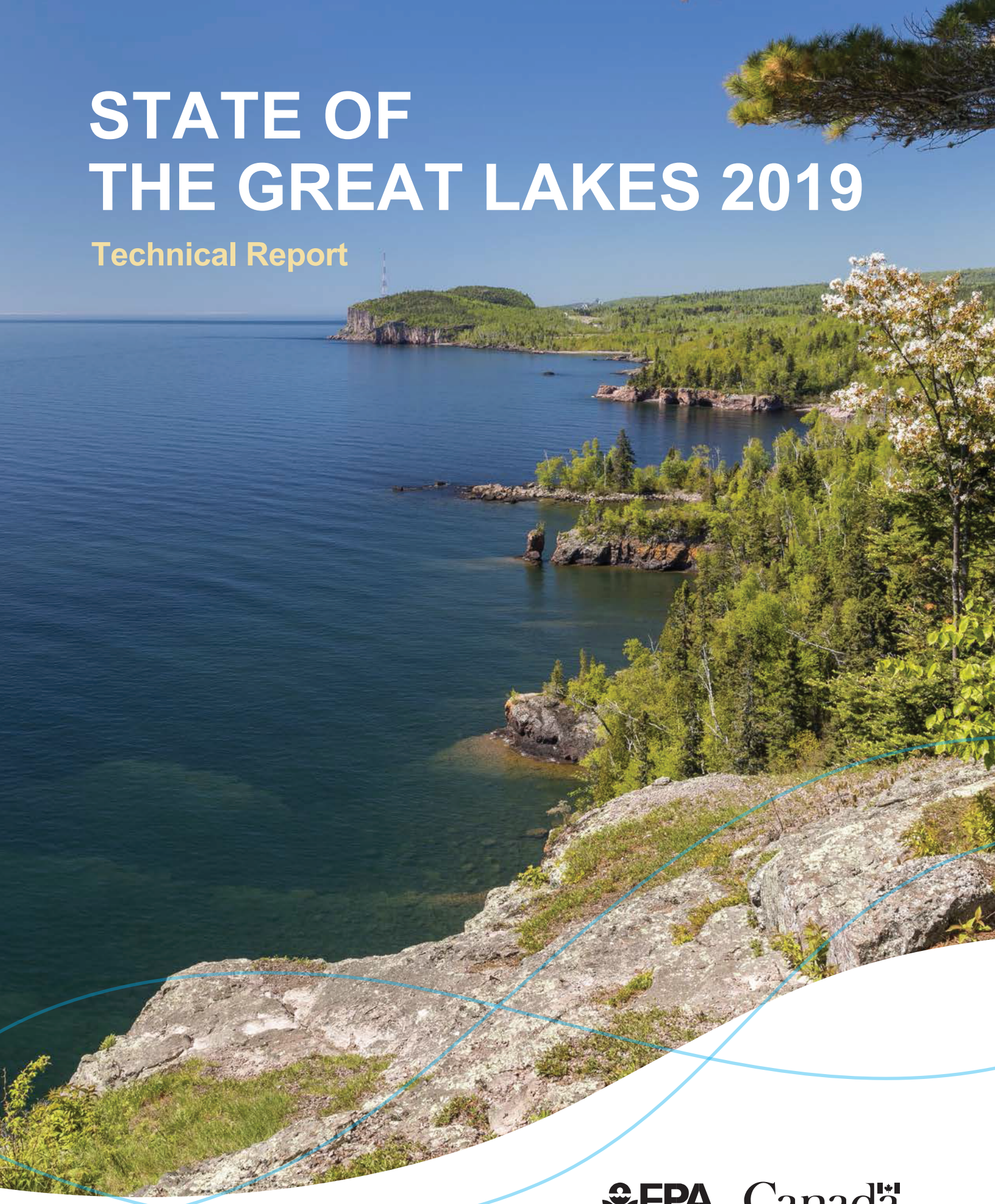
STATE OF THE GREAT LAKES 2019

Technical Report



EPA Canada

Environment and Climate Change Canada and the U.S. Environmental Protection Agency

Cat. No.: En161-3/1E-PDF (Online)
ISSN 2292-1222 (Online)
EPA 905-R-20-044

Suggested Citation: Environment and Climate Change Canada and the U.S. Environmental Protection Agency. 2021. *State of the Great Lakes 2019 Technical Report*. Cat No. En161-3/1E-PDF. EPA 905-R-20-044. Available at binational.net.

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State of the Great Lakes 2019

By the Governments of

Canada

and the

United States of America

Prepared by Environment and Climate Change Canada

and the

U.S. Environmental Protection Agency

Table of Contents

Drinking Water Indicator: Sub-Indicator Technical Reports

Treated Drinking Water sub-indicator report **1**

Beaches Indicator: Sub-Indicator Technical Reports

Beach Advisories sub-indicator report **11**

Fish Consumption Indicator: Sub-Indicator Technical Reports

Contaminants in Edible Fish sub-indicator report **33**

Toxic Chemicals Indicator: Sub-Indicator Technical Reports

Toxic Chemicals in Sediment sub-indicator report **47**

Toxic Chemicals in Water sub-indicator report **59**

Toxic Chemicals in Whole Fish sub-indicator report **73**

Toxic Chemicals in Herring Gull Eggs sub-indicator report **104**

Toxic Chemicals in the Atmosphere sub-indicator report **127**

Habitat and Species Indicator: Sub-Indicator Technical Reports

Coastal Wetland Invertebrates sub-indicator report **156**

Coastal Wetland Fish sub-indicator report **171**

Coastal Wetland Amphibians sub-indicator report **190**

Coastal Wetland Birds sub-indicator report **212**

Coastal Wetland Plants sub-indicator report **235**

Coastal Wetlands: Extent and Composition sub-indicator report **249**

Aquatic Habitat Connectivity sub-indicator report **256**

Phytoplankton sub-indicator report **264**

Zooplankton sub-indicator report **271**

Benthos sub-indicator report **282**

Diporeia sub-indicator report **294**

Lake Sturgeon sub-indicator report **303**

Prey Fish sub-indicator report **322**

Lake Trout sub-indicator report **335**

Walleye sub-indicator report **349**

Fish-Eating and Colonial Nesting Waterbirds sub-indicator report **359**

Nutrients and Algae Indicator: Sub-Indicator Technical Reports

Nutrients in Lakes sub-indicator report **367**

Harmful Algal Blooms sub-indicator report **380**

Cladophora sub-indicator report **395**

Water Quality in Tributaries sub-indicator report **411**

Invasive Species Indicator: Sub-Indicator Technical Reports

Rate of Invasion of Aquatic Non-Indigenous Species sub-indicator report **421**

Impacts of Aquatic Invasive Species sub-indicator report **438**

Sea Lamprey sub-indicator report **453**

Dreissenid Mussels sub-indicator report **463**

Terrestrial Invasive Species sub-indicator report **474**

Groundwater Indicator: Sub-Indicator Technical Reports

Groundwater Quality sub-indicator report **492**

Watershed Impacts and Climate Trends Indicator: Sub-Indicator Technical Reports

Forest Cover sub-indicator report **517**

Land Cover sub-indicator report **528**

Hardened Shorelines sub-indicator report **544**

Watershed Stressors sub-indicator report **564**

Baseflow Due to Groundwater sub-indicator report **581**

Tributary Flashiness sub-indicator report **591**

Human Population sub-indicator report **607**

Precipitation Amounts sub-indicator report **624**

Water Levels sub-indicator report **631**

Surface Water Temperature sub-indicator report **643**

Ice Cover sub-indicator report **654**

Sub-Indicator: Treated Drinking Water

Overall Assessment
Status: Good
Trends:
10-Year Trend (2007-2017)*: Unchanging
Long-term Trend (2004-2017): Unchanging**
Rationale: Eight and a half million Canadians and about 30 million Americans get their drinking water from the Great Lakes. The overall quality of source water and treated water in the Great Lakes basin is "Good". In the period 2007 – 2017, Ontario's source water monitoring network rarely found source water chemical contaminant levels above Ontario Drinking Water Quality Standards (ODWQS), and never found source water radiological contaminant levels above ODWQS. From 2004 – 2017, the percentage of treated water test results that exceeded ODWQS was consistently low. ** Note: The long-term trend is based on Ontario data only.

An average of 95% of the total human population living in the Great Lakes States of Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania and Wisconsin on public water supplies was served with drinking water that met all applicable health-based drinking water standards from 2015 to 2017. The trend is "Unchanging" with no significant increase or decrease in treated drinking water quality based on calculated indices of 95.8% (2012) to 96.0% (2017) of the population served receiving drinking water meeting all health-based standards for the Great Lakes states. * Note: A 6-year trend will be used as an estimate for the 10-year trend as only 6 years of consistent data have been used for the calculation in this report. Assessment of drinking water quality status and trend is the same as previous reporting.

Lake-by-Lake Assessment
Individual lake basin assessments were not prepared for this report.

Status Assessment Definitions
Ontario:
Good: At least 99% of treated drinking water tests meet standards.
Fair: 97-98.9% of treated drinking water tests meet standards.
Poor: Less than 97% of treated drinking water tests meet standards.
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

U.S.:
Good: At least 95% of the population served received treated drinking water meeting all health-based standards.
Fair: 90%-95% of the population served received treated drinking water meeting all health-based standards.
Poor: 90% of the population served received treated drinking water meeting all health-based standards.
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions
Ontario:
Improving: Increase in % of treated drinking water test results meeting standards.
Unchanging: No change in % of treated drinking water tests meeting standards.
Deteriorating: Decrease in % of treated drinking water tests meeting standards.
Undetermined: Data is not available to determine the trend in treated drinking water test results meeting standards.

U.S.:
Improving: Increase in % of population served receiving treated drinking water meeting all health-based standards.
Unchanging: No change in % of population served receiving treated drinking water meeting all health-based standards.
Deteriorating: Decrease in % of population served receiving treated drinking water meeting all health-based standards.
Undetermined: Trend in treated drinking water violations is not determined or data is 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
Treated drinking water supplies should be free from harmful microbial, chemical and radiological contaminants and be safe to drink.

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
This sub-indicator will measure concentrations of chemical substances such as metals (e.g. lead) and other inorganic compounds, pesticides, radionuclides (e.g. tritium), and drinking water disinfection by-products (e.g., trihalomethanes) as well as microbial parameters such as bacteria in treated drinking water and raw water where available. Microbiological measures to include: total coliform and *E.coli* (Ontario and U.S.), and Giardia and cryptosporidium (U.S. only). Chemical contaminants to include: fluoride, lead, selenium, trichloroethene, microcystin-LR, nitrilotriacetic acid, atrazine, nitrate and nitrite, arsenic, uranium, barium and possibly taste and odour compounds.

Please note: turbidity is usually measured by on-line monitors at the WTP. Turbidity is an operational parameter, and most turbidity data resides at the WTP's and is not available for reporting. There are no water quality standards for total organic carbon and dissolved organic carbon.

Ontario:

- Percentage of drinking water tests meeting standards – treated drinking water
- Percentage of drinking water systems meeting standards – raw water

U.S.: Treated Drinking Water is assessed by calculating the percentage population served for a given year during which community water systems provide drinking water that meets all applicable health-based water standards, as reported by the Safe Drinking Water Information System (SDWIS) for the Great Lakes basin. Health-related violations may occur in the finished, or treated, drinking water at a treatment facility or elsewhere through the distribution system. Violations may be due to elevated levels of chemicals (e.g., atrazine, nitrate, nitrite, heavy metals) or treatment requirements.

Ecological Condition
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 as low as possible, continual efforts should be made to decrease microbial, chemical and radiological contamination of source water.

Ontario
The Ontario Ministry of the Environment, Conservation and Parks provided the data for the Canadian component of this report. The Ministry's Drinking Water Surveillance Program (DWSP) provided source water data and the Ministry's Drinking Water and Environmental Compliance Division (DWECD) provided treated drinking water data. DWSP is a scientific program run in partnership with volunteer municipalities and First Nations. DWECD has lead responsibility for program and operational activities related to the protection and provision of safe drinking water in Ontario, from source to tap.

The source water data is from select municipal residential and First Nation drinking water systems. The treated water data is from all municipal residential drinking water systems, and therefore represents the vast majority of water consumed by Ontarians. Both the source water and treated water data are from systems whose sources include not

only the Great Lakes, but also inland lakes, rivers and groundwater.

This report compares source water and treated water sample results to the Ontario Drinking Water Quality Standards (ODWQS). The ODWQS are Ontario's human health standards for microbial, chemical and radiological parameters in treated drinking water.

Source Water

Good quality source water is an important part of the drinking water safety net. Drinking water sources in the Great Lakes basin include the Great Lakes, inland lakes, rivers and groundwater. Potential contaminants of drinking water sources include microbes, chemicals and radioactive substances. Source water must be treated to make it safe to drink. Surface water generally requires more extensive treatment than groundwater because surface water isn't naturally filtered like groundwater.

Table 1 presents source water results from DWSP for thirteen parameters and a radiation screening test. The Ontario portion of the sub-indicator report compares source water contaminant levels to treated water quality standards.The first twelve parameters (arsenic to uranium) are chemical parameters. They were chosen because they have occurred at high concentrations in source water in the U.S. or Canada, or because they represent a class of contaminants (atrazine for pesticides and microcystin-LR for algal toxins). The thirteenth parameter, tritium, is a radiological parameter and was included to examine potential impacts from the nuclear power industry. The radiation screening test determined whether any other radiological parameters exceeded Ontario drinking water quality standards (ODWQS). Microbes account for most occurrences of source water levels above ODWQS; however, DWSP source water monitoring does not include microbial parameters because DWSP focuses on chemical and radiological parameters.

Table 1 summarizes data from 109 to 118 drinking water systems, depending on the year. In 2007 two of the systems were First Nations systems and that number increased to four by 2015.

In the period 2007 – 2017, the chemical parameters that had source water concentrations greater than ODWQS were trichloroethene, fluoride, lead, selenium, microcystin-LR, nitrilotriacetic acid and arsenic. Whenever high levels of a given parameter occurred in more than one year, they were always repeat occurrences at the same one or two drinking water systems. The high levels of trichloroethene, fluoride, lead, and selenium occurred in groundwater whereas the high levels of microcystin-LR, nitrilotriacetic acid and arsenic occurred in surface water. The only high levels to occur in the Great Lakes were for microcystin-LR (at the one drinking water system where high levels occurred) and nitrilotriacetic acid (at one system out of two where high levels occurred).

The high levels of fluoride, lead and selenium were caused by naturally occurring geologic deposits. The high levels of trichlorethene and nitrilotriacetic acid were from man-made pollution. Microcystin-LR comes from algae blooms, but high microcystin-LR levels are mostly the result of man-made phosphorus pollution which causes algae to grow into excessively large blooms. It is not known what the source of the arsenic was at the one system where high levels occurred. In total, from 2007 – 2017 ten drinking water systems (9% of systems) had occurrences of source water chemical concentrations greater than ODWQS.

Only one drinking water system had source water microcystin-LR levels above the Ontario drinking water quality standard. That drinking water system is perhaps the most impacted by algal blooms in Ontario, and it was added to the DWSP network in 2015. Therefore the sudden occurrence of high levels of microcystin-LR in Table 1 in 2015 and 2016 is due to a change in DWSP network design and does not necessarily indicate worsening source water quality.

There are no atrazine results for 2012 – 2016 and no nitrilotriacetic acid results for 2016 – 2017 because DWSP did not monitor those parameters in those years.

In summary, throughout the period 2007-2017, a small percentage of drinking water systems had source water chemical concentrations greater than Ontario drinking water quality standards (ODWQS). No radiological parameters were found at levels above ODWQS. Ontario's source water quality is "Good".

Treated Water

Figure 1 presents treated water test results for the period 2004 – 2017. The results include microbial, chemical and radiological tests and are from all of Ontario's municipal residential drinking water systems. The number of munici-

pal residential systems in Ontario decreased from 729 in 2004 to 658 in 2017 due mainly to amalgamation of systems. Since 2004, the percentage of tests meeting ODWQS has remained in the range 99.74% - 99.88%. Ontario's treated drinking water quality is "Good".

United States

The information provided by the United States for this report focuses on finished (treated) drinking water. In the United States, the Safe Drinking Water Act Reauthorization of 1996 requires all drinking water utilities to provide yearly water quality information to their consumers. To satisfy this obligation, U.S. water utilities produce an annual Consumer Confidence/Water Quality Report (CC/WQR). These reports provide information regarding source water type (i.e., surface water, groundwater); the availability of source water assessments; and, a brief summary of drinking water systems' susceptibility to:

- Potential sources of contamination;
- Water treatment process;
- Contaminants detected in finished drinking water;
- Violations that occurred; and,
- Other relevant information.

Records of the number and type of health-based violations are also recorded in the nationwide U.S. EPA Safe Drinking Water Information System (SDWIS). Health-based violations in the U.S. include violations of: a Maximum Contaminant Level (MCL) (the highest level of a contaminant that is allowed in drinking water); a Maximum Residual Disinfectant Level (MRDL) (the highest level of a disinfectant allowed in drinking water); and, Treatment Technique (TT) (a required process intended to reduce the level of contaminants in drinking water). (SOGL 2011 Technical Report, Drinking Water Quality, Pg. 235)

In 2012, the total human population living in Great Lakes basin counties in the States of Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania and Wisconsin was 27,348,902 and was served by a total of 4,269 water supply systems. It is important to note that not everyone living in a border county in which the Great Lakes basin boundary goes through receives Great Lakes water, therefore, these population numbers are not representative of the number of people who draw their drinking water from the Great Lakes. Water systems over time become inactive, are consolidated with other water systems or new systems are established. All of these factors can contribute to the differences in the number of water supply systems that are referenced below.

In 2012, 95.8% of the total population had drinking water meeting all applicable health-based drinking water quality standards with 93.6% of the water supply systems in compliance with drinking water quality regulations.

In 2013, the total human population living in the Great Lakes basin counties of the eight Great Lakes States was 26,817,725 and was served by a total of 4,240 water supply systems. 95.8% of the total population had drinking water meeting all health-based drinking water quality standards with 94% of the water supply systems in compliance with drinking water quality regulations.

In 2014, the total human population living in the Great Lakes basin counties of the eight Great Lakes States is 26,672,882 and is served by a total of 4,148 water supply systems. 95.4% of the total population had drinking water that met all health-based drinking water quality standards with 93.8% of the water supply systems in compliance with drinking water regulations.

In 2015, the total human population living in the Great Lakes basin counties of the eight Great Lakes States is 26,654,888 and is served by a total of 4,148 water supply systems. 93.2% of the total population had drinking water that met all health-based drinking water quality standards with 94.6% of the water supply systems in compliance with drinking water regulations.

In 2016, the total human population living in the Great Lakes basin counties of the eight Great Lakes States is 26,746,111 and is served by a total of 4,151 water supply systems. 96.9% of the total population had drinking water that met all health-based drinking water quality standards with 96.4% of the water supply systems in compliance with drinking water regulations.

In 2017, the total human population living in the Great Lakes basin counties of the eight Great Lakes States is 26,797,143 and is served by a total of 4,134 water supply systems. 96.0% of the total population had drinking water that met all health-based drinking water quality standards with 96.0% of the water supply systems in compliance with drinking water regulations.

Figure 2 shows the number of violations and the number of community water systems in violation from 2012 – 2017.

Figure 3 shows the average % that community drinking water systems and population did not have any health based violations in U.S. Great Lakes counties.

Figure 4 is a comparative breakdown by Great Lakes States of the total number of human population impacted by drinking water quality exceedances in Great Lakes states in 2012 - 2017.

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, and 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.

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 (Ontario)	X* (US)		

Clarifying Notes:
* = U.S. EPA is aware of inaccuracies and underreporting of some data in the U.S. EPA Safe Drinking Water Information System. U.S. EPA is working with the states to improve the quality of the data.

Data Limitations

Most contaminants in drinking water rarely exceed guidelines and many are below their analytical detection limit. Since the absolute concentration of some contaminants may not be determinable, it is difficult to show fluctuations in their concentration levels.

The treated water data is from all Ontario municipal residential drinking water systems, and therefore represents the vast majority of water consumed by Ontarians; note however that it does not include data from non-municipal sys-

tems, non-residential systems and private wells. Both the source water data and the treated water data are from systems whose sources include not only the Great Lakes, but also inland lakes, rivers and groundwater; therefore Great Lakes source water quality is better than what Table 1 shows. U.S. source water monitoring is not required or regulated.

For the U.S. data it is important to note that not everyone living in a border county in which the Great Lakes basin boundary goes through receives Great Lakes water, therefore, these population numbers are not representative of the number of people who draw their drinking water from the Great Lakes but rather the populations living in the Great Lakes counties.

Additional Information

Raw water data (source water samples collected inside the WTP) is included in this report where available as it is relevant to the state of the environment and source water quality and protection. Good source water quality is an important part of assuring the safety and quality of drinking water and determines how much treatment is needed. Although good source water quality is very important, treatment is critical for providing safe, high-quality drinking water to Great Lakes basin residents. The main goal of the information presented in this sub-indicator is to ensure that all drinking water provided is protected at its source and treated in such a way that it is safe to drink.

Both treated and raw water, where available, are included in this report to provide a more accurate representation of water quality health in the Great Lakes. In addition, there are other indicators that help to convey the overall water quality health in the Great Lakes basin ecosystem and combined, these indicators provide the overall water quality status and trends in the Great Lakes.

In Ontario and in the U.S., health-based criteria are generally not established for source waters. The Ontario portion of the sub-indicator report compares source water contaminant levels to treated water quality standards.

It would be beneficial for both the United States and Canada to continue efforts in arriving at an agreed-upon and standardized methodology in assessing drinking water quality in the Great Lakes for comparability of metrics.

This indicator is assessing the quality of treated drinking water but note that drinking water quality problems can occur if water distribution systems are compromised.

Acknowledgments

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Enforcement and Compliance History Online (ECHO) SDWA Dashboard: https://echo.epa.gov/trends/comparative-maps-dashboards/drinking-water-dashboard

List of Tables

Table 1. Percentage of drinking water systems where DWSP source water results for select parameters met Ontario Drinking Water Quality Standards.
Source: Ontario Ministry of the Environment, Conservation and Parks, Drinking Water Surveillance Program

List of Figures

Figure 1. Trend in percentage of treated drinking water tests meeting Ontario Drinking Water Quality Standards, for municipal residential drinking water systems.
Sources:
Ontario Ministry of the Environment and Climate Change, *Chief Drinking Water Inspector Annual Report, 2016 – 2017*, https://www.ontario.ca/page/chief-drinking-water-inspector-annual-report-2016-2017
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Figure 2. Health Based Violations and Systems in Violation
Source: U.S. EPA Safe Drinking Water Information System.
Figure 3. Percent of Systems and Population that Met All Health Based Standards
Source: U.S. EPA Safe Drinking Water Information System
Figure 4. Population Impacted by Drinking Water Exceedances
Source: U.S. EPA Safe Drinking Water Information System

Last Updated

State of the Great Lakes 2019 Technical Report

	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Arsenic	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	99%
Atrazine	100%	100%	100%	100%	100%	---	---	---	---	---	100%
Barium	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Fluoride	99%	99%	99%	99%	99%	99%	99%	99%	99%	99%	99%
Lead	99%	99%	100%	100%	99%	100%	100%	100%	99%	100%	100%
Microcystin-LR*	100%	100%	100%	100%	100%	100%	100%	100%	94%	94%	100%
Nitrilotriacetic acid	100%	98%	100%	100%	100%	100%	100%	100%	100%	---	---
Nitrate	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Nitrite	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Selenium	99%	99%	99%	99%	99%	99%	99%	99%	100%	100%	100%
Trichloroethene	97%	99%	98%	98%	98%	98%	98%	99%	99%	98%	98%
Uranium	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Tritium	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Other radiological	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%

'*Only one system had microcystin-LR levels above the standard

Table 1. Percentage of drinking water systems where DWSP source water results for select parameters met Ontario Drinking Water Quality Standards.
Source: Ontario Ministry of the Environment, Conservation and Parks, Drinking Water Surveillance Program

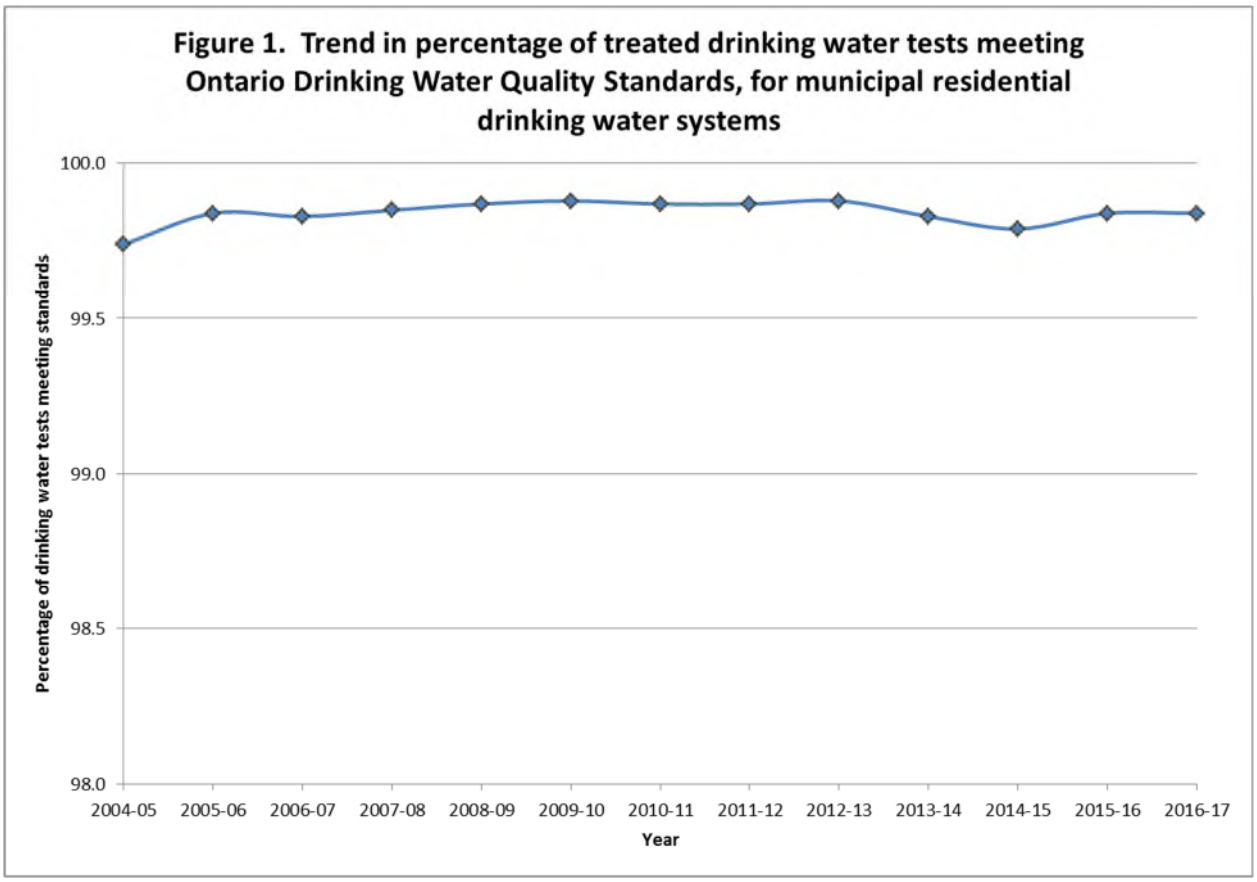


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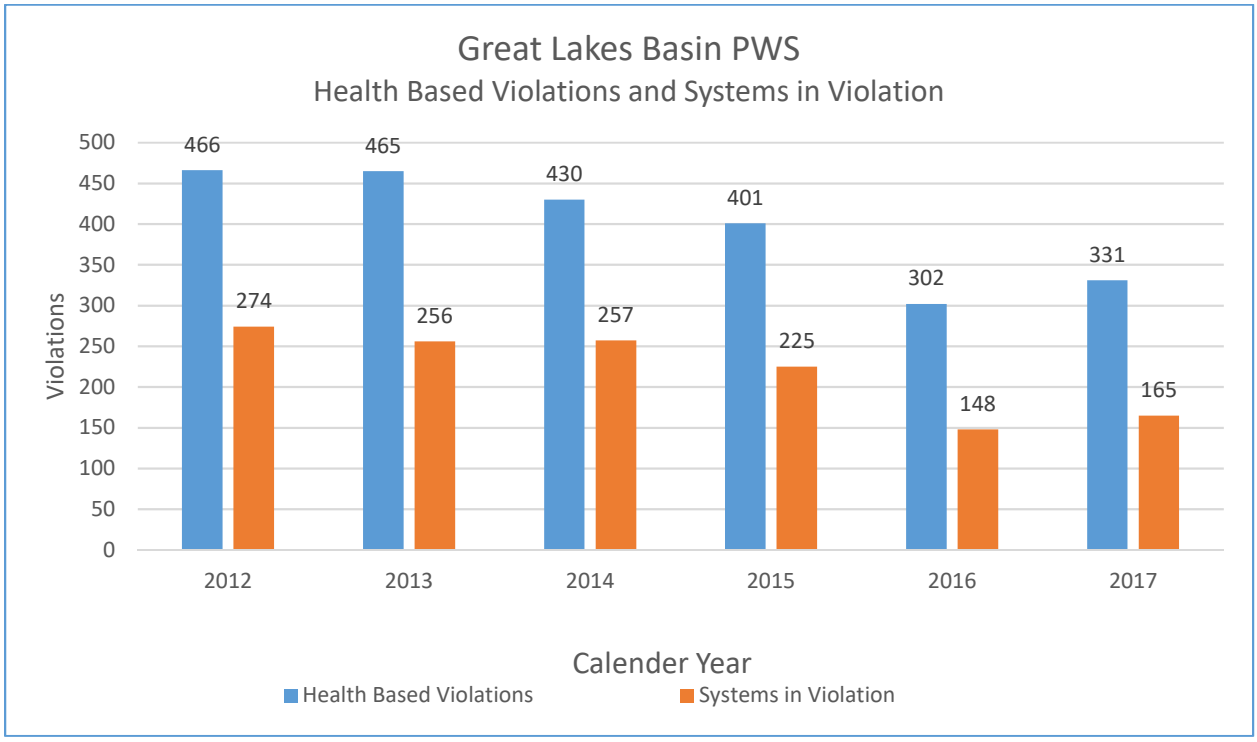


Figure 2. Health Based Violations and Systems in Violation
Source: U.S. EPA Safe Drinking Water Information System.

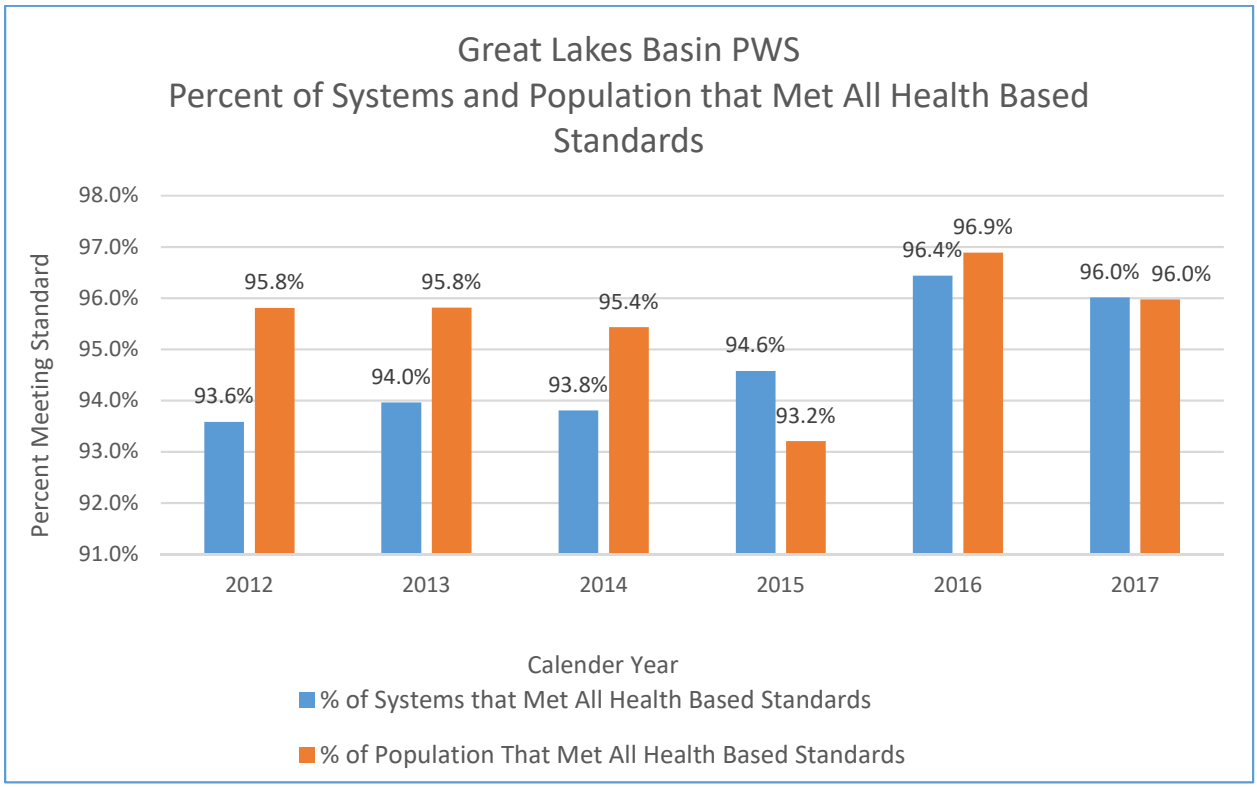


Figure 3. Percent of Systems and Population that Met All Health Based Standards
Source: U.S. EPA Safe Drinking Water Information System

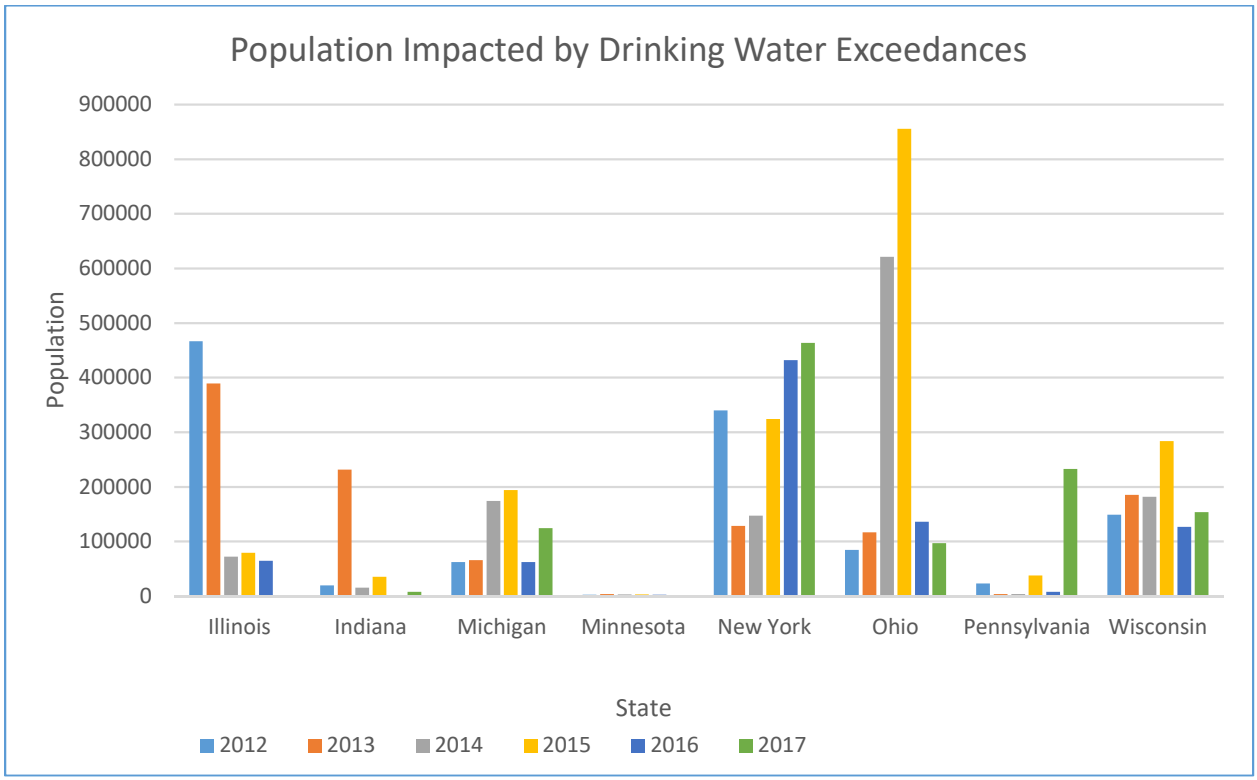


Figure 4. Population Impacted by Drinking Water Exceedances
Source: U.S. EPA Safe Drinking Water Information System

Sub-Indicator: Beach Advisories

Overall Assessment
Status: Good
Trends:
10-Year Trend: Unchanging
Long-term Trend (1999-2017)*: Unchanging
Rationale: From 2015 to 2017, monitored Canadian Great Lakes beaches were open and safe for swimming for an average of 82% of the swimming season. Canadian Great Lakes beaches were open and safe for swimming 4% more overall in the 2015 to 2017 period compared to previous reporting period (2011-2014). From 2015 to 2017, monitored US Great Lakes beaches were open and safe for swimming for an average of 93% of the swimming season, and the trend has been described as Unchanging since 2008. Overall, monitored Great Lakes beaches are binationally assessed as Good, and the 10-year trend for the percentage of Great Lakes beaches open and safe for swimming is described as Unchanging. Although the status since the 2017 report has improved from Fair-Good to Good for Great Lakes monitored beaches, the 10-year trend is described as Unchanging because the overall changes have been minimal over the 10-year period. The long-term trend for Canadian Great Lakes monitored beaches is also described as Unchanging. *Note: long-term trends are based only on Canadian data for the overall and lake-by-lake assessments.

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 to Deteriorating
Long-term Trend (2005-2017)*: Unchanging
Rationale: From 2015 to 2017, monitored Canadian Lake Superior beaches were assessed as Good because these beaches met Ontario bacterial standards and were open and safe for swimming an average of 90% of the swimming season. Monitored Canadian Lake Superior beaches were open and safe for swimming 88% in 2008-2010 and 89% in 2011-2014, and the 10-year Canadian trend is described as Unchanging. From 2015 to 2017, US Lake Superior beaches were assessed as Good because beaches were open and safe for swimming for an average of 95% of the swimming season. Monitored US Lake Superior beaches were open and safe for swimming 98% in 2008-2010 and 96% from 2011-2014, and there has been a slightly Deteriorating US trend since 2008. Overall, monitored Lake Superior beaches are binationally assessed as Good, and the 10-year trend is described as Unchanging to Deteriorating. *Note: the long-term trend is based only on Canadian data for the Lake Superior assessment.

Lake Michigan
Status: Good
10-Year Trend: Unchanging
Long-term Trend (2000-2017): Unchanging*
Rationale: From 2015 to 2017, monitored Lake Michigan beaches were assessed as Good because beaches were open and safe for swimming for an average of 95% of the swimming season. Monitored US Lake Michigan beaches were also open and safe for swimming 95% in 2008-2010 and 2011-2014. As a result, the 10-year trend is described as Unchanging. * Brought forward with data from figures in the 2017 report.

Lake Huron (including St. Marys River)
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1999-2017)*: Unchanging
Rationale: From 2015 to 2017, monitored Canadian Lake Huron beaches were assessed as Good because these beaches met Ontario bacterial standards and were open and safe for swimming for an average of 83% of the

swimming season. Monitored Canadian Lake Huron beaches were open and safe for swimming 83% in 2008-2010 and 82% in 2011-2014, and the 10-year Canadian trend is described as Unchanging. From 2015 to 2017, monitored US Lake Huron beaches were assessed as Good because beaches were open and safe for swimming for an average of 98% percent of the swimming season. Monitored US Lake Huron beaches were open and safe for swimming 97% in 2008-2010 and 2011-2014, and the 10-year US trend is described as Unchanging. Overall, monitored Lake Huron beaches are binationally assessed as Good, and the 10-year trend is described as Unchanging. *Note: the long-term trend is based only on Canadian data for the Lake Huron assessment.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1999-2017)*: Deteriorating
Rationale: From 2015 to 2017, monitored Canadian Lake Erie beaches were assessed as Fair because beaches met Ontario bacterial standards and were open and safe for swimming an average of 74% of the swimming season. Monitored Canadian Lake Erie beaches were open and safe for swimming 78% in 2008-2010 and 69% in 2011-2014, and the 10-year Canadian trend is described as Unchanging. Although the 10-year Canadian trend is Unchanging, the percentage of days that beaches were open and safe for swimming increased 5% since the last reporting period (2011-2014), illustrating recent improvements. From 2015 to 2017, monitored US Lake Erie beaches were assessed as Fair because beaches were open and safe for swimming for 82% of the swimming season. Monitored US Lake Erie beaches were open and safe for swimming 85% in 2008-2010 and 80% in 2011-2014, and the 10-year US trend is described as Unchanging. Overall, monitored Lake Erie beaches are binationally assessed as Fair, and the 10-year trend is described as Unchanging. *Note: the long-term trend is based only on Canadian data for the Lake Erie assessment.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Good
10-Year Trend: Improving
Long-term Trend (1999-2017)*: Unchanging
Rationale: From 2015 to 2017, monitored Canadian Lake Ontario beaches were assessed as Good because beaches met Ontario bacterial standards and were open and safe for swimming for an average of 83% of the swimming season. Monitored Canadian Lake Ontario beaches were open and safe for swimming 75% in 2008-2010 and 77% in 2011-2014, and the Canadian 10-year trend is described as Improving. From 2015 to 2017, monitored US Lake Ontario beaches were assessed as Good because beaches were open and safe for swimming for 94% of the swimming season, and the 10-year US trend is described as Improving. Overall, monitored Lake Ontario beaches are binationally assessed as Good, and the 10-year trend is described as Improving. *Note: the long-term trend is based only on Canadian 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 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 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 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
Unchanging: the percentage of beach days that meet bacterial standards shows no change.
Deteriorating: the percentage of beach days that meet bacterial standards shows a change away from acceptable conditions.
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 body 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 are open and safe for swimming. For example, a sentence stating "93% of beaches were open and safe for swimming," does not indicate that the beaches were open 93 days of the season. It indicates that the beaches were, on average, open and safe for swimming 104 days out of the 112 days in the swimming season (i.e. 93%). 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 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 (previously known as SOLEC) reports.

In Canada, beaches are sampled at a frequency ranging from once per month up to daily sampling, depending on the health unit. 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.

Ecological Condition

Background

Recreational activities like swimming, boating, and beach activities can cause people to come into contact with contaminated water. People exposed to water contaminated with pathogens (parasites, bacteria, viruses), can contract gastrointestinal illness (including diarrhea, nausea, and vomiting), as well as skin, ear, eye, respiratory and wound infections. Escherichia coli (*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 harmful bacterium. *E.coli* can also indicate the presence of other 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.

The Ontario provincial standard is 100 *E. coli* colony forming units (cfu) per 100 mL, based on the geometric mean

(GM) of a minimum of one sample per week from each of at least 5 sampling sites per beach (Recreational Water Protocol, 2014). The Beach Management Protocol states that beaches of 1000 meters of length or greater require one sampling site per 200 meters, with a minimum of 5 samples taken at each site (Beach Management Guidance Document, 2014). In some cases, local Health Units in Ontario have implemented a more frequent sampling procedure than is outlined by the provincial government. When *E. coli* levels exceed the standard, beach waters are posted as unsafe for the health of bathers until further sampling shows that the water quality is meeting the applicable water quality standards. The average swimming season in Ontario begins at the end of May and continues until the first weekend in September, but some health units may have a longer or shorter season than the norm. The difference in the swimming season length, the number of beaches sampled each season, as well as the frequency of sampling are all factors that may skew the final result of the percent of beaches open and safe for swimming throughout the season.

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 GM, and a maximum of 300 *E. coli* cfu per 100 mL based on the GM 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 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).

Status of Great Lakes Beach Advisories

Canadian Great Lakes Beaches

The percentage of days that monitored Canadian Great Lakes beaches were open and safe for swimming during 2015 through 2017 is an average of 82% of the swimming season (Figure 1). The percentage of days that monitored Canadian Great Lakes beaches met Ontario bacterial standards and were open and safe for swimming is an average of 79% of the swimming season over a 10-year period (2008-2017). The 10-year trend is described as Unchanging as there was only a 1% improvement from the percentage of beaches that were safe and open for swimming from the 2017 report, although noting that was only based on 4 years of data from 2011 to 2014. Also, Canadian Great Lakes beaches were open and safe for swimming 78% in 2011-2014 (State of the Great Lakes (SOGL) 2017 Technical Report) and 79% in 2008-2010 (SOGL 2011) report), lending to the Unchanging trend for the 2019 assessment. However, the increase from 78% (2011-2014 data) to 82% for the most recent 3 years of data (2015-2017) reflects an Improving trend for Beach Advisories in the Canadian Great Lakes in the short-term.

The long-term trend for Canadian Great Lakes beaches was assessed as Unchanging. The long-term average of beaches open and safe for swimming during the swimming season was 83%. The percentage of monitored beaches

meeting standards in the 2019 report is 1% below the long-term average and 3% above the 10-year average. This analysis is based on the number of days within a swimming season that beaches are open and safe for swimming. Note that this report focuses on the actual number of days that a beach is open and safe for swimming, so results are slightly different than the overall health unit reports. Due to a delay in receiving results from beach water sampling, beaches are sometimes not posted by health units on days that a beach would actually be unsafe for swimming. Since the *E .coli* geometric data is viewed after the season is complete, this report focuses on those beach days in a swimming season that are actually safe for swimming over the entire swimming season. Depending on each health unit's protocol and the beach, the length of the swimming season in which the beach is actively monitored may vary along with the frequency in monitoring. Over the 2015-2017 period, the average length of the swimming season was 91 days. Lake Erie and Lake Ontario had longer average swimming seasons compared to Lake Huron and Lake Superior. The more southern location of Lake Erie and Lake Ontario provide them with a longer period of warmer temperatures permitting an earlier start and later finish to their swimming seasons. Based on the average swimming season length for monitored Canadian Great Lakes beaches for 2015-2017, for a beach with a status of Good it would be open 73-91 days of the swimming season. A beach with a status of fair would be open 63-72 days, and a beach with a poor status would be open less than 63 days of the average 2015-2017 swimming season.

The average percentage of all monitored Canadian Great Lakes beaches open the entire (100%) swimming season from 2015-2017 was 22%, slightly decreasing from the 26% in 2011-2014. The percentage of monitored Canadian Great Lakes beaches with a status of Good (open 80% or more of the swimming season) during 2015-2017 was 64% overall (Figure 3), improving from the 58% in 2011-2014. From 2015-2017, an average of 111 (64%) of beaches had a status of Good, 26 beaches (15%) had a status of Fair (open 70-79.9% of the swimming season), and 36 beaches (21%) had a status of Poor (open less than 70% of the swimming season) (Figure 4).

Canadian Lake Superior Beaches

During 2015 through 2017, monitored Canadian Lake Superior beaches were assessed as Good overall. These beaches met Ontario bacterial standards and were open and safe for swimming 90% of the swimming season (Figure 6). The average percentage of monitored Canadian Lake Superior beaches open the entire (100%) swimming season from 2015-2017 was 38%, which is a decrease from 51% in 2011-2014. The percentage of monitored Canadian Lake Superior beaches with a status of Good during the 2015-2017 period was 85%, an improvement from 76% in 2011–2014. In 2015-2017, an average of 22 Canadian Lake Superior beaches were monitored. Of those 22 beaches, an average of 19 (85%) has a status of Good, 3 beaches (12%) had a status of Fair, and 1 beach (3%) had a status of Poor (Figure 5). Based on data from 2005-2017, the long-term average for monitored Canadian Lake Superior beaches that were open and safe for swimming was 89% and the trend is considered Unchanging. The 2015-2017 data indicates that Lake Superior was only 1% above the long-term average, while the 10-year average was consistent with the long-term average at 89%.

Canadian Lake Huron Beaches

During 2015 through 2017, monitored Canadian Lake Huron beaches were assessed as Good overall. These beaches met Ontario bacterial standards and were open and safe for swimming 83% of the swimming season (Figure 10). The average percentage of monitored Canadian Lake Huron beaches open the entire (100%) swimming season from 2015-2017 was 29%, which is a decrease from the 36% in 2011-2014. The average percentage of monitored Canadian Lake Huron beaches with a status of Good (open 80% or more of the swimming season) during 2015-2017 was 66%, a slight improvement from 65% in 2011-2014. In 2015-2017 an average of 55 Canadian Lake Huron beaches were monitored, less than the 63 monitored in 2011-2014. Of those 55 beaches, an average of 37 beaches (66%) had a status of Good (open 80% or more of the swimming season), an average of 7 beaches (13%) had a status of Fair (open 70-79.9% of the swimming season), and 12 beaches (21%) had a status of Poor (open less than 70% of the swimming season) (Figure 9).

Canadian Lake Erie Beaches

During 2015 through 2017, monitored Canadian Lake Erie beaches were assessed as Fair overall. These beaches met Ontario bacterial standards and were open and safe for swimming 74% of the swimming season (Figure 13). Based on data from 2015-2017 the average percentage of monitored Canadian Lake Erie beaches open the entire (100%) swimming season from was 11%, improving from 9% in 2011-2014. The percentage of monitored Canadian Lake Erie beaches with a status of Good (open 80% or more of the swimming season) during 2015-2017 was 48%, an

improvement from 36% in 2011-2014. The 2015-2017 data indicates that an average of 41 Canadian Lake Erie beaches were monitored more than the 38 monitored in 2011-2014. Of those 41 beaches, an average of 20 beaches (48%) had a status of Good (open 80% or more of the swimming season), 8 beaches (20%) had a status of Fair (open 70-79.9 of the swimming season), and 13 beaches (32%) had a status of Poor (open less than 70% of the swimming season) (Figure 12).

Canadian Lake Ontario Beaches
During 2015 through 2017, monitored Canadian Lake Ontario beaches were assessed and assigned a status of Good overall. These beaches met Ontario bacterial standards and were open and safe for swimming an average of 83% of the swimming season (Figure 16). The percentage of monitored Canadian Lake Ontario beaches open the entire (100%) swimming season from 2015-2017 was 15%, which is a decrease from 17% in 2011-2014. The percentage of monitored beaches open 80% or more of the swimming season during 2015-2017 was 65% overall. In 2015-2017 an average of 56 monitored Canadian Lake Ontario beaches were monitored, less than the 64 monitored in 2011-2014. Of those 56 beaches, an average of 36 (65%) had a status of Good (open 80% or more of the swimming season), 8 beaches (15%) had a status of Fair (open 70-79.9% of the swimming season), and 11 beaches (20%) had a status of Poor (open less than 70% of the swimming season) (Figure 15).

Annual variability in weather, the number of beaches monitored, high water levels, and the length of the swimming season may affect the variability in days open and safe for swimming during each swimming season. Comparisons of the frequency of beach postings between Canada and the U.S. will be limited due to the use of different water quality criteria in the Great Lakes.

US Great Lakes Beaches
Since 2008, US Great Lakes beaches have been assessed as Good, and the trend in the percentage of monitored US Great Lakes beaches open and safe for swimming is described as Unchanging (Figure 2). Overall, the percentage of monitored US Great Lakes beaches that were open and safe for swimming from 2015 to 2017 was an average of 93%, which is similar to the 2011-2014 (93%) and the 2008-2010 (94%) reporting periods. US Lake Superior beaches are assessed as Good, and the trend in the percentage of days that US Lake Superior beaches are open and safe for swimming is slightly Deteriorating (Figure 7). US Lake Huron and Lake Michigan beaches are assessed as Good and the trend in the percentage of days open and safe for swimming since 2008 is described as Unchanging (Figures 8 & 11). US Lake Erie beaches are assessed as Fair and show an overall Unchanging trend in the percentage of days open and safe for swimming since 2008 (Figure 14). However, US Lake Erie beaches in 2016 and 2017 showed an improvement in the percentage of days open since 2012 to 2014. US Lake Ontario beaches are assessed as Good, and the trend is slightly Improving (Figure 17).

Linkages
Water Levels
- At the end of May 2017, Lake Ontario had water levels 82cm (32in) above average, the highest it's been since the records started in 1918, leading to widespread flooding and erosion along the Lake Ontario shoreline. Lake Ontario's high water levels reduced beach access, submerged docks, and launches in marinas in the spring of 2017. All other Great Lakes had water levels at least 22cm (8.7in) above average. (Great Lakes Quarterly Climate Impacts and Outlook: June 2017, 2017)
- Nine monitored Canadian Lake Ontario beaches were not included in the 2017 data set because they were closed for the entire 2017 swimming season due to high water levels, erosion, debris and/or unsafe conditions. Three Canadian Lake Ontario beaches and one Canadian Lake Erie beach were closed for part of the 2017 swimming season due to high water levels or unsafe conditions. In 2016, one Canadian Lake Ontario beach and one Canadian Lake Huron beach were both closed for part of the 2016 swimming season due to high water levels or unsafe conditions. The beaches closed for only part of the swimming season were included in the 2015-2017 assessment.
- Water levels at the end of August 2017 were above average for all the Great Lakes. Water levels were above average between 25-48cm (10-19in) with Lake Ontario 41cm (16in) above average and Lake Erie 48 cm (19in) above average. In 2016, the water levels at the end of August in the Great Lakes ranged from 2cm (0.8) below average to 28 cm (11in) above average. Lake Ontario water levels were 2cm below average at the end of August 2016.
- Extreme weather events during the swimming season in August 2016 including a tornado outbreak that impacted parts of Indiana, southwestern Ontario, and northwest Ohio, may have contributed to the high water

levels and unsafe conditions that resulted in partial season beach closures along Lake Ontario and Lake Erie. Water level averages are based on water levels from 1918 to the year before, for example the 2017 data is compared to the water level average from 1918-2016.

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).
- 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).
- The spring and summer of 2017 both had above average precipitation levels with March to May 2017 receiving 126% of average precipitation during the spring and June to August having 110% of average precipitation during the summer, the highest month being June with 135% average precipitation.
- Untreated water was released into Lake Ontario to mitigate flooding due to heavy rainfall events. In May 2017, there were reports that Toronto Harbor had *E.coli* levels exceeding safe levels for human contact or boating by 16-30 times. (Great Lakes Quarterly Climate Impacts and Outlook: June 2017, 2017)

Cladophora/Harmful Algal Blooms

- One Canadian Lake Erie beach was closed for part of the 2017 swimming season due to excessive algae that prevented sampling, but it was not sampled to determine if it was toxic algae. The beach was included as part of the 2017 data set and 2015-2017 assessment.
- Blooms which produce toxins deteriorate water quality in the Great Lakes, which can have adverse health effects. Thereby, the availability of recreation and safe swimming waters decrease with the presence of harmful algal blooms (Centers for Disease Control and Prevention, 2018). HABs have the ability to blanket the surface of a lake and block sunlight. Sun rays penetrate the surface of the water and cause cellular damage to bacteria beneath the surface. Thereby, HABs can cause greater amounts of bacterial organisms in the Great Lakes (Ali, 2018).
- *Cladophora* can facilitate harmful bacteria such as *E.coli*. Further, *Cladophora* mats house *E.coli*, which allows the bacteria to survive in recreational waters (Englebert et al., 2008).
- *E. coli* also has a unique, strong association with *Cladophora*. *Cladophora* provides a suitable habitat for indicator bacteria and potentially pathogens to persist and grow, which may in turn impact recreational beach water quality (USGS, 2009).

Invasive Species – Dreissenid Mussels

- Michigan State University examined the water quality of 45 beaches on Lake Michigan from 2000 to 2014 and found that *E. coli* levels have decreased over the time period likely as a result of quagga mussels (Ali, 2018). As a result, there have been fewer beach advisories and closures.
- 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 explains the decrease in *E. coli* levels across Lake Michigan (Ali, 2018). However, cellular damage can also impact "good" bacteria, which are responsible for sustaining the health of the ecosystem by recycling carbon and nutrients.

Citizen Science and/or Other Bodies of Knowledge

The University of Windsor's Great Lakes Institute for Environmental Research (GLIER) has been conducting a water quality analysis project to identify the location of high *E.coli* levels and to determine why the levels are high. GLIER has enlisted citizen scientists to help with the collection of water samples along Lake Erie, Lake St. Clair, the Detroit River, and waterways in Windsor-Essex County. The citizen scientists were provided with instructions on water sampling procedures, water sampling kits, sampling dates, and sampling drop off locations. In 2016, approximately 20 citizen scientists collected water samples for the project over the course of the summer. In 2017, a Water Blitz event was held on August 19, where citizen scientists collected water samples on the same day in 450

locations. The collected samples analyzed by GLIER researchers using a DNA technique to identify microbes including *E.coli*. Researchers at GLIER hope to use this information for the mapping and modelling of *E.coli* hotspots in Windsor-Essex County. Identifying hotspots could help determine when to issue water advisories, post beach closures, and help public health officials identify probable sources of microbial outbreaks (Heath, 2017; Roberts, 2017).

The Lake Huron Centre for Coastal Conservation has a program called Coast Watchers that enlists volunteer citizen scientists to collect observational and measured data along the beaches of Lake Huron. Some of the data Coast Watchers collect includes wave height and direction, beach debris including algae, carcasses, garbage,-as well as the presence and population of shorebirds and gulls on the beach. Through the Coast Watchers program a long-term dataset can be establish that may allow for the tracking of conditions and trends in Lake Huron's coastal health. The Lake Huron Centre for Coastal Conservation hosts community workshops to educate local residents on stewardship of the coastline and engage members of the local community to observe and assist in improving the quality of nearshore waters (The Lake Huron Centre for Coastal Conservation, 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 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 feasible 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.
- 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 GM, and a maximum of 300 *E. coli* cfu per 100 mL), whereas Canada posts a beach as unsafe if *E. coli* levels are above the Ontario criteria of 100 *E. coli* per 100ml.
- At the provincial level in Canada, the Ontario Ministry of Health and Long-Term Care (MOHLTC) raised the provincial benchmark in January 2018. The new recreational water quality standard is a geometric mean concentration (minimum of five samples) of less than 200 *E. coli* per 100 mL, and a single-sample maximum concentration of less than 400 E. coli per 100 mL Thereby, Canada and U.S. *E. coli* standards will be similar, making the data more comparable in future binational reports.
- The long-term trend assesses the entire data set available for Canadian Great Lakes beaches. Lake Erie, Lake Ontario, and Lake Huron have quality data sets starting in 1999. However, Lake Superior beach monitoring data was not collected consistently prior to 2005. Therefore, the long-term trend assessment for Lake Superior includes data from 2005-2017 while the other lakes are assessed on monitoring data from 1999-2017.
- 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 but for this sub-indicator it will track 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 health unit resources, frequency of public use, exceedance history of beach, and the current conditions. Monitored beaches may be closed because of exceedances for longer than that is necessary, or beaches may remain open if *E. coli* levels rise above the safe levels between monitoring days. Ontario Public Health Units are generally tasked with monitoring beaches in their county, but health units don't necessarily monitor all beaches in their area.
- At this time, the Canadian content in this report includes beach water quality data from Ontario Public Health Units and does not include Provincial Park or other sources of beach water quality data.
- Average percentage of days that US Great Lakes beaches were open and safe for swimming for 2008-2014 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 2014 may be different from data accessed in 2019. Data in this report were accessed from BEACON in March-April 2019.

Additional Information

Additional point and non-point source pollution at coastal areas due to 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 sources 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.

Genomics research from Dr. Tom Edge at Environment and Climate Change Canada may also become an increasingly important technique in the beach analysis area. Research with the Genomics Research and Development Initiative (GRDI) is using DNA markers to identify bacteria found in the gut of seagulls to distinguish seagull fecal contamination from that of other birds and animals. In urban centers with large seagull populations, seagull fecal contamination was identified in water samples from beaches, runoff from parking lots and more. This is a promising cost-effective and targeted solution to measure seagull fecal contamination in water samples that could better target the sources of *E. coli* in the Great Lakes. Beach managers are now considering seagulls as a source of contamination along with sewer systems and runoff when there is a beach closure for *E.coli* exceedance. Beach managers in Ottawa, Britannia, and Petrie Island have taken measures to limit seagulls' access to beaches with the aim to reduce their fecal contamination and improve water quality. This research is continuing under the GRDI with DNA sequencing techniques being used to better characterize and understand the microbial diversity in

water samples. More comprehensive information about aquatic microorganisms will help identify pollution threats and inform decision making on water quality protection (Government of Canada, 2016).

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 feasible 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.

Bluffer's Park Beach

As previously mentioned, 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 water conservation efforts. As a result, some of these beaches now meet water quality standards, and are now open and safe for swimming 80-100% of the swimming season.

For example, Bluffer's Park Beach, located in Scarborough, was once unsafe for swimming 93% of the swimming season. The primary causes of poor water quality were as a result of urbanization and land practices. In 2006, Lake Ontario Waterkeeper, Toronto and Region Conservation Authority (TRCA), and the City of Toronto completed a thorough investigation of the beach. These parties worked together and created a dune and wetland system to mitigate high concentrations of *E.coli* (Wallis, 2016). Today, Bluffer's Beach is considered a Blue Flag beach, thereby passing water quality tests over 90% of the time, and fulfilling environmental management, and safety standards (Lake Ontario Waterkeeper, n.d).

