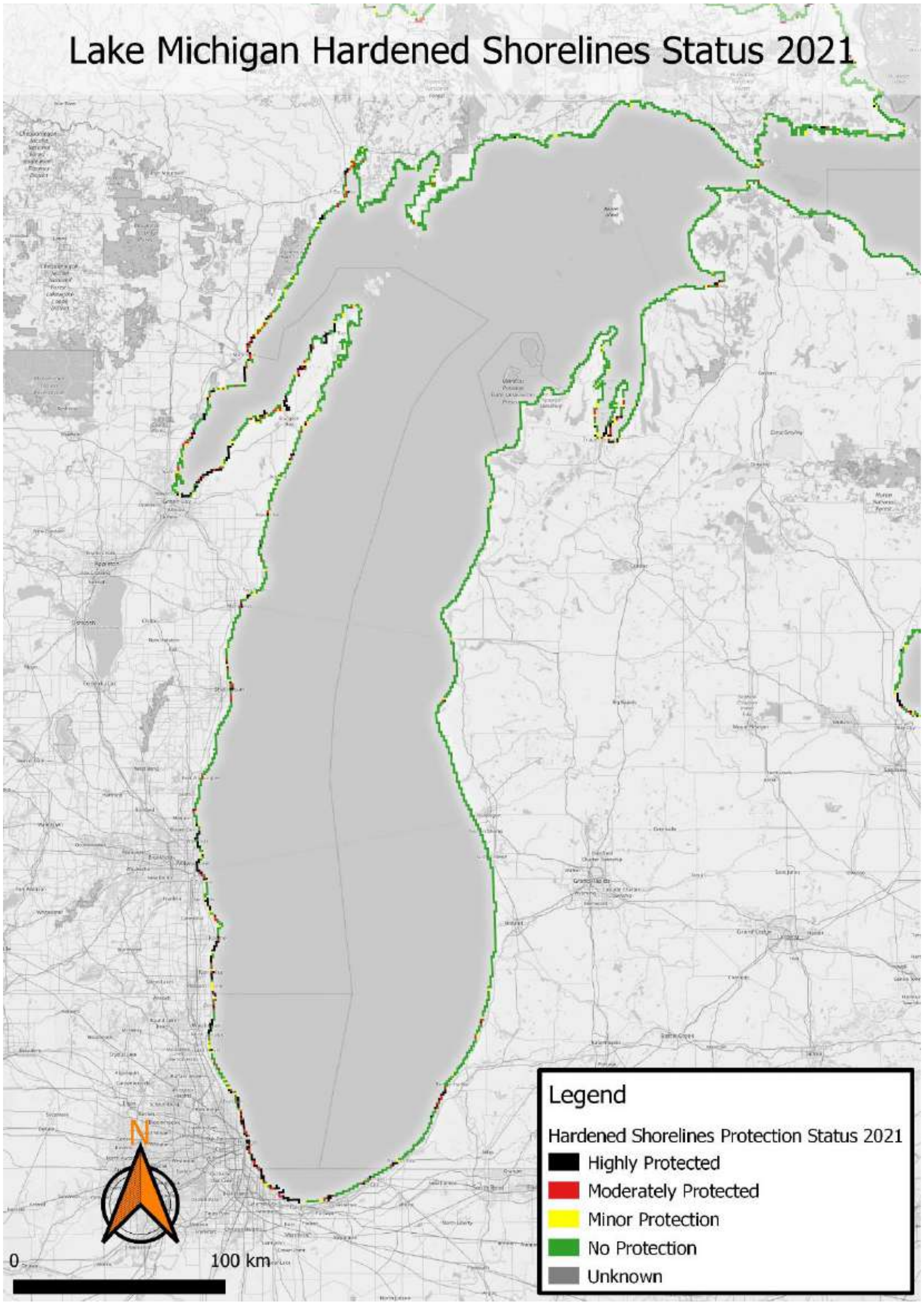


Figure 4. Map of Lake Michigan hardened shoreline protection status classification. Note islands were not included in the hardened shorelines analysis. Sources: National Oceanic and Atmospheric Administration (2019). Basemap: Open Street Map.

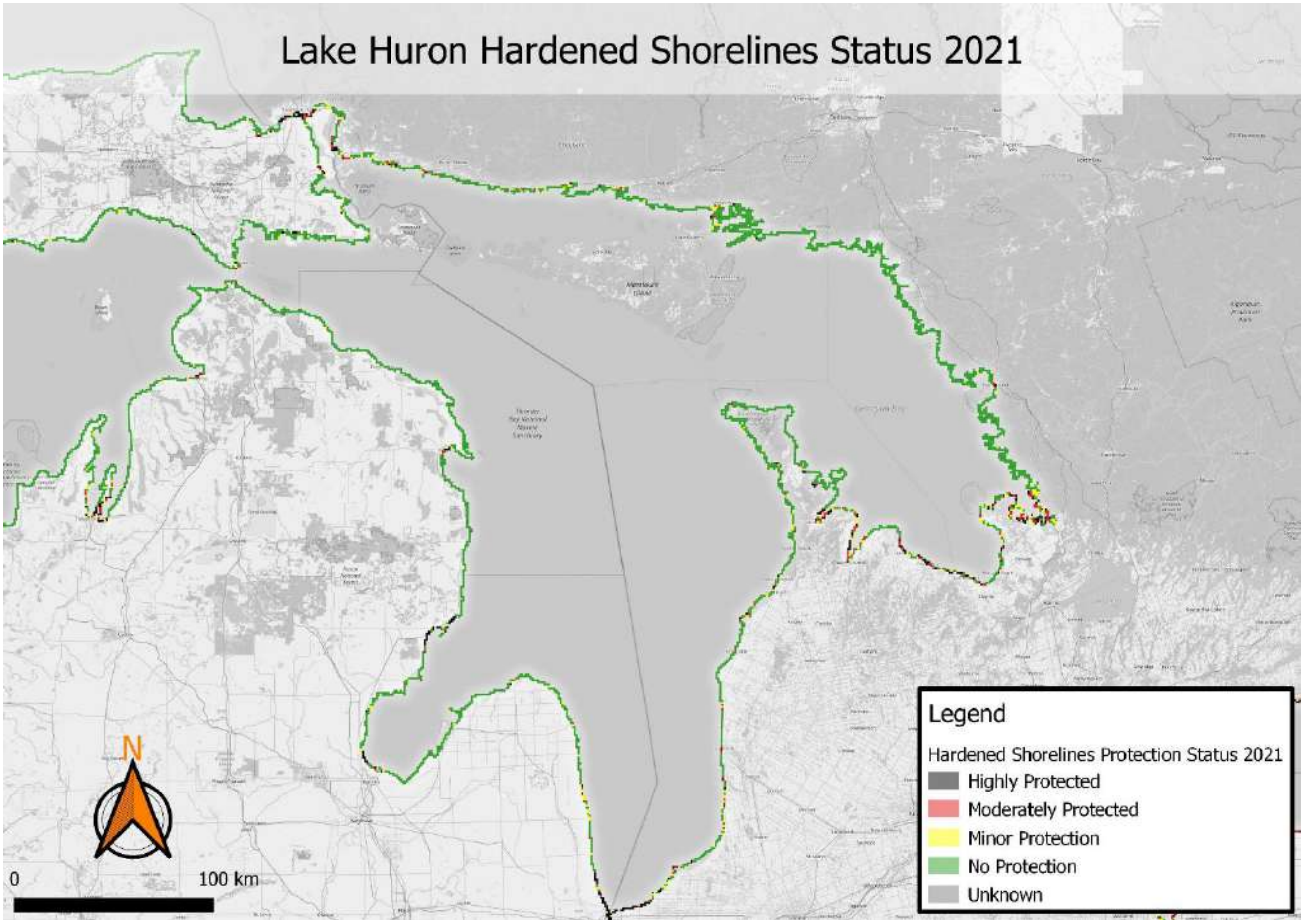


Figure 5. Map of Lake Huron, including St. Marys River, hardened shoreline protection status classification. Note islands were not included in the hardened shorelines analysis. Sources: Environment and Climate Change Canada (2020 *in* prep), National Oceanic and Atmospheric Administration (2019), Zuzek Inc. (2018). Basemap: Open Street Map.

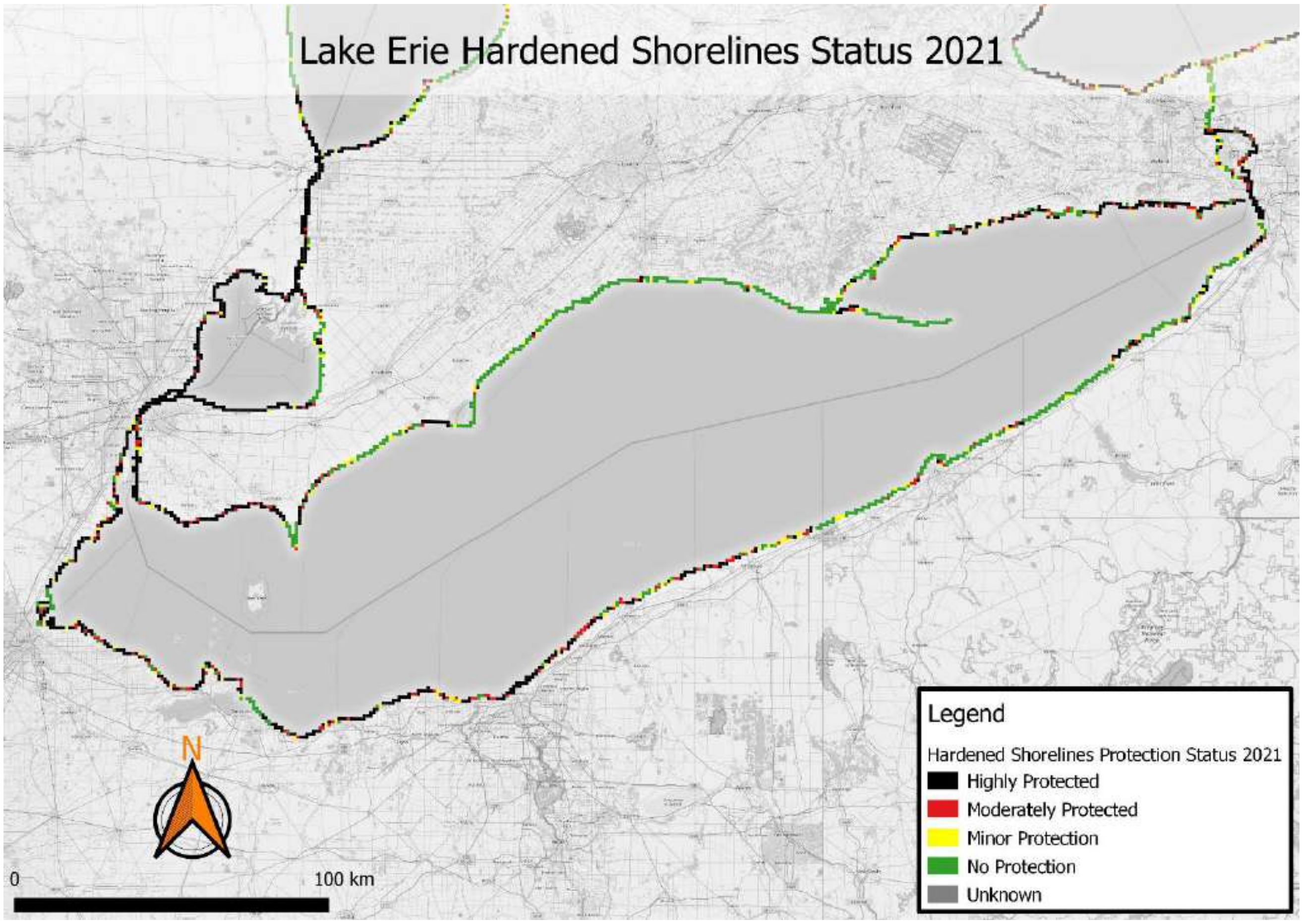


Figure 6. Map of Lake Erie, including St. Clair-Detroit River Ecosystem, hardened shoreline protection status classification. Note islands were not included in the hardened shorelines analysis. Sources: Environment and Climate Change Canada (2020 *in prep*), National Oceanic and Atmospheric Administration (2019), Zuzek Inc. (2018). Basemap: Open Street Map.

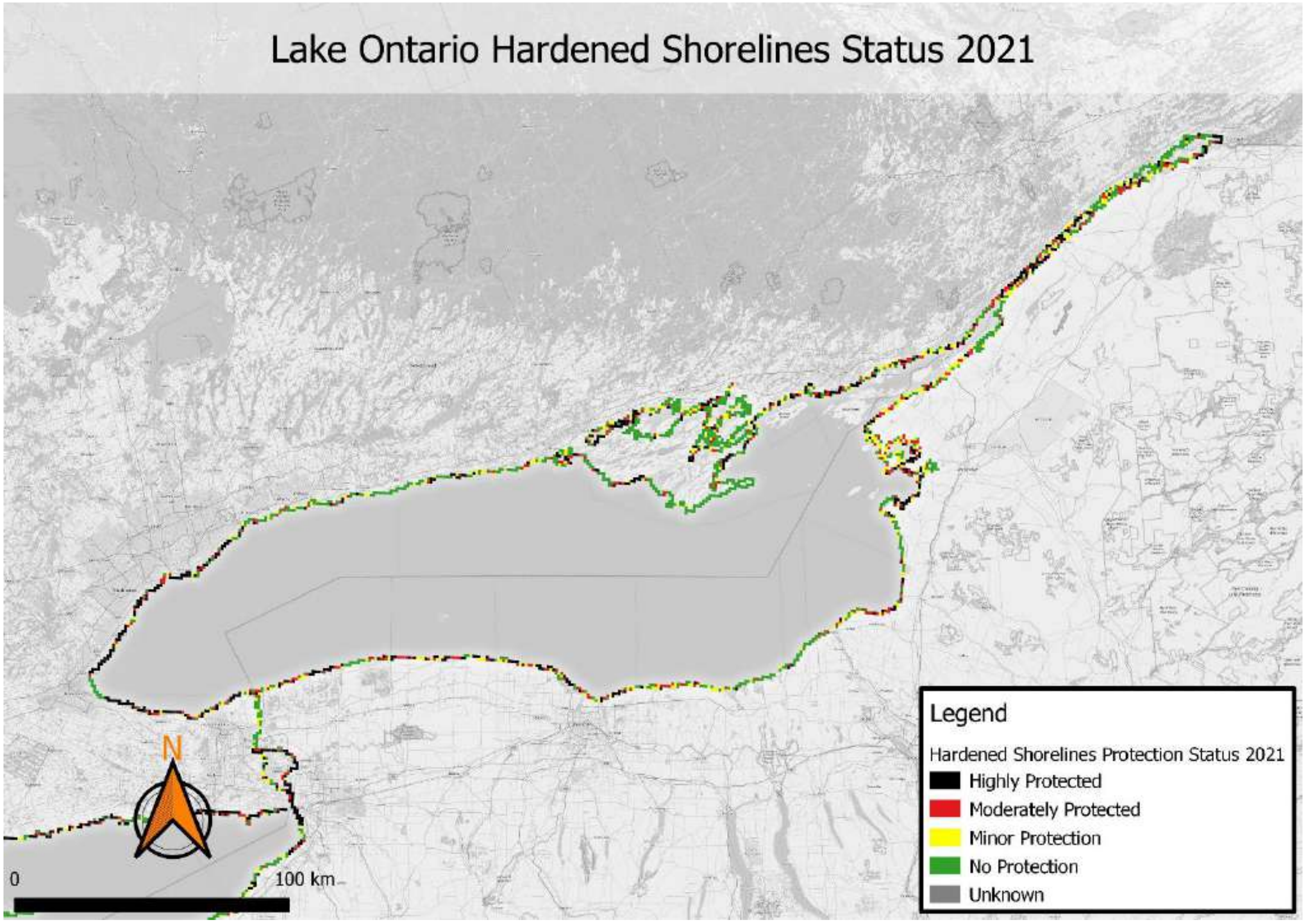
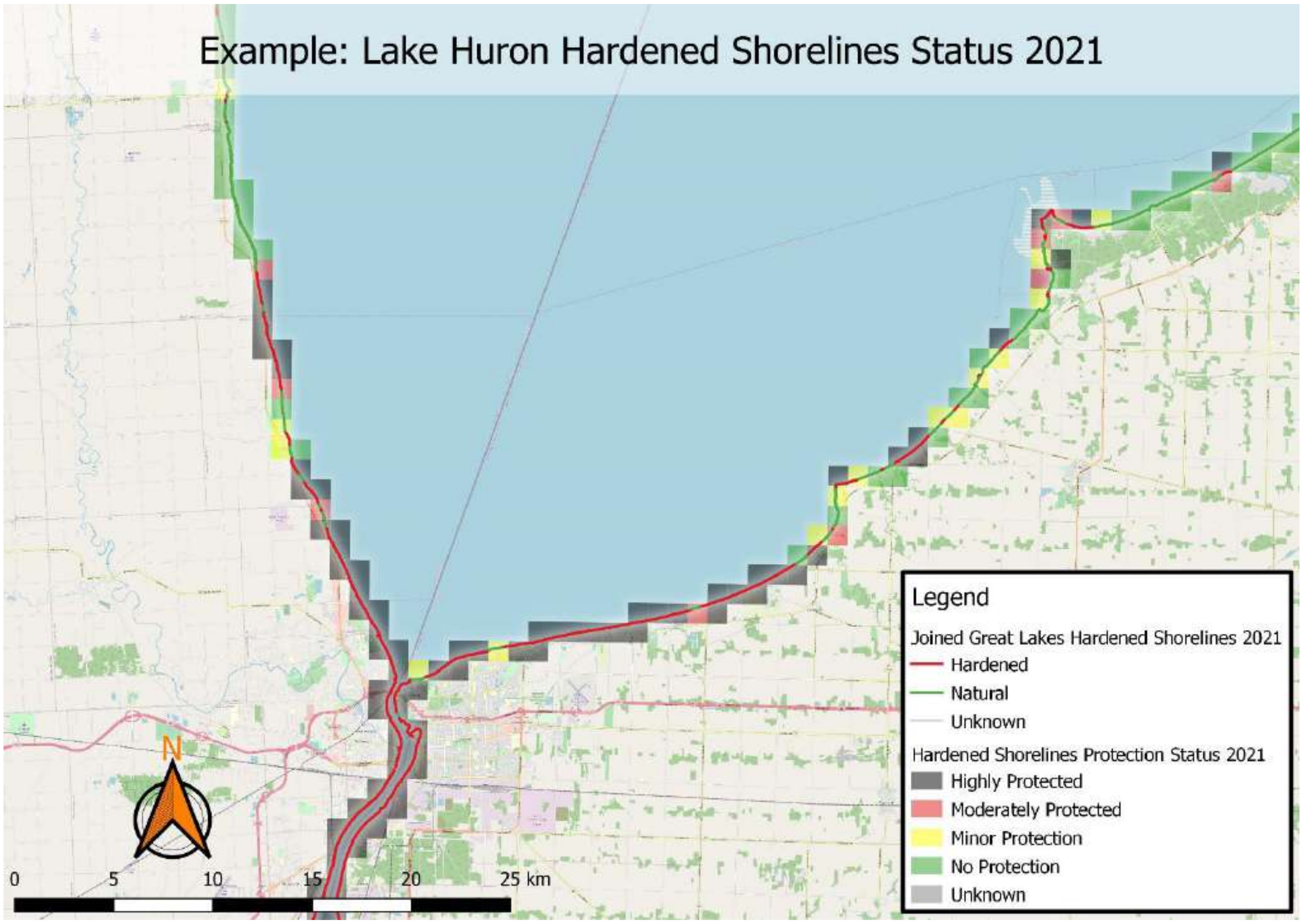


Figure 7. Map of Lake Ontario, including Niagara River and International section of the St. Lawrence River, hardened shoreline protection status classification. Note islands were not included in the hardened shorelines analysis. Sources: Environment and Climate Change Canada (2020 *in prep*), National Oceanic and Atmospheric Administration (2019). Basemap: Open Street Map.



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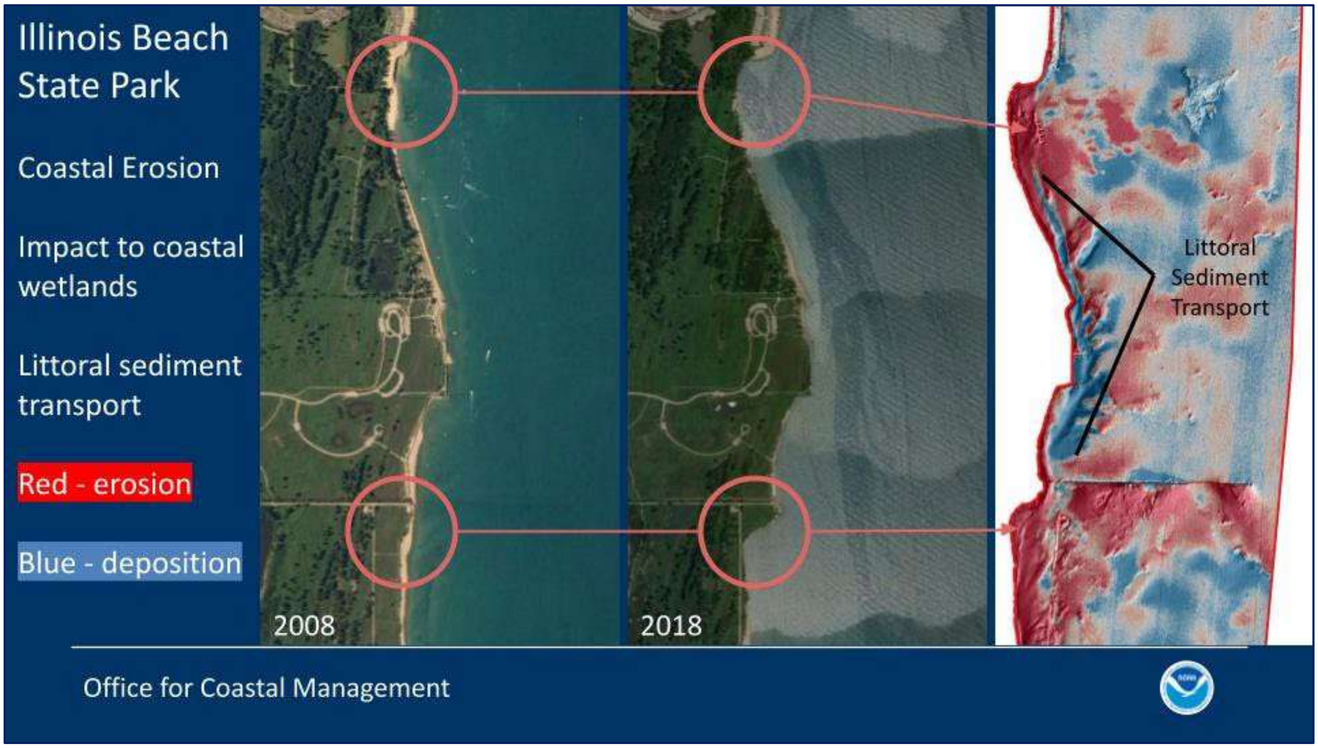

Figure 9. Images of Illinois Beach State Park. Image on left is from 2008, image in middle is from 2018, and image on right is bathymetry elevation surface difference between 2018 and 2008 with red showing areas of erosion and blue showing areas of deposition in the littoral zone. Notice the erosion nick points downstream of the hardened shorelines. Sources: Google Earth imagery and bathymetric surfaces developed from Joint Airborne Lidar Bathymetry Technical Center of Expertise (JALBTCX) topobathy lidar. Credit: NOAA Office for Coastal Management.

Sub-Indicator: Water Quality in Tributaries

Overall Assessment – Canadian Only

Status: Fair

Trends:

10-Year Trend (2010-2019): Unchanging

Long-term Trend (1970 - 2019): Undetermined

Rationale: The overall status is based on the Water Quality Index (WQI) which was computed for 72 Canadian tributaries to the Great Lakes spanning a gradient of land cover/uses from forestry to agricultural and urban to determine current water quality status. The WQI is based on eight water quality constituents (ammonia, chloride, copper, iron, nitrate, nitrite, phosphorus and zinc). The overall water quality status of tributaries to the Great Lakes varies considerably in this large geographic region but can be described as Fair if taking the average WQI from all 72 tributaries (WQIavg=63, WQIrange=16-100). This overall assessment could be biased since Lake Superior and the northern portion of Lake Huron is currently underrepresented. Broken down by category, 18% of the tributaries were categorized as having Good water quality, 63% as Fair, and 19% as Poor.

A separate analysis for a different selection of water quality sites (n=133) was conducted analyzing 10-year- and long- term trends for three water quality constituents (nitrate, total phosphorus and chloride). The 10-year trend is reported as Unchanging. Trend results for three parameters for the entire Great Lakes basin indicate undetermined trends for this time period at most sites. The long-term trend is reported as undetermined since the trends of the various components are mixed, with the majority of trend sites showing increasing chloride concentrations, no trend in nitrate concentrations and decreasing total phosphorus concentrations, however, it should be noted that Lake Superior and Lake Huron are underrepresented.

Lake-by-Lake Assessment

Lake Superior – Canadian Only

Status: Undetermined

10-Year Trend: Undetermined

Long-term Trend: Undetermined

Rationale: The average WQI score for three tributaries (down from four tributaries in the previous State of the Great Lakes (SOGL) report) was 79. WQI scores ranged from 70 to 92 (Fair to Good). There were only a few sites monitored and all three sites were clustered near Thunder Bay, therefore it is difficult to assign an overall lake assessment value based on so few tributaries.

Lake Michigan – Not Applicable (assessment is Canadian Only)

Status: Not Applicable

10-Year Trend: Not Applicable

Long-term Trend: Not Applicable

Rationale: No tributaries to Lake Michigan are monitored by the Ontario Provincial Water Quality Monitoring Network (PWQMN)

Lake Huron (including St. Mary's River) – Canadian Only

Status: Fair

10-Year Trend: Unchanging

Long-term Trend: Undetermined

Rationale: The average WQI score for 20 tributaries was 76. WQI scores ranged from 55 to 86 (Fair to Good). The 10-year trend is reported as unchanging: majority of sites show no trend for both total phosphorus and nitrate while just under half of trend sites show an increasing chloride trend. Long-term trends are undetermined since the trends are mixed: most sites show a decreasing trend for total phosphorus, no trend for nitrate, and an increasing trend for chloride concentrations.

Lake Erie (including St. Clair-Detroit River Ecosystem) – Canadian Only

Status: Poor

10-Year Trend: Unchanging

Long-term Trend: Undetermined

Rationale: The average WQI score for 15 tributaries was 43. WQI scores ranged from 16 to 68 (Poor to Fair). The 10-year trend is reported as unchanging since the majority of trend sites are showing no trend for all three parameters tested (total phosphorus, nitrate, and chloride). Long-term trend is reported as undetermined since the trends are mixed: the majority of trend sites show decreasing trends in total phosphorus concentrations, no trend in nitrate concentrations, and increasing trends in chloride concentrations.

Lake Ontario (including Niagara River and International section of the St. Lawrence River) – Canadian Only

Status: Fair

10-Year Trend: Unchanging

Long-term Trend: Undetermined

Rationale: The average WQI score for 34 tributaries was 62. WQI scores ranged from 27 to 100 (Poor to Good). The 10-year trend is reported as unchanging: majority of sites show no trend for both total phosphorus and nitrate while just over half of trend sites show an increasing chloride trend. The long-term trend is reported as undetermined since the trends are mixed: the majority of trend sites show decreasing total phosphorus concentrations, no trend for nitrate concentrations, and increasing chloride concentrations.

Supporting Information

Status Assessment Definitions

The calculated WQI values fit into five categories that describe water quality conditions as used by the Canadian Council of Ministers of the Environment (CCME):

Excellent (95-100);

Good (80-94);

Fair (65-79);

Marginal (45-64); and

Poor (0-44).

For this sub-indicator, the five original categories developed by CCME were dissolved into three descriptive categories:

Good: 80-100

Fair: 45-79

Poor: 0-44

Trend Assessment Definitions

Improving: Metrics show a change toward more acceptable condition.

Unchanging: Metrics generally show no overall change in condition.

Deteriorating: Metrics show a change away from acceptable condition.

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets

Desirable outcomes are the absence of undesirable water quality conditions in streams. The Water Quality Index (WQI) score is from 0 to 100 with rankings for poor, fair and good. The category ranges describe sites where the water quality complies with criteria virtually all of the time (Good) or hardly any of the time (Poor).

Sub-Indicator Purpose

The purpose of this sub-indicator is to communicate water quality status relative to guidelines based on eight water quality parameters and also to determine short-term and long-term trends in three water quality parameters to support the evaluation of aquatic ecosystem health in Great Lakes tributaries.

Ecosystem Objective

The surface waters in the Great Lakes Basin should be of a quality that is protective of aquatic life and healthy aquatic ecosystems.

This sub-indicator best supports work towards General Objective #6 and #4 of the 2012 Great Lakes Water Quality Agreement. General Objective #6 states that the Waters of the Great Lakes should "be free from nutrients that directly or indirectly enter the water as a result of human activity, in amounts that promote growth of algae and cyanobacteria that interfere with aquatic ecosystem health, or human use of the ecosystem". General Objective #4 states that the Waters of the Great Lakes should "be free from pollutants in quantities or concentrations that could be harmful to human health, wildlife, or aquatic organisms, through direct exposure or indirect exposure through the food chain".

Measure

Current status

Water Quality Index (WQI) calculations for Ontario tributaries to the Great Lakes were computed using monitoring data from the Provincial Water Quality Monitoring Network (Figures 1-2) (PWQMN; https://data.ontario.ca/dataset/provincial-stream-water-quality-monitoring-network). The WQI may be calculated independently for U.S. tributaries if data are available.

The WQI provides a mathematical framework for synthesizing water quality monitoring results for multiple samples and parameters into a single value representing overall water quality conditions at a given site. The WQI is based on three measures (factors) of compliance with water quality criteria (guidelines and objectives) for the protection of aquatic life.

Scope: measures the percentage of the number of parameters that comply with water quality criteria

Frequency: measures the percentage of individual water quality tests that comply with criteria

Magnitude: measures by how much criteria are exceeded

The three factors are combined into a single unit-less value between 0 and 100 where higher numbers indicate better water quality. The WQI is computed using the Canadian Council of Ministers of the Environment's Water Quality Index (v. 1.2; CCME 2011a), which is described in detail in CCME (2001a, b).

For the calculation of the WQI, the water quality results are compared with guidelines from the Canadian Council of Ministers of the Environment (CCME)'s Water Quality Guidelines for the Protection of Aquatic Life (CCME, 2011b) or, in the absence of CCME Guidelines, the Ontario Interim Provincial Water Quality Objectives (PWQO) (i.e., for total phosphorus) (OMOE, 1994) (Table 1). All guidelines except for total phosphorus are for the protection of aquatic life whereas the guideline for total phosphorus is to avoid excessive plant growth in rivers and streams.

While the specific parameters that are selected for use in the calculation of the WQI and the number of parameters included may vary by jurisdiction at the discretion of the water quality manager, a minimum of four parameters that are sampled at least four times per year is recommended by CCME (2001) for calculating the WQI. Considerations for selecting parameters should include:

1. If parameters are relevant to the water body that is being assessed;

2. Whether the parameters are comparable to a water quality criterion such as the CCME's Water Quality Guidelines for the Protection of Aquatic Life;
3. If an appropriate number of samples (measurements) have been collected at each site at a suitable frequency for each parameter; and
4. If it is possible to maintain consistency in calculating and reporting the WQI within and between jurisdictions.

The parameters selected to calculate the WQI for this report will be at the discretion of the agencies collecting and analyzing the data, considering the above and ensuring consistency with criteria used in previous SOGL reports. The index was calculated at sites with four years of data and a minimum of 10 observations for the following eight parameters: ammonia (unionized), chloride, copper, iron, nitrates, nitrites, phosphorus, and zinc. These eight parameters were selected because they were consistently monitored at all sites used in this assessment and can all be compared to water quality criterion. These eight parameters also reflect a range of issues/stressors in the watershed. Nutrients (nitrate, nitrite, un-ionized ammonia and total phosphorus) are included as higher concentrations can be indicative of both urban and agricultural influences and can contribute to nutrient enrichment. Chloride is an important indicator of urban disturbance in watersheds; the main anthropogenic source is the application of road salts during the winter months. Elevated levels of metals (copper, iron, zinc) can be toxic to aquatic organisms and are usually indicative of urban / industrial sources in the watershed. Inland stream water quality results for these parameters were acquired from the Ontario Provincial Water Quality Monitoring Network (PWQMN) (OMOE, 2013). For the calculation of the WQI, the water quality results are compared with guidelines from the Canadian Council of Ministers of the Environment (CCME)'s Water Quality Guidelines for the Protection of Aquatic Life (CCME, 2011b) or, in the absence of CCME Guidelines, the Ontario Interim Provincial Water Quality Objectives (PWQO) (i.e., for total phosphorus) (OMOE, 1994) (Table 1).

Water quality data are typically collected from March to November over several years as they represent environmental exposure during various climatic regimes (wet/dry; warm/cool) that might stress biological communities.

The WQI was calculated for the most downstream monitoring site for streams draining to the Great Lakes, including tributaries to the Great Lake connecting channels and the St. Lawrence River as an indication of water quality entering the Great Lakes. Data for the period 2015 – 2019 were used to determine the status in this report.

Because the WQI synthesizes multiple water quality parameters into one value, comparing WQI values across time may not be the best indicator of changes in water quality. This version of the sub-indicator report includes a separate assessment of water quality trends in Great Lakes tributaries.

Short and long-term trends

A separate analysis was conducted to determine short- and long- term trends; results are shown in Table 2 and Figure 3. The WQI (explained above) was used to determine current water quality status. Trends were determined for Great Lakes tributaries by using a flow-adjusted Mann Kendall trend test at suitable sites. The criteria used for the selection of WQI sites (noted in the sub-section above) differed from the criteria used to select sites for the short- and long- term trend analyses. As such, a different selection of sites were used for the trend analyses (see Figure 3) compared to the sites used for the WQI calculation (see Figure 1). While only downstream (i.e., closest to the outlet) PWQMN sites were selected for the WQI used for the current status, the criteria were expanded for sites suitable for trend analyses. Long-term sites (>= 10 years of data) that were co-located at or near a Water Survey of Canada hydrometric gauge (HYDAT) (https://www.canada.ca/en/environment-climate-change/services/water-overview/quantity/monitoring/survey/data-products-services/national-archive-hydat.html) were used in the analysis. Further, any sites with gaps in the long-term record greater than 1/3 were removed from the analysis. Three water quality parameters were used in these trend assessments: total phosphorus, nitrate and chloride.

In order to remove any influences of flow on trends, the method described in Helsel et al., (2020) was used. A LOWESS model was fitted to concentration against flow and a Mann-Kendall Trend Test was used on the residuals from this model. Long-term trends were analyzed for the complete record at each station, starting from 1970 onwards. The complete record varies for each station and is not consistent across sites. Please see Table 3 for the time period used for each site for long-term trends. 10-year trends were assessed for the period: 2010-2019.

Ecological Condition

Background

The Ontario Ministry of the Environment, Conservation and Park's Provincial (Stream) Water Quality Monitoring Network (PWQMN) measures water quality in rivers and streams at hundreds of sites across Ontario in partnership with Ontario's Conservation Authorities. Most of these sites are located in the Great Lakes basin, and many are located at or near the outlets of tributaries to the Great Lakes. Stream water samples are collected on an approximately monthly basis from April to November and delivered to the Ministry of the Environment, Conservation and Park's laboratory where they are analyzed using consistent analytical methods for a consistent suite of water quality indicators. Water quality indicators are selected to indicate the influence of land-use activities (primarily agricultural and urban influences) on stream water quality. For example, chloride is measured as an indicator of the influence of salt loading from winter de-icing. Field measurements including water temperature and pH are also taken at the time of sample collection using portable water quality meters. Water quality data for all stream monitoring sites is available on the Ontario Ministry of the Environment, Conservation and Parks public website (https://data.ontario.ca/dataset/provincial-stream-water-quality-monitoring-network).

Status of Water Quality in the Great Lakes Tributaries

The WQI was computed for 72 Canadian tributaries to the Great Lakes. The overall water quality status of tributaries to the Great Lakes can be described as Fair (WQIavg=63, WQIrange=16-100). Broken down by category: 18% of the tributaries were categorized as having Good water quality, 63% as Fair, and 19% were Poor (Figures 1 and 2).

Good water quality was found in certain tributaries to Lakes Huron, Superior, and Ontario and the St. Lawrence River. Poor water quality was found in certain tributaries to Lakes Erie and Ontario. The WQI scores ranged from 16 (Sturgeon River, Lake Erie) to 100 (Moira River, Lake Ontario).

On a lake-by-lake basis, tributaries to Lake Superior can be described as having an Undetermined status. Tributaries to Lakes Huron (WQI_{avg}=76, WQI_{range}=55-86, n=20) and Ontario and St. Lawrence River (WQI_{avg}=62, WQI_{range}=27-100, n=34) can be described as having Fair water quality. Tributaries to Lake Erie (WQI_{avg}=43, WQI_{range}= 16-68, n=15) were categorized as having Poor water quality. The designation of the water quality for tributaries (based on the WQI results) has not changed for any of the Great Lake basins since the last report.

10 year- and long- term trends

Short (10-year) and long-term trend analyses for three water quality parameters (total phosphorus, nitrate and chloride) known to be influenced by anthropogenic activities are included with this sub-indicator report to estimate trends in water quality conditions. Assessing a singular trend indication on a basin-level is challenging considering the gradient in land uses/covers in various watersheds within each of the Great Lakes basin. In many cases, trends are mixed since the concentrations of the different water quality parameters evaluated here often vary in opposite directions over time (Table 2). For Lakes Huron and Ontario, the 10-year trend is reported as unchanging since the majority of trend sites show no trend for total phosphorus and nitrate. Slightly less than half of the trend sites in Lake Huron and slightly more than half of the trend sites in Lake Ontario had an increasing trend in chloride concentrations in the past 10 years. However, greater than 70% of the sites used in the trend analysis are unchanging for total phosphorus and nitrate, suggesting that there is no strong evidence towards deteriorating conditions based on this analysis (Table 2). Long-term trends are mixed and therefore assessed as undetermined: most sites show a decreasing trend for total phosphorus, no trend for nitrate, and increasing trend for chloride. The 10-year trend for Lake Erie is reported as unchanging since the majority of trend sites are showing no trend for all three parameters tested (total phosphorus, nitrate, and chloride). Long-term trend is reported as undetermined due to mixed trends: majority of trend sites show a decreasing trend in total phosphorus concentrations, no trend in nitrate concentrations, and increasing trend in chloride concentrations.

There are insufficient monitored sites (under the PWQMN) in the Lake Superior Basin to determine trends in tributary water quality for this basin.

Linkages

The WQI values for the 72 tributaries show a statistically significant negative correlation (Spearman's Rank Correlation; $p<0.001$) with percent of the watershed occupied by human land uses (%agriculture and %urban land covers combined) (Figure 4). This relationship suggests that overall water quality in the Great Lakes tributaries is influenced by human land use where minimally developed watersheds have higher WQI scores than more heavily developed watersheds.

The WQI scores suggest the potential for substances in stream water to impact aquatic life based on compliance with water quality criteria. However, the WQI values are not a direct measure of impacts to aquatic communities, such as changes in fish and benthic invertebrate communities. The WQI values (and the water quality in tributaries) also infer the potential for discharge of nutrients or other substances from tributaries into the Great Lakes and the associated impacts of these discharges, particularly at the tributary mouths and nearby nearshore areas.

It should be noted that there are some linkages that can be made to impacts on aquatic life. Elevated concentrations above water quality criteria for parameters such as nitrates, un-ionized ammonia, chloride, and metals indicate potential impacts to aquatic life. For example, freshwater mussels are particularly sensitive to chloride (a component of road salt) exposure compared to other aquatic life, especially during their early life stages. Chloride concentrations in many of our rivers and streams have been increasing since the mid-1970s. (Water Quality in Ontario, 2014 Report).

Linkages to other sub-indicators in the indicator suite include:

- Cladophora – phosphorus inputs from tributaries can have an effect on *Cladophora* growth.
- Harmful Algal Blooms – nutrient inputs from tributaries contribute to algal blooms in receiving water bodies.
- Groundwater Quality – in areas where groundwater and surface water interact, groundwater may influence water quality.
- Nutrients in Lakes (open water) – tributaries transport nutrients and other constituents to downstream receiving water bodies.

- Treated Drinking Water – where drinking water systems use surface water (rivers) as drinking water sources.
- Watershed Stressors – watershed land use and stressors directly impact water quality in receiving tributaries.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada				X
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://data.ontario.ca/dataset/provincial-stream-water-quality-monitoring-network		

Data Limitations

The WQI is a simple communication tool intended for the public. However, the WQI has its limitations. One potential challenge is combining water quality parameters with different ecological impacts (mixing eutrophication parameters with toxics) into one index value, as opposed to sub-indices to represent issues that have common management expectations (i.e., nutrient management programs both point and non-point). Some water quality programs do not have sufficient parameters to use the Index without resulting in the inadvertent influence of one parameter (with few parameters, the influence of one can dominate the final Index score). Some water quality parameters are naturally-occurring in certain areas, such as trace metals, and inclusion in the Index may require the development of site-specific guidelines.

The computed WQI score is dependent on the adequacy of the data and the water chemistry parameters that represent the risk based on the human activities within the watershed. Given the cost of some water quality parameters, they may not be included in routine programs and the Index score may not reflect the entire risk.

Because the WQI can be influenced by factors other than water quality (i.e., the particular parameters selected for the calculation, the number of parameters included, the specific sites used, and the water quality criteria for a given jurisdiction), using changes in the WQI scores over time to identify trends can be potentially more indicative of changes based on how the index was calculated than changes in the quality of the water. For this reason, subsequent SOGL reports will report on water quality trends using more robust methods. While the same 8 parameters were used to calculate the WQI as in previous SOGL reports, the number of stations reporting has decreased to 72 in this report, down from 73 in the previous report.

Water quality criteria can be exceeded in areas that are naturally rich in a given nutrient or metal. The calculation of the WQI does not take into account naturally-occurring elevated concentrations of some parameters.

Furthermore, the overall status for the entire Great Lakes Basin may not be accurate given the spatial bias in the water quality sites used in this assessment. While there is good spatial coverage for both the Lakes Erie and Ontario watersheds, Lake Superior and the northern shore of Lake Huron are underrepresented.

10 year- and long- term trend analyses were conducted using the Mann Kendall trend test which tests for a monotonic trend in the data. Non-monotonic trends (i.e., two different trends in the same time period) may exist in the data but would not be depicted by the Mann Kendall results. For example, a site showing a no trend result from the Mann Kendall test may in fact have two different and significant trends for the time period in question (e.g. increasing then decreasing or decreasing then increasing).

Most of the PWQMN's monitoring sites are purposefully located where water quality impacts are known or expected, such as areas with a high population or where land is used for agriculture. Minimally-impacted reference watersheds are likely under-represented in this sub-indicator. The sub-indicator also under-represents tributaries to the upper Great Lakes (especially Lake Superior). For future reports, a redundancy or other analysis could be undertaken to eliminate some sites from the lower Great Lakes to ensure all lakes are more equally represented

For this Water Quality in Tributaries report, the WQI has been computed for Canadian tributaries only. Work is underway to include data from the U.S. side of the Great Lakes Basin in future versions of this report. This may require changes in condition and trend assessment metrics and definitions.

Additional Information

The WQI is a communication tool that was designed to report complex water quality information about multiple variables in a simplified format. While the WQI can provide a broad overview of water quality, it is not intended to replace rigorous technical analyses of water quality data for water resource management.

This current Water Quality in Tributaries report is a status update from 2019 (ECCC and USEPA, 2019). This report uses the same eight (8) site-relevant parameters as the previous report. The WQI was recalculated for this report using the most recent water quality monitoring results for these parameters with current water quality criteria for the protection of aquatic life. For this current report, WQI scores are computed for 72 tributaries whereas scores were computed for 73 tributaries used previously (ECCC and USEPA, 2019). This version of the report also includes summary results from trend analyses of individual water quality parameters.

Acknowledgments

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agriculture and urban land uses

Source: Ontario Ministry of the Environment, Conservation and Parks

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Table 1. Water quality criteria for the eight indicators used in the CCME Water Quality Index (WQI) calculations. Source: Ontario Ministry of the Environment, Conservation and Parks

Parameter	Criterion	Source
Ammonia (un-ionized)	0.0152 mg L-1-N	CCME
Chloride	120 mg L-1	CCME
Copper	2 µg L-1 at water hardness of 0-120 mg L-1-CaCO3 3 µg L-1 at water hardness of 120-180 mg L-1-CaCO3 4 µg L-1 at water hardness of >180 mg L-1-CaCO3	CCME
Iron	300 µg L-1	CCME
Nitrate	2.9 mg L-1-N	CCME
Nitrite	0.06 mg L-1-N	CCME
Phosphorus	0.03 mg L-1	OMOE
Zinc	30 µg L-1	CCME

Table 2. Summary of short (10 year-) and long-term water quality trends (% of sites) in each of the Great Lakes basins based on Mann Kendall trend tests (number of sites used in the analysis are shown in square brackets).

	Total phosphorus		Nitrate		Chloride	
	10-year	**Long-term**	**10-year**	**Long-term**	**10-year**	**Long-term**
Lake Ontario	[63]	[59]	[62]	[56]	[63]	[57]
Decrease	5	59	5	34	0	5
Increase	24	8	15	25	56	79
No trend	71	32	81	41	44	16
Lake Erie	[38]	[35]	[38]	[35]	[38]	[36]
Decrease	5	66	5	23	0	6
Increase	3	3	18	29	42	78
No trend	92	31	76	49	58	17
Lake Huron	[30]	[28]	[30]	[28]	[30]	[28]
Decrease	10	64	10	21	13	11
Increase	3	7	20	18	47	64
No trend	87	29	70	61	40	25
Lake Superior	[1]		[1]		[1]	
Decrease	0		0		0	
Increase	0		0		0	
No trend	100		100		100	

Table 3. WQI Index results, trend results and time period of record (long-term trends) for individual sites used in analyses. The ↑ symbol indicates an increasing trend, ↓ indicates a decreasing trend, and ↔ indicates no trend.

			CHLORIDE TREND			NITRATE TREND			TOTAL PHOSPHORUS TREND		
STATION	Great Lake Basin	WQI	Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range
04001000302	Erie	35.9	↔	↔	1991 - 2019	↔	↔	1991 - 2019	↔	↔	1991 - 2019
04001301702	Erie		↑			↔	↔	2003 - 2019	↔		
04001302502	Erie		↔	↑	1970 - 2019	↑	↑	1984 - 2019	↔	↓	1970 - 2019
04001302902	Erie		↔	↑	1972 - 2019	↓	↓	1984 - 2019	↔	↓	1972 - 2019
04001304102	Erie		↔	↑	1975 - 2019	↓	↔	1984 - 2019	↔	↔	1975 - 2019
04001304402	Erie		↔	↑	1975 - 2019	↔	↑	1984 - 2019	↔	↔	1975 - 2019
04001305802	Erie		↑	↑	1976 - 2019	↔	↔	1984 - 2019	↔	↓	1976 - 2019
04001306402	Erie		↔			↔			↔		
04001306602	Erie		↔	↑	1979 - 2019	↔	↑	1984 - 2019	↔	↑	1979 - 2019
04001308002	Erie		↔	↑	1988 - 2019	↔	↔	1988 - 2019	↔	↔	1988 - 2019
04001308102	Erie		↔	↔	2006 - 2016	↔	↓	2006 - 2016	↓	↓	2006 - 2016
04001308202	Erie	53.6									
04001308302	Erie		↑	↑	2005 - 2019	↑	↑	2005 - 2019	↔	↔	2005 - 2019
04001309002	Erie		↔	↑	2006 - 2019	↔	↔	2006 - 2019	↔	↔	2006 - 2019
04002700902	Erie		↔	↔	2006 - 2019	↔	↔	2006 - 2019	↔	↔	2006 - 2019
04002701202	Erie		↑	↑	1975 - 2019	↔	↔	1984 - 2019	↔	↓	1975 - 2019
04002701602	Erie		↑	↔	2002 - 2019	↔	↓	2002 - 2019	↔	↔	2002 - 2019
04002701702	Erie	50.9									
10000100302	Erie	44.1									
10000200202	Erie	35.2	↔	↑	1976 - 2019	↔	↔	1984 - 2019	↔	↓	1976 - 2019

STATION	Great Lake Basin	WQI	CHLORIDE TREND Short-term	Long-term	Long-term range	NITRATE TREND Short-term	Long-term	Long-term range	TOTAL PHOSPHORUS TREND Short-term	Long-term	Long-term range
16001800202	Erie	48									
16002700102	Erie	16.3									
16003200102	Erie	37.8									
16007200102	Erie	34.3									
16008700502	Erie		↔	↓	2004 - 2019	↔	↑	2004 - 2019	↔	↓	2004 - 2019
16008701002	Erie	38.3									
16010900702	Erie		↑	↑	1980 - 2019	↑	↑	1984 - 2019	↔	↓	1980 - 2019
16010900802	Erie	49.4	↑	↑	2002 - 2019	↑	↑	2002 - 2019	↔	↓	2002 - 2019
16012401102	Erie	67.8	↔	↔	2002 - 2018	↑	↑	2002 - 2018	↓	↓	2002 - 2018
16012401202	Erie		↑	↑	2002 - 2019	↑	↔	2002 - 2019	↔	↔	2002 - 2019
16015900302	Erie	53.9	↑	↑	1981 - 2019	↑	↑	1995 - 2019	↔	↓	1975 - 2019
16016400102	Erie	49.5	↔	↑	1970 - 2019	↔	↓	1995 - 2019	↔	↔	1970 - 2019
16018401602	Erie		↔	↓	1970 - 2019	↔	↓	1994 - 2019	↔	↓	1970 - 2019
16018403202	Erie		↔	↑	1970 - 2018	↔	↓	1994 - 2018	↔	↓	1970 - 2018
16018403402	Erie		↑	↑	1970 - 2019	↔			↔	↓	1970 - 2019
16018403502	Erie	37.1									
16018403602	Erie		↑	↑	1972 - 2019	↔	↑	1994 - 2019	↔	↓	1972 - 2019
16018403702	Erie		↑	↑	1972 - 2019	↔	↔	1994 - 2019	↔	↓	1972 - 2019
16018406702	Erie		↔	↑	1978 - 2019	↔			↔	↓	1975 - 2019
16018407402	Erie		↔	↔	2007 - 2019	↔	↔	2007 - 2019	↔		
16018407702	Erie		↔	↑	1980 - 2019	↔	↔	1994 - 2019	↑	↓	1975 - 2019
16018409302	Erie		↑	↑	1977 - 2015	↔	↓	1994 - 2015	↔	↓	1977 - 2015

STATION	Great Lake Basin	WQI	CHLORIDE TREND Short-term	Long-term	Long-term range	NITRATE TREND Short-term	Long-term	Long-term range	TOTAL PHOSPHORUS TREND Short-term	Long-term	Long-term range
16018409902	Erie		↑	↑	1978 - 2019	↔	↔	1994 - 2019	↔	↓	1978 - 2019
16018410202	Erie		↑	↑	1979 - 2019	↔	↓	1994 - 2019	↔	↓	1979 - 2019
16018410302	Erie		↔	↑	1980 - 2019	↔	↔	1994 - 2019	↔	↓	1980 - 2019
16018410402	Erie		↔	↑	2006 - 2019	↔	↔	2006 - 2019	↔	↔	2006 - 2019
16018410602	Erie		↑	↑	1980 - 2019	↔	↔	1994 - 2019	↔	↓	1980 - 2019
03001500202	Huron	70.9									
03001600302	Huron	85.5	↓	↑	1975 - 2019	↔	↔	1984 - 2019	↔	↔	1975 - 2019
03001700202	Huron	78.3									
03003000202	Huron	85.6	↔	↑	1975 - 2019	↑	↓	1984 - 2019	↓	↓	1975 - 2019
03003601002	Huron	85.5	↓	↓	2002 - 2019	↑	↔	2002 - 2019	↓	↓	2002 - 2019
03005300102	Huron	85.4	↑	↔	2006 - 2019	↔	↔	2006 - 2019	↔	↔	2006 - 2019
03005702502	Huron	82.7									
03005702902	Huron		↑	↑	2002 - 2019	↔	↔	2002 - 2019	↔	↓	2002 - 2019
03006600102	Huron	69.4									
03007000302	Huron	78.3									
03007000402	Huron		↔	↔	2002 - 2019	↔	↔	2002 - 2019	↔	↓	2002 - 2019
03007300202	Huron	77.3	↑	↑	2002 - 2019	↑	↑	2002 - 2019	↔	↓	2002 - 2019
03007500202	Huron	85.3									
03007600302	Huron	77	↑	↔	2002 - 2019	↑	↔	2002 - 2019	↔	↔	2002 - 2019
03007600502	Huron	78.1	↑	↑	2002 - 2019	↔	↔	2002 - 2019	↔	↓	2002 - 2019
03007702902	Huron		↔	↑	1977 - 2019	↔	↔	2002 - 2019	↔	↓	1977 - 2019
03007703902	Huron		↑	↑	2002 - 2019	↔	↔	2002 - 2019	↔	↓	2002 - 2019

STATION	Great Lake Basin	WQI	CHLORIDE TREND			NITRATE TREND			TOTAL PHOSPHORUS TREND		
			Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range
03013301902	Huron		↔	↑	1974 - 2018	↔	↓	2003 - 2018	↔	↓	1974 - 2018
03013400102	Huron		↑	↑	1970 - 2019	↔	↓	1995 - 2019	↔	↓	1970 - 2019
08002100202	Huron	54.8									
08002201002	Huron		↑	↑	2006 - 2019	↔	↔	2006 - 2019	↔	↔	2006 - 2019
08002201602	Huron		↑	↑	1984 - 2019	↓	↔	1984 - 2019	↓	↑	1984 - 2019
08002202002	Huron		↑	↑	2003 - 2019	↔	↔	2003 - 2019	↔	↓	2003 - 2019
08004000802	Huron	69.8	↔	↑	1975 - 2019	↔	↓	1984 - 2019	↔	↓	1975 - 2019
08005600102	Huron	77.9									
08005600202	Huron		↑	↑	1984 - 2019	↔	↑	1984 - 2019	↔	↓	1984 - 2019
08005600302	Huron		↔	↑	1970 - 2019	↔	↑	1984 - 2019	↔	↓	1970 - 2019
08005600902	Huron		↑	↑	1970 - 2019	↔	↔	1984 - 2019	↔	↓	1970 - 2019
08005603702	Huron		↔	↔	2004 - 2019	↓	↓	2004 - 2019	↔	↔	2004 - 2019
08007600102	Huron	71									
08010300102	Huron	61.5	↑			↔	↔	2002 - 2019	↔		
08012303002	Huron	63.7	↔	↓	2002 - 2019	↑	↔	2002 - 2019	↔	↔	2002 - 2019
08012305102	Huron		↔	↔	2005 - 2019	↔	↑	2005 - 2019	↔	↔	2005 - 2019
08012305702	Huron		↔	↔	2005 - 2019	↑	↑	2005 - 2019	↑	↑	2005 - 2019
08013500302	Huron	78.3	↓	↑	1970 - 2019	↔	↔	1984 - 2019	↔	↓	1970 - 2019
08013500502	Huron		↔	↔	2002 - 2019	↔	↔	2002 - 2019	↔	↔	2002 - 2019
14002800302	Huron		↔	↑	1977 - 2019	↓	↓	1995 - 2019	↔	↓	1977 - 2019

STATION	Great Lake Basin	WQI	CHLORIDE TREND			NITRATE TREND			TOTAL PHOSPHORUS TREND		
			Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range
14002801202	Huron		↑	↓	1976 - 2019	↔			↔	↓	1976 - 2019
14002802802	Huron		↓			↔			↔		
06000300102	Ontario	38.1									
06001700102	Ontario	85.3									
06002400102	Ontario	41.2	↔	↑	1970 - 2019	↔	↔	2003 - 2019	↔	↑	1970 - 2019
06006000702	Ontario	62.7									
06006100102	Ontario	56.4	↔	↑	2002 - 2019	↔	↓	2002 - 2019	↔	↔	2002 - 2019
06006300102	Ontario	59.4									
06006301102	Ontario		↔	↑	1975 - 2019	↔	↓	1994 - 2019	↔	↔	1975 - 2019
06006800102	Ontario	37.6									
06007600302	Ontario		↑	↑	1988 - 2019	↔	↑	1994 - 2019	↔	↓	1988 - 2019
06007600402	Ontario		↔	↑	1970 - 2019	↔	↑	1994 - 2019	↔	↓	1970 - 2019
06007600802	Ontario		↔	↓	1987 - 2014	↑	↑	1994 - 2014	↓	↓	1987 - 2014
06007601002	Ontario		↑	↑	1982 - 2019	↔	↓	1994 - 2019	↔	↓	1982 - 2019
06007602202	Ontario		↑	↑	1979 - 2019	↓	↔	1994 - 2019	↔	↓	1979 - 2019
06007602302	Ontario		↔	↑	1979 - 2019	↔	↓	1994 - 2019	↔	↑	1979 - 2019
06007605002	Ontario	45.4									
06008000602	Ontario	35.4	↔	↑	2002 - 2019	↔	↔	2002 - 2019	↑	↑	2002 - 2019
06008200302	Ontario	32.9									
06008300902	Ontario		↔			↔	↔	2004 - 2019	↑		
06008301902	Ontario	45.8	↔	↑	1988 - 2019	↔	↓	1994 - 2019	↔	↓	1979 - 2019

STATION	Great Lake Basin	WQI	CHLORIDE TREND Short-term	Long-term	Long-term range	NITRATE TREND Short-term	Long-term	Long-term range	TOTAL PHOSPHORUS TREND Short-term	Long-term	Long-term range
06008310302	Ontario		↑	↑	2002 - 2019	↔	↓	2002 - 2019	↔	↓	2002 - 2019
06008310402	Ontario		↑	↑	2002 - 2019	↔	↑	2002 - 2019	↔	↔	2002 - 2019
06008501402	Ontario	26.6	↔	↑	1984 - 2019	↑	↓	1994 - 2019	↑	↓	1979 - 2019
06009701102	Ontario	48.6									
06010400102	Ontario	59.7									
06010800102	Ontario	67.9	↑	↑	1970 - 2018	↓	↓	1994 - 2018	↑	↓	1970 - 2018
06011200302	Ontario	67.5									
06011600102	Ontario	62.6	↑	↑	1970 - 2019	↔			↑	↓	1970 - 2019
06011700302	Ontario	78.3	↑	↑	1973 - 2019	↑	↔	1994 - 2019	↑	↓	1973 - 2019
06012900102	Ontario	78.2									
06012900202	Ontario		↑			↔	↓	2002 - 2019	↔		
06012900502	Ontario		↑	↑	2002 - 2019	↑	↓	2002 - 2019	↔	↔	2002 - 2019
06013000102	Ontario	71									
06013300402	Ontario	70.7	↑	↑	2002 - 2019	↔	↑	2002 - 2019	↑	↔	2002 - 2019
06013300502	Ontario		↑	↔	2006 - 2019	↔	↓	2006 - 2019	↔	↔	2006 - 2019
06017200102	Ontario		↔			↔			↔		
06018000302	Ontario		↔						↔		
06018300202	Ontario	74.2	↔	↑	1970 - 2019	↔	↑	1994 - 2019	↔	↔	1970 - 2019
09000100502	Ontario	51.2	↔			↔			↔		
09000800502	Ontario	54.4	↔	↔	2002 - 2019	↔	↓	2002 - 2019	↔	↔	2002 - 2019
09000800602	Ontario		↑	↑	2002 - 2019	↔	↓	2002 - 2019	↔	↔	2002 - 2019

STATION	Great Lake Basin	WQI	CHLORIDE TREND			NITRATE TREND			TOTAL PHOSPHORUS TREND		
			Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range
09000800702	Ontario		↑	↑	2005 - 2019	↓	↓	2005 - 2019	↓	↓	2005 - 2019
09000902402	Ontario	52.7	↑	↔	2002 - 2019	↔	↓	2002 - 2019	↔	↓	2002 - 2019
12000200402	Ontario	62									
12001700102	Ontario	85.1									
12003100102	Ontario	76.3	↑	↔	1986 - 2019	↔	↔	1994 - 2019	↔	↓	1986 - 2019
12003400102	Ontario	54.8	↑	↑	1989 - 2019	↔	↔	1994 - 2019	↔	↓	1989 - 2019
12007300302	Ontario	61.7	↑	↑	1976 - 2019	↔	↔	1994 - 2019	↔	↓	1976 - 2019
12008600102	Ontario		↑	↔	2003 - 2019	↔	↔	2003 - 2019	↔	↓	2003 - 2019
17002100302	Ontario		↔			↔			↔	↓	1970 - 2019
17002103802	Ontario		↑	↑	1970 - 2019	↑	↑	1995 - 2019	↓	↓	1970 - 2019
17002107102	Ontario		↑	↑	2011 - 2019	↔	↔	2011 - 2019	↔	↔	2011 - 2019
17002107502	Ontario		↑	↑	1972 - 2019	↔	↔	1996 - 2019	↔	↓	1972 - 2019
17002107702	Ontario		↑	↑	2006 - 2019	↑	↔	2006 - 2019	↔	↔	2006 - 2019
17002109502	Ontario		↔			↔			↑	↔	1976 - 2019
17002112802	Ontario		↔	↔	2002 - 2019	↔	↔	2002 - 2019	↔	↓	2002 - 2019
17002113302	Ontario		↑	↑	2002 - 2019	↔	↔	2002 - 2019	↔	↓	2002 - 2019
17002113702	Ontario		↑	↑	2006 - 2019	↔	↑	2006 - 2019	↑	↔	2006 - 2019
17002114502	Ontario		↑	↔	2007 - 2019	↔	↑	2007 - 2019	↑	↑	2007 - 2019
17002600102	Ontario	100									
17002600602	Ontario		↔	↔	2002 -	↔	↔	2002 -	↔	↓	2002 -

STATION	Great Lake Basin	WQI	CHLORIDE TREND Short-term	Long-term	Long-term range	NITRATE TREND Short-term	Long-term	Long-term range	TOTAL PHOSPHORUS TREND Short-term	Long-term	Long-term range
					2019			2019			2019
17002600702	Ontario		↔	↔	2006 - 2019	↔	↑	2006 - 2019	↔	↔	2006 - 2019
17002600902	Ontario		↑	↑	1970 - 2017	↔	↔	1994 - 2017	↔	↓	1970 - 2017
17002601302	Ontario		↔	↑	1970 - 2019	↔	↔	1994 - 2019	↔	↓	1970 - 2019
17002601902	Ontario		↔	↑	1970 - 2019	↔	↔	1994 - 2019	↔	↓	1970 - 2019
17003100102	Ontario	92.8	↔	↑	1970 - 2019	↔	↓	1994 - 2019	↑	↓	1970 - 2019
17003500202	Ontario	92.6									
17003700302	Ontario	85.3	↔	↓	1987 - 2019	↑	↓	1994 - 2019	↔	↓	1975 - 2019
18003300302	Ontario		↑	↑	1970 - 2019	↔	↓	1994 - 2019	↔	↓	1970 - 2019
18003303102	Ontario		↑	↑	1972 - 2019	↔	↔	1994 - 2019	↔	↓	1972 - 2019
18003303602	Ontario		↑	↑	1980 - 2019	↑	↔	1994 - 2019	↔	↔	1980 - 2019
18207010002	Ontario			↑	1974 - 2019					↓	1974 - 2019
18207014502	Ontario		↑	↑	1980 - 2016	↔	↑	1986 - 2016	↔	↓	1980 - 2016
18337012102	Ontario		↑	↑	1986 - 2019	↔	↔	1994 - 2019	↔	↓	1983 - 2019
18343000102	Ontario		↑	↑	2007 - 2019	↔	↑	2007 - 2019	↔	↔	2007 - 2019
18343000202	Ontario		↔	↑	2007 - 2019	↔	↔	2007 - 2019	↔	↔	2007 - 2019
18343006102	Ontario		↔	↑	1983 - 2019	↑	↔	1994 - 2019	↑	↓	1983 - 2019
18343017502	Ontario		↑	↑	2003 - 2019	↔	↑	2003 - 2019	↔	↔	2003 - 2019
18343023002	Ontario		↑	↑	1988 - 2019	↔	↓	1994 - 2019	↑	↔	1988 - 2019

STATION	Great Lake Basin	WQI	CHLORIDE TREND			NITRATE TREND			TOTAL PHOSPHORUS TREND		
			Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range	Short-term	Long-term	Long-term range
18343061002	Ontario		↔	↓	2003 - 2019	↔	↑	2003 - 2019	↑	↑	2003 - 2019
18369001002	Ontario		↔	↑	1970 - 2018	↔			↑	↓	1970 - 2018
01010400202	Superior	92.1									
01010500102	Superior	69.9	↔			↔			↔		
01010600202	Superior	75.4									

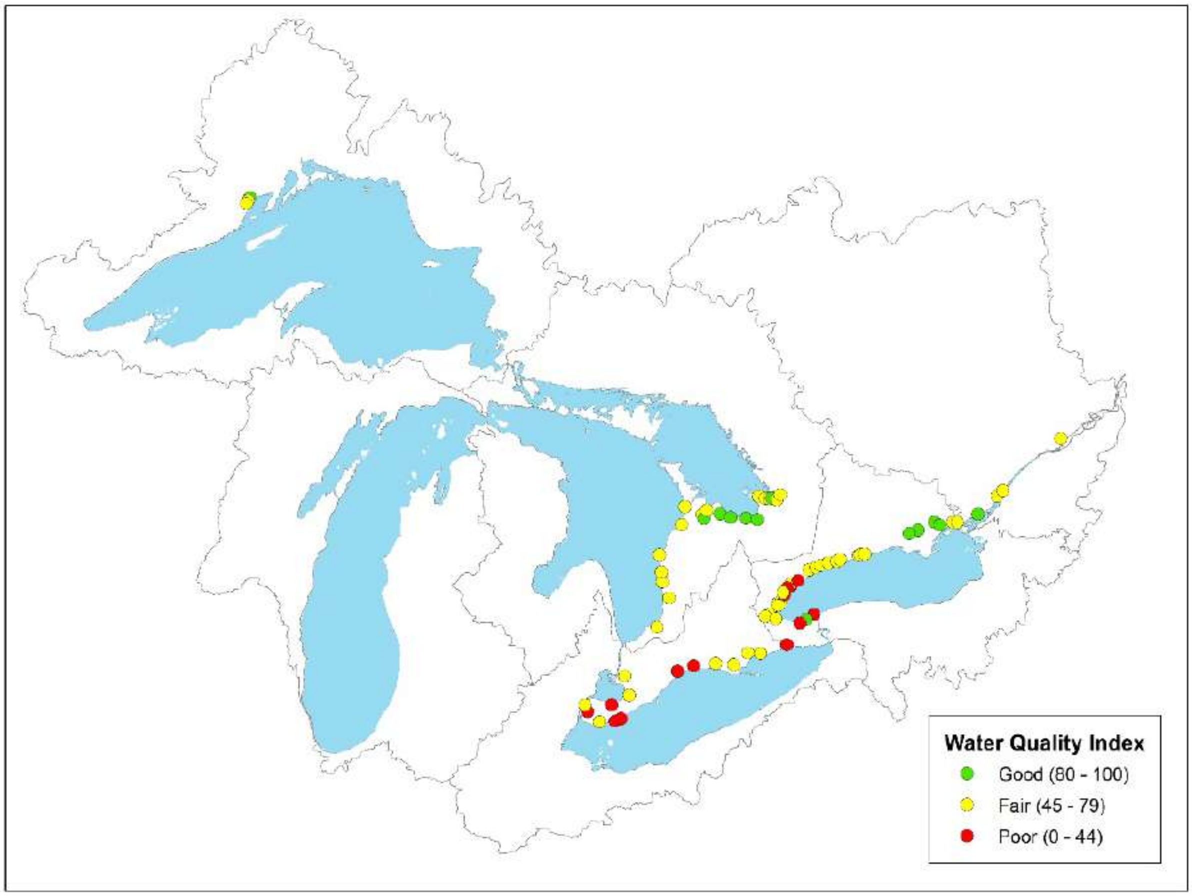

Figure 1. Map showing CCME Water Quality Index (WQI) values for 72 Canadian tributaries to the Great Lakes. Source: Ontario Ministry of the Environment, Conservation and Parks.

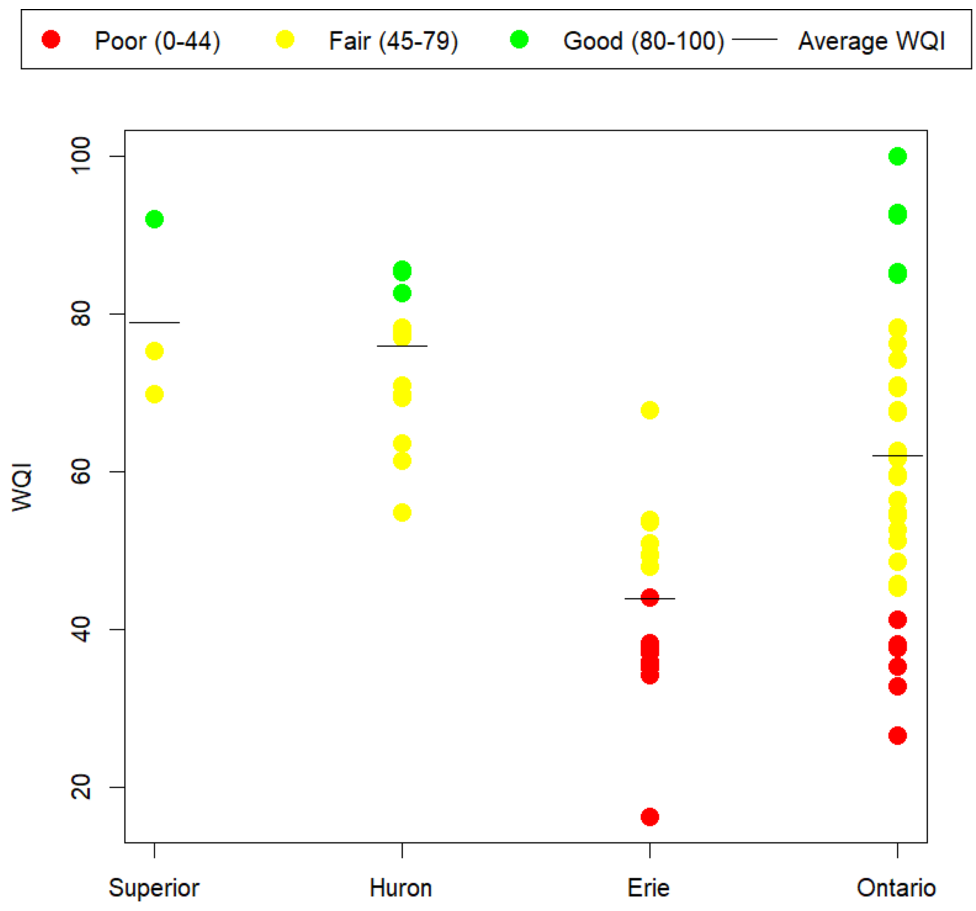

Figure 2. CCME Water Quality Index (WQI) values for Canadian Great Lakes tributaries by lake basin. Source: Ontario Ministry of the Environment, Conservation and Parks.

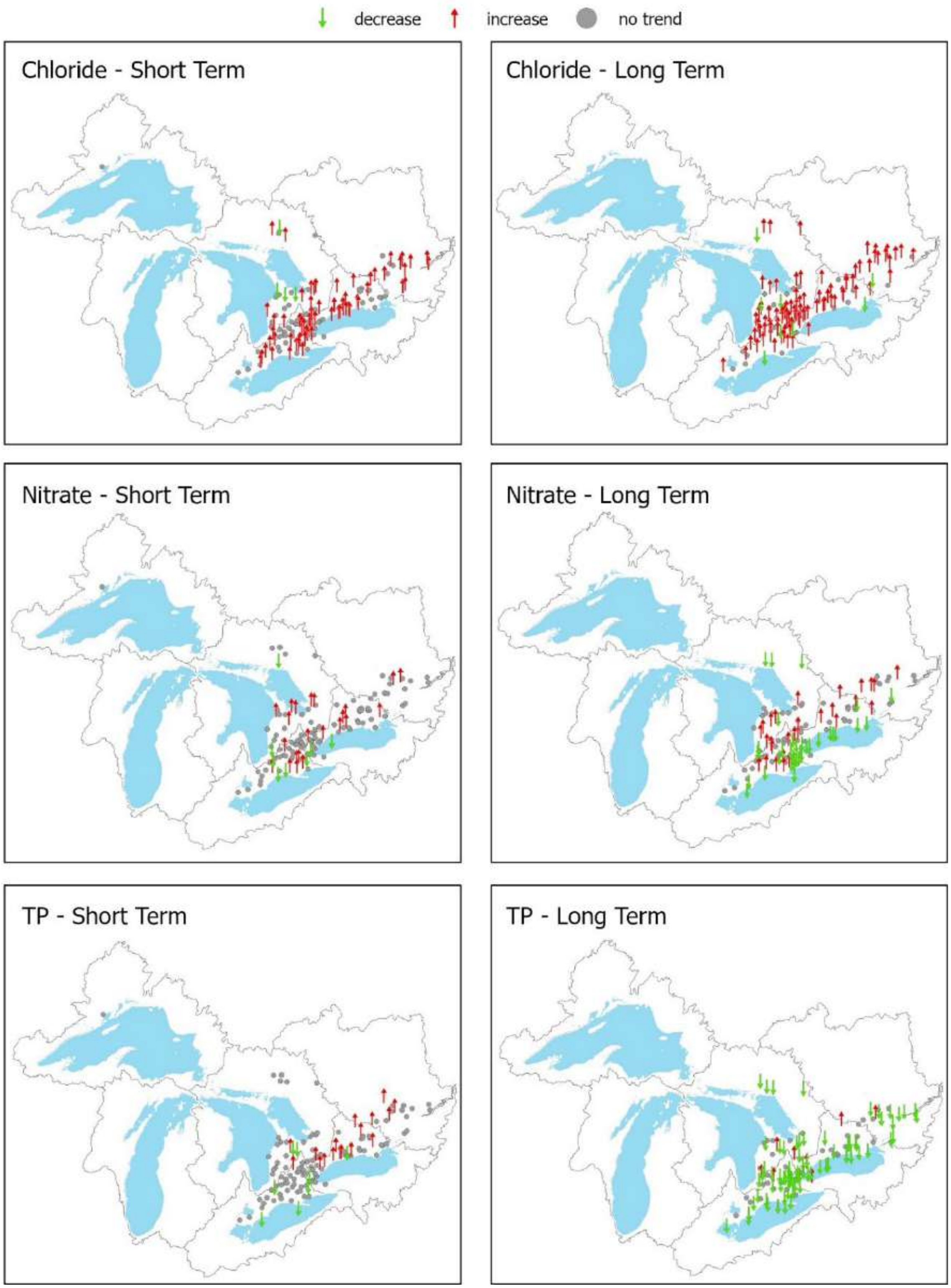

Figure 3. Short- and long-term trends for three water quality parameters (chloride, nitrate, and total phosphorus). Source: Ontario Ministry of the Environment, Conservation and Parks.

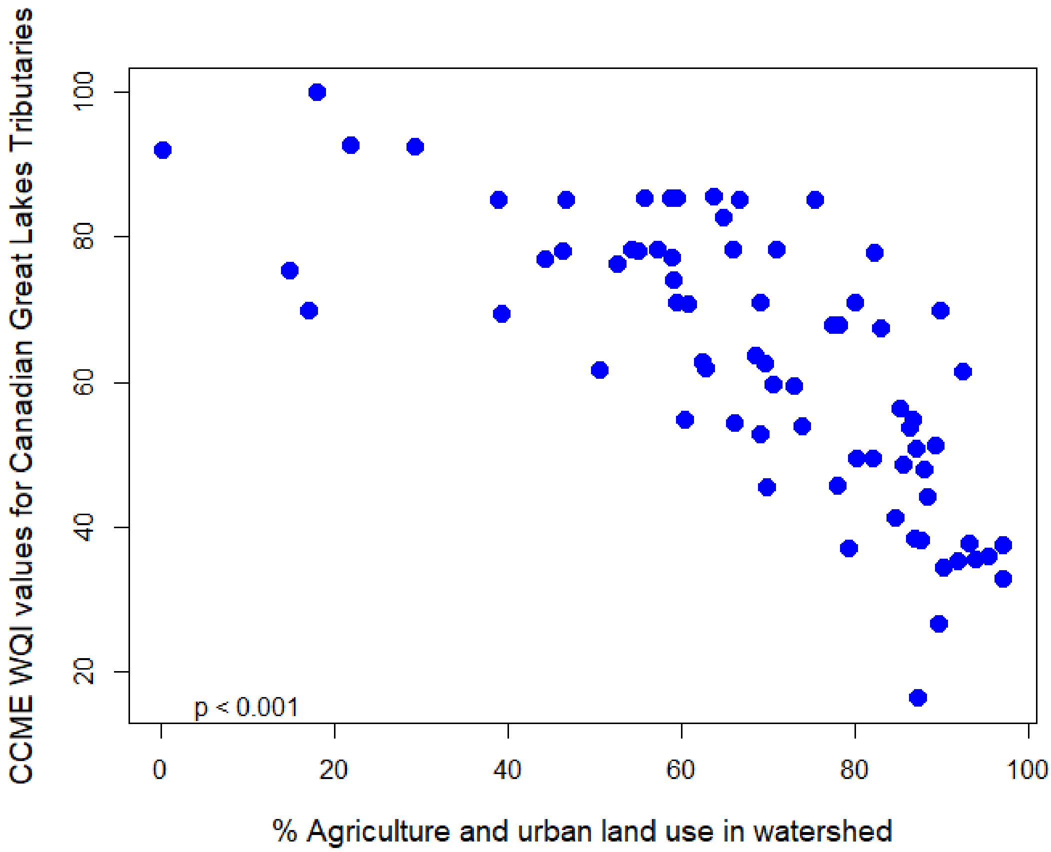

Figure 4. CCME Water Quality Index (WQI) values for Canadian Great Lakes tributaries (n=73) versus percent agriculture and urban land uses. Source: Ontario Ministry of the Environment, Conservation and Parks

Sub-Indicator: Human Population

Overall Assessment

Trends:

10-year Trend (2010-2020 U.S. & 2011-2021 Canada): Increasing

30-year Trend (1990-2020 U.S. & 1991-2021 Canada): Increasing

Long-term (50-year) Trend (1970-2020 U.S. & 1971-2021 Canada): Increasing

Rationale: Based on the 2020 U.S. census and the 2021 Canadian census of population, the resident population within the Great Lakes Basin was 35,371,814. As shown in Table 1, this represents an increase of 1,569,562 (4.6%) residents since the 2010 U.S. census and 2011 Canadian census. Due to a greater than one percent increase in population from the 2010/2011 census reports, the 10-year trend is "Increasing".

For the 30-year trend, the resident population increased by 4,441,359 (14.4%) from 30,930,455 residents in 1990/1991 to 35,371,814 residents in 2020/2021. Therefore, the 30-year trend is "Increasing" (Figure 3).

For the long-term (50-year) trend, the resident population increased by 7,405,631 (26.5%) from 27,966,183 residents in 1970/1971 to 35,371,814 residents in 2020/2021. Therefore, the long-term trend is "Increasing" (Figure 4).

Lake-by-Lake Assessment

Lake Superior

10-year Trend (2010-2020 U.S. & 2011-2021 Canada): Unchanging

30-year Trend (1990-2020 U.S. & 1991-2021 Canada): Decreasing

Long-term (50-year) Trend (1970-2020 U.S. & 1971-2021 Canada): Unchanging

Rationale: The Lake Superior Basin is the least populous among the five Great Lakes sub-basins. Based on the 2020 U.S. census and the 2021 Canadian census of population, the resident population within the Lake Superior Basin was 594,370 people, an increase of 3,473 residents (0.6%) since the 2010 U.S. census and 2011 Canadian census (Table 1). Due to a less than one percent change in the basin's resident population since the 2010/2011 census, the 10-year trend is "Unchanging."

For the 30-year trend, the resident population decreased by 26,368 (-4.2%) from 620,738 residents in 1990/1991 to 594,370 residents in 2020/2021 (Table 1). Therefore, the 30-year trend is "Decreasing" (Figure 3).

For the long-term (50-year) trend, the resident population decreased by 3,125 (-0.5%) from 597,495 residents in 1970/1971 to 594,370 residents in 2020/2021 (Table 1). The population within the Lake Superior basin increased from 1970/1971 until the 1990/1991 census reporting period. After 1990/1991, the population within the basin decreased and is similar to the 1970/1971 census reporting period. Therefore, the long-term trend is "Unchanging" (Figure 4).

Lake Michigan[1]

10-year Trend (2010-2020): Increasing

30-year Trend (1990-2020): Increasing

Long-term (50-year) Trend (1970-2020): Increasing

Rationale: The Lake Michigan Basin lies wholly within the United States and has no drainage area within Canada (Table 4). Based on the 2020 U.S. census, the resident population within the Lake Michigan Basin was 8,011,470 people, which is an increase of 255,799 (3.2%) residents since the 2010 U.S. census (Table 1). Due to a greater than one percent increase in population from the 2010 census, the 10-year trend is "Increasing."

For the 30-year trend, the resident population increased by 704,211 (9.6%) from 7,307,259 residents in 1990 to 8,011,470 residents in 2020 (Table 1). Therefore, the 30-year trend is "Increasing" (Figure 3).

For the long-term (50-year) trend, the resident population increased by 1,585,983 (24.7%) from 6,425,487 residents in 1970 to 8,011,470 residents in 2020 (Table 1). Therefore, the long-term trend is "Increasing" (Figure 4).

Lake Huron (including St. Marys River)

10-year Trend (2010-2020 U.S. & 2011-2021 Canada): Increasing

30-year Trend (1990-2020 U.S. & 1991-2021 Canada): Increasing

Long-term (50-year) Trend (1970-2020 U.S. & 1971-2021 Canada): Increasing

Rationale: Based on the 2020 U.S. census and the 2021 Canadian census of population, the population within the Lake Huron Basin was 3,199,891 residents, which is an increase of 138,502 (4.5%) since the 2010 U.S. census and 2011 Canadian census (Table 1). Due to a greater than one percent increase in population from the 2010/2011 census, the 10-year trend is "Increasing."

For the 30-year trend, the resident population increased by 477,515 (17.5%) from 2,722,376 residents in 1990/1991 to 3,199,891 residents in 2020/2021 (Table 1). Therefore, the 30-year trend is "Increasing" (Figure 3).

For the long-term (50-year) trend, the resident population increased by 891,318 (38.6%) from 2,308,573 residents in 1970/1971 to 3,199,891 residents in 2020/2021 (Table 1). Therefore, the long-term trend is "Increasing" (Figure 4).

Lake Erie (including St. Clair-Detroit River Ecosystem)

10-year Trend (2010-2020 U.S. & 2011-2021 Canada): Increasing

[1] Beginning in the early 1800s, the Chicago diversion, located in Chicago, Illinois, diverts water from the Lake Michigan watershed into the Upper Mississippi River basin, reducing the land drainage area of the Lake Michigan watershed. Consequently, the Lake Michigan Basin is narrow to the surface water of Lake Michigan in northern Illinois, northwest Indiana, and southeast Wisconsin, which include parts of several densely populated metro areas, such as Chicago, Illinois, and Milwaukee, Wisconsin. For this report, new geospatial methods were used to determine population, which only includes the residents within the basin and excludes the surrounding metro population (Figure 1). Previous SOGL reports employed different methods to estimate population of these metro areas, which intentionally included metro residents outside the basin because their drinking water source was Lake Michigan and was an overestimate of basin population. For the population served drinking water sourced from the surface waters of the Great Lakes and connecting rivers, see the Drinking Water sub-indicator report.