Acknowledgements

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Peggy Donnelly, U.S. EPA Region 5
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Holiday Wirick, U.S. EPA, Region 5

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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List of Figures

Figure 1. Percentage of days that monitored Canadian Great Lakes beaches were open and safe for swimming from 2008-2017. Good = 80% or >; Fair = 70 to 79.99%; and Poor = <69.99%.
Source**:** Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 2. Percentage of days that US Great Lakes beaches were open and safe for swimming from 2008 to 2017. Good = 90% or >; Fair = 80 to 89.99%; and Poor = <79.99%.
Figure 3. Percentage of days that monitored Canadian Great Lakes beaches were assessed with a status of Good, Fair, or Poor from 2008-2017.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017

Figure 4. Percentage of days that monitored Canadian Great Lakes beaches were assessed with a status of Good, Fair, or Poor from 2015-2017.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 5. Percentage of days that monitored Canadian Lake Superior beaches were assessed with a status of Good, Fair, or Poor from 2008-2017.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 6. Percentage of days that monitored Canadian Lake Superior beaches were open and safe for swimming from 2008-2017. Good = 80% or >; Fair = 70 to 79.99%; and Poor = <69.99%.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 7. Percentage of days that US Lake Superior beaches were open and safe for swimming from 2008 to 2017. Good = 90% or >; Fair = 80 to 89.99%; and Poor = <79.99%.
Source: Data collected from US states and reported to US EPA's Beach Advisory and Closing Online Notification (BEACON) system
Figure 8. Percentage of days that US Lake Michigan beaches were open and safe for swimming from 2008 to 2017. Good = 90% or >; Fair = 80 to 89.99%; and Poor = <79.99%.
Source: Data collected from US states and reported to US EPA's Beach Advisory and Closing Online Notification (BEACON) system
Figure 9. Percentage of days that monitored Canadian Lake Huron beaches were assessed with a status of Good, Fair, or Poor from 2008-2017.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 10. Percentage of days that monitored Canadian Lake Huron beaches were open and safe for swimming from 2008-2017. Good = 80% or >; Fair = 70 to 79.99%; and Poor = <69.99%.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 11. Percentage of days that US Lake Huron beaches were open and safe for swimming from 2008 to 2017. Good = 90% or >; Fair = 80 to 89.99%; and Poor = <79.99%.
Source: Data collected from US states and reported to US EPA's Beach Advisory and Closing Online Notification (BEACON) system
Figure 12. Percentage of days that monitored Canadian Lake Erie beaches were assessed with a status of Good, Fair, or Poor from 2008-2017.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 13. Percentage of days that monitored Canadian Lake Erie beaches were open and safe for swimming from 2008-2017. Good = 80% or >; Fair = 70 to 79.99%; and Poor = <69.99%.
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Figure 15. Percentage of days that monitored Canadian Lake Ontario beaches were assessed with a status of Good, Fair, or Poor from 2008-2017.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017
Figure 16. Percentage of days that monitored Canadian Lake Ontario beaches open and safe for swimming during the swimming season from 2008-2017. Good = 80% or >; Fair = 70 to 79.99%; and Poor = <69.99%.
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Figure 17. Percentage of days that US Lake Ontario beaches were open and safe for swimming from 2008 to 2017. Good = 90% or >; Fair = 80 to 89.99%; and Poor = <79.99%.

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State of the Great Lakes 2019 Technical Report

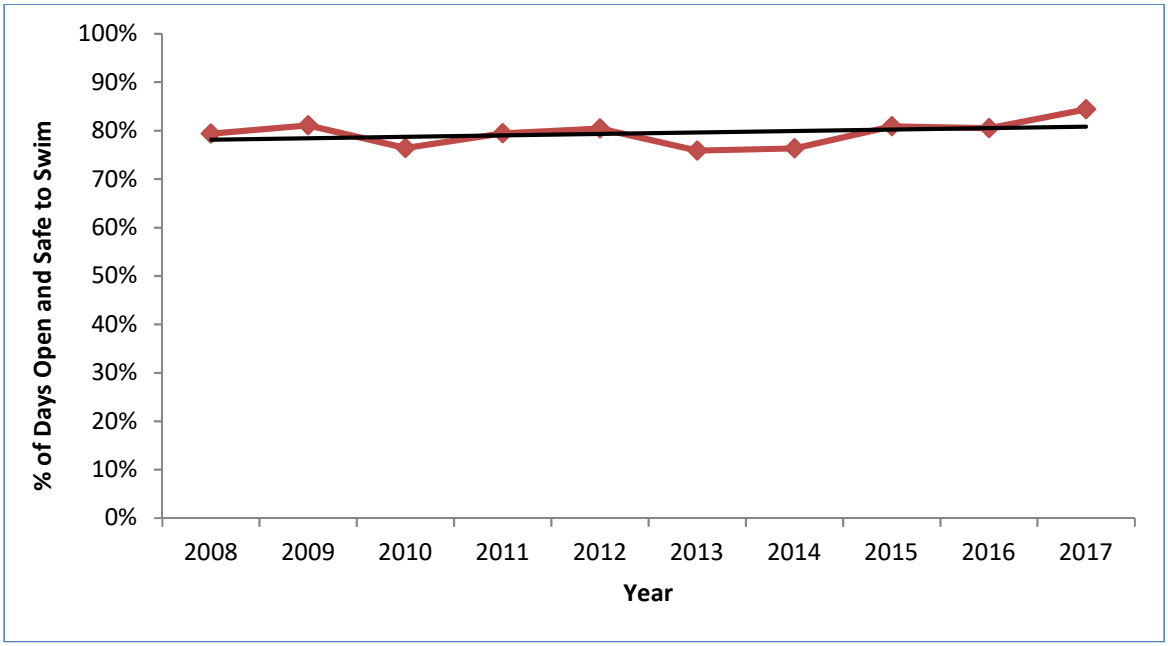


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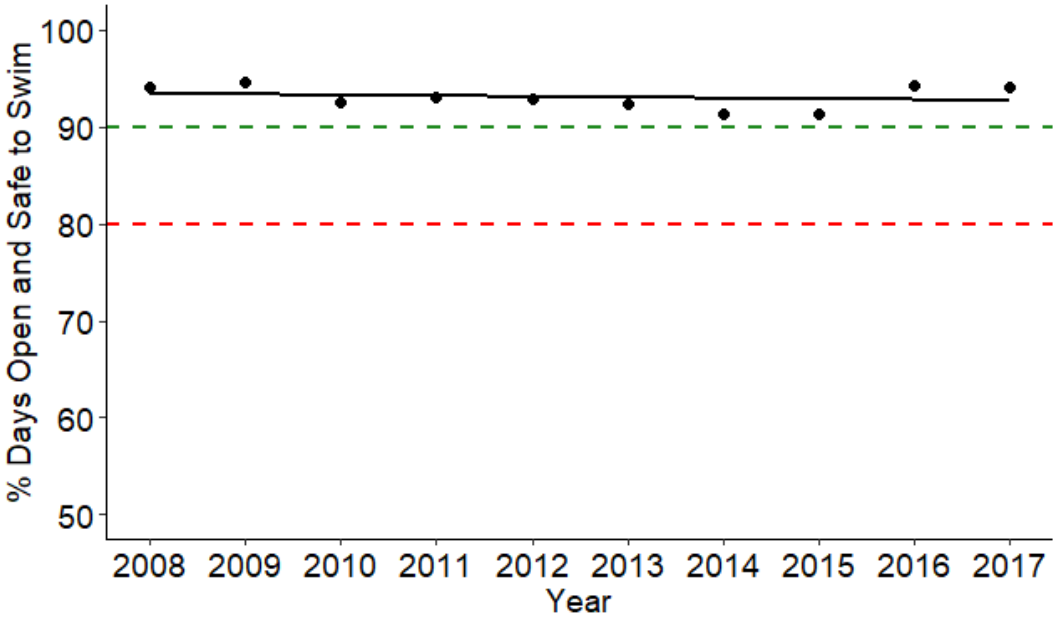


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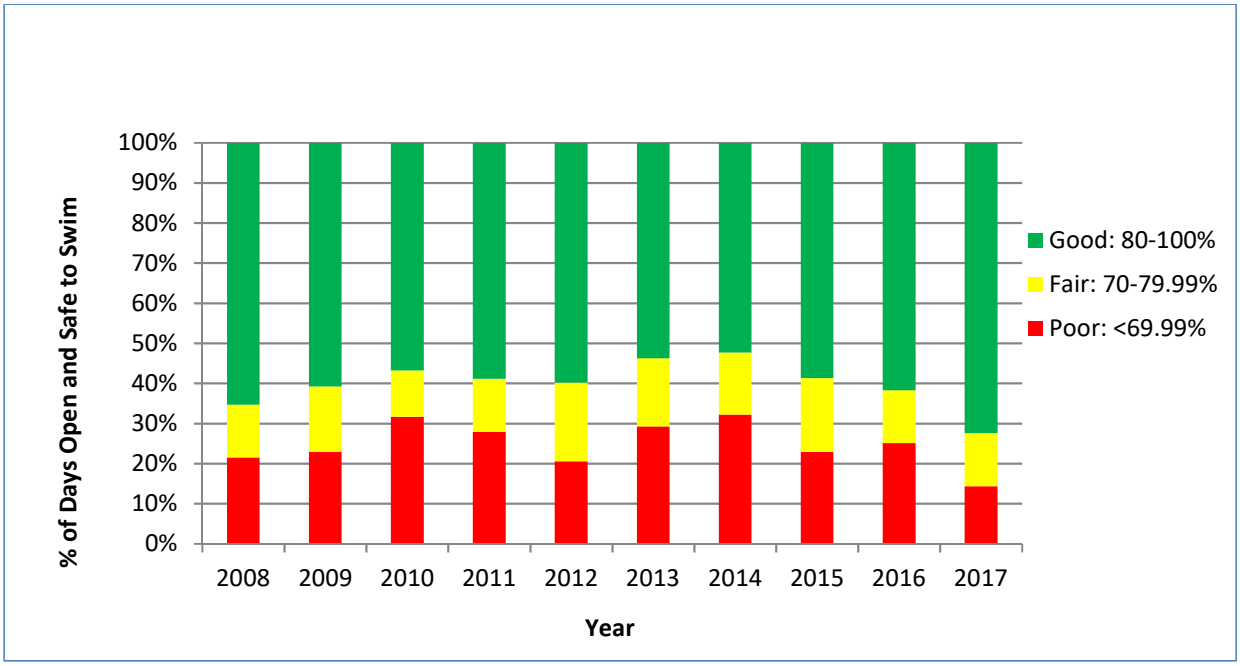


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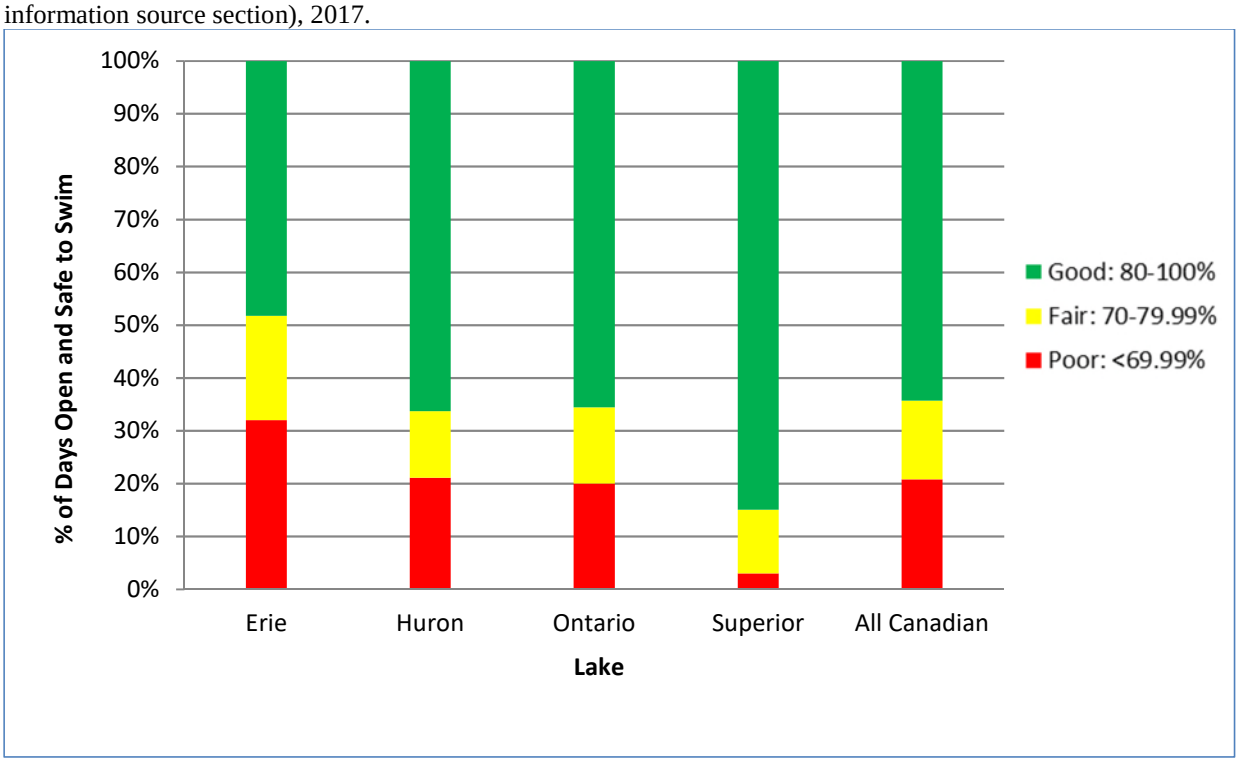


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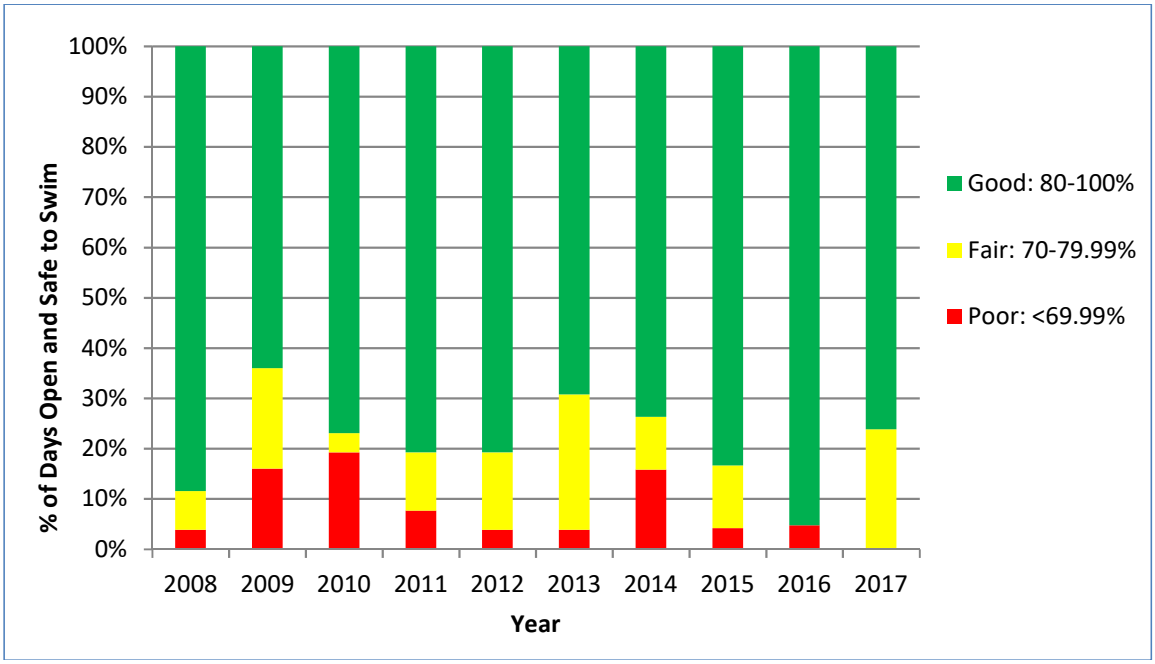


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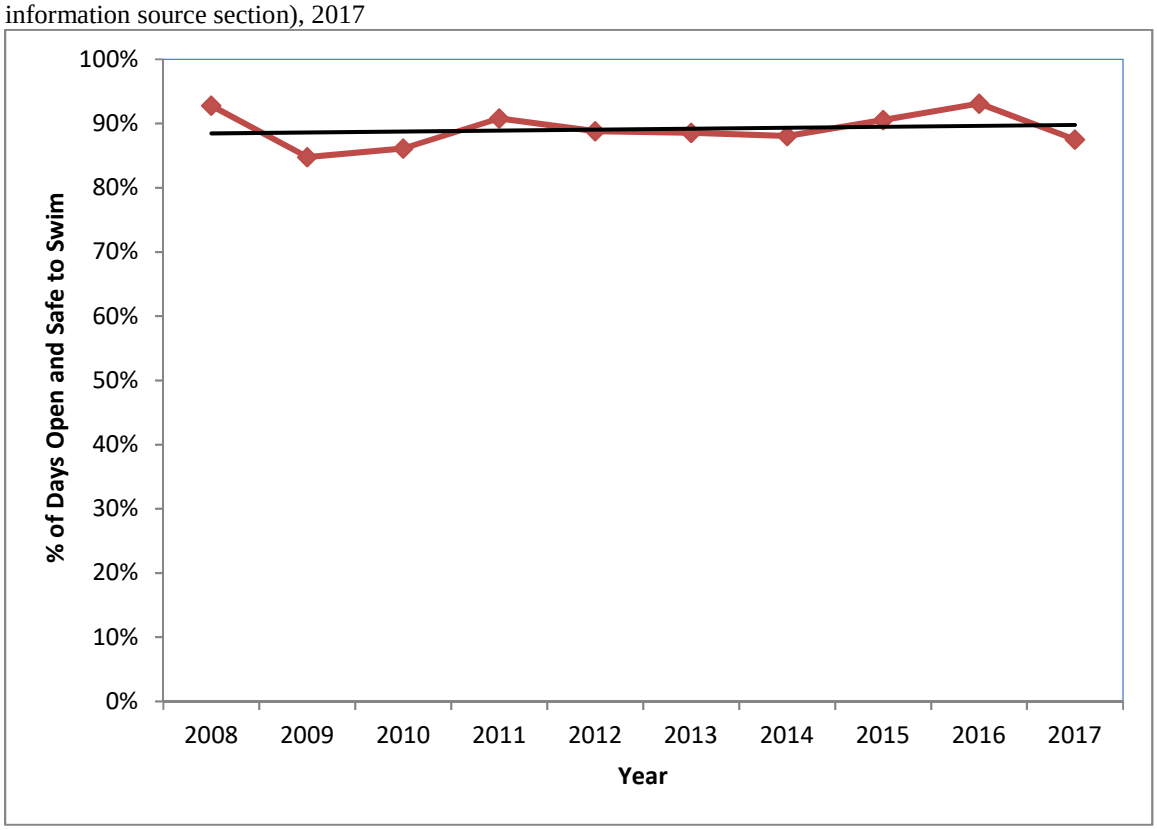


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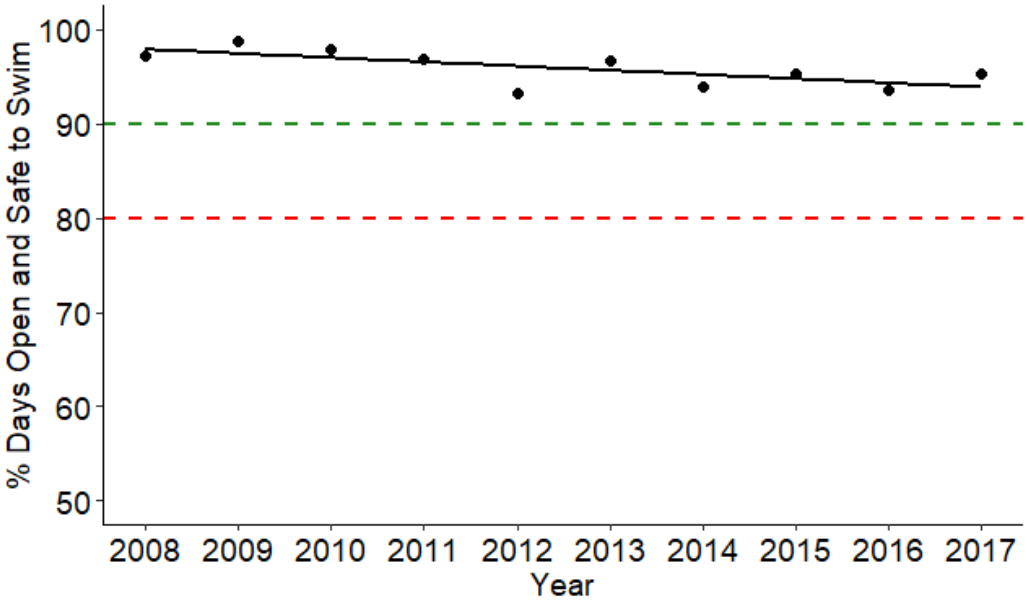


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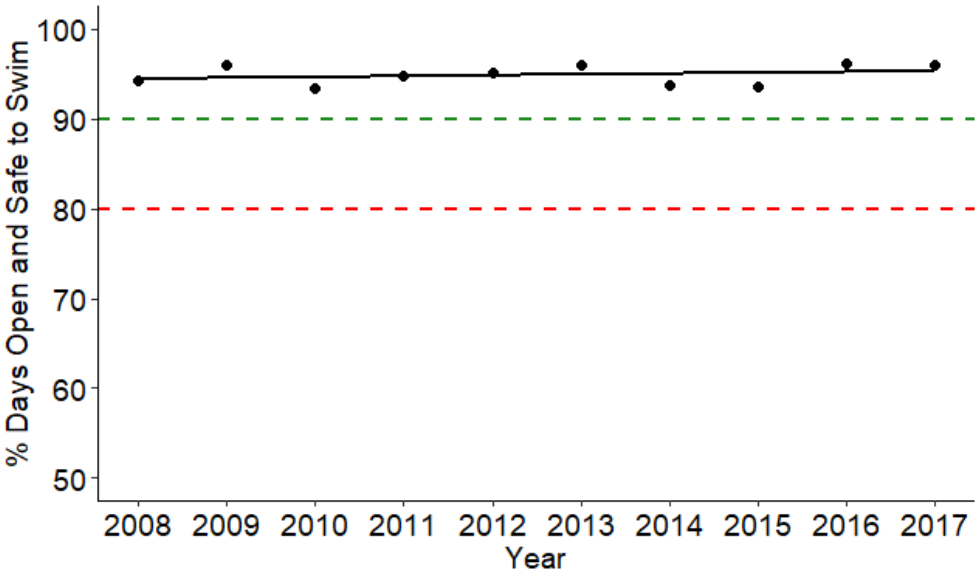


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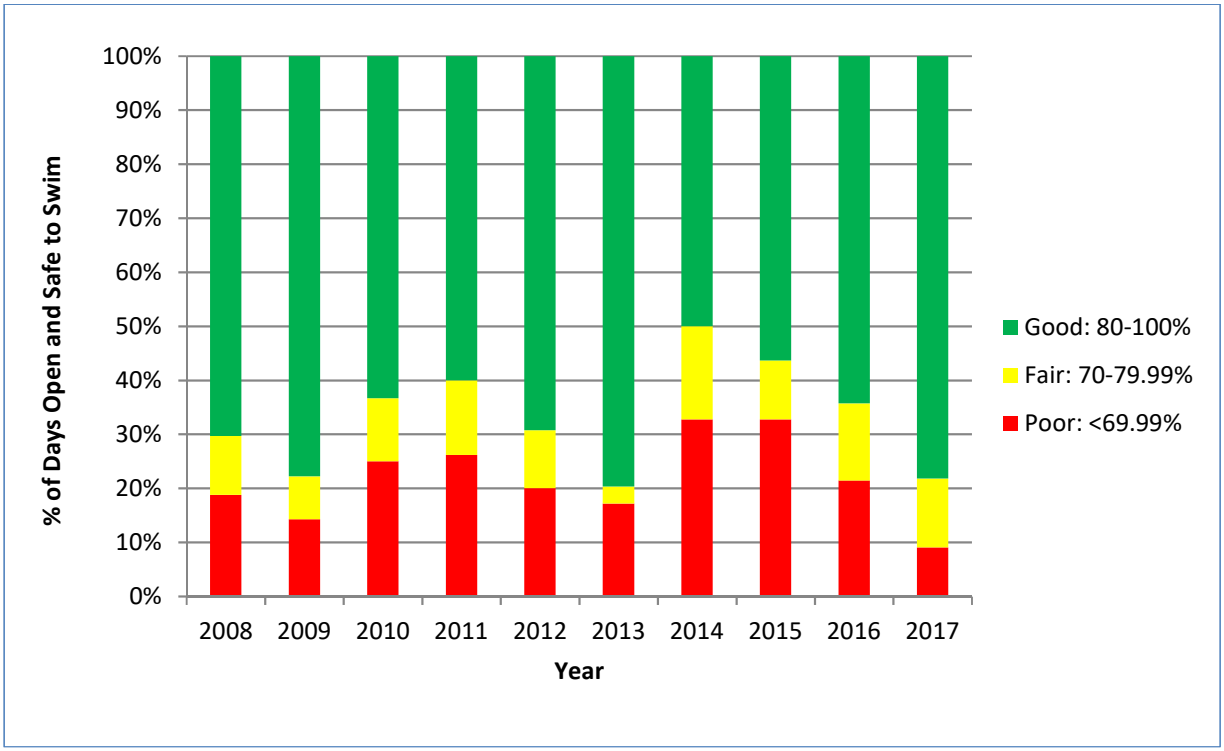


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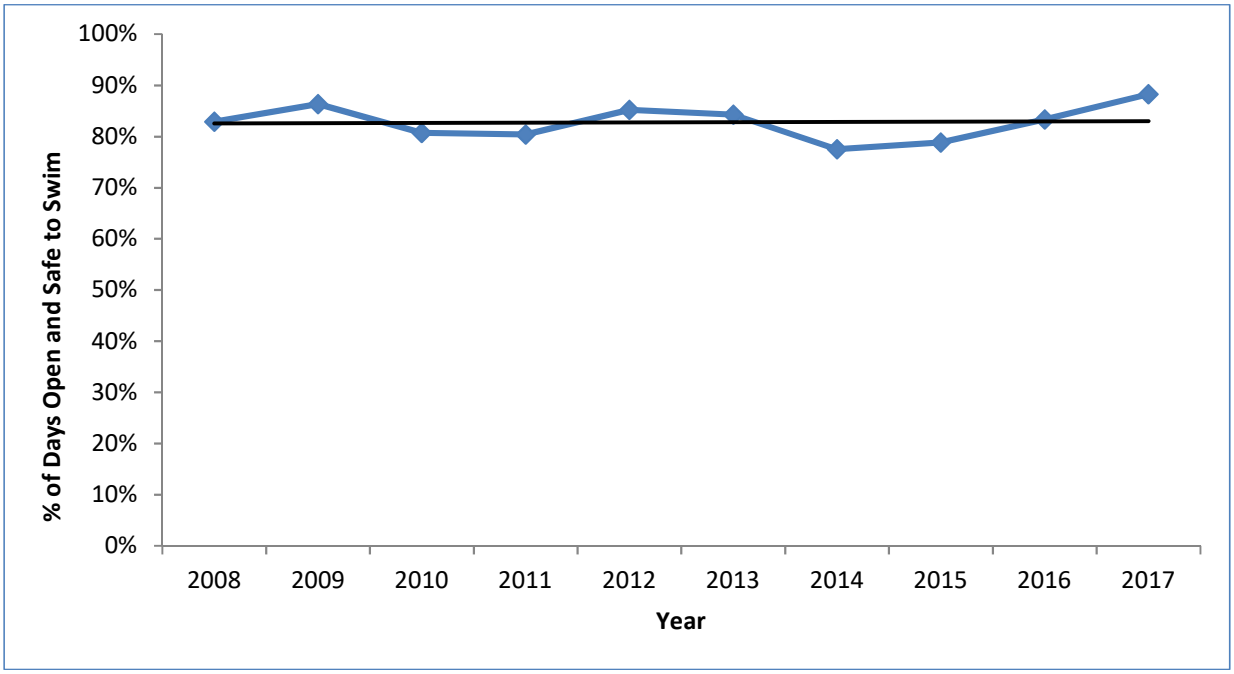


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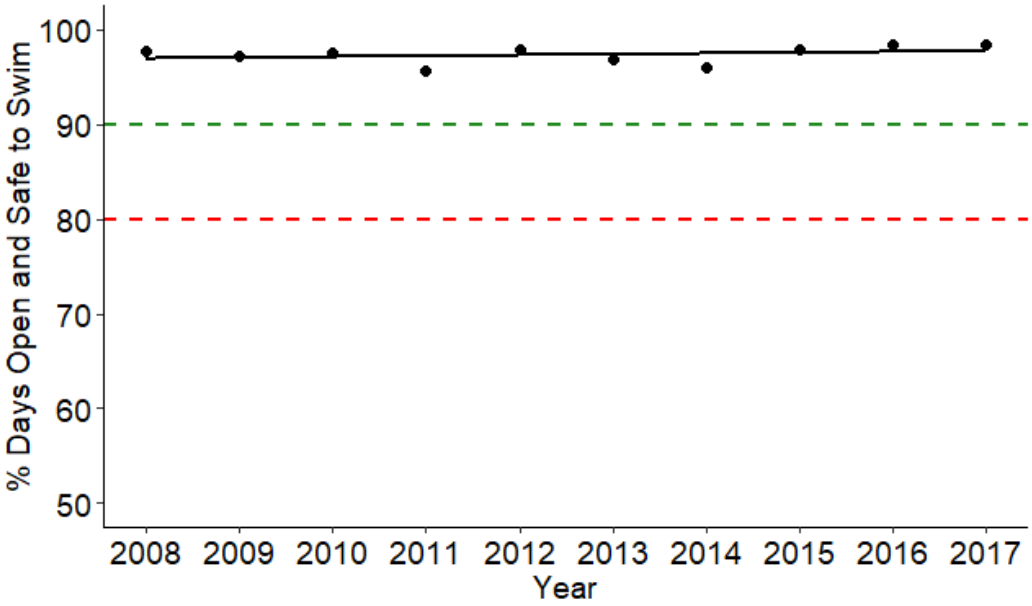


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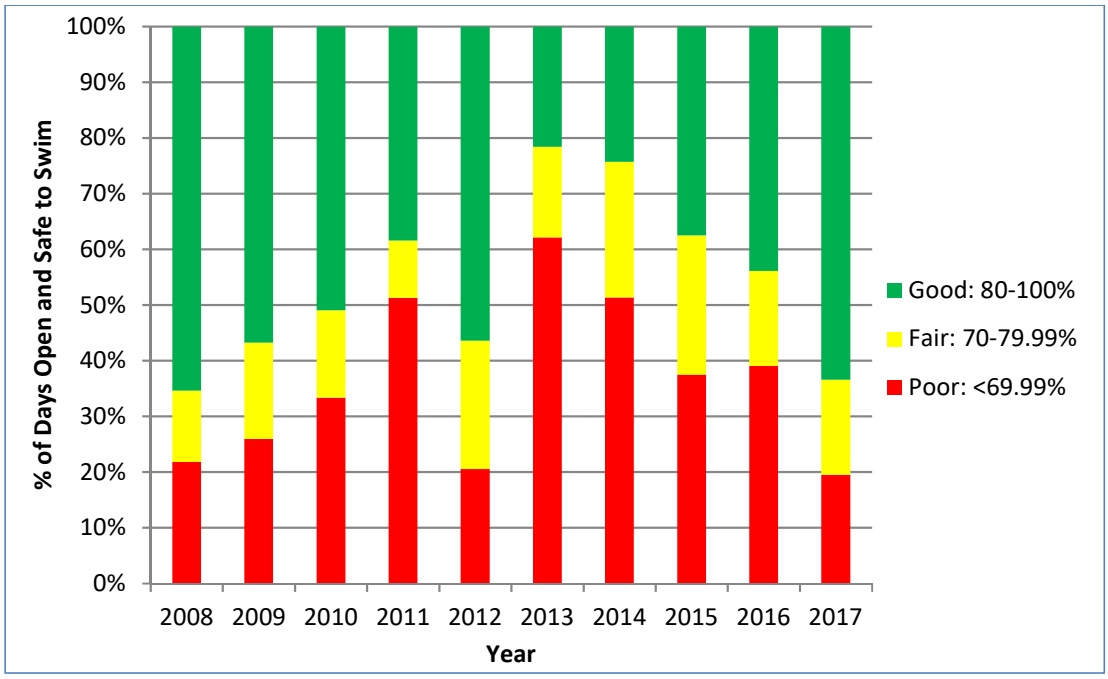


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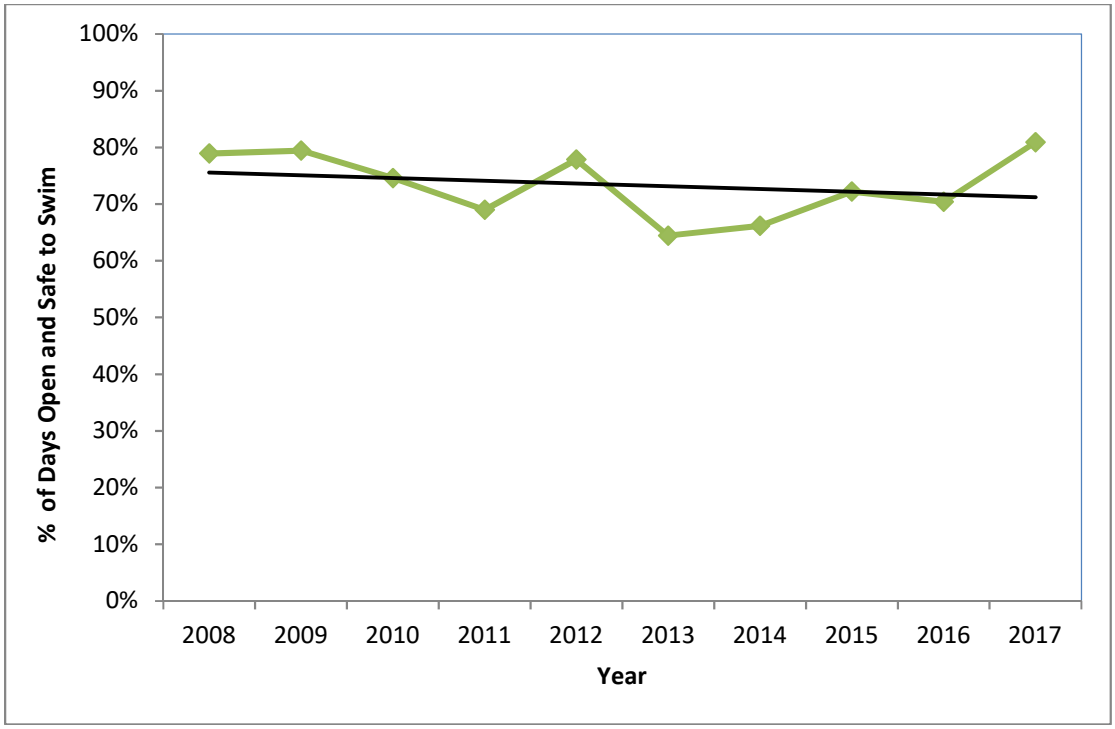


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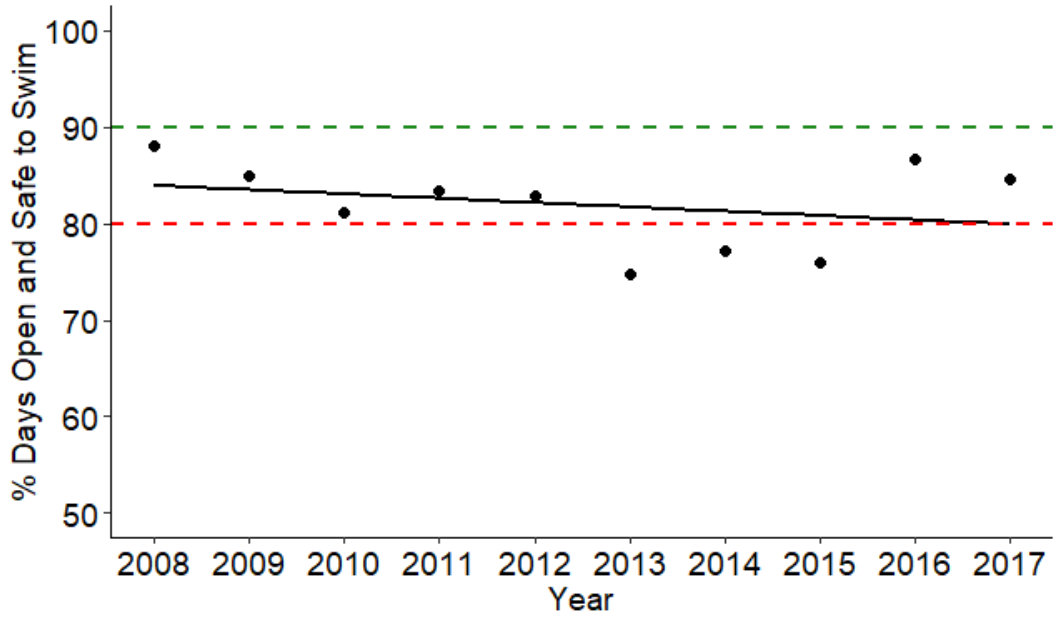


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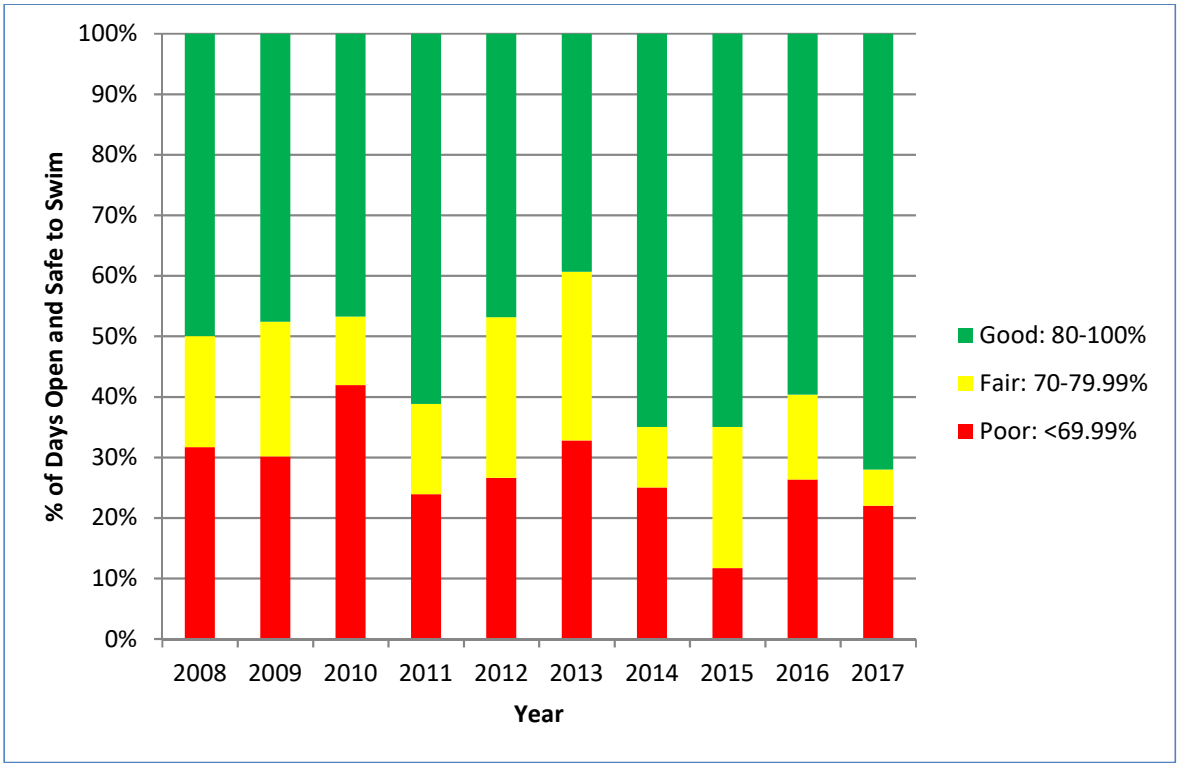


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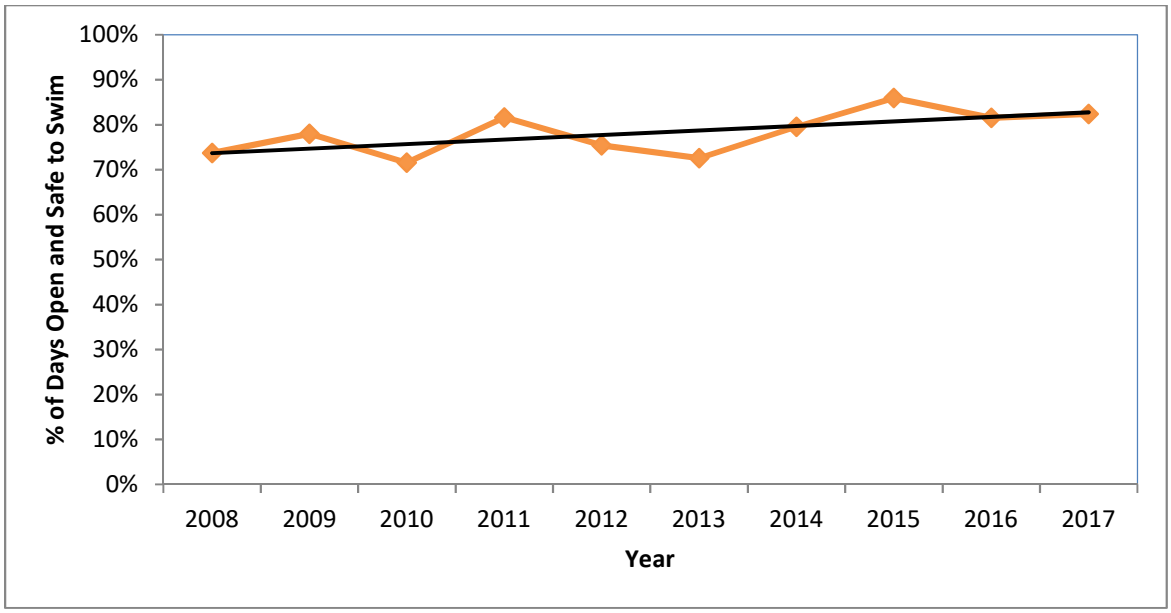


Figure 16. Percentage of days that monitored Canadian Lake Ontario beaches open and safe for swimming during the swimming season from 2008-2017. Good = 80% or >; Fair = 70 to 79.99%; and Poor = <69.99%.
Source: Data collected from Ontario Health Units located along the Great Lakes (see Health units listed in information source section), 2017

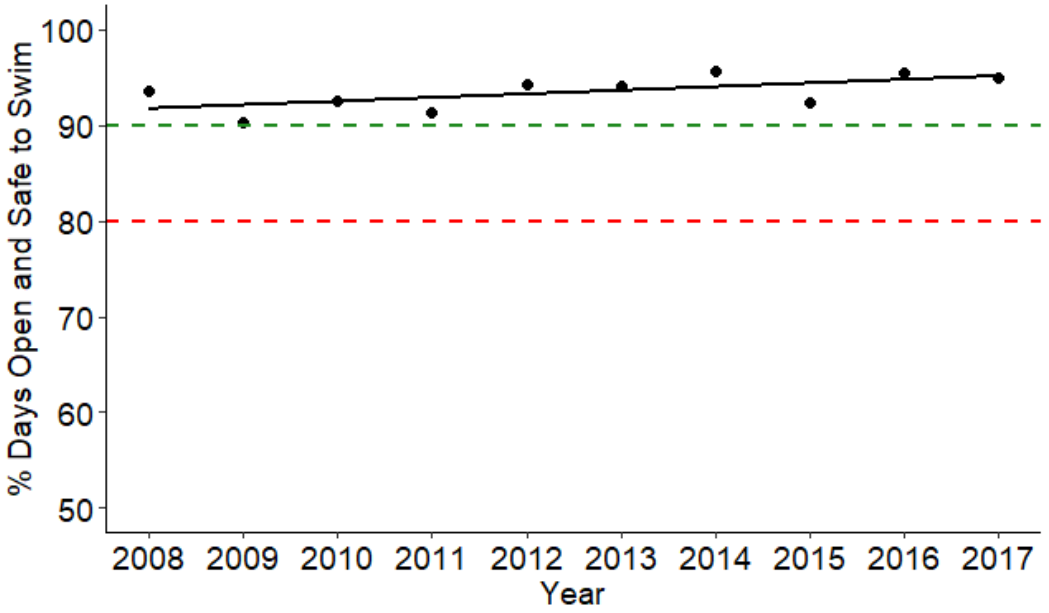


Figure 17. Percentage of days that US Lake Ontario beaches were open and safe for swimming from 2008 to 2017. Good = 90% or >; Fair = 80 to 89.99%; and Poor = <79.99%.
Source: Data collected from US states and reported to US EPA's Beach Advisory and Closing Online Notification (BEACON) system. Data accessed March-April 2019.

Sub-Indicator: Contaminants in Edible Fish

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Unchanging
Long-term Trend (1975-2016): Improving
Rationale: Polychlorinated biphenyls (PCBs) are responsible for the majority of fish consumption advice for all of the Great Lakes. PCB levels appear to have declined in Lakes Ontario and Michigan fish in recent years; however, they appear to be stable in Lake Erie, Lake Huron and Lake Superior fish over the 10-year period. Overall, PCB levels in edible portion of Great Lakes fish have declined substantially (more than 90% in some cases) since its ban in the late 1970s. However, their current levels are still above certain advisory benchmarks. Depending on fish species and lake, mercury levels are declining or stable; and would result in restrictions on eating fish. Recent levels of any other legacy contaminant or known contaminants of emerging concern (CECs) typically do not result in restrictions on fish consumption. Since the last report, the most recent PCB measurements for Lakes Michigan and Ontario, where highest levels among the Great Lakes were observed in the past, are lower and support continued improvements. Lake Huron PCB levels also declined since the last reporting cycle in 2017. Lake Superior and Lake Erie PCB levels appear to be stable but Lake Erie levels now appear to be higher than the other Great Lakes.

Status and trend assessments are based on data generated by the Ontario Ministry of the Environment, Conservation and Parks for Lakes Superior, Huron, Erie and Ontario. Lake Michigan status and trend assessments were generated based on data provided by the States of Michigan, Wisconsin, Illinois, and Indiana. Reliance upon a single data source, such as the Province, allows for more consistent comparison between reporting cycles but may not be reflective of status and/or trends at the State level. Identification and inclusion of additional U.S. State data will be a priority for future reporting.

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: Unchanging
Long-term Trend (1975-2016): Improving
Rationale: There have been substantial improvements in PCB and mercury levels in Lake Superior fish over the last 40+ years; however, the levels appear to be unchanging since the 1990s. About 44% of the advisories for both general and sensitive population (women of child-bearing age and children) for the Canadian waters of the Great Lakes still have restrictions of <8 meals per month. The recent measurements (since the *State of the Great Lakes 2017* Technical Report (SOGL 17 TR)) are undetermined for PCB and mercury levels in the recent years. Toxaphene is still present in some Lake Superior fish and causes a few restrictions of between 1-2 meals per year and 4 meals per month.

Lake Michigan
Status: Fair
10-Year Trend: Improving
Long-term Trend (1975-2016): Improving
Rationale: Historically, PCB levels in fish from Lake Michigan were the worst among the Great Lakes. However, substantial declines over time have resulted in levels close to or, in some cases, lower than the other lakes in more recent years. Both PCBs and mercury appear to be mostly declining and continue to be the contaminants most responsible for driving fish consumption advisories. Since the 2017 SOGL report, continued improvements in both PCB and mercury are evident from the most recent measurements.

Lake Huron
Status: Fair

10-Year Trend: Unchanging
Long-term Trend (1976-2016): Improving
Rationale: Lake Huron fish experienced declines in both PCBs and mercury during the 1970s and 1980s; however, levels of mercury and, to a certain extent, PCBs appear to be stable since that time. Disturbances in the food web structure of Lake Huron due to invasive species may have contributed to the slower declines in the recent years. About 46% of the advisories for both general and sensitive population for the Canadian waters of the Great Lakes are still restrictive (<8 meals per month). Although levels of mercury and PCBs have been more-or-less stable over the last two decades, it was encouraging to observe lower PCB levels in the most recent measurement since the 2017 report; however, mercury levels were either stable or slightly higher.

Lake Erie
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1976-2015): Improving-Unchanging
Rationale: Concentrations of PCBs and mercury have been historically lower in Lake Erie fish compared to the other Great Lakes. Trends in PCBs and mercury concentrations are generally unchanging over the last ten years. The long-term trend for PCB is improving. The long-term trend for mercury is unchanging, due to slightly increased concentrations in mercury since the mid-1990s. Alteration of the Lake Erie food web by invasive species, such as dreissenid mussels and round goby, have likely impacted the contaminant levels in Lake Erie fish. About 60% of the advisories for both general and sensitive population for the Canadian waters of the Great Lakes still have restrictions of <8 meals per month.

Lake Ontario
Status: Fair
10-Year Trend: Improving
Long-term Trend (1975-2016): Improving
Rationale: Similar to Lake Michigan, PCB levels in fish from Lake Ontario were historically among the worst in the Great Lakes. However, substantial declines (more than 90% in many cases) over time have resulted in almost similar or, in some cases, lower levels compared to the other lakes more recently. Both PCBs and mercury appear to be mostly declining. About 62% of the advisories for both general and sensitive population for the Canadian waters of the Great Lakes still have restrictions of <8 meals per month. The most recent mercury levels since the SOGL 2017 report appear to be similar to the observations during the last 5 years, while the PCB levels demonstrated further improvements.

Status Assessment Definitions
Good: Concentrations of contaminants of concern in fish fillets allow for consumption of 8 or more meals per month.
Fair: Concentrations of contaminants of concern in fish fillets allow for consumption of between 1 and 4 meals per month.
Poor: Concentrations of contaminants of concern in fish fillets allow for consumption of less than 1 meal per month.
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions
Improving: Concentrations of contaminants of concern are declining with statistical significance.
Unchanging: Concentrations of contaminants of concern are not declining or increasing with sufficient statistical power to detect a 5% annual rate of change 80% of the time.
Deteriorating: Concentrations of contaminants of concern are increasing with statistical significance.
Undetermined: Concentrations of contaminants of concern do not indicate a clear overall trend and/or there is insufficient statistical power to detect a 5% annual rate of change.

Endpoints and/or Targets
An assessment of compounds of interest in fish fillet 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 will include trend evaluation of contaminants of concern (e.g. PCBs and mercury) in fish fillets. A declining trend for these compounds is representative of an improvement in the environmental condition and the potential for reduced exposure to contaminants from consumption of Great Lakes fish. The elimination of restrictive fish consumption advisories in the Great Lakes may also be considered to be an appropriate endpoint.

Sub-Indicator Purpose
The purpose of this sub-indicator is to assess levels of compounds that pose a risk to human health and infer the potential harm through the consumption of Great Lakes fish. Special emphasis will be paid to compounds that are incorporated into fish consumption advisories such as persistent, bioaccumulative, and toxic (PBT) compounds including mercury and PCBs in fillets of Great Lakes fish.

This 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 originated 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 levels that result in consumption advice. The outcome is then used to relate the ecosystem status to General Objective #3 of the GLWQA.

Measure
Concentrations of contaminants of concern (e.g., PCBs, mercury) in fish fillet of species most consumed by Great Lakes basin citizens are used as an indicator of exposure to these compounds. Temporal trends of compounds and assessment of the advisory levels associated with these compounds 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 Ontario Ministry of the Environment, Conservation and Parks (OMECP). US samples are collected according to the Great Lakes State programs. Levels and trends of PBT chemicals in the most edible portion of fish (i.e., fillets) are assessed and compared against advice categories identified in the Guide to Eating Ontario Fish and the Protocol for a Uniform Great Lakes Sport Fish Consumption Advisory.

Since the 1970s, there have been declines 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 a 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 that they can be eliminated from regular monitoring to prioritize resources for other purposes such as monitoring CECs (Gandhi et. al 2014, 2015). Because PCBs and mercury are currently the most limiting compounds in eating Great Lakes fish, the assessment is largely evaluated based on findings of PCBs and mercury. PCB concentrations are high enough that they are the predominant chemical driving the fish consumption advice. When providing advice due to PCBs 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 compounds 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 fish fillets 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 compounds (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 the widest temporal and spatial scale for results. 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 Ministry of the Environment, Conservation and Parks (MECP) collect the fish, which are then analyzed by MECP 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 2015-2016 edition of 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 2,300 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. All data from 2010 has been published and 2015 data has been shared with State collaborators. Publications for these data are expected shortly. This study is being considered to be repeated in 2020.

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. 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 Ontario Ministry of the Environment, Conservation and Parks, Canada, 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 2000 ng/g 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 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 <500 ng/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 200 ng/g, 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 the Great Lakes fish.

Although PCBs appear to be declining in many cases, especially for Lakes Ontario and Michigan where fish PCB levels were historically the highest, a stable trend is emerging for Lakes Erie and Superior. Although 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 stable trends, no substantial declines in PCB for Lakes Erie, Huron and Superior have occurred since the late 1980s.

Mercury

Mercury concentrations in fish historically exceeded fish consumption advisory benchmarks more frequently (Figure 2). The levels have generally declined over the last four decades, and the concentrations in the selected sizes of fish from the five Great Lakes are now below 0.2-0.3 µg/g (Figure 2). These levels would allow 4-8 meals per month consumption for the sensitive population (Table 1).

Walleye can be considered among the best indicator species for mercury. The monitoring data show that the mercury levels continue to decline in Lakes Ontario and Michigan. However, similar to PCB, the levels appear to be stable for Lakes Erie and Huron. A stabilizing trend for mercury in Lake Superior fish is also evident from the monitoring data collected for Lake Trout and Chinook Salmon.

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. Province of Ontario, and States of Minnesota and Michigan are investigating the sources of perfluoroalkyl acids in fish, and have site-specific advice for where fish have been tested for elevated perfluorooctane sulfonic acid (PFOS), a chemical of the PFAS family. Similarly, elevated levels of PFOS were found in fish at 11 inland locations in Ontario, for which site-specific advice has been issued. 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). However, concentrations of PFOS in common Great Lakes fish species have not resulted in advisories that would be more restrictive compared to those due to PCBs or mercury (Gandhi et al. 2017b). For this reason, a detailed assessment was not conducted for PFASs this reporting cycle, but shall be considered in the future reports as necessary.

Toxaphene and Mirex

Previous reporting cycles identified that 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. For this reason, 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 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 are evidences 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 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 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.

Future reporting of this sub-indicator will continue to focus on beneficial compounds, such as fatty acids in species of fish most consumed by Great Lakes citizens and may allow for comparison of risks and benefits of fish consumption and tracking of concentrations of these compounds over time. As noted in the Guide to Eating Ontario Fish 2015-2016, "it is clear that fish consumption can present both benefits and risks. So the real question is: do the benefits of eating fish outweigh the risks to our health? Well, it depends. This is because various factors such as the contaminant of concern, its level in fish, and the levels of various nutrients (e.g., Omega-3 fats, vitamins) in fish vary widely from one fish species and size to another, and are location-specific. Although scientific studies have begun to evaluate the health benefits against the risks of eating contaminated fish, our understanding is still very 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 advice in this guide continues to be based only on contaminant risk…" (from http://www.ontario.ca/environment-and-energy/guide-eating-ontario-fish). 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 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 appears 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 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 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.

Additional Information

At present, PCBs, mercury, and chlordane are the only PBT chemicals that have uniform fish advisory protocols across the U.S. Great Lakes Basin. The U.S. Great Lakes Sport Fish Advisory Task Force is currently drafting additional uniform PBT advisories in order to limit confusing the public that can result from issuing varying advisories for the same species of fish across the basin.

There are differences in the way fish samples are analyzed for contaminants and consumption advisories are developed in the U.S. and Canada. This means that the data and 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 sport fish fillets, 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 first year of this program was in 2011, and results are being shared through various outlets, including State of the Great Lakes reporting, as they are received.