30-year Trend (1990-2020 U.S. & 1991-2021 Canada): Increasing

Long-term (50-year) Trend (1970-2020 U.S. & 1971-2021 Canada): Increasing

Rationale: The Lake Erie Basin is the most populous among the five Great Lakes sub-basins. Based on the 2020 U.S. census and the 2021 Canadian census of population, the population within the Lake Erie Basin was 12,399,519 residents, which is an increase of 363,401 (3.0%) since the 2010 U.S. census and the 2011 Canadian census (Table 1). Due to a greater than one percent increase in population from the 2010/2011 census, the 10-year trend is "Increasing."

For the 30-year trend, the resident population increased by 655,962 (5.6%) from 11,743,557 residents in 1990/1991 to 12,399,519 residents in 2020/2021 (Table 1). Therefore, the 30-year trend is "Increasing" (Figure 3).

For the long-term (50-year) trend, the resident population increased by 642,654 (5.5%) from 11,756,865 residents in 1970/1971 to 12,399,519 residents in 2020/2021 (Table 1). Therefore, the long-term trend is "Increasing" (Figure 4).

Lake Ontario (including Niagara River)

10-year Trend (2010-2020 U.S. & 2011-2021 Canada): Increasing

30-year Trend (1990-2020 U.S. & 1991-2021 Canada): Increasing

Long-term (50-year) Trend (1970-2020 U.S. & 1971-2021 Canada): Increasing

Rationale: The Lake Ontario Basin saw the largest increases for all 3 periods (10-year, 30-year and 50-year). Based on the 2020 U.S. census and the 2021 Canadian census of population, the population within the Lake Ontario Basin was 11,166,564 residents, which is an increase of 808,387 (7.8%) since the 2010 U.S. census and 2011 Canadian census (Table 1). Due to a greater than one percent increase in population from the 2010/2011 census, the 10-year trend is "Increasing."

For the 30-year trend, the resident population increased by 2,630,039 (30.8%) from 8,536,525 residents in 1990/1991 to 11,166,564 residents in 2020/2021 (Table 1). Therefore, the 30-year trend is "Increasing" (Figure 3).

For the long-term (50-year) trend, the resident population increased by 4,288,801 (62.4%) from 6,877,763 residents in 1970/1971 to 11,166,564 residents in 2020/2021 (Table 1). Therefore, the long-term trend is "Increasing" (Figure 4).

Trend Assessment Definitions

Human Population information in the State of the Great Lakes indicator suite is not assessed in the same manner as other sub-indicators. The intent of this sub-indicator is to identify population fluctuations across the years and not to assess these conditions as “Good”, “Fair”, or “Poor”.

10-year Trends

Increasing: Net increase of the human population in the Great Lakes Basin >1%

Unchanging: Net change in the human population in the Great Lakes Basin <1%

Decreasing: Net decrease of the human population in the Great Lakes Basin >1%

Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend

30-year and Long-term (50-year) Trends

Increasing: Regression analysis indicates an increasing population trend.

Unchanging: Regression analysis does not indicate a trend in the population data.

Decreasing: Regression analysis indicates a decreasing population trend.

Undetermined: Data are not available to report on a trend.

Endpoints and/or Targets

No endpoints are assessed for the human population sub-indicator because multiple factors would need to be considered, such as available infrastructure and sustainable practices, to determine the effects and set thresholds for population growth within the Great Lakes Basin, which goes beyond the purpose of this sub-indicator report.

Sub-Indicator Purpose

Human activities can stress the Great Lakes ecosystem and are often the driving force of environmental degradation. The purpose of this sub-indicator is to present the human population counts and changes within the Great Lakes Basin over time to provide context for understanding anthropogenic drivers of ecosystem trends.

Ecosystem Objective

The human population sub-indicator is related to all the General Objectives described in the 2012 Great Lakes Water Quality Agreement. However, this sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should “be free from other substances, materials, or conditions that may negatively impact the chemical, physical or biological integrity of the Waters of the Great Lakes.”

Measure

This sub-indicator includes analyses of changes in resident population across the Great Lakes Basin from 1970/1971 to 2020/2021 using census data reported by the U.S. Census Bureau and Statistics Canada.

Statistics Canada conducts the census of population every five years on the 1st and 6th year of each decade. Census surveys include population by geographic areas (e.g., census division, dissemination area, dissemination block), total private dwellings, and private dwellings occupied by usual residents. Our results included the resident population and excluded secondary residences (e.g., seasonal cottages). The most recent Canadian census was conducted in 2021, and the initial data release was in February 2022.

The U.S. Census Bureau (USCB) is the official federal office to perform the decennial census of the U.S. population and conducts the American Community Survey (ACS), which provides intercensal population estimates. The most recent census was conducted in 2020 and the initial data release was in April 2021.

Both nations' census agencies report census data by geographical areas, including reporting units of state/province, county/census division, census tracks, block groups/dissemination areas, and blocks (least granular to most granular). Statistics Canada has published population by watershed for each quinquennial census since 1971. Conversely, the U.S. Census Bureau does not calculate population by watershed. Therefore, population within the U.S. portion of the Great Lakes Basin must be calculated by the EPA.

Notably, the methods for determining the U.S. population within the Great Lakes Basin have changed from previous SOGL reports. Instead of calculating the population at the county level and performing multiple proportional adjustments for straddling counties within the basin, the new method used census data at a finer geographic scale (block group data) and geospatial analysis software (ArcGIS Pro, v.2.8, ESRI) to more accurately determine the population of U.S. residents strictly within the Great Lakes Basin. Geospatial layers of population at the census block group level for 2020 and 2010 were used and are publicly available on ArcGIS Online (esri_dm, 2021). The geospatial layers and tabular data of population at the census block group level for 1970, 1980, 1990, and 2000 are publicly available from IPUMS National Historical Geographic Information System (Manson et al., 2020).

The results of population within the Great Lakes Basin are presented in 'Table 1. Population Trends within the Great Lakes Basin: 10-year trend, 30-year trend, & 50-year trend' and 'Table 2. Population within the Great Lakes Basin, 1970-2020, every 10 years.' Because Canada conducts a census every five years, the results are shown in 'Table 3. Population within the Great Lakes Basin in Canada, 1971-2021, every 5 years.' The land drainage areas for the Great Lakes Basin and each its sub-basins, which were used to calculate population density, are reported in Table 4.

The percent population change was used to determine the 10-year trend as indicated in the Trends Assessment Definition section. Long-term (50-year) trends and 30-year trends within the data were determined using linear or quadratic modeling, depending on the shape of the data, with an F-test with an alpha value of 0.05 using R software (R Core Team, 2020). If the p-value from the F-test was greater than the alpha value, the trend was considered "Unchanging," and no trendline was included in Figure 3 and Figure 4. The Shapiro-Wilk test for normality was used to ensure the normality of the model residuals. Due to the different frequency of U.S. census (decadal) and Canadian census (every five years), the U.S. data contained fewer data points and thus a model may not have fit the data. In these cases, the trend analysis for the Canadian population and the percent change in the U.S. population were used to determine if the population was "Increasing," "Decreasing," or "Unchanging."

Ecological Condition

Population within the Great Lakes Basin

The Great Lakes Basin lies within part of eight U.S. states – Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania, and Wisconsin – and the Province of Ontario, Canada (Figure 1). Based on our analysis of the 2020 U.S. census and the 2021 Canadian census of population, the population within the Great Lakes Basin was 35,371,814 residents. As shown in Table 1, this represents an increase of 1,569,562 (4.6%) residents since the 2010 U.S. census and 2011 Canadian census. By nation, the U.S. resident population increased by 369,367 (1.6%) to 22,943,937 residents in 2020, and the Canadian resident population increased by 1,200,195 (10.7%) to 12,427,877 residents in 2021, which represents 64.9% and 35.1% of the basin population, respectively (Table 2).

Considering the land drainage area, the Great Lakes Basin is 515,209 square kilometers (198,923 square miles) (Table 4). The average binational population density across the Great Lakes Basin was 68.7 residents per square kilometer (177.8 residents per square mile) (Table 2). By nation, the U.S. portion of the Great Lakes Basin is 289,827 square kilometers (111,903 square miles) and the Canadian portion of the basin is 225,383 square kilometers (87,021 square miles), encompassing 56.3% and 43.7% of the basin area, respectively. In 2020, the average U.S. population density was 79.2 residents per square kilometer (205.0 residents per square mile) whereas the average Canadian population density in 2021 was 55.1 residents per square kilometer (142.8 residents per square mile). Maps of the distribution of the 2020 U.S. and 2021 Canadian population across the Great Lakes Basin and Great Lakes Region are shown in Figure 1 and Figure 2, respectively.

There are currently more residents in the U.S. side of the basin, and U.S. population density is greater despite having slightly more land drainage area than the Canadian portion of the basin. However, Canada's population within the Great Lakes Basin grew faster than the U.S. population within the basin, both in terms of population count and percent change, over the short-term (10-year), 30-year, and long-term (50-year) time frames. In fact, Canada's population growth rates were more than three times greater than the U.S. growth rates within the basin (Table 1).

The Great Lakes Basin population has continued to increase over the 30-year and long-term (50-year) periods. Since the 1990/1991 census reports, the resident population within the Great Lakes Basin increased by 4,441,359 (14.4%) from 30,930,455 residents in 1990/1991 to 35,371,814 residents in 2020/2021, which is an average increase of 148,045 residents annually over those 30 years (Table 1). By nation, the U.S. resident population increased by 711,457 (3.2%) from 22,232,480 residents in 1990 to 22,943,937 residents in 2020, an average increase of 23,715 residents per year. The Canadian resident population increased by 3,729,902 (42.9%) from 8,697,975 residents in 1991 to 12,427,877 in 2021 – an average increase of 124,330 residents per year, which was more than five times the growth compared to the U.S portion of the basin (Figure 3).

Since 1970/1971 census reports, the Great Lakes Basin resident population increased by 7,405,631 (26.5%) from 27,966,183 residents in 1970/1971 to 35,386,661 residents in 2020/2021, which was an increase of 148,410 residents annually over those 50 years (Table 1). By nation, the U.S. resident population increased by 1,560,082 (7.3%) from 21,383,855 residents in 1970 to 22,943,937 residents in 2020 – an average increase of 31,202 annually. The Canadian resident population increased by 5,845,328 (88.8%) from 6,582,328 residents in 1971 to 12,427,877 in 2021 – an average increase of 116,911 residents per year, which was nearly four times the growth rate compared to the U.S portion of the basin (Figure 4).

Maps of the distribution of the 2020 U.S. and 2021 Canadian population across the Great Lakes Basin and Great Lakes Region are shown in Figure 1 and Figure 2, respectively.

The larger Great Lakes Region, comprised of eight U.S. states – Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania, and Wisconsin – and the Province of Ontario, has also experienced a substantial population increase (Table 5). According to the 2020 U.S. Census, of the 331,449,281-resident population of the United States,[2] 86,278,976 (26.0%) resided within the eight Great Lakes states (USCB, 2021e). Of those Great Lakes states' residents, 22,943,937 (26.6%) resided within the Great Lakes Basin, which comprise 7.0 percent of the nationwide population (Table 2 and Figure 2). According to the 2021 Canadian census of population, of the 36,991,981-resident population of Canada, 14,223,942 (38.5%) resided in the Province of Ontario, and 12,427,877 (87.4%) of Ontarians resided within the Great Lakes Basin, which comprise 33.6 percent of the nationwide population (Statistics Canada, 2022) (Table 2 and Figure 2).

Trend Analysis by Lake Basin

Lake Superior Basin

The Lake Superior Basin is shared by the Province of Ontario and the states of Minnesota, Wisconsin, and Michigan, and is the least populous among the five Great Lakes sub-basins (Figure 1). Based on the 2020 U.S. census and the 2021 Canadian census of population, the resident population within the Lake Superior Basin was 590,897 people, an increase of 3,473 residents (0.6%) since 2010 U.S. census and 2011 Canadian census (Table 1). By nation, the U.S. resident population increased by 2,013 (0.5%) to 424,473 residents in 2020 and the Canadian resident population increased by 1,460 (0.9%) to 169,897 residents in 2021, which represent basin population proportions of 71.4% and 28.6%, respectively.

Considering the land drainage area, the Lake Superior Basin is 126,136 square kilometers (48,701 square miles), which ranks second behind only the Lake Huron Basin (Table 4). Based on the 2020/2021 population counts, the binational population density across the Lake Superior Basin was 4.7 residents per square kilometer (12.2 residents per square mile), which was the least dense among the five Great Lakes sub-basins (Table 2). By nation, the U.S. portion of the basin is 42,896 square kilometers (16,562 square miles) and the Canadian portion of the basin is 83,240 square kilometers (32,139 square miles), which are 34.0% and 66.0% of the basin land drainage, respectively. In 2020, the U.S. population density was 9.9 residents per square kilometer (25.6 residents per square mile) whereas the Canadian population density was 2.0 residents per square kilometer (5.3 residents per square mile) in 2021. This demonstrates that not only are there more residents in the U.S. portion of the basin, but also the U.S. population density was nearly five times greater than that of Canada's population density within the basin. This difference in population density is due in part to the smaller portion of the land drainage area within the U.S compared to Canada. Other explanatory factors, such as socio-cultural, economic, and environmental factors, should be explored.

For the 30-year trend, the resident population decreased by 26,368 (-4.2%) from 620,738 residents in 1990/1991 to 594,370 residents in 2020/2021 (Table 1). By nation, the U.S. resident population decreased by 13,408 (-3.1%) from 437,881 residents in 1990 to 424,473 residents in 2020, with a large decline between 1990 and 2000 followed by a slight increasing trend from 2000 to 2020. Likewise, the Canadian resident population decreased by 12,960 (-7.1%) from 182,857 residents in 1991 to 169,897 in 2021. It should be noted that the decline in the Canadian population was more consistent during this period than the U.S. population (Figure 3).

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[2] Includes the resident population of the 50 states and the District of Columbia and excludes the 3,641,780-resident population of the five U.S. Territories (American Samoa, Guam, Northern Mariana Islands, Puerto Rico, and U.S. Virgin Islands), as ascertained by the 2020 U.S. Census.

For the long-term (50-year) trend, the resident population decreased by 3,125 (-0.5%) from 597,495 residents in 1970/1971 to 594,370 residents in 2020/2021 (Table 1). The Lake Superior Basin population increased from 597,495 residents in 1970/1971 and to 620,738 residents at 1990/1991 census surveys. Between the 1990/1991 and 2020/2021 census surveys, the population declined from 620,738 residents to 594,370 residents, which is similar to the population of 1970/1971. By nation, the U.S. resident population within the Lake Superior Basin increased from 426,752 residents in 1970 to 437,881 residents in 1990, and then declined to 424,473 residents in 2020. Likewise, the Canadian resident population increased from by 170,743 residents in 1971 to 182,857 in 1991, and then declined to 169,897 residents in 2021 (Table 2).

In summary, these trends demonstrate that the change of resident population within the Lake Superior Basin increased from 1970/1971 to 1990/1991, followed by a period of population decrease between 1990/1991 and 2020/2021 to levels similar to 50 years ago.

Lake Michigan Basin

The Lake Michigan Basin is shared by the states of Wisconsin, Illinois, Indiana, and Michigan, and is the third most populous among the five Great Lakes sub-basins. Uniquely, the Lake Michigan Basin lies wholly within the United States and has no drainage area within Canada (Table 4 and Figure 1). Based on the 2020 U.S. census, the resident population within the Lake Michigan Basin was 8,011,470 people, which is an average increase of 255,799 (3.2%) residents since the 2010 U.S. census (Table 1).

Considering land drainage area, the Lake Michigan Basin is 116,392 square kilometers (44,939 square miles), which ranks third among the five Great Lakes sub-basins (Table 4). In 2020, the population density was 68.8 residents per square kilometer (178.3 residents per square mile), which also ranks third among the five Great Lakes sub-basins (Table 2).

For the 30-year trend, the resident population increased by 704,211 (9.6%) from 7,307,259 residents in 1990 to 8,011,470 residents in 2020, which was an average increase of 23,474 residents annually over those 30 years (Table 1 and Figure 3). This was the second largest increase in population count over this time frame, trailing only the increase of 2.7 million residents within the Lake Ontario Basin.

For the long-term (50-year) trend, the resident population increased by 1,585,983 (24.7%) from 6,425,487 residents in 1970 to 8,011,470 residents in 2020, which was an average increase of 31,720 residents annually over those 50 years (Table 1 and Figure 4). This was the second largest increase in population count over this time frame, trailing only the increase of 4.2 million residents within the Lake Ontario Basin.

In summary, these trends demonstrate that the growth of the U.S. resident population within the Lake Michigan Basin has slowed slightly since 1970 yet maintains steady growth with an average of 25,000 new residents annually over the past decade.

Notably, the Chicago diversion, located in Chicago, Illinois, diverts water from the Lake Michigan watershed into the Upper Mississippi River basin, reducing the land drainage area of the Lake Michigan watershed. The Chicago diversion began in the early 1800s and increased in 1900 after the Chicago Sanitary and Ship Canal was completed.[3] Consequently, the Lake Michigan Basin is narrow to the surface water of Lake Michigan in northeast Illinois, northwest Indiana, and southeast Wisconsin, which include parts of several densely populated metro areas, such as Chicago, Illinois, and Milwaukee, Wisconsin. For this report, new geospatial methods were used to determine population, which only included the residents within the basin and excluded the surrounding metro

[3] For an overview of Great Lakes diversions, see https://www.ijc.org/en/lsbc/watershed/great-lakes-diversions.

population outside the basin (Figure 1). Previous SOGL reports employed different methods to estimate population of metro areas, which intentionally included metro residents outside the basin because their drinking water source was Lake Michigan and was an overestimate.

Lake Huron Basin (including St. Marys River)

The Lake Huron Basin is shared by the Province of Ontario and the State of Michigan, and is the second least populous among the five Great Lakes sub-basins, greater than only the resident population within the Lake Superior Basin (Figure 1). Based on the 2020 U.S. census and the 2021 Canadian census of population, the population within the Lake Huron Basin was 3,199,891 residents, which is an increase of 138,502 (4.5%) since the 2010 U.S. census and 2011 Canadian census (Table 1). By nation, the U.S. resident population decreased 10,928 (-0.7%) to 1,563,597 residents whereas the Canadian resident population increased 149,430 (10.1%) to 1,636,294 residents, which represent basin population proportions of 48.9% and 51.1%, respectively. The 2021 Canadian census of population marks the first time that the Canadian population within the Lake Huron Basin is more than the U.S. population.

Considering land drainage area, the Lake Huron Basin is 133,360 square kilometers (51,491 square miles), which is the largest among the five Great Lakes sub-basins (Table 4). Based on the 2020/2021 population counts, the binational population density across the Lake Huron Basin was 24.0 residents per square kilometer (62.1 residents per square mile), which only surpassed the population density within the Lake Superior Basin (Table 2). By nation, the U.S. portion of the basin is 42,125 square kilometers (16,264 square miles) and the Canadian portion of the basin is 91,236 square kilometers (35,226 square miles), which are 31.6% and 68.4% of the basin land drainage area, respectively. In 2020, the U.S. population density was 37.1 residents per square kilometer (96.1 residents per square mile) whereas Canadian population density was 17.9 residents per square kilometer (46.5 residents per square mile) in 2021 (Table 2). This demonstrates that although the Canadian population within the basin is slightly more than the U.S. resident population, the U.S. population density is double the Canadian population density. This difference in population density is due to the U.S portion of the basin being half the land drainage area compared the Canadian portion of the basin.

For the 30-year trend, the resident population across the Lake Huron Basin increased by 477,515 (17.5%) from 2,722,376 residents in 1990/1991 to 3,199,891 residents in 2020/2021, which was an average increase of 15,917 residents annually over those 30 years (Table 1). By nation, the U.S. resident population increased by 60,379 (4.0%) from 1,503,218 residents in 1990 to 1,563,597 residents in 2020 – an average increase of 2,013 residents annually. By contrast, the Canadian resident population increased 417,136 (34.2%) from 1,219,158 residents in 1991 to 1,636,294 in 2021 – an average increase of 13,905 residents annually. This demonstrates the Canadian population within the Lake Huron Basin grew nearly seven times faster than the population within the U.S portion of the basin over the past 30 years.

For the long-term (50-year) trend, the resident population across the Lake Huron Basin increased by 891,318 (38.6%) from 2,308,573 residents in 1970/1971 to 3,199,891 residents in 2020/2021, which was an average increase of 17,826 residents annually over those 50 years (Table 1). By nation, the U.S. resident population increased by 189,544 (13.8%) from 1,374,053 residents in 1970 to 1,563,597 residents in 2020 – an average increase of 3,791 residents annually. In contrast, the Canadian resident population increased by 701,774 (75.1%) from 934,520 residents in 1971 to 1,636,294 in 2021 – an average increase of 14,035 residents annually. This demonstrates the Canadian population within the Lake Huron Basin grew nearly four times faster than the population within the U.S portion of the basin over the past 50 years.

In summary, the growth of the U.S. resident population within the Lake Huron Basin has been slowing since 1970 and shrunk by nearly 1,100 residents annually since 2010, even as the State of Michigan's population grew by nearly 200,000 residents (Table 5). This decrease in U.S. population within the basin over the past decade may have been due to the Flint Water Crisis, driving the continued decline in Flint's population, the most populous city in the basin, among other factors (Figure 1). In contrast, the growth of the Canadian population within the basin has maintained steady growth since 1971, including a sharp increase of 94,165 new residents between 2016 and 2021 – the second greatest five-year population growth within the Canadian portion of the Lake Huron Basin since 1971, the earliest population by watershed is reported (Table 3).

Lake Erie Basin (including St. Clair-Detroit River Ecosystem)

The Lake Erie Basin is shared by Ontario and the states of Indiana, Michigan, Ohio, Pennsylvania, and New York, and is the most populous among the five Great Lakes sub-basins (Figure 1). Based on the 2020 U.S. census and the 2021 Canadian census of population, the population within the Lake Erie Basin was 12,399,519 residents, which is an increase of 363,401 (3.0%) since the 2010 U.S. census and the 2011 Canadian census (Table 1). By nation, the U.S. resident population increased by 104,014 (1.1%) to 9,946,913 residents in 2020 and the Canadian resident population increased by 259,387 (11.8%) to 2,452,606 in 2021, which represent basin population proportions of 80.2% and 19.8%, respectively. This is the largest difference in population proportion between the two countries among the four binationally shared Great Lakes sub-basins. This difference is due to the heavily populated metro areas on the U.S. side of the basin (e.g., Detroit, Toledo, Cleveland) compared to substantially smaller cities on the Canadian side (e.g., Windsor, Kitchener, London) (Figure 1).

Considering land drainage area, the Lake Erie Basin is 76,441 square kilometers (29,514 square miles), which is the second smallest, superseding only the Lake Ontario Basin (Table 4). Based on the 2020/2021 population counts, the binational population density across the Lake Erie Basin was 162.2 residents per square kilometer (420.1 residents per square mile), which was the second most dense among the five Great Lakes sub-basins, ranking behind only the Lake Ontario Basin's population density (Table 2). By nation, the U.S. portion of the basin is 53,894 square kilometers (20,809 square miles) and the Canadian portion of the basin is 22,547 square kilometers (8,705 square miles), which are 70.5% and 29.5% of the basin's land drainage area, respectively. In 2020, the U.S. population density was 184.6 residents per square kilometer (478.0 residents per square mile), which was the second most densely populated area of the Great Lakes Basin trailing only the Canadian portion of the Lake Ontario Basin. In comparison, the Canadian population density was 108.8 residents per square kilometer (281.7 residents per square mile) in 2021 (Table 2).

For the 30-year trend, the resident population increased by 655,962 (5.6%) from 11,743,557 residents in 1990/1991 to 12,399,519 residents in 2020/2021, which was an average increase of 21,865 residents annually over those 30 years (Table 1). By nation, the U.S. resident population increased by 28,191 (0.3%) from 9,918,722 residents in 1990 to 9,946,913 residents in 2020 – an average increase of only 940 residents annually. Conversely, the Canadian resident population increased by 627,771 (34.4%) from 1,824,835 residents in 1991 to 2,452,606 in 2021 – an average increase of 20,926 residents annually. This demonstrates the Canadian population within the Lake Erie Basin grew nearly 22 times faster than the population within the U.S portion of the basin over the past 30 years.

For the long-term (50-year) trend, the resident population increased by 642,654 (5.5%) from 11,756,865 residents in 1970/1971 to 12,399,519 residents in 2020/2021, which was an average increase of 12,853 residents annually over those 50 years (Table 1). By nation, the U.S. resident population decreased by 328,263 (-3.2%) from 10,275,176 residents in 1970 to 9,946,913 residents in 2020 – an average decrease of 6,565 residents annually. In contrast, the Canadian resident population increased by 970,917 (65.5%) from 1,481,689 residents in 1971 to 2,452,606 in 2021 – an average increase of 19,418 residents annually over the past 50 years.

In summary, the U.S. resident population within the Lake Erie Basin has declined since 1970. Although the U.S. population within the basin has grown by 10,000 new residents annually this past decade, it still has not returned to its 1970 population count. In contrast, the Canadian population within the basin grown tremendously since 1971, including a sharp increase of 186,339 new residents between 2016 and 2021 – the greatest five-year population growth within the Canadian portion of the Lake Erie Basin since 1971, the earliest population by watershed is reported (Table 3).

Lake Ontario Basin (including Niagara River)

The Lake Ontario Basin experienced the greatest increase in growth among all five Great Lakes sub-basins, both in terms of population count and percent change, for the short-term (10-year), 30-year, and long-term (50-year) time frames.

The Lake Ontario Basin is shared by the province of Ontario and the states of Pennsylvania and New York, and is the second most populous among the five Great Lakes sub-basins, trailing only the population across the Lake Erie Basin (Figure 1). Based on the 2020 U.S. census and the 2021 Canadian census of population, the population within the Lake Ontario Basin was 11,166,564 residents, which is an increase of 808,387 (7.8%) since the 2010 U.S. census and 2011 Canadian census of population (Table 1). By nation, the U.S. resident population increased by 18,469 (0.6%) to 2,997,484 residents in 2020 whereas the Canadian resident population increased by 789,918 (10.7%) to 8,169,080 in 2021, which represent basin population proportions of 26.8% and 73.2%, respectively. This is the second largest difference in population proportion between the two countries, trailing only the Lake Erie Basin.

Notably, the Lake Ontario Basin is one of only two sub-basins (joined by the Lake Huron Basin as of 2021) where there are more Canadian residents than U.S. residents. In fact, the Canadian population within the Lake Ontario Basin has outnumbered the U.S. population since at least 1970/1971 – the earliest that populations by watershed are reported.

Considering land drainage area, the Lake Ontario Basin is 62,879 square kilometers (24,278 square miles), which is the smallest among the five Great Lakes sub-basins (Table 4). Based on the 2020/2021 population counts, the binational population density across the Lake Ontario Basin was 177.6 residents per square kilometer (459.9 residents per square mile), which was the densest among the five Great Lakes sub-basins (Table 2). By nation, the U.S. portion of the basin is 34,519 square kilometers (13,328 square miles) and the Canadian portion of the basin is 28,360 square kilometers (10,950 square miles), which are 54.9% and 45.1% of the basin's land drainage area, respectively. In 2020, the U.S. population density was 86.8 residents per square kilometer (224.9 residents per square mile). In comparison, the Canadian population density was 288.0 residents per square kilometer (746.0 residents per square mile) in 2021, which was the most densely populated area of the Great Lakes Basin (Table 2). The difference between the Canadian and U.S. population density is due to the more densely populated metro areas in the Canadian portion of the basin (e.g., Hamilton, Toronto).

For the 30-year trend, the resident population increased by 2,630,039 (30.8%) from 8,536,525 residents in 1990/1991 to 11,166,564 residents in 2020/2021 (Table 1). By nation, the U.S. resident population decreased by 67,916 (-2.2%) from 3,065,400 residents in 1990 to 2,997,484 residents in 2020 – an average decrease of 2,264 residents annually. In contrast, the Canadian resident population increased by 2,630,039 (49.3%) from 5,471,125 residents in 1991 to 8,169,080 in 2021 – an average increase of 89,932 residents annually. This demonstrates for every U.S. resident that left the basin, nearly 39 Canadians were added to the basin annually over the past 30 years.

For the long-term (50-year) trend, the resident population increased by 4,288,801 (62.4%) from 6,877,763 residents in 1970/1971 to 11,166,564 residents in 2020/2021 (Table 1). By nation, the U.S. resident population increased by 115,097 (4.0%) from 2,882,387 residents in 1970 to 2,997,484 residents in 2020 – an average increase of 2,302 residents annually. In contrast, the Canadian resident population increased 4,173,704 (104.5%) from 3,995,376 residents in 1971 to 8,169,080 in 2021 – an average increase of 83,474 annually. This demonstrates the Canadian population within the Lake Ontario Basin grew 36 times faster than the population within the U.S portion of the basin over the past 50 years.

Linkages

Human population is related to many stressors around the Great Lakes Basin and tracking the resident population within the basin provides important context for many ecosystem changes. The following linkages are a few examples of possible connections between human population and the nine high-level indicators analyzed through State of the Great Lakes reporting.

- Drinking Water:
    - Rapid growth may strain water systems due to increased demand for water treatment infrastructure including wastewater and drinking water treatment systems (Mikovits et al., 2014).
- Beaches:
    - Without proper infrastructure investments, population growth could lead to increased fecal pollution, including *E. coli*, due to increased strain on wastewater treatment and also changes to agricultural practices (Santo-Domingo and Ashbold, 2008).
    - Increases in the human population are often associated with increases in impermeable surfaces, potentially leading to higher amounts of runoff that can cause combined sewer overflows and beach advisories (Chithra et al., 2015; Salerno et al., 2018).
- Fish Consumption:
    - Increases in human populations will likely increase fishing trips within the Great Lakes (Hunt et al., 2021), leading to increased sport fish consumption.
- Toxic Chemicals:
    - Toxic Chemicals in the Atmosphere sub-indicator: Atmospheric deposition is a significant pathway of entry for pollutants within the Great Lakes, including persistent organic pollutants. Urban areas are a major source of persistent organic pollutants in the atmosphere due to increased use and emission rates (Melymuk et al., 2011).
- Habitat and Species:
    - Rapid population growth can lead to the development of naturalized areas causing habitat loss and increased anthropogenic stress on the species endemic to the Great Lakes. Additional stressors related to areas of high population density include increases in plastic pollution and other pollution sources (Baldwin et al., 2016; Jenny et al., 2020).
    - Fish Related Sub-Indicators: The direct and indirect effects of climate change in the Great Lakes Basin may cause changes in thermal habitat, prey availability, alter reproductive phenology, and the fish community structure (Collingsworth et al., 2017).

- Nutrients and Algae:
  - Nutrients in Lakes and Harmful Algal Blooms sub-indicators: Activities such as agricultural production to feed a growing population, urbanization, wastewater discharge, and the use of fossil fuels and household products contribute to excess nutrients in the Great Lakes Basin that can spur the growth of harmful algal blooms (HABs) (Wurtsbaugh et al., 2019).
- Invasive Species:
  - Changes in climate from anthropogenic climate change may remove the temperature barriers for warmer water invasive species in the Great Lakes (Van Zuiden et al., 2016).
- Groundwater:
  - Highly populated areas can affect groundwater in various ways. Increased impervious areas can reduce natural aquifer replenishment through the soil-zone transfer of precipitation. At the same time, pollution from leakage from the water supply, sewage pipes, and septic systems may enter shallower groundwater systems. Other pollution sources associated with residential areas, such as the application of road salt and pesticides, can also make their way into shallow aquifers that may discharge directly into the Great Lakes (Howard and Gerber, 2018).
- Watershed Impacts and Climate Trends:
  - Structures associated with densely populated areas, including dams, hardened stream channels, and increased impervious surfaces, can change the hydrology of streams and the transport of materials from the watershed into the Great Lakes (Graf, 1999; Trudeau et al., 2015; DeBues et al., 2019).
  - Surface Water Temperatures and Ice-Cover sub-indicators: The effects of anthropogenically induced climate change on the Great Lakes will be wide-ranging, including decreased ice cover and changes to the timing of stratification (Mason et al., 2016; Collingsworth et al., 2017). In fact, the Great Lakes are already experiencing warming winters and an earlier onset of stratification (Anderson et al., 2021).
  - Hardened Shorelines and Land Cover sub-indicators: Increased populations may increase the development of natural land, which can lead to a decrease in biodiversity, overall species abundance, reductions in habitat, and alterations in the transport of shoreline sediments (Shear et al., 2003; Gittman et al., 2016).

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable, and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada		X		
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Refer to Statistics Canada and U.S. Census Bureau sources in "Information Sources" Contact the authors for the digital files of the Tables and Figures.		

Data Limitations

- Census data from the United States and Canada do not perfectly align because their censuses are conducted in different years and at different frequencies. The United States conducts a decennial census on year zero of each decade (e.g., 2000, 2010, 2020). Canada conducts a census every five years on year 1 and 6 of the decade (e.g., 2011, 2016, 2021).
- Statistics Canada calculates population by watershed so the Canadian population within the Great Lakes Basin included in this report is reproduced without adjustment. Conversely, the U.S. Census Bureau does not report population by watershed, and thus the population within the U.S. portion of the Great Lakes Basin must be calculated by the EPA.
- Differing from previous SOGL reports, the method employed to calculate the resident population within the U.S. portion of the Great Lakes Basin used population counts at the finest geographic unit available (block group level) and geospatial software to conduct the analysis. For the SOGL 2025 and 2028, the American Community Survey (ACS) 2020-2024 5-year estimates at the block group level is the finest intercensal data product that is expected to be available and should be used to calculate the U.S. resident population within the Great Lakes Basin (USCB, 2021b).
- Due to changes in methods to determine the U.S. population within the Great Lakes Basin, U.S. population estimates (and thus basin-wide population counts) described in this report are not comparable to previous SOGL reports. Due to recalculations in the watershed level data by Statistics Canada in 2017, the Canadian population counts described in this report are not comparable to previous SOGL reports. See the tables for the most recent data available and their sources.

Additional Information

According to the United Nations, key global demographic trends are population growth, population ageing, migration, and urbanization, which all hold important implications for economic and social development and environmental sustainability (United Nations, 2019). Due to the high impact urban areas may have on the surrounding environment – such as the development of natural or agricultural land (i.e., land-use change), increased demand on water infrastructure, and increased nutrient inputs (Furberg and Ban, 2012; Mikovits et al., 2014) – it is important to examine these changes across the population centers within the Great Lakes Basin. In the Province of Ontario, the Greater Toronto Area (GTA) – comprised of municipalities (i.e., census subdivision) along the northwestern shore of Lake Ontario – has been and is projected to continue to be the fastest growing region in the Great Lakes Basin. This area is projected to increase of 2.6 million residents (36.7% growth) to reach a population over 9.5 million by 2046 (Ministry of Finance, 2021). This means that nearly half of all Canadian residents within the Great Lakes Basin are anticipated to reside in the GTA by 2046.

Over the past decade, the population of most U.S cities (with populations greater than 50,000) within the Great Lakes Basin decreased, including Chicago, IL (-0.1%), Detroit, MI (-6.1%), Milwaukee, WI (-0.8%), Cleveland, OH (-4.0%), Toledo, OH (-5.0%), Buffalo, NY (-2.3%), Rochester, NY (-2.3%), Flint, MI (-6.7%), Erie, PA (-6.2%) and Gary, IN (-6.7%). Despite the decrease, Chicago remains the third most populous city in the United States followed by Detroit (24th) and Milwaukee (31st). Notably, all the other U.S. cities near the top of the population rankings are growing while the Great Lakes cities are shrinking (USCB, 2021d). However, this ranking is of the population of cities, which excludes the populations of the surrounding metro areas. Upon examining the metropolitan statistical areas (MSA) in the Great Lakes Basin over the last decade, the population of the Chicago-Naperville-Elgin (IL-IN-WI) metro area was unchanged and the population of the Detroit-Warren-Dearborn, MI metro area grew slightly (0.5%) (USCB, 2021c). These two examples demonstrate the migration of residents from cities into the surrounding metro areas (i.e., urban sprawl), which is consistent with the longer-term population shift away from “Rust Belt” cities – i.e., industrial cities along the Great Lakes whose populations peaked and sharply declined since the 1970s (Hartley, 2013).

Urban sprawl has also been documented in Canada as population growth between 2011 and 2016 was higher among peripheral municipalities (+6.9%) compared with central municipalities (+5.8%) (Statistics Canada, 2019). The binational urban sprawl observed in the past decade is consistent with previously reported trends within the basin (Great Lakes Science Advisory Board et al., 2009).

While increasing populations in urban areas can mean greater stress on the Great Lakes, this is not necessarily the case. Sustainable development requires that cities expand and grow in an environmentally sensitive way and not compromise the environment for future generations. Adapting sustainable practices such as stormwater treatment, sustainable waste management, and green infrastructure can limit anthropogenic pressures (Pickett et al., 2013). Determining the effects of population growth on the surrounding environment and tracking sustainable practices are complex issues beyond the purpose of this sub-indicator. Many of the other sub-indicators in the State of the Great Lakes report assess the effects of the human population on those specific topics. For example, the land-use sub-indicator analyzes the changes in urbanized areas around the Great Lakes, and the toxic chemicals sub-indicators assess the concentrations of chemicals within the Great Lakes resulting from human activity. This sub-indicator serves as a record of population change within the Great Lakes Basin and does not aim to assess the effects of those population changes.

Indigenous Populations

The first inhabitants of the Great Lakes Basin arrived over 10,000 years ago as the mile-high glaciers receded from the region. In 1620, when Étienne Brûlé became the first European to land at the site of Sault Ste. Marie, the population in present-day Michigan was about 15,000 Indigenous peoples. Other researchers have estimated the population of Indigenous peoples in the Great Lakes Basin – termed *Nayaano-nibiimaang Gichigamiin* in Anishinaabemowin, translating to *The Five Freshwater Seas* – was between 60,000 and 117,000 in the 16th century, when Europeans began their search for a passage to the Orient through the Great Lakes (Fuller, 1995).

Approximately 120 First Nations and Tribes, as well as many Métis communities, have occupied the Great Lakes Basin over the course of history. Currently, in Ontario alone, there are 61 First Nation communities located within the Great Lakes basin (Union of Ontario Indians, 2016). Within the Great Lakes Basin, there are also 21 Métis Nation of Ontario (MNO) community councils in addition to the Jackfish Métis Association, French River Métis, Historic Métis of Saugeen, and the Red Sky Métis Nation. In 2020, the 61 First Nation communities in the Canadian Great Lakes Basin accounted for approximately 153,000 residents (Government of Canada, 2021). While the exact population of Métis citizens in the Great Lakes basin is unknown, 120,585 people in Ontario identified as Métis in the 2016 Canadian census of population (Statistics Canada, 2017a).

In the United States, there are 42 Indian Tribal Nations recognized by the U.S. Government whose reservations are across Minnesota, Wisconsin, Michigan, and New York, and/or who may retain treaty guaranteed rights to hunt, fish or gather in areas ceded to the United States in various treaties (Figure 5). In 2005, the total enrolled membership of these Tribal Nations was about 175,000 (GLRC, 2005).

U.S. Tribes in the Great Lakes Region

This list includes Tribes with present day reservations and/or ceded territory usufructuary rights and Inter-Tribal treaty resource/environmental organizations in the U.S. Great Lakes region (EPA, 2021a; EPA, 2021b; U.S. Caucus of the Traditional Ecological Knowledge Task Team, 2021).

Tribes in Michigan

- Bay Mills Indian Community
- Grand Traverse Band of Ottawa and Chippewa Indians
- Hannahville Indian Community
- Keweenaw Bay Indian Community
- Lac Vieux Desert Band of Lake Superior Chippewa Indians
- Little River Band of Ottawa Indians
- Little Traverse Bay Bands of Odawa Indians
- Match-E-Be-Nash-She-Wish Band of Potawatomi Indians (Gun Lake Tribe)
- Nottawaseppi Huron Band of Potawatomi
- Pokagon Band of Potawatomi
- Saginaw Chippewa Indian Tribe of Michigan
- Sault Ste. Marie Tribe of Chippewa Indians

Tribes in Minnesota

- Bois Forte Band of Chippewa
- Fond du Lac Band of Lake Superior Chippewa Indians

- Grand Portage Band of Lake Superior Chippewa Indians
- Leech Lake Tribe of Ojibwe
- Lower Sioux Community
- Mille Lacs Band of Ojibwe
- Prairie Island Indian community
- Red Lake Band of Chippewa
- Shakopee Mdewakanton Sioux Community
- Upper Sioux Community
- White Earth Band of Chippewa

Tribes in New York

- Cayuga Nation
- Oneida Indian Nation
- Onondaga Nation
- Saint Regis Mohawk Tribe
- Seneca Nation of Indians
- Shinnecock Indian Nation
- Tonawanda Band of Seneca
- Tuscarora Nation

Tribes in Wisconsin

- Bad River Band of Lake Superior Chippewa Indians
- Forest County Potawatomi Community
- Ho-Chunk Nation
- Lac Courte Oreilles Band of Lake Superior Chippewa Indians
- Lac du Flambeau Band of Lake Superior Chippewa Indians
- Menominee Indian Tribe of Wisconsin
- Oneida Nation of Wisconsin
- Red Cliff Band of Lake Superior Chippewa Indians
- Saint Croix Chippewa Indians of Wisconsin
- Sokaogon Chippewa Community (Mole Lake)
- Stockbridge-Munsee Indian Community

First Nation Communities in Canada listed by Great Lakes sub-basin

Lake Superior

- Animbiigoo Zaagi'igan Anishinaabek First Nation
- Biinjitiwaabik Zaaging Anishinaabek First Nation
- Biigtigong Nishnaabeg
- Bingwi Neyaashi Anishinaabek
- Brunswick House First Nation
- Chapleau Cree First Nation
- Chapleau Ojibway First Nation
- Fort William First Nation

- Ginoogaming First Nation
- Kiashke Zaaging Anishinaabek First Nation
- Long Lake #58 First Nation
- Missanabie Cree First Nation
- Michipicoten First Nation
- Netmizaaggamig Nishnaabeg
- Pays Plat First Nation
- Red Rock Indian Band
- Whitesand First Nation

Lake Huron

- Aamijwnaang First Nation
- Atikameksheng Anishnawbek
- Aundeck Omni Kaning First Nation
- Batchewana First Nation
- Beausoleil First Nation
- Chippewas of Georgina Island First Nation
- Chippewas of Kettle and Stony Point First Nation
- Chippewas of Rama First Nation
- Chippewas of Nawash Unceded First Nation
- Dokis First Nation
- Garden River First Nation
- Henvey Inlet First Nation
- M'Chigeeng First Nation
- Magnetawan First Nation
- Mississauga First Nation
  Moose Deer Point First Nation
- Nipissing First Nation
- Sagamok Anishnawbek First Nation
- Saugeen First Nation
- Serpent River First Nation
- Shawanaga First Nation
- Sheguiandah First Nation
- Sheshegwaning First Nation
- Thessalon First Nation
- Wahnapitae First Nation
- Wahta Mohawks
- Wasauksing First Nation
- Whitefish River First Nation
- Wikwemikong Unceded First Nation
- Zhiibaahaasing First Nation

Lake Erie

- Caldwell First Nation
- Chippewas of the Thames First Nation

- Delaware Nation of Moraviantown
- Mississaugas of the Credit First Nation
- Munsee-Delaware Nation
- Oneida Nation of the Thames
- Six Nations of the Grand
- Walpole Island First Nation

Lake Ontario

- Alderville First Nation
- Curve Lake First Nation
- Hiawatha First Nation
- Mississauga's of Scugog Island
- Mississaugas of the Credit First Nation
- Mohawks of the Bay of Quinte

From U.S. census records, data on race have been collected since the first U.S. decennial census in 1790. The 1860 Census was the first to enumerate American Indians as a separate race group, and the 1890 Census was the first to count American Indians throughout the country. Alaska Natives, in Alaska, have been counted in various respects since the 1880 Census, generally under the American Indian category, but were enumerated as a separate group starting with the 1940 Census. All states began collecting data separately for Eskimos and Aleuts in 1980. Since the 2000 Census, a combined response category – "American Indian or Alaska Native" – was used, and a dedicated write-in line to collect information on the American Indian and Alaska Native population. In 2010, there were 5,220,579 people across the U.S. who identified as American Indian or Alaska Native, which was 1.7 percent of the total U.S. population. Across the eight Great Lakes States, there were 870,686 American Indians and Alaska Natives, which was 16.7 percent of the U.S. American Indians and Alaska Natives population. By 2060, the projected U.S. American Indian and Alaska Native population is estimated to reach 10 million people, or approximately 2.4 percent of the U.S. population (Norris, 2012). Reported in the 2020 Census, 9.7 million people identified as American Indian and Alaska Native, which is 2.9 percent of the U.S. population (USCB, 2021a).