The Ontario Ministry of the Environment, Conservation and Parks continues to monitor contaminants of long term concern such as PCBs, 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
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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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Baline Snyder, TetraTech
Chip McCarty, Computer Sciences Corporation
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Contacts for State Data included in this sub-indicator
Brian Koch, Illinois Environmental Protection Agency
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Joe Bohr, Michigan Department of Environmental Quality

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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)

List of Figures

Figure 1. Total PCB concentrations (ng/g) in five fish species from the Great Lakes observed between 1975 and 2016 with focus on 2000 onwards. Lake Michigan measurements were for skin-on fillets, while skin-removed fillets for the other lakes. Dashed red lines represent an approximate binational health related benchmark for the general population.
Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Environmental Protection Agency

Figure 2. Total mercury concentrations (µg/g) in five fish species from the Great Lakes observed between 1975 and 2016. Lake Michigan measurements were for skin-on fillets, while skin-removed fillets for the other lakes. Red and green dashed lines represent approximate binational health related benchmarks for the general and sensitive populations, respectively.
Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Environmental Protection Agency

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

Last Updated

State of the Great Lakes 2019 Technical Report

Meals per month	PCB (ug/g)			Hg (ug/g)		
	Sensitive*	Sensitive**	General**	Sensitive*	Sensitive**	General**
32	0 - .05	<.026	<.026	0 <=.05	<0.06	<0.15
16		.026-.053	.026-.053		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

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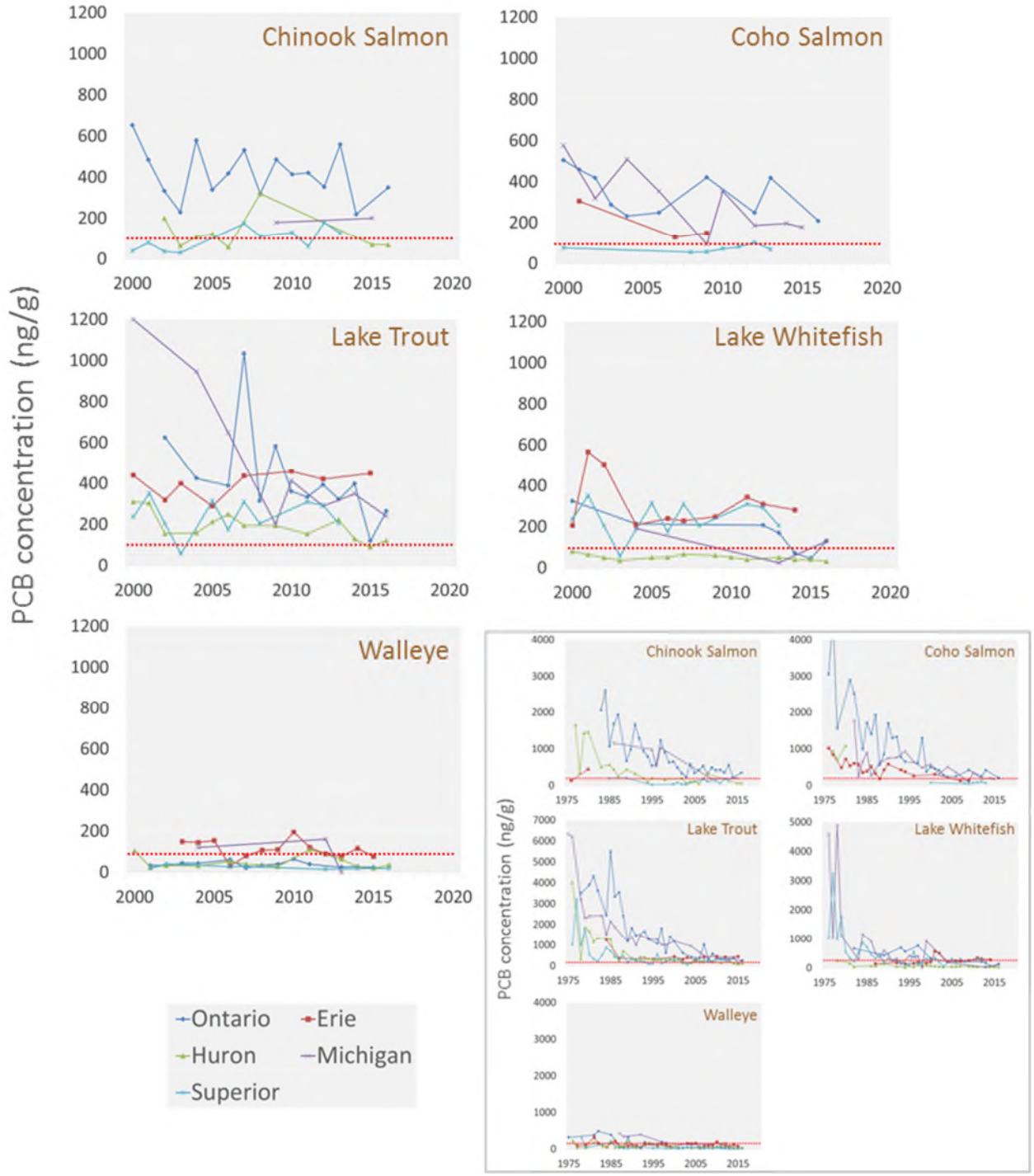


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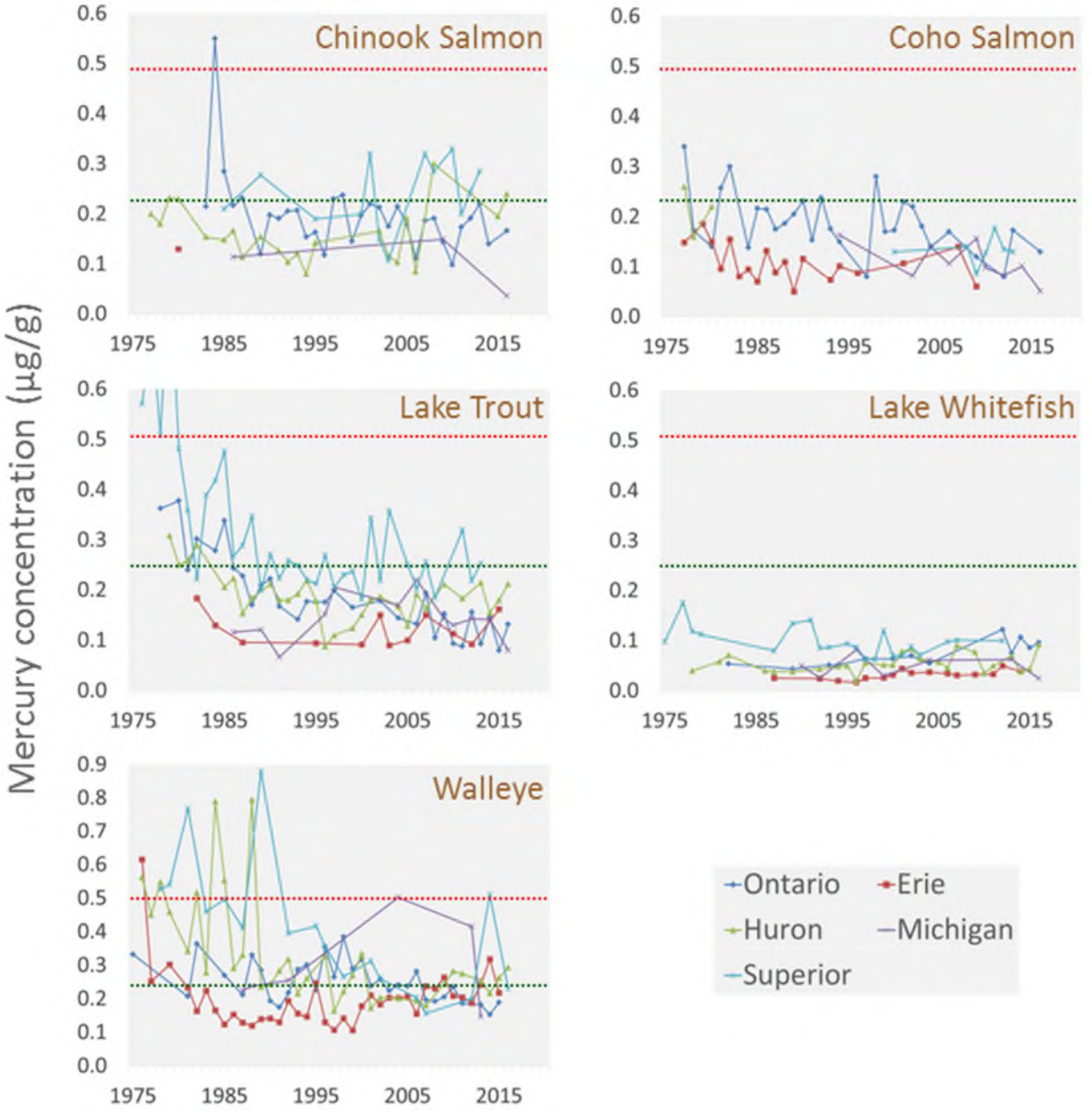


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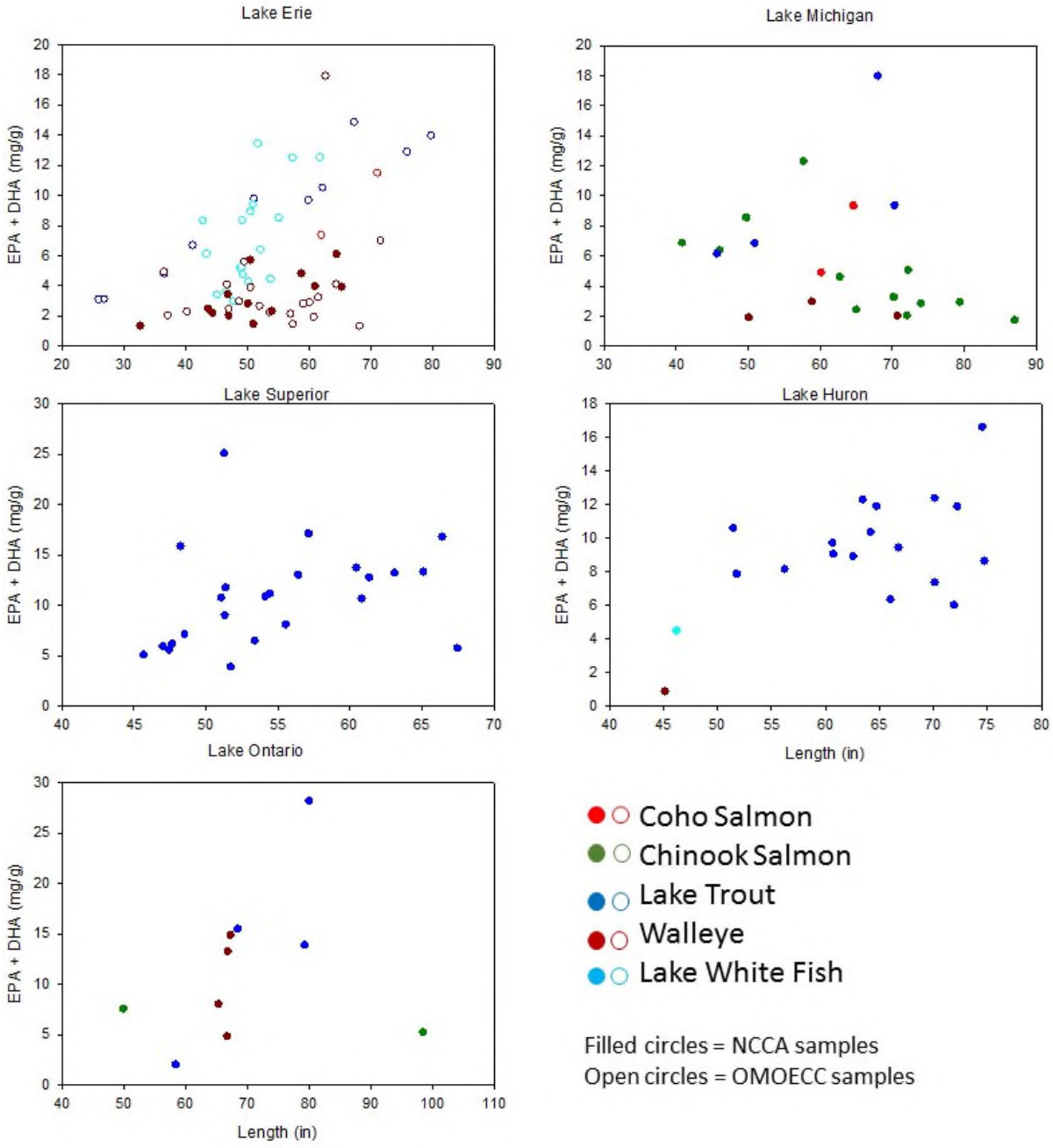


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Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Environmental Protection Agency

Sub-Indicator: Toxic Chemicals in Sediment

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 on 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 (previously knowns as SOLEC) 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 centres 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 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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List of Tables

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

List of Figures

Figure 1. Spatial distribution of mercury contamination in surface sediments of open-lake areas and connecting channels of the Great Lakes. Sources: 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 acid**s** (PFSAs) and Perfluorooctane sulfonic acid in Great lakes sediment.
Data Source: Environment and Climate Change Canada

Last Updated

State of the Great Lakes 2019 Technical Report

	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 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)

% 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 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.

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.

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

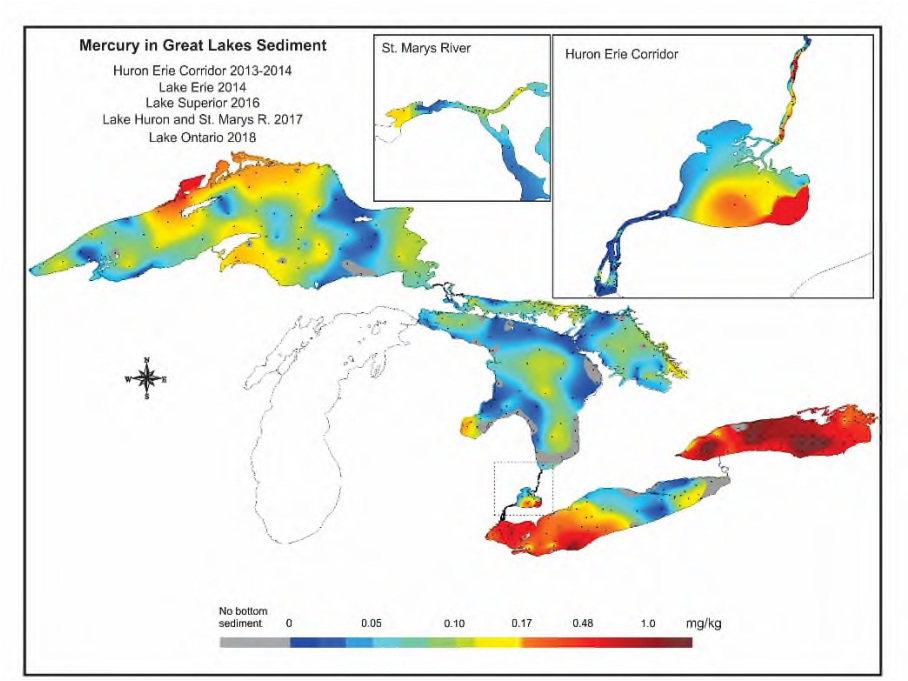


Figure 1. Spatial distribution of mercury contamination in surface sediments of open-lake areas and connecting channels of the Great Lakes.
Source: Environment and Climate Change Canada

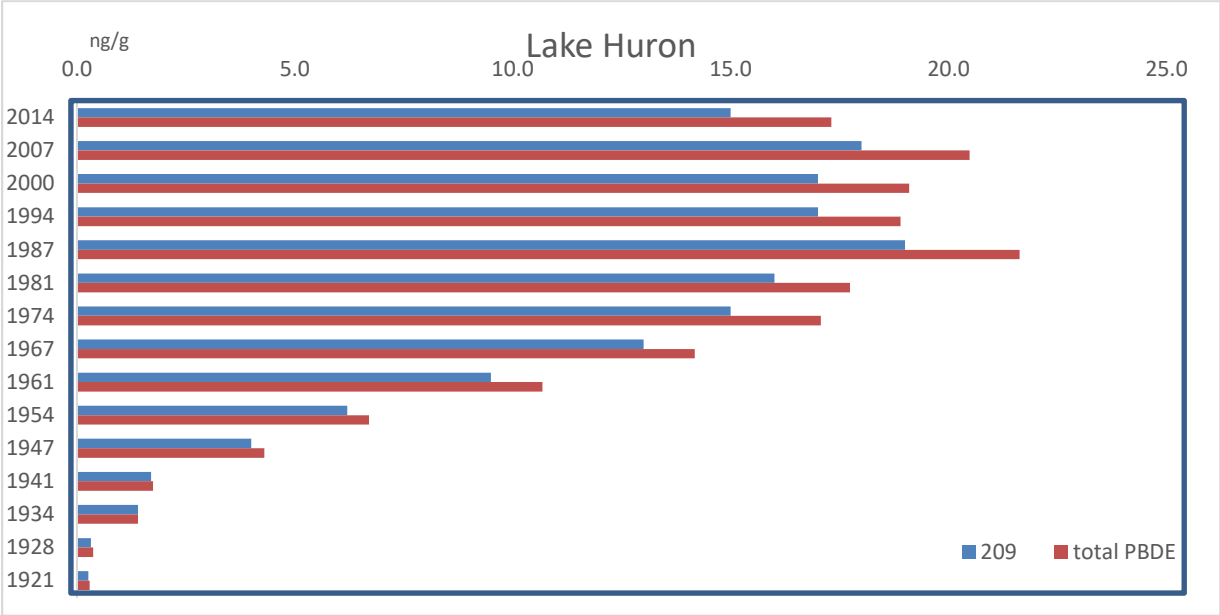


Figure 2. Temporal trends of polybrominated diphenyl ethers in Lake Huron sediment.
Data Source: Environment and Climate Change Canada

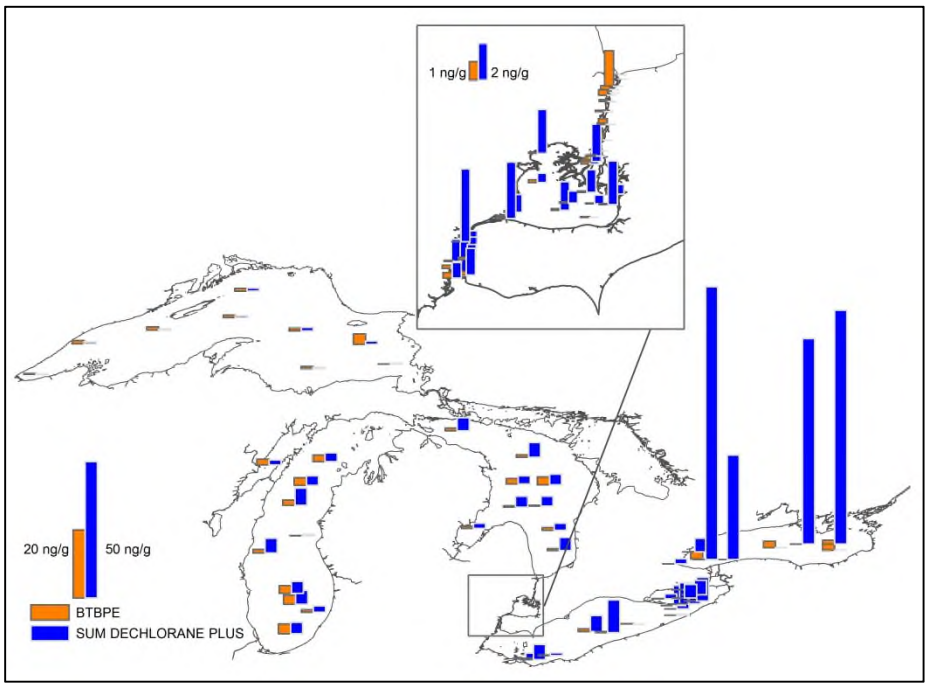


Figure 3.Spatial distribution of BTBPE and dechlorane plus (sum of syn and anti) in Great Lake sediment (sampled in 2010-2014).
Source: Lakes Superior, Michigan and Huron-Guo (2015); St. Clair River, Lake St. Clair, Detroit R. and Lake Erie - Environment and Climate Change Canada; Lake Ontario -Yang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

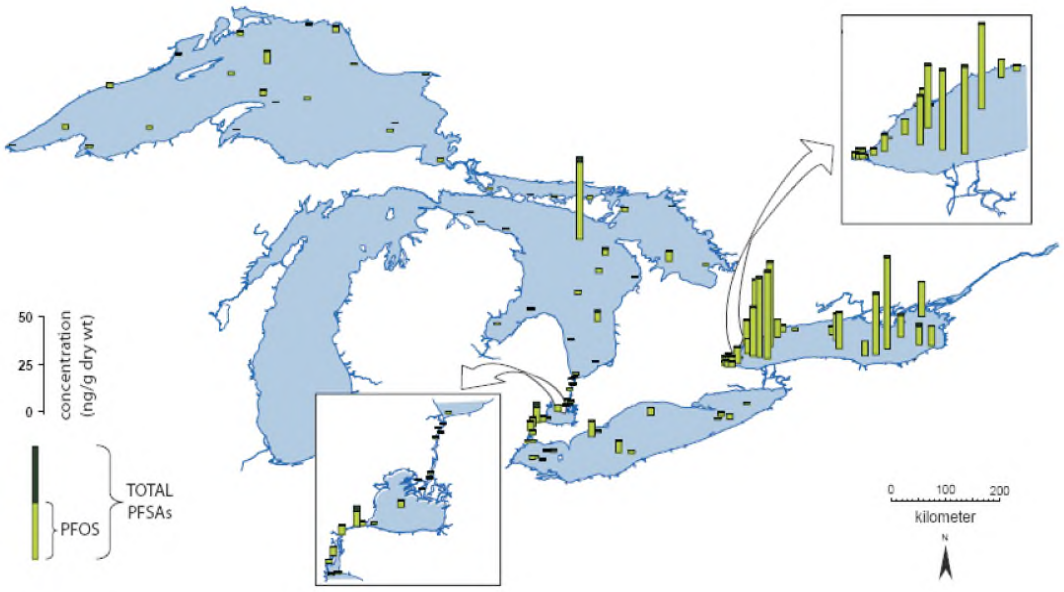


Figure 4. Total Perfluorinated sulfonic acids (PFSAs) and Perfluorooctane sulfonic acid in Great lakes sediment.
Data Source: Environment and Climate Change Canada

Sub-Indicator: Toxic Chemicals in Water

(Previously entitled Toxic Chemical Concentrations)

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			
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.				

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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List of Tables

Table 1. Monitoring of Chemicals of Mutual Concern in Great Lakes waters. The most recent year(s) of monitoring is indicated.
Source: Environment and Climate Change Canada

Table 2. Legacy organic contaminants monitored in Great Lakes surface water for Environment and Climate Change Canada's Great Lakes Surveillance. Parameters are monitored during spring cruises from dissolved (filtered) large volume (16 – 24 L) samples using clean techniques. Those detected in more than 10% of samples are indicated with an "×".
Source: Environment and Climate Change Canada

List of Figures

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

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.
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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

Last Updated

State of the Great Lakes 2019 Technical Report

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	Georgian 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

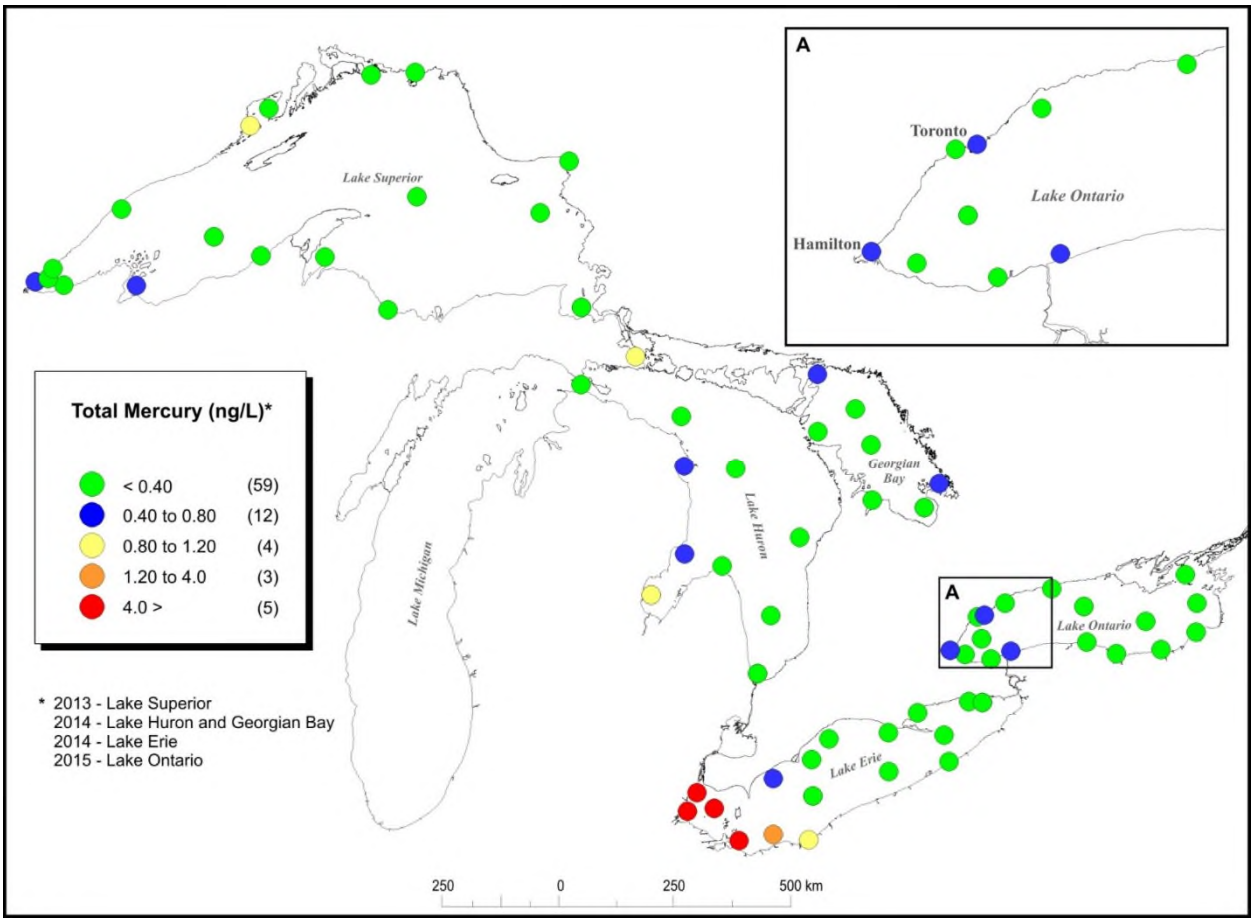


Figure 1. Spatial distribution of total mercury in the Great Lakes. Data are the most recent available spring, surface data for all stations. * = years of data included for total mercury analysis.
Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance Program

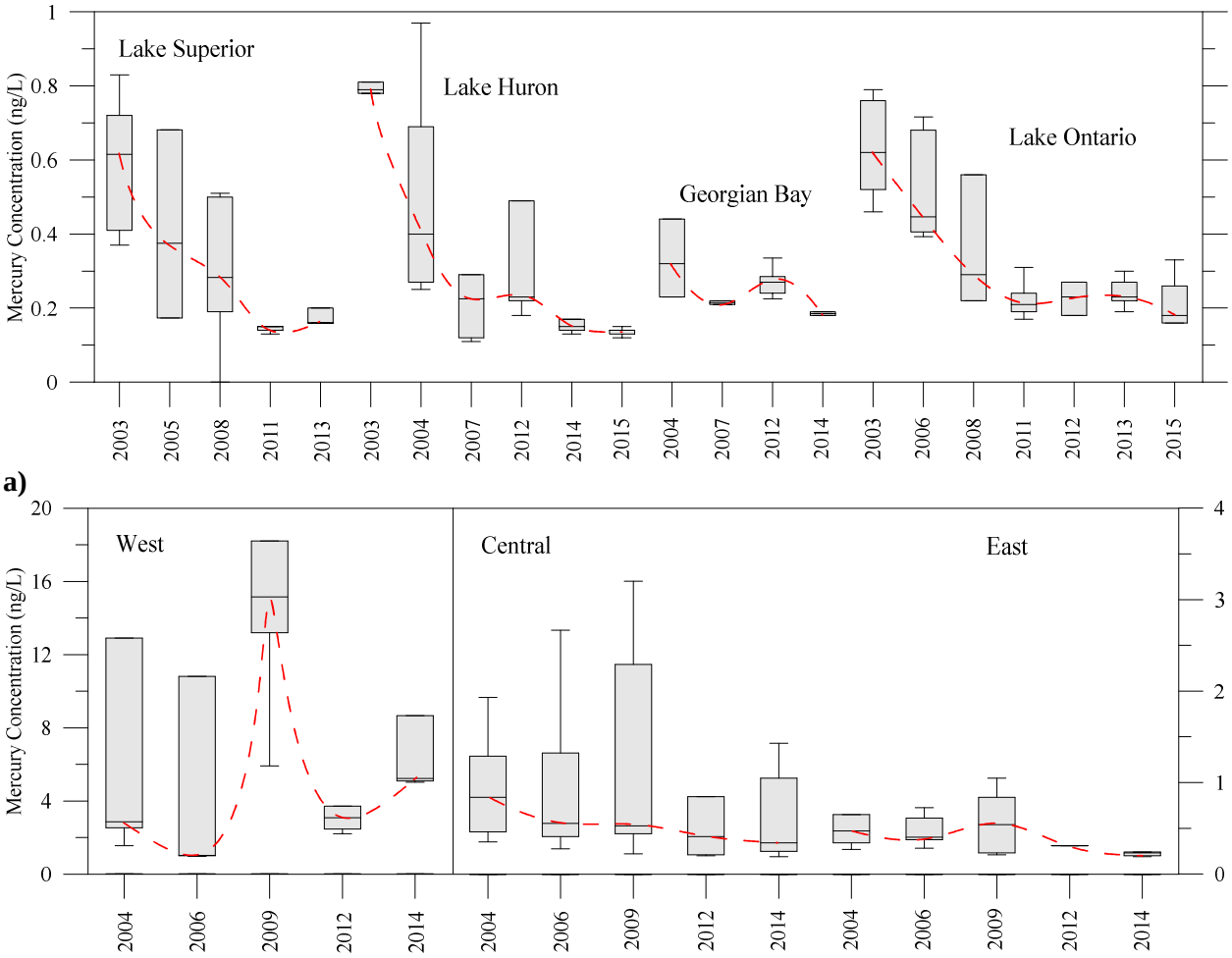


Figure 2. Temporal changes of total mercury in the Great Lakes. Data are a) Great Lakes spring, surface data from offshore stations and b) Lake Erie spring, surface data from all stations by basin. Lake Erie west basin data for are scaled using the left-hand vertical axis and central and east basin data 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

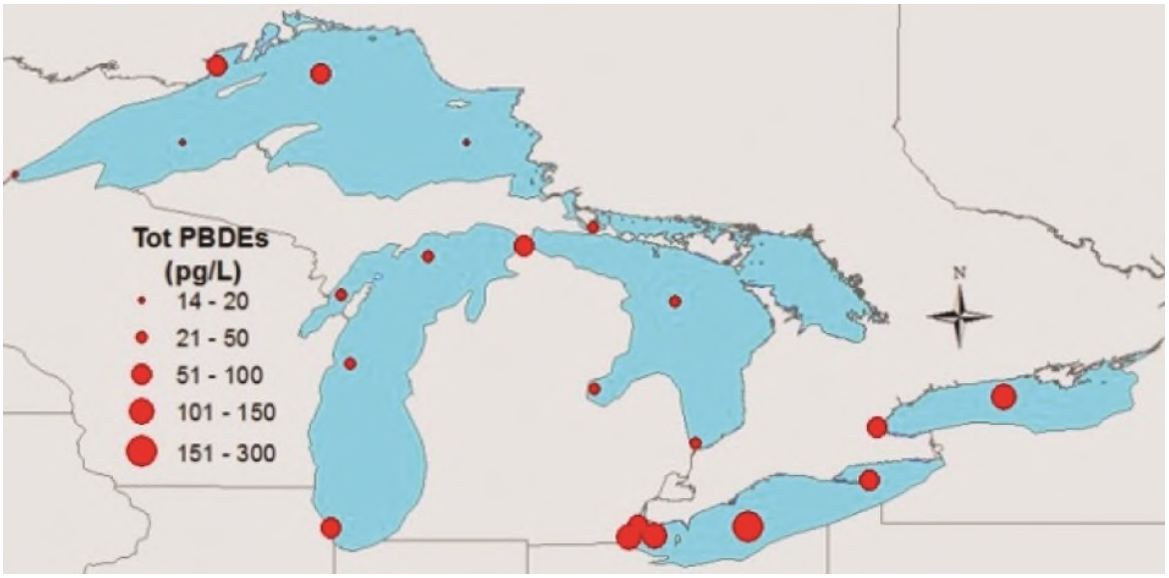


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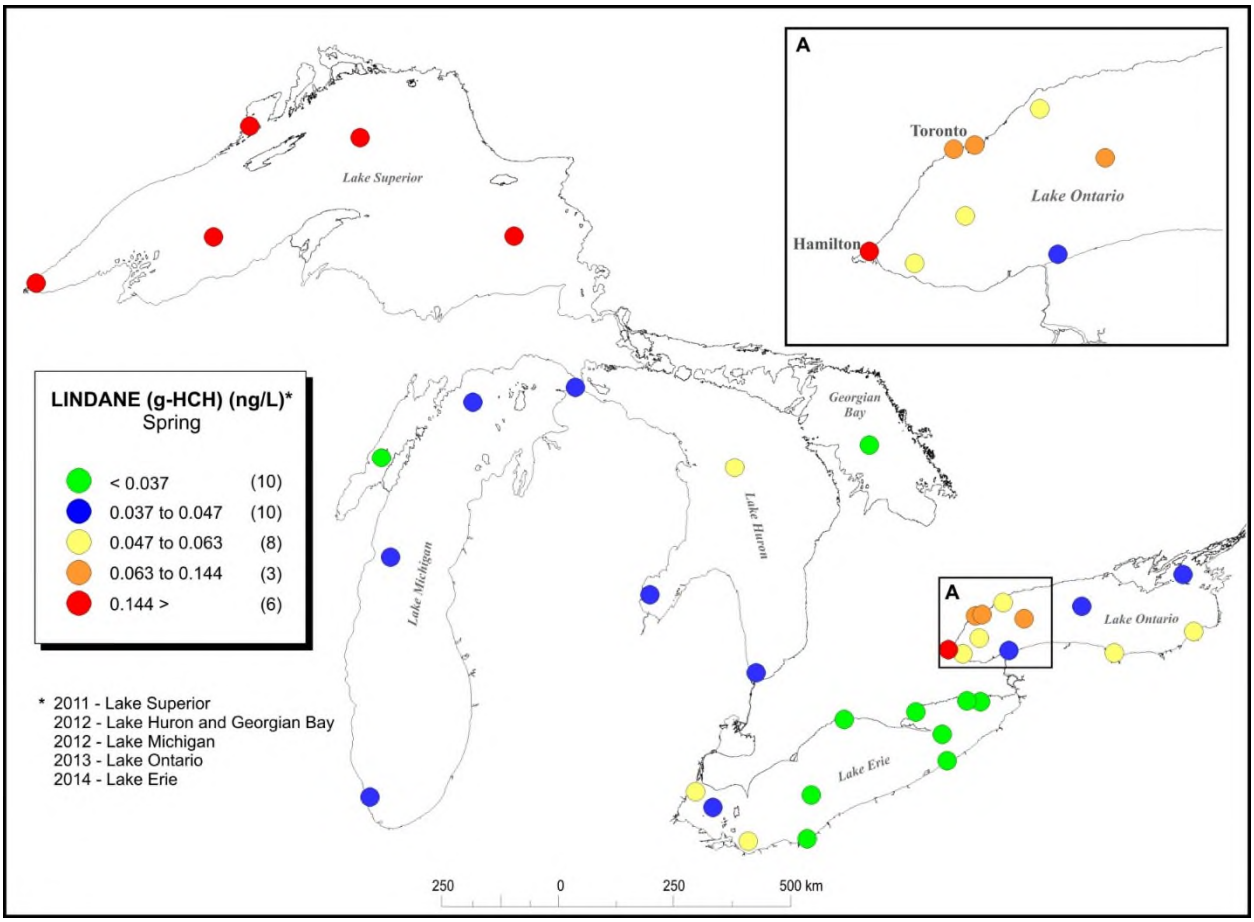


Figure 4. Spatial distribution of dissolved lindane (gamma hexachlorocyclohexane) in the Great Lakes. Data are the most recent available spring, surface data for all stations. * = years of data included for lindane analysis.
Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance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

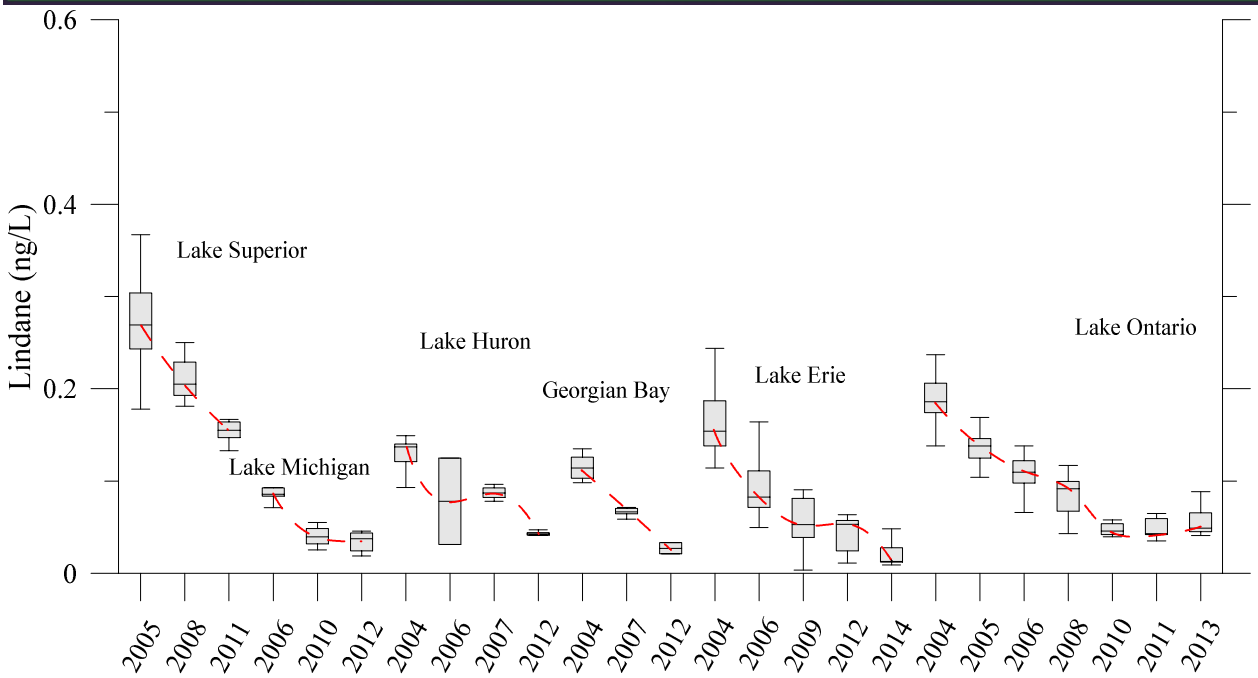


Figure 5. Temporal changes of dissolved lindane in the Great Lakes. Data are spring, surface data at offshore stations. Boxes show central median and 25% and 75% values, whiskers show 1.5x interquartile range.
Source: Data are from Environment and Climate Change Canada's Great Lakes Surveillance Program

Sub-Indicator: Toxic Chemicals in Whole Fish

Overall Assessment
Status: Fair
Trends:
10-Year Trend (2007 – 2016): Unchanging
Long-term Trend (1977 – 2016): Improving
Rationale: While concentrations of some Chemicals of Mutual Concern (CMCs) continue to exceed environmental guidelines or targets, on average the levels of CMCs are near their guidelines and the average MDR for CMCs in whole fish remains in the yellow zone of the Mean Deviation Ratio (MDR) chart, meaning it is neither greater or less than zero. The overall status of the Great Lakes basin is assessed as "Fair". 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 is "Unchanging". See Ecological Condition section for further explanation on the selection and purpose of CMC reporting. Improvements to the MDR method have been made since the last report, including changes to the chemicals included in the MDR (removal of DDT and addition of HBCDD/HBCD) and slight changes to the method used to calculate the MDR. Therefore, direct comparison of the MDR since the last report at the Great Lakes and lake basin scale, is not possible and will not be discussed in the lake by lake assessments. PCBs, PeBDE, and PFOS are above environmental quality guidelines or targets across the Great Lakes basin.

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

Lake-by-Lake Assessment

Lake Superior
Status: Fair
Trends:
10-Year Trend (2007 – 2016): Unchanging
Long-Term Trend (1977 – 2016): 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 (PCBs, PeBDE, and PFOS) are still above environmental guidelines or targets (Table 1) in Lake Superior. The MDR falls within the yellow zone of the MDR chart indicating that is not greater or less than zero and that long term overall conditions are "Fair". Despite the increase in the MDR around 2000, which was a result of the introduction of PBDEs into our monitoring programs, conditions are "Improving" since monitoring began in 1977. A statistically significant trend was not present in the last 10-years of monitoring indicating that the condition is "Unchanging".

Lake Michigan
Status: Fair
Trends:
10-Year Trend (2007 – 2016): Unchanging
Long-Term Trend (1983 – 2016): Improving
Rationale: In Lake Michigan, the MDR indicates that on average the concentrations of CMCs are above environmental targets or guidelines (Table 1). PCBs, PeBDE, and HxBDE are above guidelines or targets while those of mercury, TeBDE, and HBCD are below environmental guidelines or targets. The MDR report falls within the yellow zone of the MDR chart indicating that is not greater or less than zero and that overall conditions are "Fair". Since monitoring began, conditions are "Improving" 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 is "Unchanging".

Lake Huron
Status: Fair
Trends:
10-Year Trend (2007 – 2016): Unchanging
Long-Term Trend (1977 – 2016): 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, and HBCD concentrations are below guidelines or targets while those of PCBs, HxBDE, PeBDE, and PFOS were above environmental guidelines or targets. The MDR falls within the yellow zone of the MDR chart indicating that is not greater or less than zero and that overall conditions are "Fair". Since monitoring began, conditions are "Improving" 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 is "Unchanging".

Lake Erie
Status: Fair
Trends:
10-Year Trend (2007 – 2016): Unchanging
Long-Term Trend (1977 – 2016): 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 falls within the yellow zone of the MDR chart indicating that is not greater or less than zero and that overall conditions are "Fair". Since monitoring began, conditions are "Improving" 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 is "Unchanging".

Lake Ontario
Status: Fair
Trends:
10-Year Trend (2007 – 2016): Unchanging
Long-Term Trend (1977 – 2016): 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 falls within the yellow zone of the MDR chart indicating that is not greater or less than zero and that overall conditions are FAIR. Since monitoring began, conditions are "Improving" 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 is "Unchanging".

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

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.

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 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 Chemicals of Mutual Concern (CMC), from multiple species, in all 5 of the Great Lakes over time. The Mean Deviation Ratio (MDR) has been refined from the previous reporting cycle and is used to determine an overall assessment of status and trend using 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 will be 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. 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. 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 (by age or size cohort) to determine mean and variance for each 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 > 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 < zero indicate that conditions are below the targets or guidelines and interpreted as "Good" condition. The MDR is calculated as presented in equation 1:

$$(1) \qquad MDR = \frac{\sum_{j=1}^{P}\left(\frac{\sum_{i=1}^{n}\log(\frac{Ci}{T})}{n}\right)_j}{P}$$

Where, P is the number of parameters measured, n is the number of measurements in a time period for each parameter, C is the measured concentration of each parameter, and T is the target or environmental guideline. 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. 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 representing "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 (Thorsten Pohlert (2018). trend: Non-Parametric Trend Tests and Change-Point Detection. R package version 1.1.0.). 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 (Steffen Moritz 2018).

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 and/or Walleye) 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 Ecosystem Objectives. The *Chemicals of Mutual Concern in Great Lakes 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 represent not just quantities of contaminants 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).

As an example, the GLFMSP recently completed an interlaboratory comparison study of multiple age structures to identify the most appropriate age estimation structure for the Program. The Lake Trout (*Salvelinus namaycush*) maxillae was selected, over the otolith, as the most 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 chemicals 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 in to 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. However, ECCC has identified increasing levels of mercury in in the western basin of Lake Erie. Through the Annex 3 process, new chemicals designated as CMCs will be further incorporated into monitoring

and surveillance programs, when applicable. 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 and PBDEs continue to exceed environmental quality guidelines and/or objectives. 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). As reported in previous indicators, 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 were 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)

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.

Total polychlorinated biphenyls (PCBs)

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 target of 100 ng/g ww in the 1987 amendment to the GLWQA. 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). 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).

Since the last State of the Great Lakes (SOGL) sub-indicator report in 2017, PCBs have only been measured in Rainbow Smelt in Lakes Huron and Superior. Average total PCB concentrations in composited Rainbow Smelt measured by Environment and Climate Change Canada were 12 ng/g ww in Lake Superior (2016) and 18 ng/g ww in Lake Huron (2017). 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 say whether the levels of PCBs in Rainbow Smelt in Lakes Huron and Superior have increased or decreased since the last assessment.

Total Mercury (Hg)

Concentration trends for total mercury in Great Lakes top predator fish have been variable, rate of decline and

statistical significance, 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 300 ng/g 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, and Michigan with slight increases observed in Lakes Huron and Erie with mixed significance (Table 3). Short term concentration trends are mixed without significance in any lake (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 Δ^{200} mercury signatures in predatory fish from the upper Great lakes reveals that bioaccumulated Hg is more atmospherically similar to atmospherically derived mercury than a lake's sediment. The data suggest that in some cases, atmospherically derived Hg may be a more important source of methyl mercury 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 also occurs in the Great Lakes, possibly transforming recently deposited atmospheric mercury deposition. The degree of photochemical processing of Hg is likely controlled by phytoplankton uptake rather than by dissolved organic carbon quantity among lakes (Lepak et al. 2018).

ECCC recently completed an assessment of total mercury levels aquatic birds 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 (*Salvelinus namaycush*), Walleye (*Sander vitreus*), and Rainbow Smelt (*Osmerus mordax*). 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 18). Concentration of mercury in smelt are 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. Mercury was designated as a Chemical of Mutual Concern by the Parties, Canada and the United States of America, through the GLWQA.

Polybrominated Diphenyl Ethers (PBDEs)

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 10). Concentrations of the penta BDE mixture (BDE-99 + 100) in all five Great Lakes continue to remain above the FEQG of 13 ng/g ww (Figure 11). 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 12). 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 mixed 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.

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. PFOS, perfluorocarboxylates and their precursors are used primarily in water, oil, soil, and grease repellents for paper and packaging, carpets, and fabrics, as well as in aqueous film forming foam (AFFF) for fighting fuel fires. PFOS was voluntarily phased-out of production by their primary supplier in 2002. However, PFOS use in Canada and the U.S. continues due to specific use exemptions. 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 13). This trend is likely due to variations in analytical technique as methods continue to be developed and refined. 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). A sufficient number of years of data (>10) was not available for Lake Michigan fish at the time of sub-indicator development and therefore are not included in the overall assessment for this chemical.

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 as a result of it being designated as a CMC. Current and retrospective analyses for this compound was 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 significantly 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 collected 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 ΣHBCDD concentrations compared to locations to the west or east. HBCD has 16 possible stereo-isomers with different biological activities. The greatest mean ΣHBCDD concentration was 72.6 ng/g lw 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 ΣHBCDD increased significantly during 1979 to 2015. The present study provided novel information on the isomer-specific HBCDDs in Canada freshwater fish. (Su et al. 2018).

Short-Chain Chlorinated Paraffins (SCCPs) & Long-Chain-PFCAs
No new data is currently available for either the U.S. or Canadian monitoring and surveillance program. While this class of compounds are of mutual concern, concentrations from previous studies are well below the FEQG for SCPCAs of 2700 ng/g lipid. No status or trend information can be provided on these compounds. However, a summary of peer reviewed literature for these compounds is provided. A previous study, 2015, on the levels of PCAs (Perfluorinated carboxylic acids in fish from Canadian lakes showed that fish from the Great Lakes had higher levels of the medium chain (C14-C17) PCAs (MCPCAs) than short-chain (C10-C13) PCAs (SCPCAs) (Saborido Basconcillo, Backus et al. 2015). Medium chains may be an important component to monitor and/or research in the Great Lakes. The levels of Medium-Chain Perfluorinated carboxylic acids (MCPCAs) 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, SCPCAs 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. 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, thus status and trend statements are not possible at this time. However, the authors do 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)

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 developed to accurately report contaminant trends due to significant Lake Trout age structure changes, 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).

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 ΣPBDE, 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).

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. However, limited knowledge exists on their abundance in the edible portion of a variety of Great Lakes fish to aid in understanding their potential risk to human consumers. Fish from the Canadian Monitoring and Surveillance program were studied for levels, patterns, trends and significance of PCNs. In total, 470 fillet samples of 18 fish species collected from the Canadian waters of the Great Lakes between 2006 and 2013. A limited comparison of fillet and whole-body concentrations in Carp and Bullhead was also conducted. The ΣPCN ranged from 0.006–6.7 ng/gwet weight (ww) and 0.15–190 ng/g lipid weight (lw) with the dominant congeners being PCN-52/60 (34%), -42 (21%) and -66/67 (15%). The concentrations spatially varied in the order of the Detroit River > Lakes Erie > Ontario > Huron > Superior. PCN-66/67 was the dominating congener contributing on average 76–80% of toxic equivalent concentration (TEQPCN). Contribution of TEQPCN to TEQ Total (TEQ Dioxins + Furans + dioxin-like PCBs + PCNs) was mostly < 15%, especially at higher TEQ Total, and PCB-126 remains the major congener contributing to TEQ Total. The congener pattern suggests that impurities in PCB formulations and thereby historical PCB contamination, instead of unintentional releases from industrial thermal processes, could be an important source of PCNs in Great Lakes fish. A limited temporal change analysis indicated declines in the levels of PCN-66/67 between 2006 and 2012, complemented by previously reported decrease in PCNs in Lake Ontario Lake Trout between 1979 and 2004. The whole-body concentrations were 1.4–3.2 fold higher than the corresponding fillets of carp and bullhead. Overall, the study results suggest that only targeted monitoring of PCNs in Great Lakes fish, especially at the Detroit River, Lake Erie and Lake Ontario, is necessary to assess continued future improvements of this group of contaminants of concern (Gewurtz et al. 2018).

PCNs have also been studied in whole fish and herring gull eggs. Spatial and temporal trends (1979–2013) of PCN concentrations were 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, whose effects appear to manifest themselves 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).

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. The present study addresses these knowledge gaps by reporting SDPAs and BZT-UVs in herring gull (*Larus argentatus*) eggs, Lake Trout (*Salvelinus namaycush*), 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. For Herring Gull eggs, the samples from upper Great Lakes contained significantly greater levels of SDPAs than those eggs from lower lakes, possibly due to the differences in terrestrial food in diet. Interestingly, the predominant SDPAs in herring gull eggs were dinonyl- (C9C9) and monononyl-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 also 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 declines of zooplankton populations in Lake Huron are suspected to be the cause for slower growing Lake Trout, resulting in higher chemical concentrations of Persistent Bioaccumulative Toxics. 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.

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 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.

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 or changes in the food web composition 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.

Acknowledgments

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Information Sources

We would like to acknowledge all the State, Tribal, and Provincial partners that contribute samples and information to the monitoring and surveillance programs in both the United States and Canada.

We would also like to acknowledge the Ship's Captains and Crew of the Research Vessel Kelso, Environment and Climate Change Canada, and the Research Vessel Lake Guardian, U.S. EPA, for the participation in sample collection.

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

List of Tables

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
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
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
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

List of Figures

Figure 1. Mean Deviation Ratio (MDR) 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.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 3. Mean Deviation Ratio (MDR) for Lake Michigan.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 4. Mean Deviation Ratio (MDR) for Lake Huron.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 5. Mean Deviation Ratio (MDR) for Lake Erie.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 6. Mean Deviation Ratio (MDR) for Lake Ontario.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

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
Figure 8. Total PCB concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 9. Total mercury concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada, U.S. Environmental Protection Agency
Figure 10. Mean TeBDE concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 11. Mean PeBDE concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 12. Mean HxBDE concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 13. Mean PFOS concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
Figure 14. Average concentrations of total mercury (dots) measured in composite samples of Rainbow Smelt by Environment Canada. Lines show the three-year moving average. Note that most rates of change are not significantly different from zero base on p-values >0.05.
Source: Environment and Climate Change Canada
Figure 15. Total HBCD concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Last Updated
State of the Great Lakes 2019 Technical Report

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
Short Chane 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		

Table 1. Contaminant criteria for environmental monitoring and surveillance programs.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency
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European Union Environmental Quality Standards - https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32013L0039

Compound or Class	Agency		Great Lake				
	ECCC	US EPA	Ontario	Erie	Huron	Michigan	Superior
4-n-octylphenol (OP)	X		X				
4-nonlyphenol monoethoxylate (NP1EO)	X		X				
4-nonylphenol (NP)	X		X				
4-nonylphenol diethoxylate (NP2EO)	X		X				
Hexabromocyclododecane (α-, γ-HBCD)	X		X				
Polychlorinated naphthalenes (PCN)[1]	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**
Chemicals detected greater than 10 % frequency identified through Monitoring and Surveillance							

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

Mercury	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	1.000	0.972	0.079	1.000
Sen's slope (%)	-0.58%	0.01%	0.01%	-0.38%	-0.02%

HxBDE	Ontario	Huron	Erie	Superior	Michigan
P value	**0.002**	0.545	0.837	0.108	0.232
Sen's slope (%)	-1.97%	-0.33%	0.39%	-2.65%	-0.95%

PeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	**0.000**	0.069	0.651	**0.004**	**0.009**
Sen's slope (%)	-2.00%	-1.39%	-0.41%	-3.20%	-3.92%

TeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	**0.000**	**0.000**	0.902	0.053	**0.000**
Sen's slope (%)	-2.96%	-3.51%	0.06%	-2.09%	-5.98%

HBCDD	Ontario	Huron	Erie	Superior	Michigan
P value	0.668	0.876	0.436	0.087	**0.029**
Sen's slope (%)	-0.40%	0.57%	1.29%	-2.06%	2.16%

PFOS	Ontario	Huron	Erie	Superior	Michigan
P value	0.474	0.436	**0.003**	**0.003**	-
Sen's slope (%)	-0.81%	1.81%	-7.32%	5.82%	-

PCB	Ontario	Huron	Erie	Superior	Michigan
P value	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Sen's slope (%)	-3.01%	-2.82%	-1.19%	-2.13%	-2.42%

LEGEND

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

Long Term Data Set

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	1.000	1.000	0.474	0.211	0.858
Sen's slope (%)	0.00%	0.02%	-0.79%	-2.21%	0.20%

HxBDE	Ontario	Huron	Erie	Superior	Michigan
P value	0.243	0.474	**0.007**	0.152	0.721
Sen's slope (%)	-1.33%	1.23%	-3.75%	-3.64%	0.91%

PeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	0.362	1.000	0.107	0.107	0.592
Sen's slope (%)	0.83%	0.32%	-1.57%	-3.90%	0.73%

TeBDE	Ontario	Huron	Erie	Superior	Michigan
P value	0.059	**0.007**	0.074	0.152	0.107
Sen's slope (%)	-2.29%	-3.69%	-1.94%	-3.12%	-3.69%

HBCDD	Ontario	Huron	Erie	Superior	Michigan
P value	0.059	1.000	0.858	0.152	0.074
Sen's slope (%)	-2.17%	0.14%	0.48%	-2.04%	2.12%

PFOS	Ontario	Huron	Erie	Superior	Michigan
P value	0.474	1.000	**0.000**	**0.012**	-
Sen's slope (%)	-0.81%	0.05%	-6.54%	5.90%	-

PCB	Ontario	Huron	Erie	Superior	Michigan
P value	0.419	0.858	0.721	0.371	0.074
Sen's slope (%)	-0.02%	0.85%	-0.86%	-2.25%	-2.17%

LEGEND

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

Short Term Data Set

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

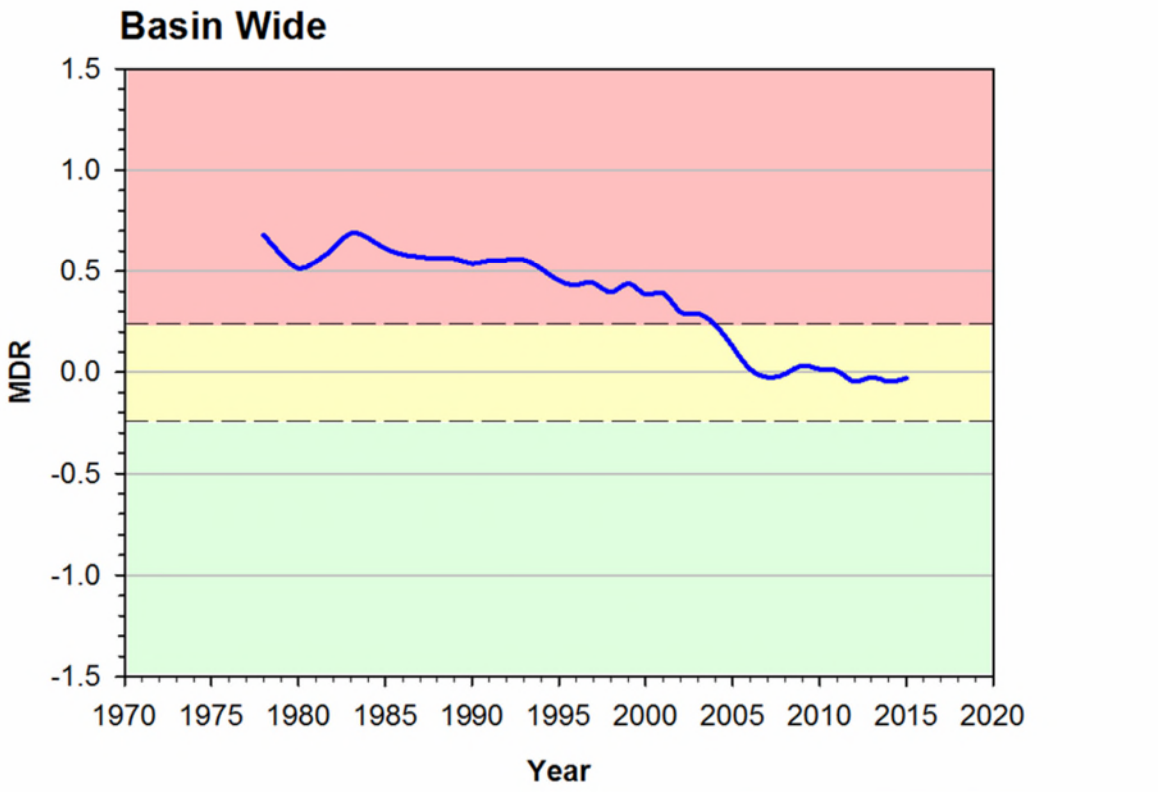


Figure 1. Mean Deviation Ratio (MDR) for the Great Lakes basin.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

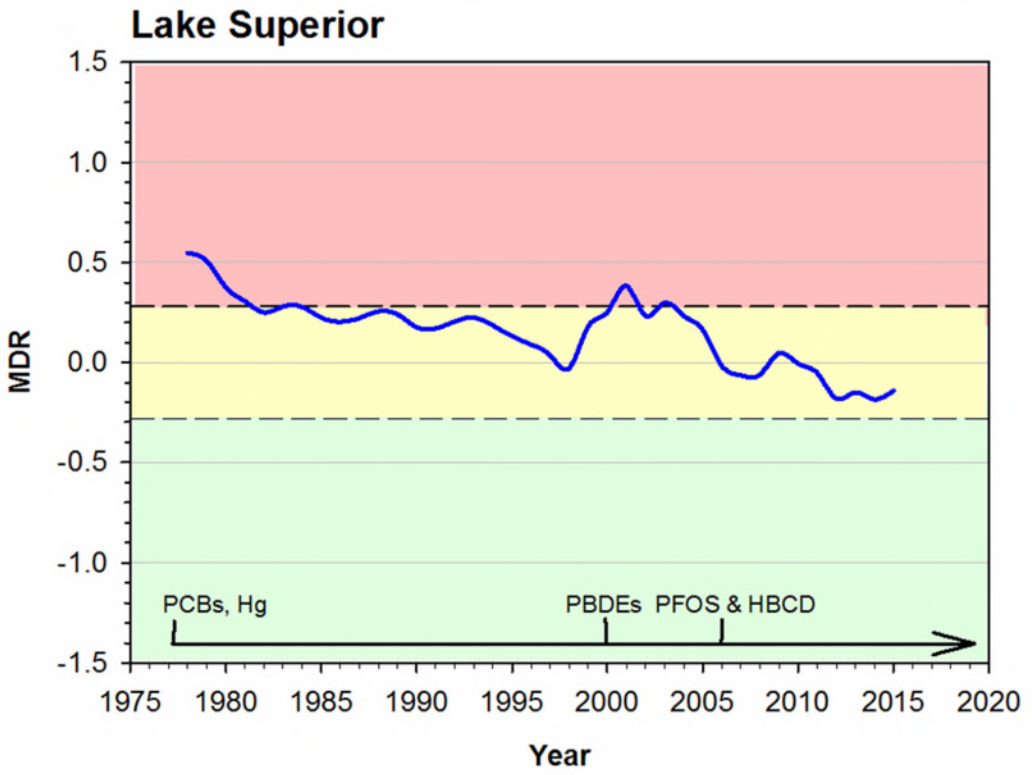


Figure 2. Mean Deviation Ratio (MDR) for Lake Superior.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

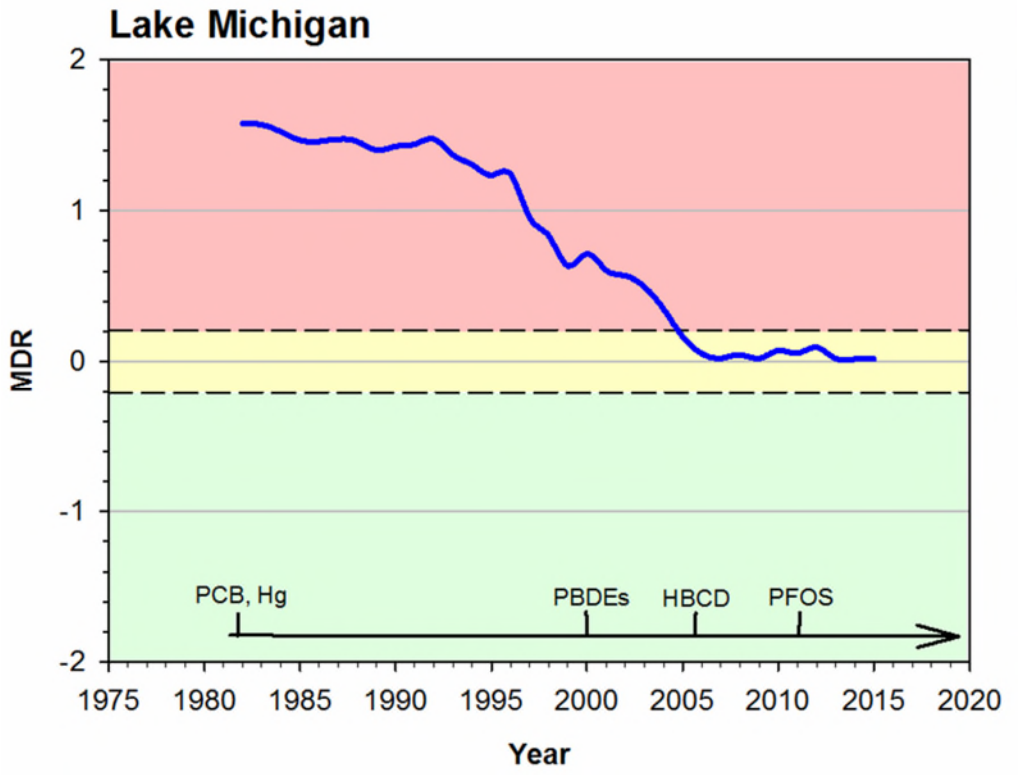


Figure 3. Mean Deviation Ratio (MDR) for Lake Michigan.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

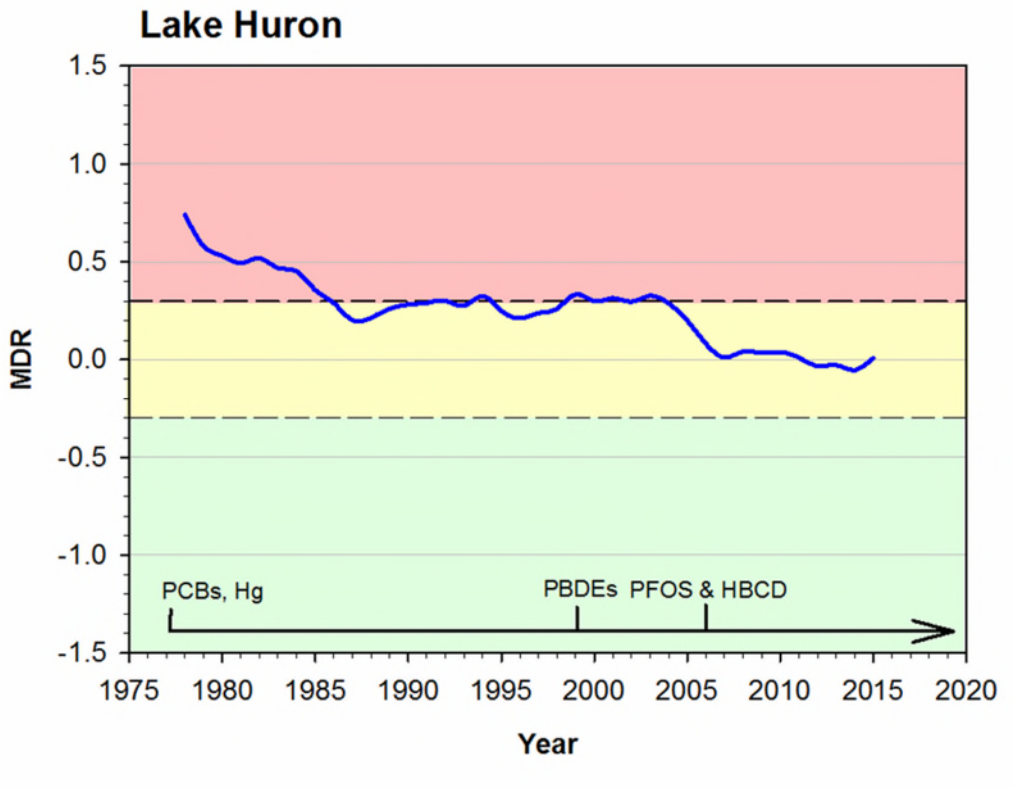


Figure 4. Mean Deviation Ratio (MDR) for Lake Huron.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

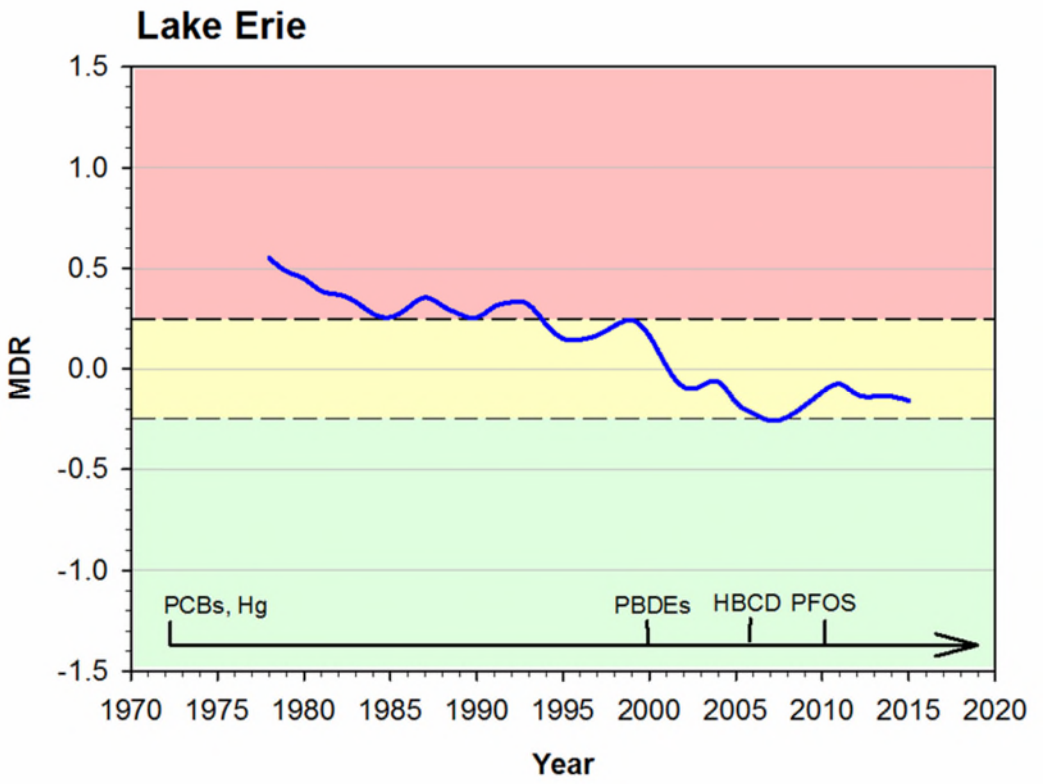


Figure 5. Mean Deviation Ratio (MDR) for Lake Erie.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

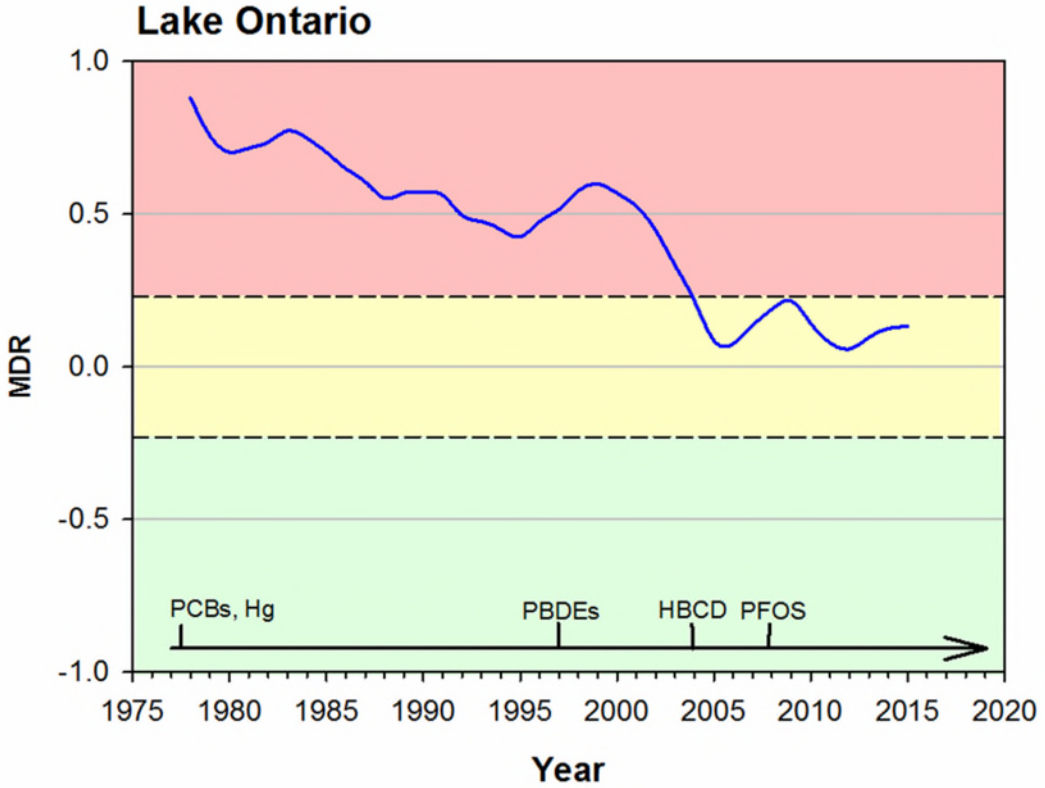


Figure 6. Mean Deviation Ratio (MDR) for Lake Ontario.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

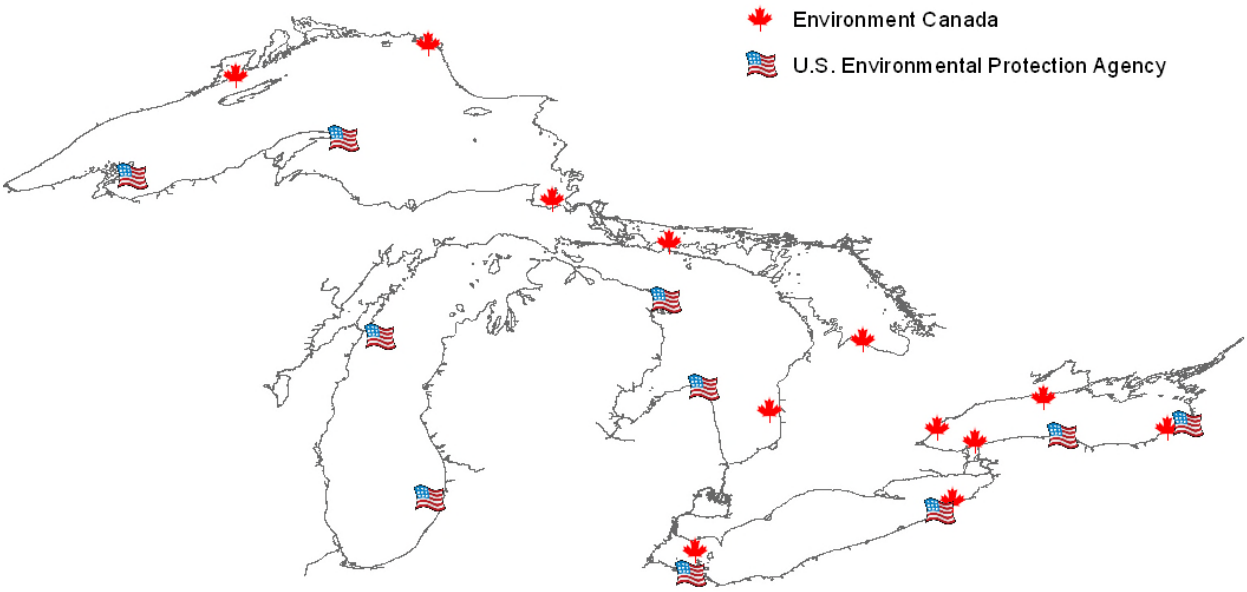


Figure 7. Map of Great Lakes showing Environment and Climate Change Canada and U.S. Environmental Protection Agency monitoring stations for fish contaminants.
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

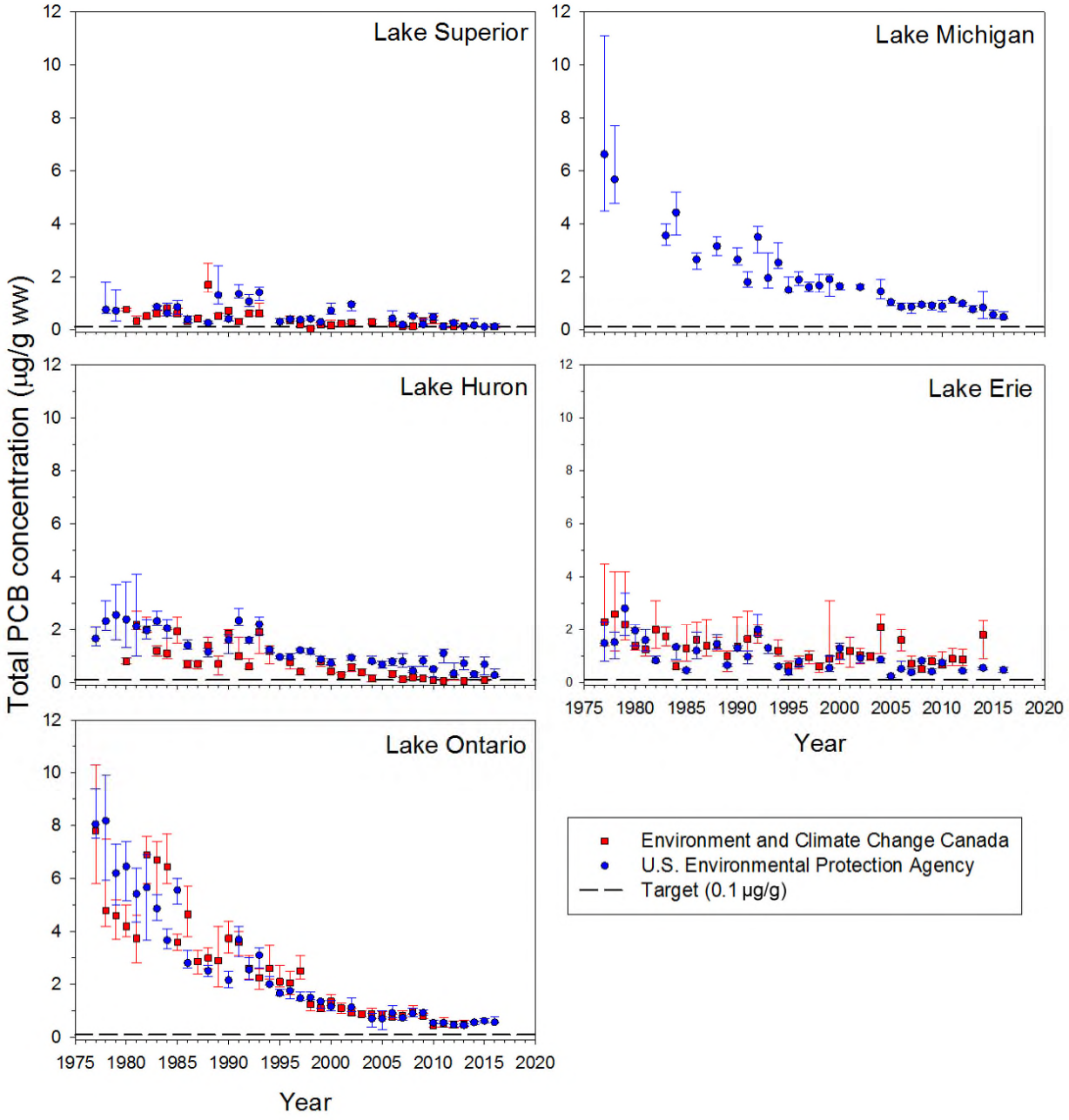


Figure 8. Total PCB concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

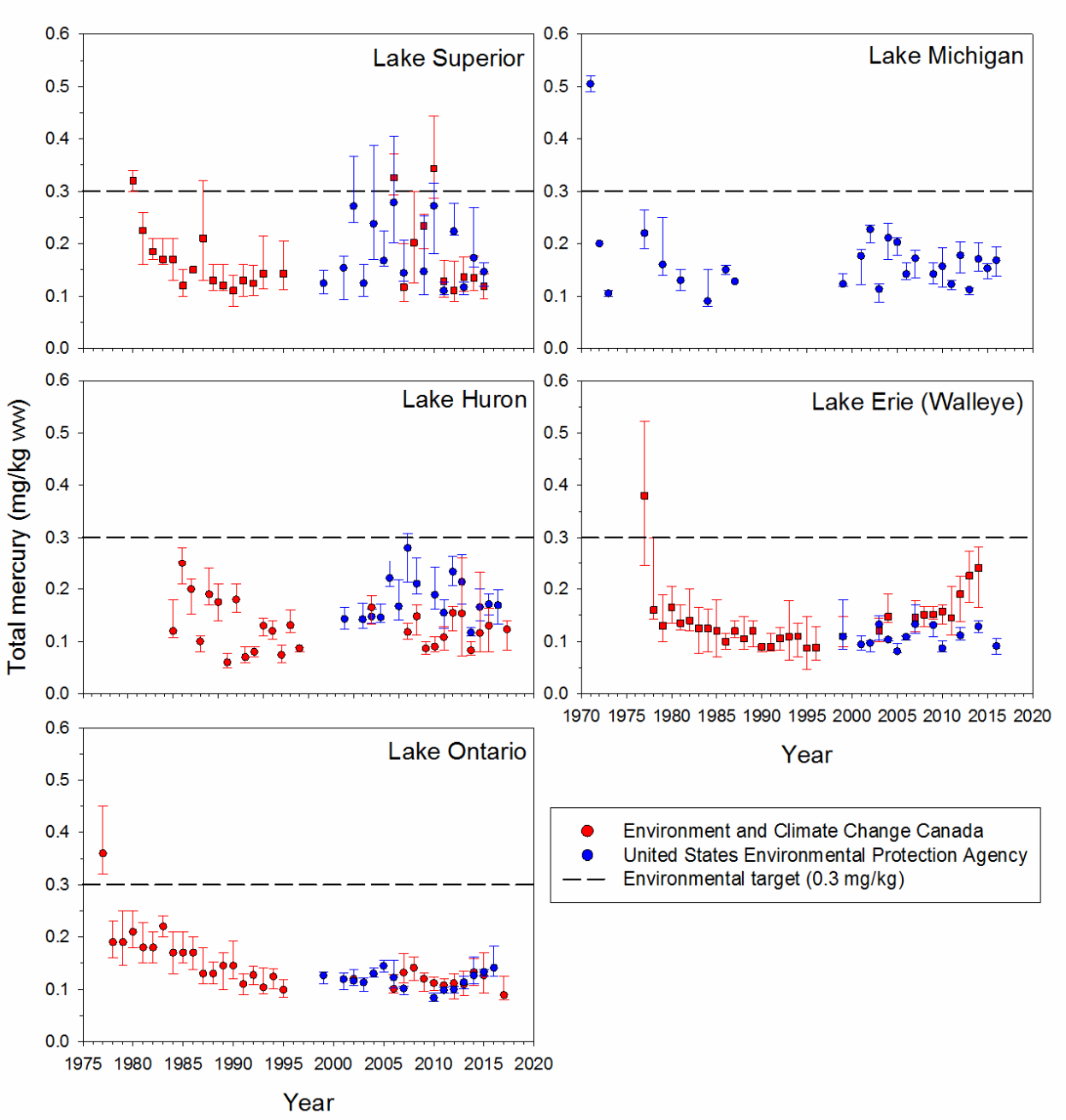


Figure 9. Total mercury concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada, U.S. Environmental Protection Agency

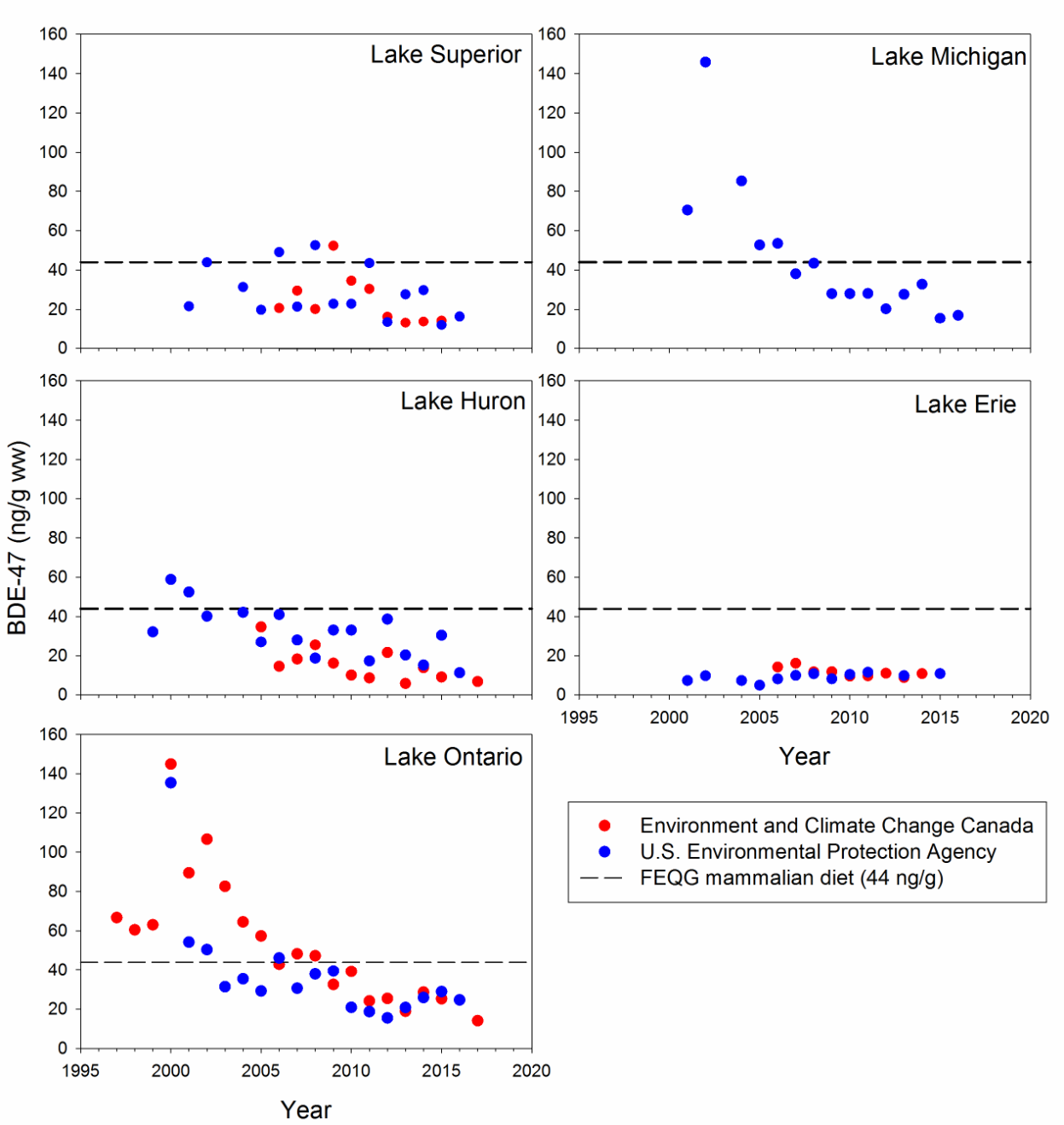


Figure 10. Mean TeBDE concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

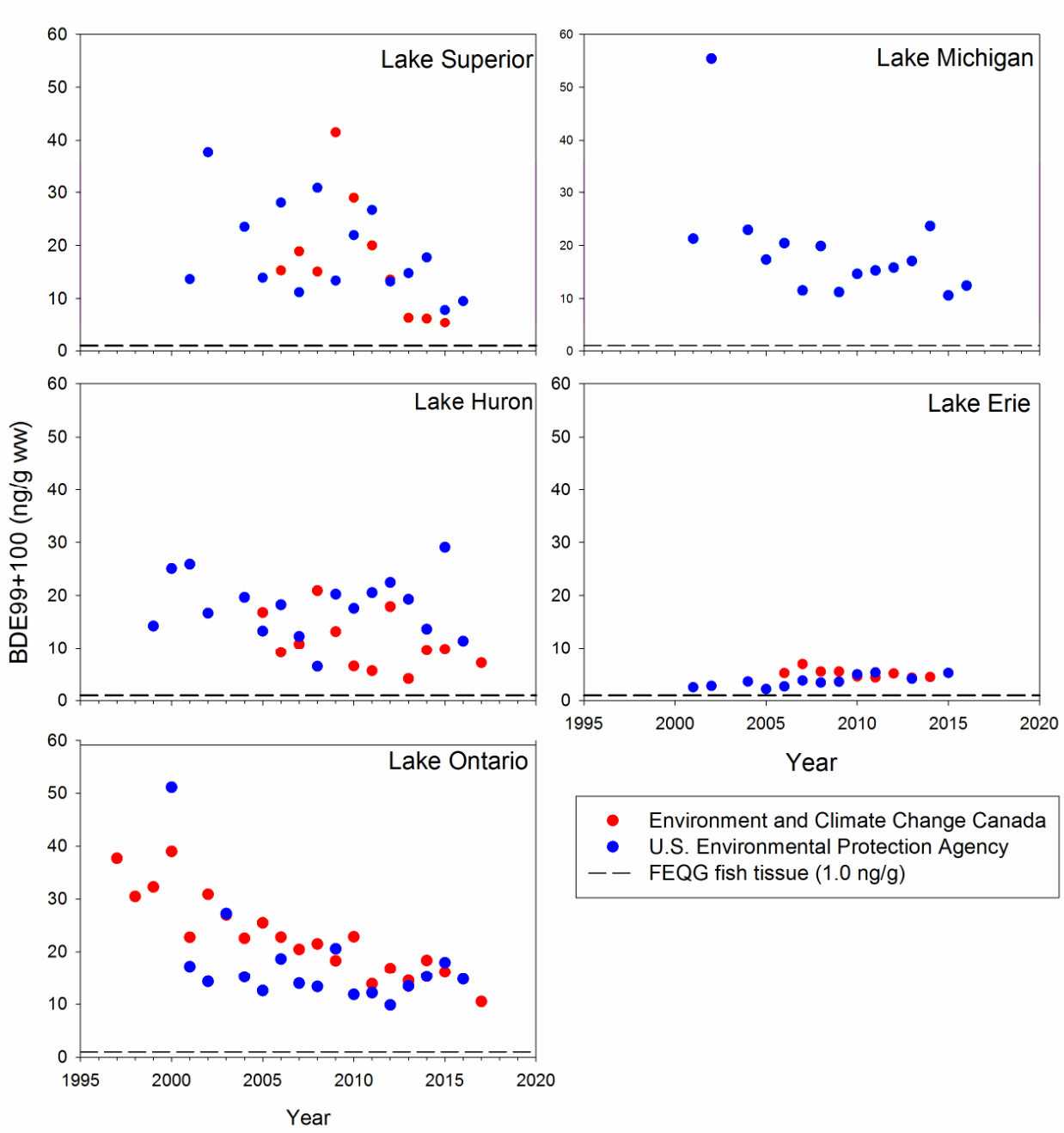


Figure 11. Mean PeBDE concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

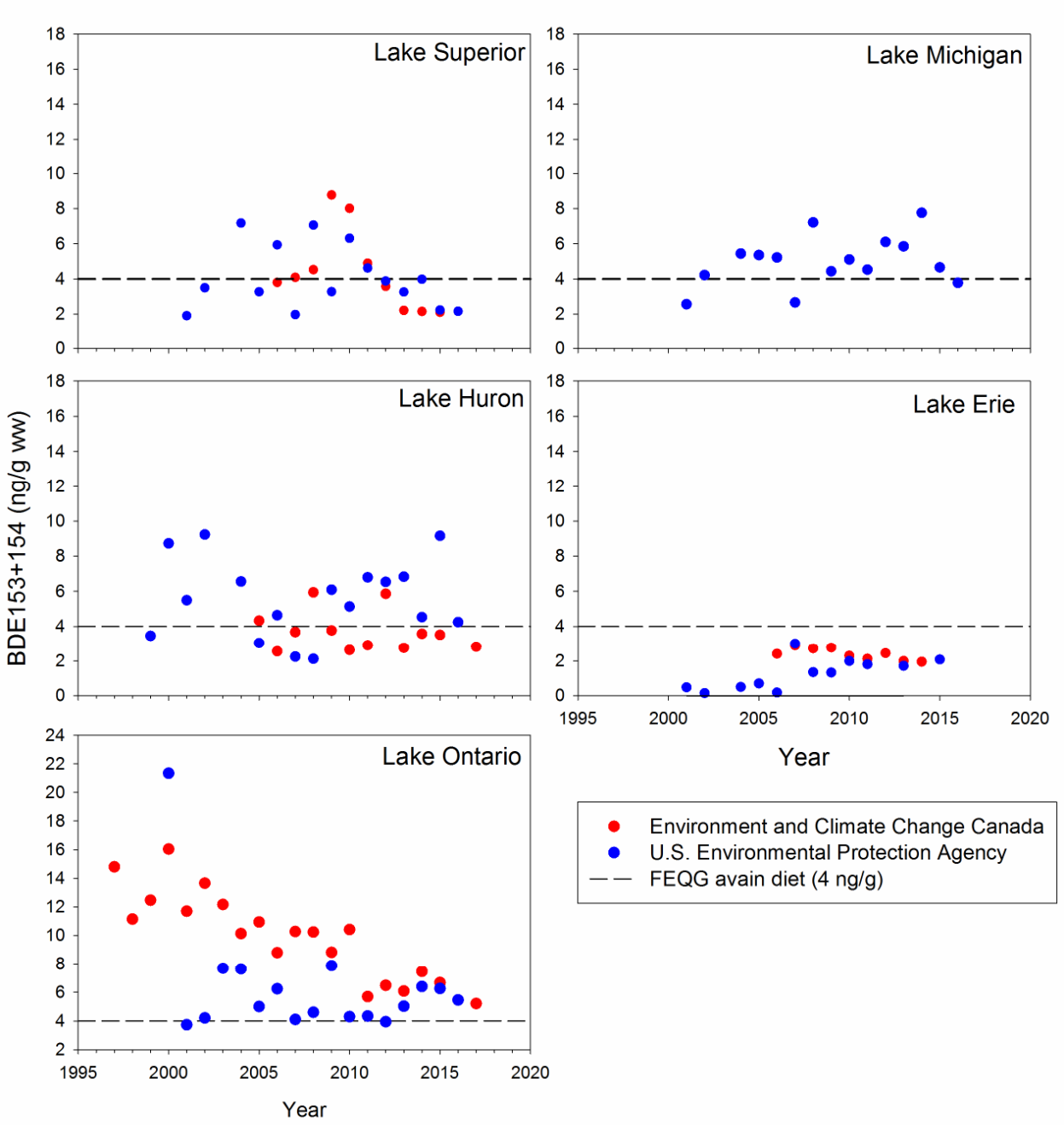


Figure 12. Mean HxBDE concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

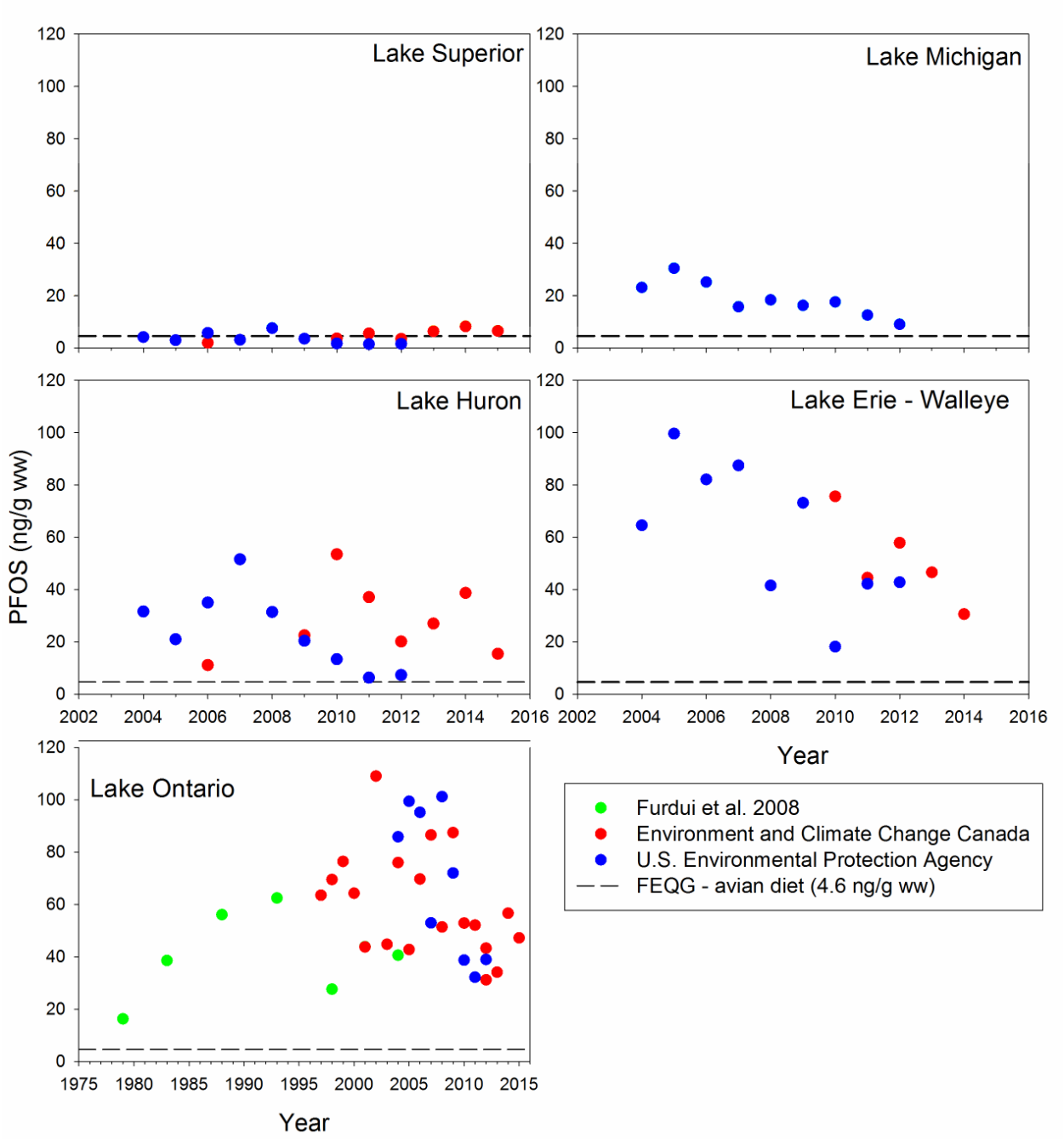


Figure 13. Mean PFOS concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

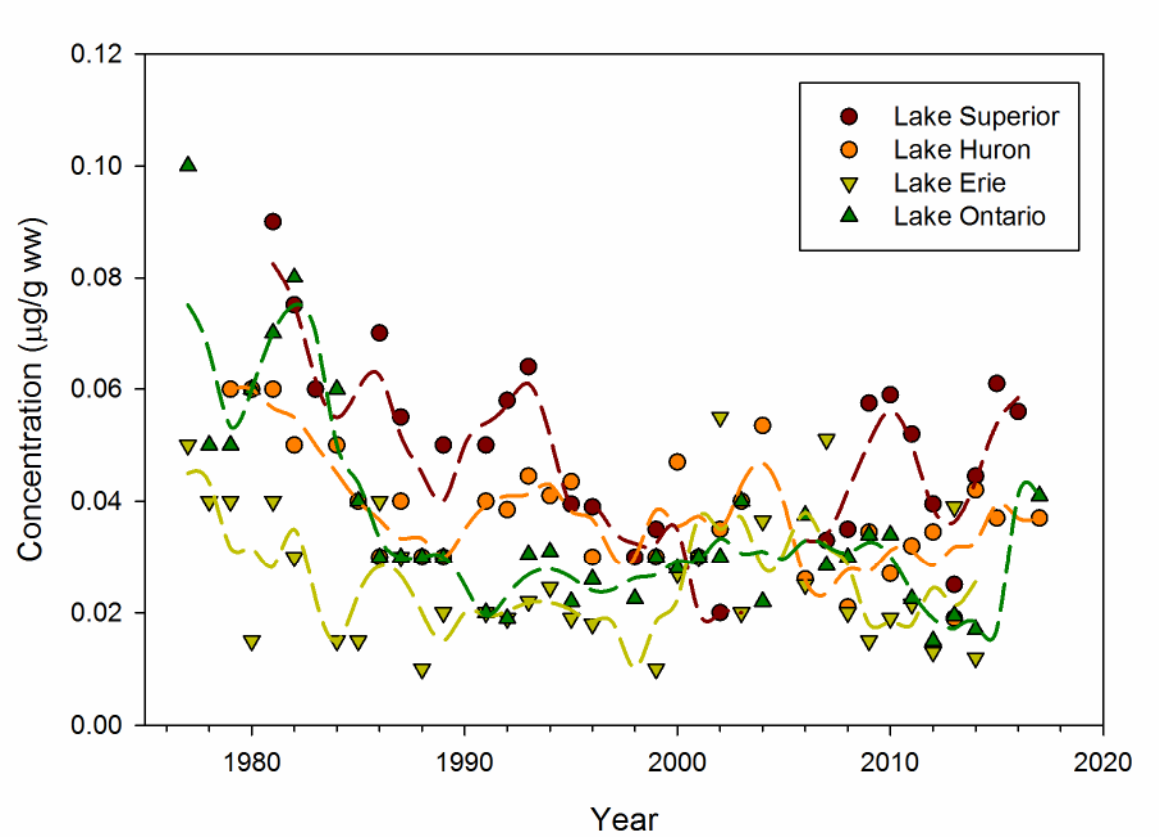


Figure 14. Average concentrations of total mercury (dots) measured in composite samples of Rainbow Smelt by Environment Canada. Lines show the three-year moving average. Note that most rates of change are not significantly different from zero base on p-values >0.05.
Source: Environment and Climate Change Canada

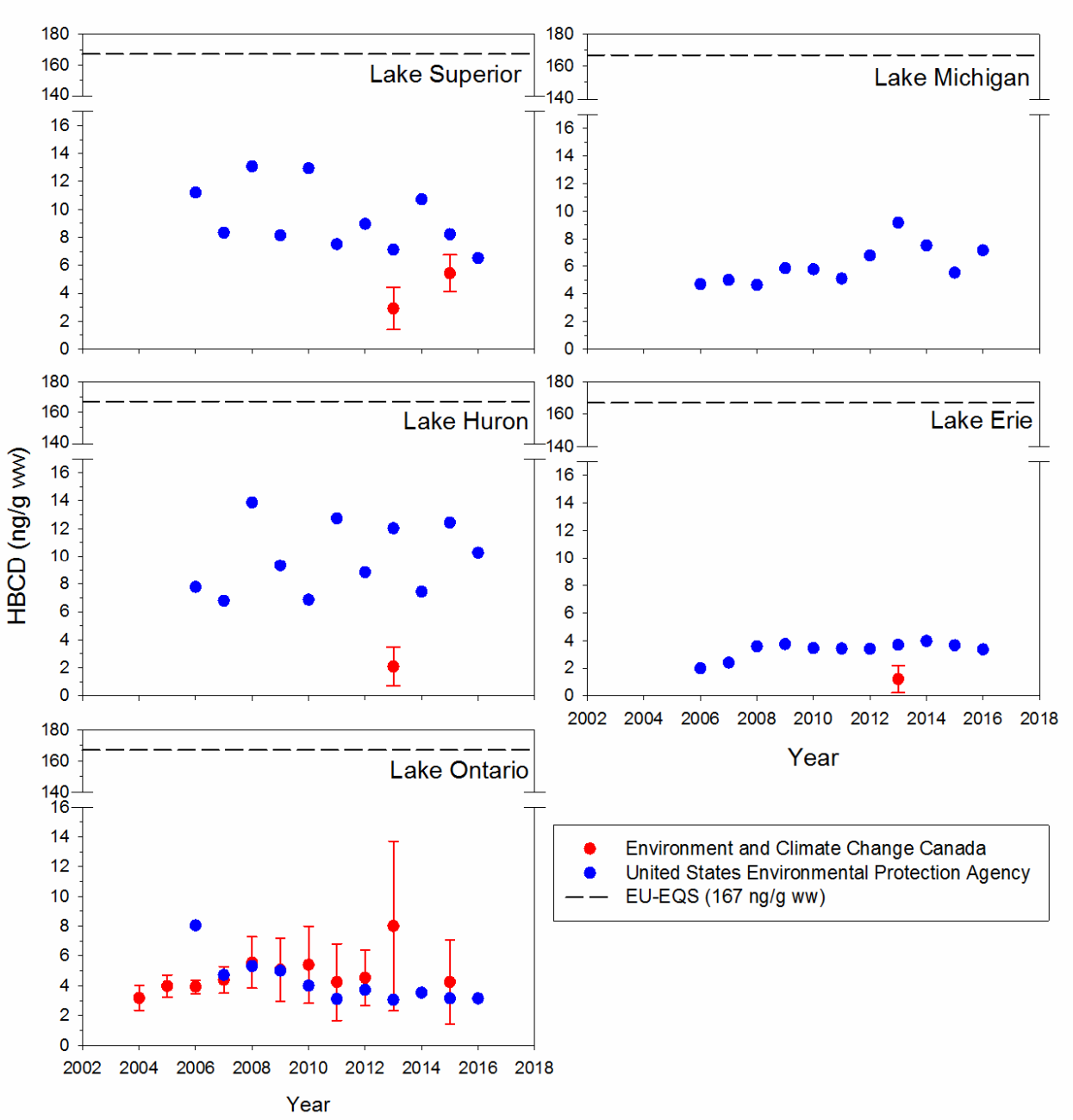


Figure 15. Total HBCD concentrations for individual (Environment and Climate Change Canada) and composited (U.S. Environmental Protection Agency) whole body Lake Trout or Walleye (Lake Erie) collected from each of the Great Lakes. Figures with dashed trend lines are shown where slopes are not statically different than zero. Solid lines denote slopes that are statistically greater or less than zero ($\alpha = 0.05$).
Source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Sub-Indicator: Toxic Chemicals in Herring Gull Eggs

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 contaminant (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 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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List of Tables

Table 1. Concentrations of PCBs (sum of 33 congeners) and select organochlorine pesticides (µg/g, wet weight) in Herring Gulls 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]
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]

List of Figures

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

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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 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	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 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]

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

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]

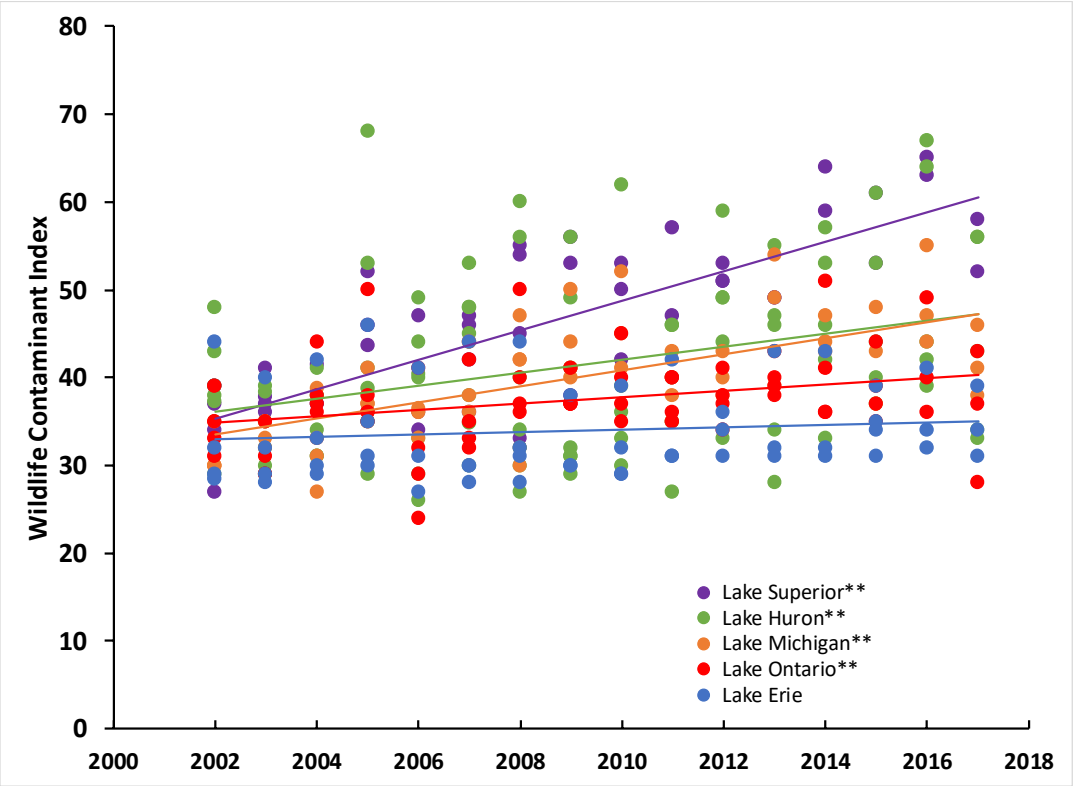
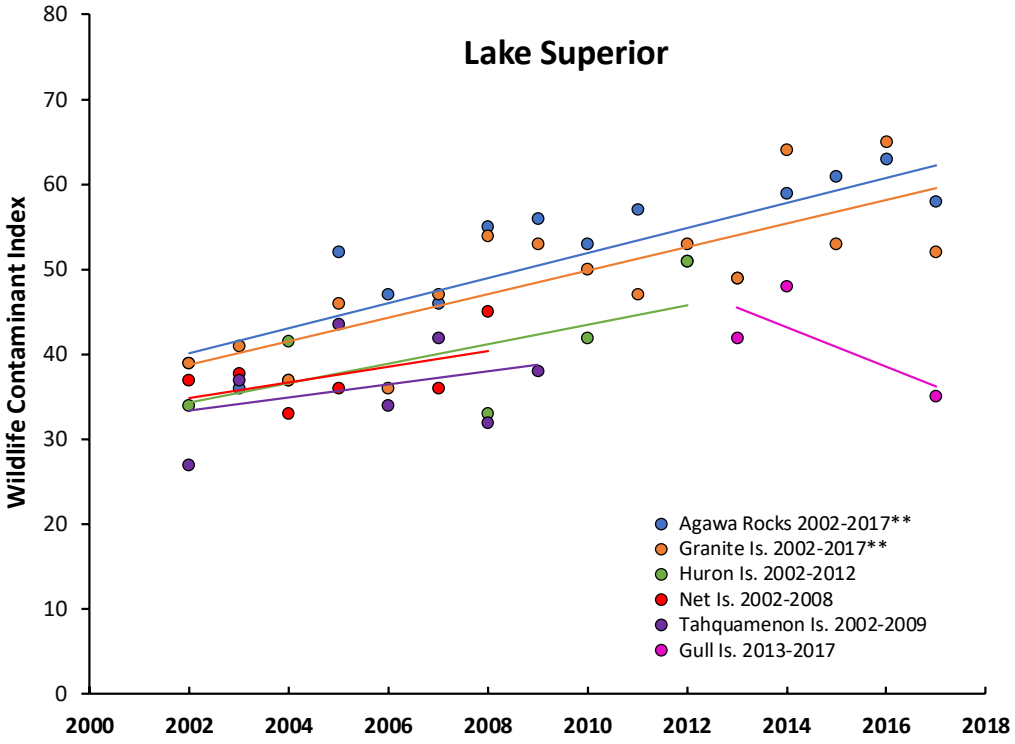
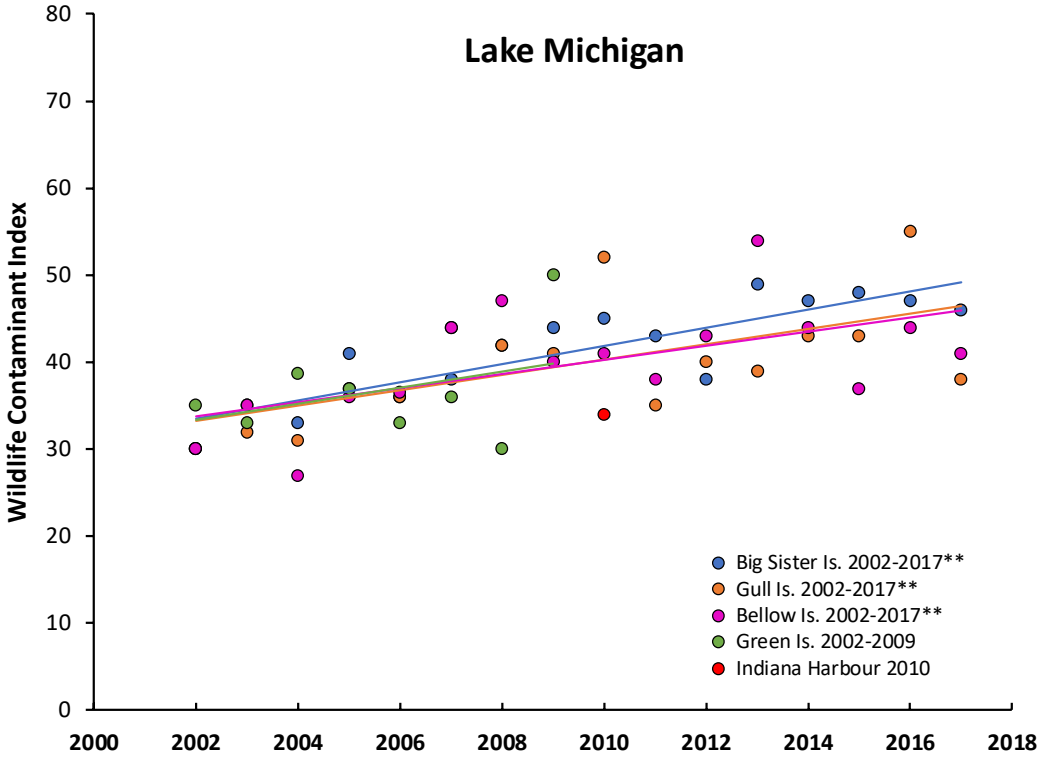
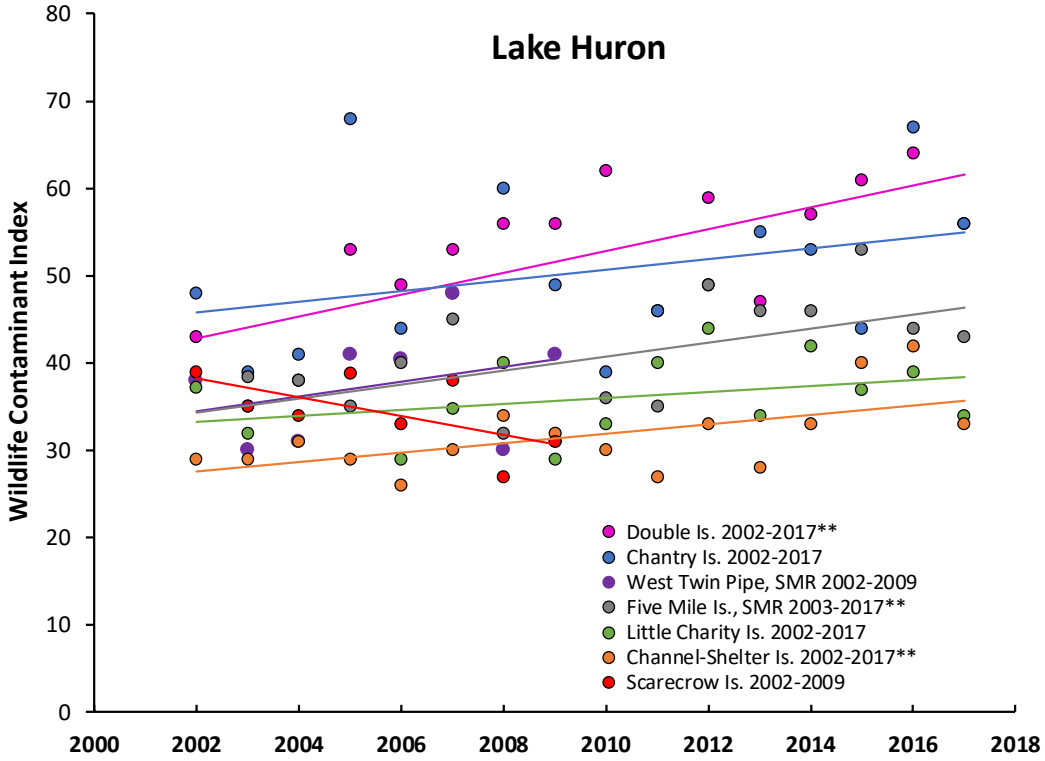
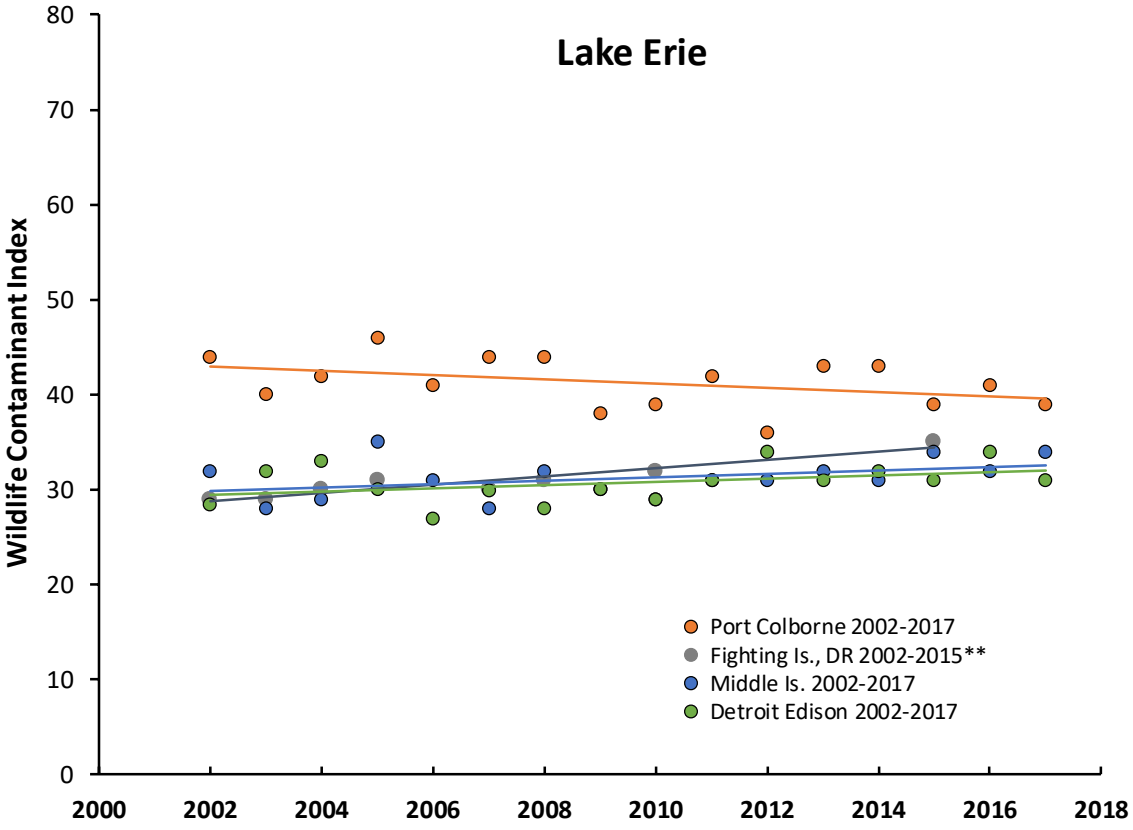


Figure 1. Temporal trends for the Wildlife Contaminant Index (WCI), based upon PCBs, PCDD/Fs and selected organochlorine pesticides, of Herring Gull colonies pooled by Great Lake. WCIs are significantly increasing which is indicative of improving conditions in all lakes as designated by "**", except 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
80
70
60
50
40
30
20
10
0
2000
2002
2004
2006
2008
2010
2012
2014
2016
2018
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
80
70
60
50
40
30
20
10
0
2000
2002
2004
2006
2008
2010
2012
2014
2016
2018
Port Colborne 2002-2017
Fighting Is., DR 2002-2015**
Middle Is. 2002-2017
Detroit Edison 2002-2017

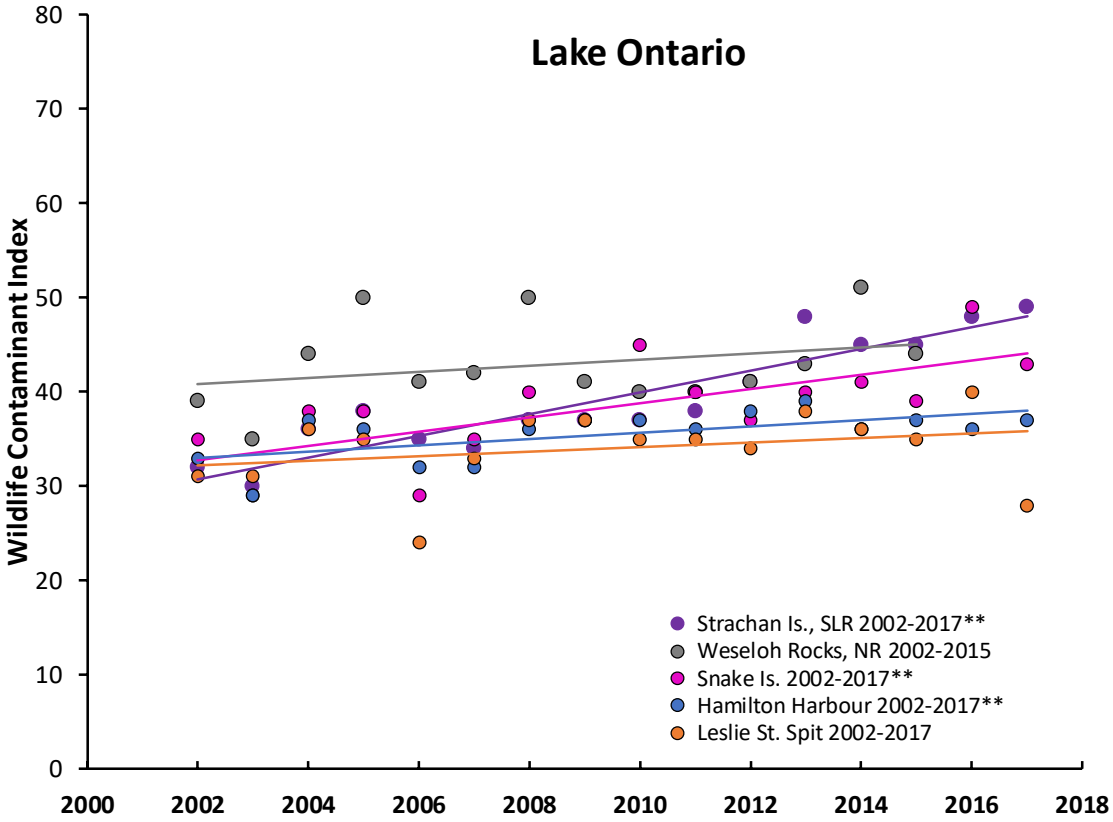


Figure 2. Temporal trends for the Wildlife Contaminant Index of each Herring Gull colony within each Great Lake, based upon PCBs, PCDD/Fs and selected organochlorine pesticides. "**" indicates colonies where linear regression analysis is significantly different from zero.
Source: Great Lakes Herring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

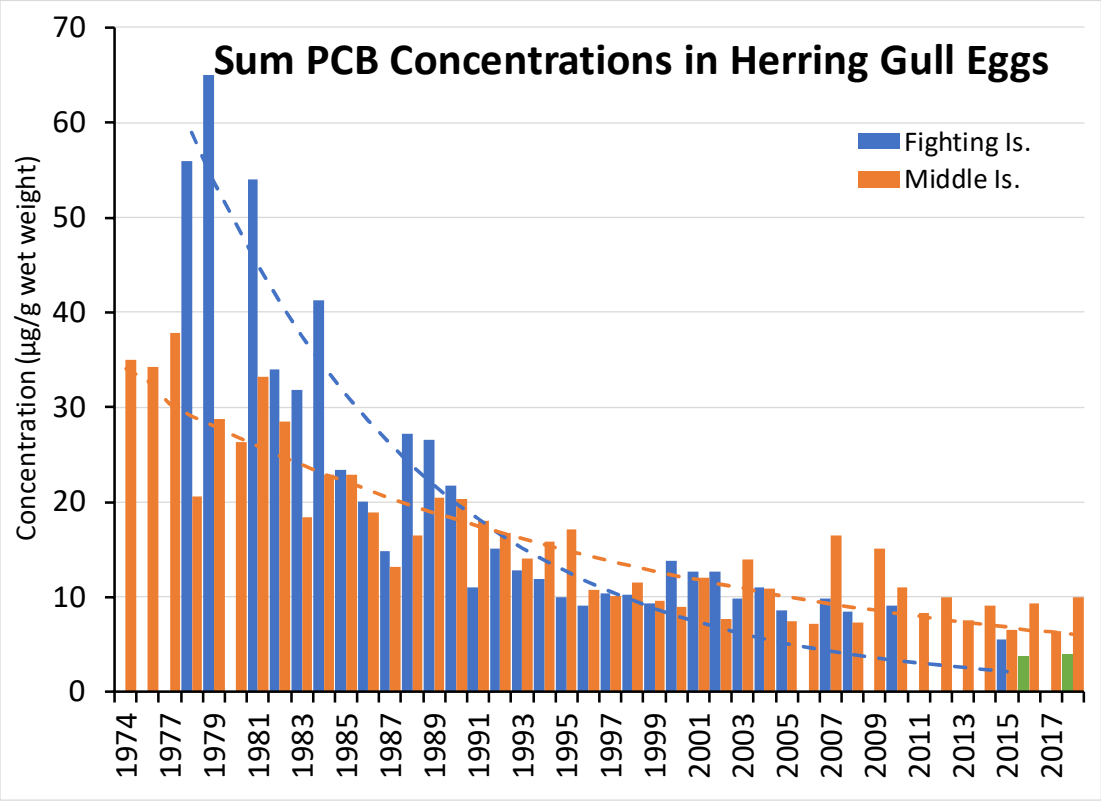


Figure 3. Sum PCBs in herring gull eggs from Detroit River (Fighting Island) and western Lake Erie (Middle Island) from 1974-2018. The green bars indicate PCB concentrations in cormorant eggs from Fighting Island in 2016 and 2018 since no data are available for Herring Gulls.
Source: Great Lakes Herring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