From Canadian census records, there were 1,673,785 Indigenous Peoples (i.e., First Nations people, Métis, and Inuit) counted in the 2016 Canadian census of population, accounting for 4.9 percent of the total population. This was up from 3.8 percent in 2006 and 2.8 percent in 1996. Since 2006, the population of Indigenous peoples has grown by 42.5 percent – more than four times the growth rate of the non-Indigenous population over the same period. According to population projections, the number of Indigenous peoples will continue to grow quickly. In the next two decades, the population of Indigenous peoples is likely to exceed 2.5 million persons (Statistics Canada, 2017a).

Additional Methods Information

Previously reported estimates of U.S. population within the Great Lakes Basin were the sum of the populations of the 209 counties with any land area within the Great Lakes Basin, which included the resident population outside the Great Lakes Basin within counties that are only partially within the Great Lakes watershed (i.e., straddling counties). This method is precise because population counts at the county level have been recorded since the first U.S. census in 1790, but it produces less accurate results of the resident population strictly within the Great Lakes Basin. See Table 6 for the population of U.S. counties within the Great Lakes Basin from 1900 to 2020.

In more recent SOGL reports, to estimate the resident population strictly within the Great Lakes Basin, the populations of straddling counties were ratio adjusted based on the counties' land drainage area within the Great

Lakes Basin. However, smaller geographic units of population (e.g., block groups) are published and allow for a more accurate estimate of the population strictly within the Great Lakes Basin.

To determine the resident population within the Great Lakes Basin and its five sub-basins, we used the following methods in GIS software (ArcGIS Pro 2.9. Redlands, CA: Environmental Systems Research Institute, Inc., 2021). The 'Clip (Analysis)' and 'Summarize Within (Analysis)' tools were used to identify the population within the Great Lakes Basin from the larger population of the eight Great Lakes states. Census block groups (smallest geographic unit published) that straddle the Great Lakes Basin boundaries were ratio adjusted by enabling geometric ratio with the 'Make Feature Layer (Data Management)' tool. Thus, only residents within the Great Lakes Basin were included in our population estimate. For a detailed methods document for ArcGIS procedures, please contact the authors.

Notably, due to our change in methods, the U.S. population within each sub-basin varies from previous reports, most notably the population within the Lake Michigan Basin. Beginning in the early 1800s, the Chicago diversion, located in Chicago, Illinois, diverts water from Lake Michigan watershed to the Upper Mississippi River basin, reducing the land drainage area of the Lake Michigan watershed. Consequently, the Lake Michigan Basin is narrow through Chicagoland, resulting in much of the densely populated Chicago Metro Area falling outside the Lake Michigan Basin and in the Upper Mississippi River watershed instead. Thus, aiming to include only the residents strictly within the Great Lakes Basin, the metro Chicago residents outside of the Lake Michigan Basin were excluded from these results. This change caused substantial variation in population results reported previously, which were intentionally included in earlier SOGL reports because their drinking water source was Lake Michigan and was an overestimate of basin population. For the population served drinking water sourced from the surface waters of the Great Lakes and connecting rivers, see the Drinking Water sub-indicator report.

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Last Updated

State of the Great Lakes 2022 Technical Report

Table 1. Population Trends within the Great Lakes Basin: 10-year trend, 30-year trend, & 50-year trend. Note: The '10-year Δ', '30-year Δ' and '50-year Δ' is the change from the census years indicated in parentheses compared with the 2020/2021 population count. This table is a subset from 'Table 2. Population within the Great Lakes Basin, 1970-2020, every 10 years.' See Table 2 for more information and data sources.

Basin	Portion	Current Data	10-year Trend (2010-2020 U.S. & 2011-2021 Canada)				30-year Trend (1990-2020 U.S. & 1991-2021 Canada)				Long-term (50-year) Trend (1970-2020 U.S. & 1971-2021 Canada)			
		2020/2021 Population Count	2010/2011 Population Count	10-Year Δ	10-Year % Δ	Avg Δ per annum	1990/1991 Population Count	30-Year Δ	30-Year % Δ	Avg Δ per annum	1970/1971 Population Count	50-Year Δ	50-Year % Δ	Avg Δ per annum
Superior	Binational	594,370	590,897	3,473	0.6%	347	620,738	-26,368	-4.2%	-879	597,495	-3,125	-0.5%	-63
	Ontario	169,897	168,437	1,460	0.9%	146	182,857	-12,960	-7.1%	-432	170,743	-846	-0.5%	-17
	USA	424,473	422,460	2,013	0.5%	201	437,881	-13,408	-3.1%	-447	426,752	-2,279	-0.5%	-46
Michigan	Binational	8,011,470	7,755,671	255,799	3.3%	25,580	7,307,259	704,211	9.6%	23,474	6,425,487	1,585,983	24.7%	31,720
	Ontario	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	USA	8,011,470	7,755,671	255,799	3.3%	25,580	7,307,259	704,211	9.6%	23,474	6,425,487	1,585,983	24.7%	31,720
Huron	Binational	3,199,891	3,061,389	138,502	4.5%	13,850	2,722,376	477,515	17.5%	15,917	2,308,573	891,318	38.6%	17,826
	Ontario	1,636,294	1,486,864	149,430	10.1%	14,943	1,219,158	417,136	34.2%	13,905	934,520	701,774	75.1%	14,305
	USA	1,563,597	1,574,525	-10,928	-0.7%	-1,093	1,503,218	60,379	4.0%	2,013	1,374,053	189,544	13.8%	3,791
Erie	Binational	12,399,519	12,036,118	363,401	3.0%	36,340	11,743,557	655,962	5.6%	21,865	11,756,865	642,654	5.5%	12,853
	Ontario	2,452,606	2,193,219	259,387	11.8%	25,939	1,824,835	627,771	34.4%	20,926	1,481,689	970,917	65.5%	19,418
	USA	9,946,913	9,842,899	104,014	1.1%	10,401	9,918,722	28,191	0.3%	940	10,275,176	-328,263	-3.2%	-6,565
Ontario	Binational	11,166,564	10,358,177	808,387	7.8%	80,839	8,536,525	2,630,039	30.8%	87,668	6,877,763	4,288,801	62.4%	85,776
	Ontario	8,169,080	7,379,162	789,918	10.7%	78,992	5,471,125	2,697,955	49.3%	89,932	3,995,376	4,173,704	104.5%	83,474
	USA	2,997,484	2,979,015	18,469	0.6%	1,847	3,065,400	-67,916	-2.2%	-2,264	2,882,387	115,097	4.0%	2,302
Overall		**35,371,814**	**33,802,252**	**1,569,562**	**4.6%**	**156,956**	**30,930,455**	**4,441,359**	**14.4%**	**148,045**	**27,966,183**	**7,405,631**	**26.5%**	**148,113**
Ontario portion		12,427,877	11,227,682	1,200,195	10.7%	120,020	8,697,975	3,729,902	42.9%	124,330	6,582,328	5,845,549	88.8%	116,911
USA portion		22,943,937	22,574,570	369,367	1.6%	36,937	22,232,480	711,457	3.2%	23,715	21,383,855	1,560,082	7.3%	31,202

Table 2. Population within the Great Lakes Basin, 1970-2020, every 10 years. *Continued next page.*

Basin	Portion	1970/1971 Population						1980/1981 Population					
		Count	% Basin	Decadal Δ	% Δ	Density		Count	% Basin	Decadal Δ	% Δ	Density	
						SQKM	SQMI					SQKM	SQMI
Superior	Binational	597,495	100%	-	-	4.7	12.3	612,908	100%	15,413	2.6%	4.9	12.6
	Ontario	170,743	28.6%	-	-	2.1	5.3	177,599	29.0%	6,856	4.0%	2.1	5.5
	USA	426,752	71.4%	-	-	9.9	25.8	435,309	71.0%	8,557	2.0%	10.1	26.3
Michigan	Binational	6,425,487	100%	-	-	55.2	143.0	6,699,412	100%	273,925	4.3%	57.6	149.1
	Ontario	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	USA	6,425,487	100%	-	-	55.2	143.0	6,699,412	100%	273,925	4.3%	57.6	149.1
Huron	Binational	2,308,573	100%	-	-	17.3	44.8	2,575,902	100%	267,329	11.6%	19.3	50.0
	Ontario	934,520	40.5%	-	-	10.2	26.5	1,058,199	41.1%	123,679	13.2%	11.6	30.0
	USA	1,374,053	59.5%	-	-	32.6	84.5	1,517,703	58.9%	143,650	10.5%	36.0	93.3
Erie	Binational	11,756,865	100%	-	-	153.8	398.4	11,535,352	100%	-221,513	-1.9%	150.9	390.8
	Ontario	1,481,689	12.6%	-	-	65.7	170.2	1,623,278	14.1%	141,589	9.6%	72.0	186.5
	USA	10,275,176	87.4%	-	-	190.7	493.8	9,912,074	85.9%	-363,102	-3.5%	183.9	476.3
Ontario	Binational	6,877,763	100%	-	-	109.4	283.3	7,404,032	100%	526,269	7.7%	117.7	305.0
	Ontario	3,995,376	58.1%	-	-	140.9	364.9	4,550,906	61.5%	555,530	13.9%	160.5	415.6
	USA	2,882,387	41.9%	-	-	83.5	216.3	2,853,126	38.5%	-29,261	-1.0%	82.7	214.1
Overall		**27,966,183**	**100%**	**-**	**-**	**54.3**	**140.6**	**28,827,606**	**100%**	**861,423**	**3.1%**	**56.0**	**144.9**
Ontario portion		6,582,328	23.5%	-	-	29.2	75.6	7,409,982	25.7%	827,654	12.6%	32.9	85.2
USA portion		21,383,855	76.5%	-	-	73.8	191.1	21,417,624	74.3%	33,769	0.2%	73.9	191.4

Table 2. Population within the Great Lakes Basin, 1970-2020, every 10 years (continued). Continued next page.

Basin	Portion	1990/1991 Population						2000/2001 Population					
		Count	% Basin	Decadal Δ	% Δ	Density SQKM	Density SQMI	Count	% Basin	Decadal Δ	% Δ	Density SQKM	Density SQMI
Superior	Binational	620,738	100%	7,830	1.3%	4.9	12.7	594,197	100%	-26,541	-4.3%	4.7	12.2
	Ontario	182,857	29.5%	5,258	3.0%	2.2	5.7	173,716	29.2%	-9,141	-5.0%	2.1	5.4
	USA	437,881	70.5%	2,572	0.6%	10.2	26.4	420,481	70.8%	-17,400	-4.0%	9.8	25.4
Michigan	Binational	7,307,259	100%	607,847	9.1%	62.8	162.6	7,558,850	100%	251,591	3.4%	64.9	168.2
	Ontario	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	USA	7,307,259	100%	607,847	9.1%	62.8	162.6	7,558,850	100%	251,591	3.4%	64.9	168.2
Huron	Binational	2,722,376	100%	146,474	5.7%	20.4	52.9	2,948,705	100%	226,329	8.3%	22.1	57.3
	Ontario	1,219,158	44.8%	160,959	15.2%	13.4	34.6	1,358,006	46.1%	138,848	11.4%	14.9	38.6
	USA	1,503,218	55.2%	-14,485	-1.0%	35.7	92.4	1,590,699	53.9%	87,481	5.8%	37.8	97.8
Erie	Binational	11,743,557	100%	208,205	1.8%	153.6	397.9	12,096,594	100%	353,037	3.0%	158.2	409.9
	Ontario	1,824,835	15.5%	201,557	12.4%	80.9	209.6	2,018,716	16.7%	193,881	10.6%	89.5	231.9
	USA	9,918,722	84.5%	6,648	0.1%	184.0	476.7	10,077,878	83.3%	159,156	1.6%	187.0	484.3
Ontario	Binational	8,536,525	100%	1,132,493	15.3%	135.8	351.6	9,316,990	100%	780,465	9.1%	148.2	383.8
	Ontario	5,471,125	64.1%	920,219	20.2%	192.9	499.7	6,354,843	68.2%	883,718	16.2%	224.1	580.4
	USA	3,065,400	35.9%	212,274	7.4%	88.8	230.0	2,962,147	31.8%	-103,253	-3.4%	85.8	222.2
Overall		**30,930,455**	**100%**	**2,102,849**	**7.3%**	**60.0**	**155.5**	**32,515,336**	**100%**	**1,584,881**	**5.1%**	**63.1**	**163.5**
Ontario portion		8,697,975	28.1%	1,287,993	17.4%	38.6	100.0	9,905,281	30.5%	1,207,306	13.9%	43.9	113.8
USA portion		22,232,480	71.9%	814,856	3.8%	76.7	198.7	22,610,055	69.5%	377,575	1.7%	78.0	202.1

Table 2. Population within the Great Lakes Basin, 1970-2020, every 10 years (continued)

Basin	Portion	2010/2011 Population						2020/2021 Population					
		Count	% Basin	Decadal Δ	% Δ	Density		Count	% Basin	Decadal Δ	% Δ	Density	
						SQKM	SQMI					SQKM	SQMI
Superior	Binational	590,897	100%	-3,300	-0.6%	4.7	12.1	594,370	100%	3,473	0.6%	4.7	12.2
	Ontario	168,437	28.5%	-5,279	-3.0%	2.0	5.2	169,897	28.6%	1,460	0.9%	2.0	5.3
	USA	422,460	71.5%	1,979	0.5%	9.8	25.5	424,473	71.4%	2,013	0.5%	9.9	25.6
Michigan	Binational	7,755,671	100%	196,821	2.6%	66.6	172.6	8,011,470	100%	255,799	3.3%	68.8	178.3
	Ontario	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	USA	7,755,671	100%	196,821	2.6%	66.6	172.6	8,011,470	100%	255,799	3.3%	68.8	178.3
Huron	Binational	3,061,389	100%	112,684	3.8%	23.0	59.5	3,199,891	100%	138,502	4.5%	24.0	62.1
	Ontario	1,486,864	48.6%	128,858	9.5%	16.3	42.2	1,636,294	51.1%	149,430	10.1%	17.9	46.5
	USA	1,574,525	51.4%	-16,174	-1.0%	37.4	96.8	1,563,597	48.9%	-10,928	-0.7%	37.1	96.1
Erie	Binational	12,036,118	100%	-60,476	-0.5%	157.5	407.8	12,399,519	100%	363,401	3.0%	162.2	420.1
	Ontario	2,193,219	18.2%	174,503	8.6%	97.3	251.9	2,452,606	19.8%	259,387	11.8%	108.8	281.7
	USA	9,842,899	81.8%	-234,979	-2.3%	182.6	473.0	9,946,913	80.2%	104,014	1.1%	184.6	478.0
Ontario	Binational	10,358,177	100%	1,041,187	11.2%	164.7	426.7	11,166,564	100%	808,387	7.8%	177.6	459.9
	Ontario	7,379,162	71.2%	1,024,319	16.1%	260.2	673.9	8,169,080	73.2%	789,918	10.7%	288.0	746.0
	USA	2,979,015	28.8%	16,868	0.6%	86.3	223.5	2,997,484	26.8%	18,469	0.6%	86.8	224.9
Overall		**33,802,252**	**100%**	**1,286,916**	**4.0%**	**65.6**	**169.9**	**35,371,814**	**100%**	**1,569,562**	**4.6%**	**68.7**	**177.8**
Ontario portion		11,227,682	33.2%	1,322,401	13.4%	49.8	129.0	12,427,877	35.1%	1,200,195	10.7%	55.1	142.8
USA portion		22,574,570	66.8%	-35,485	-0.2%	77.9	201.7	22,943,937	64.9%	369,367	1.6%	79.2	205.0

Note: 'Decadal Δ' is the decade-over-decade change in population count. '% Basin' is the proportion of the resident population within each basin.

Sources:

Canadian data:

2021 data: Statistics Canada, Environment and Energy Statistics Division, special tabulation from the 2021 Census of Population. Received February 23, 2022. (*Preliminary and subject to revision.*)

1971-2016 data: Statistics Canada. (2017). *Table 17-10-0117-01 Selected population characteristics, Canada, major drainage areas and sub-drainage areas.* https://doi.org/10.25318/1710011701-eng (accessed June 22, 2021).

U.S. data: (U.S. population counts were derived from authors' geospatial analyses of census block groups within the Great Lakes Basin from the following datasets)

2010 to 2020 data: esri_dm. (2021, October 1). *USA Census Block Groups Boundaries.* ArcGIS Online. https://esri.maps.arcgis.com/home/item.html?id=1c924a53319a491ab43d5cb1d55d8561 (accessed February 17, 2022).

1970 to 2000 data: Steven Manson, Jonathan Schroeder, David Van Riper, Tracy Kugler, and Steven Ruggles. *IPUMS National Historical Geographic Information System: Version 15.0 [Database].* Minneapolis, MN: IPUMS. 2020. http://doi.org/10.18128/D050.V15.0 (accessed June 17, 2021).

glc_esri. (2020, August 20). *Great Lakes & St. Lawrence River Basins.* ArcGIS Online. https://epa.maps.arcgis.com/home/item.html?id=df1de3fdd82047a79864b45c9ec1180b (accessed June 17, 2021).

Table 3. Population within the Great Lakes Basin in Canada, 1971-2021, every 5 years

Continued next page.

Basin	1971 Census					1976 Census					1981 Census				
	Count	5-year Δ	% Δ	Density SQKM	Density SQMI	Count	5-year Δ	% Δ	Density SQKM	Density SQMI	Count	5-year Δ	% Δ	Density SQKM	Density SQMI
Superior	170,743	-	-	2.1	5.3	175,602	4,859	2.8%	2.1	5.5	177,599	1,997	1.1%	2.1	5.5
Huron	934,520	-	-	10.2	26.5	1,018,331	83,811	9.0%	11.2	28.9	1,058,199	39,868	3.9%	11.6	30.0
Erie	1,481,689	-	-	65.7	170.2	1,578,664	96,975	6.5%	70.0	181.3	1,623,278	44,614	2.8%	72.0	186.5
Ontario	3,995,376	-	-	140.9	364.9	4,307,275	311,899	7.8%	151.9	393.4	4,550,906	243,631	5.7%	160.5	415.6
Overall	**6,582,328**	**-**	**-**	**29.2**	**75.6**	**7,079,872**	**497,544**	**7.6%**	**31.4**	**81.4**	**7,409,982**	**330,110**	**4.7%**	**32.9**	**85.2**

Basin	1986 Census					1991 Census					1996 Census				
	Count	5-year Δ	% Δ	Density SQKM	Density SQMI	Count	5-year Δ	% Δ	Density SQKM	Density SQMI	Count	5-year Δ	% Δ	Density SQKM	Density SQMI
Superior	178,858	1,259	0.7%	2.1	5.6	182,857	3,999	2.2%	2.2	5.7	180,086	-2,771	-1.5%	2.2	5.6
Huron	1,079,199	21,000	2.0%	11.8	30.6	1,219,158	139,959	13.0%	13.4	34.6	1,305,544	86,386	7.1%	14.3	37.1
Erie	1,679,536	56,258	3.5%	74.5	192.9	1,824,835	145,299	8.7%	80.9	209.6	1,921,043	96,208	5.3%	85.2	220.7
Ontario	4,885,860	334,954	7.4%	172.3	446.2	5,471,125	585,265	12.0%	192.9	499.7	5,887,739	416,614	7.6%	207.6	537.7
Overall	**7,823,453**	**413,471**	**5.6%**	**34.7**	**89.9**	**8,697,975**	**874,522**	**11.2%**	**38.6**	**100.0**	**9,294,412**	**596,437**	**6.9%**	**41.2**	**106.8**

Basin	2001 Census					2006 Census					2011 Census				
	Count	5-year Δ	% Δ	Density SQKM	Density SQMI	Count	5-year Δ	% Δ	Density SQKM	Density SQMI	Count	5-year Δ	% Δ	Density SQKM	Density SQMI
Superior	173,716	-6,370	-3.5%	2.1	5.4	171,433	-2,283	-1.3%	2.1	5.3	168,437	-2,996	-1.7%	2.0	5.2
Huron	1,358,006	52,462	4.0%	14.9	38.6	1,444,504	86,498	6.4%	15.8	41.0	1,486,864	42,360	2.9%	16.3	42.2
Erie	2,018,716	97,673	5.1%	89.5	231.9	2,129,870	111,154	5.5%	94.5	244.7	2,193,219	63,349	3.0%	97.3	251.9
Ontario	6,354,843	467,104	7.9%	224.1	580.4	6,863,620	508,777	8.0%	242.0	626.8	7,379,162	515,542	7.5%	260.2	673.9
Overall	**9,905,281**	**610,869**	**6.6%**	**43.9**	**113.8**	**10,609,427**	**704,146**	**7.1%**	**47.1**	**121.9**	**11,227,682**	**618,255**	**5.8%**	**49.8**	**129.0**

Table 3. Population within the Great Lakes Basin in Canada, 1971-2021, every 5 years (continued)

Basin	2016 Census					2021 Census				
	Count	5-year Δ	% Δ	Density		Count	5-year Δ	% Δ	Density	
				SQKM	SQMI				SQKM	SQMI
Superior	168,732	295	0.2%	2.0	5.3	169,897	1,165	0.7%	2.0	5.3
Huron	1,542,129	55,265	3.7%	16.9	43.8	1,636,294	94,165	6.1%	17.9	46.5
Erie	2,266,267	73,048	3.3%	100.5	260.3	2,452,606	186,339	8.2%	108.8	281.7
Ontario	7,780,152	400,990	5.4%	274.3	710.5	8,169,080	388,928	5.0%	288.0	746.0
Overall	**11,757,280**	**529,598**	**4.7%**	**52.2**	**135.1**	**12,427,877**	**670,597**	**5.7%**	**55.1**	**142.8**

Note: The Lake Michigan Basin is entirely within the United States, and thus omitted from this table.
Sources:
2021 data: Statistics Canada, Environment and Energy Statistics Division, special tabulation from the 2021 Census of Population. Received February 23, 2022. (*Preliminary and subject to revision.*)
1971-2016 data: Statistics Canada. (2017). *Table 17-10-0117-01 Selected population characteristics, Canada, major drainage areas and sub-drainage areas.* https://doi.org/10.25318/1710011701-eng (accessed June 22, 2021).
Data note: The following sub-drainage areas reported in Table 17-10-0117-01 comprise the Great Lakes sub-basins in Canada:
Lake Superior:

- Northwestern Lake Superior
- Northeastern Lake Superior

Lake Huron:

- Northern Lake Huron
- Wanapitei and French, Ontario
- Eastern Georgian Bay
- Eastern Lake Huron

Lake Erie:

- Northern Lake Erie

Lake Ontario:

- Lake Ontario and Niagara Peninsula

Table 4. Great Lakes Basin Land Drainage Area

Note: % is the proportion of the basin's land drainage area within each nation.

These values were derived from our geospatial analysis by subtracting the surface area of the Great Lakes and connecting channels from the whole watershed area (land and water). Inland waterbodies (lakes and streams) were not removed from these land drainage area values. These land drainage values are consistent with values previously reported by USEPA (https://www.epa.gov/greatlakes/physical-features-great-lakes).

These values were used to calculate population density reported in Table 2 & Table 3.

Basin	Portion	Land Drainage Area		
		SQKM	SQMI	%
Superior	Binational	126,136	48,701	100%
	Ontario	83,240	32,139	66.0%
	USA	42,896	16,562	34.0%
Michigan	Binational	116,392	44,939	100%
	Ontario	0	0	0%
	USA	116,392	44,939	100%
Huron	Binational	133,360	51,491	100%
	Ontario	91,236	35,226	68.4%
	USA	42,125	16,264	31.6%
Erie	Binational	76,441	29,514	100%
	Ontario	22,547	8,705	29.5%
	USA	53,894	20,809	70.5%
Ontario	Binational	62,879	24,278	100%
	Ontario	28,360	10,950	45.1%
	USA	34,519	13,328	54.9%
Overall		**515,209**	**198,923**	**100%**
Ontario portion		225,383	87,021	43.7%
USA portion		289,827	111,903	56.3%

Table 5. Great Lakes Region Population, 1900-2020, every 10 years. *Continued next page.*

Census Year	1900				1910				1920			
State / Province	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region
Illinois	4,821,550	995,199	26.0%	14.4%	5,638,591	817,041	16.9%	14.2%	6,485,280	846,689	15.0%	14.1%
Indiana	2,516,462	324,058	14.8%	7.5%	2,700,876	184,414	7.3%	6.8%	2,930,390	229,514	8.5%	6.4%
Michigan	2,420,982	327,093	15.6%	7.2%	2,810,173	389,191	16.1%	7.1%	3,668,412	858,239	30.5%	8.0%
Minnesota	1,751,394	449,568	34.5%	5.2%	2,075,708	324,314	18.5%	5.2%	2,387,125	311,417	15.0%	5.2%
New York	7,268,894	1,271,041	21.2%	21.7%	9,113,614	1,844,720	25.4%	23.0%	10,385,227	1,271,613	14.0%	22.6%
Ohio	4,157,545	485,229	13.2%	12.4%	4,767,121	609,576	14.7%	12.0%	5,759,394	992,273	20.8%	12.5%
Ontario	2,182,947	68,626	3.2%	6.5%	2,527,292	344,345	15.8%	6.4%	2,933,662	406,370	16.1%	6.4%
Pennsylvania	6,302,115	1,044,101	19.9%	18.8%	7,665,111	1,362,996	21.6%	19.3%	8,720,017	1,054,906	13.8%	19.0%
Wisconsin	2,069,042	382,162	22.7%	6.2%	2,333,860	264,818	12.8%	5.9%	2,632,067	298,207	12.8%	5.7%
Totals	**33,490,931**	**5,347,077**	**19.0%**	**100%**	**39,632,346**	**6,141,415**	**18.3%**	**100%**	**45,901,574**	**6,269,228**	**15.8%**	**100%**

Census Year	1930				1940				1950			
State / Province	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region
Illinois	7,630,654	1,145,374	17.7%	14.3%	7,897,241	266,587	3.5%	14.0%	8,712,176	814,935	10.3%	13.8%
Indiana	3,238,503	308,113	10.5%	6.1%	3,427,796	189,293	5.8%	6.1%	3,934,224	506,428	14.8%	6.2%
Michigan	4,842,325	1,173,913	32.0%	9.0%	5,256,106	413,781	8.5%	9.3%	6,371,766	1,115,660	21.2%	10.1%
Minnesota	2,563,953	176,828	7.4%	4.8%	2,792,300	228,347	8.9%	4.9%	2,982,483	190,183	6.8%	4.7%
New York	12,588,066	2,202,839	21.2%	23.5%	13,479,142	891,076	7.1%	23.8%	14,830,192	1,351,050	10.0%	23.4%
Ohio	6,646,697	887,303	15.4%	12.4%	6,907,612	260,915	3.9%	12.2%	7,946,627	1,039,015	15.0%	12.6%
Ontario	3,431,683	498,021	17.0%	6.4%	3,787,655	355,972	10.4%	6.7%	4,597,542	809,887	21.4%	7.3%
Pennsylvania	9,631,350	911,333	10.5%	18.0%	9,900,180	268,830	2.8%	17.5%	10,498,012	597,832	6.0%	16.6%
Wisconsin	2,939,006	306,939	11.7%	5.5%	3,137,587	198,581	6.8%	5.5%	3,434,575	296,988	9.5%	5.4%
Totals	**53,512,237**	**7,610,663**	**16.6%**	**100%**	**56,585,619**	**3,073,382**	**5.7%**	**100%**	**63,307,597**	**6,721,978**	**11.9%**	**100%**

Table 5. Great Lakes Region Population, 1900-2020, every 10 years (continued). *Continued next page.*

Census Year	1960				1970				1980			
State / Province	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region
Illinois	10,081,158	1,368,982	15.7%	13.6%	11,113,976	1,032,818	10.2%	13.6%	11,426,518	312,542	2.8%	13.6%
Indiana	4,662,498	728,274	18.5%	6.3%	5,193,669	531,171	11.4%	6.3%	5,490,224	296,555	5.7%	6.6%
Michigan	7,823,194	1,451,428	22.8%	10.6%	8,875,083	1,051,889	13.4%	10.9%	9,262,078	386,995	4.4%	11.1%
Minnesota	3,413,864	431,381	14.5%	4.6%	3,804,971	391,107	11.5%	4.7%	4,075,970	270,999	7.1%	4.9%
New York	16,782,304	1,952,112	13.2%	22.7%	18,236,967	1,454,663	8.7%	22.3%	17,558,072	-678,895	-3.7%	21.0%
Ohio	9,706,397	1,759,770	22.1%	13.1%	10,652,017	945,620	9.7%	13.0%	10,797,630	145,613	1.4%	12.9%
Ontario	6,236,092	1,638,550	35.6%	8.4%	7,703,106	1,467,014	23.5%	9.4%	8,625,107	922,001	12.0%	10.3%
Pennsylvania	11,319,366	821,354	7.8%	15.3%	11,793,909	474,543	4.2%	14.4%	11,863,895	69,986	0.6%	14.2%
Wisconsin	3,951,777	517,202	15.1%	5.3%	4,417,731	465,954	11.8%	5.4%	4,705,767	288,036	6.5%	5.6%
Totals	**73,976,650**	**10,669,053**	**16.9%**	**100%**	**81,791,429**	**7,814,779**	**10.6%**	**100%**	**83,805,261**	**2,013,832**	**2.5%**	**100%**

Census Year	1990				2000				2010			
State / Province	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region	Count	Decadal Δ	% Δ	% Region
Illinois	11,430,602	4,084	0.04%	13.2%	12,419,293	988,691	8.6%	13.4%	12,830,632	411,339	3.3%	13.3%
Indiana	5,544,159	53,935	1.0%	6.4%	6,080,485	536,326	9.7%	6.6%	6,483,802	403,317	6.6%	6.7%
Michigan	9,295,297	33,219	0.4%	10.8%	9,938,444	643,147	6.9%	10.7%	9,883,640	-54,804	-0.6%	10.2%
Minnesota	4,375,099	299,129	7.3%	5.1%	4,919,479	544,380	12.4%	5.3%	5,303,925	384,446	7.8%	5.5%
New York	17,990,455	432,383	2.5%	20.8%	18,976,457	986,002	5.5%	20.5%	19,378,102	401,645	2.1%	20.0%
Ohio	10,847,115	49,485	0.5%	12.6%	11,353,140	506,025	4.7%	12.2%	11,536,504	183,364	1.6%	11.9%
Ontario	10,084,885	1,459,778	16.9%	11.7%	11,410,046	1,325,161	13.1%	12.3%	12,851,821	1,441,775	12.6%	13.3%
Pennsylvania	11,881,643	17,748	0.1%	13.8%	12,281,054	399,411	3.4%	13.2%	12,702,379	421,325	3.4%	13.1%
Wisconsin	4,891,769	186,002	4.0%	5.7%	5,363,675	471,906	9.6%	5.8%	5,686,986	323,311	6.0%	5.9%
Totals	**86,341,024**	**2,535,763**	**3.0%**	**100%**	**92,742,073**	**6,401,049**	**7.4%**	**100%**	**96,657,791**	**3,915,718**	**4.2%**	**100%**

Table 5. Great Lakes Region Population, 1900-2020, every 10 years (continued)

Census Year	2020			
State / Province	**Count**	**Decadal Δ**	**% Δ**	**% Region**
Illinois	12,812,508	-18,124	-0.1%	12.7%
Indiana	6,785,528	301,726	4.7%	6.8%
Michigan	10,077,331	193,691	2.0%	10.0%
Minnesota	5,706,494	402,569	7.6%	5.7%
New York	20,201,249	823,147	4.2%	20.1%
Ohio	11,799,448	262,944	2.3%	11.7%
Ontario	14,223,942	1,372,121	10.7%	14.2%
Pennsylvania	13,002,700	300,321	2.4%	12.9%
Wisconsin	5,893,718	206,732	3.6%	5.9%
Totals	**100,502,918**	**3,845,127**	**4.0%**	**100%**

Note: Population data for Ontario are from the 1901, 1911, 1921, 1931, 1941, 1951, 1961, 1971, 1981, 1991, 2001, 2011, and 2021 censuses. 'Decadal Δ' is the decade-over-decade change in population count. '% Region' is the proportion of each state/province's population compared to the Great Lakes Region's total population.

Sources:

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Steven Manson, Jonathan Schroeder, David Van Riper, Tracy Kugler, and Steven Ruggles. IPUMS National Historical Geographic Information System: Version 16.0 [dataset]. Minneapolis, MN: IPUMS. 2021. http://doi.org/10.18128/D050.V16.0 (accessed February 28, 2022).

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Table 6. Population of U.S. Counties within the Great Lakes Basin, 1900-2020, every 10 years. *Continued next page.*

Census Year	1900				1910				1920			
State	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population
Illinois	1,873,239	657,082	54.0%	38.9%	2,460,291	587,052	31.3%	43.6%	3,127,302	667,011	27.1%	48.2%
Indiana	431,193	58,364	15.7%	17.1%	526,586	95,393	22.1%	19.5%	648,218	121,632	23.1%	22.1%
Michigan	2,420,982	328,088	15.7%	100%	2,810,173	389,191	16.1%	100%	3,668,412	858,239	30.5%	100%
Minnesota	121,275	62,487	106.3%	6.9%	233,637	112,362	92.7%	11.3%	295,910	62,273	26.7%	12.4%
New York	2,082,272	211,242	11.3%	28.6%	2,343,323	261,051	12.5%	25.7%	2,629,238	285,915	12.2%	25.3%
Ohio	1,764,942	287,093	19.4%	42.5%	2,108,311	343,369	19.5%	44.2%	2,814,328	706,017	33.5%	48.9%
Pennsylvania	192,737	18,561	10.7%	3.1%	206,811	14,074	7.3%	2.7%	235,292	28,481	13.8%	2.7%
Wisconsin	1,236,613	262,775	27.0%	59.8%	1,451,313	214,700	17.4%	62.2%	1,671,735	220,422	15.2%	63.5%
Totals	**10,123,253**	**1,885,692**	**22.9%**	**32.3%**	**12,140,445**	**2,017,192**	**19.9%**	**32.7%**	**15,090,435**	**2,949,990**	**24.3%**	**35.1%**

Census Year	1930				1940				1950			
State	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population
Illinois	4,086,510	959,208	30.7%	53.6%	4,184,436	97,926	2.4%	53.0%	4,687,889	503,453	12.0%	53.8%
Indiana	860,609	212,391	32.8%	26.6%	919,770	59,161	6.9%	26.8%	1,116,819	197,049	21.4%	28.4%
Michigan	4,842,325	1,173,913	32.0%	100%	5,256,106	413,781	8.5%	100%	6,371,766	1,115,660	21.2%	100%
Minnesota	297,828	1,918	0.6%	11.6%	313,454	15,626	5.2%	11.2%	307,198	-6,256	-2.0%	10.3%
New York	2,966,637	337,399	12.8%	23.6%	3,045,304	78,667	2.7%	22.6%	3,399,147	353,843	11.6%	22.9%
Ohio	3,350,851	536,523	19.1%	50.4%	3,430,475	79,624	2.4%	49.7%	3,999,956	569,481	16.6%	50.3%
Pennsylvania	255,746	20,454	8.7%	2.7%	270,734	14,988	5.9%	2.7%	315,146	44,412	16.4%	3.0%
Wisconsin	1,937,506	265,771	15.9%	65.9%	2,061,375	123,869	6.4%	65.7%	2,281,231	219,856	10.7%	66.4%
Totals	**18,598,012**	**3,507,577**	**23.2%**	**37.1%**	**19,481,654**	**883,642**	**4.8%**	**36.9%**	**22,479,152**	**2,997,498**	**15.4%**	**38.3%**

Table 6. Population of U.S. Counties within the Great Lakes Basin, 1900-2020, every 10 years (continued). *Continued next page.*

Census Year	1960				1970				1980			
State	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population
Illinois	5,423,381	735,492	15.7%	53.8%	5,875,007	451,626	8.3%	52.9%	5,694,027	-180,980	-3.1%	49.8%
Indiana	1,423,492	306,673	27.5%	30.5%	1,592,825	169,333	11.9%	30.7%	1,658,563	65,738	4.1%	30.2%
Michigan	7,823,194	1,451,428	22.8%	100%	8,875,083	1,051,889	13.4%	100%	9,262,078	386,995	4.4%	100%
Minnesota	343,771	36,573	11.9%	10.1%	329,293	-14,478	-4.2%	8.7%	345,644	16,351	5.0%	8.5%
New York	3,956,907	557,760	16.4%	23.6%	4,296,068	339,161	8.6%	23.6%	4,214,448	-81,620	-1.9%	24.0%
Ohio	4,934,886	934,930	23.4%	50.8%	5,397,075	462,189	9.4%	50.7%	5,316,425	-80,650	-1.5%	49.2%
Pennsylvania	345,121	29,975	9.5%	3.0%	361,391	16,270	4.7%	3.1%	386,375	24,984	6.9%	3.3%
Wisconsin	2,698,705	417,474	18.3%	68.3%	3,013,700	314,995	11.7%	68.2%	3,139,541	125,841	4.2%	66.7%
Totals	**26,949,457**	**4,470,305**	**19.9%**	**39.8%**	**29,740,442**	**2,790,985**	**10.4%**	**40.1%**	**30,017,101**	**276,659**	**0.9%**	**39.9%**

Census Year	1990				2000				2010			
State	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population	Count	Decadal Δ	% Δ	% State Population
Illinois	5,621,485	-72,542	-1.3%	49.2%	6,021,097	399,612	7.1%	48.5%	5,898,137	-122,960	-2.0%	46.0%
Indiana	1,668,139	9,576	0.6%	30.1%	1,811,632	143,493	8.6%	29.8%	1,892,087	80,455	4.4%	29.2%
Michigan	9,295,297	33,219	0.4%	100%	9,938,444	643,147	6.9%	100%	9,883,640	-54,804	-0.6%	100%
Minnesota	316,307	-29,337	-8.5%	7.2%	334,248	17,941	5.7%	6.8%	342,664	8,416	2.5%	6.5%
New York	4,237,527	23,079	0.5%	23.6%	4,225,309	-12,218	-0.3%	22.3%	4,215,173	-10,136	-0.2%	21.8%
Ohio	5,197,486	-118,939	-2.2%	47.9%	5,324,946	127,460	2.5%	46.9%	5,221,280	-103,666	-1.9%	45.3%
Pennsylvania	378,458	-7,917	-2.0%	3.2%	389,289	10,831	2.9%	3.2%	386,788	-2,501	-0.6%	3.0%
Wisconsin	3,246,927	107,386	3.4%	66.4%	3,525,047	278,120	8.6%	65.7%	3,685,228	160,181	4.5%	64.8%
Totals	**29,961,626**	**-55,475**	**-0.2%**	**39.3%**	**31,570,012**	**1,608,386**	**5.4%**	**38.8%**	**31,524,997**	**-45,015**	**-0.1%**	**37.6%**

Table 6. Population of U.S. Counties within the Great Lakes Basin, 1900-2020, every 10 years (continued)

Census Year	2020			
State	**Count**	**Decadal Δ**	**% Δ**	**% State Population**
Illinois	5,989,883	91,746	1.6%	46.8%
Indiana	1,959,533	67,446	3.6%	28.9%
Michigan	10,077,331	193,691	2.0%	100%
Minnesota	342,530	-134	-0.04%	6.0%
New York	4,216,538	1,365	0.03%	20.9%
Ohio	5,196,537	-24,743	-0.5%	44.0%
Pennsylvania	371,210	-15,578	-4.0%	2.9%
Wisconsin	3,770,046	84,818	2.3%	64.0%
Totals	**31,923,608**	**398,611**	**1.3%**	**37.0%**

Notes: This table shows the population of the 209 U.S. counties with any land area within the Great Lakes Basin, including straddling counties with population outside the Great Lakes Basin.

'Decadal Δ' is the decade-over-decade change in population count. '% State Population' is the proportion of the State's resident population within Great Lakes Basin counties compared to the State's total population (displayed in Table 5).

See Figure 1 for a map of the U.S. counties with any land area within the Great Lakes Basin.

For the table of population for all 209 U.S. counties within the Great Lakes Basin from the 1790 census to the 2020 census, contact Ryan Graydon (graydon.ryan@epa.gov).

Sources:

Steven Manson, Jonathan Schroeder, David Van Riper, Tracy Kugler, and Steven Ruggles. IPUMS National Historical Geographic Information System: Version 16.0 [dataset]. Minneapolis, MN: IPUMS. 2021. http://doi.org/10.18128/D050.V16.0 (accessed February 28, 2022).

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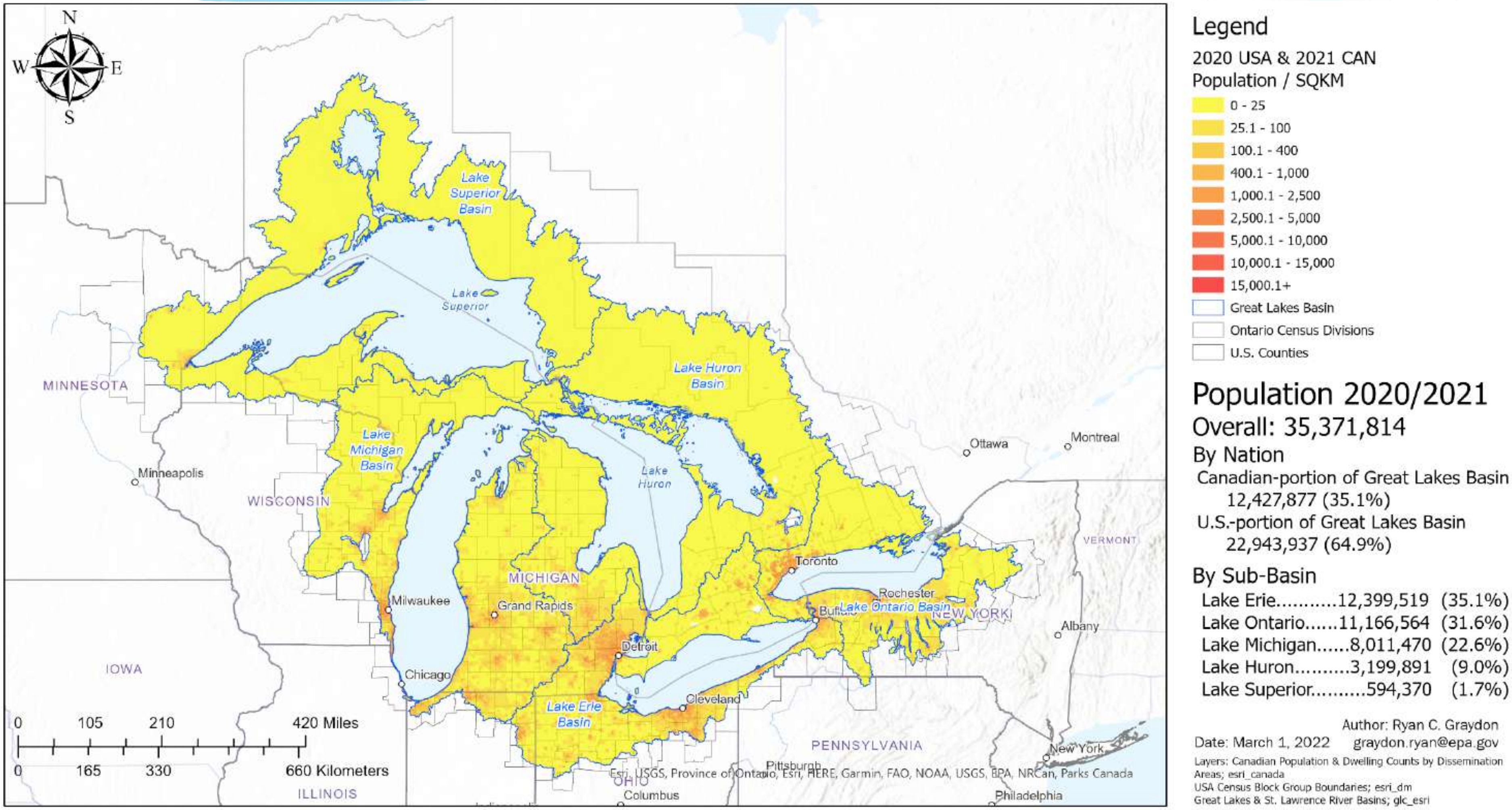

Figure 1. Geographic Distribution of Population Across the Great Lakes Basin (2020/2021)

Note: The symbology displays the population density (per square kilometer) by census block group in the U.S. and by dissemination area in Canada.

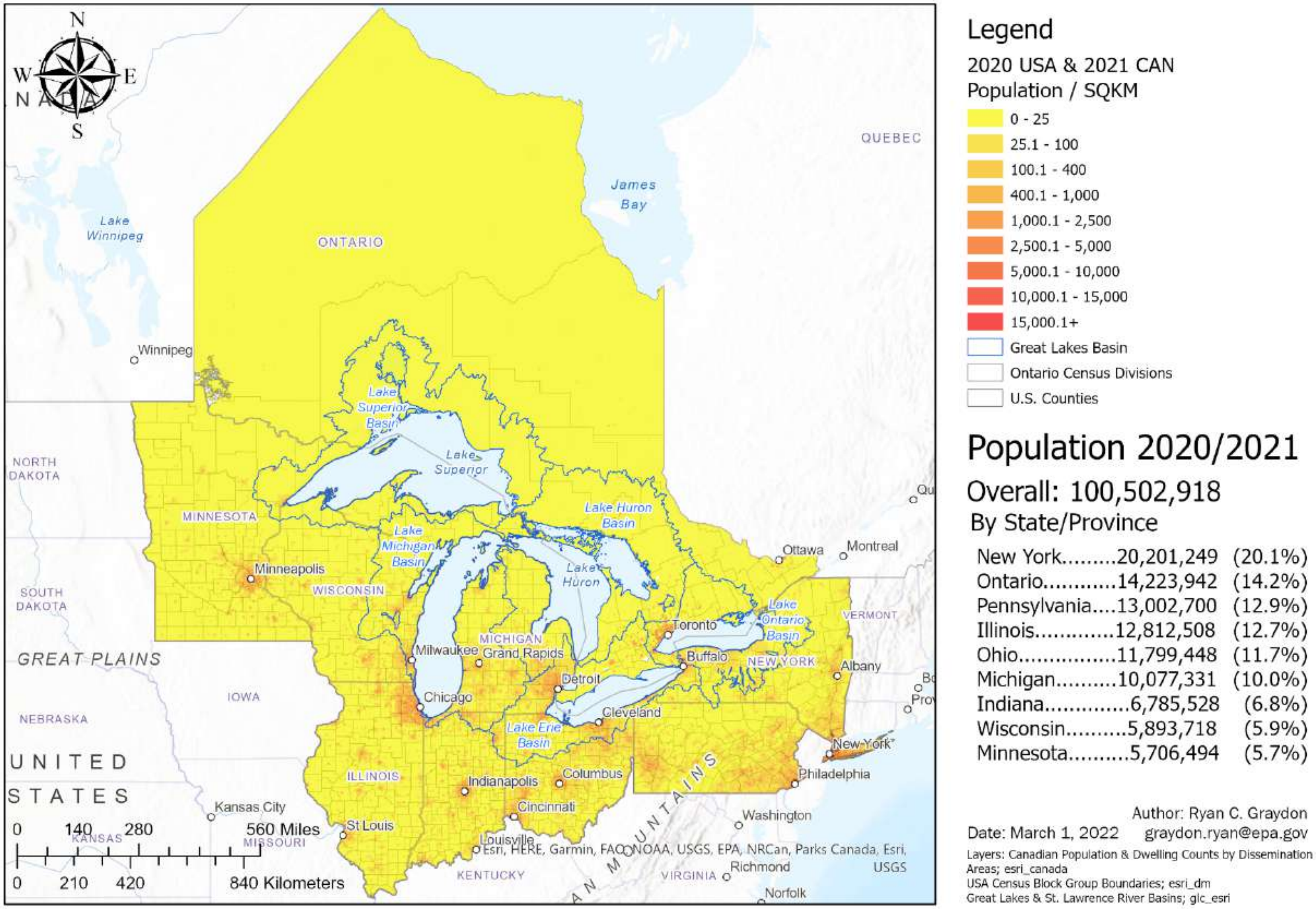

Figure 2. Geographic Distribution of Population Across the Great Lakes Region (2020/2021)

Note: The symbology displays the population density (per square kilometer) by census block group in the U.S. and by dissemination area in Canada.

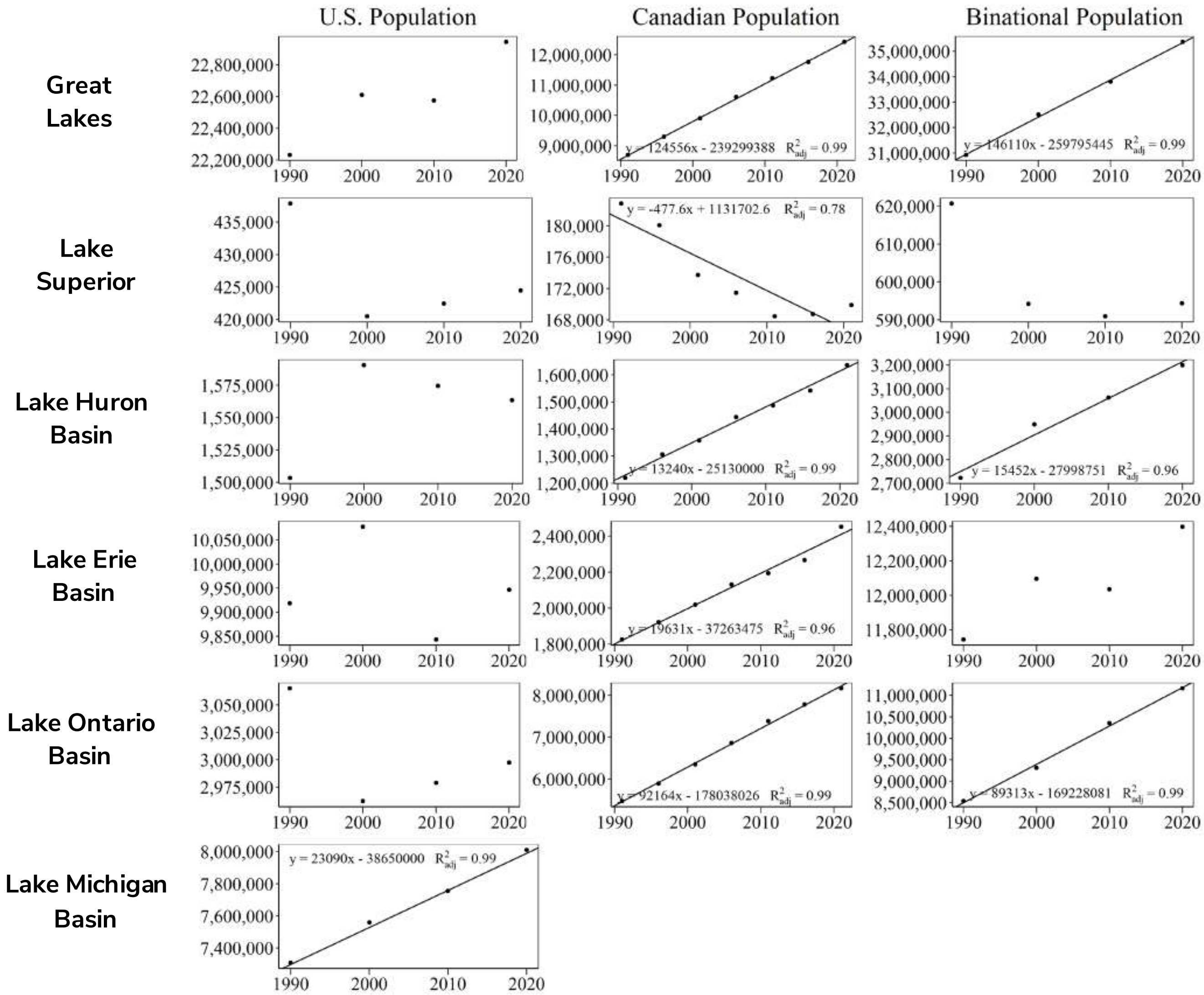

Figure 3. 30-year population trends in the Great Lakes Basin (1990/1991-2020/2021)
Notes: The Lake Michigan Basin is entirely within the United States. The data points in the figure are reported in Table 2 and Table 3.
Sources:
Canadian data:
2021 data: Statistics Canada, Environment and Energy Statistics Division, special tabulation from the 2021 Census of Population. Received February 23, 2022. (*Preliminary and subject to revision.*)
1971-2016 data: Statistics Canada, *Table 17-10-0117-01 Selected population characteristics, Canada, major drainage areas and sub-drainage areas.* https://doi.org/10.25318/1710011701-eng (accessed June 22, 2021).
U.S. data: U.S. population within the Great Lakes Basin were derived from authors' geospatial analyses of census block groups from the following datasets:
2010 to 2020 data: esri_dm. (2021, October 1). *USA Census Block Groups Boundaries*. ArcGIS Online. https://esri.maps.arcgis.com/home/item.html?id=1c924a53319a491ab43d5cb1d55d8561 (accessed February 17, 2022).

1970 to 2000 data: Steven Manson, Jonathan Schroeder, David Van Riper, Tracy Kugler, and Steven Ruggles. *IPUMS National Historical Geographic Information System: Version 15.0 [Database]*. Minneapolis, MN: IPUMS. 2020. http://doi.org/10.18128/D050.V15.0 (accessed June 17, 2021).
glc_esri. (2020, August 20). *Great Lakes & St. Lawrence River Basins*. ArcGIS Online. https://epa.maps.arcgis.com/home/item.html?id=df1de3fdd82047a79864b45c9ec1180b (accessed June 17, 2021).

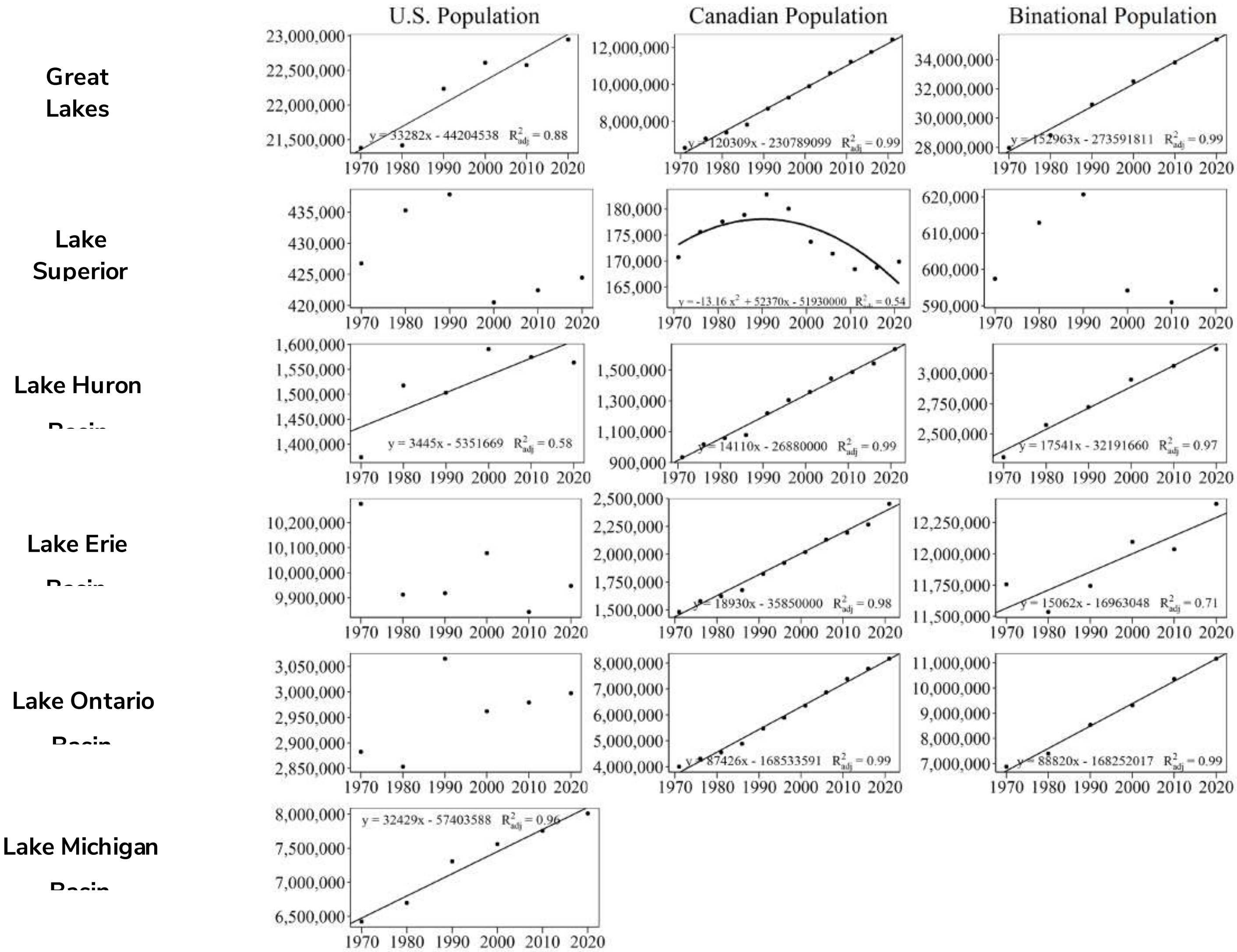

Figure 4. Long-term (50-year) population trends in the Great Lakes Basin (1970/1971-2020/2021)
Notes: The Lake Michigan Basin is entirely within the United States. The data points in the figure are reported in Table 2 and Table 3.
Sources:
Canadian data:
2021 data: Statistics Canada, Environment and Energy Statistics Division, special tabulation from the 2021 Census of Population. Received February 23, 2022. (*Preliminary and subject to revision.*)
1971-2016 data: Statistics Canada, *Table 17-10-0117-01 Selected population characteristics, Canada, major drainage areas and sub-drainage areas*. https://doi.org/10.25318/1710011701-eng (accessed June 22, 2021).
U.S. data: U.S. population within the Great Lakes Basin were derived from authors' geospatial analyses of census block groups from the following datasets:
2010 to 2020 data: esri_dm. (2021, October 1). *USA Census Block Groups Boundaries*. ArcGIS Online. https://esri.maps.arcgis.com/home/item.html?id=1c924a53319a491ab43d5cb1d55d8561 (accessed February 17, 2022).

1970 to 2000 data: Steven Manson, Jonathan Schroeder, David Van Riper, Tracy Kugler, and Steven Ruggles. *IPUMS National Historical Geographic Information System: Version 15.0 [Database]*. Minneapolis, MN: IPUMS. 2020. http://doi.org/10.18128/D050.V15.0 (accessed June 17, 2021).
glc_esri. (2020, August 20). *Great Lakes & St. Lawrence River Basins*. ArcGIS Online. https://epa.maps.arcgis.com/home/item.html?id=df1de3fdd82047a79864b45c9ec1180b (accessed June 17, 2021).

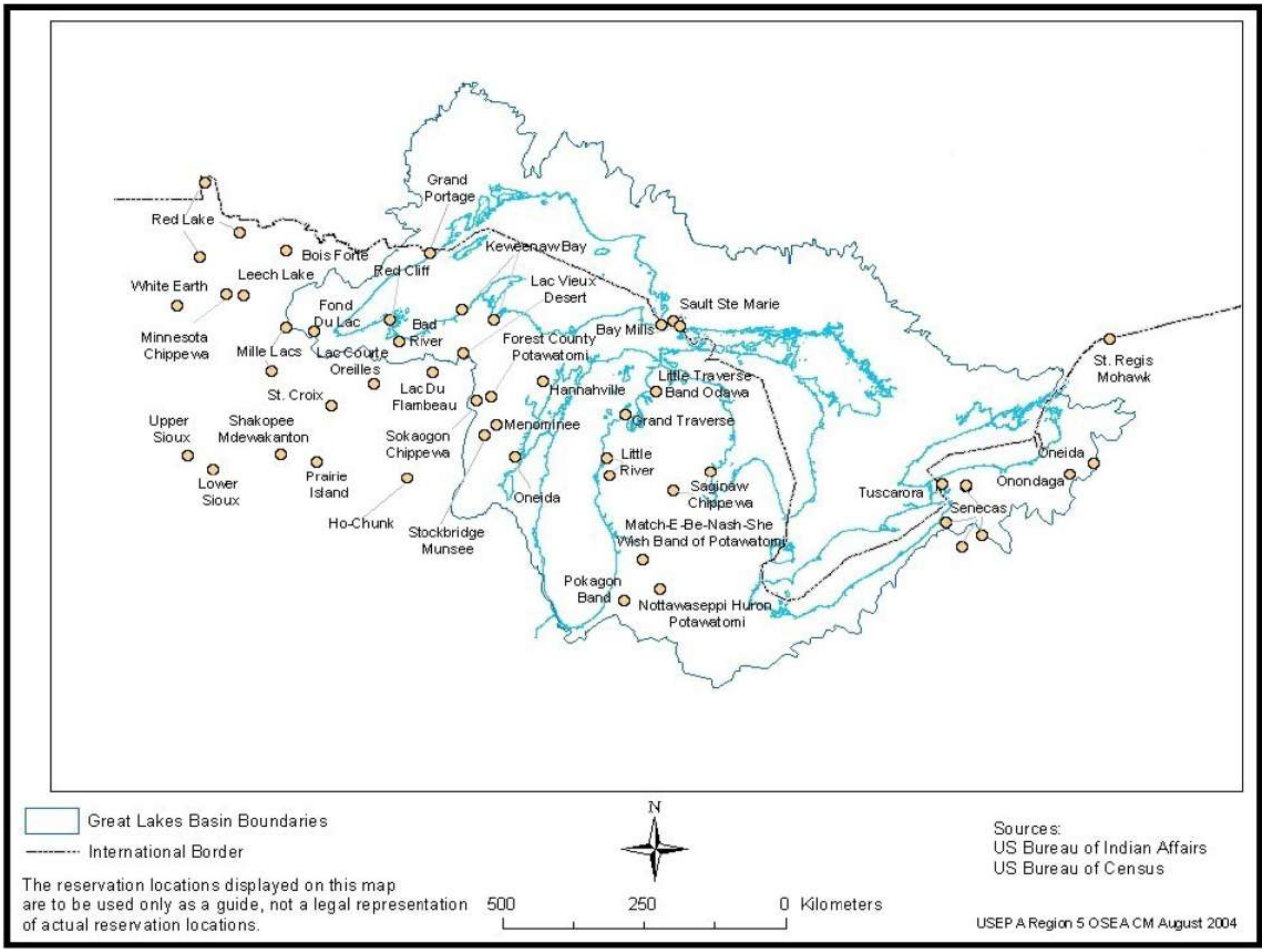

Figure 5. U.S. Tribal Nations in the Great Lakes Region. *Source: Great Lakes Regional Collaboration. (2005).* Tribal Nations Issues and Perspectives, Version 1.0. https://gsgp.org/media/lgydin3m/glrc-tribal-briefing-paper.pdf

Sub-Indicator: Precipitation Amounts in the Great Lakes Basin

Overall Assessment

Trends:

10-Year Trend (2011-2020): Unchanging

30-Year Trend (1991-2020): Increasing

Long-term Trend (1950-2020): Increasing

Rationale: Annual precipitation anomalies based on US and Canadian based stations aggregated across the entire Great Lakes basin relative to 1961-1990 means display a statistically significant trend for both the 30 year period (+3.9% per decade) and for the long-term period (+2.2% per decade). Total precipitation during 2011-2020 exceeded that of any other 10 year period since 1950. Although the overall assessment is based on annual values, there was evidence of long term increasing trends in summer (+1.5% per decade) and fall (+4.1% per decade).

Lake-by-Lake Assessment

Note: The lake-by-lake trend assessments are based solely on the annual precipitation anomalies, and not on seasonal precipitation anomalies. Some additional information on seasonal trends are included in the rationale sections, however this is only to provide additional context.

Lake Superior

10-Year Trend (2011-2020): Unchanging

30-Year Trend (1991-2020): Unchanging

Long-term Trend (1950-2020): Unchanging

Rationale: The annual precipitation anomaly trend for Lake Superior was positive in all three periods but statistically significant for none. Total precipitation during 2011-2020 exceeded that of any other 10-year period since 1950 for Lake Superior. Although the annual trend was not significant, the seasonal trend was statistically significant for the fall 10-year period (+37.9% per decade), and the fall long-term period (+4.1% per decade).

Lake Michigan-Huron

10-Year Trend (2011-2020): Unchanging

30-Year Trend (1991-2020): Increasing

Long-term Trend (1950-2020): Increasing

Rationale: The annual precipitation anomaly trend for Lake Michigan-Huron was statistically significant for the 30 year period (+3.4% per decade) and the long-term period (+2.5% per decade). Total precipitation during 2011-2020 exceeded that of any other 10-year period since 1950 for Lake Michigan-Huron. The long term trend was also increasing for winter (+2.6% per decade), spring (+2.8% per decade), and fall (+4.0% per decade).

Lake Erie (including St. Clair-Detroit River Ecosystem)

10-Year Trend (2011-2020): Unchanging

30-Year Trend (1991-2020): Unchanging

Long-term Trend (1950-2020): Increasing

Rationale: The annual precipitation anomaly trend for Lake Erie was statistically significant for the long-term period (+2.6% per decade). Although the annual trend was not significant for the 30-year period, the seasonal trend was statistically significant for the winter 30-year period (+7.3% per decade). The summer long-term trend was also increasing (+2.7% per decade).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

10-Year Trend (2011-2020): Unchanging

30-Year Trend (1991-2020): Unchanging

Long-term Trend (1950-2020): Increasing

Rationale: The annual precipitation anomaly trend for Lake Ontario was statistically significant for the long-term period (+2.7% per decade). Although the annual trend was not significant for the 30-year period, the seasonal trend was significant for the winter 30-year period (+6.2% per decade). Significant long-term seasonal trends were also observed in winter (+2.3% per decade), summer (+3.1% per decade), and fall (+4.5% per decade).

Trend Assessment Definitions

Increasing: The calculated historical trend in annual precipitation amount is positive and large enough to be distinguished from the variability of the data (is statistically significant at the 0.05 level)

Unchanging: The calculated historical trend in annual precipitation amount is too small to be distinguished from the natural variability of the data (is not statistically significant at the 0.05 level)

Decreasing: The calculated historical trend in annual precipitation amount is negative and large enough to be distinguished from the variability of the data (is statistically significant at the 0.05 level)

Undetermined: Data are not available to report on a trend or the time period is too short for viable precipitation trend analysis

Note that significant trends in seasonal precipitation are also described in the rationale, but trend assessment definitions are based on annual precipitation.

Sub-Indicator Purpose

The purpose of this precipitation amounts sub-indicator is to assess whether the amount of precipitation falling within the Great Lakes basin is above or below a reference period, both annually and seasonally, both Great Lakes wide and for individual lakes and to infer the potential impact that varying precipitation amounts due to climate change will have on ecosystem components.

Ecosystem Objective

To maintain the ecosystem of the Great Lakes and surrounding region by allowing the hydrologic system of the Great Lakes basin to continue to follow historic patterns. Changes in the frequency, seasonal distribution, or magnitude of precipitation will affect the hydrologic system of the entire Great Lakes basin, resulting in impacts such as altered water levels or changes in the rates and patterns of storm runoff, thereby influencing the distribution of pollutants, nutrients, and invasive species.

This sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should "be free from other substances, materials, or conditions that may negatively impact the chemical, physical, or biological integrity of the Waters of the Great Lakes." This sub-indicator also relates to all of the General Objectives in the Agreement because it has broad ecosystem implications.

Measure

This sub-indicator estimates the annual and seasonal precipitation anomalies (from the 1961-1990 reference period) for each period of study (1950–2020, 1991-2020 and 2011-2020) in the Great Lakes basin as well as each individual lakes' basin ("basin" hereafter refers to the combined area including both the land-surface of the watershed and the lake surface). Data from 1950-1990 are from the Coordinating Committee for Great Lakes Basic Hydraulic and Hydrologic Data. For details on how those data were derived, see Hunter et al. (2015). Data from 1991-2020 was aggregated from approximately 1350 stations throughout the Great Lakes. 1200 of these stations were from the Global Historical Climatology Network daily (Menne et al. 2012) and 150 were from the Global Hourly Integrated Surface Database (Smith et al. 2011).

Data for these analyses come from the Great Lakes Seasonal Hydrologic Forecast System (GLSHFS, described in Gronewold et al. 2017), which was implemented to calculate basin-wide precipitation values using data from various surface meteorological stations on both sides of the U.S.-Canada border as its input. GLSHFS uses a Thiessen Polygon weighting technique to derive historical precipitation over sub-basins which can then be aggregated into lake basin averages for the entire lake (Croley and Hartmann 1985). The Thiessen weighting algorithm is the primary interpolation approach applied in the development of the precipitation dataset for the Great Lakes Coordinating Committee for Great Lakes Basin Hydraulic and Hydrologic Data, 1948 to present period (for details on the Coordinating Committee, see Gronewold et al., 2018). Analysis was performed using the Coordinating Committee's precipitation dataset from 1950-1989 and precipitation output from the GLSHFS for 1990-2020. GLSHFS was used for the later period due to recent improvements in data quality control which were not implemented in previous versions of GLSHFS. These new improvements help to ensure that erroneous station data are not used to compute basin-wide precipitation values.

This sub-indicator, as well as the other climate trend sub-indicators (Water Levels, Ice Cover, and Surface Water Temperatures), have been determined using an anomaly-based approach for estimating trends. An anomaly-based approach is a more representative way of describing the state of an ecosystem variable where the typical value is not necessarily common knowledge. It also allows for easy comparison of the state of variables that are not necessarily similar, such as precipitation in different seasons or different geographic locations. In addition, the anomaly-based approach helps put the longer periods of record into context by identifying whether the deviations or extremes are increasing or decreasing over the various time periods. This is valuable in reporting on recent climate changes and impacts in the Great Lakes.

For this sub-indicator, an annual or seasonal precipitation anomaly is the difference between the precipitation amount and the reference value, which is then divided by the reference value and multiplied by 100. The anomaly is therefore represented by a percentage of deviation from the reference value and the trend is expressed as a percent rate of change. Since the precipitation mean of the base period is a constant, the trends within the anomalies are identical to the trends determined from the original data.

The reference values used in this sub-indicator are the annual and seasonal precipitation totals for the base period of 1961–1990. Seasons are defined as winter (December, January, February); spring (March, April, May); summer (June, July, August); and fall (September, October, November). The 1961–1990 base period was selected following the World Meteorological Organization recognition as a standard reference period for long-term climate change assessments (WMO 2007). Moreover, this period includes a relatively high fraction of manual and rigorously quality controlled observations before the automation of observations (Mekis et al, 2018).

The significance of trends was evaluated using a Mann-Kendall test (Kendall, 1955). For a computed trend to be deemed statistically significant, it must be large enough to stand out from the variability of the data. Statistical significance was computed at the 0.05 level. While trends that were not significant at the 0.05 level may still be true, there is also a tangible chance that they instead represent cyclic or other variations in the data.

A 9-year moving average for the climate trend sub-indicators is displayed on the figures, as an aid for visualizing trends.

Ecological Condition

Overall

Annual precipitation anomalies for the Great Lakes Region for 1950–2020 is shown in Figure 1. It displays a statistically significant (at the 0.05 level) increasing trend of 2.2% per decade (Table 1, 15.4% total over the period). The trend for the 30-year period (1991-2020) was also found to be significant and had an increasing trend of 3.9% per decade (11.7% total over the period). While the trend during the 10-year period (2011-2020) was not statistically significant, it is important to note that it was the wettest 10-year period since 1950.

Precipitation anomalies can also be analyzed on a seasonal basis. The 9-year running means of the seasonal precipitation anomalies for the Great Lakes basin over the period of record (1950-2020) are shown in Figure 2. Summer and fall were found to have statistically significant trends of increase for the long-term period but not the 30 or 10 year period.

Lake-by-Lake

The four lakes' trend analysis results can be found in Table 1. The 9-year running means of anomaly series are plotted for annual and seasonal precipitation in Figures 3 and 4, respectively. The long term annual precipitation trends for individual lakes were statistically significant for Michigan-Huron, Erie and Ontario. (Though a p-value of 0.05 was used for this study, please note that all three lakes above were also significant at the 0.01 level.) There were also significant trends found for long-term seasonal precipitation as shown in Table 1. Fall showed the most number of significant trends (Superior, Michigan-Huron and Ontario), while spring showed the least with only Michigan-Huron.

For the 30 and 10 year periods fewer significant trends were found: Michigan-Huron was the only lake with a significant annual trend in the 30 year period and there were none in the 10 year period. Lakes Erie and Ontario had positive 30-year trends for winter and Lake Superior had a positive 10-year trend for fall. Despite the fewer significant trends in the shorter time periods, the slope of trends were, in general, greater than those displayed in the long term. Most notable was Superior's 10-year trend in fall (+37.9% per decade) which was an order of magnitude higher than that of other lakes and periods (Table 1).

Linkages

Increased precipitation amounts seen in recent decades in the Great Lakes basin is the result of both increasing frequency and magnitude of precipitation. An increase in global temperature enhances the ability of the atmosphere to store and transport water vapour which intensifies the hydrologic cycle and affects storm evolution and geographical distribution of precipitation (Wuebbles et al, 2020). This change, which is projected to continue into the future (Wuebbles et al., 2020) affects the hydrological system of the entire basin. The impacts of such changes would be numerous; some examples specific to an increase in precipitation event magnitude (precipitation extremes) include crop loss due to storm-damage, erosion, and flooding.

Locally, changes in ice cover and surface water temperature will directly impact precipitation event magnitude and frequency through mechanisms such as lake-effect precipitation.

Precipitation Amounts in the Great Lakes Basin link directly to almost all other sub-indicators in the suite given their role as a driving force in hydrology, nutrient and toxin distribution, and shoreline and wetland health. An increase in intense precipitation events increase the risk of flooding and higher soil erosion rates, causing entrainment and delivery of sediments, nutrients, pesticides and other contaminants in surface waters.

Some specific examples include:

- Beach Advisories – runoff following precipitation events and related bacteria loading is a major concern to beach safety and human health.
- Harmful Algal Blooms – more rain in the future over the Great Lakes basin means more runoff and sewer discharge into the lakes, increasing the potential occurrence of harmful algal blooms and hypoxia.
- Coastal Wetland Sub-indicators (Coastal Wetland Amphibians, Coastal Wetland Birds, Coastal Wetland Fish, Coastal Wetland Invertebrates, Coastal Wetland Plants, Coastal Wetland: Extent and Composition) – change in precipitation event frequency or intensity will have a direct impact on coastal wetlands by altering the extent and quality of wetland ecosystems through processes such as altering water levels, changing nutrient availability, or affecting pollutant concentrations. Extreme precipitation events could also lead to additional upstream erosion resulting in increased sedimentation, loss of wetland vegetation, and habitat loss.
- Invasive Species – cumulative stresses related to climate changes, including precipitation amounts, may encourage the spread of invasive species in coastal wetlands.
- Tributary Flashiness – precipitation events, especially extreme ones, can increase flashiness and can increase and/or change the flow of contaminants resulting in reduced water quality.

In addition, Precipitation Amounts in the Great Lakes Basin potentially have direct or indirect impacts on all 14 Beneficial Use Impairments included in Annex 1 (Areas of Concern) of the Great Lakes Water Quality Agreement Protocol of 2012.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes*	Data can be found here: 1950-1990: www.greatlakescc.org/wp36/home/coordinating-committee-products-and-datasets/ *valid for 1950-1990		

Data Limitations

Precipitation data suffers from geographic and temporal discontinuities due to factors such as lack of observing sites, changes in observation method, equipment wear and replacement, station relocation and automation. Additionally, the solid precipitation undercatch due to mainly windy conditions can lead to further missing precipitation amount in all downstream applications. To mitigate the effect of discontinuities, station data undergo QA/QC prior to use as input for the Thiessen Polygon weighting algorithm run by GLSHFS.

Precipitation naturally varies greatly on multiple time-scales including daily, seasonally, and annually. This is a challenge in precipitation trend analysis, especially when analyzing time series of a few decades or even shorter, as only very large trends will be deemed statistically significant by tests. In addition, the analysis of trends does not fully depict the anomaly observed in recent years--despite being nearly always anomalously wet, the trends were not statistically significant.

Additional Information

In the next century, annual precipitation is expected to increase by up to 20% across the Great Lakes basin with greater annual precipitation projected for Lake Superior (Lofgren et al. 2002; McKenney et al. 2011). Lake-effect precipitation continues to be observed in future projections and is expected to increase due to decreasing ice cover on lakes (Burnett et al. 2003; Notaro et al. 2014). The form of precipitation is also expected to change, with more precipitation falling as rain and freezing rain and less as snow. Shifts in the timing of precipitation are expected, where rainfall will increase in the spring but decrease in the summer (Kling et al. 2003; Hayhoe et al. 2010).

Future reporting cycles could benefit from an analysis of the number of extreme storms—defined as any extratropical cyclone-related event occurring during ice-free and ice-breakup periods which meets two of the following three criteria:

- Total daily precipitation that exceeds the 95th percentile of the climatological reference period.
- Central atmospheric pressure less than 980mb
- Recorded maximum wind gusts in a 24hr period exceeding the 95th percentile

Acknowledgments

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Figure 2. Nine-year running means of seasonal precipitation anomalies (from the 1961–1990 seasonal means) for the Great Lakes basin during the period of record (1961–2020). Note that the mean for a particular 9-year interval is centered on the middle year, meaning the first year for which the running mean can be defined is 1954 and the last is 2016.

Figure 3. Annual precipitation anomaly (from the 1961–1990 mean) and nine-year running mean for each Great Lake during the 1950–2020 period. Note that the mean for a particular 9-year interval is centred on the middle year, meaning the first year for which the running mean can be defined is 1954 and the last is 2016.

Figure 4. Nine-year running means of seasonal precipitation anomalies (from the 1961–1990 seasonal means) for each Great Lake individually during the period of record (1950–2020).

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Table 1: The slope of the linear trend (units: % anomaly per decade) for each period and lake. The anomaly is based on the deviation from the reference value which is shown in table 2. The trends that are statistically significant at the 0.05 level are colored dark green. Though the 0.05 level is used exclusively to determine statistical significance throughout this report, two more significance levels, 0.01 (bold) and 0.1 (light green) are also shown for additional context.

	Annual	Winter	Spring	Summer	Fall
LONG TERM: (1950-2020)					
Total	**2.2**	1.9	2	1.5	**4.1**
Superior	1.2	-0.3	0	0.5	4
Michigan-Huron	**2.5**	2.6	2.8	1.4	**4**
Erie	**2.6**	3	2.2	2.7	5.2
Ontario	**2.7**	2.3	1.6	**3.1**	**4.5**
30 year: (1991-2020)					
Total	3.9	7.1	4	3.4	3.7
Superior	4.9	12.8	1.3	2.9	5.2
Michigan-Huron	3.4	5	5.6	2.4	3.7
Erie	4.6	7.3	4.3	4.9	5.3
Ontario	3.1	6.2	-0.1	6.9	1.1
10 year: (2011-2020)					
Total	6.5	12.2	-1	9.4	7.1
Superior	18.1	46.2	-24.2	8.3	37.9
Michigan-Huron	7.8	5	6.1	15.5	3.4
Erie	-6.1	-8.6	17.3	-3.4	-18.9
Ontario	-0.1	15.1	6.6	-2.9	-5.4

Table 2: The reference value (mean 1961-1990) for each lake expressed as total annual and seasonal precipitation in millimeters.