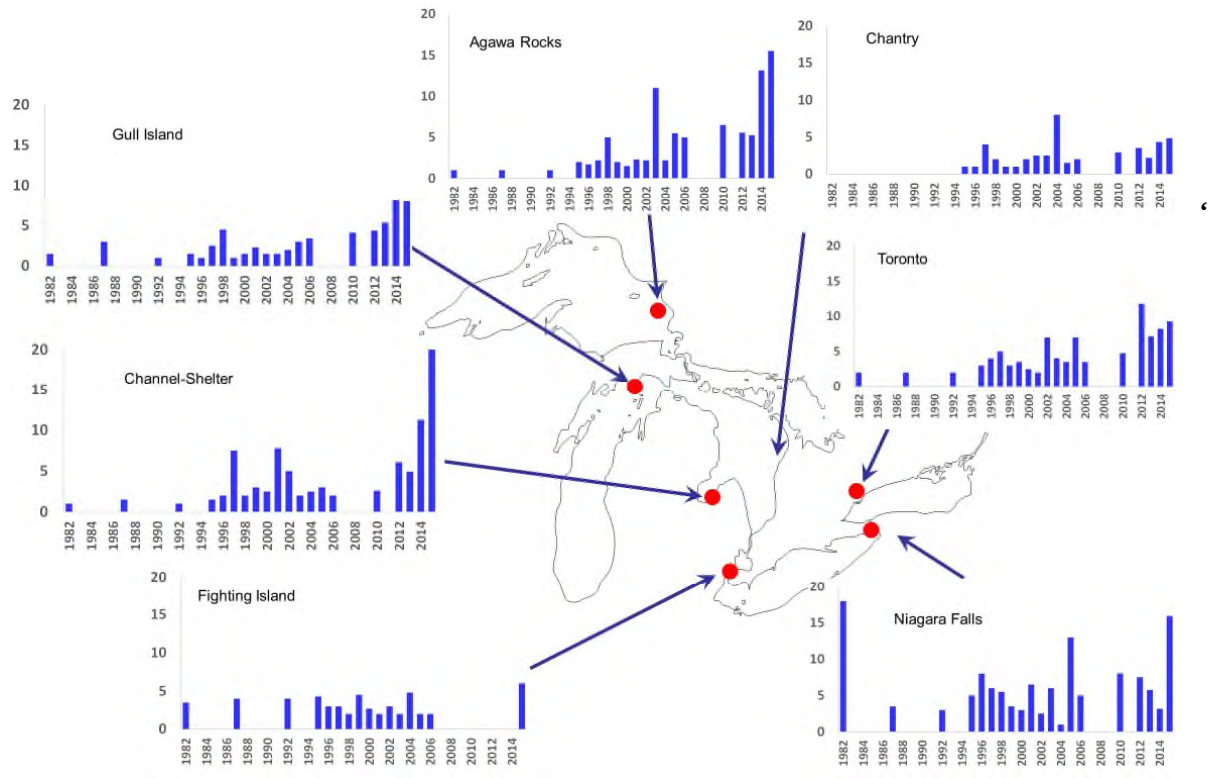


Figure 4. Concentrations of Dechlorane Plus (ng/g, wet weight) in Herring Gull eggs from selected colonies from the Great Lakes in 1982 to 2015.
Source: Great Lakes Herring Gull Monitoring Program [Environment and Climate Change Canada]

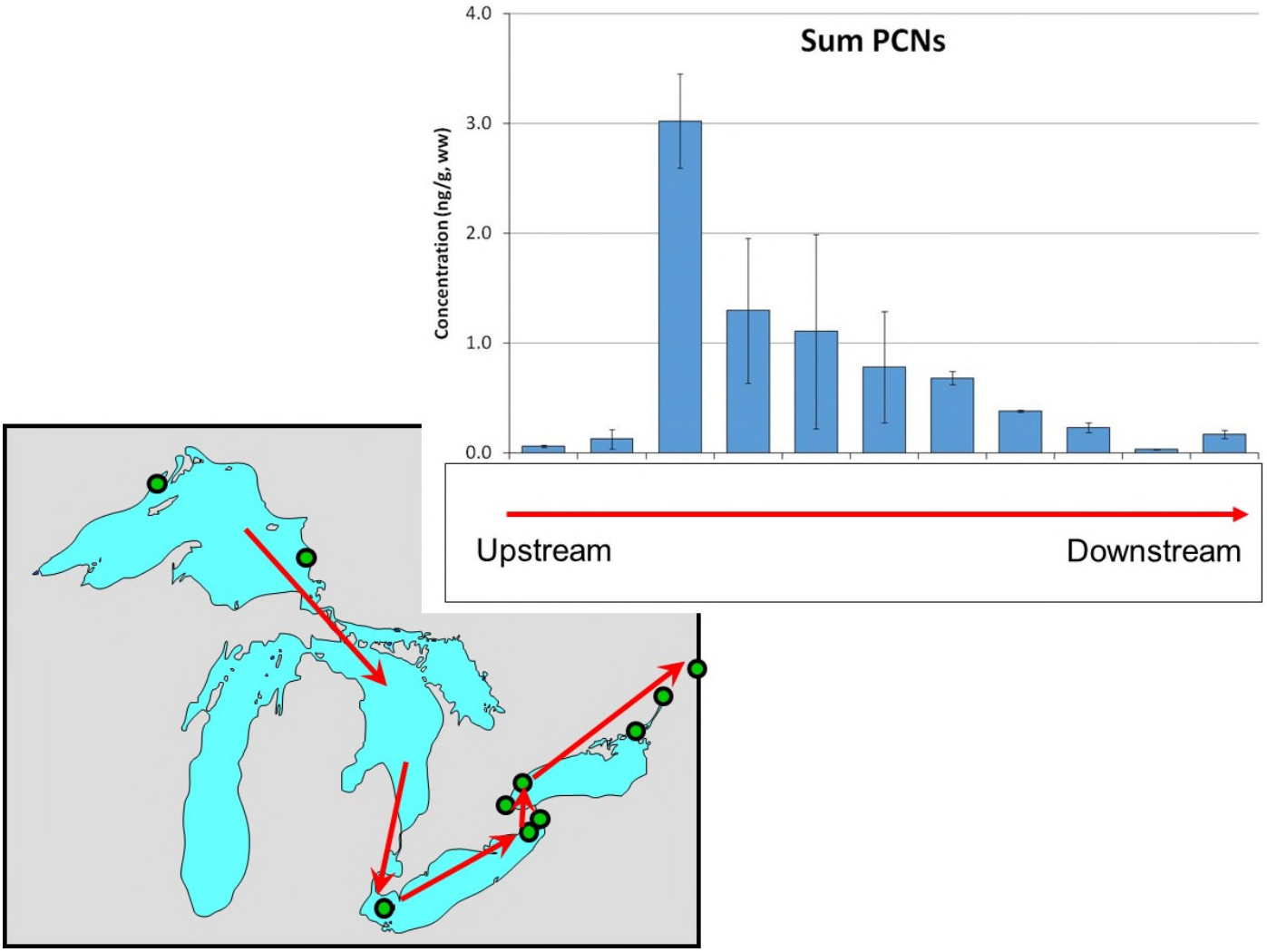


Figure 5. Concentrations of Polychlorinated naphthalenes (ng/g, wet weight) in Herring Gull eggs from selected colonies from the Great Lakes in 2011-2013 (unpublished data, ECCC; based upon McGoldrick et al, 2018). Colonies are ordered left-right from upstream-downstream; the peak corresponds to eggs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

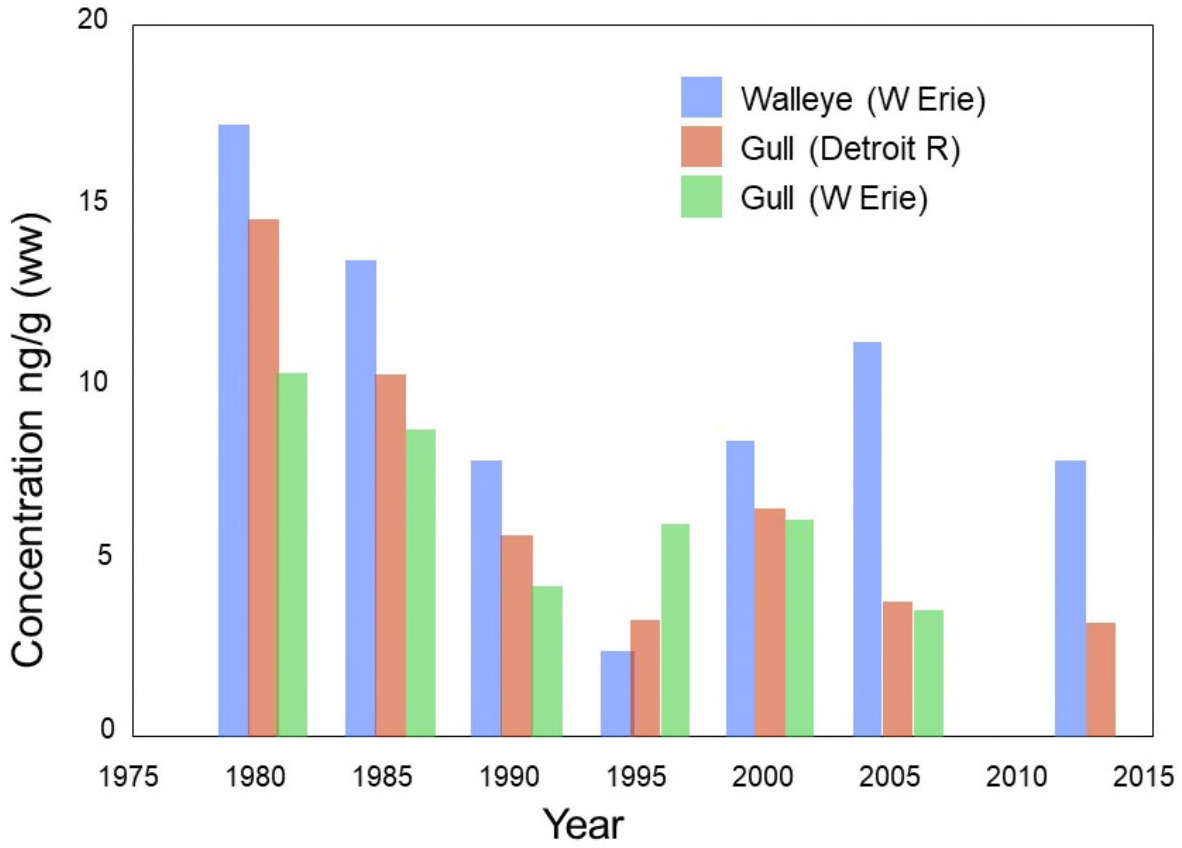


Figure 6. Concentrations of Polychlorinated naphthalenes (ng/g, wet weight) in Herring Gull eggs from the Detroit River and western Lake Erie, and Walleye from Western Lake Erie, from 1980 to 2015. In the mid to late-1990s, there was substantial dredging of PCN contaminated sediment from the Detroit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

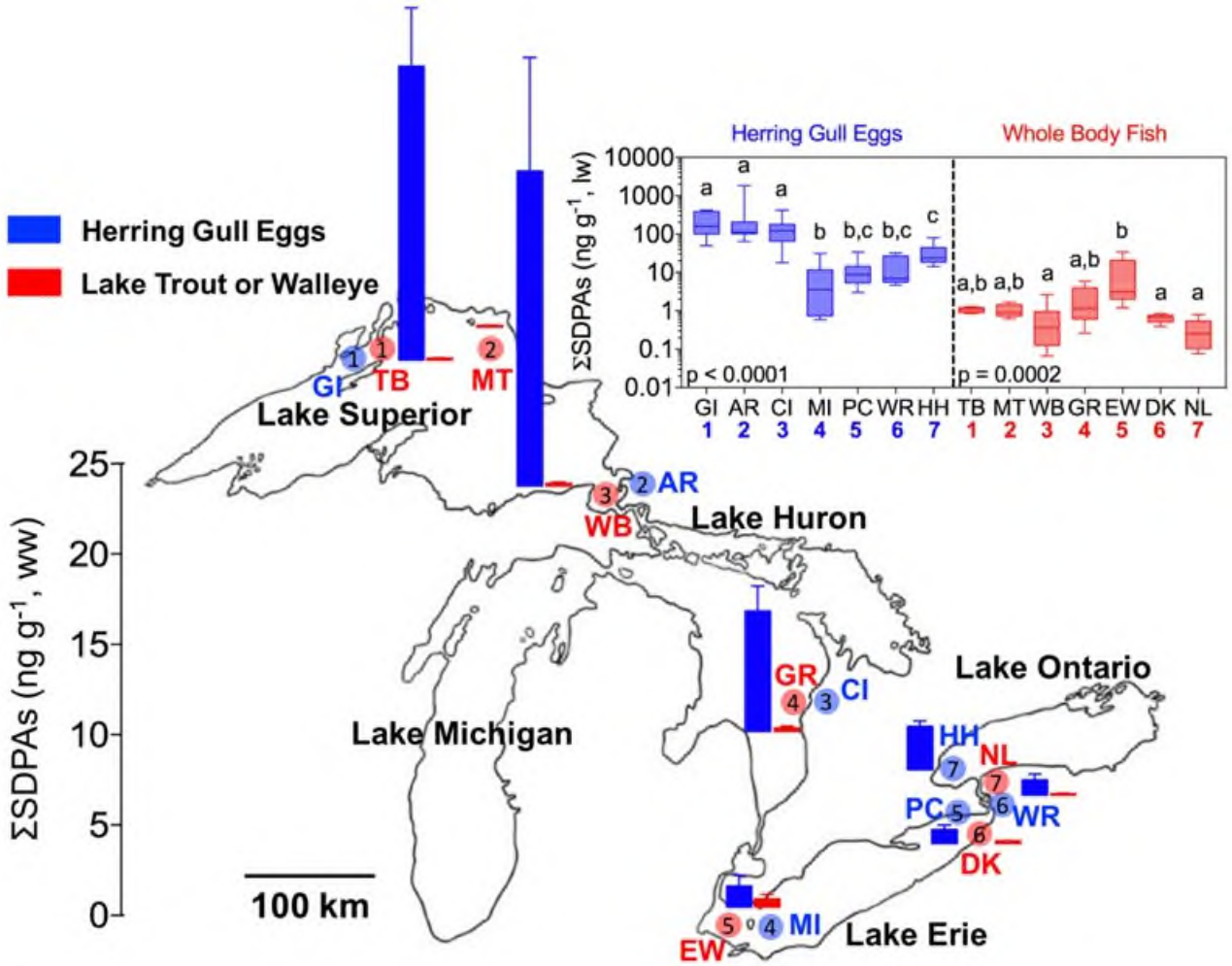


Figure 7. Concentration (mean ± standard error bar (ww) of ΣSDPAs in Herring Gull eggs (blue) and Lake Trout and Walleye whole body homogenate (red) in the Great Lakes of North America. Inset shows boxplot of lipid normalized ΣSDPAs concentrations.
Source: Chemical Monitoring Program [Environment and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

Sub-Indicator: Toxic Chemicals in the Atmosphere

(Previously entitled Atmospheric Deposition of Toxic Chemicals)

Overall Assessment
Status: **Fair**
Trends:
10-Year Trend: Improving
Long-term Trend (1990-2016): Improving
Rationale: Although levels of toxic chemicals in air are generally low, the large surface area of the Great Lakes results in significant atmospheric inputs. 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. Eastern Lake Erie tends to show higher levels than at other remote sites, most likely since it is located closer to an urban area (Buffalo, NY) than the other master stations. It may also receive some influence from the East Coast of the U.S. For the 2019 sub-indicator report, 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 2017 and 2011 Atmospheric Deposition of Toxic Chemicals reports. 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 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 (11 years of data available). Precipitation samples were collected, which provides good indications of atmospheric PFAS levels due to the polar characteristic 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 remain 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 HBCDD and SCCP.

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). Calculated loadings for each lake, including trends over time, are also available (Guo et al. 2018 and Shunthirasingham et al. 2016).

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.

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.

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 time which is most relevant to virtual elimination is the longest halving time of the measured chemicals.

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. 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 or negative, 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 will measure the concentrations of Chemicals of Mutual Concern (CMCs) in the atmosphere and in precipitation of the Great Lakes region. This sub-indicator will track whether concentrations are decreasing, staying the same, or increasing over time where data are available. The sub-indicator will also assess whether the spatial trends in concentrations are changing or staying the same. The monitoring data was used by the Parties 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 will primarily report 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 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 will also consider the most recent atmospheric loadings as estimated by IADN and GLB (see data limitations section).

For the current CMCs relevant to atmospheric deposition, air and atmospheric loading information (where available) for PCBs, mercury, LC-PFCAs - C9-C20, PFOA, PFOS, PBDEs, HBCDD and SCCPs gathered from these monitoring efforts and scientific literature will be included in this sub-indicator. For CMCs that do not have sufficient information to base a determination, air and atmospheric deposition information will be 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 will also be presented in this sub-indicator as additional information but only data on the CMCs will be used for the status and trend assessments.

Precipitation samples are integrated over each month. Annual average concentrations will be calculated and reported using these measurements. Spatial and temporal trend analyses will be 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 will be 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 will be calculated and reported using this data. Spatial and temporal trend analyses will be performed on the MDN data using a variety of statistical tools (Risch et al., 2012; Prestbo and Gay, 2009).

Air concentrations respond rapidly to changes in emissions, but it is not known how rapidly the lakes and the fishes in them will respond ("environmental hysteresis"). This makes atmospheric measurements a very useful tool for tracking progress as a result of chemical control efforts.

Another component of the sub-indicator in the future could be to assess the relative rate of change of the loading with time. The more contaminant levels decline, the faster the goal of virtual elimination will be achieved.

Since air has a short response time to changes in atmospheric emissions and is a relatively well-mixed environmental medium, this sub-indicator is most suitable in providing early warning for chemicals that could become CMCs. The occurrence of potentially harmful substances in Great Lakes air and precipitation will be presented here but will not be 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. Since the 1990s, over a million measurements of the concentrations of PCBs, pesticides, PAHs, flame retardants, and trace metals have been made at these sites. 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, other contaminants (non-CMCs) were also considered in assessing the status and trend. Despite the difference in chemicals used for this 2019 assessment 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 2019 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 where possible, 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 (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)

Atmospheric PCB concentrations are decreasing relatively slowly with halving times in the range of 9-40 years at Canadian sites (Shunthirasingham et al. 2016) and about 11-27 years at U.S. sites; Figure 2. Previously, no differences in the halving times of PCBs among the five U.S. sites and Point Petre were found, suggesting a relatively homogeneous decrease rate in the Great Lakes region (Salamova et al, 2015). 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 (Shanahan et al. 2015, and others), 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. 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; Shunthirasingham et al. 2016).

Volatilization of PCBs from the lakes is also 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 3 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 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 coming under control. 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; this is a not insignificant 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. This net transfer is 46% lower than that estimated in 1994−1995 which is an improvement. 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. PBDE data were analyzed using a multiple linear regression model to determine significant temporal trends in these atmospheric concentrations, and some of these data are shown in Figure 6. The concentrations of PBDEs were decreasing at the urban sites at Chicago and Cleveland, but were generally unchanging at the remote sites, Sleeping Bear Dunes and Eagle Harbor.

GLB data (2005-2014) showed that BDE-47, BDE-99, and BDE-209 were the dominant PBDE congeners found at Point Petre, Burnt Island and Egbert (a satellite station between Lake Huron and Lake Ontario) (Shunthirasingham et al., 2018). The total concentrations of the 15 congeners (Σ_{15}PBDEs) ranged from 1.2 to 160 pg/m^3, with an annual average of 5.7–22 pg/m^3 and an annual median of 5.1–15 pg/m^3 at Point Petre; from 1.1 to 1000 pg/m^3 with an annual average of 14–60 pg/m^3 and an annual median of 6.3–23 pg/m^3 at Burnt Island; and from 1.9 to 1300 pg/m^3 with an annual average of 11–63 pg/m^3 and an annual median of 9.0–21 pg /m^3 at Egbert. 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 close to urban centers where control measures are in place. Atmospheric PBDE concentrations are decreasing slowly, with half-lives in the range of 2–16 years (some of these trends are shown in Figure 7). Faster declining trends of PBDEs were observed at Point Petre rather than at Burnt Island. As Point Petre is closer to urban centers, faster declining trends may reflect the phase out of technical BDE mixtures in urban centers while long-range atmospheric transport sustains the air concentrations at Burnt Island. A passive air and water sampling study in Lake Superior in 2011 showed that atmospheric (gaseous) and dissolved PBDEs, in particular BDE-47, were greatest near urban and populated sites (Ruge et al. 2015). Net gaseous deposition of BDE-47 was observed at coastal sites, while the central open lake and at Lake Superior's master station of Eagle Harbor generally displayed volatilization of PBDEs into the atmosphere, mainly of BDE-47.

Rauert et al. (2018a) reported air monitoring results of PBDEs from the Global Atmospheric Passive Sampling (GAPS) Network from 2014 and compared these concentrations with previous measurements. At Toronto, higher Σ13PBDEs levels were seen in 2014 (16-31 pg/m^3) than in 2005 (<MDL-10 pg/m^3) but levels were lower than an active air sampling campaign at this site in 2010−2011 (<0.4-180 pg/m^3) (Shoeib et al., 2014). BDE-209 levels in Toronto were higher in the 2014 GAPS samples (13-20 pg/m^3) than in 2010−2011 (<0.24-11 pg/m^3), but in line with levels seen around the Great Lakes (Gouin et al., 2005; Salamova and Hites, 2011; Liu et al., 2016). The results may be indicating increasing BDE-209 atmospheric concentrations at Toronto, but future monitoring is needed to confirm this tendency.

Perfluorooctane Sulfonate (PFOS)

PFOS measured in precipitation at Point Petre, Burnt Island/Evansville and Sibley under GLB show slow declining trends ranging from 6.9 to 11 years. At the Sibley station, the concentrations were mostly below detection limit. An example of PFOS trend in precipitation at Point Petre derived using Digital Filtration method (DF) is shown in Figure 8. The median concentrations at the 3 sites ranged from 0.59 to 1.5 ng/L. These concentrations were similar to those found in precipitation samples (June 2006 – October 2008) from Japan (average 0.36-0.81 ng/L), but higher than those measured in the USA (average 0.18-0.27 ng/L), Hong Kong (average 0.32 ng/L), India (0.04 ng/L) and France (average 0.11 ng/L) (Kwok et al. 2010). They are similar to the concentrations in rain (November to December 2009) measured in a non-industrial river catchment in Zurich, Switzerland (Median 0.65 ng/L) (Müller et al. 2011).

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). Under GAPS, PFOS concentrations were measured in Toronto using passive air samplers and were found to be 1.5 pg/m^3 (samples from Genualdi et al., 2010 reanalyzed using UPLC-MS/MS), 0.29-3.3 pg/m^3 and 4.0 pg/m^3

for Apr-Jul 2009, Apr-Dec 2013 and Apr-Jul 2015, respectively (Rauert et al., 2018b); also implying a potentially increasing tendency over time.

Perfluorooctanoic Acid (PFOA)

PFOA is also measured in precipitation under GLB at Point Petre, Burnt Island/Evansville and Sibley (2006-2016). The median concentrations of PFOA at the 3 sites ranged from 0.38 to 0.64 ng/L. These concentrations are lower than those reported in precipitation collected from June 2006 to October 2008 in Japan (average 1.81-1.94 ng/L) and USA (average 1.79-3.40 ng/L); and similar to those reported in Hong Kong (average 0.32 ng/L), India (average 0.21 ng/L) and France (average 0.21 ng/L) (Kwok et al. 2010). They are similar to the concentrations in rain (November to December 2009) measured in a non-industrial river catchment in Zurich, Switzerland (median 0.78 ng/L) (Müller et al. 2011). Consistent declining trends were observed at the 3 sites. An example of PFOA trend in precipitation measured at Point Petre derived using the DF Method is given in Figure 8. Halving times range from 4.3 to 7.0 years.

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). In the GAPS program, PFOA measured in air using passive samplers showed concentrations of <3 pg/m^3, 1.0-9.9 pg/m^3 and 15 pg/m^3 for Apr-Jul 2009, Apr-Dec 2013 and Apr-Jul 2015, respectively (Rauert et al., 2018b) implying potentially increasing concentrations over time.

Long-chain Perfluorinated Carboxylic Acids (LC-PFCAs)

Under GLB, LC-PFCAs (C9-12) concentrations in precipitation are available at Point Petre (2006-2016), Burnt Island/Evansville (2006-2016) and Sibley (2006-2010, 2013-2016) (Figure 1). The precipitation sampler has been moved from Burnt Island, which shutdown on March 31, 2013, to the nearby Evansville site in July 2013. This move has not compromised the temporal trends of chemicals found in precipitation. PFOS, PFOA and LC-PFCAs are ionic per- and polyfluoroalkyl substances (PFASs) which are water soluble, thus, it is appropriate to monitor them in precipitation. Their concentrations in air are usually much lower than their precursors which are neutral PFASs. These neutral PFASs are more volatile and are often found in air (see *Other Chemicals of Interest Section* below). At the 3 sites, the medians of Σ_4LC-PFCA (C9-12) ranged from 0.49 to 0.57 ng/L. Perfluorononanoic acid (PFNA) (C9) is the only LC-PFCA that is showing consistent declining trends with halving times ranging from 6.7 to 11 years (unpublished data). Precipitation concentrations of the other measured LC-PFCAs (C10-C12) were not showing consistent increasing or decreasing trends with time.

Rauert et al. (2018b) reported LC-PFCA air concentrations derived from passive air samples collected under the GAPS network. These LC-PFCAs include C9-18 compounds. Their concentrations are given in Table 1.

Mercury

Total Gaseous Mercury (TGM) has been measured at the Egbert station in Ontario (between Lake Huron and Lake Ontario; Figure 1) from December 1996 to December 2016 and the mean concentrations was 1.51 ± 0.31 ng/m^3. During this measurement period, the temporal trend was found to be decreasing at -1.7 (range of -1.9 to -1.5) % per year. Wet deposition was also measured at the same site between March 2000 and December 2015 with a reported mean concentration of 8.1 ng/L. The annual trend for wet deposition was found to be insignificant with a range of -1.2 % to +0.6 % per year (Amanda Cole, personal communication).

Previously, TGM was measured at Point Petre (Nov 1996 – Nov 2007) and Burnt Island (May 1998 –Nov 2007) and the reported average concentrations were 1.75 ± 0.33 ng/m^3 and 1.55 ± 0.22 ng/m^3, respectively, with declining trends of -1.7 (range of -2.2 to -1.2) % per year and -2.5 (-3.4 to -1.6) % per year, respectively. Wet deposition of mercury was also measured at 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 (Amanda Cole, personal communication). Zhang et al. (2016) also reported that atmospheric mercury concentrations (Hg^0) and mercury wet deposition (Hg^{II}) fluxes have generally declined since the 1990s. Atmospheric Hg^0 concentrations have decreased about 2% per year since 2005 as measured in Canada's Experimental Lakes Area (west of Lake Superior). Wet deposition measurements from the North American Mercury Deposition Network follow these trends with fluxes decreasing about 1.6% per year since 1996. Zhang et al. (2016) suggest that reduced emissions from utilities over the past few decades and the phase-out of mercury in many commercial products has led to lower global anthropogenic emissions and associated deposition to ecosystems.

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 Δ^{200}Hg, 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 a lake's sediment. This finding suggests that atmospherically derived Hg may be a more important source of methyl mercury, which 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)
The median HBCDD concentrations were 2.0 pg/m3, 2.1 pg/m3, 1.7 pg/m3 and 5.2 pg/m3 for Chicago, Cleveland, Sturgeon Point and Sleeping Bear Dunes, respectively. Higher levels of HBCDD were observed at the remote site of Sleeping Bear Dunes with comparable levels at the remaining three sites. a-HBCDD and g-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 (2009-2012) were found to be 0.17-0.50 pg/m^3 and 0.054-0.38 pg/m^3 for Burnt Island and Point Petre, respectively. HBCDD was only detected in less than 40% of all samples and the data are too sparse to develop temporal trends (Shunthirasingham et al., 2018). HBCDD is also measured in air in Toronto under the GAPS network with passive air sampler, but in 2014, HBCDD was non-detectable in all samples (Rauert et al., 2018a). HBCDD is listed under Annex A (Elimination) of the Stockholm Convention on POPs with its production and use to be eliminated globally. Although HBCDD is a CMC, it is not included in the assessment of this indicator due to the lack of data.

Short-Chain Chlorinated Paraffins (SCCPs)
Under GAPS, SCCPs have been measured in air using passive air sampling method at the Toronto site. The concentrations found were 221.1 pg/m^3 in 2006, 5.5 pg/m^3 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. 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 Canadian sits (Shunthirasingham et al. 2016). Concentrations of α-HCH and γ-HCH are decreasing rapidly in air, with halving times of 4-5 years at Canadian sites (Shunthirasingham et al. 2016) and about 4 years at U.S. sites (Salamova et al. 2015); see Figure 9 and 10. These are the most rapid halving times observed for any compound measured as part of IADN/GLB.

The insecticides, α-endosulfan and β-endosulfan were taken off the market in July 2016. Its vapor phase atmospheric concentrations around the Great Lakes are decreasing with halving times ranging from 7 to 12 years (Salamova et al. 2015, Shunthirasingham et al. 2016) (Figure 10). Based on estimated use rates of endosulfan in the U.S. from 1997 to 2009, Salamova et al. 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. Its concentrations in precipitation are decreasing with halving times of about 8 years at all sites (Venier et al., 2016). The satellite station of Egbert, located between Lakes Ontario and Huron and surrounded by agricultural cropland, showed high concentrations of dichlorodiphenyltrichloroethanes (DDTs), dieldrin, γ-HCH (lindane), and endosulfan compared to the more remote master stations on Lakes Huron and Ontario. This observation was attributed to historical (DDTs, lindane, and dieldrin) and current (endosulfan) agricultural applications of these OCPs in the area. These observations suggest that agricultural areas are a source of OCPs to the lakes (Shunthirasingham et al. 2016).

Isomer-specific data provide insights on the temporal trends and possible sources of specific compounds. The relative proportion of *o,p'*-DDT to *p,p'*-DDT in air has increased significantly at five U.S. sites and 2 Canadian sites over the last two decades. It is suggested that dicofol (a pesticide manufactured from DDT), which has higher *o,p'*-

/*p,p'*-DDT ratios than the commercially-used technical DDT mixture, may now be a significant, additional source of DDT to the Great Lakes (Venier and Hites 2014; Shunthirasingham et al. 2016). The average ratio of the concentration of γ-HCH (lindane) versus the sum of the concentrations of γ-HCH + α-HCH did not vary significantly with time, but it did show an urban signature, suggesting that cities may be more important sources of these compounds than previously suspected.

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 (see Figure 3) (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 (endosulfan was taken off the market in July 2016). 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).

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 alternative halogenated flame retardants [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. The atmospheric EH-TBB and BEH-TEBP concentrations were also significantly and rapidly increasing at all the five sites, with doubling times of 2–6 years (see Figure 6C for an example of these trends).

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-tri-bromophenyl 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)]. The temporal trends for non-PBDE HFRs in air at the Canadian sites are not clear and the absences of clear temporal effects suggest that there are continued inputs of non-PBDE HFRs into the environment. PBT concentrations are decreasing at Burnt Island, with a half-life of 2.1 years but seem to be increasing at Point Petre, with a doubling time of 6.8 years (Figure 11). TBP-AE and HBBz concentrations at both Point Petre and Burnt Island were not changing. The concentrations of *anti*-DDC-CO in the atmosphere are decreasing, with a half-life of 3.4 years at Burnt Island, but the concentrations are increasing

very slowly at Point Petre, with a doubling time of 17 years. *Syn*-DDC-CO levels are decreasing, with a half-life of 15 years at the Point Petre, but the levels are increasing, with a doubling time of 7.6 years at the Burnt Island.

Organophosphate esters (OPE) are industrial chemicals that are widely used as flame retardants, plasticizers, anti-foaming agents, and as additives in hydraulic liquids, lacquers, and floor polishes. These chemicals have been in heavy use for decades, and due to worldwide restrictions on brominated flame retardants, OPE production and use are expected to increase in the future. 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; Figure 12 (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).

At the GAPS site of Toronto, several HFRs and OPEs were detectable in passive air samples (Rauert et al., 2018a, Lee et al., 2016). Results are summarized in Table 2.

Per- and polyfluoroalkyl substances (PFASs) (other than PFOS, PFOA and LC-PFCAs)
PFAS precursor compounds, such as fluorotelomer alcohols (FTOHs) and fluorinated sulfonamides and sulfonamidoethanols (FOSAs and FOSEs), can transform or degrade to PFOA and PFOS in the environment. They are more volatile than the ionic PFASs and are often found in air. PFOS precursors measured in the air of Toronto showed average concentrations of *N*-methyl perfluorooctanesulfonamidoethanol (*N*-MeFOSE) of 101 pg/m^3 and *N*-ethyl perfluorooctanesulfonamidoethanol (*N*-EtFOSE) of 205 pg/m^3 (Martin et al. 2002). Rauert et al. (2018b) found that 6:2FTOH and 8:2FTOH dominated the profiles of all PFASs measured in passive air samples collected under GAPS in the urban site of Toronto. FOSAs were detected more frequently than the FOSEs. A lower 8:2FTOH ratio to other FTOHs indicates direct emissions from localized sources because 8:2 FTOH has the longest atmospheric residence time (Wang et al., 2014). Lower 8:2FTOH ratios have been reported in Toronto (Ahrens et al., 2012, 2013) as compared to remote relations, e.g. in the Arctic region, consistent with urbanized region as a source of FTOHs.

At Toronto, levels of perfluorosulfonic acids (PFSAs, including PFOS) from an active air sampling campaign in 2010 of 0.38-3.12 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 (0.29-4.69 pg/m^3) under GAPS, but lower than the levels seen in 2015 (8.1 pg/m^3). The GAPS study results from 2013 to 2015 indicates a significant increase in concentrations ($p < 0.05$) of the PFSAs (Rauert et al., 2018b).

Polycyclic Aromatic Hydrocarbons (PAH)
IADN data for total PAH concentrations (see Figure 13) 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 deposition 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 have decreased slowly over the period 1992–2015 (see Figure. 3). 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).

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). 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).

Lead

Lead was measured on airborne particles in the Great Lakes Basin (GLB) at three stations, namely Point Petre (1988-2017), Burnt Island (1992-2013) and Egbert (1988-2006) (unpublished data GLB). It was found that temporal trends of lead in airborne particles are generally declining in Great Lakes air with half-lives ranging from 13 to 15 years (Figure 14).

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 would 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 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.
- 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 by other researchers 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 and SCCP 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 UNEP Minamata Convention on Mercury, with 98 parties including U.S. and Canada, came into effect on 16th August, 2017, is a legally-binding international treaty to protect human health and the environment from anthropogenic emissions and release of mercury and mercury compounds. Major regulatory activities of the Minamata Convention include a ban on new mercury mines, the phase-out of existing ones, the phase out and phase down of mercury use in a number of products and processes, control measures on emissions to air and on releases to land and water, and the regulation of the informal sector of artisanal and small-scale gold mining. The Convention also addresses interim storage of mercury and its disposal once it becomes waste, sites contaminated by mercury as well as health issues.

The agricultural chemical lindane was recently phased out in the U.S. and Canada, and endosulfans were phased out in the U.S. and Canada by 2016 (Federal Register, 2010; Health Canada Pest Management Regulatory Agency, 2011).

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.

World-wide, the largest remaining source of mercury emissions to the atmosphere is coal-fired power plants. Regionally, many sources 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. 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 emerging chemicals of concern.

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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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@canada.ca, 613-995-9031.
Link to IADN websites: http://epa.gov/greatlakes/monitoring/air2/index.html

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List of Tables

Table 1. LC-PFCAs measured in air in Toronto (pg/m^3) using passive air samplers
Source: Rauert et al. (2018b)
Table 2. HFRs and OPEs measured in air in Toronto (pg/m^3) using passive air samplers
Source: Rauert et al. (2018a), Lee et al. (2016)

List of Figures

Figure 1. Great Lakes Basin map showing the United States' Integrated Atmospheric Deposition Network (IADN) and Canada's Great Lakes Basin (GLB) Monitoring and Surveillance master monitoring stations Eagle Harbour, Sleeping Bear Dunes, Burnt Island (closed March 2013, replaced by Evansville started 2014), Sturgeon Point and Point Petre, one near each of the Great Lakes; urban stations of Chicago and Cleveland; satellite station of Egbert and precipitation measurement station at Sibley.
Source: Hayley Hung, personal communication
Figure 2. A. Annual geometric average vapor phase concentrations of all measured PCB congeners summed together and plotted as a function of sampling year. The open circles in the Chicago (2007) and Sturgeon Point (1996) panels represent outliers and were not used for the regressions shown here. Note the concentration scales for the Chicago and Cleveland data are 10 times higher than for the other sites. Plots from Salamova et al, (2015) were updated to include data through 2016.
B. Trends of vapour phase concentrations of PCB52 and PCB101 at Burnt Island and Point Petre. r^2 is the linear coefficient of determination with the fitted trend line derived using the Digital Filtration method.
Plots from Shunthirasingham et al. (2016) updated to include Point Petre data through 2015. Burnt Island station was shut down in 2013, therefore no new data for update.
Source: Salamova et al. (2015) (data updated through 2016), Shunthirasingham et al. (2016) (data updated to 2015)
Figure 3. Total yearly flows of PCB-52, γ-HCH, *p,p'*-DDT, and phenanthrene into and out of the five Great Lakes between 1992 and 2015. Black lines are first order rate curves ($t_{1/2}$ is a halving time and t_2 is a doubling time, in years). Lines that are missing indicate that the exponential regression was not significant ($p > 0.05$). (Figure modified from Guo et al., 2018).
Source: Guo et al. (2018)
Figure 4. Ratio of the concentrations of PCB-11 relative to those of total Aroclor-PCBs. The data have been corrected for population variations among the six sampling sites. The red line shows the periodicity of the ratios, and the green line shows the long-term temporal trend in the ratios. These ratios are doubling every 13 ± 1 years.
Source: Hites (2018)
Figure 5. Estimated total PCB mass budget flows (kg/yr) and inventories (kg) for 2010-2015 and comparison to the 1994-1995 mass balance results based on the MICHTOX model (in parentheses) in Lake Michigan. The blue and green layers represent water and sediment layers, respectively. The thickness of the arrows indicates the magnitude

of flows in 2010-2015.
Source: Guo et al. (2017)
Figure 6. Vapor plus particle phase, atmospheric concentrations of (A) BDE-47 and (B) BDE-209 measured at Cleveland, OH, every 12 days starting in January 2005 and ending in December 2015 and (C) EHTBB and BEHTBP (concentrations summed together) measured at Cleveland starting in January 2008 and ending in December 2015. The values are given as the natural logarithms of the concentrations in pg/m^3. The regression lines (red) were all significant at $P < 0.005$. The halving times ($t_{1/2}$) and doubling times (t_2) are given in years. Plots from Venier et al. (2015) were updated to include data through 2015.
Source: Venier et al. (2015) (data updated to 2015)
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Source: Shunthirasingham et al. (2018) (data updated to 2016)
Figure 12. Atmospheric concentrations of total OPE (ΣOPE), shown as their natural logarithms, in the particle phase at the urban (Chicago and Cleveland) and at the rural and remote (Sturgeon Point, Sleeping Bear Dunes, and Eagle Harbor) IADN sites. The red lines show the significant regressions. The regression lines (red) were all significant at $P < 0.005$. The halving times ($t_{1/2}$) and doubling times (t_2) are given in years. The Julian Day is the number of days starting from 1 January 2012.
Source: Salamova et al. (2016)
Figure 13. Annual geometric average of vapor plus particle phase concentrations of all the measured polycyclic aromatic hydrocarbons (PAH) summed together and plotted as a function of sampling year. The open circles in the Cleveland (2012) and Point Petre (2008) panels represent outliers and were not used for the regressions shown here. The regression for the Sleeping Bear Dunes data was not statistically significant at $P < 0.05$. Plots from Salamova et al. (2015) were updated to include data through 2016.
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Source: Hayley Hung, personal communication

Last Updated
State of the Great Lakes 2019 Technical Report

Sampling period	PFNA (C9)	PFDA (C10)	PFUnDA (C11)	PFDoOA (C12)	PFTriDA (C13)	PFTeDA (C14)	PFHxDA (C16)	PFODA (C18)
2009 Apr-Jul	<0.2	0.24	0.12	<0.07	<0.08	<0.08	<0.08	<0.08
2013 Apr-Dec	<0.2-0.33	<0.3-2.2	<0.05-1.1	<0.07-0.10	<0.08	<0.08	<0.08	<0.09
2015 Apr-Jul	5.3	7.4	2.3	0.84	0.44	0.39	<0.07	<0.08

Table 1. LC-PFCAs measured in air in Toronto (pg/m^3) using passive air samplers
Source : Rauert et al. (2018b)

Sampling period	EH-TBB	HBBz	BTBPE
2014	5.2-10	0.30-0.36	<MDL-1.6
2005	<0.09-1.6	<0.02-0.53	

Sampling period	TEP	TnBP	TCEP	TCPP	TPP	m-TTP	TBEP	TDCPP	TEHP
2014 Jan-Jun	3.7-13	75-94	24-130	86-710	9.3-31	17-25	82-170	4.7-17	<0.06-2

For the HFRs, 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB), hexabromobenzene (HBBz), 1,2-bis(2,4,6-tribromophenoxy)ethane (TBEBTBPE).

For the OPEs, tri-ethyl phosphate (TEP), tri-nbutyl phosphate (TnBP), tris(2-chloroethyl) phosphate (TCEP), tris(chloroisopropyl) phosphate (TCPP), triphenyl phosphate (TPP), tri-m-tolyl phosphate (m-TTP), tris(2-butoxyethyl) phosphate (TBEP), tris(1,3-dichloro-2-propyl) phosphate (TDCPP), tris(2-ethyl hexyl) phosphate (TEHP).

Table 2. HFRs and OPEs measured in air in Toronto (pg/m^3) using passive air samplers
Source: Rauert et al. (2018a), Lee et al. (2016)



Figure 1. Great Lakes basin map showing the United States' Integrated Atmospheric Deposition Network (IADN) and Canada's Great Lakes Basin (GLB) Monitoring and Surveillance master monitoring stations Eagle Harbour, Sleeping Bear Dunes, Burnt Island (closed March 2013, replaced by Evansville started 2014), Sturgeon Point and Point Petre, one near each of the Great Lakes; urban stations of Chicago and Cleveland; satellite station of Egbert and precipitation measurement station at Sibley.
Source: Hayley Hung, personal communication

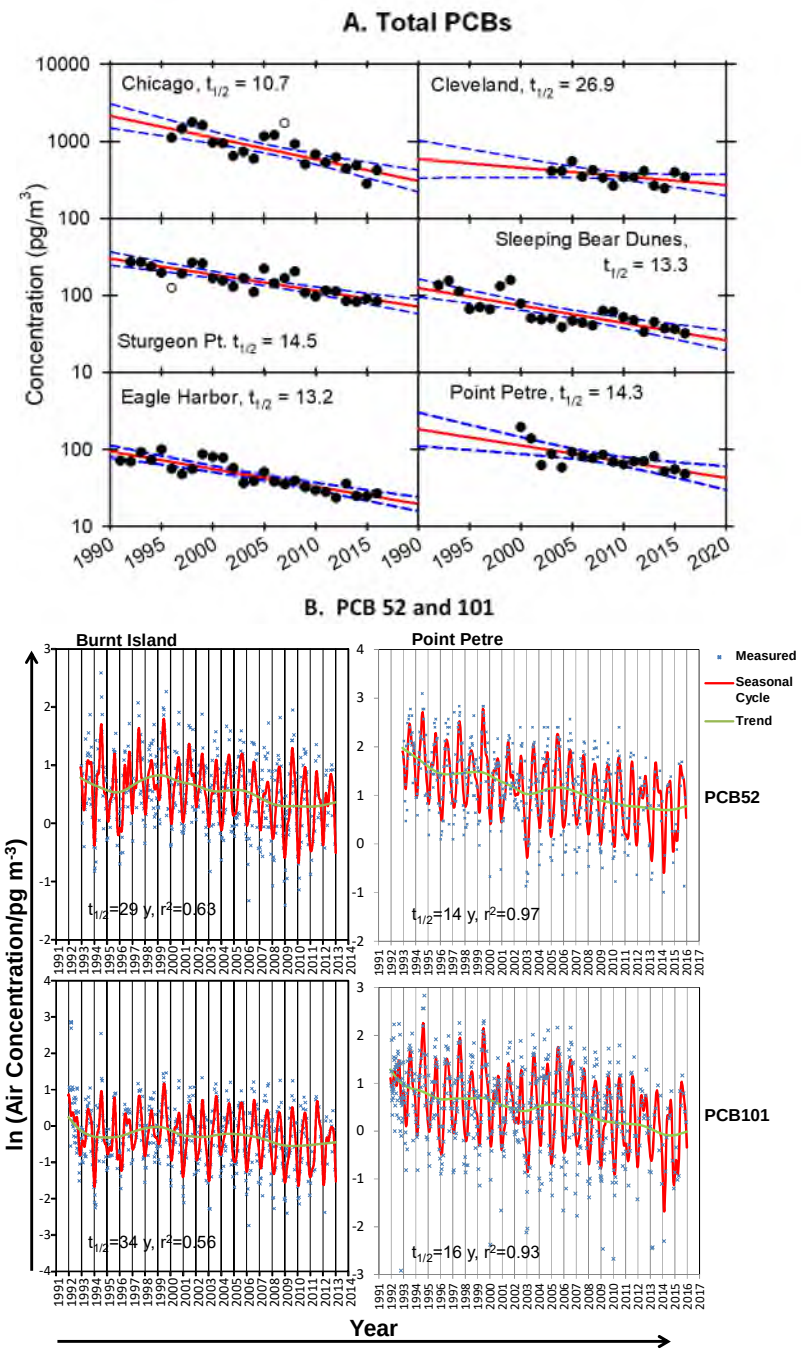


Figure 2. A. Annual geometric average vapor phase concentrations of all measured PCB congeners summed together and plotted as a function of sampling year. The open circles in the Chicago (2007) and Sturgeon Point (1996) panels represent outliers and were not used for the regressions shown here. Note the concentration scales for the Chicago and Cleveland data are 10 times higher than for the other sites. Plots from Salamova et al, (2015) were updated to include data through 2016.
B. Trends of vapour phase concentrations of PCB52 and PCB101 at Burnt Island and Point Petre. r^2 is the linear coefficient of determination with the fitted trend line derived using the Digital Filtration method.
Plots from Shunthirasingham et al. (2016) updated to include Point Petre data through 2015. Burnt Island station was shut down in 2013, therefore no new data for update.
Source: Salamova et al. (2015) (data updated through 2016), Shunthirasingham et al. (2016) (data updated to 2015)

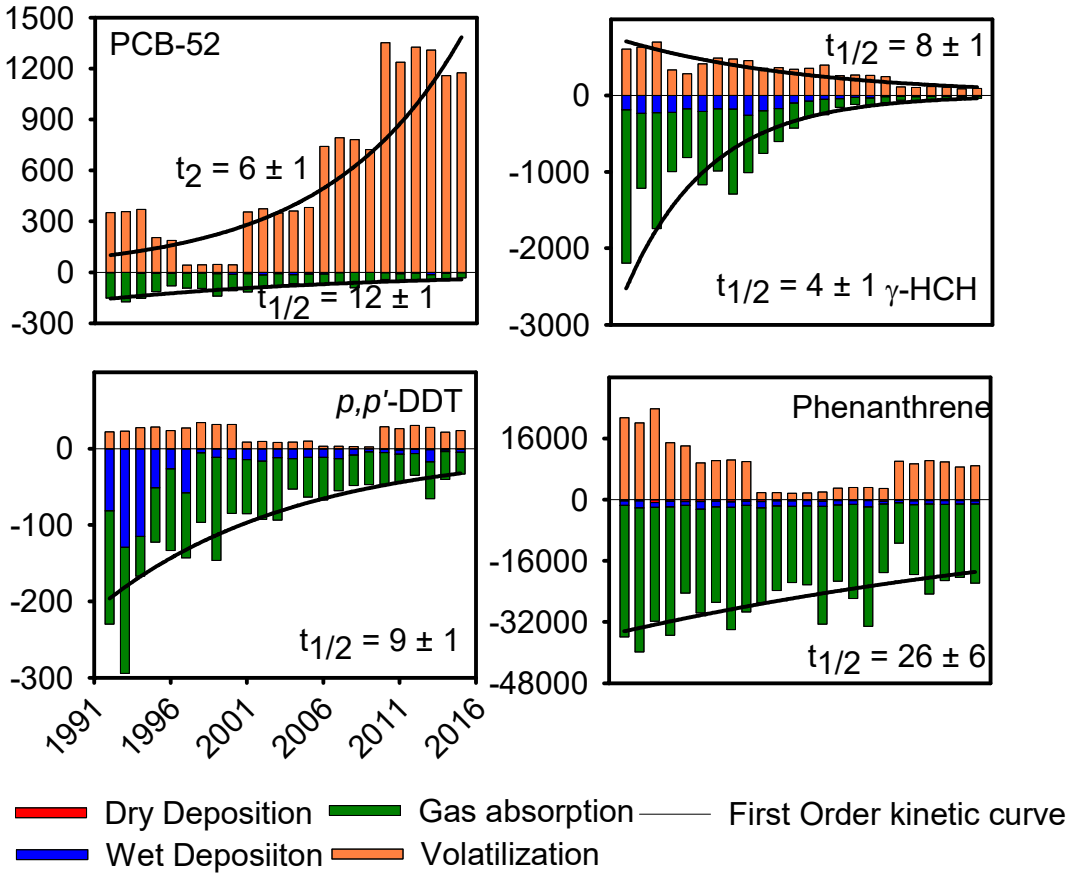


Figure 3. Total yearly flows of PCB-52, γ-HCH, *p,p'*-DDT, and phenanthrene into and out of the five Great Lakes between 1992 and 2015. Black lines are first order rate curves ($t_{1/2}$ is a halving time and t_2 is a doubling time, in years). Lines that are missing indicate that the exponential regression was not significant ($p > 0.05$). (Figure modified from Guo et al., 2018).
Source: Guo et al. (2018)

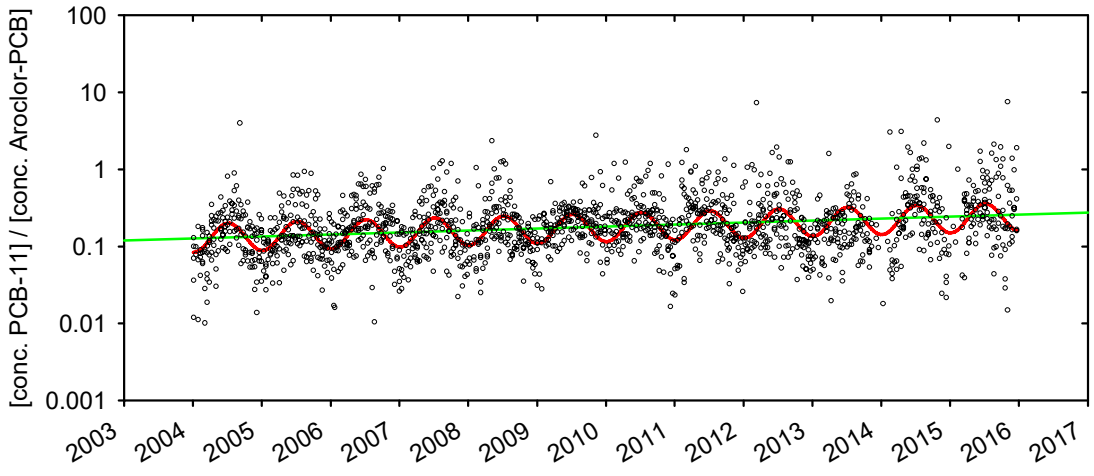


Figure 4. Ratio of the concentrations of PCB-11 relative to those of total Aroclor-PCBs. The data have been corrected for population variations among the six sampling sites. The red line shows the periodicity of the ratios, and the green line shows the long-term temporal trend in the ratios. These ratios are doubling every 13 ± 1 years.
Source: Hites (2018)

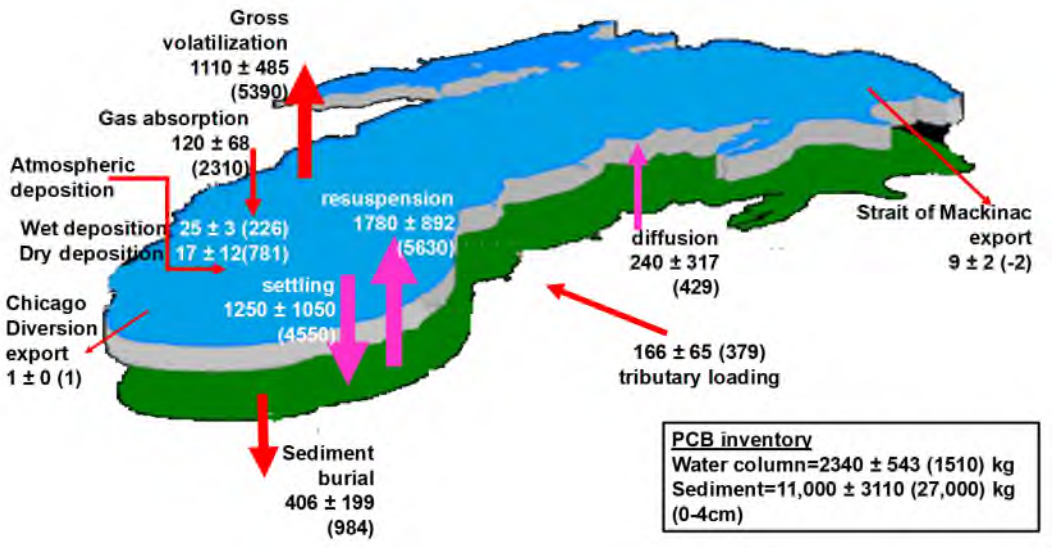


Figure 5. Estimated total PCB mass budget flows (kg/yr) and inventories (kg) for 2010-2015 and comparison to the 1994-1995 mass balance results based on the MICHTOX model (in parentheses) in Lake Michigan. The blue and green layers represent water and sediment layers, respectively. The thickness of the arrows indicates the magnitude of flows in 2010-2015.
Source: Guo et al. (2017)

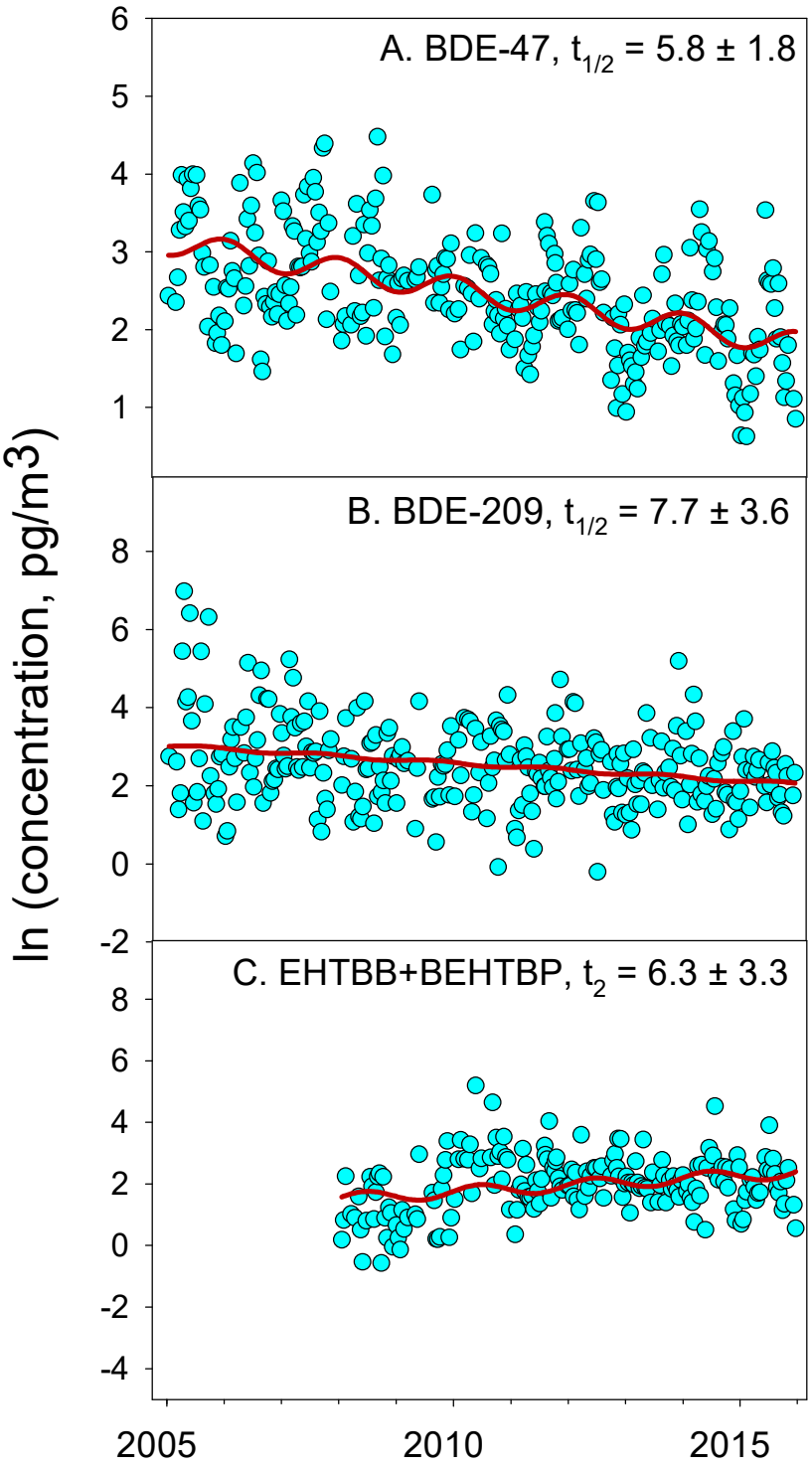


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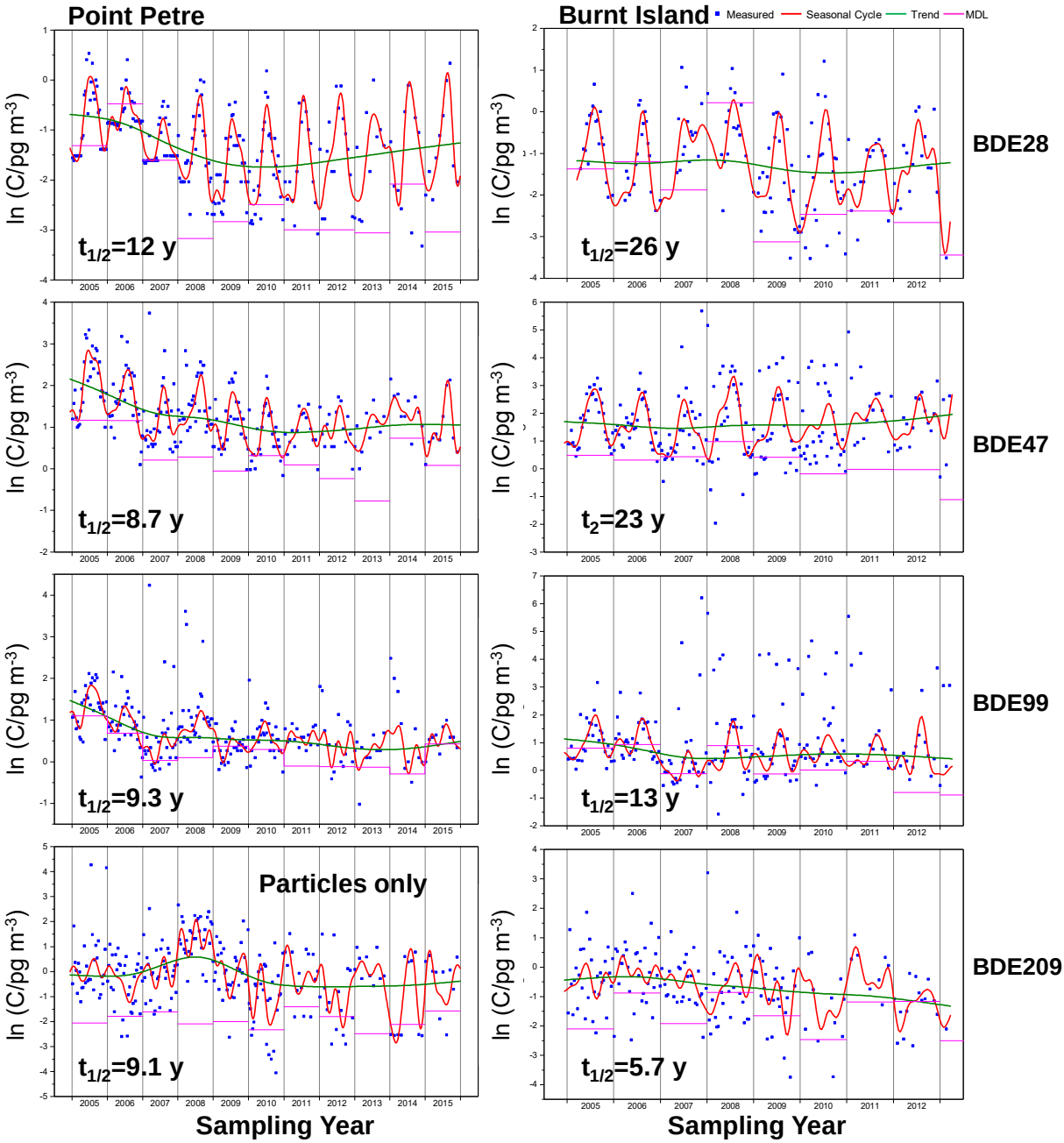