	Annual	Winter	Spring	Summer	Fall
Total	878.4	173	201.1	257.2	247.1
Superior	815.8	154.5	169.8	259.4	232.1
Michigan-Huron	852.7	165.3	197.5	245	244.8
Erie	914.7	182.3	231.5	264.4	236.6
Ontario	927.4	207	217.2	244.6	258.6

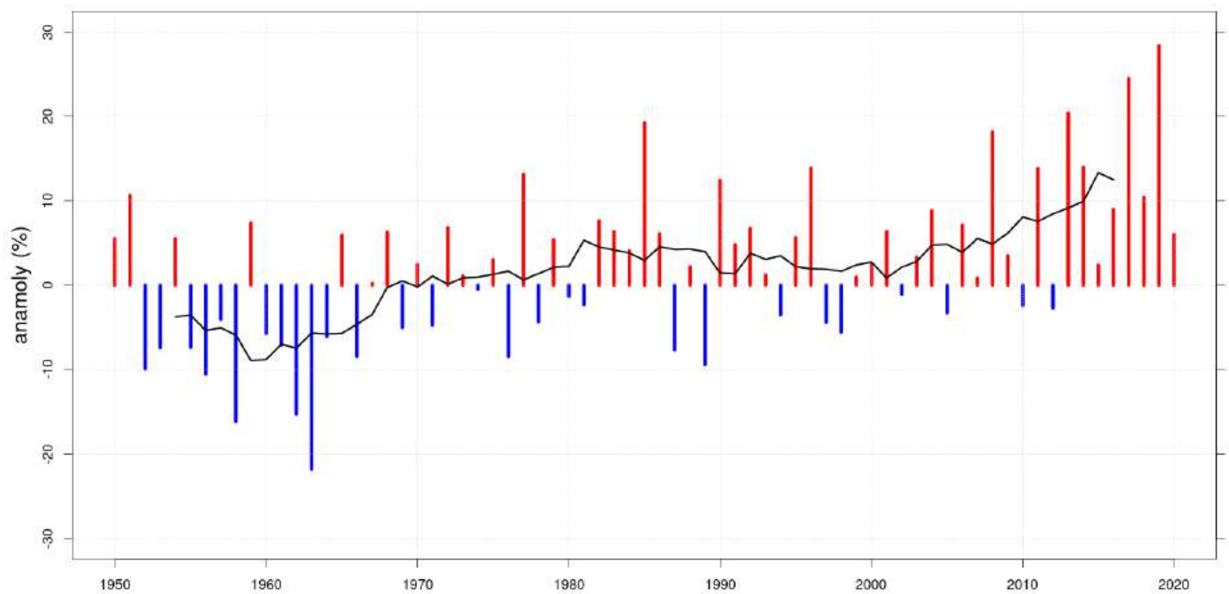
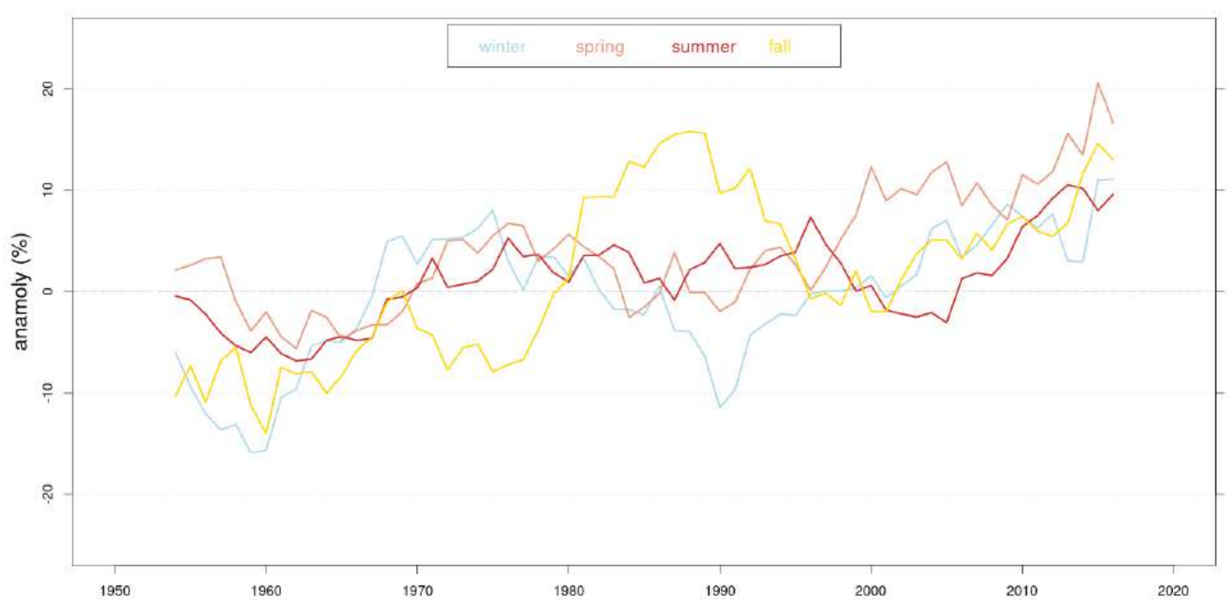


Figure 1. Annual precipitation anomaly (from the 1961–1990 mean) and nine-year running mean for the Great Lakes basin during the 1950–2020 period. Note that the mean for a particular 9-year interval is centered on the middle year, meaning the first year for which the running mean can be defined is 1954 and the last is 2016.



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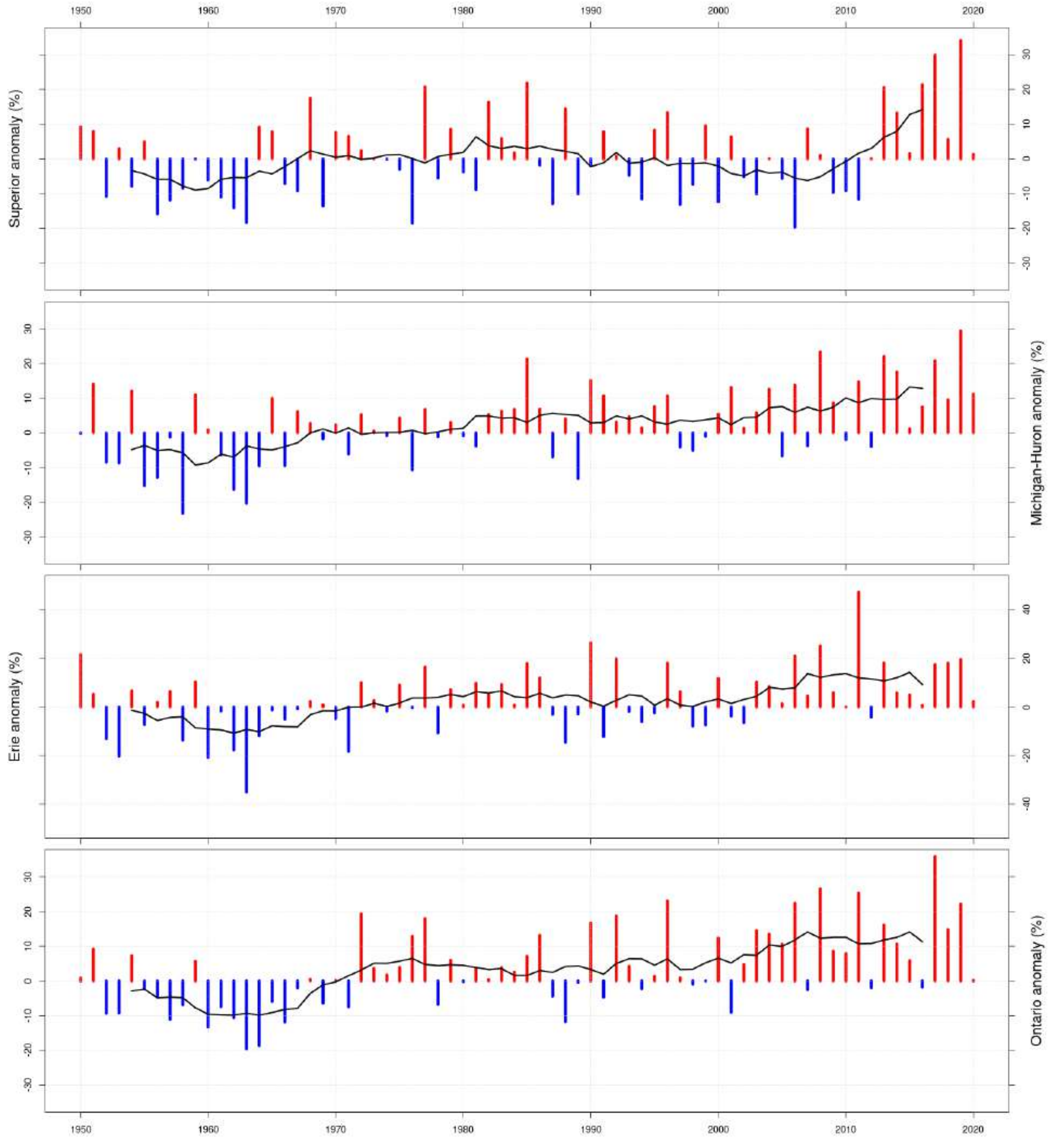

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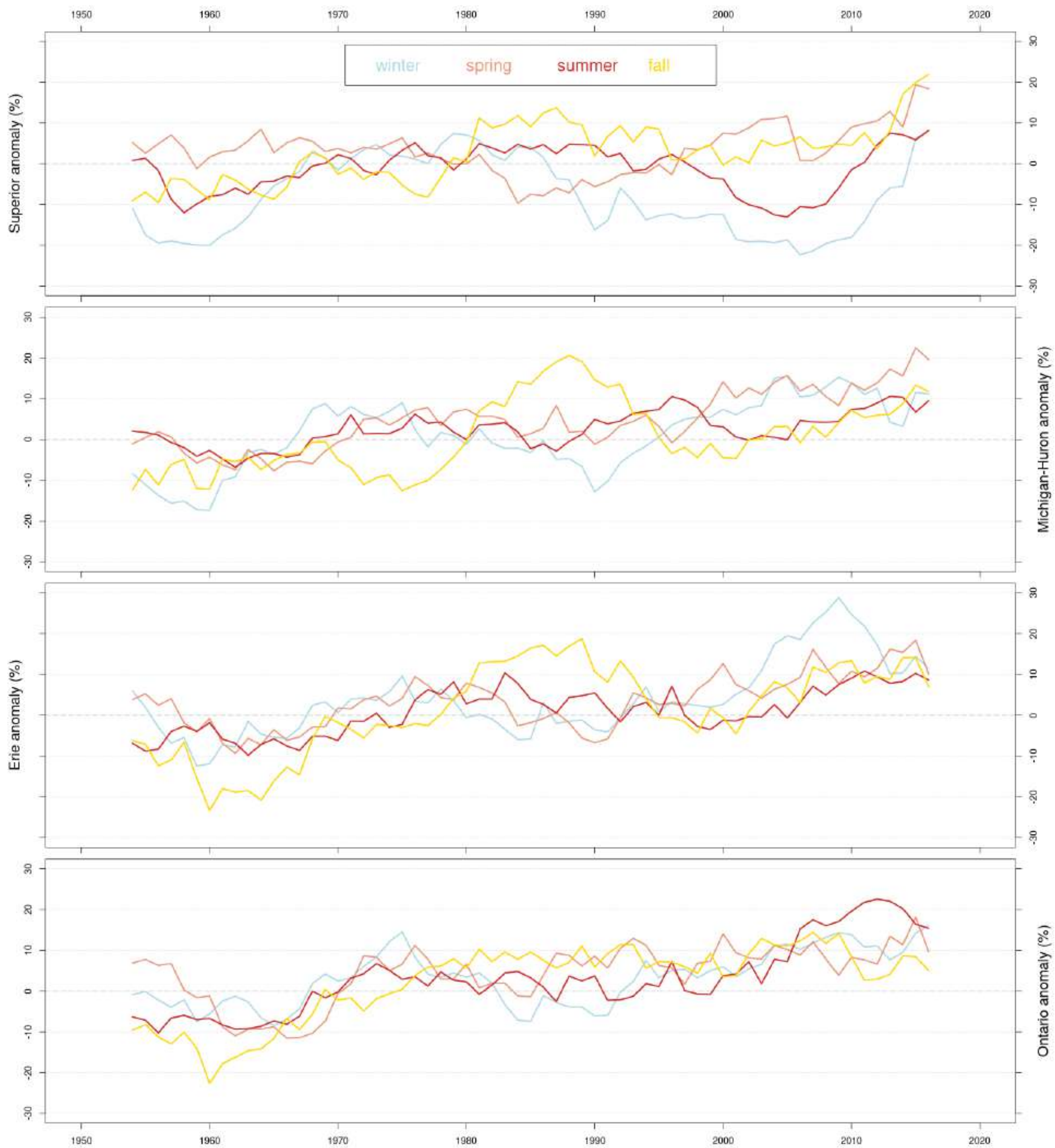

Figure 4. Nine-year running means of seasonal precipitation anomalies (from the 1961–1990 seasonal means) for each Great Lake individually during the period of record (1950–2020).

Sub-Indicator: Water Levels

Overall Assessment

Trends:

10-Year Trend (2011-2020): Increasing on all lakes

30-Year Trend (1991-2020): Unchanging on all lakes

Long-term Trend (1918- 2020): Unchanging to Increasing, depending on lake

Rationale: Notable basin-scale water level dynamics for the 1918-2020 period of record include a significant decline in the late 1990s, persistent and record-low monthly mean levels on Superior (including record lows in August and September 2007) and Michigan-Huron (including record lows in December 2012 and January 2013), followed by a surge in water levels on all lakes that resulted in very high water levels from 2017 to 2020 and included new record high water levels on all lakes in 2017, 2019, and/or 2020, depending on the lake. Water level data on the Great Lakes date back to the 1800s. Any assessment of temporal trends in water levels depends on the period selected from this historical record. For the purposes of this report, an anomaly or deviations based approach has been used to compare 10-year, 30-year and long-term (1918 to 2020) trends for each individual Great Lake. There is some difficulty in determining whether there is a single significant trend over the period of record for all of the Great Lakes. Each individual Great Lake has unique conditions impacting its water levels, so an individual analysis of the water levels of each lake was thought to be more meaningful for assessment of ecological changes due to lake level changes. The base period chosen for comparison purposes was 1918 to 1990 because the water level record begins in 1918, and in order to be consistent with other State of the Great Lakes sub-indicators the base period ends in 1990. Due to the relatively much longer response time of the Great Lakes to changes in other climate inputs compared to other climate sub-indicators, to obtain accurate results, continued monitoring of lake levels is suggested to ensure differentiation of trends. Significant increasing trends in water levels relative to the base period were found for the past 10 years for all of the lakes. However, some caution is advised when using this short a period to predict water level trends into the future, as there is a reasonable possibility that these trends will not continue. For the 103-year period from 1918 to 2020, only Lake Erie showed a statistically significant upward trend in water level.

Lake-by-Lake Assessment

Lake Superior

10-Year Trend (2011-2020): Increasing

30-Year Trend (1991-2020): Unchanging

Long-term Trend (1918-2020): Unchanging

Rationale: Regulation of Lake Superior outflows has had some influence on water levels on the lake since 1916, however, natural hydrological factors such as precipitation, runoff and evaporation continue to be the main determinants in water levels.

Additionally, water diversion from the Hudson Bay watershed into Lake Superior has been occurring since 1939.

Lake Superior annual mean and monthly mean water levels were predominantly below long-term average values (period of record beginning in 1918) between 1999 and 2013. In 2013 and 2014, however, Lake Superior's water level rose at a very high rate, surpassing average values by mid-2014, and were only a few centimeters below record high values by the end of 2017. The rise during 2013 and 2014 was the largest rise observed over a 2-calendar-year period in the historical record since 1918. Very high water supply to Lake Superior in 2017 and again in 2019 culminated in record high water levels from May to September in 2019. Neither the 30-year nor the long-term trends showed a significant trend relative to the base period.

Lake Michigan-Huron

10-Year Trend: Increasing

30-Year Trend: Unchanging

Long-term Trend (1918-2020): Unchanging

Rationale: Lake Michigan and Lake Huron are commonly considered one lake system from a hydrological perspective due to direct connection of the lakes through the Straits of Mackinac, and therefore are referenced collectively as Lake Michigan-Huron. Lake Michigan-Huron's annual mean and monthly mean water levels were predominantly below long-term average values between the late 1990s and late 2014. In late 2014, however, as part of a very rapid 6-year rise that began in early 2013, water levels rose above long-term average values and have remained well above average to the end of 2020. These high water levels included an eight month period from January of 2020 to August of 2020, when the monthly average water level was the highest for each month in the period of record (1918-2020).

Both the 30-year and long-term trend suggest that there is no significant long-term trend in water level relative to the base period.

Lake Erie (including St. Clair-Detroit River Ecosystem)

10-Year Trend: Increasing

30-Year Trend: Unchanging

Long-term Trend (1918-2020): Increasing

Rationale: Monthly mean and annual mean water levels on Lake Erie oscillated above and below long-term average values for the past decade, including from 2011 through 2015, but the 10-year period ending in 2020 shows an upward trend relative to the base period. In 2019, the average lake level for June was the highest lake level ever recorded on Lake Erie for any month, while in 2020 the 4 months between February and May set individual monthly records. Located between Lake Huron and Erie, Lake St. Clair also experienced record high monthly lake levels during 2019 and 2020.

Lake Ontario (including Niagara River and the International section of the St. Lawrence River)

10-Year Trend: Increasing

30-Year Trend: Unchanging

Long-term Trend (1918-2020): Unchanging

Rationale: Regulation of outflow from Lake Ontario has influenced water levels on the lake and the St. Lawrence River since 1960, however, natural hydrological factors such as precipitation, runoff and evaporation continue to be the main determinants in water levels. Generally, the overall range and inter-annual variability of water levels on

Lake Ontario has been reduced due to regulation when compared to pre-regulation water levels. A new water regulation plan (Plan 2014) was implemented in early 2017, that was designed to result in a similar overall range in Lake Ontario water levels relative to the previous regulation plan, but an increase in variability of water levels on Lake Ontario closer to that of natural conditions based on historical water supplies. In the years since the implementation of Plan 2014, the basin has experienced some of the wettest conditions on record, resulting in very high lake levels, especially in 2019. During that year the level for June was the highest ever monthly average recorded in the period of record (1918-2020).

Trend Assessment Definitions

Trend reporting was based on comparison of annual mean water level to the long-term average annual water level for each lake from 1918 to 1990. Anomalies were calculated between the average water levels and a trend line was established by fitting a linear model to the anomaly data. The significance of the trend was determined using p-values from a bootstrap based Mann-Kendall model to account for significant autocorrelation identified in the data. This is a change in methodology from the previous report, which identified trends using p-values from a linear model fit. A trend was considered significant for probability of occurrence less than 0.05 ($P<0.05$) (Table 1). It was noted that the analysis was relatively sensitive to the period of analysis (1918-2020), which is not surprising due to interannual and multi-year variability of Great Lake water levels.

Increasing: an increasing anomaly trend line with a $P<0.05$ indicating that water levels are increasing relative to baseline water level.

Unchanging: an anomaly trend line with $P>0.05$ indicating that there was no statistically significant trend

Decreasing: a decreasing trend line with a $P<0.05$ indicating that water levels are decreasing relative to baseline water level.

Undetermined: data are not available to report on a trend

Sub-Indicator Purpose

- The purpose of this sub-indicator is to track seasonal, inter-annual, and long-term (i.e. decadal) trends in lake-wide average water levels across each of the Great Lakes.
- The water levels sub-indicator is used in support of the climate change category, as well as general objective #9 of the Great Lakes Water Quality Agreement (GLWQA) and the climate change, lake-wide management, and habitat and species annexes of the GLWQA.

Ecosystem Objective

Water level fluctuations have strong influences on Great Lakes habitats and the biological communities associated with them. Impacts of alterations in water level fluctuations on shoreline ecosystems (particularly coastal wetlands) are widely-documented, and underscore important additional (but less apparent) relationships between ecosystem response, human intervention, and climate change.

This sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should be "free from other substances, materials, or conditions that may negatively impact the chemical, physical or biological integrity of the Waters of the Great Lakes."

Measure

This water level sub-indicator measures:

1. Long-term water level variability: an assessment of trends in the anomalies of annual mean water levels relative to the base period of 1918-1990 over the period of record for each of the Great Lakes (data from 1918-2020).

The set of measures associated with this sub-indicator are calculated from existing estimates of lake-wide average water levels based on gage measurements since 1918. This formal network of gages for each lake was established and has served as the basis for an internationally-coordinated set of monthly mean lake-wide average water level measurements. The 1918 to 2020 period was chosen as the analysis period, because data for this period have been internationally coordinated by the Coordinating Committee for Great Lakes Basic Hydraulic and Hydrologic Data. Although the basin has experienced elevation changes as a result of post-glacial isostatic rebound, these changes are accounted for in the long term coordinated water level record.

Water levels based on gage measurements are also available dated back to 1860, the year in which at least one gage ("master gage") was installed along the shoreline of each of the Great Lakes. The use of only one gage to represent a lakewide average may not account for local variations in water levels. However, an analysis of the available data over the 1860 to 2020 period was performed and differences in long-term trends were found compared to using the 1918 to 2020 period. Using data over the 1860 to 2020 period resulted in Lake Michigan-Huron having a significant decreasing trend, and both Lakes Erie and Ontario showing an unchanging trend. This indicates sensitivity of the analysis to period of record chosen for the analysis.

Similar to the 2019 reports, an anomaly-based approach to reporting on the climate trend sub-indicators, which include Water Levels, Precipitation Amounts, Ice Cover and Surface Water Temperatures, has been implemented for 2022 State of the Great Lakes reports.

The approach includes identifying a defined base period and comparing all data (the full period of record) to this base period to identify "anomalies" or "deviations" from the base period mean. The base periods selected in the reports include up to 1990 but not beyond as it is expected that accelerated "climate change" effects would be occurring after 1990. The base periods vary in length for these 4 reports, but includes a minimum of 15 years of data that are reliable and comprehensive to calculate the base values. The base period for the water level sub-indicator was chosen to be from 1918 to 1990 as this period's data is considered reliable and is coordinated between Canada and United States. Note that the trends (if not the magnitude of the annual anomalies) resulting from the analysis of anomaly data are the same regardless of base period.

Ecological Condition

Changes in Great Lakes water levels take place over a variety of time scales ranging from hourly fluctuations to those taking place over hundreds and even thousands of years. Most of the monthly and interannual changes documented over the past 150 years are the result of changes in the Great Lakes hydrologic cycle. For example, the recent lake level fluctuations in the mid- to late-2010s, were found to be the result of weather extremes and climate variability (Gronewold and Rood, 2019). These are influenced by natural and anthropogenic factors, and long-term climate trends (Baedke and Thompson 2000; Booth and Jackson 2003). Anthropogenic factors include items such as the regulation of the outflow and inter-basin diversions, although it is generally considered that these have much less of an influence than the natural factors. Changes in Great Lakes water levels not only impact the shoreline, which will have impacts on the area of accessible and suitable habitat, but also the complex and dynamic coastal processes of the lakes. The potential for impacts of water level changes to physical habitats around the Great Lakes are dependent on a complex interrelationship of many factors which include erodibility of shoreline substrates, sediment supply and shoreline management practices. Many of the shorelines of the Great Lakes are dynamic, with the sorting action of erosion and accretion playing an important role in habitat creation, while other areas consist of natural bedrock or engineered structures which will have less potential for erosion with associated lower sediment supply to replace features that are susceptible to erosion. Higher water levels can cause more erosion along shorelines consisting of unconsolidated substrates as wave energy works further up the shore on less stable banks. However, the sediment supplied by erosion is important in maintaining dynamic shoreline features such as beaches and underwater gravel areas. During times of low water levels, wave action will tend to move unconsolidated substrates, redistributing beach material over time. When considering ecological impacts due to changes of water levels, changes in shoreline dynamics and the associated change in shoreline structure for short and long-term periods should be considered. Additionally, the impact that changing water levels may have on shoreline development practices such as shoreline protection structures, water pollution sources, dredging and in water work with their potential changes in habitat may also be considered.

Water level conditions have historically varied, and continue to vary considerably between each of the Great Lakes and temporally over the period that Great Lakes levels have been recorded (1860 to present). Various modifications to the Great Lakes watersheds and their connecting channels have had some impact on water levels to varying degrees in the last 100 years, in addition to the potential impacts due to climate change. It should be noted that the impacts due to anthropogenic changes along with the flow characteristics of the Great Lakes basin were not separated for the purposes of this analysis, but are discussed in the report. Water level fluctuations at finer spatial scales (such as the St. Marys River) were not considered in this report. In general, it can be assumed that higher levels on Lake Superior will provide generally higher flow to the St. Marys River. However, specific impacts to the connecting channel in regard to water depths, velocities and wetted perimeter would need more detailed study.

The summary below is an account of the overall factors affecting the Great Lakes water balance (Neff and Killian 2003), and ultimately the water levels, and a limited discussion on water level history and variability.

The natural factors associated with long-term water level changes in the Great Lakes include environmental processes that contribute to inflow, outflow, and storage in the system. Within broad scales, water inflow and outflow are dictated by climatically-induced changes that affect the components of the hydrologic cycle. These components include over-lake precipitation, runoff, over-lake evaporation, and connecting channel flows (i.e. the flow of water into and out of each lake through the upstream and downstream connecting channel). Groundwater flows are assumed to have an insignificant contribution to long-term water level variability relative to over-lake precipitation, runoff, and evaporation.

An additional natural factor that affects the accurate measurement of water levels, as well as the wetted perimeter of the lakes with its associated impacts on habitat, is glacial isostatic adjustment (GIA), which is the response of the earth's crust to removal of the weight of the last glacial ice sheets that crossed the area (Wilcox et al. 2007; IUGLS 2009). Unlike hydrologic factors, GIA impacts on water levels vary from one location to another around a lake. At some locations, water levels appear to be rising as a result of GIA, while levels appear to be falling at other locations on the same lake. This has an implication for the analysis of historic water level data. Measurement error due to GIA is minimized by using a representative set of gauges around each lake to calculate a lake wide average level and by periodically adjusting the datum for level measurement. As well as introducing some water level measurement error, GIA will also change the wetted perimeter of each lake over time, generally with the more northern portions of the lake drying and southern portions of the lakes seeing more wetted area. The changes in wetted perimeter of the lakes due to GIA was not specifically considered in this analysis, however for comparison the northern part of Lake Superior is rising at a vertical velocity of about 50 cm/century and part of the southwest shoreline of Lake Michigan is falling at a vertical velocity of about 14 cm/century.

Lake Superior water levels have been regulated since 1916. In its 1914 Order of Approval, the International Joint Commission (IJC) established the International Lake Superior Board of Control and delegated to it responsibility for setting Lake Superior outflows. The Board of Control established a regulation plan that has undergone several revisions. The regulation plan currently in place incorporates the concept of balancing Lake Superior and Lake Michigan-Huron levels.

With the approval by the IJC of the hydropower project at Cornwall, Ontario and Massena, New York under the Order of Approval of 1952, Lake Ontario's outflow became subject to regulation. The first regulation plan became operational in 1960. The subsequent reduction of the variability in Lake Ontario water levels has been shown to diminish wetland plant diversity and the habitats they support (LOSLR Study Board, 2006). A new water regulation plan (Plan 2014) was implemented in early 2017, that was designed to result in a similar overall range in Lake Ontario water levels relative to the previous regulation plan, but an increase in variability of water levels on Lake Ontario closer to that of natural conditions based on historical water supplies. An expected outcome of this more natural variability in Lake Ontario water levels is to increase the suitability of habitat in wetlands for a number of plant and wildlife species. The plan has not been in place for a long enough duration to make any conclusions about its long-term effect on the variability of lake levels.

Water levels are measured at several locations along the shore of the Great Lakes and their connecting channels by the National Oceanic and Atmospheric Administration (NOAA) in the United States and by the Canadian Hydrographic Service (CHS) in Canada. Several gauges in the current network of multiple gauges have been in operation only since 1918, while others have gauge records (some less reliable) extending back to the 1860s.

Status and Trends in Lake Levels

Graphs of water level anomalies relative to the base period (Figure 1) show some similarities of interest. Generally, periods of higher levels occurred in the late 1920s, the mid-1950s, from the early 1970s to mid-1990s, and now in the late 2010s and early 2020s. Pronounced low lake level periods occurred in the mid-1920s, the mid-1930s, the mid-1960s, and the 2000s. Though less-well documented, low levels also occurred in the late 1890s, following a long period of high lake levels. In addition to computing trends, Figure 1 also includes a line showing the 10-year rolling average anomaly.

Based on the historical record as shown in Figure 2, there appears to be a range within which the lake levels remain, but paleo records indicate a range that may have been greater (Brown et al. 2012). There is considerable uncertainty in how climate change, particularly changes in precipitation, runoff, and evaporation, may impact net basin water supplies and water levels and flows in the Great Lakes-St. Lawrence River region. The current state-of-the-art in climate models indicates that water levels are likely to continue to fluctuate both above and below their long-term averages in the future, but that there is not strong evidence for a pronounced shift in the long-term mean (e.g. Notaro et al., 2015, Lofgren et al., 2016, Lofgren et al., 2011, MacKay and Seglenieks, 2013). Generally, the models also indicate that the lakes may experience more extreme levels (both high and low) in the future climate.

Other Spatial Scales

Water level fluctuations at finer spatial scales (including the fluctuations on Lake St. Clair) are not considered in this report.

Linkages

- Coastal Wetland sub-indicators (Coastal Wetland Invertebrates, Coastal Wetland Fish, Coastal Wetland Birds, Coastal Wetland Amphibians, Coastal Wetland Plants, and Coastal Wetlands: Extent and Composition) – water levels have a major influence on undiked coastal wetlands and are basic to any analysis of wetland change trends.
- Phytoplankton, Zooplankton, Benthos, *Diporeia*, Native Preyfish Diversity, Lake Trout, Walleye, Lake Sturgeon, and Fish Eating and Colonial Nesting Waterbirds – water levels are related to habitat available for food, cover and travel corridors for these animals.
- Ice Cover- Evaporation rates from the lakes are in part related to ice cover in the winter.
- Surface Water Temperature – ice cover and lake evaporation are linked to surface water temperature.
- Precipitation Amounts – are linked to lake levels as this is the source of water for the Great Lakes watershed.
- Tributary Flashiness – are linked to lake levels as water from the surrounding tributaries feeds the lake.
- Climate Change – linked to changes in hydroclimate conditions that impact water levels on short and long-term time scales.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: http://www.greatlakescc.org/wp36/home/coordinating-committee-products-and-datasets/		

Data Limitations

Great Lakes long-term water level data are based on an internationally-coordinated set of measurements from shoreline gaging stations, and are explicitly intended to be comparable for both the United States and Canada. Issues regarding the integrity of the monitoring network are typically addressed by the federal agencies that participate in the Coordinating Committee on Great Lakes Basic Hydraulic and Hydrologic Data.

Additional Information

Maintenance and operation of the Great Lakes water level monitoring network is critical to understanding the Great Lakes regional water budget, and is currently a central regional mission of both the National Oceanic and Atmospheric Administration - National Ocean Service (NOAA-NOS), and the Fisheries and Oceans Canada - Canadian Hydrographic Survey (CHS). Synthesizing and communicating lake-wide average water level data is coordinated through the Coordinating Committee on Great Lakes Basic Hydraulic and Hydrologic Data, a regional partnership led by the United States Army Corps of Engineers (USACE) Detroit District, and Environment and Climate Change Canada (ECCC). Continuation of an accurate Great Lakes water level monitoring network is important in determining future changes in the Great Lakes.

This analysis provides an indication of water level trends over different time scales, and there are dozens (if not hundreds) of important connections to regional climate trends; commerce, such as commercial shipping and hydropower capacity; and ecosystem and human health. These connections are likely more clearly identified within other ecosystem indicators (including, for example, shoreline integrity, coastal wetlands, and ice cover), and should continue to be emphasized in future iterations of this effort.

An anomaly-based approach was chosen for this sub-indicator to be consistent through the report and is a more representative way of describing the state of an ecosystem variable where the typical value is not necessarily common knowledge. It also allows for easy comparison of the state of variables that are not necessarily similar, such as precipitation in different seasons or different geographic locations. Since the base period mean is a constant, trends determined using the anomalies are identical to trends determined from the original data. The anomaly-based approach helps put the longer periods of record into context and compares the historical data to more recent changes that are occurring in the ecosystem.

Water level variability across different time scales can serve as an index of significant changes in regional meteorology and climate and a reflection of anthropogenic influence (including regulation of outflows from Lakes Superior and Ontario), and an important indicator of potential impacts on coastal ecosystems, hydropower capacity, and other socioeconomic factors.

Water levels on each of the Great Lakes follow a strong seasonal pattern in which water levels tend to rise in the spring (as a result of increasing precipitation, melting of snow from the previous winter, and decreasing over-lake evaporation) until a peak is reached in mid-summer. Water levels then typically decline through the summer into the fall months (primarily through increased evaporation rates and reduced runoff), reaching a typical seasonal low in early winter. Persistent shifts in the timing of either the seasonal maximum or minimum may reflect shifts in the regional water budget (including changes in the timing and magnitude of precipitation, tributary flows, and evaporation) and provide insight into potential impacts on aquatic plants and fish spawning habitats, and other sensitive aspects of the coastal ecosystem.

A persistent increase in the magnitude of spring rise might reflect increasing “flashiness” in tributary inflows, while periods of decreased declines in the fall may reflect cooler water temperatures and diminished evaporation rates.

These drivers of the water budget reflect changes in the regional climate system and have important implications for the magnitude of seasonal rises and declines, and impacts on the coastal ecosystem. Long-term changes in Great Lakes water levels often occur through persistent above or below average water level changes in the spring and fall. For example, systematic increases in long-term water levels are often a consequence of consistent above average runoff rates in the spring and below average evaporation rates in the fall. In addition, the magnitude of seasonal rise and decline (within a given year) has important implications for coastal recreational activities and the design of coastal infrastructure. It also has implications for biological phenology and sediment-water nutrient exchange.

Differences between the water levels of each of the lakes may follow a relatively consistent and predictable pattern; anomalies in these differences may suggest an imbalance in the regional water budget, physical changes in the channels that connect the lakes, or the apparent and physical impacts of glacial isostatic adjustment on recorded water levels.

Other anthropogenic changes in the last 100 years have had some impacts on Great Lakes water levels outside of those that may be caused by climate change. Some of these include water diversions between watersheds (e.g. Long Lac and Ogoki water diversions have diverted water from the James Bay watershed into Lake Superior watershed, and the Chicago diversion diverting water from Lake Michigan to the Mississippi River), channelization and dredging (e.g. dredging of the Detroit - St. Clair River System for ship navigation, construction of the Welland Canal) and changing regulation plans (e.g. Lake Superior regulation since 1916 and Lake Ontario regulation since 1960). Specific analysis has not been undertaken to separate these effects for this report.

Changes in Great Lakes water levels, either due to anthropogenic or climate-change induced factors, provide an important key to assessing future potential ecological vulnerabilities. The interaction between ecosystem components and physical water levels, along with other climatic variables, is extremely complex, and is an evolving science. As the science of ecosystem assessment evolves, methods such as critical threshold criteria should continue to be considered in assessment of potential impacts of varying lake water levels. This could include consideration of critical thresholds for lake levels as well as development of finer scale understanding of depths and velocities of the connecting rivers (e.g. St. Marys River, Detroit-St. Clair River, Niagara River and St. Lawrence River).

The authors concur with previous State of the Great Lakes reports that additional future reporting cycles may want to focus on explicit connections between water level variability, coastal processes and ecosystem response. Given the water level patterns over the past 10 years, there might be a significant opportunity to improving that understanding if and when hydrological, coastal process and ecological data becomes available.

The existing long-term water level monitoring network on the Great Lakes is robust, and is a core component of the operational mission of federal agencies in both the United States and Canada. In the near-term, both countries' agencies are cooperating on a major update to the International Great Lakes Datum (IGLD); the benchmark against which surface water elevations across the Great Lakes are measured. It will be important for future water level inference to take the IGLD adjustments into consideration. Alternative technologies for monitoring surface water elevations in the Great Lakes, including satellite telemetry, are and will continue to be available in the future, however it is unlikely that (at least in the near-term) they will replace the data provided by the long-term shoreline-based gaging network.

In the last report, one measure that was recommended as having the potential to provide additional insight into the recent water level rises seen across the Great Lakes was changes in the magnitude of the seasonal rises and declines throughout the period of record (1918-2020). In this report, we provide an analysis of seasonal rise (Figure 3) and seasonal decline (Figure 4) using a similar anomaly-based approach to what was used for annual average water levels in Figure 1. The seasonal rise was calculated by computing the change in level from the seasonal low to the seasonal peak water level. The seasonal decline was calculated by computing the change in level from the seasonal peak to the seasonal low in the following fall/winter/spring. The base period used for these graphics were 1919-1990 for the seasonal rise and 1918-1990 for the seasonal decline. The seasonal declines are referenced to the year that the peak occurred. For example, if the seasonal peak level occurred in July 1918 and the seasonal low occurred in the following year in February, the decline would be referenced to the year the maximum occurred (July 1918). The seasonal rises are also referenced to the year the seasonal peak occurred.

Overall, the only significant ($p<0.05$) trends found in this rise/decline analysis using a linear model fit were long-term trends in the seasonal rise (increasing) and seasonal decline (increasing) for Lake Ontario. However, by plotting the magnitude of the seasonal rise and declines across all of the lakes it becomes apparent that the recent high water period was not only a result of strong water level rises in recent years, but also smaller than normal seasonal declines, except on Lake Ontario, which had larger than average seasonal decline in recent years. The large rises are noticeable on Lakes Superior and Lake Michigan-Huron in 2013 and 2014, as well as 2017 and 2019 on Lake Michigan-Huron. Lake Erie had above normal seasonal rises in all years from 2013 to 2019, with 2015 and 2017 being anomalously high. Lake Ontario experienced very large rises in 2017 and 2019, which also corresponded to the record high levels seen during those two years. Although in recent years the amount the water levels have risen have been higher than normal, the water level rises in 2020 were below normal on all lakes. The most recent seasonal decline in 2019 was less than normal on Lakes Superior and Michigan-Huron, which led to the record high water levels experienced in the beginning of 2020. However, Lakes Erie and Ontario experienced a greater than normal seasonal decline in 2019. Throughout the transition from low to high water, Lake Superior experienced below normal water level declines from 2013 to 2019, while Lake Michigan-Huron also had below normal water level declines from 2013 to 2015 and 2017 to 2019. In recent years, Lake Erie has predominantly seen near normal water level declines, with the exception of 2018, where the decline was below normal. Lake Ontario has experienced above normal water level declines from 2011 to 2019, with especially high declines in 2017 and 2019, which can be attributed to reduced net basin supply and increased Lake Ontario outflows.

Other measures to consider in the future could include:

1. Timing of seasonal water level maximum and minimum: an assessment of changes over time in the month in which the seasonal water level maximum and minimum occur. (data from late-1990s-2020)

2. Lake-to-lake water level difference: an assessment of long-term trends in the difference between the monthly mean water level for each lake and the monthly mean water level for the downstream lake. (data from 1918-2020)

3. Changes in coastal processes on the Great Lakes with changes in water levels and associated impacts on habitat structures.

4. The conclusions of this report are based on the best available data set at the time of writing this report (i.e. 1918 to 2020). Trends should be reevaluated as data becomes available.

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Table 1. Trend line p-values for lake level anomalies for 1918-2020 computed using 1918-1990 baseline period. A value of p<0.05was considered significant. Data from USACE.

Trend	Great Lake			
	Superior	Michigan-Huron	Erie	Ontario
10-year	0.00	0.00	0.01	0.02
30-year	0.79	0.82	0.49	0.70
Long-term	0.80	0.53	0.03	0.09

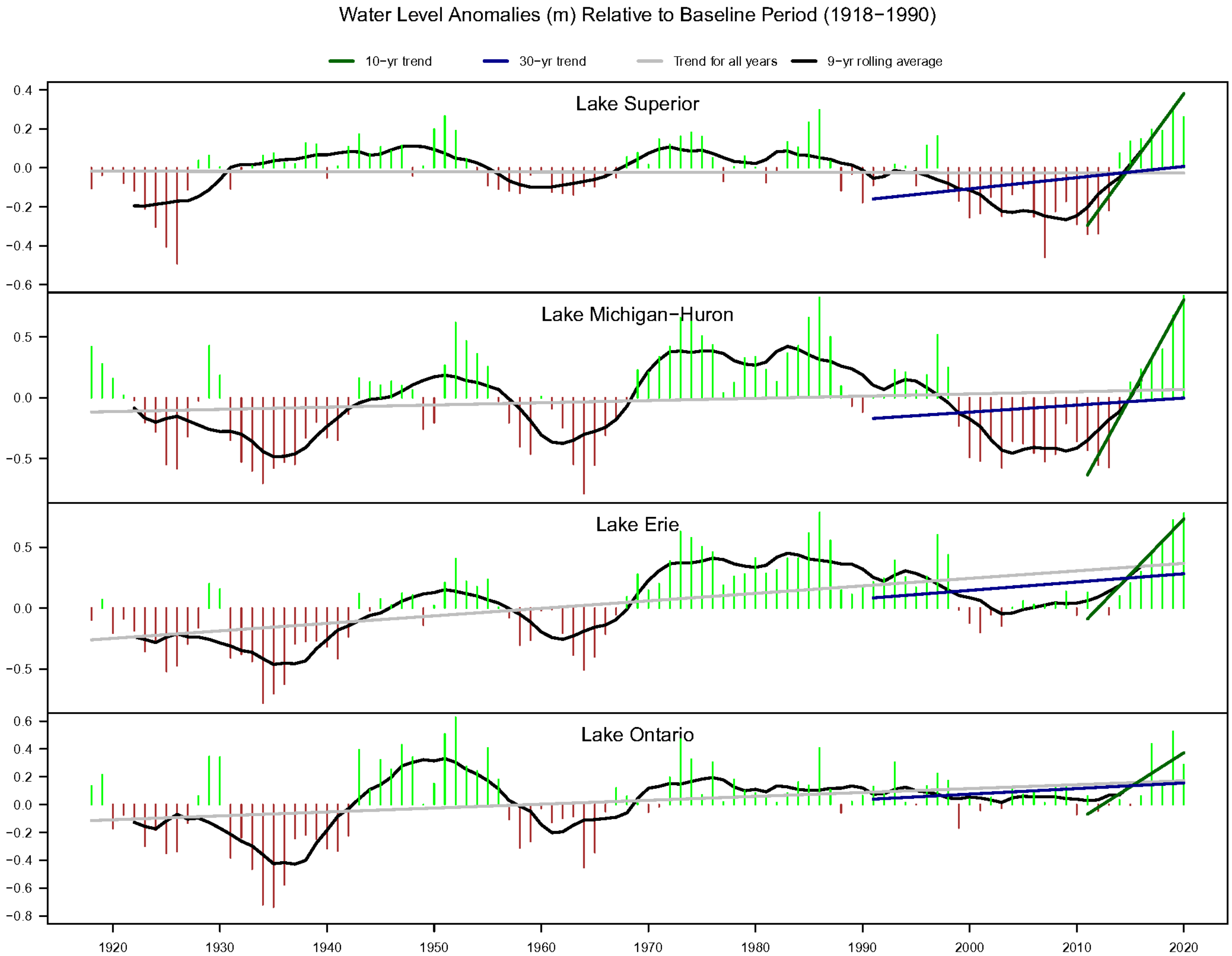

Figure 1. Water level anomalies (m) relative to average of the baseline period 1918-1990 for each Great Lake over the period 1918-2020. Note that the mean of each 9-year interval is centered on the middle year so the first year of the running mean is 1922. Data from USACE.

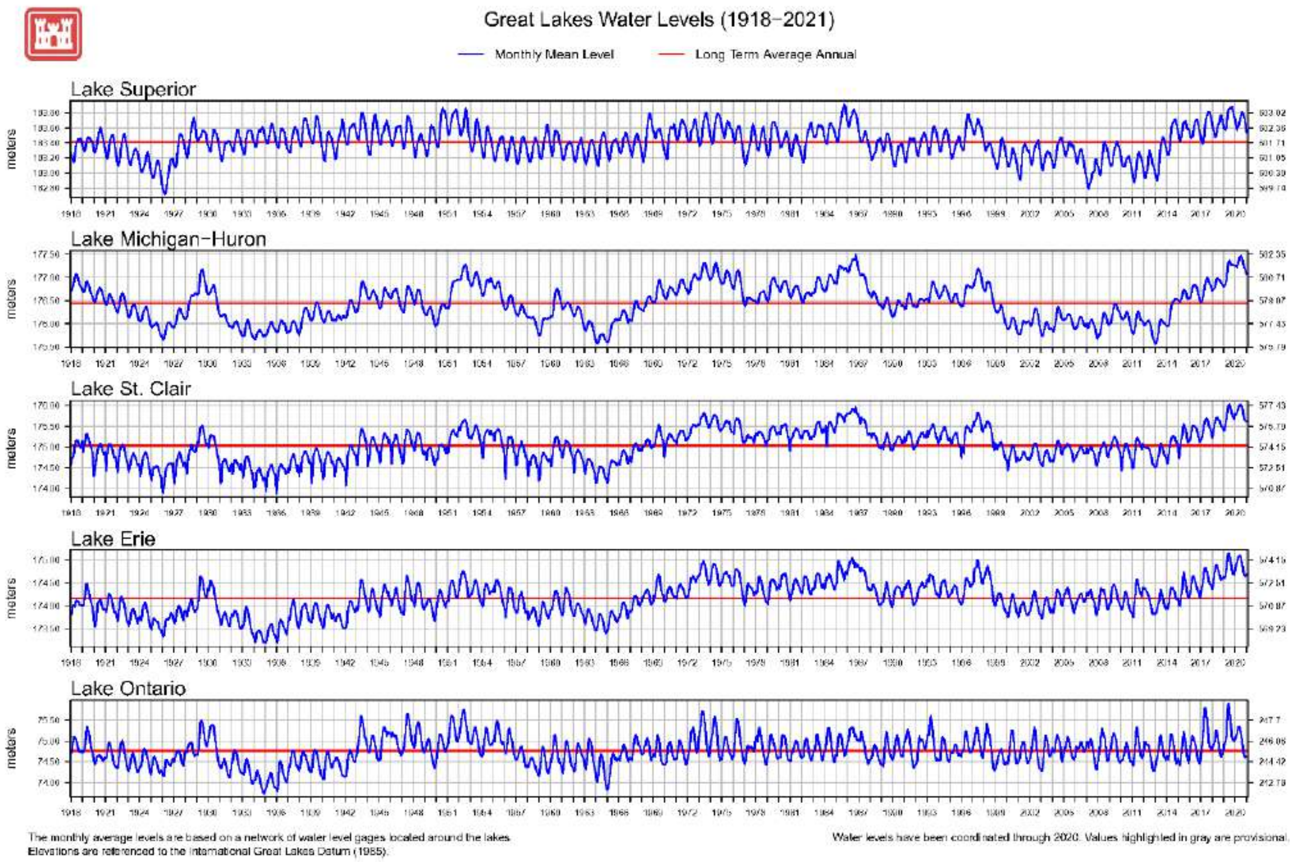

Figure 2. Monthly and yearly mean annual water levels (m) for the Great Lakes.
Source: U.S. Army Corps of Engineers, Detroit District.

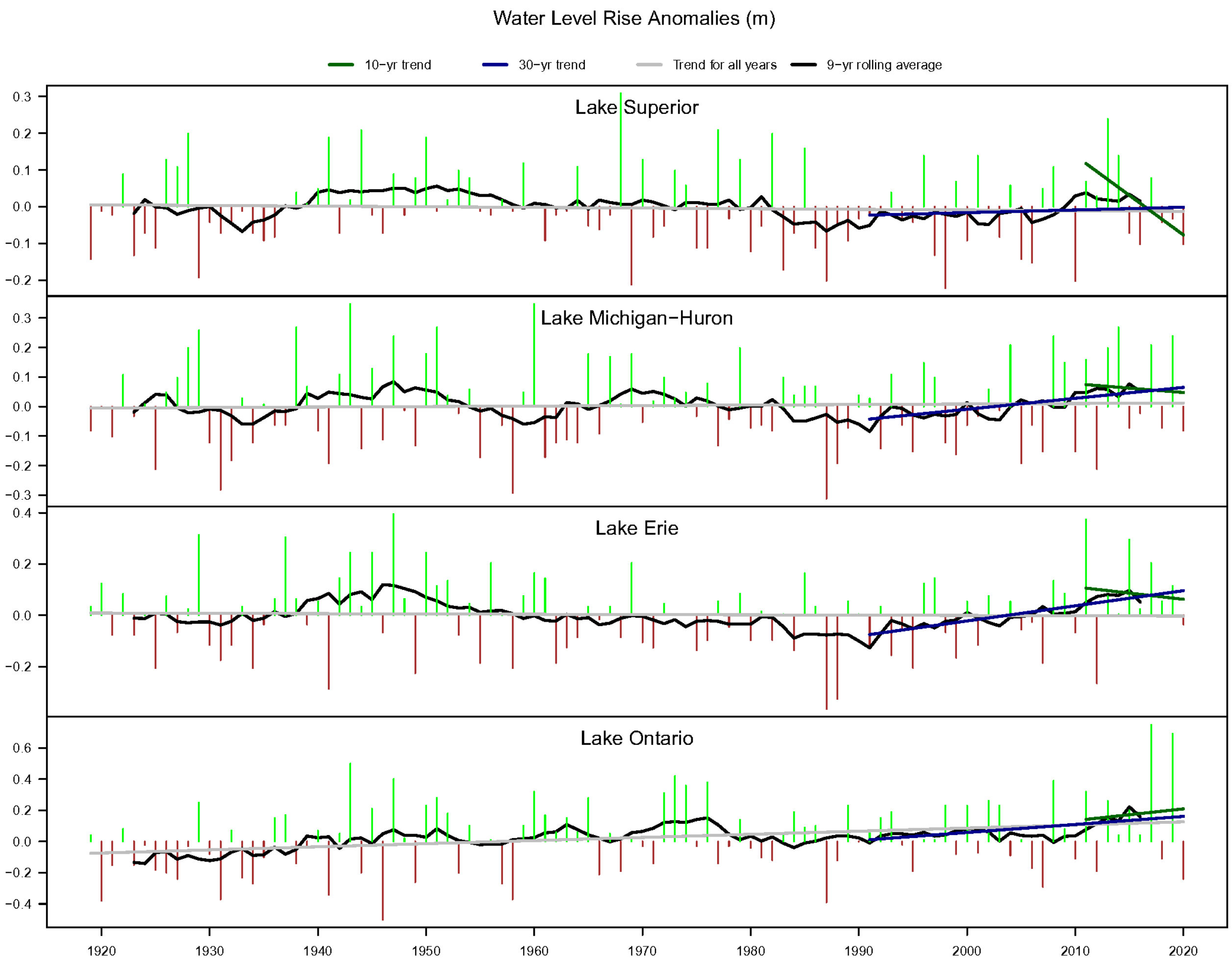

Figure 3. Water level rise anomalies (m) relative to the average rise of the baseline period 1919-1990 for each Great Lake over the period 1919-2020. Note that the mean of each 9-year interval is centered on the middle year so the first year of the running mean is 1923.

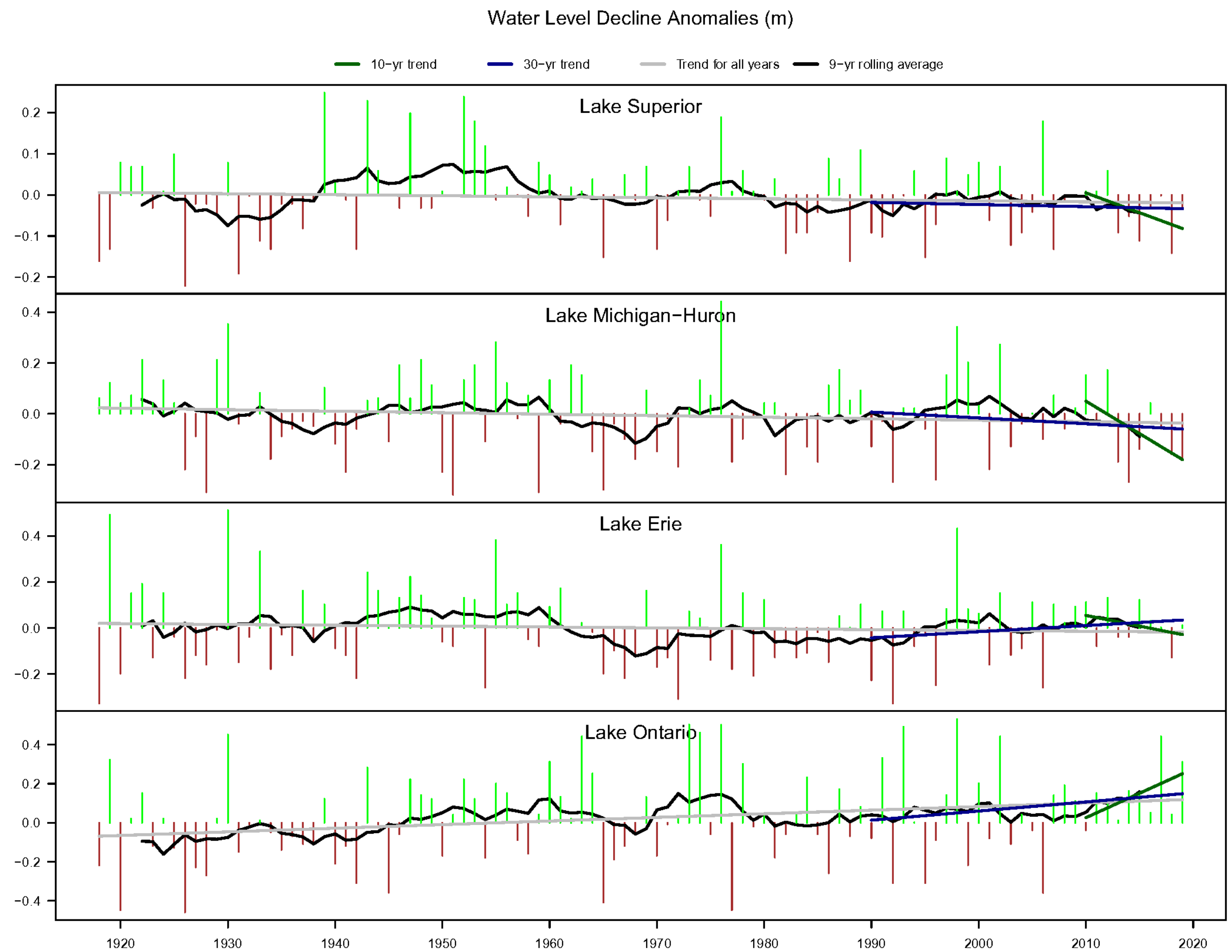

Figure 4. Water level decline anomalies (m) relative to the average decline of the baseline period 1918-1990 for each Great Lake over the period 1918-2019. Note that the mean of each 9-year interval is centered on the middle year so the first year of the running mean is 1922.

Sub-Indicator: Surface Water Temperature

Overall Assessment

Trends:

10-Year Trend: Unassessed

30-Year Trend: Increasing

Long-term Trend (1980-2020): Increasing

Rationale: An overall assessment is challenging to assign given that the lakes' surface water temperature responses to changes in metrological forcing and inflows (i.e., rivers and groundwaters) are geographically different. However, in general, based on the buoy observations (Figures 1-3), it appears that the upper Great Lakes (Superior, Michigan, Huron) show consistent trends towards higher temperatures over the last 30 years and will be classified as "Increasing", with an average trend of ~0.06℃ yr^{-1}. In the lower Great Lakes (Erie and Ontario), the rate of temperature increases on average was much slower at ~ 0.03°C yr^{-1}. Lake Erie (Figure 2) positive trend was spatially variable ranging from ~0.02°C yr^{-1} in the west basin to ~0.05°C yr^{-1} in central and eastern basins with fewer data available. Lake Ontario (Figure 3), while displaying positive trends at two eastern sites (0.05°C yr^{-1} and 0.09°C yr^{-1}), has significantly fewer data available, with less than 20 years. Model output is generally consistent with the buoy output (Figure 4) but shows distinct biases at some locations. Due to the magnitude of interannual variability compared to the long-term trend, measures such as the 10-year trend will not be assessed.

The year 1980 is the start of buoy-based surface water temperature observations in the Great Lakes – see Table 1 for more details, and the term long-term was used to reflect the most extended historical observation period to present.

Status and Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Lake-by-Lake Assessment

Lake Superior

10-Year Trend: Unassessed

30-Year Trend: Increasing

Long-term Trend (1980-2020): Increasing

Rationale: All three buoys deployed on Lake Superior show trends towards warmer summer surface water temperature, all on the order of 0.07 ± 0.03 °C yr^{-1}. The average temperature over the last 10 years, estimated by the Large Lakes Thermodynamic Model (LLTM), is approximately 3°C higher than the baseline period (1960-1990) average. As determined by the model, the average date on which the lake reached the temperature of 3.98°C (T_{MD}, maximum density) over the last 10 years is 25 days earlier.

Lake Michigan

10-Year Trend: Unassessed

30-Year Trend: Increasing

Long-term Trend (1980-2020): Increasing

Rationale: Both buoys deployed on Lake Michigan show trends towards higher summer surface water temperatures, on the order of 0.06 ± 0.02 °C yr^{-1}, similar to Lake Superior trends. The average temperature over the last 10 years, estimated by the LLTM, is approximately 4°C higher than the baseline period average. The average date of the lake reaching T_{MD} over the last 10 years is 39 days earlier.

Lake Huron (including St. Marys River)

10-Year Trend: Unassessed

30-Year Trend: Increasing

Long-term Trend (1980-2020): Increasing

Rationale: Both buoys in Lake Huron show trends towards higher summer surface water temperatures on the order of 0.06 ± 0.02°C yr^{-1}. The average temperature over the last 10 years estimated by the numerical model is approximately 3°C higher than the baseline period average. The average date of the lake reaching T_{MD} over the last 10 years is 18 days earlier. As with Michigan and Superior, 2014 stands out as a particularly cold summer. Model output from LLTM confirms this, with temperatures after 1997 significantly higher than the "baseline" temperature.

Lake Erie (including St. Clair-Detroit River Ecosystem)

10-Year Trend: Unassessed

30-Year Trend: Increasing

Long-term Trend (1980-2020): Increasing

Rationale: Three buoys (one NDBC, two ECCC) show trends towards warmer summer surface water temperatures. The trend at the NDBC buoy (0.02± 0.01°C yr^{-1}) is substantially weaker than those observed in the upper lakes. The two ECCC buoys have significantly shorter periods of reporting, both starting in 1994. The average temperature over the last 10 years estimated by the LLTM is approximately 1°C higher than the baseline period average. The average date of the lake reaching T_{MD} over the last 10 years is 20 days earlier.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)

10-Year Trend: Unassessed

30-Year Trend: Increasing

Long-term Trend (1980-2020): Increasing

Rationale: Of the three buoys on Lake Ontario, two displayed weak positive trends in summer surface water temperature; at a third buoy, the uncertainty in the estimate of the slope exceeded the best estimate of the slope. The time series at all three buoys are significantly shorter than those available for other lakes, so a long-term trend cannot be assessed, and the 30-year trend carries less weight than it does on other lakes. A very low average temperature reported in 1992 at 45135 appears to be due to a large wind-driven upwelling event, rather than being reflective of seasonally cool surface waters. The average temperature over the last 10 years, estimated by the numerical model, is approximately 1°C higher than the baseline period average. The average date of the lake reaching T_{MD} over the last 10 years is 13 days earlier than during the baseline period.

Trend Assessment Definitions

Increasing: A site was classified as "Increasing" if the best estimate of the trend of temperature as a function of time as determined by linear regression is positive and exceeds the standard error of the trend estimate.

Undetermined: A site was classified as "Undetermined" if the best estimate of the trend of temperature as a function of time as determined by linear regression did not exceed the standard error of the trend estimate.

Decreasing: A site was classified as "Decreasing" if the best estimate of the trend of temperature as a function of time as determined by linear regression is negative and exceeds the standard error of the trend estimate.

Sub-Indicator Purpose

The purpose of this sub-indicator is to assess trends in surface water temperature for each of the five Great Lakes by measuring changes in water temperature using long-term data and to infer the impact of climate change on the Great Lakes Region. This sub-indicator measures the thermal properties of the Great Lakes that affect the ecosystems' function and influence water evaporation from the lakes that affect the lake's water level (if higher surface water temperatures persists, this may potentially lead to reduced winter ice cover and increased water evaporation from the lakes resulting in lower water levels).

Ecosystem Objective

There should be no change in temperature that would adversely affect any local or general use of the waters.

This sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement, which states that the Waters of the Great Lakes should "be free from other substances, materials, or conditions that may negatively impact the chemical, physical, or biological integrity of the Waters of the Great Lakes."

Measure

The purpose of the surface water temperature sub-indicator is to assess the long-term thermal response of the Great Lakes to changes in climate. Three sources will be used: data collected from a set of nine buoys operated by the National Data Buoy Center (hereafter NDBC, operated by the US National Oceanic and Atmospheric Administration), which goes back as far as 1979, data collected from a set of four buoys operated by Environment and Climate Change Canad The U.S. National Oceanic and Atmospheric Administration (NOAA) National Data Buoy Center (NDBC) and the Canadian Environment and Climate Change Canada (ECCC) operate surface buoys in the spring, summer, and fall in all five Great Lakes. Among other fields, the buoys measure near-surface (within the upper meter) water temperature. Most deployments have water temperature available on an hourly basis; starting around 2016, some NDBC buoys started sampling at 10-minute intervals. For each year, if more than 90% of the expected data points are present, a mean of summer water temperature (July, August, September; JAS) inclusive is taken to provide a yearly value. Most of these buoys are far enough offshore that they are not affected by coastal processes such as upwelling, which could result in anomalously low estimates of surface water temperature, although there is evidence of this at some of the coastal sites. Simple linear regression was used to assess the trends, and the standard errors of the trends were used to make the determination as to the nature of the trend.

While we present the results in terms of anomalies from the mean, we subtract the mean of all of the data, rather than attempting to use a fixed baseline period due to the relatively short time period available for the buoy data.

Output from the Large Lake Thermodynamic Model (LLTM; Croley 1989a,b) is operated by the Great Lakes Environmental Research Laboratory (GLERL) and is used to estimate trends over a longer time interval, as output from this model goes back to 1950. The LLTM is a basin-averaged model- i.e., it models the lake in one dimension and does not account for lateral variability. The LLTM estimates the lake response to available meteorological data around and over the lakes, which is available over a much longer timer period than is water temperature. For the numerical model output, 1960-1990 will be used as a baseline period in order to display results in terms of anomaly. We do not intend to suggest that this baseline period was "normal" in any sense of the word. As an additional metric, the last day in the spring when the surface water temperature is below 4 °C is used. On this date, the heat content of the entire water column is highly constrained and is a robust measure of the thermal condition of the lake. In deep lakes like the Laurentian Great Lakes, this corresponds to the onset of summer stratification; in shallower lakes this may not be the case. This is used as supporting information rather than for the determination of trends. In general, the date of onset of stratification plays a significant role in mean and maximum temperatures in the subsequent summer (Austin and Colman 2007).

Ecological Condition

Overall, the Great Lakes surface water temperature seems to be increasing between 1980 and 2020, at rates that are geographically different. For example, as shown in Table 1 and Figures 1-7, the warming rates for Lake Superior is ~0.06 °C yr^{-1}, which is ~2 times more than Lake Erie with an average rate of ~0.03 °C yr^{-1}; suggesting that Lake Superior's mean surface water temperature on average is getting warmer ~2 times faster than Lake Erie. Meanwhile, lakes are experiencing earlier onset of stratification (T_{MD} = 3.98°C) over the last 10 years ranging from 39 days for Lake Michigan to 13 days for Lake Ontario: indicating earlier adverse or favorable environmental conditions for a variety of physical, biochemical, and biogeochemical processes. For example, a likely temporal shift in the development of seasonal stratification with the ability to limit the vertical transport of dissolved oxygen and nutrients, a possible temporal shift in optimum temperature for seasonal algal blooms, and the fact that the dissolved oxygen concentrations in water decreases with temperature – i.e., effects of changes in water temperature on oxygen solubility (Wetzel 2001; Chapra 2008).

Surface water temperature is directly dependent on regional air temperatures and hence regional climate. Upward trends in surface water temperatures have been documented on the Laurentian Great Lakes (e.g., Austin and Colman 2007, Huang et al. 2012) as well as on lakes around the world (O'Reilly et al. 2015). Water temperature is a primary ecosystem driver, affecting a wide range of processes, including nutrient uptake, metabolism rates, and defines fish habitat. Surface heat and moisture fluxes (evaporation) are also a strong function of surface water temperature. Summer surface water temperatures are a reflection of not only summer air temperatures but ice conditions the previous winter (Austin and Colman 2007). In addition, the onset of summer stratification provides a robust, integrated measure of winter conditions, in which higher-ice winters tend to result in a later onset of stratification and low-ice winters result in earlier onset of stratification. In lakes without significant ice formation (e.g., Michigan, Ontario), the onset of stratification is a reflection more of the winter thermal storage of the lake, again with colder years resulting in a later onset of stratification. The date of the onset of stratification is a strong predictor of the summer surface water temperature, and the results in this report are consistent with each other: the date of the onset of stratification is getting earlier, and summer surface water temperatures are increasing.

There is a great deal of natural inter-annual variability superimposed on top of the warming trend, resulting in relatively low values of the correlation coefficients for most linear fits. Several features are consistent across the lakes. First and perhaps most importantly, a significant jump occurs between 1997 and 1998, a strong El Niño year. It has been pointed out (van Cleave et al. 2014) that taken separately, summer water temperature prior to 1998 and from 1998 to the present have no significant trends, but a strong discontinuity between the average water temperature between these two time periods. The offset between these two time periods for the upper lakes is on the order of 2 °C.

In the absence of historical field observations, we rely on LLTM simulated lake water temperature since 1950. It is worth noting that while the LLTM results suggest that the past decade has been anomalously warm, the same could be said for the decades *preceding* the baseline period as well. In fact, it appears that for the span of available model output (i.e., 1950, which is in turn limited by the availability of meteorological data with which to force the model) appears to be an anomalously cold period, with lower average temperatures and later stratification onset. This may somewhat bias the anomalies for recent years towards higher values.

Linkages

There is a clear link between the onset of summer stratification and average summer water temperatures. Further, the onset of summer stratification is closely tied to ice cover in lakes that form ice cover (Austin and Colman 2007). Taking this a step further, recent work (Titze, 2016; Anderson et al., 2021) has shown a strong but unsurprising link between average winter air temperatures and the amount of ice cover, suggesting a series of causal linkages (winter air temperatures → winter ice cover/heat content → onset of stratification → summer water temperatures) which may prove to be a useful predictive tool for resource managers. It should be noted that one of the characteristics of the emerging climate change trend is for winter air temperatures to increase faster than the annual average. Due to the connectivity noted, this suggests that winter climate trends are going to impact not only winter conditions such as ice cover, but summer water temperatures as well, as measured in this report. The impact of lake level on water temperature, or the reverse, is not clear.

Trends towards earlier stratification onset (and later breakdown) imply that the period of stratification is increasing. Separate research (Austin and Colman 2008) suggests that over the period 1906-2006, the length of the period of summer stratification has increased from roughly 145 days to 170 days, an increase of about 20%. This is going to have significant implications for primary productivity in the lakes, as well as oxygen depletion in shallower, more productive parts of the Great Lakes.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: www.ndbc.noaa.gov www.meds-sdmm.dfo-mpo.gc.ca		

Data Limitations

Due to the relatively short time frame in which data on surface water temperatures exist, it is difficult to attribute climatic patterns to either natural cycles or anthropogenic activity. The numerical model output used in this report is a one-dimensional model and does not fully take into account the three-dimensional nature of the lakes. The numerical model is limited by the availability and quality of meteorological forcing data. A comparison of the modeled and observed average summer surface water temperatures (Figure 6) shows that while the model captures the gross structure of interannual variability at most sites, there can be, in some instances, significant bias between the two. This may be because the model is attempting to provide a lake-wide average temperature, whereas the buoy observations are made at distinct points within these large, spatially heterogeneous systems.

The field season was considerably foreshortened in 2020 due to COVID-related restrictions on the use of research vessels, so that reliable buoy-based data was limited.

Additional Information

Subsurface temperature data is not available on a long-term basis necessary for determining lake heat content or trends therein. However, due to an unusual thermodynamic property of freshwater, the heat content can be determined using just a surface temperature in one specific circumstance. Specifically, when the surface water temperature reaches its temperature of maximum density (3.98°C) in the spring (or early summer), the entire water column must also be at the same temperature. Subsequent to this, lakes tend to form stratification in which a layer of warm water sits on top of cooler water below; hence, this date is often referred to as the onset of spring stratification. While this only gives us a glimpse of the heat content, the date can be used at which this event happens as a stable proxy for inter-annual variability in heat content. In warm years, this event will occur early, and in cold years it will be delayed. All lakes are experiencing earlier onset of stratification relative to the baseline period of 1960-1990. The associated lengthening of stratification periods that accompany the warming of water temperatures may result in an increased period of oxygen depletion in the deep waters of some of the Great Lakes, such as Lake Erie.

While there are some groups that periodically deploy equipment over the winter, there are no structures in place to guarantee funding for systematic, year-round measurements of temperature in the Great Lakes during the winter months. The dearth of data, and in fact knowledge of these systems in the winter has been the focus of recent work (Ozersky et al. 2021). As these systems have strong seasonal connectivity, developing a long-term program for year-round measurements should be a priority. Likewise, there are very few long-term measurements of thermal structure (temperature throughout the water column) so little if anything is known about trends in features like thermocline depth.

Acknowledgments

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Figure 5. Date of reaching the temperature of maximum density (4°C) from LLTM model output, 1950 to 2020. Shaded region represents baseline period.

Source: Large Lake Thermodynamic Model.

Figure 6. Model output- buoy data comparison, 1981-2020. WT = Water Temperature.

Source: data: NDBC and ECCC. Model: Large Lake Thermodynamic Model.

Figure 7. Location of surface buoys used in this report.

Source: National Data Buoy Center (NDBC) and Environment and Climate Change Canada (ECCC).

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Table 1. Great Lakes buoys operated by NOAA and ECCC with a summary of surface water temperature trends.

Designation	Location	Trend ± 1 S.E., °C yr^{-1}	Correlation coeff. r	Start of years operational	number of years operational	Source
45006	W. Superior	0.05 ± 0.03	0.30	1981	37	NDBC
45001	C. Superior	0.07 ± 0.04	0.28	1979	37	NDBC
45004	E. Superior	0.07 ± 0.04	0.25	1980	34	NDBC
45002	N. Michigan	0.06 ± 0.02	0.41	1980	37	NDBC
45007	S. Michigan	0.05 ± 0.02	0.39	1981	35	NDBC
45003	N. Huron	0.06 ± 0.02	0.41	1980	37	NDBC
45008	S. Huron	0.06 ± 0.02	0.45	1982	35	NDBC
45005	W. Erie	0.02 ± 0.01	0.25	1980	36	NDBC
45132	C. Erie	0.05 ± 0.02	0.43	1994	22	ECCC
45142	E. Erie	0.04 ± 0.02	0.38	1994	22	ECCC
45159	W. Ontario	(-0.04 ± 0.10)	-0.14	2002	9	ECCC
45012	E. Ontario	0.05 ± 0.04	0.28	2002	18	NDBC
45135	E. Ontario	0.09 ± 0.03	0.64	1990	19	ECCC

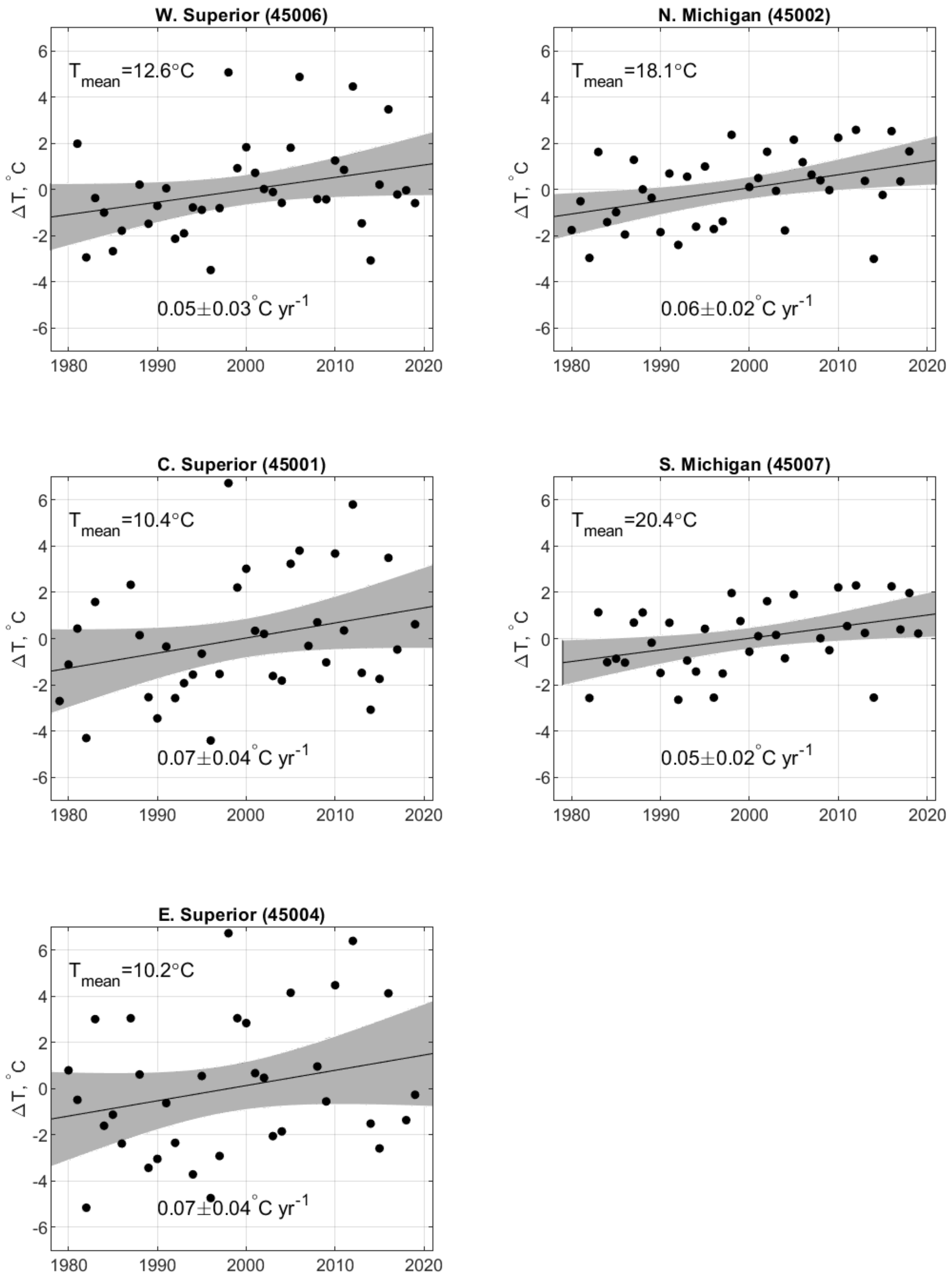

Figure 1. Summer surface water temperature anomalies at buoys in Lake Superior (first column) and Lake Michigan (second column). Solid line is the best estimate of linear trend; shaded area is region of one standard error for fit. Data from NOAA NDBC.

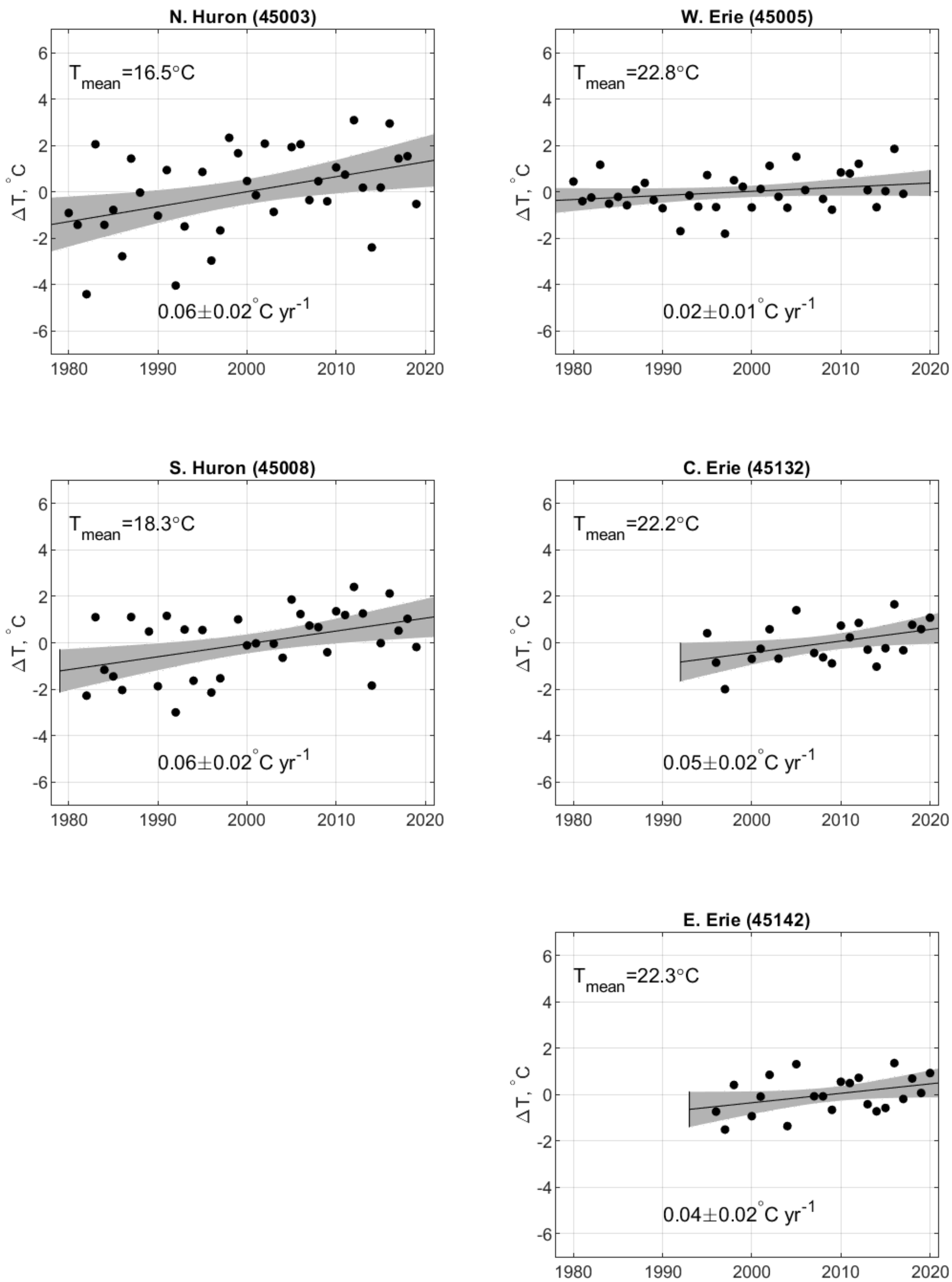

Figure 2. Summer surface water temperature anomalies at buoys in Lake Huron (first column) and Lake Erie (second column). Solid line is the best estimate of linear trend; shaded area is region of one standard error for fit. Data from NDBC and ECCC.

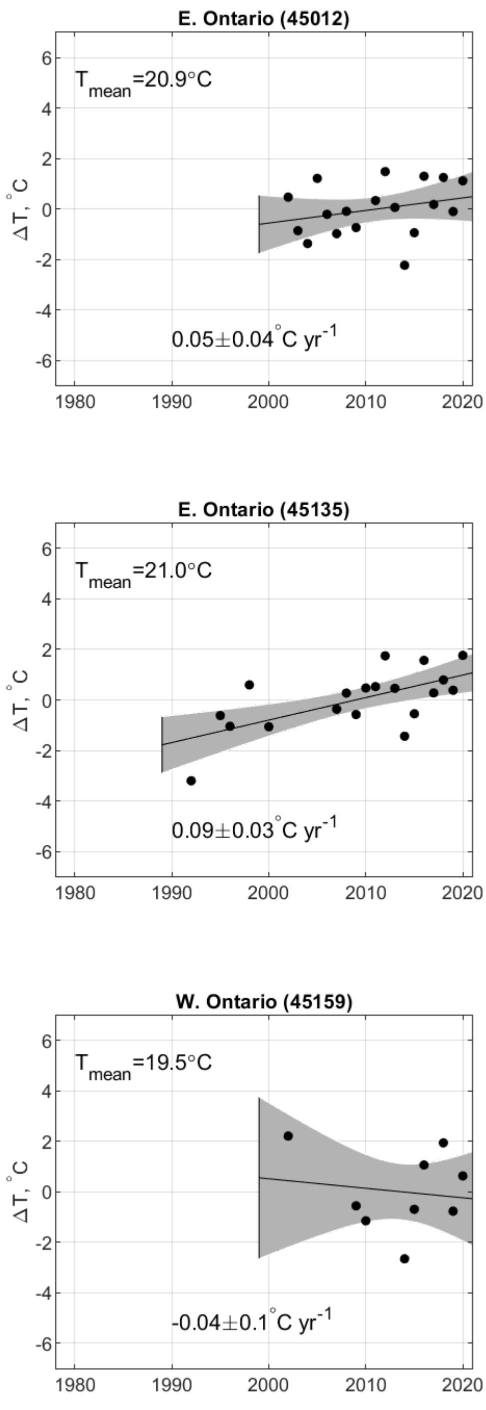

Figure 3. Summer surface water temperature anomalies at buoys in Lake Ontario. Solid line is the best estimate of linear trend; shaded area is region of one standard error for fit. Data from NDBC and ECCC.

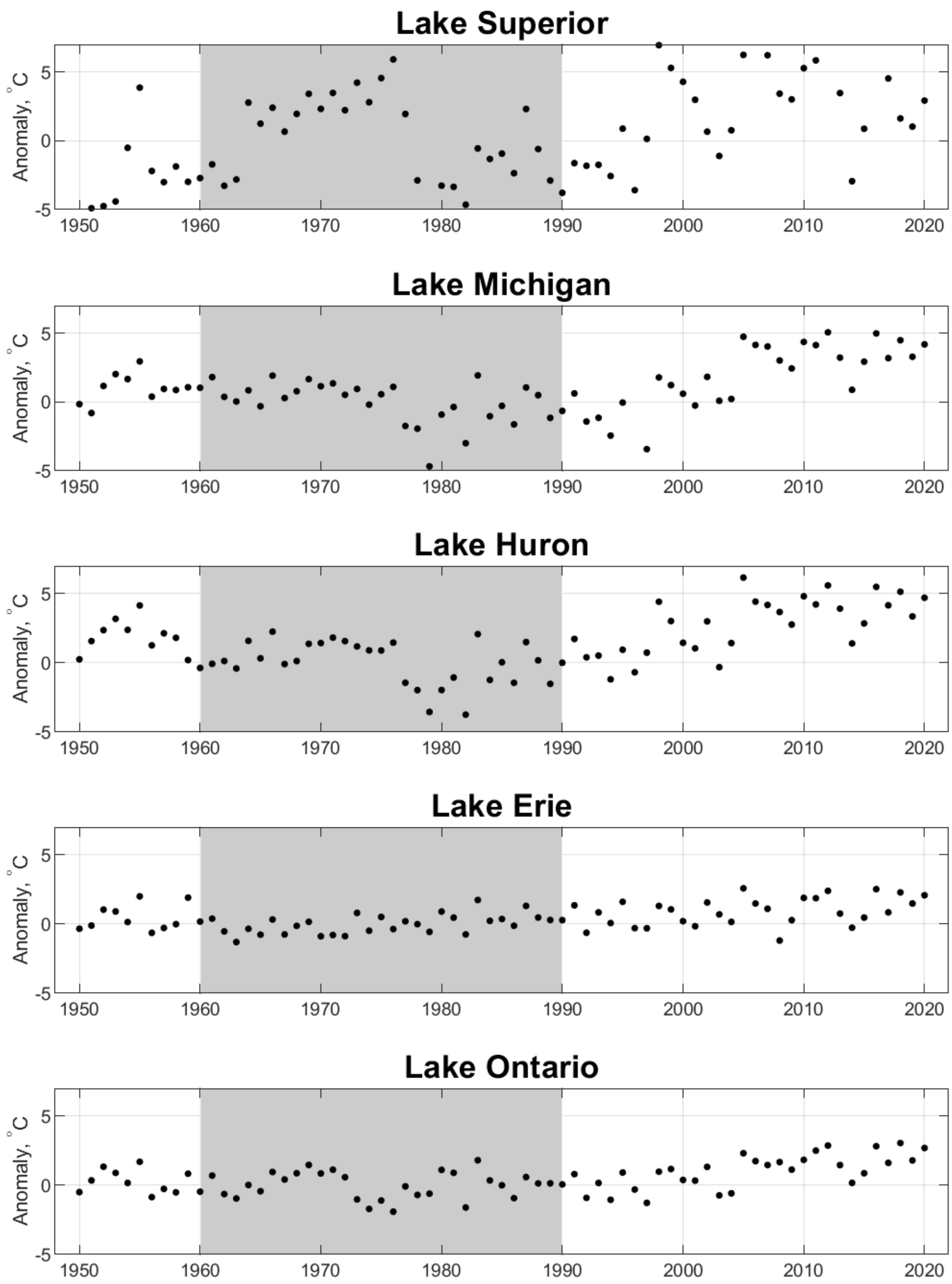

Figure 4. Mean summer (JAS) surface water temperature anomalies from LLTM output, 1950 to 2020. Shaded region represents baseline period. Source: Large Lake Thermodynamic Model.