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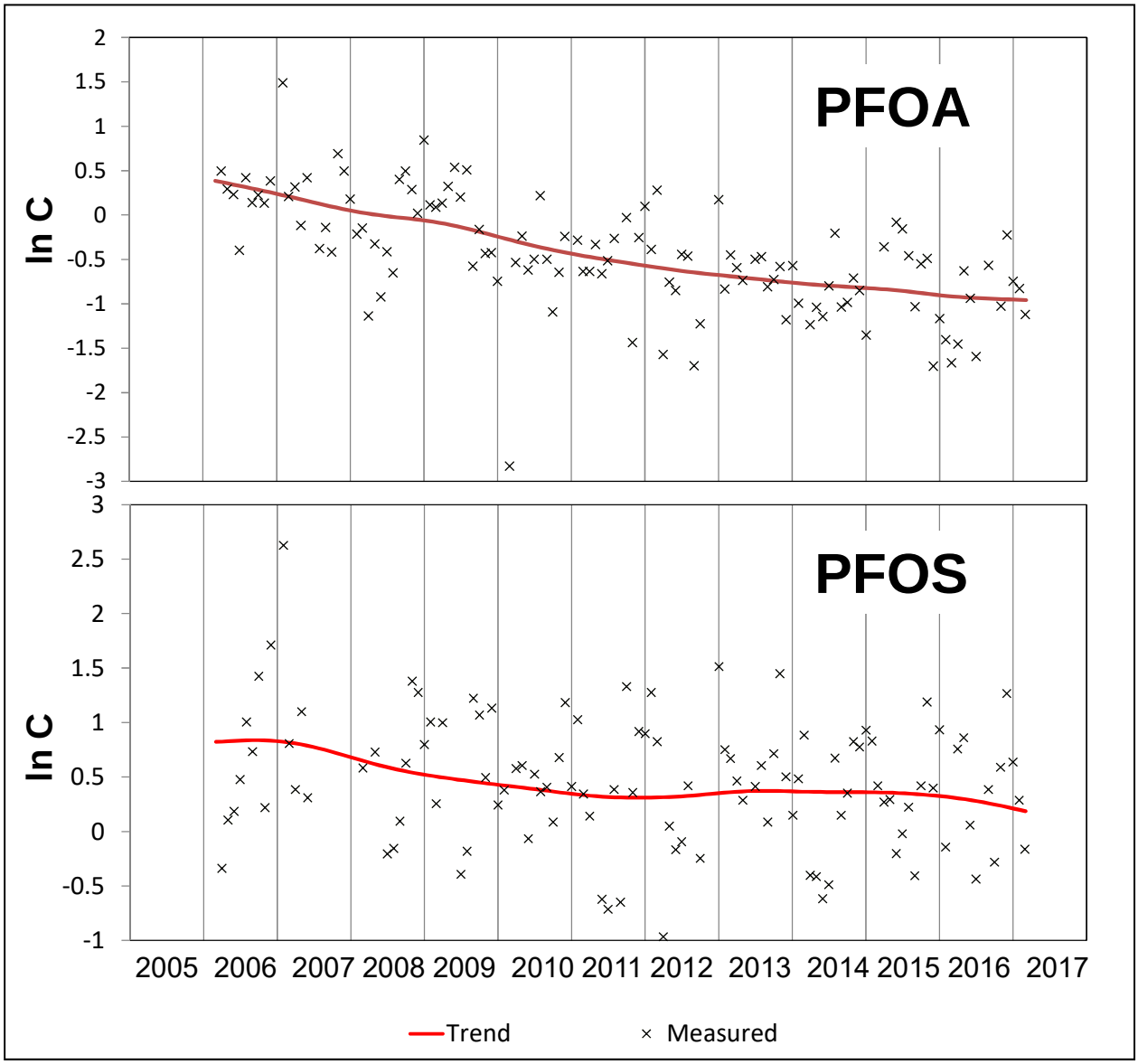


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Source: Sean Backus and Lisa Bradley, personal communication; Hayley Hung, personal communication

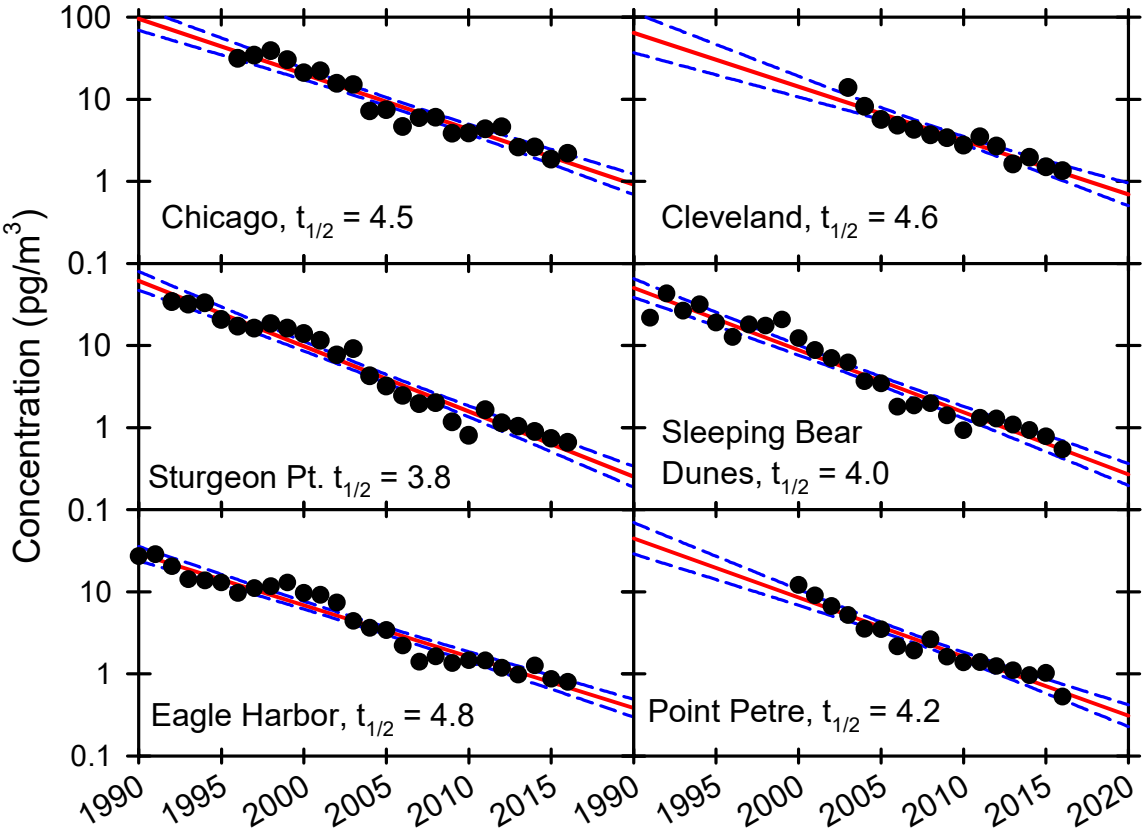


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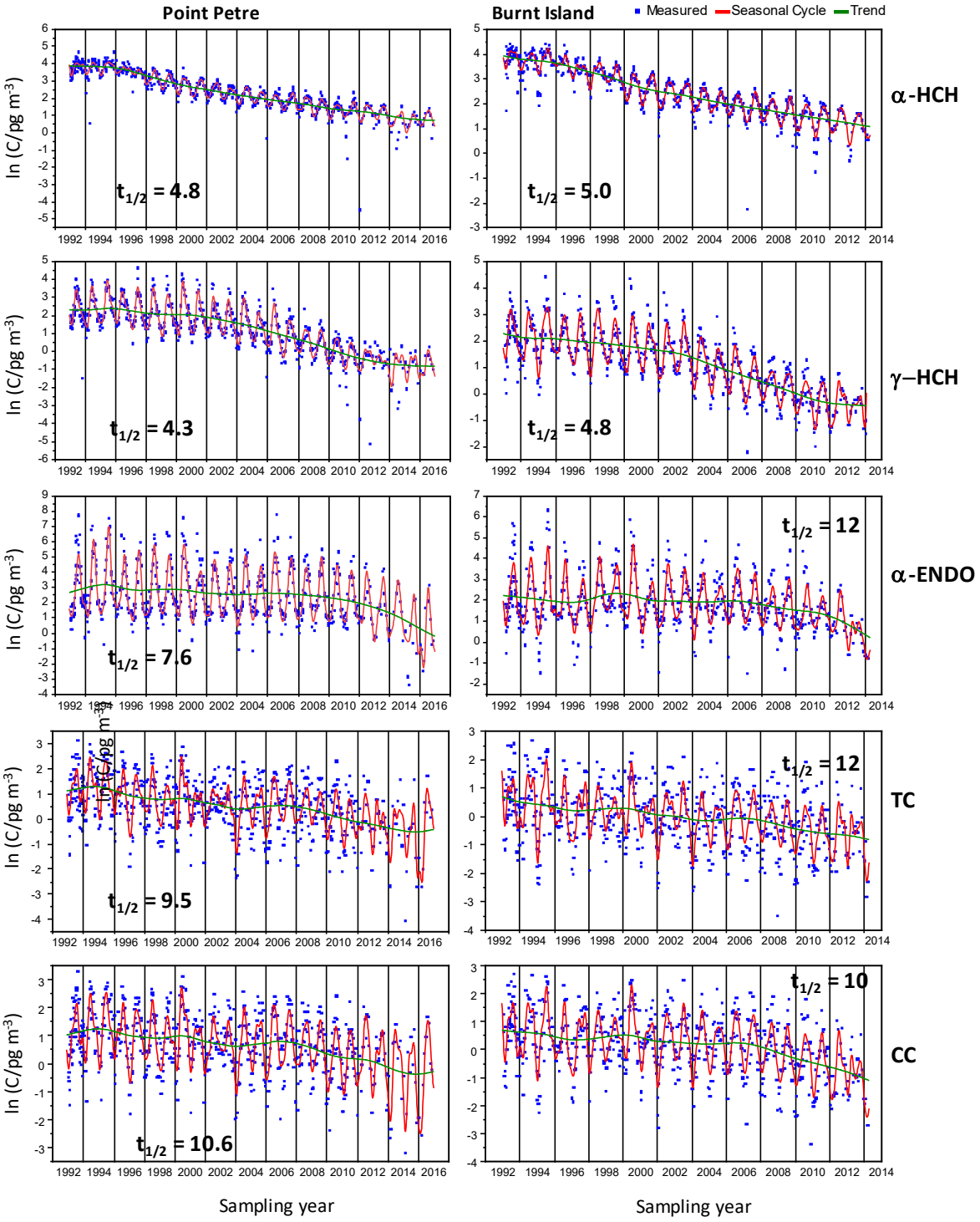


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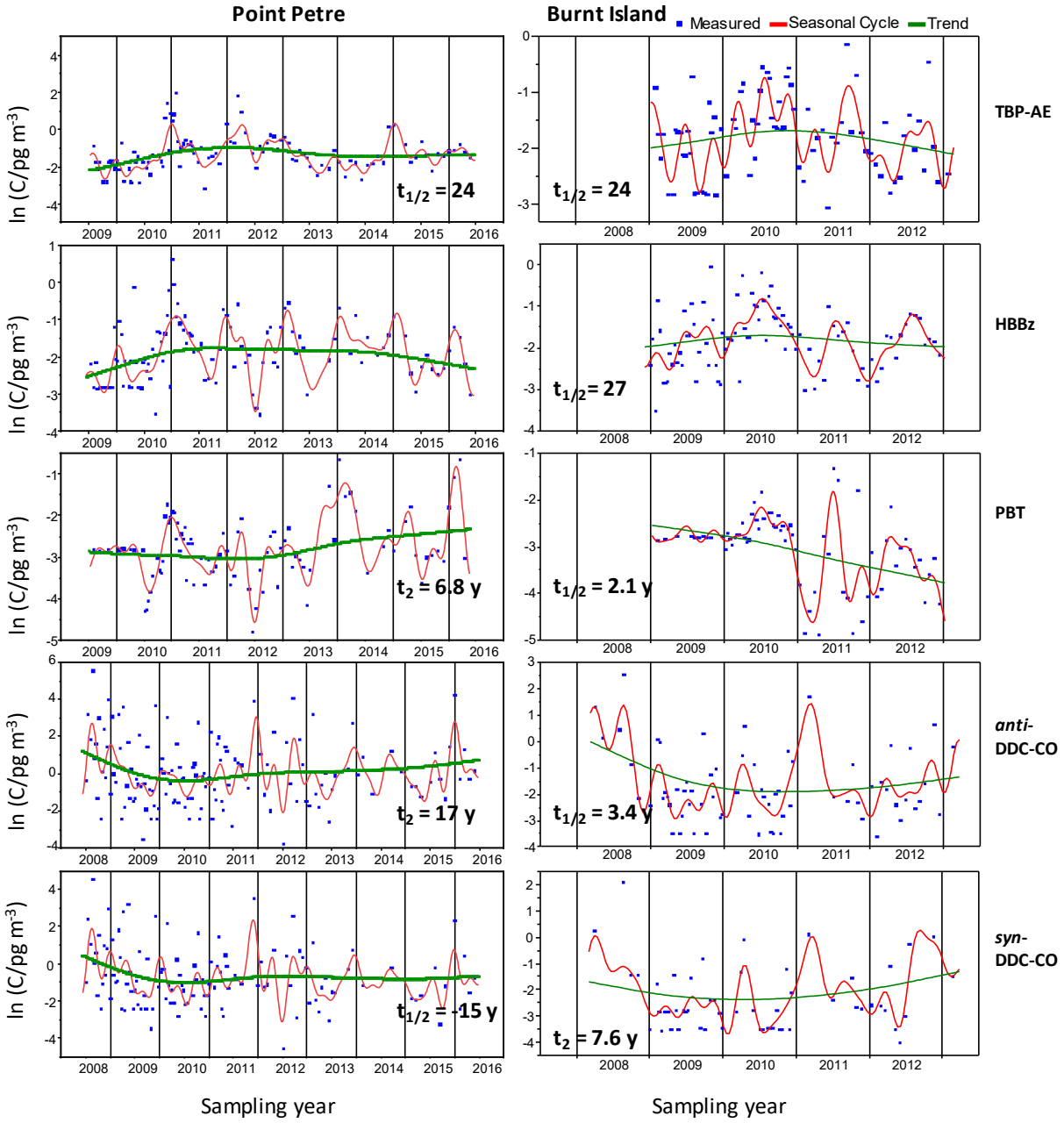


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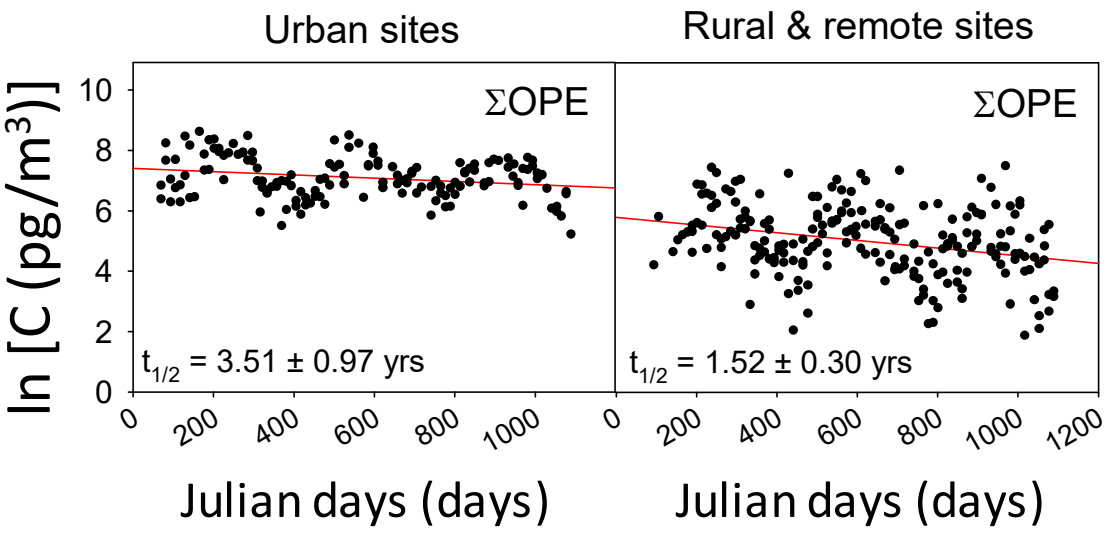


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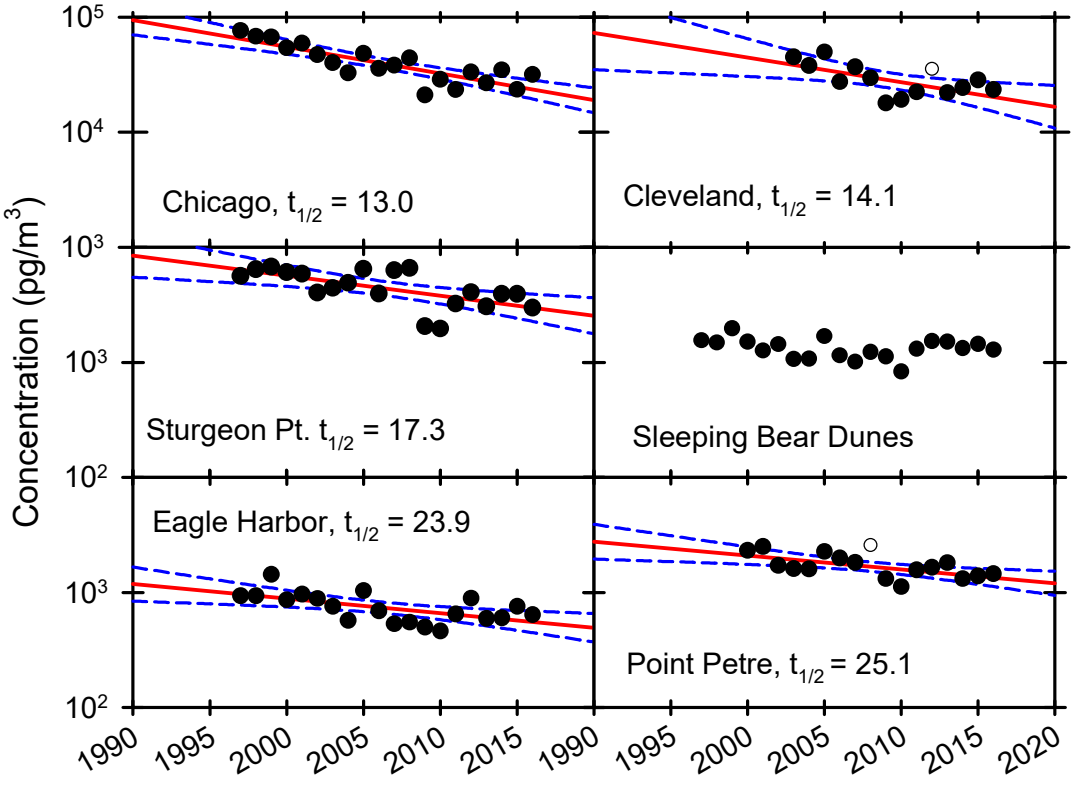


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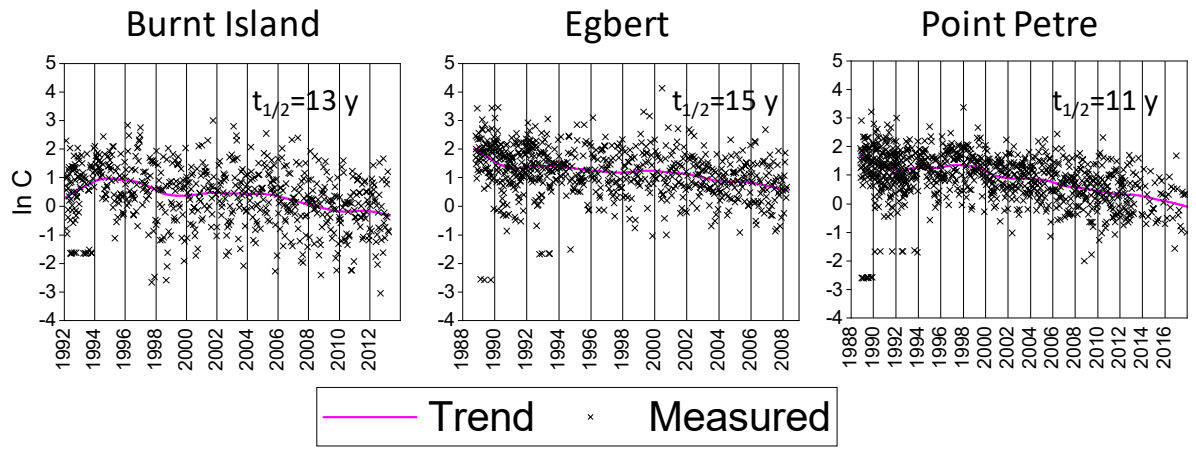


Figure 14. Temporal trends of lead measured in air-borne particles at Burnt Island, Egbert and Point Petre. Source: Hayley Hung, personal communication

Sub-Indicator: Coastal Wetland Invertebrates

Overall Assessment
Status: Fair
Trends:
10-Year Trend (2011-2017)*: Undetermined
Long-term Trend: Undetermined
Rationale: As of 2017, the vast majority of wetland sites are not in the degraded or extremely degraded categories (based on six categories: reference conditions, mildly impacted, moderately impacted, moderately degraded, degraded, and extremely degraded) based on the health of coastal wetland invertebrate communities. The health of macroinvertebrate communities appears to respond to local conditions, permitting highly degraded and less degraded wetlands to persist in the same region. Approximately 48% of wetlands sampled between 2011 and 2017 were of reference or mildly impacted quality based on invertebrate communities; this proportion changed throughout the sampling period, peaking in 2012 at 78% and lowest in 2016 at 37%. In total, 3% of sampled wetlands were of degraded or extremely degraded quality and ranged from 0-6% over the 7 years. The proportion of moderately impacted and moderately degraded sites fluctuated over the years with no discernable pattern. In the previous report, the health of macroinvertebrate communities appeared to be degrading based on a comparison over a 3-year period. With the inclusion of 7 years of data, it appears that conditions fluctuate year-to-year. In part, this may be due to each year having a different pool of sites sampled and assessed. Furthermore, macroinvertebrate communities appear to respond to localized conditions.* Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected. In previous reports, no lake-by-lake assessments were provided.

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 (2011-2017)*: Deteriorating
Long-term Trend: Undetermined
Rationale: No Lake Superior coastal wetland sites fall into the degraded or extremely degraded category based on the health macroinvertebrate communities (Figures 3 and 4). Furthermore, only 15.25% of sites were determined to be moderately degraded. Overall, 18.6% of sites sampled were of reference quality. However, the number of reference quality sites has decreased over the 7-year sampling period and in 2015-2017 no sites were of reference quality. During these years there was an increase in the proportion of mildly impacted and moderately impacted sites. This trend should be interpreted with caution as data are only available for 5 to 13 sites each year. *Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Michigan
Status: Fair
10-Year Trend (2011-2017)*: Undetermined
Long-term Trend: Undetermined
Rationale: The vast majority of Lake Michigan coastal wetlands are not of degraded quality based on macroinvertebrate community health. In total, there were no extremely degraded sites, 7.6% were degraded, 9.1% moderately degraded, 37.9% moderately impacted, 33.3% mildly impacted and 12.1% in reference condition (Figure 5). In 2011 and 2013 no sites were of reference quality and in 2013, 2014 and 2017 there were no degraded sites. The proportion of sites falling into each category varied from year-to-year within the lake (Figure 6) and each year a different subset of sites were sampled. This observation suggests localized impacts on invertebrate communities. Data are available for only 4-14 sites each year so an overall assessment of wetlands of Lake Michigan should be considered

with caution. *Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend (2011-2017)*: Deteriorating
Long-term Trend: Undetermined
Rationale: The vast majority of Lake Huron coastal wetlands are not of degraded quality based on macroinvertebrate community health. One wetland, in 2016, was found to be extremely degraded and, in total, 2.2% of Lake Huron wetlands were degraded (Figure 7 and 8). The proportion of moderately degraded wetlands fluctuated across years but remained below 19%. Between 2011 and 2013, 21.4-37.5% of wetlands were of reference quality. Reference quality sites were less common in subsequent years (3-9.7%) and the proportion of mildly impacted or moderately impacted sites increased over the same time interval. * Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Status and trends could not be determined for Lake Erie as data were only available for 11 sites over the 7 year period.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Status and trends could not be determined for Lake Ontario as data were only available for 20 sites over the 7 year period.

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 category.
Poor: The vast majority of the wetlands are in the degraded category.
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.

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 health by measuring the composition and density 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 and agricultural development and waste disposal activities should be identified, preserved and, where necessary, rehabilitated. Conducting monitoring and surveillance activities will gather definitive information on the location, severity, aerial or volume extent, and frequency of the monitoring of Great Lakes 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).

Measure

This sub-indicator will measure the relative abundance of sensitive taxa (e.g., mayflies, caddisflies), tolerant taxa (e.g. *Chironomini* as a proportion of total *Chironomidae* abundance, *Isopoda*), richness of specific taxa, and functional feeding groups (e.g., herbivores, detritivores, carnivores). Macroinvertebrate samples should be collected annually from the dominant plant zones in each wetland using dip nets in accordance with standard protocols as initially developed by the Great Lakes Coastal Wetland Consortium (GLCWC) and further developed by the Great Lakes Coastal Wetlands Monitoring Program (GLCWMP). Plant zones are defined as patches of vegetation in which a particular plant type or growth form dominates the plant community based on visual coverage estimates. Numerous replicate samples are collected from each plant zone within each wetland. Samples should be collected annual and depending on latitude and wetland type during either June, July, or August when vegetation has developed. Southern drowned river mouths should be sampled during June while lacustrine sites should be sampled during July in the south latitudes and during August in the northern latitudes.

The GLCWC developed indices of biological integrity (IBIs) in 2002 and protocols were finalized in 2008 (GLCWC 2008). Since 2010 the GLCWMP has continued development (Uzarski et al. 2017). Individual IBIs can be used independently as a measure of coastal wetland health, based on a percentage of points possible reflected as 'reference conditions' to 'extremely degraded'. The sub-indicator has been used basin wide (U.S. and Canada wide) over the past four years and much longer in some regions. This sub-indicator can also be 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 indicators in combination, because they function and indicate anthropogenic disturbance at different spatial and temporal scales and have varying resolution of detection. For example, invertebrates detect much more local disturbance of the lakeward portion of the wetland within regions. This indicator in particular detects very local changes to the system, and therefore, it is not uncommon for two adjacent wetlands with similar structure to have very different indicator values. This is a particular strength of the invertebrate-based IBI.

The invertebrate IBI has been applied to coastal wetlands basin-wide by a syndicate of universities, government agencies, and a private partner from 2011 to 2017. IBI scores were primarily based on richness and relative abundance of Odonata; richness and relative abundance of Crustacea plus Mollusca taxa; total genera richness; relative abundance of Gastropoda; relative abundance of Sphaeriidae; richness of Ephemeroptera plus Trichoptera taxa; relative abundance Isopoda; relative abundance of Amphipoda; Evenness; Shannon Diversity Index; and Simpson Index. See https://www.greatlakeswetlands.org/Home.vbhtml 'Documents' for details on indicator metrics.

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 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).

To restore/maintain the overall biological integrity of Great Lakes coastal wetlands, the various ecological components need to be adequately represented. The Great Lakes Coastal Wetland Consortium (GLCWC)-adopted

Index of Biotic Integrity (IBI, Uzarski *et al.* 2004) approach, which was further developed by the Great Lakes Coastal Wetland Monitoring Program (GLCWMP), offers information on overall diversity of the invertebrate community and trends over time (Uzarski et al. 2017). The presence, diversity and abundance of invertebrates tend to correlate with factors such as water depth, vegetation, and sediment type. Such localized conditions influence the invertebrate community present in each wetland. Therefore, a sufficient number of representative wetlands were needed to characterize each lake basin adequately. The State of the Lakes Ecosystem Conference (SOLEC) 1998 Biodiversity Investment Areas paper on Coastal Wetland Ecosystems identified the eco-reaches from which representative wetlands were selected.

Currently, about 200 wetlands are sampled annually since 2011, which is funded by the Great Lakes Restoration Initiative (GLRI) through 2020 (about $2 million per year). A total of 176 wetlands were sampled in 2011, 206 sampled in 2012, 201 in 2013, 216 in 2014, 211 in 2015, 195 in 2016, and 210 in 2017 for a total of 1415 Great Lakes coastal wetland sampling events. For aquatic macroinvertebrates specifically, 557 different sites with one to four vegetation zones have been successfully sampled since 2011; 119 sites in 2011, 115 in 2012, 126 in 2013, 128 in 2014, 129 in 2015, 121 in 2016, and 108 in 2017. As of 2017, nearly 100% of the medium and large (> 4 hectares), hydrologically-connected coastal wetlands on the Great Lakes have been sampled. With respect to the entire Great Lakes, about 80% of coastal wetlands by count and area have been sampled.

As of 2017, the average number of macroinvertebrate taxa (taxa richness) per site was 36 (Canada) and 37 (U.S.), but some wetlands had more than twice this number (Table 1). The average number of non-native invertebrate taxa in coastal wetlands was approximately 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.

There is some variability among lakes in the mean number of macroinvertebrate taxa per wetland. Lake Ontario, Lake Erie, Lake Superior wetlands averaged between 31 and 33 taxa, respectively (Table 2), while Lakes Huron and Michigan averaged approximately 38 taxa. The maximum number of invertebrate taxa was higher in Lakes Huron and Michigan wetlands (>70) than for the most invertebrate-rich wetlands in other lakes, which have a maximum of 57 to 68 taxa. Wetlands with no or the fewest taxa in both Lake Erie and Lake Ontario are sites in need of restoration. 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. Invertebrates are less mobile than fish and reflect a different 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.

The invertebrate IBI for some regions of Lake Erie and Lake Ontario are still under development because they are specific to emergent plant zones. Many of the sites in Lakes Erie and Ontario were entirely submersed aquatic vegetation, and therefore the IBIs cannot be used in those areas.

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. Agriculture and urban development are usually less intense than local physical alteration which often results in the introduction of non-native species. 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, richer, 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 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.

- Land Cover – 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), and destruction of inland wet meadow zone by plowing and diking, and addition of herbicides.

  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. 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.

- 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. 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.

- 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.

Pressures were also described in the Coastal Wetland Plants sub-indicator

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 Limitations

Between 2011 and 2017, 1415 Great Lakes coastal wetland sampling events occurred. Some of these sites have been sampled multiple times over this 7 year period. Many sites were sampled for all taxonomic groups. There are a number of sites sampled only by bird and amphibian 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 macroinvertebrate IBIs. Furthermore, macroinvertebrate IBIs have been developed for a subset of monodominant vegetation zones in coastal wetlands, including *Schoenoplectus* and wet meadow areas. Macroinvertebrate IBI scores can only be calculated for wetlands where macroinvertebrates were captured in these *Schoenoplectus* or wet meadow dominated stands.

The sites sampled in 2017 for macroinvertebrates are shown in Figure 1. Many sites were sampled for all taxonomic groups. Sites not sampled for birds and amphibians typically were sites that were impossible to access safely, and often related to private property access issues. Most bird and amphibian crews do not operate from boats since they need to arrive at sites in the dark or stay until well after dark.

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 of Great Lakes coastal wetlands based on data from least-impacted sites that are representative of Great Lakes coastal wetlands, referred to as a reference condition. These are then compared to sites experiencing a gradient of the amount and type 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.

Acknowledgments

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List of Tables

Table 1. Total macroinvertebrate taxa in Great Lakes coastal wetlands, and non-native species; summary statistics by country. Data from 2011 through 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Table 2. 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 in 2011 through 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

List of Figures

Figure 1. The percent of all sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 2. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 across the Great Lakes Basin.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 3. The percent of Lake Superior sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 4. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 in Lake Superior.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 5. The percent of Lake Michigan sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.

Figure 6. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 in Lake Michigan.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 7. The percent of Lake Huron sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Figure 8. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 in Lake Huron.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Last Updated

State of the Great Lakes 2019 Technical Report

Country	*Sites*	*Mean*	*Max*	*Min*	*St. Dev.*
Overall					
Canada	192	35.9	72	12	11.3
U.S.	366	37.0	83	10	12.5
Non-natives					
Canada	192	1.3	4	0	0.8
U.S.	366	0.8	5	0	0.9

Table 1. Total macroinvertebrate taxa in Great Lakes coastal wetlands, and non-native species; summary statistics by country. Data from 2011 through 2017.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

		Total Macroinvertebrates			*Non-native Macroinvertebrates*		
Lake	*Sites*	*Mean*	*Max*	*Min*	*Mean*	*Max*	*Min*
Erie	66	32.7	65	11	1.0	5	0
Huron	193	38.6	76	13	0.9	4	0
Michigan	85	38.1	83	15	0.8	3	0
Ontario	187	31.7	57	10	0.5	2	0
Superior	84	32.6	68	8	0.8	3	0

Table 2. 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 in 2011 through 2017.

Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

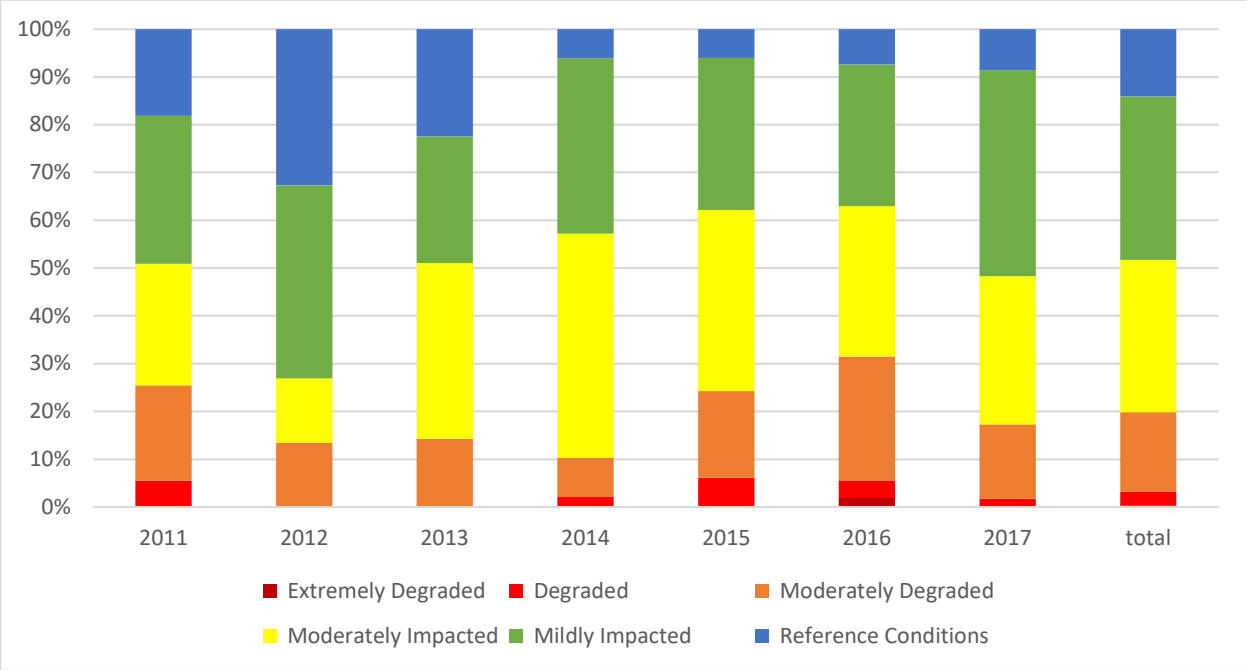


Figure 1. The percent of all sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

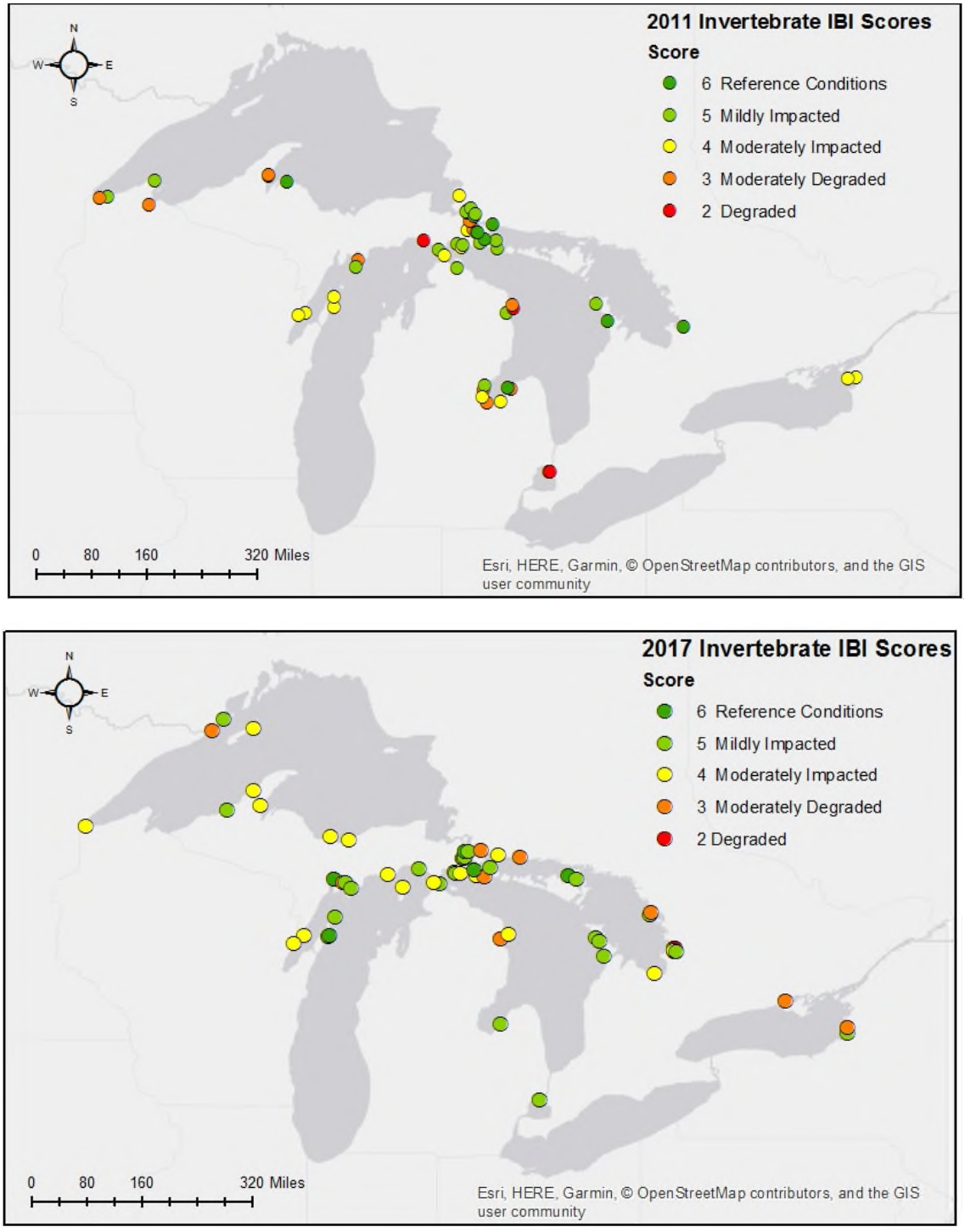


Figure 2. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 across the Great Lakes Basin.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

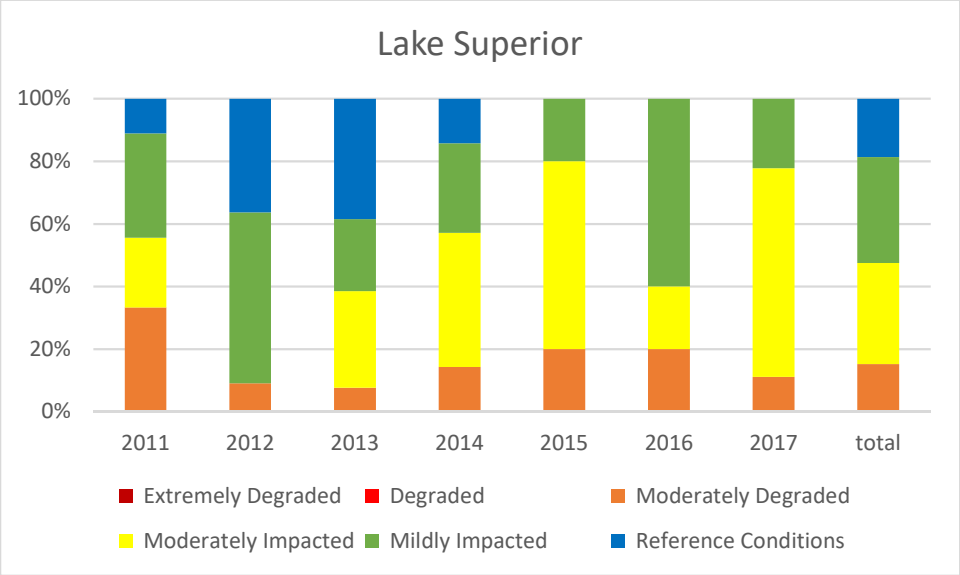


Figure 3. The percent of Lake Superior sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

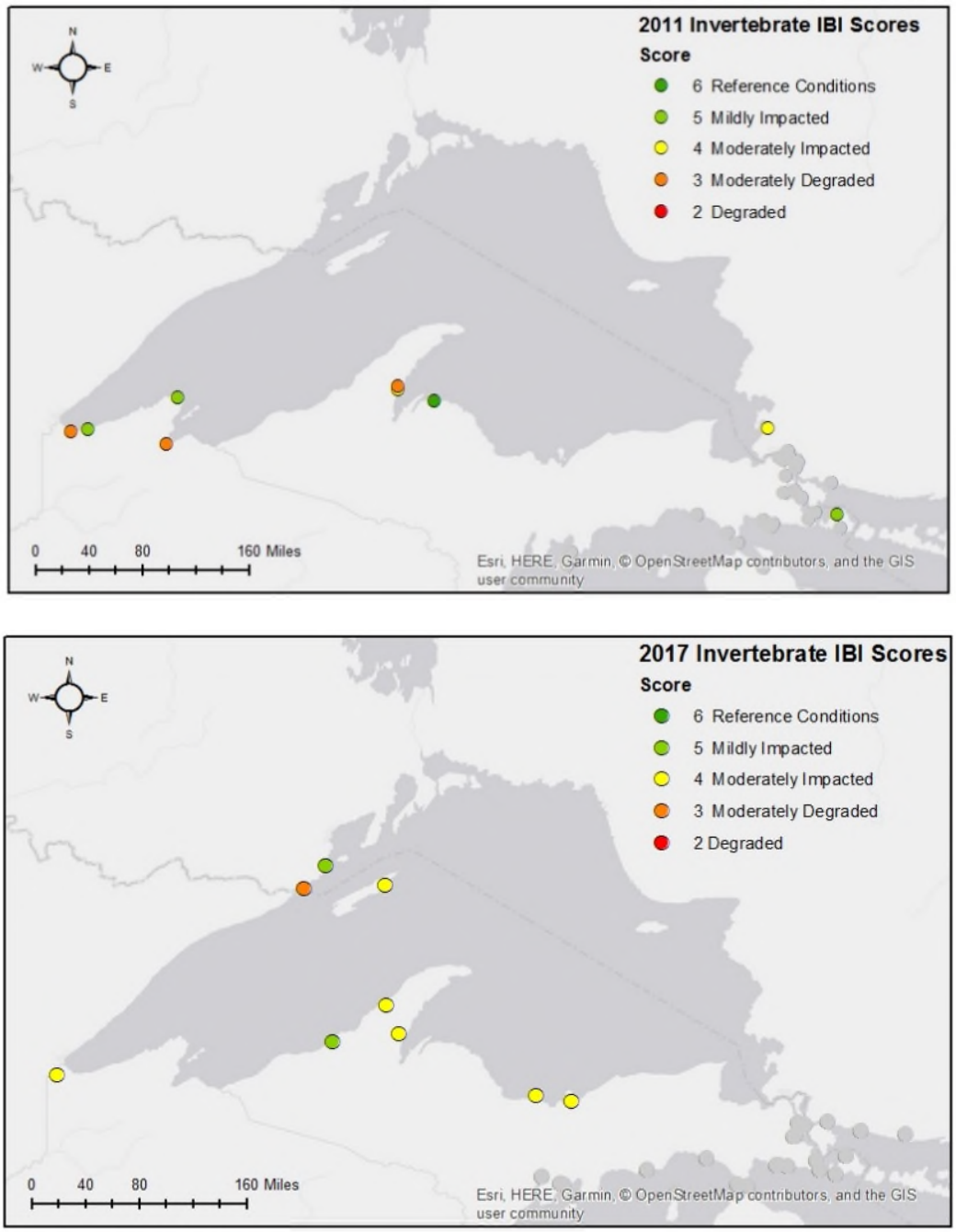


Figure 4. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 in Lake Superior.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

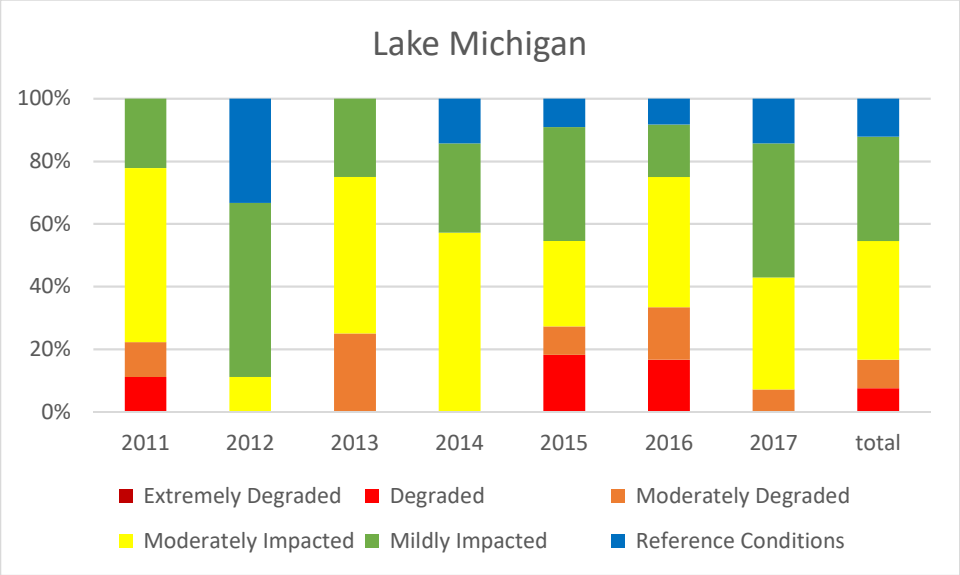


Figure 5. The percent of Lake Michigan sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

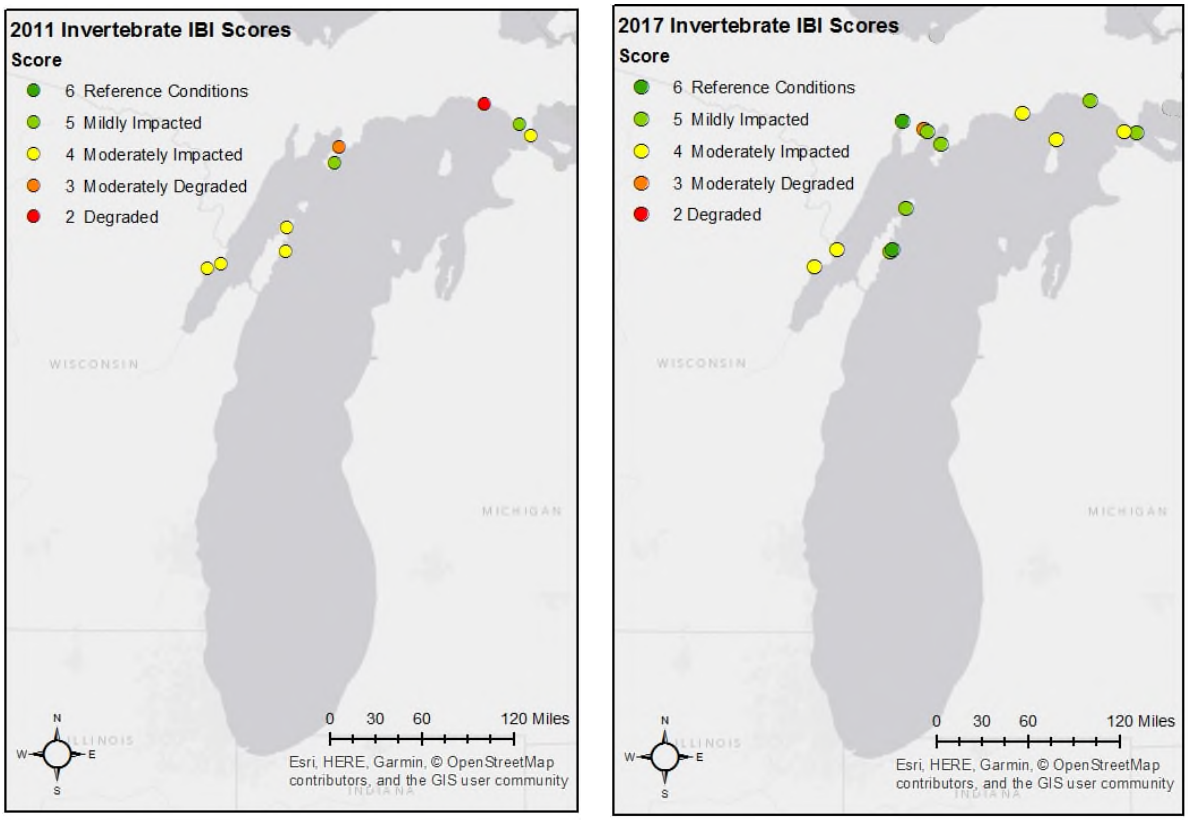


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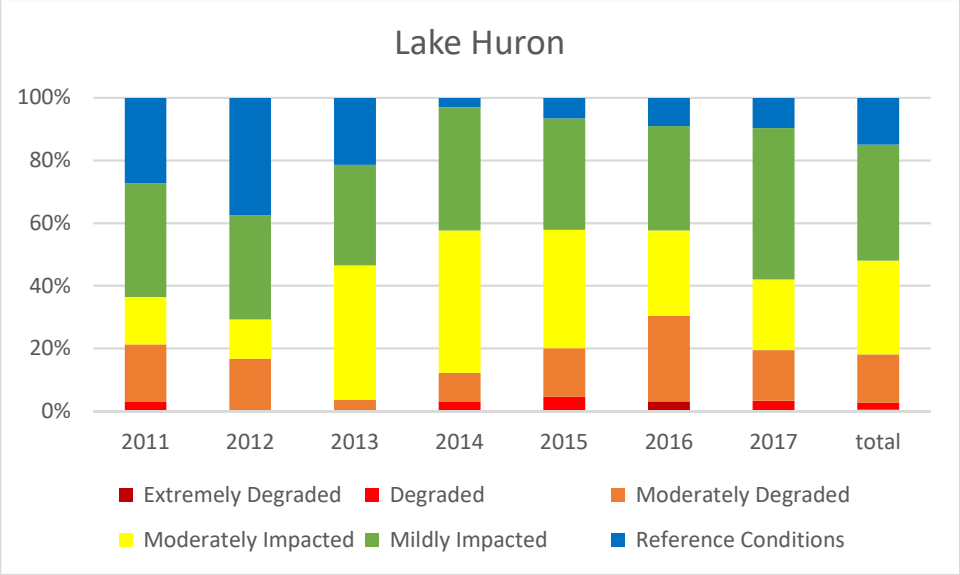


Figure 7. The percent of Lake Huron sites that fall into each IBI category based on the aquatic macroinvertbrate community each year between 2011 and 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

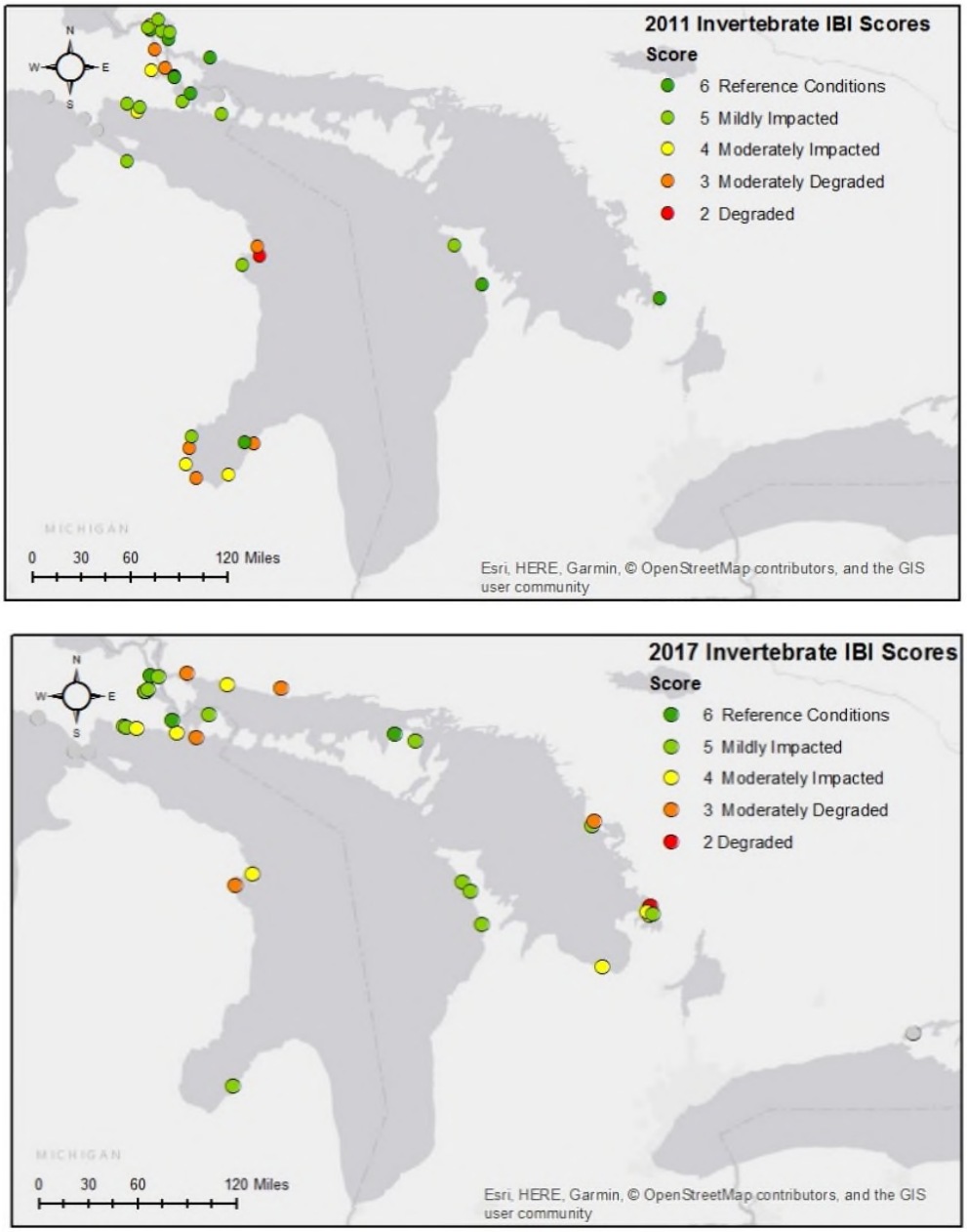


Figure 8. The condition of coastal wetland aquatic macroinvertebrate communities in 2011 compared to the condition in 2017 in Lake Huron.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Sub-Indicator: Coastal Wetland Fish

Overall Assessment
Status: Fair
Trends:
10-Year Trend (2011-2017)*: Undetermined
Long-term Trend: Undetermined
Rationale: As of 2017, the vast majority of wetlands were not degraded (based on five categories: reference conditions, mildly impacted, moderately impacted, moderately degraded, and degraded), with 11.3% of the sites falling in this category based on fish communities. The remaining sites were moderately degraded to reference quality with 20-25% of sampling events falling into each category. The trend is determined by comparing the status of coastal wetland fish over the seven year period. Between 2011 and 2017, 20-30% of wetlands were in the reference score category with no directional trend in the proportion of reference quality sites. The lack of a directional trend over the years was consistent for each condition category. Over the seven-year period 14.9-22.7 % of wetlands were mildly impacted, 18.7-27.7% were moderately impacted, 18.9-27% were moderately degraded and 7.9-18% were degraded. In the 2017 report, overall status was determined to be "Fair". Trends appeared to be improving based on a comparison over a 3 year time period. With the inclusion of all 7 years, there appears to be no substantial change in wetland status across the basin. * Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected. In previous reports, no lake-by-lake assessments were provided.

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 (2011-2017)*: Improving:
Long-term Trend: Undetermined
Rationale: As of 2017, no Lake Superior sites were in degraded condition based on fish community Indices of Biotic Integrity (IBIs) (Figures 3 and 4). This is the only Laurentian Great Lake without degraded sites. In total, 22.9% of sites sampled between 2011 and 2017 were of reference quality, 34% were mildly impacted, 20% moderately impacted, 22.9% moderately degraded. The number of moderately impacted sites decreased in the last three years. In 2014 and 2015, 42 and 40% of sites were moderately impacted but dropped significantly to 0% and 9% in 2016 and 2017, respectively. Over the same time period there was an increase in the proportion of reference quality sites. In 2014, no reference quality sites were sampled, in the following three years 20-46% of sites sampled were of reference quality. * Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Michigan
Status: Fair
10-Year Trend (2011-2017)*: Undetermined
Long-term Trend: Undetermined
Rationale: The vast majority of Lake Michigan sites did not fall in the degraded category. Over the 7-year period, only 11.7% of sites sampled were of degraded condition. In 2011, no sites were of degraded quality (Figures 5 and 6). For the following years, the percentage ranged from 11.7 to 23.5. Reference quality sites made up 5.8% to 23% of the sites sampled across the 7 years. There were no trends of increasing or decreasing quality across the whole data set. Moderately impacted conditions dominated in 2013 (53%) but otherwise made up 23-37% of sites in other years. The proportion of mildly impacted sites peaked in 2015/2016 at 0.35 and 0.037. In all other years 12.5-19% of sites were mildly impacted. No significant changes in condition occurred among years with similar sites, except for a slight increase in the proportion of degraded sites between 2011 and 2016. *Note: A 7-year trend will be used

for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend (2011-2017)*: Unchanging
Long-term Trend: Undetermined
Rationale: Based on fish communities, 14% of Lake Huron wetlands were of degraded quality between 2011 and 2017. In 2015 and 2016, few sites were of degraded quality (3% and 4% respectively) relative to other years (14-23%). There was a decrease in degraded sites and an increase in the number of reference condition sites between 2011 and 2016 (years with many overlapping sites). However, there were fewer reference quality sites in 2017 compared to 2012 (Figures 7 and 8). The number of moderately impacted and degraded sites decreased but the proportion of moderately impacted sites increased. There was year-to-year variability in wetland condition when comparing all years. However, there were no apparent increases or decreases in wetland quality over the entire 7-year period. *Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend (2011-2017)*: Undetermined
Long-term Trend: Undetermined
Rationale: The majority of wetlands did not fall into the degraded category. However, Lake Erie had the highest percentage of degraded sites (26.8%) compared to the rest of the basin (0-14%). The majority of sites fell into the moderately degraded category (40.2%). None of the sites sampled in 2011, 2015 or 2016 were of reference quality. There was no overall trend in condition across all 7 years and proportion of sites falling into each category varied across years. When comparing 2011 to 2016, years where many of the same sites were sampled, there was an increase in the number of degraded sites (Figures 9 and 10). However, there was a decrease in the proportion of degraded sites and an increase in the proportion of reference quality sites when comparing 2012 to 2017. *Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend (2011-2017)*: Improving
Long-term Trend: Undetermined
Rationale: Over the 7 year sampling period, the percent of degraded wetlands ranged from 0-12, and in total 4.7% of Lake Ontario sites were of degraded quality based on fish communities. In 2012, only 12% of sites were of reference quality, but this percentage increased over the following years, ranging from 22.7-56%. When comparing years with similar sites, there was an increase in the number of reference quality sites between 2011 and 2016 (Figures 11 and 12). Between 2012 and 2017 the percent of reference quality sites increased while the percent of degraded quality sites decreased. The percentage of mildly impacted sites was relatively consistent across years (15-25%). The percentage of moderately impacted sites decreased from 31% and 32% in 2011 and 2012 to 7.7%, 12% and 12.5% in 2015, 2016, and 2017 respectively. *Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Status Assessment Definitions
Good: No wetlands are in the degraded category but instead fall within reference conditions, mildly impacted, or moderately impacted.
Fair: The vast majority of the wetlands are not in the degraded category.
Poor: The vast majority of the wetlands are in the degraded category.
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
An endpoint for this sub-indicator was established based on fish communities of reference systems. Data were evaluated for patterns by lake, ecoregion, wetland type, and vegetation zone. An endpoint for this sub-indicator is not possible at this time.

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 and nursery areas for many fish species of ecological and economic 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." 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
Fish should be sampled using three replicate fyke nets in each major plant zone (wet meadow, emergent vegetation, and submergent vegetation) in each wetland for one net-night (Uzarski et al., 2005; Uzarski et al. 2016). Sampling locations should correspond with those for macroinvertebrate and water quality sampling. The timing of sampling should correspond with the maturity of the vegetation in each system. Fish abundance by taxon is used to calculate the Great Lakes Coastal Wetlands Monitoring Program (GLCWMP) IBI scores. The GLCWC developed indices of biological integrity (IBIs) in 2002 and protocols were finalized in 2008 (GLCWC, 2008). These were further developed by the GLCWMP (Cooper et al. 2018). The 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.

Individual IBIs can be used independently as a measure of coastal wetland health, based on a percentage of points possible reflected as 'reference conditions' to 'extremely degraded'. The sub-indicator has been used basin wide (U.S. and Canada wide) over the past eight years and much longer in some regions. This sub-indicator can also be 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. For example, fish tend to detect disturbance somewhere between the local and regional scale. See https://www.greatlakeswetlands.org/Home.vbhtml, 'Documents' for details on indicator metrics.

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 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).

In order to properly manage the Great Lakes coastal wetland fish community health there must be consistent sampling methods. Sampling was conducted no earlier than mid-June and no later than August due to migration

patterns of the fish communities. Fish should be sampled using three replicate fyke nets of 4.8 mm mesh in each major plant zone in each wetland for one net-night. 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: 0.5-m x 1-m opening and 1-m x 1-m opening. The smaller nets are placed in water that is 0.25-0.5 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, therefore, fish swimming along the edge of the vegetation zone are captured.

Any fish collected that are greater than 25 mm are identified down to species and 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 GLCWC developed indices of biological integrity (IBIs) in 2002 and protocols were finalized in 2008 (GLCWC 2008). These were further developed by the GLCWMP. The 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. Several different fish metrics are being utilized. See https://www.greatlakeswetlands.org/Home.vbhtml 'Documents' for details on indicator metrics.

The IBI provides a rigorous approach to quantify the biological condition of fish communities within the Great Lakes. It is based on reference conditions and is developed from a composite of specific measures used to describe fish community, structure, function, individual health, and abundance. Specific parameters, termed "metrics," are scored based on how similar they are to the reference condition. Individual IBIs are derived for each of the measures and can be used independently as a measure of coastal wetland health, based on a percentage of points possible reflected as 'reference conditions' to 'degraded'. The IBI also provides a narrative characterization that provides a measure of the environmental condition and will be calibrated for regional use.

Currently, about 200 wetlands are sampled annually since 2011, which is funded by the Great Lakes Restoration Initiative (GLRI) through 2020 (about $2 million per year). A total of 176 wetlands were sampled in 2011, 206 sampled in 2012, 201 in 2013, 216 in 2014, 211 in 2015, 195 in 2016, and 210 in 2017 for a total of 1415 Great Lakes coastal wetland sampling events. Not all sites were sampling for fish and the number of sites sampled per lake varied as well. To date, 82 fish sampling events have taken place in Lake Erie, 207 in Lake Huron, 102 in Lake Michigan, 169 in Lake Ontario, and 70 in Lake Superior as of 2017. As of 2017, nearly 100% of the medium and large (> 4 hectares), hydrologically-connected coastal wetlands on the Great Lakes have been sampled. With respect to the entire Great Lakes, about 80% of coastal wetlands by count and area have been sampled (Figure 1).

From 2011 to 2017, an average of 9 and 12 fish species were collected in Canadian and U.S. Great Lakes coastal wetlands, respectively (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 27 (US) fish species. The average number of non-native fish species per wetland was approximately one, though some wetlands had as many as 5 (U.S.). There are 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).

From 2011-2017, total fish species did not differ greatly by lake, averaging 9 to 12 species per wetland (Table 2). Lake Michigan wetlands had the lowest maximum number of species with 19 and Lake Huron had the highest maximum number of species (27). Lake Huron wetlands averaged the lowest mean number of non-native fish taxa (0.6) and Lake Erie had the highest (2.4). All other lakes had a similar average number of non-native fish species per wetland, which was about one (1) species.

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. Data from several studies indicate that the characteristic groups of fish species in reference wetlands from each ecological province tend to have similar water temperature and aquatic productivity preferences.

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. Most of 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).

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 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. Agriculture and urban development are usually less intense than local physical alteration which often results in the introduction of non-native species. 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, richer, 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 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.

- Land Cover – Agriculture degrades wetlands in several ways, including nutrient enrichment from fertilizers, increased sediments 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 submergent 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. 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. Common and grass carp reproductive and feeding behaviour results in loss of submergent 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 effect riverine systems most through increased sediment movements, particularly in flashy 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.

Pressures were also described in the Coastal Wetland Plants sub-indicator.

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 Limitations

This sub-indicator can only be used where there is sufficient water depth to use fyke nets and a minimum of 10 fish must be captured or the sites must be fished another net-night. Between 2011 and 2017, 1415 Great Lakes coastal wetland sampling events occurred. Some of these sites have been sampled multiple times over this 7 year period. Many sites were sampled for all taxonomic groups. There are a number of sites sampled only by bird and amphibian 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 dominated stands.

Acknowledgments

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List of Tables

Table 1. Total fish species in wetlands, and non-native species; summary statistics by country for sites sampled from 2011 through 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP), Uzarski et al. 2016
Table 2. Fish total species and non-native species found in Great Lakes coastal wetlands by lake. Mean, maximum, and minimum number of species per wetland. Data from 2011 through 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

List of Figures

Figure 1. The percentage of sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 2. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 across the Great Lakes basin.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 3. The percentage of all Lake Superior sites fallling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 4. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Superior.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 5. The percentage of Lake Michigan sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 6. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Michigan.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 7. The percentage of Lake Huron sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 8. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Huron.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 9. The percentage of Lake Erie sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 10. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Erie.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 11. The percentage of Lake Ontario sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)
Figure 12. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Ontario.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Last Updated

State of the Great Lakes 2019 Technical Report

Country	*Sites*	*Mean*	*Max*	*Min*	*St. Dev.*
Overall					
Canada	221	9.1	20	2	3.7
U.S.	504	12.0	27	1	5.0
Non-natives					
Canada		0.8	4	0	0.9
U.S.		1	5	0	1

Table 1. Total fish species in wetlands, and non-native species; summary statistics by country for sites sampled from 2011 through 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP), Uzarski et al. 2016

		Total Fish			*Non-native Fish*		
Lake	*Sites*	*Mean*	*Max*	*Min*	*Mean*	*Max*	*Min*
Erie	95	10.5	25	1	2.4	5	1
Huron	248	12.1	27	3	0.6	3	0
Michigan	115	9.1	19	2	1.1	5	0
Ontario	178	11.3	21	4	0.9	4	0
Superior	93	11.4	25	3	0.7	5	0

Table 2. Fish total species and non-native species found in Great Lakes coastal wetlands by lake. Mean, maximum, and minimum number of species per wetland. Data from 2011 through 2017.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP), Uzarski et al. 2016

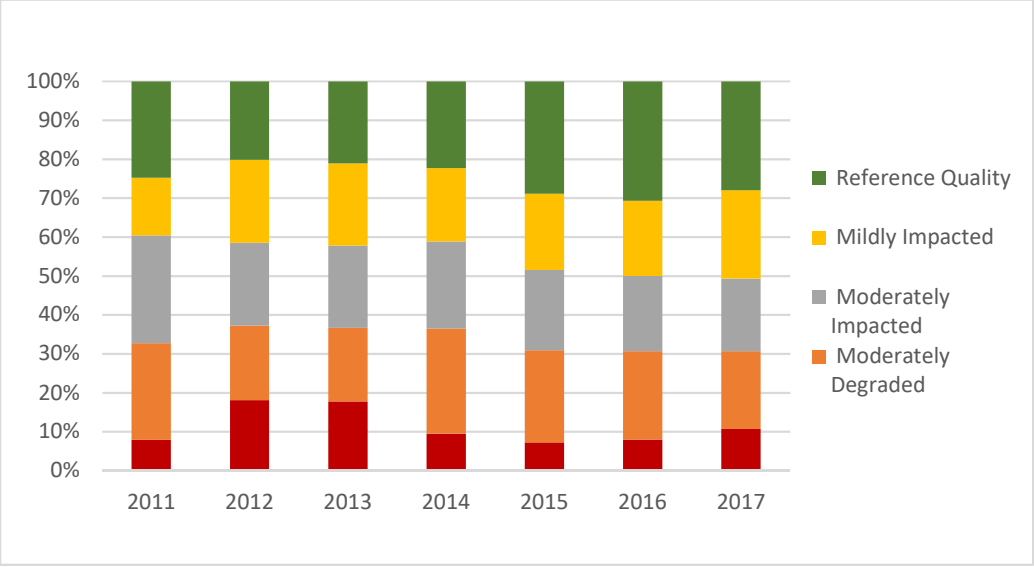


Figure 1. The percentage of sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

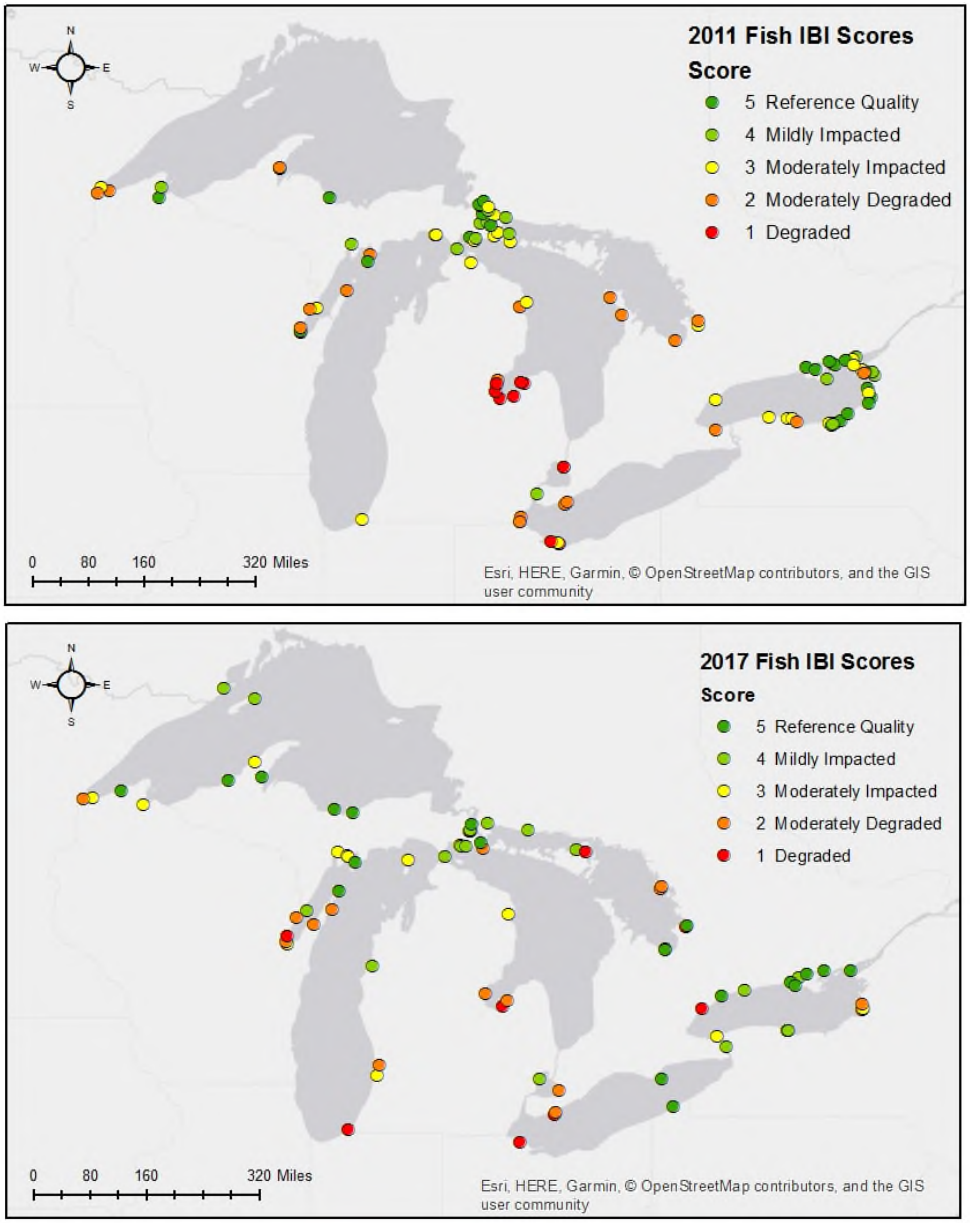


Figure 2. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 across the Great Lakes basin.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

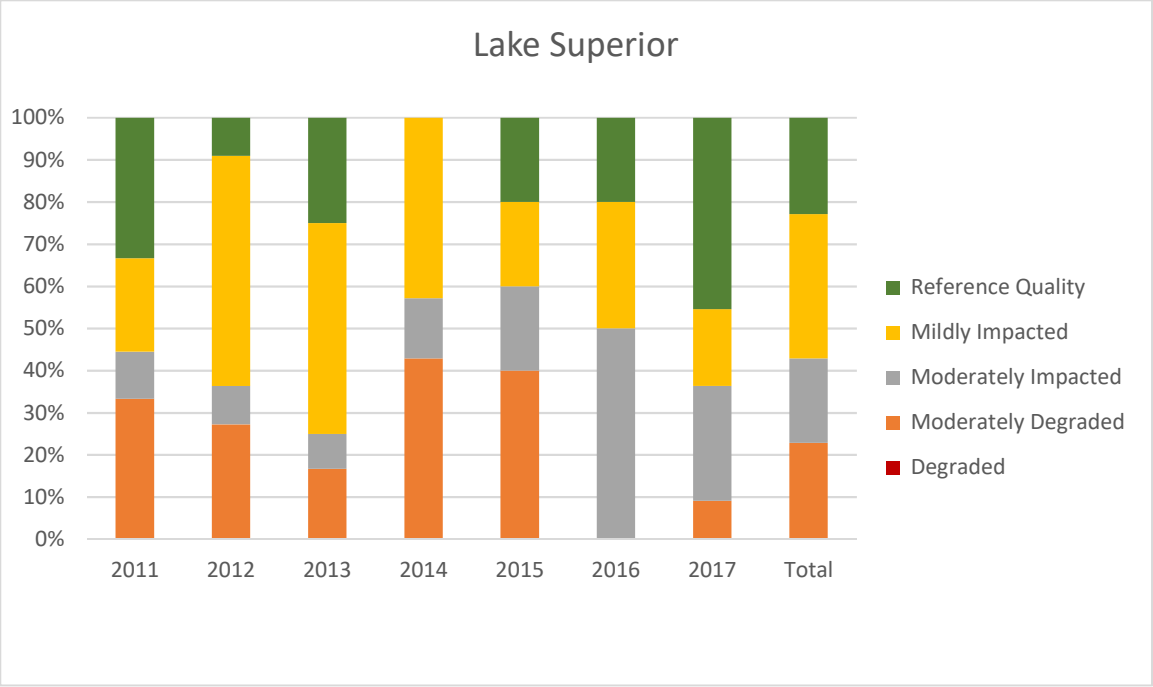


Figure 3. The percentage of Lake Superior sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

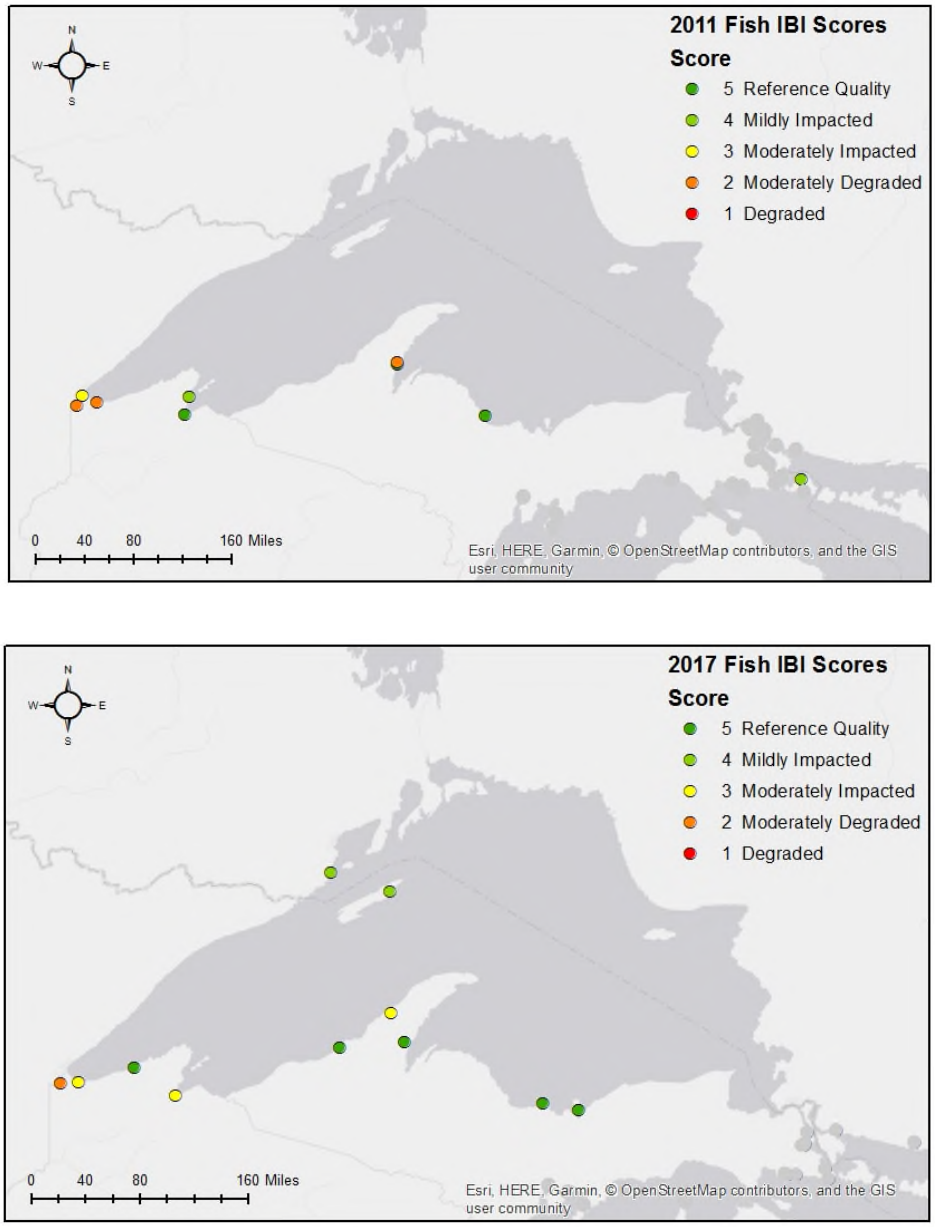


Figure 4. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Superior.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

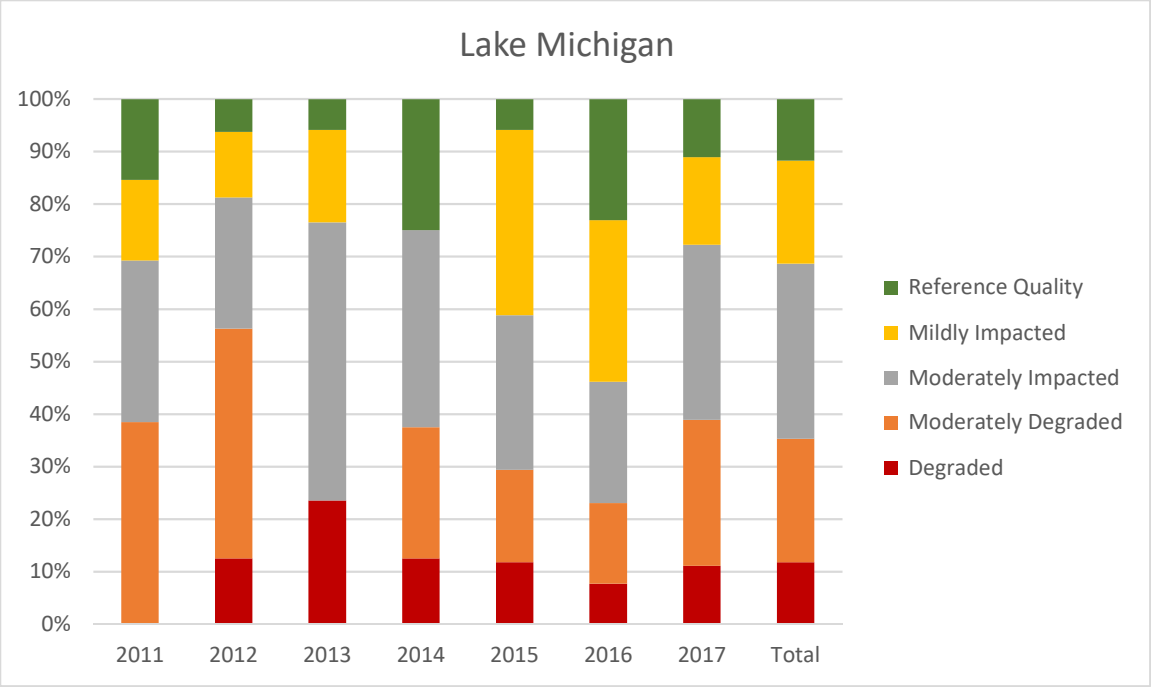


Figure 5. The percentage of Lake Michigan sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

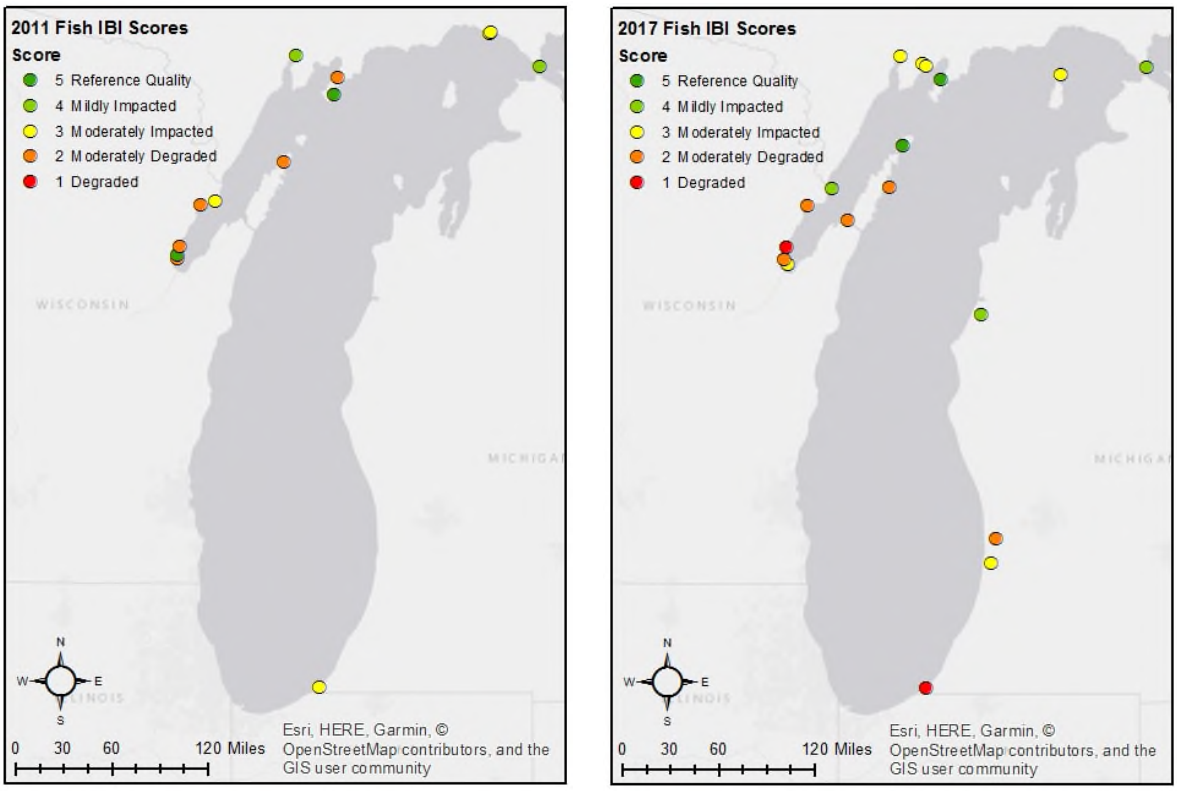


Figure 6. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Michigan.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

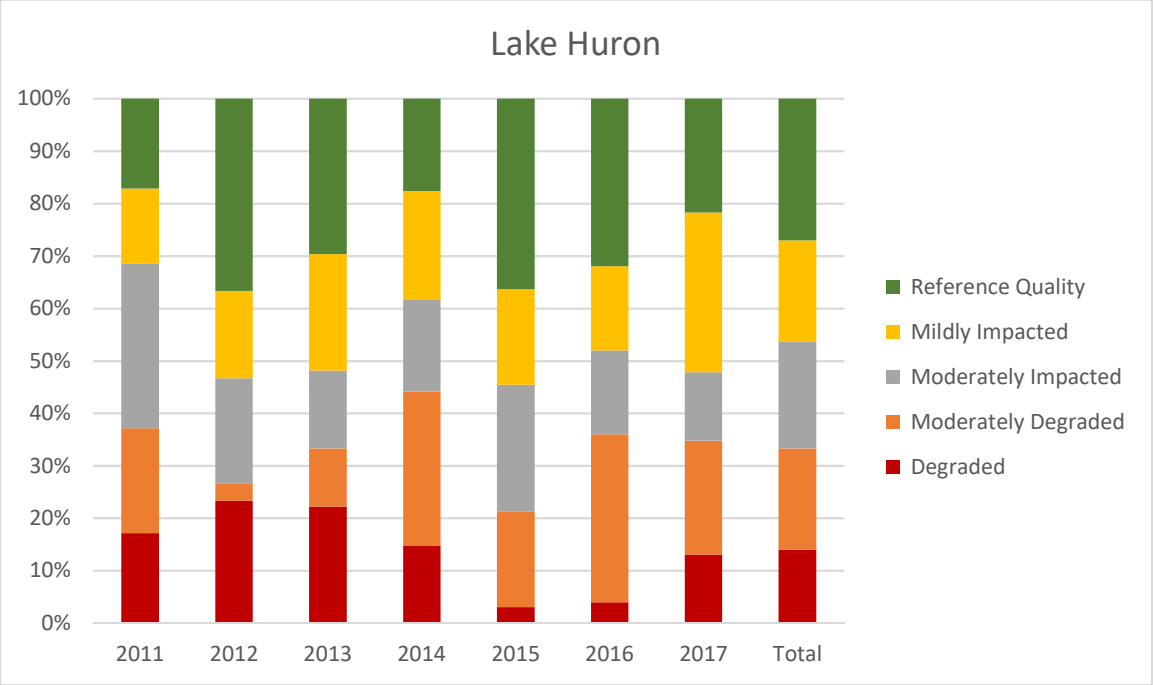


Figure 7. The percentage of Lake Huron sites falling into each IBI category based on the fish community each year. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

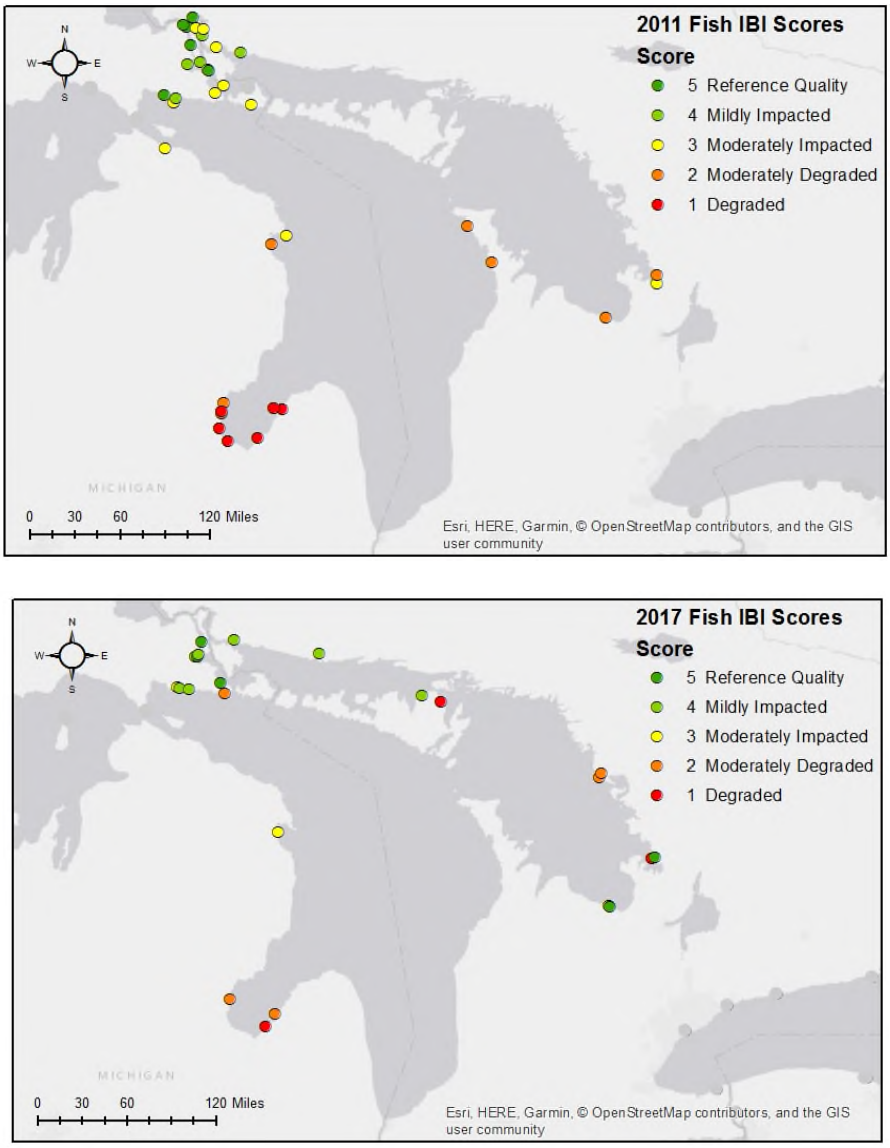


Figure 8. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Huron.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

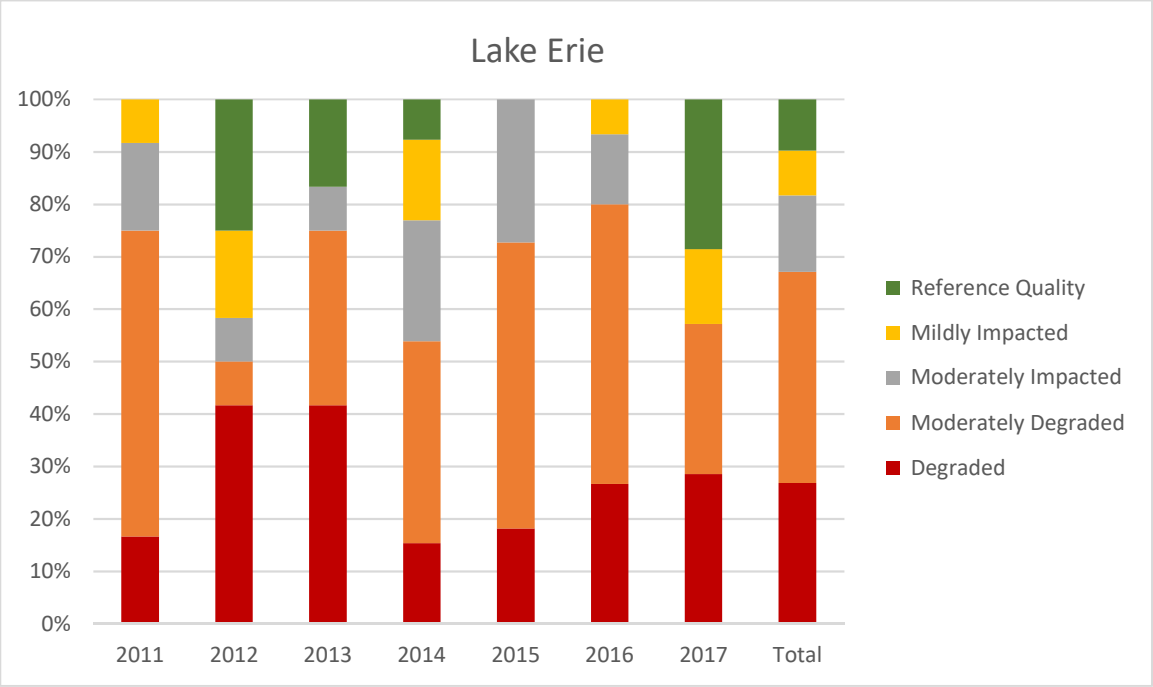


Figure 9. The percentage of Lake Erie sites falling into each IBI category based on the fish community each year. Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

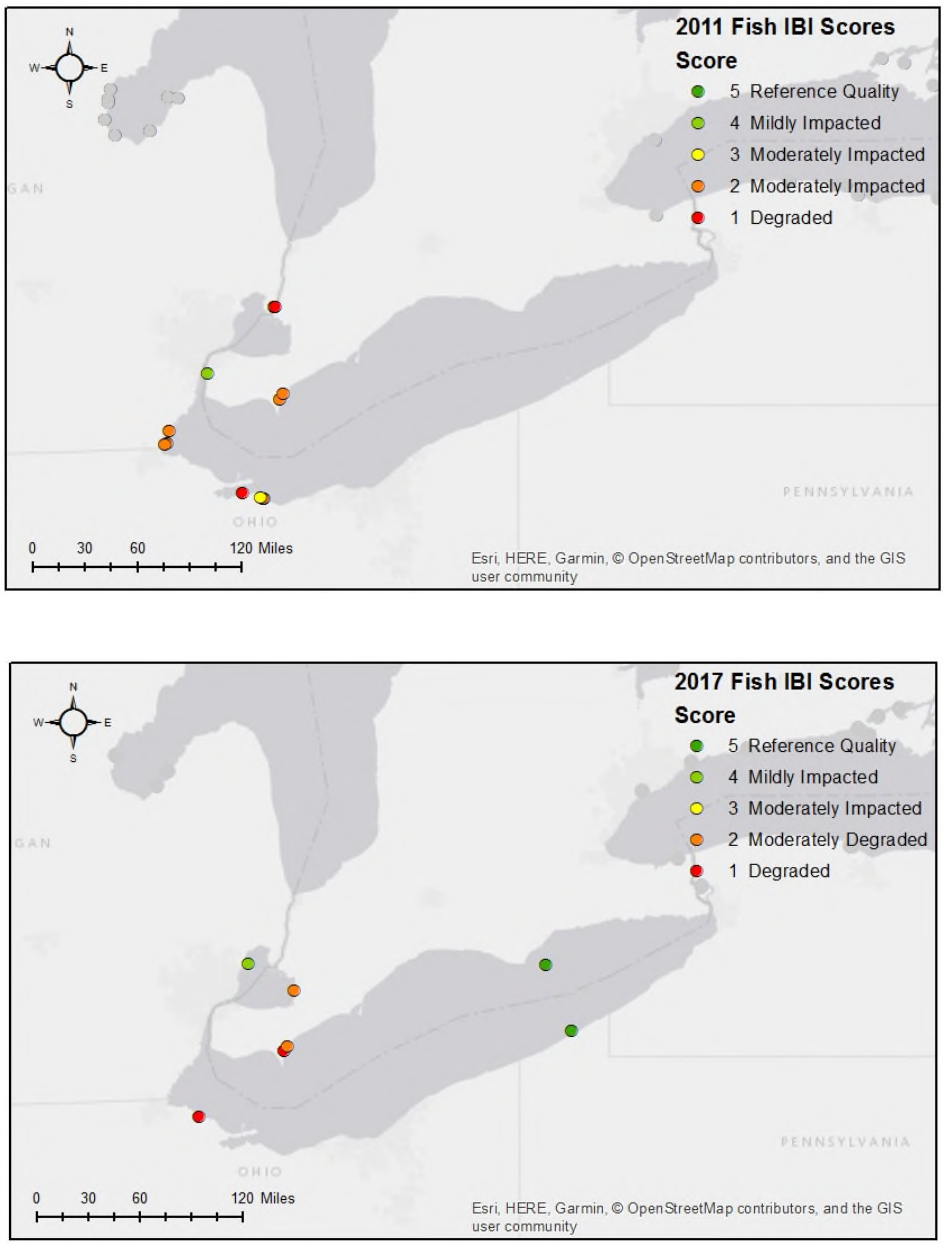


Figure 10. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Erie.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

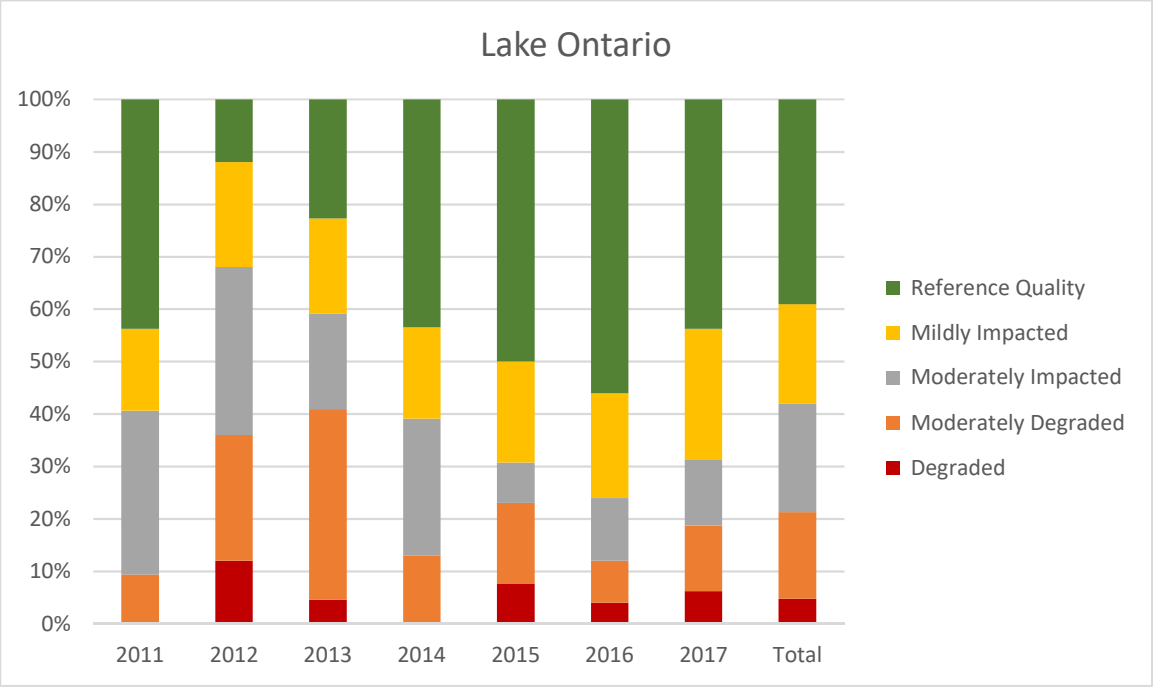


Figure 11. The percentage of Lake Ontario sites falling into each IBI category based on the fish community each year.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

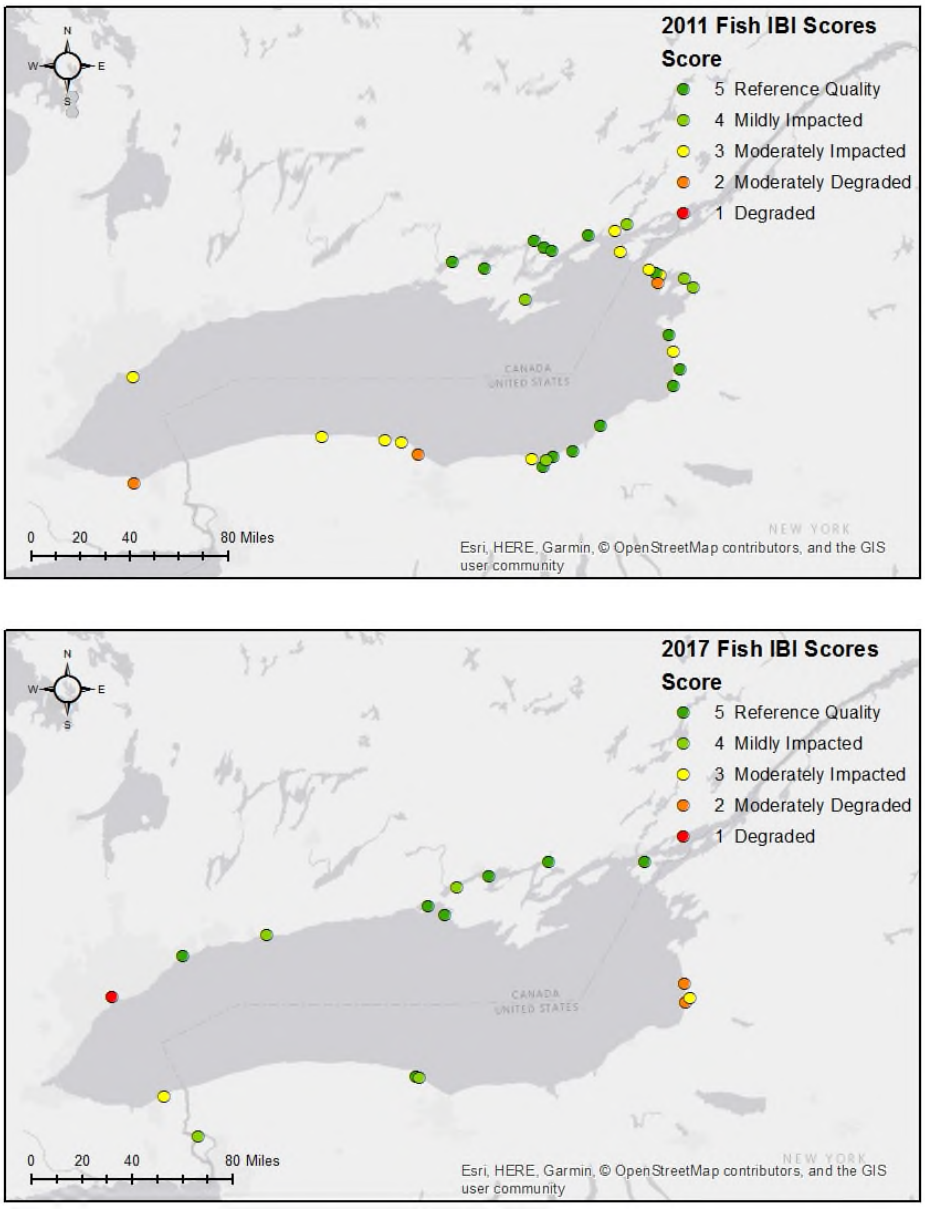


Figure 12. The condition of coastal wetland fish communities in 2011 compared to the condition in 2017 in Lake Ontario.
Source: Great Lakes Coastal Wetland Monitoring Program (CWMP)

Sub-Indicator: Coastal Wetland Amphibians

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Improving
Long-term Trend (1995-2017): Improving
Rationale: Median 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 was 6.5 (out of 10) based on data from 2017, reflecting a significant increase of 2.9%/year over the past 10 years. 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.

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: Undetermined
Long-term Trend (1995-2017): Undetermined
Rationale: Median IEC was 8.5 based on data from 2017.

Lake Michigan
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1995-2017): Undetermined
Rationale: Median IEC was 8.3 based on data from 2017 and did not significantly increase or decrease over the most recent 10 years.

Lake Huron (including St. Marys River)
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1995-2017): Undetermined
Rationale: Median IEC was 8.2 based on data from 2017 and did not significantly increase or decrease over the most recent 10 years.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Unchanging
Long-term Trend (1995-2017): Unchanging
Rationale: Median IEC was 4.4 based on data from 2017 and did not significantly increase or decrease over the most recent 10 year.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1995-2017): Improving
Rationale: Median IEC was 5.9 based on data from 2017, reflecting a significant increase of 3.8%/ over the past 10 years.

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 2019 reporting, status is based on data from 2017) 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 CWMP. Note that the values defining Poor, Fair, and Good below have changed since the previous report (Tozer et al. 2017) 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; IEC > 66th percentile (for 2019 reporting, IEC > 7.7)
Fair: Some ecosystem components are in acceptable condition; 33rd percentile ≤ IEC ≤ 66th percentile (for 2019 reporting, 5.0 ≤ IEC ≤ 7.7)
Poor: Very few or no ecosystem components are in acceptable condition; IEC < 33rd percentile (for 2019 reporting, IEC < 5.0)
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 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). The statistical significance of trends was assessed via parametric bootstrapping in R (R Core Team 2015) 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. 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 2019 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 trends of Great Lakes coastal wetland ecosystem health by directly measuring the composition and occurrence of wetland breeding anurans, 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 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. 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 Bird Studies Canada's Great Lakes Marsh Monitoring Program (GLMMP), 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). 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 Great Lakes Coastal Wetland Monitoring Program (CWMP) led by Central Michigan University was initiated in 2011 and currently is scheduled to operate until at least 2020 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 eight 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 (previously known as SOLEC) 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 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.

In this report, we assessed anuran community health based on multi-species data across the Great Lakes basin (Howe et al. 2007a, 2007b; Hanowski et al. 2007a, 2007b; Tozer 2013). Quantitative data were used for breeding anurans at thousands of sample points throughout the Great Lakes in both the U.S. and Canada. At 806 of these sample points, quantitative information was available on three potential environmental stressors (Panci et al. 2017): 1) agricultural intensity in the contributing watershed (i.e., the landscape draining into the wetland), 2) non-agricultural landscape development such as roads, buildings, and human population density in the contributing

watershed, and 3) wetland area and fragmentation, measured by the total wetland area within 1 km of the sampled wetland's centroid. For convenience, these gradients are referred to in this report as agriculture, development, and wetland area, respectively. Clearly, many other stressors affect anuran communities in coastal wetlands, but agricultural intensity, non-agricultural landscape development, and wetland area provide tractable quantitative yardsticks from which one can identify sensitive species and community variables (Brazner et al. 2007a, 2007b). Species that are sensitive to these easily-measured landscape variables are likely vulnerable to other, unmeasured stressors. Site-specific values for each of the three measured stressors (agricultural intensity, development, and wetland area/habitat destruction) were combined by principal components analysis into a single gradient that approximates the "human footprint" associated with a single wetland.

To avoid excessive zeros (absences) in the data, each of the 806 sites with accompanying environmental data and composite "human footprint" score were placed bins of 10 points each (or 6 in the last case) with similar measures of environmental stress. In other words, the 10 sites with the worst "human footprint" scores were placed in the first bin, then the 10 sites with the next best "human footprint" scores were placed in the second bin, and so on, until the last of the 81 bins was reached, which contained the last 6 sites with the very best "human footprint" scores. This created a new gradient of 81 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 multiple anuran species or anuran species groups, a stressor response curve (biotic response [BR] function) was estimated from the probabilities of occurrence in each of the 81 bins (Figure 1). The probability values were simply the proportions of sample points in each bin where each anuran species or anuran species group was present. 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 similar to the normal distribution (Gnass Giese et al. 2015). These BR functions (Figure 1) provided the basis for scoring new sites based on the presence/absence of eight species/species groups (Table 1). Parameters of the best-fit mathematical function were estimated by computer iteration in R (R Core Team 2015) with package "iec" (https://github.com/ngwalton/iec). The three parameters (mean, standard deviation, and height) describe a bell-shaped or truncated Gaussian function within the range of 0-10.

The health of coastal wetlands was subsequently evaluated using the index of ecological condition (IEC), an objective biotic indicator introduced by Howe et al. (2007a, 2007b), improved by Gnass Giese et al. (2015), and compared to other similar indices (using bird data) by Chin et al. (2015). By recording the species present at a wetland, one can essentially work backward by iteration to 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 that were actually present at the site plus the sum of one minus the probabilities (i.e., the probabilities of not finding) species that were absent at the sample point. Presence of species that have been shown previously to favour minimally-stressed wetlands will indicate ecologically healthy conditions and high IEC scores. By contrast, presence of species that favour 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. The final suite of BR functions for calculating IECs included eight species (Table 1). Rare species and species rarely found in wetland habitats were excluded, resulting in an indicator metric that directly represents the anuran assemblage associated with a coastal wetland.

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 an index of 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 adjusted for 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, but IEC metrics were also calculated for inland wetlands, and were used only for comparison, and were not used for any of the assessments of status or trend. In addition, box-and-whisker plots of wetland-level mean IECs for coastal wetlands in each basin and overall for the most recent year (2017) are reported to illustrate current status in relation to defining values for Poor, Fair, and Good status. Density plots of wetland-level mean IECs for coastal and inland wetlands in each basin based on the most recent 5 year of data (2013-2017) are reported to illustrate variation in distributions. To further illustrate distributions, these same data for coastal wetlands are also 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, and

the most recent 5 years of data (2013-2017) were used to boost sample sizes. To better illustrate general patterns on the map, the point-to-raster tool was used in ArcGIS to calculate mean values within 3000 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".

Ecological Condition

Data coverage—The dataset available for scoring sites consisted of mean annual wetland-level IECs based on 52,860 point counts conducted at 4,758 sample points in 1,749 wetlands over 23 years from 1995-2017 throughout the Great Lakes basin (Figure 2). The number of years that each wetland was surveyed varied from 1 to 23, with a mean of 3.5 ± 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 = 1,114; 64%) rather than inland (n = 635; 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 wetlands surveyed per year (268 ± 117 [mean ± SD]) ranged from 114 to 442, with substantially more wetlands surveyed from 2011-2014 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 ranged from 3.6 to 6.6 from 1995-2017, significantly increased by 2.9%/year (1.5, 4.4) [lower, upper 95% confidence limits] over the past 10 years (Table 2, Figure 4), and was 6.5 (out of 10) based on data from 2017 (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, although the Improving trend may not be entirely real as discussed in more detail in the next paragraph below and in the discussion. Generally, IECs were lower in the southern portion of the Great Lakes basin south of the Canadian Shield where agricultural and human development footprints are bigger/stronger (Figure 6-7). By contrast, median IEC at inland wetlands did not significantly increase or decrease overall over the past 10 years and was higher than in coastal wetlands in most years (Table 2; Figure 4, 7).

The status has changed from Poor to Fair, and the trend has changed from Unchanging to Improving. The differences in status and trend may be due to one or more of several factors: 1) improved sensitivity of the metrics used to calculate the IEC since the last report (Tozer et al. 2017); 2) substantial improvements by the Great Lakes Coastal Wetland Monitoring Program (CWMP) since 2011 in geographical coverage of coastal wetlands in Lake Superior and northern portions of Lake Michigan and Lake Huron, where anuran communities tend to experience fewer stressors; and 3) changes in lake levels, which may lead to dramatic changes in the area and/or quality of coastal wetland breeding anuran habitat and associated IECs. Calculation of trends will be more reliable in future reports with the continuation of the CWMP because coverage of coastal wetlands will be more consistent over the most recent 10 years. In addition, as the number of years of sampling increases, the power to detect change also increases.

Lake Superior—Median IEC in coastal wetlands ranged from 2.0 to 10.0 from 1995-2017 (Figure 4) and was 8.5 based on data from 2017 (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 Good, and the trend is Undetermined. Somewhat contrasting 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 (Figure 4, 7). 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, accounting at least partially for some of the modest scores in comparison with those from other lakes (Figure 4-7).

The status has changed from Fair to Good, and the trend has changed from Unchanging to Undetermined. 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 breeding anuran habitat and associated IECs. For this report, the current information is inadequate to perform reliable statistical significance testing for trends. Calculation of trends will be more reliable in future reports with the

continuation of the CWMP because coverage of coastal wetlands will be more consistent over the most recent 10 years. In addition, as the number of years of sampling increases, the power to detect change also increases.

Lake Michigan—Median IEC in coastal wetlands ranged from 0.3 to 8.4 from 1995-2017 (Figure 4) and was 8.3 based on data from 2017 (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 Good, and the trend is Unchanging. 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 status has changed from Poor to Good, and the trend is the same as in the last report. 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 breeding anuran habitat and associated IECs.

Lake Huron (including St. Marys River)—Median IEC in coastal wetlands ranged from 5.2 to 8.7 from 1995-2017, did not significantly increase or decrease over the past 10 years (Table 2, Figure 4), and was 8.2 based on data from 2017 (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. Similar patterns occurred at inland wetlands in the Lake Huron watershed (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 has changed from Fair to Good, and the trend is the same as in the last report. 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 breeding anuran habitat and associated IECs.

Lake Erie (including St. Clair-Detroit River Ecosystem)—Median IEC in coastal wetlands ranged from 3.5 to 5.4 from 1995-2017, did not significantly increase or decrease over the past 10 years (Table 2, Figure 4), and was 4.4 based on data from 2017 (Figure 5). Based on these patterns, the status of coastal wetland health based on anuran communities in Lake Erie is Poor, and the trend is Unchanging. Similar patterns occurred at inland wetlands in the Lake Erie watershed (Table 2; Figure 4, 7). Out of all of the lake basins, Lake Erie had the lowest median IEC in coastal wetlands, being only slightly worse than Lake Ontario (Figure 5-7).

There has been no change in status or trend since the last report.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)—Median IEC in coastal wetlands ranged from 2.1 to 5.9 from 1995-2017, significantly increased by 3.8%/year (1.0, 6.7) [lower, upper 95% confidence limits] over the past 10 years (Table 2, Figure 4), and was 5.9 based on data from 2017 (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. Similar patterns occurred at inland wetlands in the Lake Ontario watershed (Table 2; Figure 4, 7). Out of all of the lake basins, Lake Ontario had the second-lowest median IEC in coastal wetlands, being only slightly better than Lake Erie (Figure 5-7).
The status has changed from Poor to Fair, and the trend has changed from Unchanging to Improving**.** The difference in status and trend 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 breeding anuran habitat and associated IECs.

Discussion—Throughout the Great Lakes basin, the current status of coastal wetland health based on wetland breeding anurans is Fair, with current status of Lake Superior, Lake Michigan, and Lake Huron being Good, Lake Ontario being Fair, and Lake Erie being Poor (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 southern portion of Lake Michigan and all of Lake Erie and Lake Ontario compared to Lake Superior and Lake Huron (Figure 6). These patterns 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 Lake Superior and most parts of Lake Huron (Allan et al. 2013, Bourgeau-Chavez et al. 2015, Danz et al. 2007, Niemi et al. 2009). Given that Lake Erie had the lowest median coastal IEC of all of the lake basins, the health of Lake Erie's coastal wetlands appear to be especially compromised compared to coastal wetlands in the remaining lake basins. The low median coastal IEC may also indicate that Lake Erie is experiencing unique stressors or relatively high intensities of stressors compared with stressors in the other lake basins. Nonetheless, some high quality coastal wetlands are still present in all of the Great Lakes (Figure 4-7). By illustrating and documenting differences in wetland health in these ways, 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 these results are beyond the scope of this report. The condition of sites based on a multivariate "human footprint" stressor that incorporates measures of all three stressor variables (agriculture, development, and wetland area) was reported.

Throughout the Great Lakes basin, coastal wetland health based on wetland breeding anurans significantly increased over the past 10 years, 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. A contributing factor that may help explain recent increases in IECs over the past 10 years is that since 2011 the coverage of surveys in coastal wetlands throughout the Great Lakes basin has improved dramatically, especially in the upper Great Lakes, due to the implementation of the CWMP (Figure 3). This change in coverage may have confounded some or all of the Improving trends to an unknown extent. In other words, a certain amount of the improvement may be due to differences in the sites surveyed before and after 2011. Perhaps a combination of improvements in survey coverage and other unknown factors contributed to the apparent increases in IECs reported here. In addition, in an ongoing effort to improve the performance and utility of the IEC for this sub-indicator, the sensitivity was improved for the metrics used to calculate the IECs. In the 2017 report, IECs were based on 7 wetland breeding anuran species or groups of wetland breeding anuran species, whereas this 2019 report is based on 8 species or species groups, with the addition of Wood Frog (*Rana sylvatica*). This change in metrics may have also had unknown influence on some of the differences between the 2017 report and this report, such as the difference in the status and trend overall going from Poor and Unchanging in the 2017 report to Fair and Improving in this 2019 report.

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 beyond 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 (Figure 4, 7). The ability to compare coastal and inland wetlands due to differences in sample sizes was best for Lake Erie and Lake Ontario, whereas it was limited for the other lake basins. In some lake basins, similar patterns across coastal and inland wetlands were found, whereas in other lake basins somewhat contrasting patterns occurred (Table 2, Figure 4). Wetland health as represented by wetland breeding anurans may be responding to different intensities of stressors in coastal versus inland wetlands within certain watersheds. For instance, a previous study using only the GLMMP dataset observed that occupancy of certain wetland breeding anuran species was lower at coastal marshes compared to inland marshes (Tozer 2013). As well, changes in lake levels may lead to dramatic changes in the area and/or quality of coastal wetland anuran habitat and associated IECs, perhaps contributing to the Improving trends in coastal wetlands reported here. If lake levels contribute more to changes in the quality of habitat and associated frog assemblages in Great Lakes coastal wetlands compared to inland wetlands, then the more stable inland wetlands near the coastal zone might be important population reservoirs during certain periods. Thus, continued sampling of both coastal and inland wetlands throughout the Great Lakes basin is needed to completely monitor and assess the health of wetlands based on anurans throughout the entire region.

The overall Fair status and Improving trend reported for coastal wetlands throughout the Great Lakes basin contrasts with some previous reports for this sub-indicator, which noted overall Poor status and Deteriorating trends based on the prevalence of significant negative trends in occupancy among 8 species or species groups of wetland breeding anuran using the GLMMP dataset alone (e.g., Tozer 2014). The apparent discrepancy in overall status and trend between this report and previous reports is likely at least partially due to differences in sampling coverage, with

previous reports summarizing the status and trend of predominantly 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. Thus, the overall Poor status and Deteriorating trend reported previously may have only been most representative, for instance, of the current lower status reported for Lake Erie and Lake Ontario. Nevertheless, it is important to note that the patterns summarized in this report are based on a comprehensive IEC metric, which represents the collective responses of multiple wetland breeding anuran species to wetland condition. Therefore, one should not lose sight of the fact that some species, such as the Western Chorus Frog (*Pseudacris triseriata*), have experienced long-term declines at various scales in the Great Lakes (e.g., Tozer 2013); these populations may be responding in species-specific ways to environmental stressors that warrant unique actions or present unique opportunities for improving wetland health. The results show no significant relationship between Chorus Frog occurrence and the combined stressor gradient (Figure 1), so it appears that across the Great Lakes Chorus Frogs are responding to factors other than the stressors that were measured or that local or regional decreases are offset by local or regional increases elsewhere.

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 (also known as SOLEC) 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. 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. Finally, the Coastal Wetland Amphibians sub-indicator can 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).

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. Bird Studies Canada's Great Lakes Marsh Monitoring Program, which draws upon the dedication and skills of hundreds of citizen scientists 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). 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 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.

The data for this sub-indicator are collected via various projects and organizations using paid professional staff, volunteer citizen scientists, or a mix of the two. Paid staffers are useful for surveying wetlands that are difficult to access or are located too far from willing volunteers. This is especially true for many wetlands around Lake Superior and along the eastern side of Georgian Bay on Lake Huron. Paid staffers are also advantageous for detecting certain rare species or groups of species that require advanced identification knowledge, and which are required for use in certain community indices and metrics. Trained Citizen Scientists on the other hand, where available, are often excellent cost-effective reliable alternatives to paid staffers, especially when identification abilities of each observer are verified with visual and acoustical tests. Thus, the suite of data being collected for this sub-indicator can be continually improved through ongoing assessments of ways to best utilize a mix of paid staffers and Citizen Scientists. This will require a balance between cost, quality, and spatial extent of data and associated information on coastal wetland ecosystem health.

Additional Information

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 and attribution project (Albert and Simonson, 2004; Ingram and Potter, 2004). Such wetland loss and stress is especially apparent in the southern portion of the Great Lakes basin (Allan et al. 2013, Bourgeau-Chavez et al. 2015). Indeed, we found that IECs based on wetland breeding anuran community data were lowest in these southern areas, which suggests that wetland loss and stress are contributing to lower anuran-based IECs in the lower Great Lakes.

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 frog 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. The BR functions were also developed based on information from three stressor gradients: agriculture, development, and wetland area. 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.