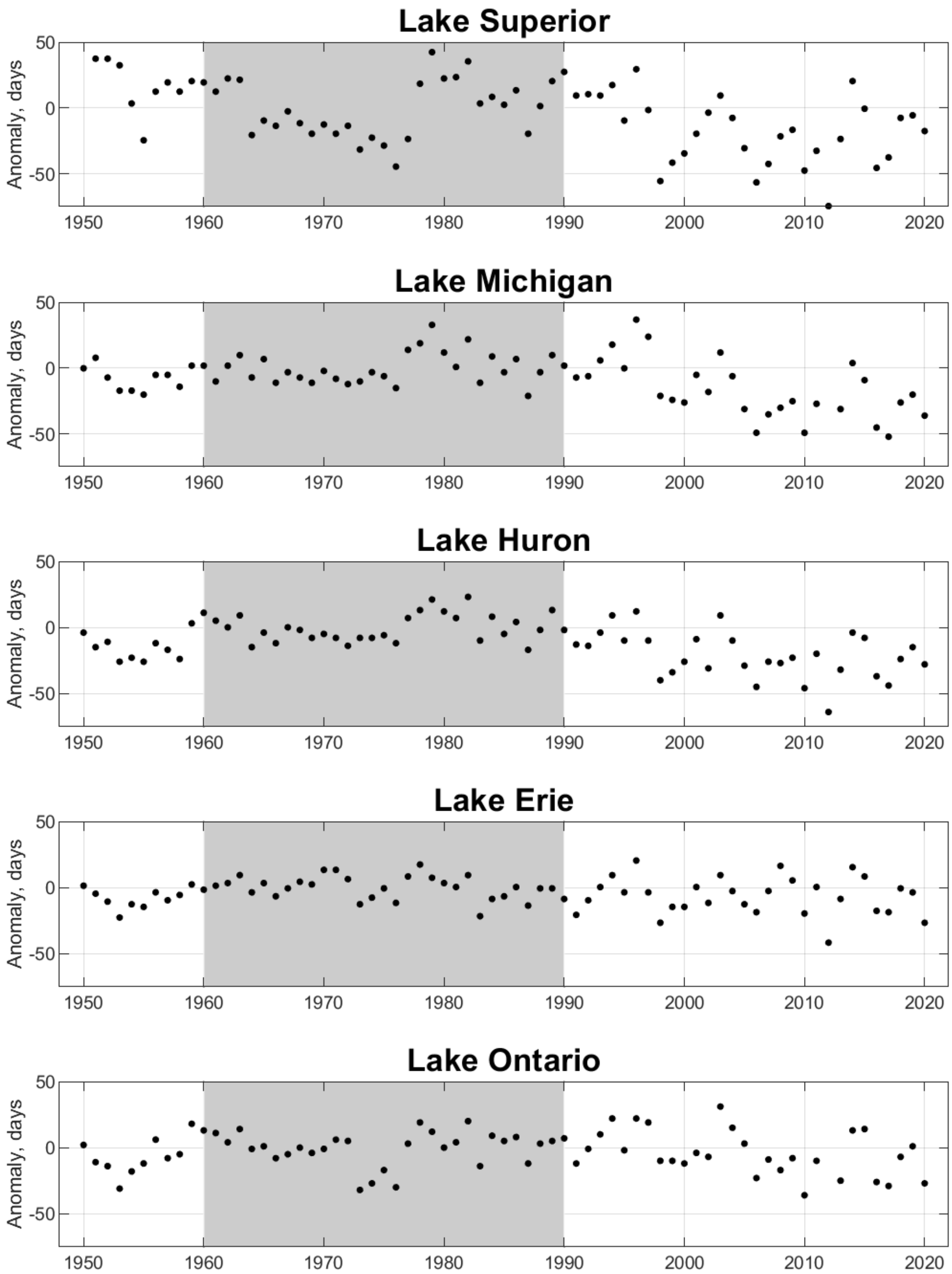

Figure 5. Date of reaching the temperature of maximum density (4C) from LLTM model output, 1950 to 2020. Shaded region represents baseline period. Source: Large Lake Thermodynamic Model.

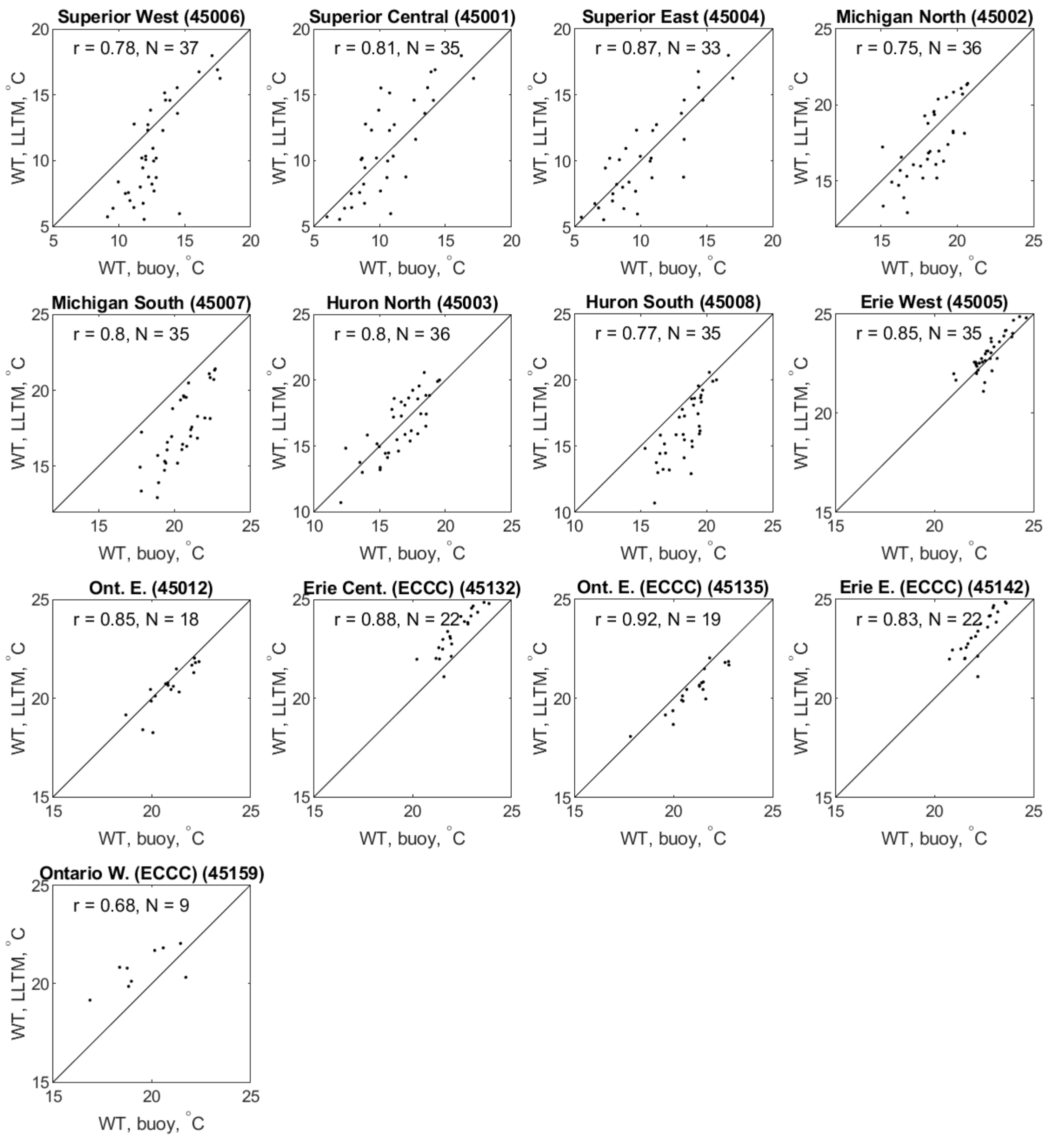

Figure 6. Model output- buoy data comparison, 1981-2020. WT = Water Temperature. Source: data: NDBC and ECCC. Model: LLTM.

Figure 7. Location of surface buoys used in this report. Source: National Data Buoy Center (NDBC) and Environment and Climate Change Canada (ECCC)

Sub-Indicator: Ice Cover

Overall Assessment

Trends:

10-Year Trend (2011-2020): Increasing

30-Year Trend (1991-2020): Decreasing

Long-term Trend (1973-2020): Decreasing

Rationale: The annual maximum ice cover (AMIC) anomaly (derived from the 1973 to 1990 base period average) for the period of 1973 to 2020 demonstrates a statistically significant (at the 0.05 level) trend of -0.46% annually. This implies a basin-wide ice loss of 22.1% over the period of 1973 to 2020. The 10-year and 30-year trends in maximum ice cover anomalies are 0.63% and -0.04% respectively (Figure 2). As shown below, in addition to overall large, natural interannual variability, the long-term trend is declining quite significantly.

Trend assessment definitions are included following the Lake-by-Lake Assessment section.

Note that the trends calculated within a specific period of time such as 1973-2020 can only be applicable to the same period, and cannot be extrapolated to the future and back to the past. It should not be interpolated to a period shorter than the time series of the data from which the trends are derived, since there are decadal and multi-decadal changes in lake ice cover (Wang et al. 2012b, 2018) and longer time scale of ~50 years (Warner et al. 2021).

So, it is important to note that annual fluctuations can affect trends – especially short-term trends. A 10-year period is relatively short when the climate trend is calculated. For example, recent heavy ice seasons in 2013/2014, 2014/2015, and 2018/2019 have opposed the trend of decreasing ice cover anomalies, which have reduced the statistical significance of the trend in the dataset.

Lake-by-Lake Assessment

Lake Superior

10-Year Trend: Increasing

30-Year Trend: Decreasing

Long-term Trend (1973-2020): Decreasing

Rationale: The long-term trend for the ice cover anomaly is decreasing at an annual rate of -0.74%, translating into a lake-wide decrease in maximum ice cover of 35.5% from 1973 to 2020. Lake Superior has experienced the highest diminishment of maximum lake ice coverage over the historical period of all the Great Lakes. The trend for the 10-year period is increasing (0.18%) and for the 30-year timeframe is decreasing (-0.42%) (Figure 3).

Note that the long-term record for the whole basin is dominated by Lake Superior, as it is the largest lake and has the strongest long-term downward trend.

Lake Michigan

10-Year Trend: Increasing

30-Year Trend: Increasing

Long-term Trend (1973-2020): Decreasing

Rationale: A similar long-term pattern in the anomaly analysis emerges for Lake Michigan as the trend in maximum ice cover is decreasing at a rate of -0.33% per year for the 1973 to 2020 period. Statistically significant trends of 0.07% for the 10-year period and 0.07% for the 30-year timeframe of maximum ice cover anomalies were noted (Figure 4).

Lake Huron

10-Year Trend: Increasing

30-Year Trend: Decreasing

Long-term Trend (1973-2020): Decreasing

Rationale: Lake Huron exhibits a decreasing trend (-0.39%) in the maximum lake ice coverage anomaly calculated for all the specified long-term period of 1973 to 2020. Maximum ice cover anomalies trend negatively on an annual basis across the 30-year (-0.08%) period of examination as well, but trend positively in the most recent decade (1.49%) (Figure 5).

Lake Erie

10-Year Trend: Decreasing

30-Year Trend: Decreasing

Long-term Trend (1973-2020): Decreasing

Rationale: A statistically significant annual decrease in the maximum ice cover anomaly of -0.50% was noted for the long-term timeframe. Shorter period indicators for Lake Erie were also deemed to be statistically significant, with a trend of -0.63% over the last 30 years and -0.41% for the last 10 years (Figure 6).

Note that as the southernmost and shallowest lake, Erie is the most sensitive and varies from 100% ice-covered to 0% in any given year

Lake Ontario

10-Year Trend: Increasing

30-Year Trend: Increasing

Long-term Trend (1973-2020): Decreasing

Rationale: Lake Ontario maximum ice cover anomalies show a decreasing (-0.24% per year) trend for the long-term period. Statistically significant 10-year and 30-year trends in maximum ice cover anomaly tendencies show an increasing (0.06%) trend for the 10-year period and an increase (0.04%) in the 30-year timeframe (Figure 7).

Status Assessment Definitions

Climate information in the State of the Great Lakes indicator suite is not assessed in the same manner as other sub-indicators. For example, the ecosystem has adapted to and needs both high and low water levels and neither condition can be assessed as “Good” or “Poor”. However, prolonged periods of high or low water levels may cause stress to the ecosystem. Therefore, climate trends for the Precipitation Amounts in the Great Lakes, Surface Water Temperatures, Ice Cover and Water Levels reports are assessed as "Increasing", "Unchanging", or "Decreasing" over a defined period of time.

Trend Assessment Definitions

Increasing: Increasing AMIC anomaly (and/or average ice concentrations) over the period of record (1973-2018).

Unchanging: No change in AMIC anomaly (and/or average ice concentrations) over the short and/or long term of the reporting cycle.

Decreasing: Decreasing AMIC anomaly (and/or average ice concentrations) over the period of record (1973-2018).

Undetermined: Data are not available to report on a trend.

Endpoints and/or Targets

No endpoint is needed for the climate-based sub-indicator reports. The ecosystem adapts to periods of high and low ice coverage, however, a prolonged period of either could pose problems.

Sub-Indicator Purpose

The overall purpose of this sub-indicator is to assess winter ice cover. The sub-indicator will also help to assess impacts on seasonal and interannual lake temperature (Anderson et al. 2021) and accompanying physical changes to each lake over time by measuring the thermal properties of the Great Lakes. Thermal properties affect the ecosystems' function (Ozersky et al. 2021) and influence water evaporation and lake effect snow (Wright et al. 2013; Cronewold et al. 2015; Fujisaki-Manome et al. 2017) from the lakes that affects water levels. This sub-indicator tracks the extent of winter ice cover for each of the five Great Lakes by measuring spatial extent of water temperature and ice cover using long-term data, which can be compared with future claimate change in the Great Lakes region (ECCC 2021; Wang, X. et al. 2017). This sub-indicator is also used to infer potential impact of climate change on wetlands since ice cover affects water levels and protects the shorelines, including wetlands (Dehghan 2019), from erosion by waves and winter storms. In addition, the role of wind in ice formation and stability is an important factor on landfast ice. Wind mixing and upwelling, as well as driving intermittent breakup of shoreline-attached ice, keep Lake Michigan and Lake Ontario from freezing in most years due to lake orientation and depth.

Ecosystem Objective

Change in lake ice cover during the winter due to climate change will affect water temperature and biomass abundance (Ozersky et al. 2021) on the Great Lakes in the following spring and summer and, in turn, affect lake ecosystems. Awareness of occurrence will encourage human response to reduce the stressor towards minimizing biological disruption.

This sub-indicator best supports work towards General Objective #9 of the 2012 Great Lakes Water Quality Agreement which states that the Waters of the Great Lakes should "be free from other substances, materials, or conditions that may negatively impact the chemical, physical, or biological integrity of the Waters of the Great Lakes."

Measure

This sub-indicator will measure annual maximum and average ice concentrations on each of the Great Lakes. The data are collected by National Oceanic and Atmospheric Administration (NOAA) and the Canadian Ice Service (CIS) of Environment and Climate Change Canada from 1973 to 2020 for the purposes of this report.

The satellite measurement of lake ice started in 1979. Quality control was applied to the dataset to remove the outliers (usually greater than 2 standard deviations). Then spatial ice cover maps and ice cover time series can be obtained (Wang et al. 2012b). From 1973 to 2020, ice cover of each lake and total ice cover for all five Great Lakes can be calculated. Furthermore, the decadal average can be constructed for reporting purposes.

According to Assel (2005), the daily spatial average ice cover for each Great Lake is calculated from daily grids. Daily grids are generated by linear interpolation of observed ice cover grids between adjacent dates for a given winter season from the date of the first ice chart to date of the last ice chart (Assel and Norton, 2001). Lake-averaged ice cover prior to date of first ice chart and after date of last ice chart is assumed to be zero. The daily lake-averaged ice cover on each Great Lake is used to calculate the seasonal average ice cover. The seasonal average ice cover is the sum of the daily lake-averaged ice cover over a winter divided by 182 (the number of days between 1 December to the following 31 May). The seasonal average ice cover is calculated using days when the lake-averaged ice cover was greater than or equal to 5%.

The seasonal average ice cover is an index of the severity of an annual ice cycle. Ancillary ice cycle variables calculated for each winter are the Julian dates that the first and last observed lake-averaged ice cover were greater than or equal to 5% and the duration of the ice cover, that is, the difference between dates of last and first ice.

Annual maximum ice cover (AMIC) is defined as a maximum percentage of ice cover in one day during an ice season (winter). This snapshot of the ice season is a realization that can be measured, and reflects the overall atmospheric cumulative effects on lake ice. Furthermore, its seasonal and interannual variability can be accurately recorded and analyzed (Bai et al. 2012). The trend of AMIC for a specified period can be calculated (Wang et al. 2012a,b; 2017a,b). However, the trend varies with different length of the time series included, because there is multi-decadal variability in lake ice caused by multi-decadal atmospheric and water thermal forcings (Wang et al. 2018). The AMIC anomaly is defined as the difference between the daily AMIC and the climatological mean of daily AMIC over the period of 1973-1990.

An anomaly-based approach to reporting on the climate trend sub-indicators, which include water levels, precipitation amounts, ice cover and surface water temperatures, has been implemented since the 2019 State of the Great Lakes reporting cycle.

The approach includes identifying a defined base period and comparing the most recent 10-year, 30-year and long-term trends (the full period of record) to this base period to identify "anomalies" or "deviations" from the base period. The base period includes up to 1990 but not beyond as it is expected that accelerated “climate change” effects would be occurring after 1990. The base period varies in length for these 4 reports, but includes a minimum of 15 years of data that are reliable and comprehensive to calculate the base values. An anomaly-based approach puts the longer periods of record into context and compares the historical data to more recent changes that are occurring in the ecosystem. This approach provides more telling information as it identifies whether the deviations or extremes are increasing or decreasing over the various time periods which is valuable in reporting on recent climate changes and impacts in the Great Lakes. Previously, the increasing and/or decreasing trends provided in the State of the Great Lakes climate trend reports looked at the historical period of record and whether long-term trends were increasing or decreasing over that time.

A moving average (length determined by the authors) for the climate trend sub-indicators is used because these measures of precipitation, ice cover, water levels and surface water temperatures, in most cases, are non-linear (e.g., there is no inherent link in precipitation amounts from one year to the next). The moving average smooths the variation in the year-to-year fluctuations to help show the trends in the data over the longer period (Wang et al. 2018).

Ecological Conditions

This sub-indicator is used as a potential assessment of climate change, particularly within the Great Lakes basin. Changes in water and air temperatures will influence ice development on the lakes and, in turn, affect coastal wetlands, nearshore aquatic environments, and inland environments. Ice cover directly influences lake water temperature change, duration of stratification, and fish behavior. Based on the observations (Figure 1), the highest maximum ice cover for each Great Lake occurred in 1977-1979, 1994, 2014, and 2015. For Lakes Michigan, Erie and Ontario, the highest maximum ice over took place in 1977, 1978 and 1979 (Figures 4, 6 and 7), respectively, depending on the spatial variability of air temperature and water depth (i.e., heat content) of each lake.

There is spatial variability in the AMIC trend. The steepest trends occur along heavily ice-covered coasts in Lake Superior, Georgian Bay, northern Lake Huron, and northern Lake Michigan. Offshore ice cover has more gradual trends, as ice cover is not continuous. Research is needed to map the grid points of a lake using GIS to calculate each grid point trend. This on-going research will reveal spatial trend distributions over the Great Lakes (Mason et al. 2016).

During the 2015/16 and 2016/17 winters, AMIC was observed to be 34% and 19%, respectively, significantly below the long-term average of 55% (Figure 1), mainly due to the simultaneous occurrence of a strong El Niño, positive North Atlantic Oscillation (NAO), warm phase of Atlantic Multi-decadal Oscillation (AMO), and warm phase of Pacific Decadal Oscillation (PDO).

Air temperatures over a lake are among the few factors that control the formation of ice on that surface. Colder winter temperatures increase the rate of heat release by the lake, thereby increasing the freezing rate of the water. Milder winter temperatures have a similar controlling effect--only the rate of heat released is slowed and the ice forms more slowly. Globally, some inland lakes appear to be freezing up at later dates, and breaking-up earlier, than the historical average, based on a study of 150 years of data (Magnuson et al. 2000). These trends add to the evidence that the earth has been in a period of global warming for at least the last 150 years.

Linkages

This sub-indicator links directly to the other sub-indicators in the Watershed Impacts and Climate Trends indicator. It is indirectly linked to other sub-indicators that track trends in wetland area, habitat change and fish species.

The freezing and thawing of lakes is a very important aspect to many aquatic and terrestrial ecosystems. Many fish species rely on the ice cover or ice duration to give their eggs protection against strong wind stirring during the late part of the ice season. Nearshore ice has the ability to change the shoreline as it can encroach upon the land during winter freeze-up times. Even inland systems are affected by the amount of ice that forms, especially within the Great Lakes basin. Less ice on the Great Lakes allows for more water to evaporate and be spread across the basin in the form of snow. This can have an effect on the foraging animals (such as deer) that need to dig through snow during the winter in order to obtain food.

Under the business as usual emission scenario (IPCC RCP 8.5), annual average air temperature over the Canadian portion of the Great Lakes basin is projected to be about 5°C warmer by the 2080s (2070-2099) than it was in the 1990s (1986-2005) (Zhu et al 2018). These changes could affect the formation of ice on the Great Lakes and could result in changes in the ecosystem as noted below.

Under the decreasing ice conditions, however, there are some benefits such as lengthened shipping duration (Kubat et al. 2021) and less destruction of infrastructure around the Great Lakes, including offshore wind farm development. Declining ice cover may not be a universally negative phenomenon for all constituencies.

Linkages to other sub-indicators in the indicator suite include:

- Surface Water Temperature – ice cover has high correlation with water temperature (r = 0.5).
- Water Levels – ice cover can influence heat and moisture exchange between the lake and the atmosphere.
- Precipitation Amounts – ice cover can strongly influence lake-effect snow.
- Coastal Wetlands: Extent and Composition – ice cover can protect coastal wetlands and reduce erosion.
- Hardened Shorelines – less ice cover exposes the shoreline to waves generated by winter storms that accelerate erosion.
- Dreissenid Mussels – ice cover can influence the water stability, vertical temperature structure and spring warming and summer stratification.
- Harmful Algal Blooms – higher water temperatures and less ice cover may be related to more and earlier algal blooms.
- Toxic Chemicals in Herring Gull Eggs – a link has been shown between contaminant levels in Herring Gull eggs and ice cover.
- Lake white fish, lake herring, and yellow perch species are dependent on ice cover, as ice cover can protect the eggs from wind mixing.

Assessing Data Quality

Data Characteristics	Agree	Neutral or Unknown	Disagree	Not Applicable
Data are documented, validated, or quality-assured by a recognized agency or organization	X			
Data are from a known, reliable and respected generator of data and are traceable to original sources	X			
Geographic coverage and scale of data are appropriate to the Great Lakes Basin	X			
Data obtained from sources within the U.S. are comparable to those from Canada	X			
Uncertainty and variability in the data are documented and within acceptable limits for this sub-indicator report	X			
Data used in assessment are openly available and accessible	Yes	Data can be found here: https://www.glerl.noaa.gov/data/ice/		

Additional Information

Ice cover is a very understandable feature. Lake ice indicates coastal wetland ice and itself affects wetlands (e.g., winter storm severity). Less ice cover exposes the shoreline to waves generated by winter storms that accelerates erosion. Ice cover reflects temperature, wind, and heat stored in a lake, and therefore, this is a good indicator of climate effects. These data are already collected annually for each lake by NOAA and CIS using satellite imagery. There is a natural variability in maximum ice extent accounted for in the interpretation.

On-going research will improve our understanding of lake ice and related climate changes, as well as its implications for ecosystems in the Great Lakes (Bai et al. 2015; Wang et al. 2012b, 2018).

Acknowledgments

Ice dataset was obtained from NOAA Great Lakes Environmental Research Laboratory (https://www.glerl.noaa.gov/data/ice/)

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All data analyzed and charts created by the authors.

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List of Figures

Figure 1. Time series of five Great Lake AMIC for the period 1973-2020, which is based on the binational NIC and CIS dataset.

Source: NOAA Great Lakes Environmental Research Laboratory (GLERL), National Ice Service (NIC) and Canadian Ice Service (CIS)

Figure 2. Time series of five Great Lakes AMIC anomalies for the periods 1973-2020, based on the binational NIC and CIS dataset. 10-year trend (orange), 30-year trend (green), long-term trend (purple) and 9-year running mean (black) of anomalies also shown.

Source: ECCC and NOAA.

Figure 3. Time series of Lake Superior AMIC anomalies for the periods 1973-2020, based on the binational NIC and CIS dataset. 10-year trend (orange), 30-year trend (green), long-term trend (purple) and 9-year running mean (black) of anomalies also shown.

Source: ECCC and NOAA.

Figure 4. Time series of Lake Michigan AMIC anomalies for the periods 1973-2020, based on the binational NIC and CIS dataset. 10-year trend (orange), 30-year trend (green), long-term trend (purple) and 9-year running mean (black) of anomalies also shown. Source: ECCC and NOAA.

Figure 5. Time series of Lake Huron AMIC anomalies for the periods 1973-2020, based on the binational NIC and CIS dataset. 10-year trend (orange), 30-year trend (green), long-term trend (purple) and 9-year running mean (black) of anomalies also shown.

Source: ECCC and NOAA.

Figure 6. Time series of Lake Erie AMIC anomalies for the periods 1973-2020, based on the binational NIC and CIS dataset. 10-year trend (orange), 30-year trend (green), long-term trend (purple) and 9-year running mean (black) of anomalies also shown.

Source: ECCC and NOAA.

Figure 7. Time series of Lake Ontario AMIC anomalies for the periods 1973-2020, based on the binational NIC and CIS dataset. 10-year trend (orange), 30-year trend (green), long-term trend (purple) and 9-year running mean (black) of anomalies also shown.

Source: ECCC and NOAA.

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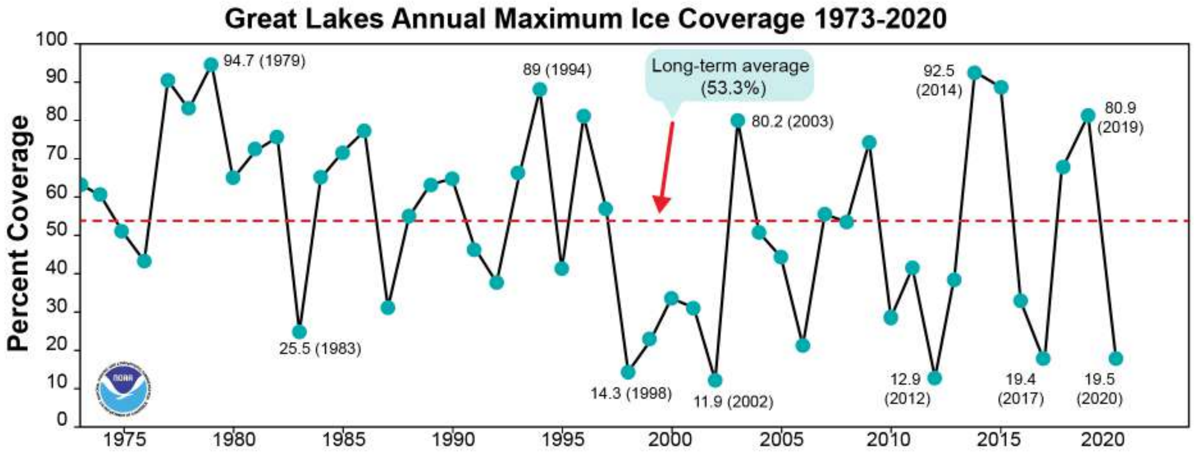

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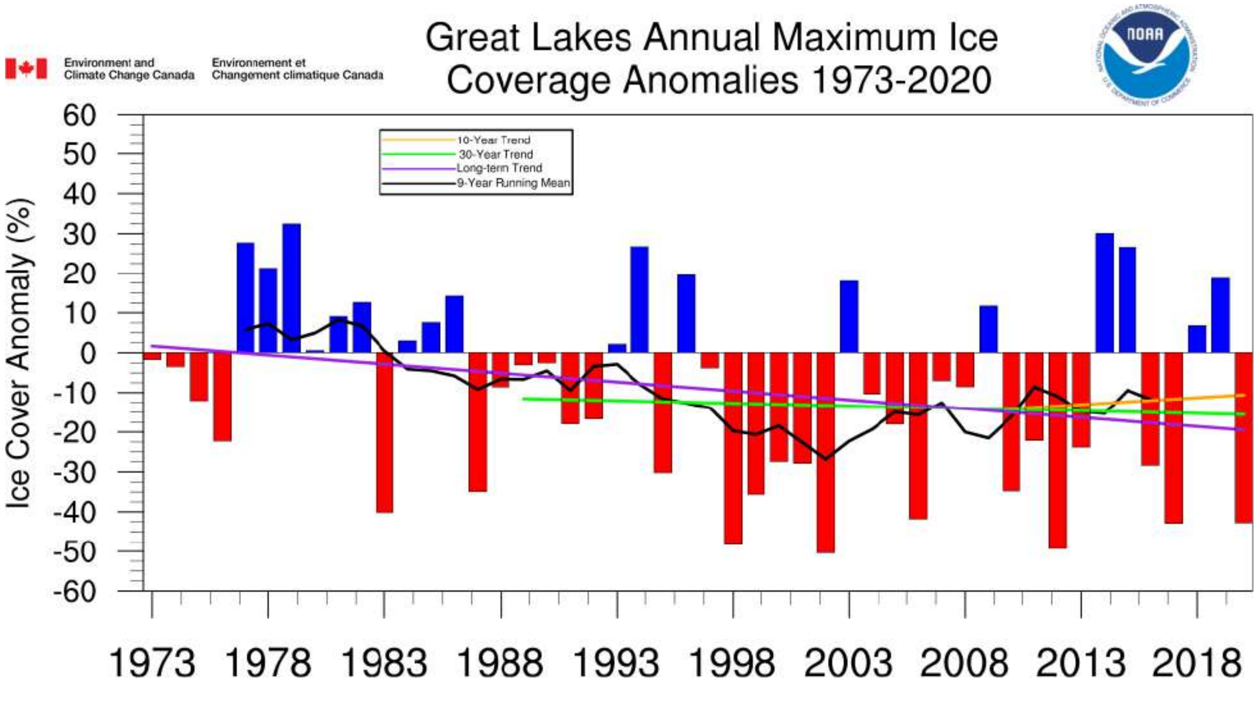

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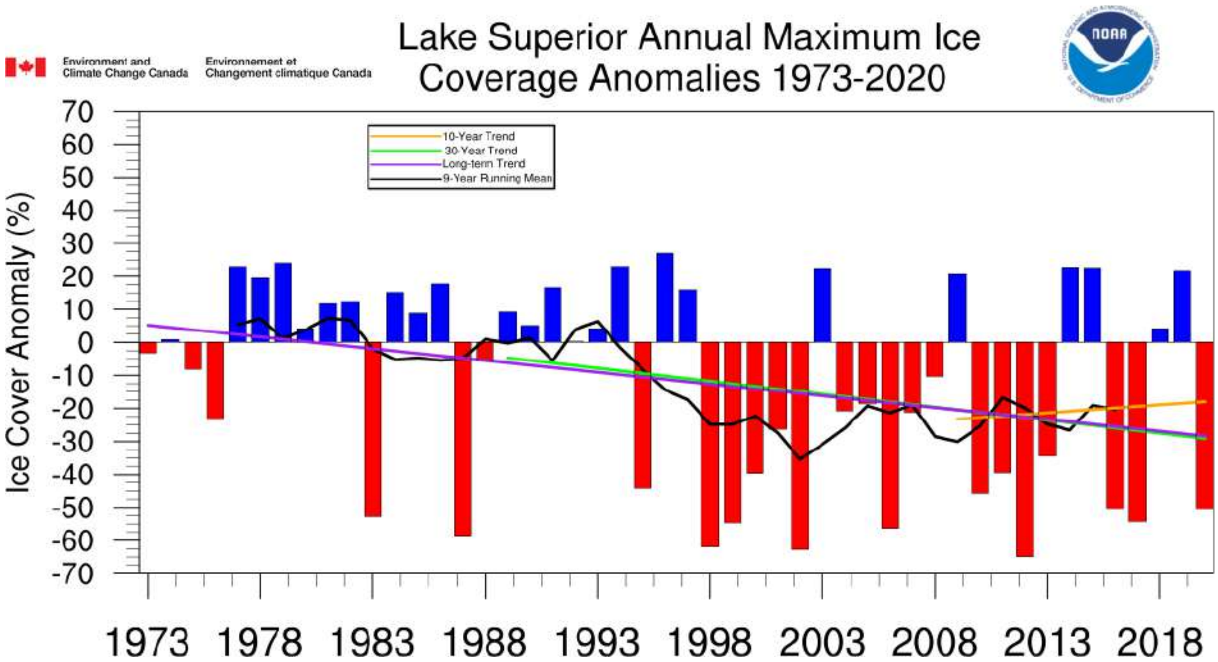

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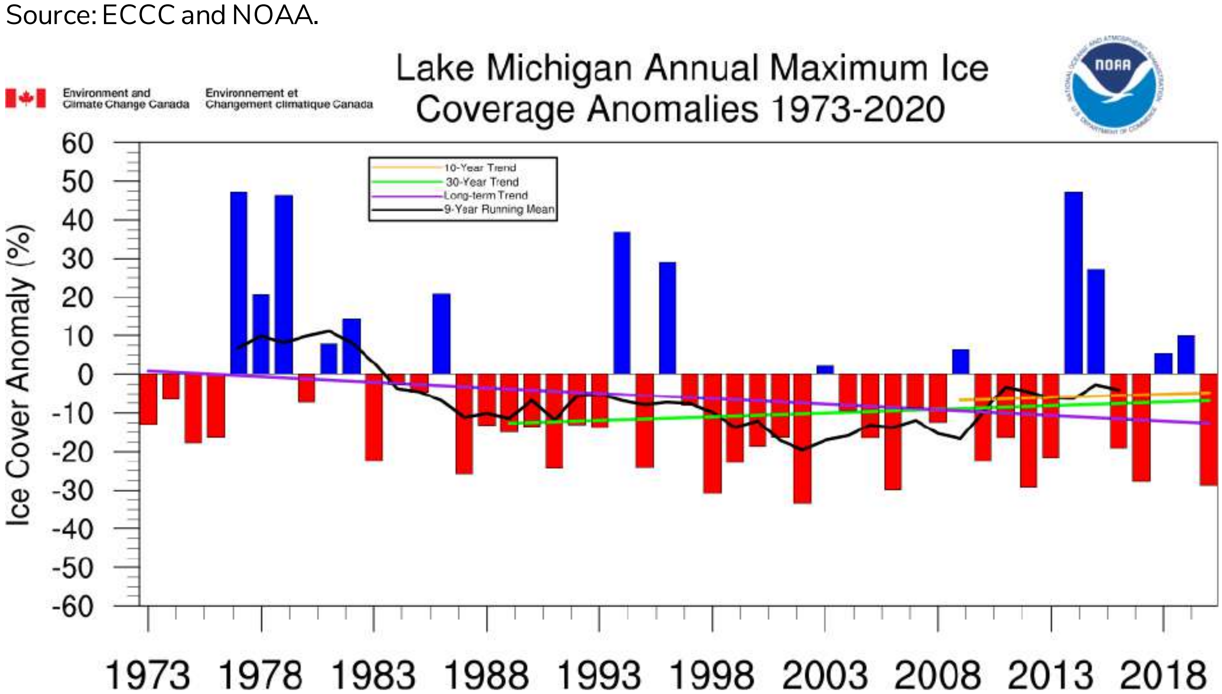

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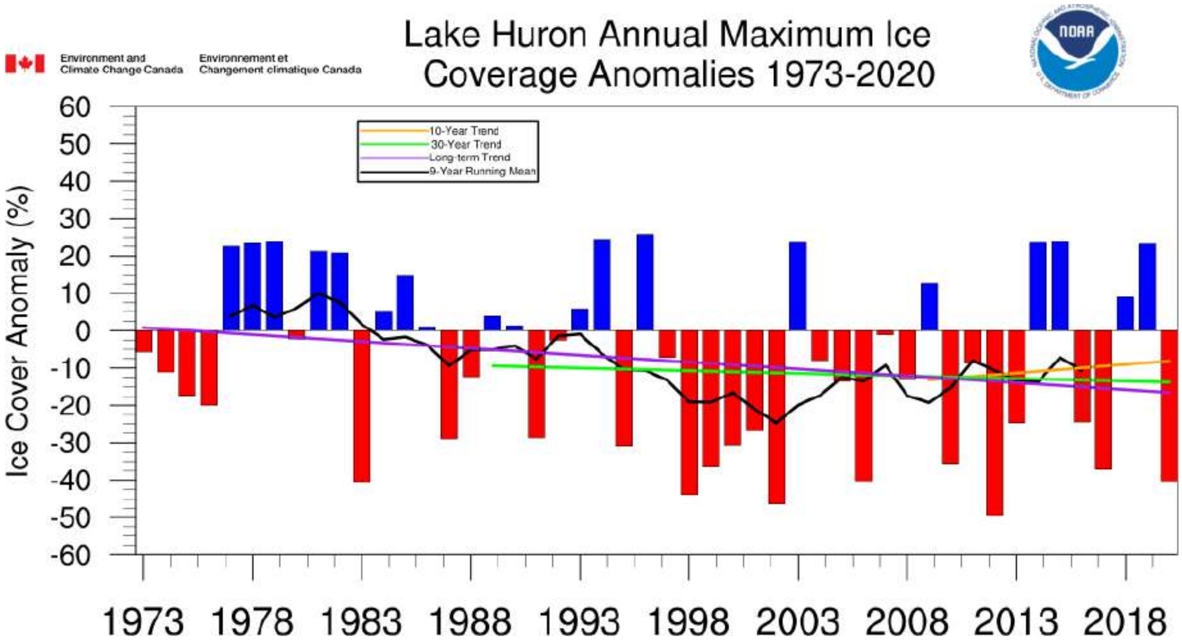

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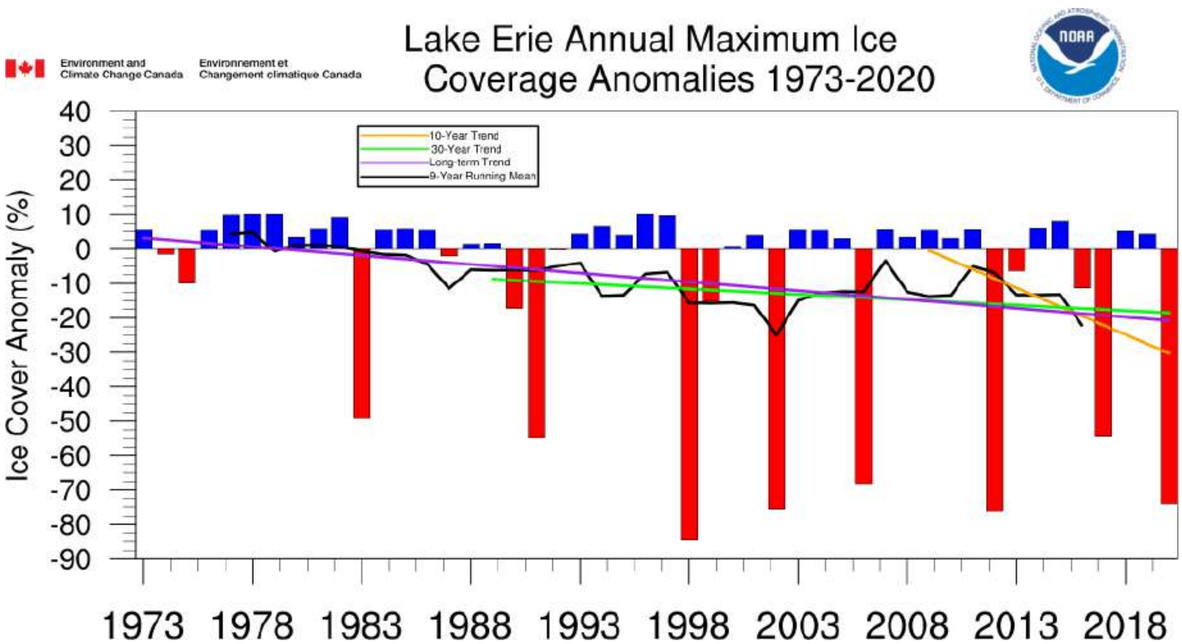

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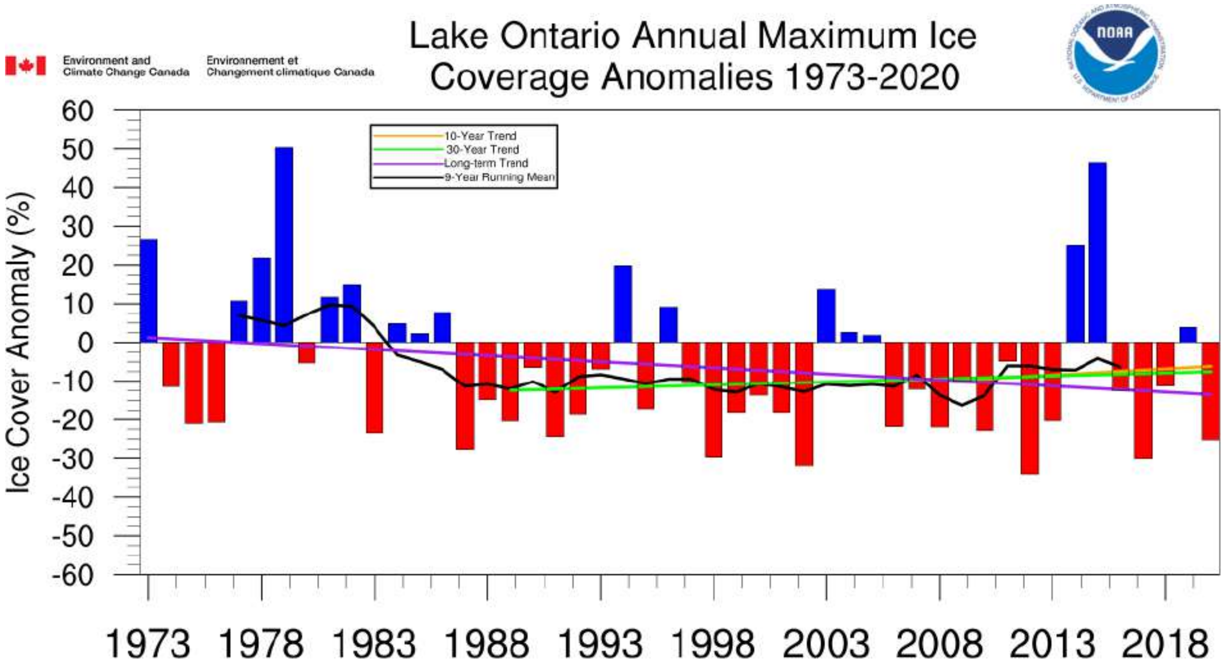

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