Acknowledgments

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List of Tables

Table 1. Wetland breeding anuran species and groups of species (in bold) (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

Table 2. Long-term (1995-2017) and 10-year (2007-2017) trends in the anuran-based 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 (%/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 calcu-

late reliable trends are shown with dashes. Trends calculated using equation 4 in Smith et al. (2014) with 1000 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

List of Figures

Figure 1. Biotic response functions (lines) for selected anuran species from coastal wetlands throughout the Great Lakes basin. Shown is the probability of occurrence as a function of a combined "human footprint" variable incorporating environmental condition due to agriculture, development, and wetland area (0 = poor condition, 10 = good condition). Species with biotic response functions shown as red dashed lines occur more frequently in wetlands in poor condition, whereas species shown with green solid lines occur more frequently in wetlands in good condition. Circles represent binned data at 10 observations per bin. See Table 1 for scientific names.
Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program

Figure 2. Wetlands surveyed for anurans from 1995-2017 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 (upper map) and as a function of coastal versus inland (lower map). Note that coastal wetlands (n = 1,114) far outnumber inland wetlands (n = 635), 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 1995-2017 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 1995-2017 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 2017 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 (2013-2017). 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 was used in ArcGIS to calculate mean values within 3000 m cells, and then used the focal statistics tool on the resulting means to calculate average values within a 2-cell circular window, which we refer to as "smoothed IEC". The most recent 5 year 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 (2013-2017) throughout 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

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No.	Common name	Scientific name(s)
1	American Toad	*Anaxyearus americanus*
2	Bullfrog	*Rana catesbeiana*
3	**Chorus Frog** (Boreal Chorus Anuran, Western Chorus Anuran)	*Pseudacris maculata, Pseudacris triseriata*
4	**Gray Treefrog** (Eastern Gray Treefrog, Cope's Gray Treefrog)	*Hyla versicolor, Hyla chrysoscelis*
5	Green Frog	*Rana clamitans*
6	Northern Leopard Frog	*Rana pipiens*
7	Spring Peeper	*Pseudacris crucifer*
8	Wood Frog	*Rana sylvatica*

Table 1. Wetland breeding anuran species and groups of species (in bold) (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

Wetland type	Basin	Start year	End year	Trend	Lower	Upper
Coastal	**Overall**	**1995**	**2017**	**2.1**	**1.2**	**3.0**
Coastal	**Overall**	**2007**	**2017**	**2.9**	**1.5**	**4.4**
Coastal	Superior	1995	2017	–	–	–
Coastal	Superior	2007	2017	–	–	–
Coastal	Michigan	1995	2017	–	–	–
Coastal	Michigan	2007	2017	3.5	-0.8	7.7
Coastal	Huron	1995	2017	–	–	–
Coastal	Huron	2007	2017	0.4	-1.8	2.6
Coastal	Erie	1995	2017	1.2	-0.2	2.6
Coastal	Erie	2007	2017	0.5	-2.0	3.1
Coastal	**Ontario**	**1995**	**2017**	**2.7**	**1.0**	**4.4**
Coastal	**Ontario**	**2007**	**2017**	**3.8**	**1.0**	**6.7**
Inland	Overall	1995	2017	-0.3	-0.8	0.3
Inland	Overall	2007	2017	0.0	-1.0	0.9
Inland	Superior	1995	2017	–	–	–
Inland	Superior	2007	2017	–	–	–
Inland	Michigan	1995	2017	–	–	–
Inland	Michigan	2007	2017	–	–	–
Inland	Huron	1995	2017	-0.2	-1.1	0.7
Inland	Huron	2007	2017	-0.7	-2.9	1.4
Inland	Erie	1995	2017	-0.4	-1.3	0.5
Inland	Erie	2007	2017	-1.0	-2.6	0.5
Inland	Ontario	1995	2017	0.1	-1.0	1.2
Inland	Ontario	2007	2017	0.4	-1.0	1.8

Table 2. Long-term (1995-2017) and 10-year (2007-2017) trends in the anuran-based 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 (%/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 1000 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

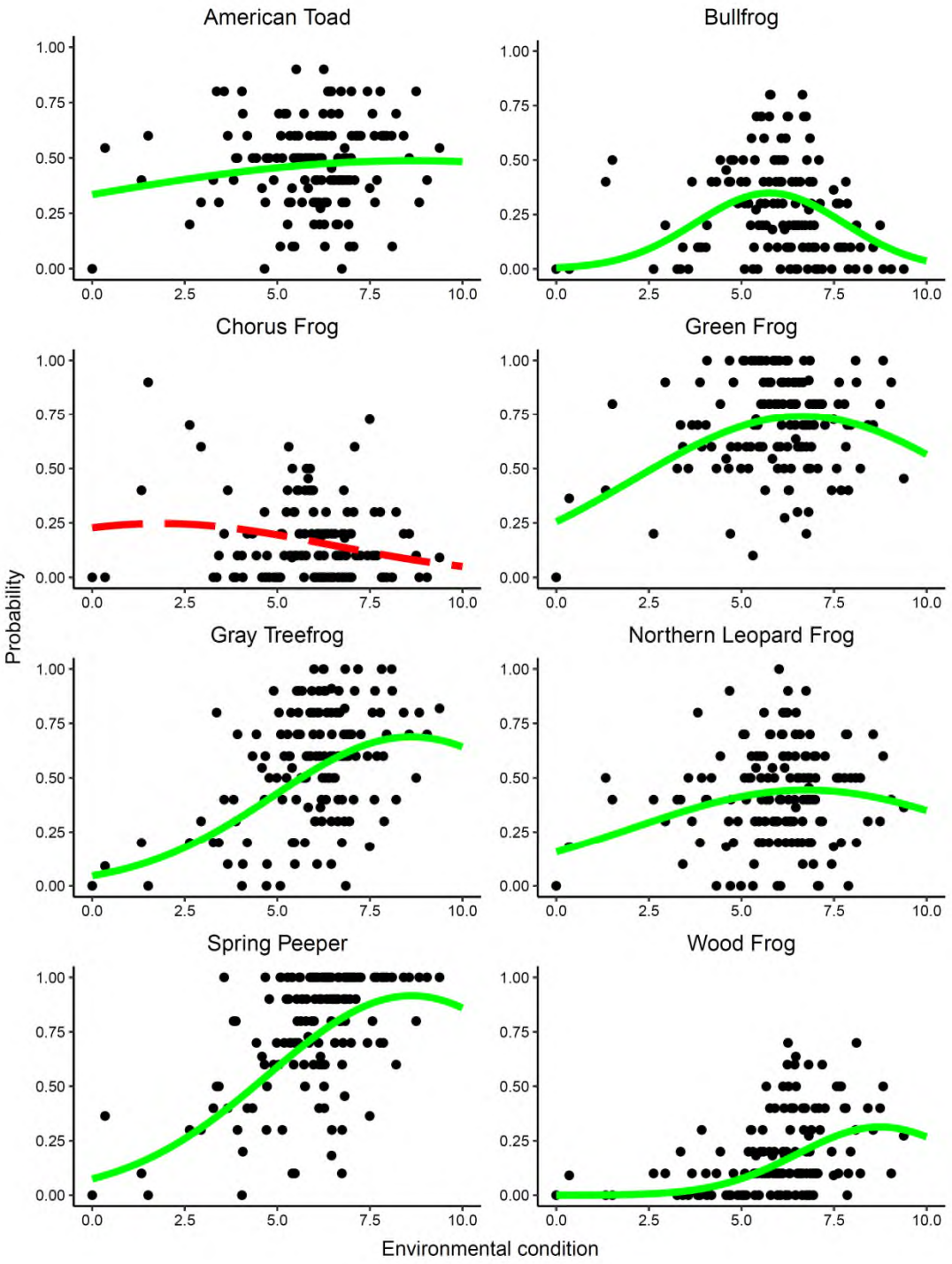


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Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program

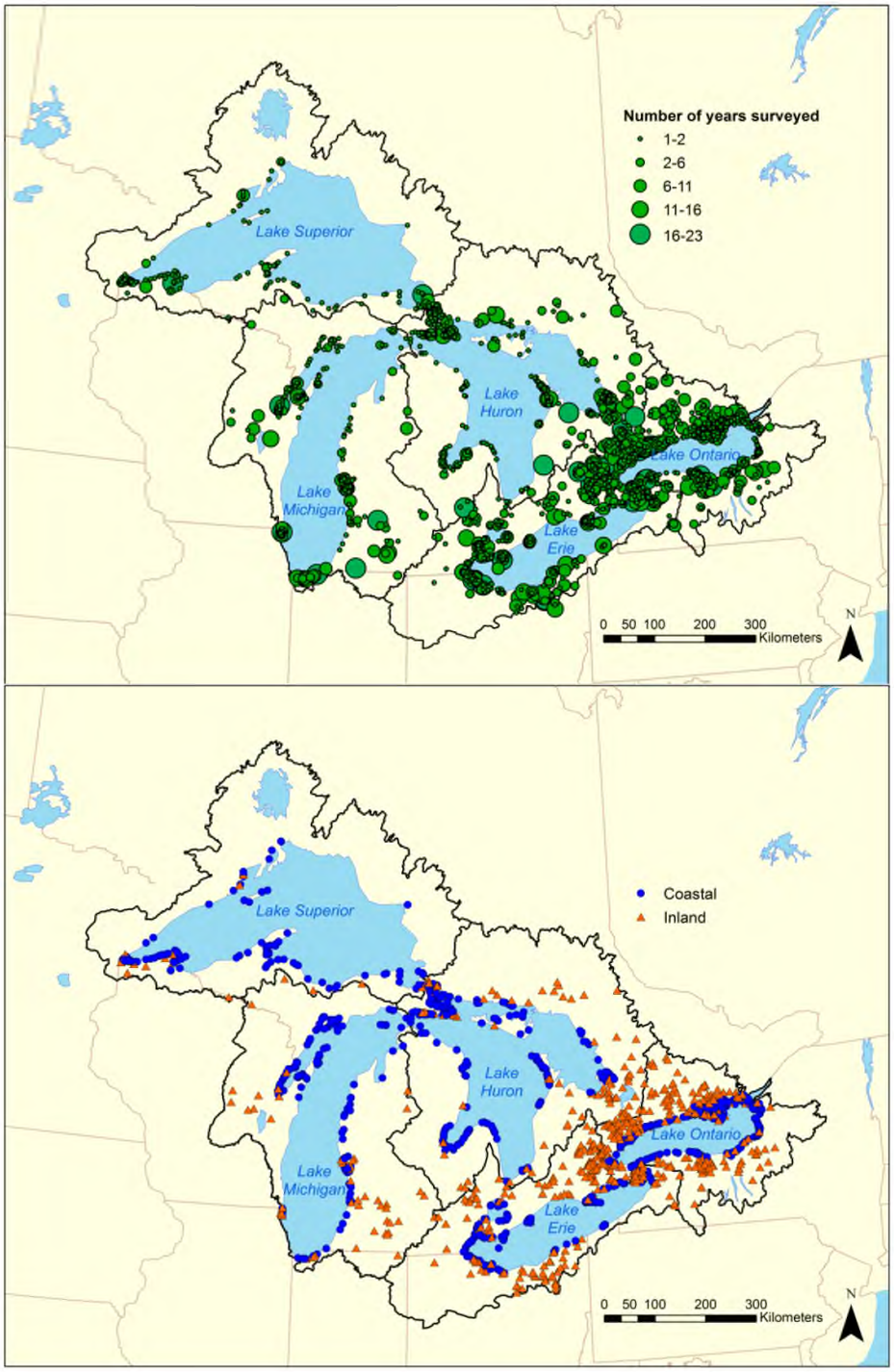


Figure 2. Wetlands surveyed for anurans from 1995-2017 throughout the Great Lakes basin for the purpose of estimating anuran-based indices of wetland health. Shown are wetlands as a function of the number of years that each wetland was surveyed (upper map) and as a function of coastal versus inland (lower map). Note that coastal wetlands (n = 1,114) far outnumber inland wetlands (n = 635), 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

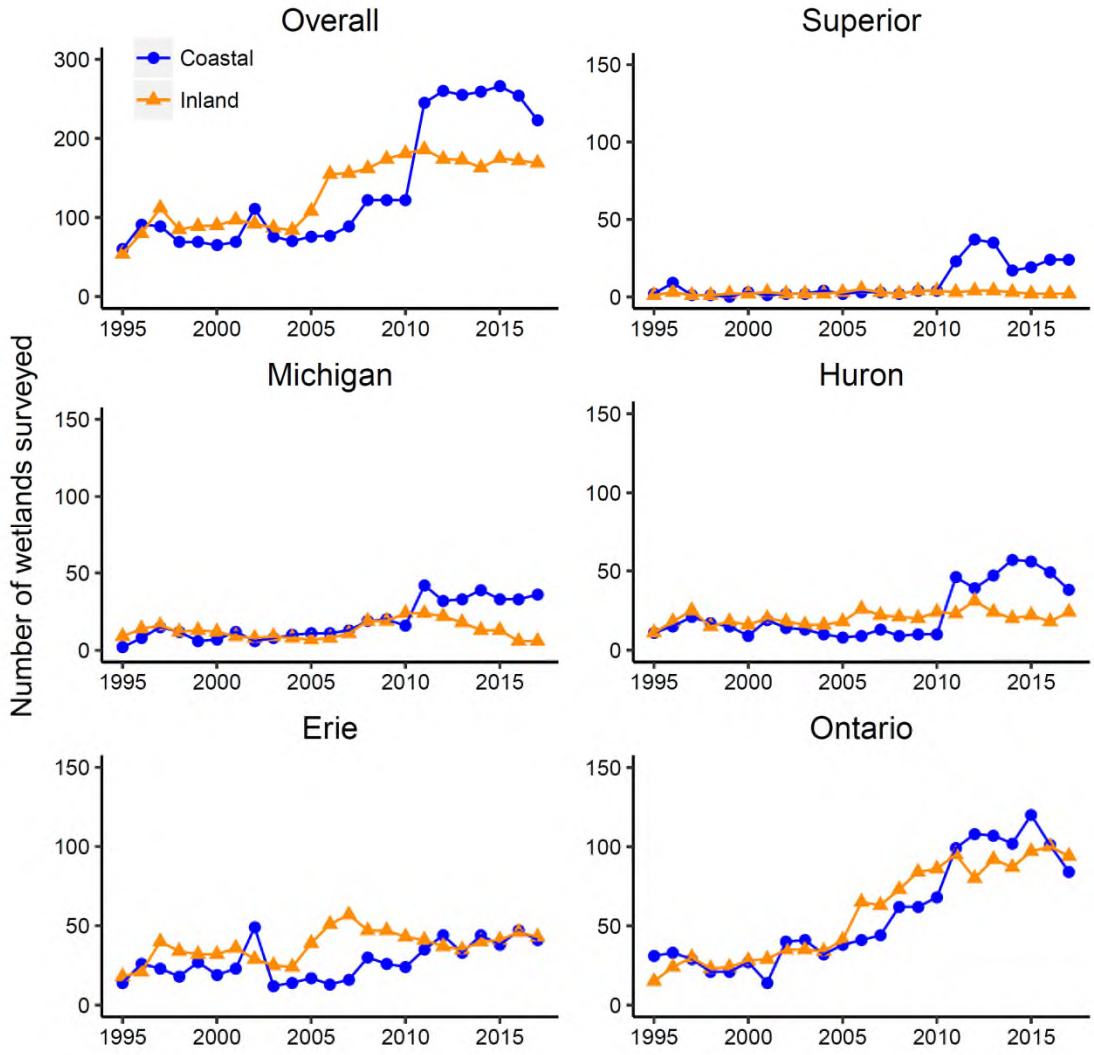


Figure 3. Number of wetlands surveyed for anurans per year from 1995-2017 throughout the Great Lakes basin for the purpose of estimating anuran-based indices of wetland health. Shown are wetlands surveyed as a function of the entire Great Lakes basin (overall) and each individual lake basin for coastal and inland wetlands.
Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program

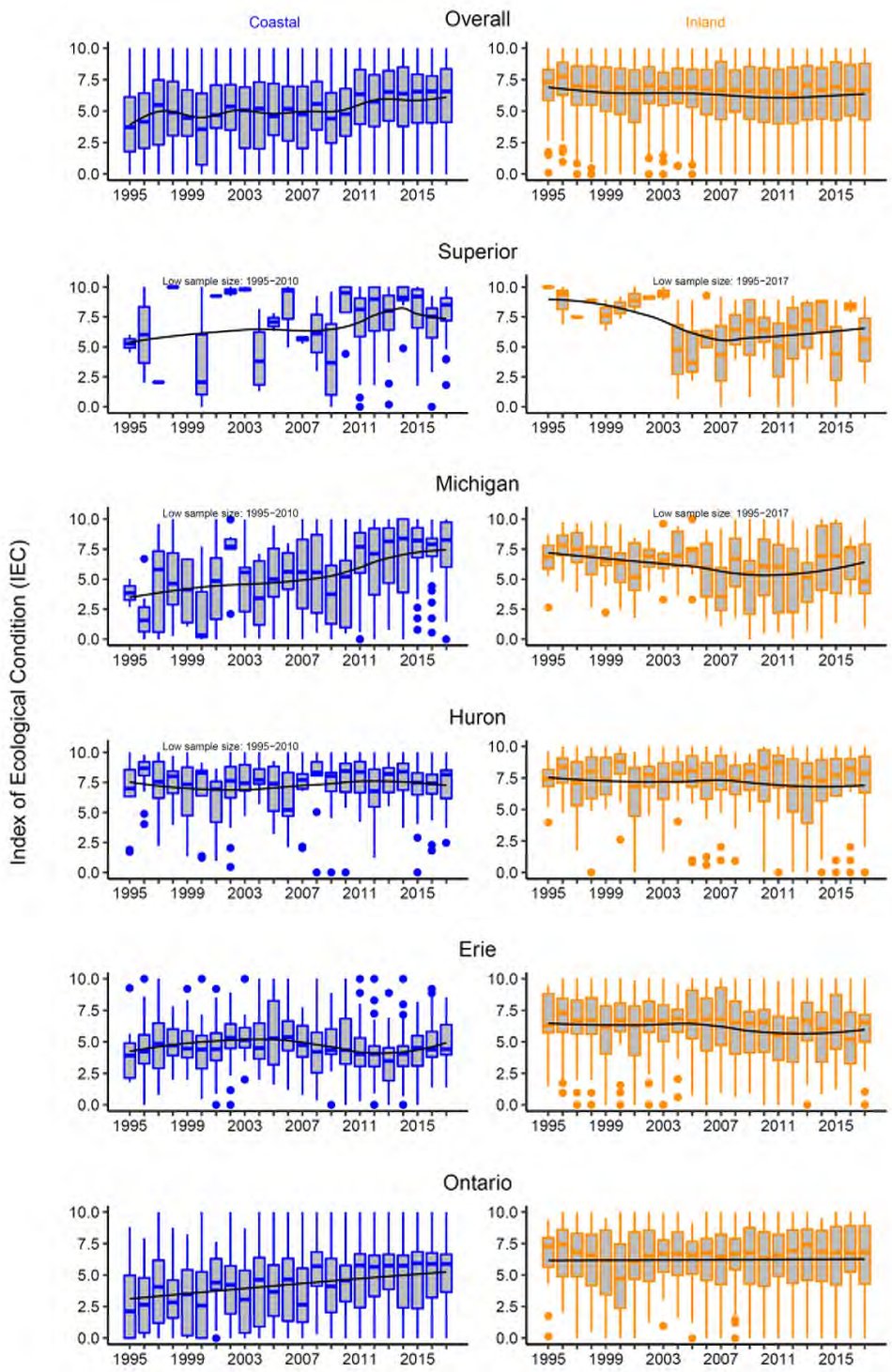


Figure 4. Temporal trends in the index of ecological condition (IEC) based on anuran community data from 1995-2017 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal (blue) and inland (orange) wetlands. Shown are box-and-whisker plots of wetland-level means in 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

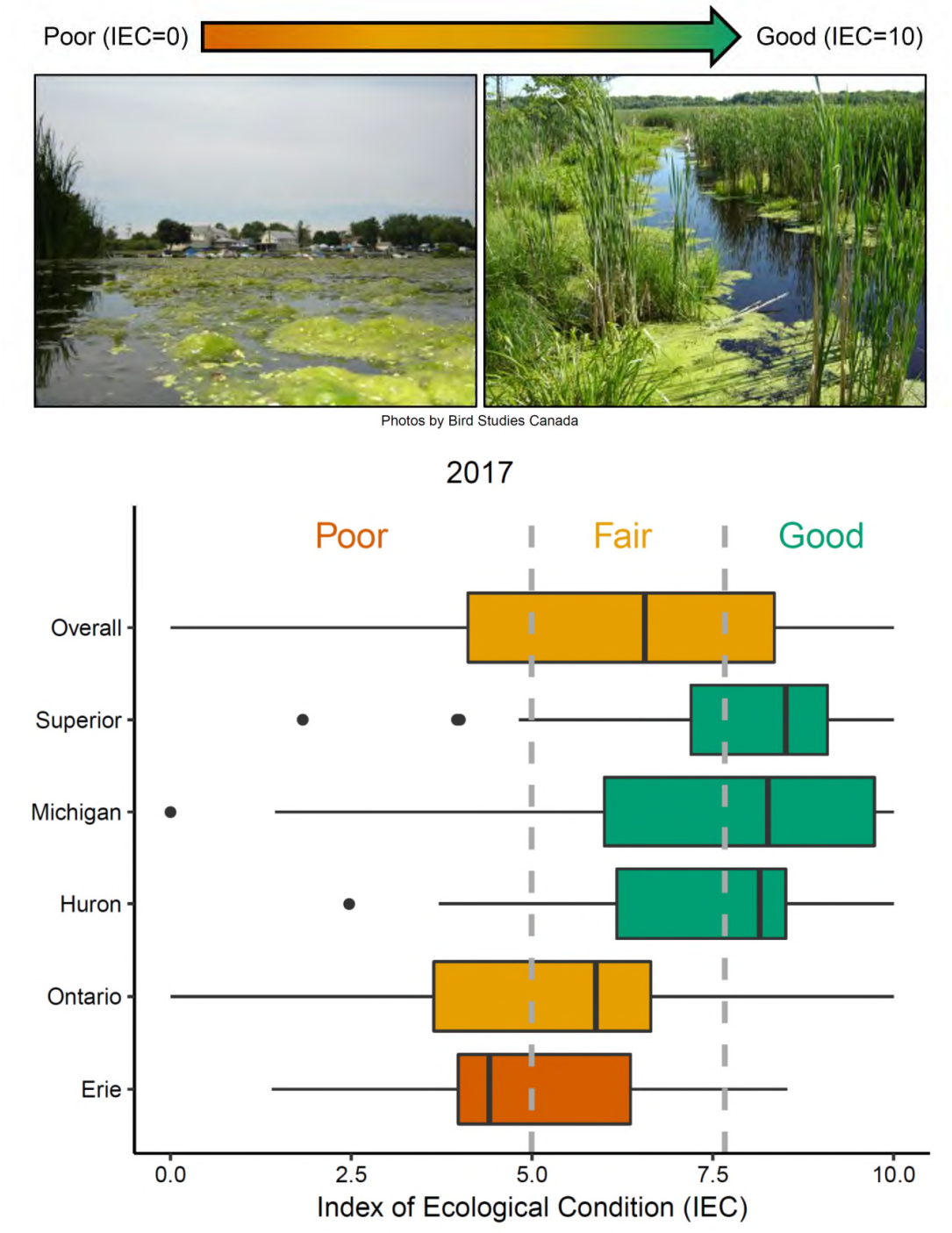


Figure 5. Distribution of the index of ecological condition (IEC) based on anuran community data in 2017 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal wetlands. Shown are box-and-whisker plots of wetland-level means. Whiskers represent 1.5 times the inter-quartile 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

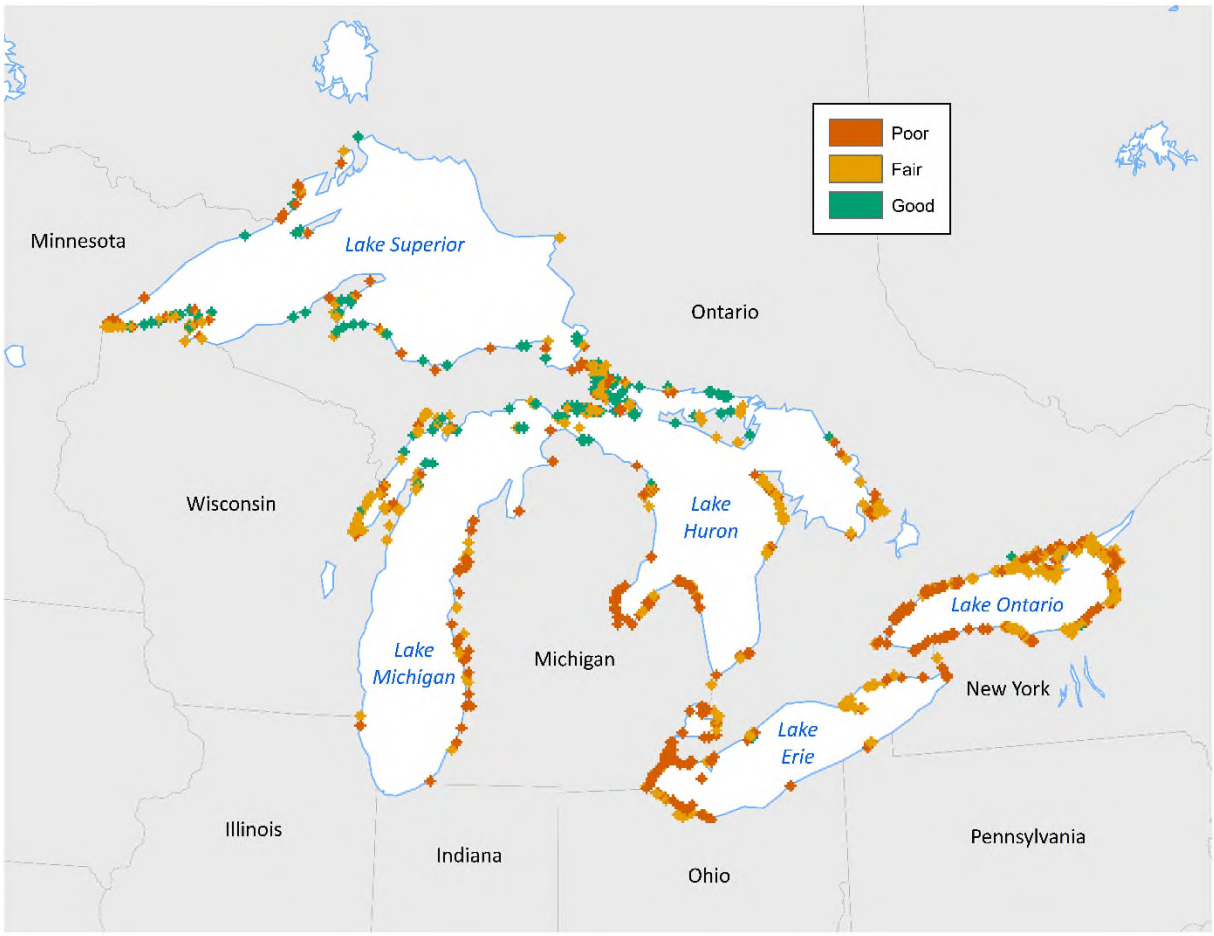


Figure 6. Index of ecological condition (IEC) throughout coastal wetlands of the Great Lakes based on anuran community data from the past 5 years (2013-2017). Shown are smoothed wetland-level mean IEC values, averaged across years for wetlands that were sampled in multiple years. To smooth the data, the point-to-raster tool was used in ArcGIS to calculate mean values within 3000 m cells, and then used the focal statistics tool on the resulting means to calculate average values within a 2-cell circular window, which we refer to as "smoothed IEC". The most recent 5 year 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

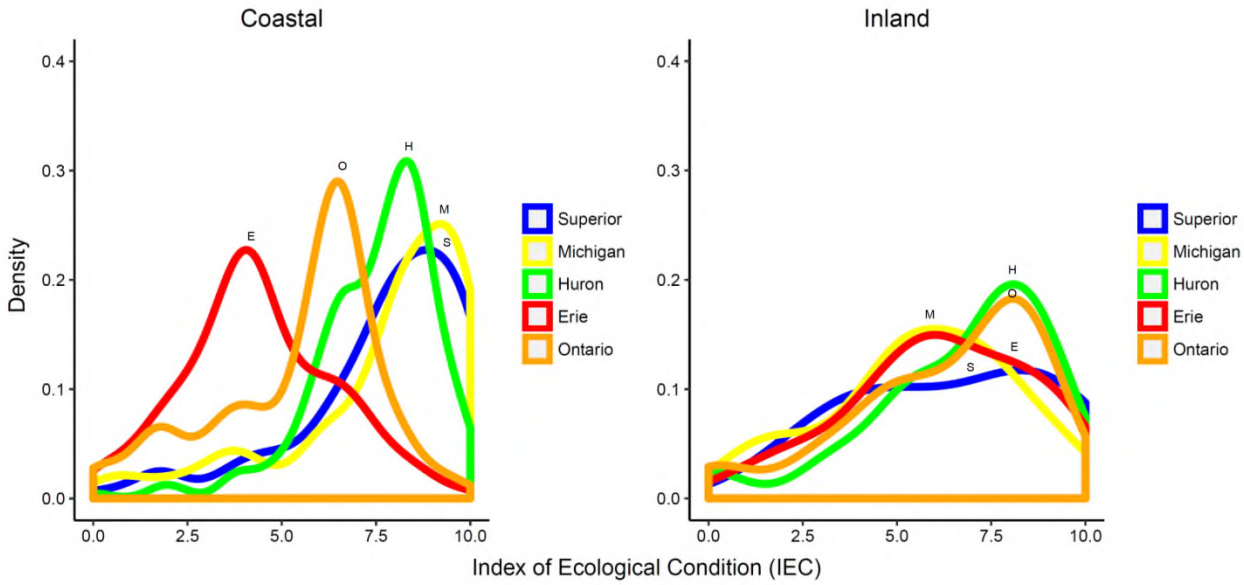


Figure 7. Density plots of the index of ecological condition (IEC) based on anuran community data from the past 5 years (2013-2017) throughout each individual lake basin for coastal and inland wetlands. Shown are plots based on wetland-level mean IEC values, averaged across years for wetlands that were 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: Improving
Long-term Trend (1995-2017): Improving
Rationale: Median index of ecological condition (IEC), an objective biotic indicator summarizing standardized observations of wetland birds in coastal wetlands was 5.5 (out of 10) based on data from 2017, reflecting a significant increase of 2.4%/year over the past 10 years. 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.

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: Undetermined
Long-term Trend (1995-2017): Undetermined
Rationale: Median IEC was 6.7 based on data from 2017.

Lake Michigan
Status: Good
10-Year Trend: Undetermined
Long-term Trend (1995-2017): Undetermined
Rationale: Median IEC was 6.5 based on data from 2017.

Lake Huron (including St. Marys River)
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1995-2017): Undetermined
Rationale: Median IEC was 6.0 based on data from 2017 and did not significantly increase or decrease over the most recent 10 years.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1995-2017): Unchanging
Rationale: Median IEC was 4.6 based on data from 2017 and did not significantly increase or decrease over the most recent 10 years.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1995-2017): Improving
Rationale: Median IEC was 5.1 based on data from 2017 and significantly increased by 4.1%/year over the past 10 years.

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 2019 reporting, status is based on data from 2017) 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 CWMP. Note that the values defining Poor, Fair, and Good below have changed since the previous report (Tozer et al. 2017b) 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 > 66th percentile (for 2019 reporting, median IEC > 5.8).
Fair: Some ecosystem components are in acceptable condition; 33rd percentile ≤ median IEC ≤ 66th percentile (for 2019 reporting, 3.9 ≤ median IEC ≤ 5.8).
Poor: Very few or no ecosystem components are in acceptable condition; median IEC < 33rd percentile (for 2019 reporting, median IEC < 3.9).
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 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 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 2015) 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. 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 2019 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; and (4) generalist predators (e.g., northern raccoon [*Procyon lotor*]) associated with anthropogenic habitats in the surrounding landscape increase (Brazner et al. 2007a, 2007b; Crosbie and Chow-Fraser 1999; Howe

et al. 2007a; Grandmaison and Niemi 2007; Naugle et al. 2000; Smith and Chow-Fraser 2010 a, 2010b; Tozer et al. 2010). 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 Bird Studies Canada's Great Lakes Marsh Monitoring Program (GLMMP), 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). 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 Great Lakes Coastal Wetland Monitoring Program (CWMP) led by Central Michigan University was initiated in 2011 and currently is scheduled to operate until at least 2020 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 birds and associated wetland health.

Surveys—Wetland birds were sampled to an unlimited distance from a 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 sampled 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 double 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, 2017a). 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) or both 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).

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 trend 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 (previously known as SOLEC) 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 like a regional epidemic that affects individuals regardless of wetland condition.

In this report, we assessed bird community health based on multi-species data from across the Great Lakes basin

(Howe et al. 2007a, 2007b; Hanowski et al. 2007a, 2007b; Tozer 2013, 2016). Quantitative data were used for wetland birds at thousands of sample points throughout the Great Lakes basin in both the U.S. and Canada. At 806 of these sample points, quantitative information was available on three potential environmental stressors (Panci et al. 2017): 1) agricultural intensity in the contributing watershed (i.e., the landscape draining into the wetland), 2) non-agricultural landscape development such as roads, buildings, and human population density in the contributing watershed, and 3) wetland area and fragmentation, measured by the total wetland area within 1 km of the sampled wetland's centroid. For convenience, these gradients are referred to in this report as agriculture, development, and wetland area, respectively. Clearly, many other stressors affect bird communities in coastal wetlands, but agricultural intensity, non-agricultural landscape development, and wetland area provide quantitative yardsticks from which one can identify sensitive species and community variables (Brazner et al. 2007a, 2007b). Species that are sensitive to these easily-measured landscape variables are likely vulnerable to other, unmeasured stressors. Site-specific values for each of the three measured stressors (agricultural intensity, development, and wetland area/habitat destruction) were combined by principal components analysis into a single gradient that approximates the "human footprint" associated with a single wetland.

To avoid excessive zeros (absences) in the data, each of the 806 sites with accompanying environmental data and composite "human footprint" score were placed bins of 10 points each (or 6 in the last case) with similar measures of environmental stress. In other words, the 10 sites with the worst "human footprint" scores were placed in the first bin, then the 10 sites with the next best "human footprint" scores were placed in the second bin, and so on, until the last of the 81 bins was reached, which contained the last 6 sites with the very best "human footprint" scores. This created a new gradient of 81 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 bird species groups, including wetland obligates (i.e., bird species that exclusively depend on wetland habitat) 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 81 bins (Figure 1). The probability values were simply the proportions of sample points in each bin where each bird species or bird species group was present. 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 similar to the normal distribution (Gnass Giese et al. 2015). These BR functions (Figure 1) provided the basis for scoring new sites based on the presence/absence of 27 species/species groups (Table 1). Parameters of the best-fit mathematical function were estimated by computer iteration in R (R Core Team 2015) with package "iec" (https://github.com/ngwalton/iec). The three parameters (mean, standard deviation, and height) describe a bell-shaped or truncated Gaussian function within the range of 0-10.

The final suite of 27 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*; Belted Kingfisher, *Megaceryle alcyon*; Red-winged Blackbird, *Agelaius phoeniceus*), and species groups such as "rails," "diving ducks," "terns," and "Alder/Willow Flycatcher" (*Empidonax alnorum*/*Empidonax traillii*). 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 (e.g., Barn Swallow, *Hirundo rustica*; Common Grackle, *Quiscalus quiscula*) are more common in degraded wetlands, whereas some relatively disturbance intolerant wetland obligate species (e.g., Sandhill Crane, *Antigone canadensis*; Sedge Wren, *Cistothorus platensis*) are more common in less degraded wetlands (Figure 1). Unless included in taxonomic groups (e.g., "rails"), rare 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 27 species or species groups used to generate BR functions for calculating IECs are listed in Table 1.

The health of coastal wetlands was subsequently evaluated using the index of ecological condition (IEC), an objective biotic indicator introduced by Howe et al. (2007a, 2007b), improved by Gnass Giese et al. (2015), and compared to other similar indices for wetland birds by Chin et al. (2015). By recording the species present at a wetland, one can essentially work backward by iteration to 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 adjusted for 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, but IEC metrics were also calculated for inland wetlands, and were used only for comparison, and were not used for any of the assessments of status or trend. In addition, box-and-whisker plots of wetland-level mean IECs for coastal wetlands in each basin and overall for the most recent year (2017) are reported to illustrate current status in relation to defining values for Poor, Fair, and Good status. 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 (2013-2017) are reported to illustrate variation in distributions. To further illustrate distributions, these same data for coastal wetlands are also 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, and the most recent 5 years of data (2013-2017) were used 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."

Ecological Condition

Data coverage—The dataset available for generating IECs consisted of 37,096 point counts conducted at 5,189 sample points in 1,784 wetlands over 23 years from 1995-2017 throughout the Great Lakes basin (Figure 2). The number of years that each wetland was surveyed varied from 1 to 22, with a mean of 3.4 ± 3.9 (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 = 1,242; 70%) rather than inland (n = 542; 30%) 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 wetlands surveyed per year (260 ± 135 [mean ± SD]) ranged from 118 to 488 with substantially more wetlands surveyed from 2011-2017 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. Annual coverage also 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 ranged from 3.5 to 5.5 from 1995-2017, significantly increased by 2.4%/year over the past 10 years (1.1, 3.6) [lower, upper 95% confidence limits for % annual change] (Table 2, Figure 4), and was 5.5 (out of 10) based on data from 2017 (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, although the Improving trend may not be entirely real as discussed in more detail in the next paragraph below and in the discussion. Generally, IECs were lower in the southern portion of the Great Lakes basin south of the Canadian Shield where agricultural and human development footprints are bigger/stronger (Figure 6-7). Similar patterns occurred at inland wetlands overall, although the rate of improvement was less (Table 2; Figure 4, 7).
The status has not changed since the last report (Tozer et al. 2017b), whereas the trend has changed from Unchanging to Improving. The difference in trend may be due to one or more of several factors: 1) improved sensitivity of the metrics used to calculate the IEC since the last report; 2) substantial improvements by the Great Lakes Coastal Wetland Monitoring Program (CWMP) since 2011 in geographical coverage of coastal wetlands in Lake Superior and northern portions of Lake Michigan and Lake Huron, where bird communities tend to experience fewer stressors; and 3) changes in lake levels, which may lead to dramatic changes in the area and/or quality of coastal wetland bird habitat and associated IECs. Calculation of trends will be more reliable in future reports with the continuation of the CWMP because coverage of coastal wetlands will be more consistent over the most recent 10 years. In addition, as the number of years of sampling increases, the power to detect change also increases.

Lake Superior—Median IEC in coastal wetlands ranged from 1.8 to 8.9 from 1995-2017 (Figure 4) and was 6.7 based on data from 2017 (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 bird communities in Lake Superior is Good, 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 (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, which may account at least partially for some of the modest scores in comparison with those from other lakes (Figure 4-7).

The status is the same as in the last report, whereas the trend has changed from Unchanging to Undetermined. For this report, the current information is inadequate to perform reliable statistical significance testing for trends. Calculation of trends will be more reliable in future reports with the continuation of the CWMP because coverage of coastal wetlands will be more consistent over the most recent 10 years. In addition, as the number of years of sampling increases, the power to detect change also increases.

Lake Michigan—Median IEC in coastal wetlands ranged from 2.6 to 6.5 from 1995-2017 (Figure 4) and was 6.5 based on data from 2017 (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 bird communities in Lake Michigan is Good, and the trend is Undetermined. Similar 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 birds 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).

Since the last report, the status has changed from Fair to Good, and the trend has changed from Unchanging to Undetermined. 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. For this report, the current information is inadequate to perform reliable statistical significance testing for trends. Calculation of trends will be more reliable in future reports with the continuation of the CWMP because coverage of coastal wetlands will be more consistent over the most recent 10 years. In addition, as the number of years of sampling increases, the power to detect change also increases.

Lake Huron (including St. Marys River)—Median IEC in coastal wetlands ranged from 3.8 to 7.1 from 1995-2017, did not significantly increase or decrease over the past 10 years (Table 2, Figure 4), and was 6.0 based on data from 2017 (Figure 5). Based on these patterns, the status of coastal wetland health based on bird communities in Lake Huron is Good, and the trend is Unchanging. Similar patterns occurred at inland wetlands in the Lake Huron watershed (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).

There has been no change in status or trend since the last report.

Lake Erie (including St. Clair-Detroit River Ecosystem)—Median IEC in coastal wetlands ranged from 3.8 to 5.2 from 1995-2017, did not significantly increase or decrease over the past 10 years (Table 2, Figure 4), and was 4.6 based on data from 2017 (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 patterns occurred at inland wetlands in the Lake Erie watershed (Table 2; Figure 4, 7). Out of all of the lake basins, Lake Erie had the lowest median IEC in coastal wetlands, being only slightly worse than Lake Ontario (Figure 5-7).

The status has changed from Poor to Fair, and the trend has changed from Deteriorating to Unchanging. The difference in status and trend 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.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)—Median IEC in coastal wetlands ranged from 2.9 to 5.3 from 1995-2017, significantly increased by 4.1%/year over the past 10 years (2.1, 6.0) (Table 2, Figure 4), and was 5.1 based on data from 2017 (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. Similar patterns occurred at inland wetlands in the Lake Ontario watershed (Table 2; Figure 4, 7). Out of all of the lake basins, Lake Ontario had the second-lowest median IEC in coastal wetlands, being only slightly better than Lake Erie (Figure 5-7).

There has been no change in status or trend since the last report.

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 Huron, and Lake Michigan being Good, and Lake Ontario and Lake Erie 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 southern portion of Lake Michigan and Lake Huron and all of Lake Erie and Lake Ontario compared to most locations in Lake Superior (Figure 6). These patterns are probably due to greater anthropogenic stress from agriculture, development, and perhaps wetland loss in Lake Michigan and Lake Huron south of the Canadian Shield, and in all of Lake Erie and Lake Ontario compared to most locations in Lake Superior (Allan et al. 2013, Bourgeau-Chavez et al. 2015, Danz et al. 2007, Niemi et al. 2009). Nonetheless, some high quality coastal wetlands are still present in all of the Great Lakes (Figure 4-7). By illustrating and documenting differences in wetland health in these ways, 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 these results are beyond the scope of this report. The condition of sites based on a multivariate "human footprint" stressor that incorporates measures of all three stressor variables (agriculture, development, and wetland area) was reported.

Throughout the Great Lakes basin, coastal wetland health based on wetland birds significantly increased over the past 10 years, with non-significant positive trends in most individual lake basins during the same period (Table 2, Figure 4). One exception was Lake Erie, with the only non-significant negative trend in coastal IECs over the past 10 years (Table 2). The cause of recent increases in IECs is unclear, although it may be at least partially due to concurrent changes in water levels for some bird species in some wetlands (Timmermans et al. 2008), or other factors (see next paragraph). By contrast, the non-significant decrease in coastal IECs in Lake Erie may be associated with increasing amounts of anthropogenic stress from agriculture, development, and perhaps wetland loss (e.g., Danz et al. 2007, Wolter et al. 2006). Thus, given that Lake Erie had the lowest median coastal IEC of all of the lake basins and was the only lake basin where coastal IECs decreased (albeit non-significantly) over time, the health of Lake Erie's coastal wetlands appear to be especially compromised compared to coastal wetlands in the remaining lake basins. The low median coastal IEC and the non-significant declining trend may also indicate that Lake Erie is experiencing unique stressors or relatively high intensities of stressors compared with stressors in the other lake basins.

A contributing factor that may help explain recent increases in coastal IECs over the past 10 years is that since 2011 the coverage of surveys in coastal wetlands throughout the Great Lakes basin has improved dramatically, especially in the upper Great Lakes, due to the implementation of the CWMP (Figure 3). This change in coverage may have confounded some or all of the Improving trends to an unknown extent. In other words, a certain amount of the increases may be due to differences in the sites surveyed before and after 2011. Perhaps a combination of changes in water levels and associated habitat for some bird species in some wetlands and improvements in survey coverage, as well as other unknown factors, contributed to the apparent increases in IECs reported here. In addition, in an ongoing effort to improve the performance and utility of the IEC for this sub-indicator, the sensitivity was improved for the metrics used to calculate the IECs. In the 2017 report, IECs were based on 52 wetland or wetland-associated bird species. By contrast, this 2019 report is based on a more restricted set of 27 wetland bird species or species groups that are more directly dependent on wetlands, and by association, more effectively represent coastal wetland health. This change in metrics may have also had unknown influence on some of the differences between the 2017 report and this report, such as the difference in the trend overall going from Unchanging in the 2017 report to Improving in this 2019 report.

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 bird habitat (Grabas et al. 2012, Grabas and Rokitnicki-Wojcik 2015, Tozer 2016), may have also contributed to the increasing coastal wetland bird IECs that

were observed in Lake Ontario and perhaps beyond 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). The ability to compare coastal and inland wetlands due to differences in sample sizes was best for Lake Erie and Lake Ontario, whereas it was limited for the other lake basins. Similar patterns occurred across coastal and inland wetlands, with the following exception. Overall and in Lake Ontario, coastal IECs significantly increased over the past 10 years, but inland IECs exhibited no significant corresponding increase (Table 2, 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. For instance, a previous study using only the GLMMP dataset observed that mean abundance of certain wetland-dependent bird species was lower at coastal wetlands compared to inland wetlands, suggesting that coastal wetlands may be influenced differently than inland wetlands (Tozer 2013). Thus, continued sampling of both coastal and inland wetlands throughout the Great Lakes basin is needed to completely monitor and assess the health of wetlands based on birds throughout the entire region.

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, which noted overall Poor status and Deteriorating trends based on the prevalence of significant negative trends in abundance among approximately 20 wetland-dependent bird species using the GLMMP dataset alone (e.g., Tozer 2014). The apparent discrepancy in overall status and trend between this report and previous reports is likely at least partially due to differences in sampling coverage. Previous reports summarizing the status and trend were predominantly 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. Thus, the overall Poor status and Deteriorating trend reported previously may have only been most representative, for instance, of the current lower status reported for Lake Erie and Lake Ontario. Nevertheless, it is important to note that the patterns summarized in this report are based on a comprehensive IEC metric, which represents the collective responses of many wetland bird species to wetland condition. Therefore, one should not lose sight of the fact that there are particular species, including bitterns (e.g., *Botaurus*), shallow- (e.g., *Porzana*) and deep-water rails (e.g., *Gallinula*), and marsh-nesting terns (e.g., *Chlidonias*), which have experienced long-term declines at various scales in the Great Lakes (e.g., Tozer 2013, 2016). These taxa may be responding to environmental stressors in species-specific ways, warranting unique actions or presenting unique opportunities for improving 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 (previously known as SOLEC) 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). 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). 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). And finally, it can be expected to be influenced by climate-related impacts and associated interactions with various other factors, such as through direct mortality of certain coastal wetland bird species as 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). Thus, 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).

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 bird monitoring programs operating throughout the Great Lakes. Bird Studies Canada's Great Lakes Marsh Monitoring Program, which draws

upon the dedication and skills of hundreds of citizen scientists 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). 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 Limitations

Wetland birds are highly mobile and most are dependent on wetlands for only portions of their life cycle. Temporal trends in local bird populations, and by association related bird-based indices of wetland health, can be influenced by factors external to Great Lakes coastal wetlands, such as conditions on the wintering grounds and during migration. For this reason, work will be required to identify impacts to marsh bird numbers during the birds' entire life cycle, including potential carry-over effects driving numbers within Great Lakes coastal wetlands.

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; 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 the 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.

Data for this sub-indicator are collected via various projects and organizations using paid professional staff, volunteer Citizen Scientists, or a mix of the two. Paid staffers are useful for surveying wetlands that are difficult to access or are located too far from willing volunteers. This is especially true for many wetlands around Lake Superior and along the eastern side of Georgian Bay on Lake Huron. Paid staffers are also advantageous for detecting certain rare species or groups of species that require advanced identification knowledge, and which are required for use in certain community indices and metrics. Trained Citizen Scientists on the other hand, where available, are often excellent cost-effective reliable alternatives to paid staffers, especially when identification abilities of each observer are verified with visual and acoustical tests. Thus, the suite of data collected for this sub-indicator can be continually improved through ongoing assessments of ways to best utilize a mix of paid staffers and Citizen Scientists. This will require a balance between cost, quality, and spatial extent of data and associated information on coastal wetland ecosystem health.

Additional Information

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 and attribution project (Albert and Simonson, 2004; Ingram and Potter, 2004). Such wetland loss and stress are especially apparent in the southern portion 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 lowest in these southern areas, which suggests that wetland loss and stress are contributing to lower bird-based IECs in the lower Great Lakes.

One approach to identifying appropriate actions and opportunities for improving the health of coastal wetlands is to identify factors that will promote colonization or that will reduce extinction 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 International Joint Commission 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 three stressor gradients: agriculture, development, and wetland area. 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.

Acknowledgments

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List of Tables

Table 1. Wetland bird species and species groups used to estimate IECs.
Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program
Table 2. Trends in IECs over time.
Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program

List of Figures

Figure 1. Biotic response functions (lines) for selected bird species from coastal wetlands throughout the Great Lakes basin. Shown is the probability of occurrence as a function of a combined "human footprint" variable incorporating environmental condition due to agriculture, development, and wetland area (0 = Poor condition, 10 = Good condition). Species with biotic response functions shown as red dashed lines occur more frequently in wetlands in Poor condition, whereas species shown with green solid lines occur more frequently in wetlands in Good condition. Circles represent binned data at 10 observations per bin. See Table 1 for scientific names.
Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program
Figure 2. Wetlands surveyed for birds from 1995-2017 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 = 1,242) far outnumber inland wetlands (n = 542), 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 1995-2017 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.
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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 2017 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 bird community data from the past 5 years (2013-2017). 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 3000 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 (2013-2017) 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

Last Updated

State of the Great Lakes 2019 Technical Report

No.	Common name	Scientific name(s)
1	**Aerial foragers other than Barn Swallow** (Bank Swallow, Cliff Swallow, Northern Rough-winged Swallow, Purple Martin, Tree Swallow)	*Riparia riparia, Petrochelidon pyrrhonota, Stelgidopteryx serripennis, Progne subis, Tachycineta bicolor*
2	Alder Flycatcher, Willow Flycatcher	*Empidonax alnorum, Empidonax traillii*
3	American Bittern	*Botaurus lentiginosus*
4	American Coot, Common Gallinule	*Fulica americana, Gallinula galeata*
5	American Goldfinch	*Spinus tristis*
6	Bald Eagle, Osprey	*Haliaeetus leucocephalus, Pandion haliaetus*
7	Barn Swallow	*Hirundo rustica*
8	Belted Kingfisher	*Megaceryle alcyon*
9	Common Grackle	*Quiscalus quiscula*
10	Common Yellowthroat	*Geothlypis trichas*
11	**Dabbling ducks other than Mallard** (American Black Duck, American Wigeon, Blue-winged Teal, Gadwall, Green-winged Teal, Northern Pintail, Northern Shoveler, Wood Duck)	*Anas rubripes, Mareca americana, Spatula discors, Mareca strepera, Anas crecca, Anas acuta, Spatula clypeata, Aix sponsa*
12	**Diving ducks** (Canvasback, Common Goldeneye, Common Merganser, Hooded Merganser, Lesser Scaup, Long-tailed Duck, Red-breasted Merganser, Redhead, Ruddy Duck)	*Aythya valisineria, Bucephala clangula, Mergus merganser, Lophodytes cucullatus, Aythya affinis, Clangula hyemalis, Mergus serrator, Aythya americana, Oxyura jamaicensis*
13	European Starling	*Sturnus vulgaris*
14	Killdeer	*Charadrius vociferus*
15	Mallard	*Anas platyrhynchos,*
16	Marsh Wren	*Cistothorus palustris*
17	Northern Harrier	*Circus cyaneus*
18	Pied-billed Grebe	*Podilymbus podiceps*
19	**Rails** (King Rail, Sora, Virginia Rail, Yellow Rail)	*Rallus elegans, Porzana carolina, Rallus limicola, Coturnicops noveboracensis*
20	Red-winged Blackbird	*Agelaius phoeniceus*
21	Sandhill Crane	*Antigone canadensis*
22	Sedge Wren	*Cistothorus platensis*
23	Song Sparrow	*Melospiza melodia*
24	Swamp Sparrow	*Melospiza georgiana*
25	**Terns** (Black Tern, Caspian Tern, Common Tern, Forster's Tern)	*Chlidonias niger, Hydroprogne caspia, Sterna hirundo, Sterna forsteri*
26	Wilson's Snipe	*Gallinago delicata*
27	Yellow Warbler	*Setophaga petechia*

Table 1. Wetland bird species and groups of species (in bold) (n = 27) 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

Wetland type	Basin	Start year	End year	Trend	Lower	Upper
Coastal	**Overall**	**1995**	**2017**	**1.4**	**0.8**	**2.0**
Coastal	**Overall**	**2007**	**2017**	**2.4**	**1.1**	**3.6**
Coastal	Superior	1995	2017	–	–	–
Coastal	Superior	2007	2017	–	–	–
Coastal	Michigan	1995	2017	–	–	–
Coastal	Michigan	2007	2017	–	–	–
Coastal	Huron	1995	2017	–	–	–
Coastal	Huron	2007	2017	1.6	-2.1	5.4
Coastal	Erie	1995	2017	0.6	-0.2	1.3
Coastal	Erie	2007	2017	-1.2	-2.7	0.3
Coastal	**Ontario**	**1995**	**2017**	**1.2**	**0.2**	**2.2**
Coastal	**Ontario**	**2007**	**2017**	**4.1**	**2.1**	**6.0**
Inland	Overall	1995	2017	0.2	-0.5	0.9
Inland	Overall	2007	2017	0.3	-1.0	1.6
Inland	Superior	1995	2017	–	–	–
Inland	Superior	2007	2017	–	–	–
Inland	Michigan	1995	2017	–	–	–
Inland	Michigan	2007	2017	–	–	–
Inland	Huron	1995	2017	0.3	-1.1	1.7
Inland	Huron	2007	2017	-0.4	-3.8	3.1
Inland	Erie	1995	2017	0.3	-1.2	1.7
Inland	Erie	2007	2017	-0.3	-2.5	2.0
Inland	Ontario	1995	2017	0.0	-1.1	1.1
Inland	Ontario	2007	2017	0.8	-1.3	2.9

Table 2. Long-term (1995-2017) and 10-year (2007-2017) 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 (%/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 1000 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

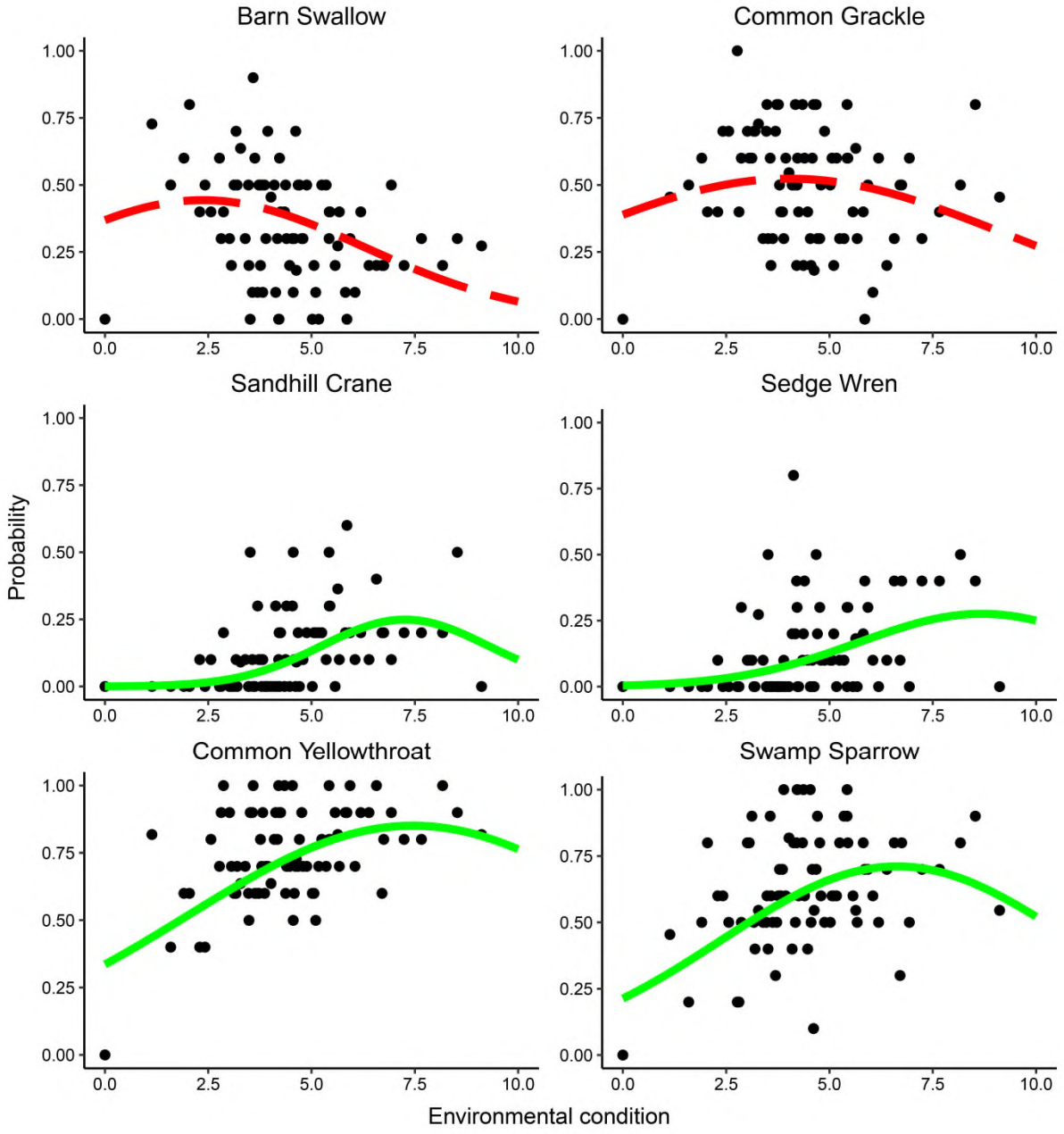


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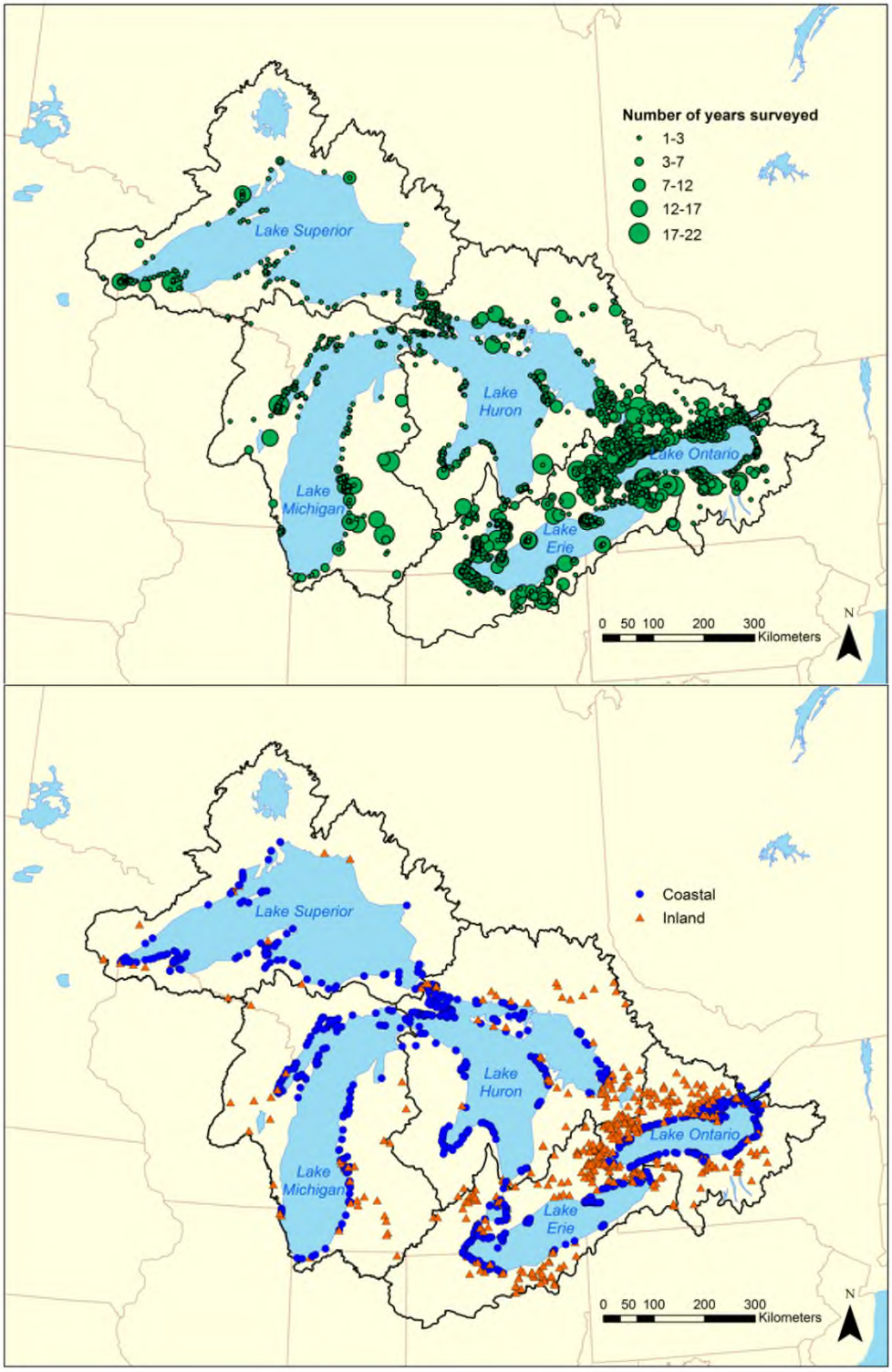


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Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program

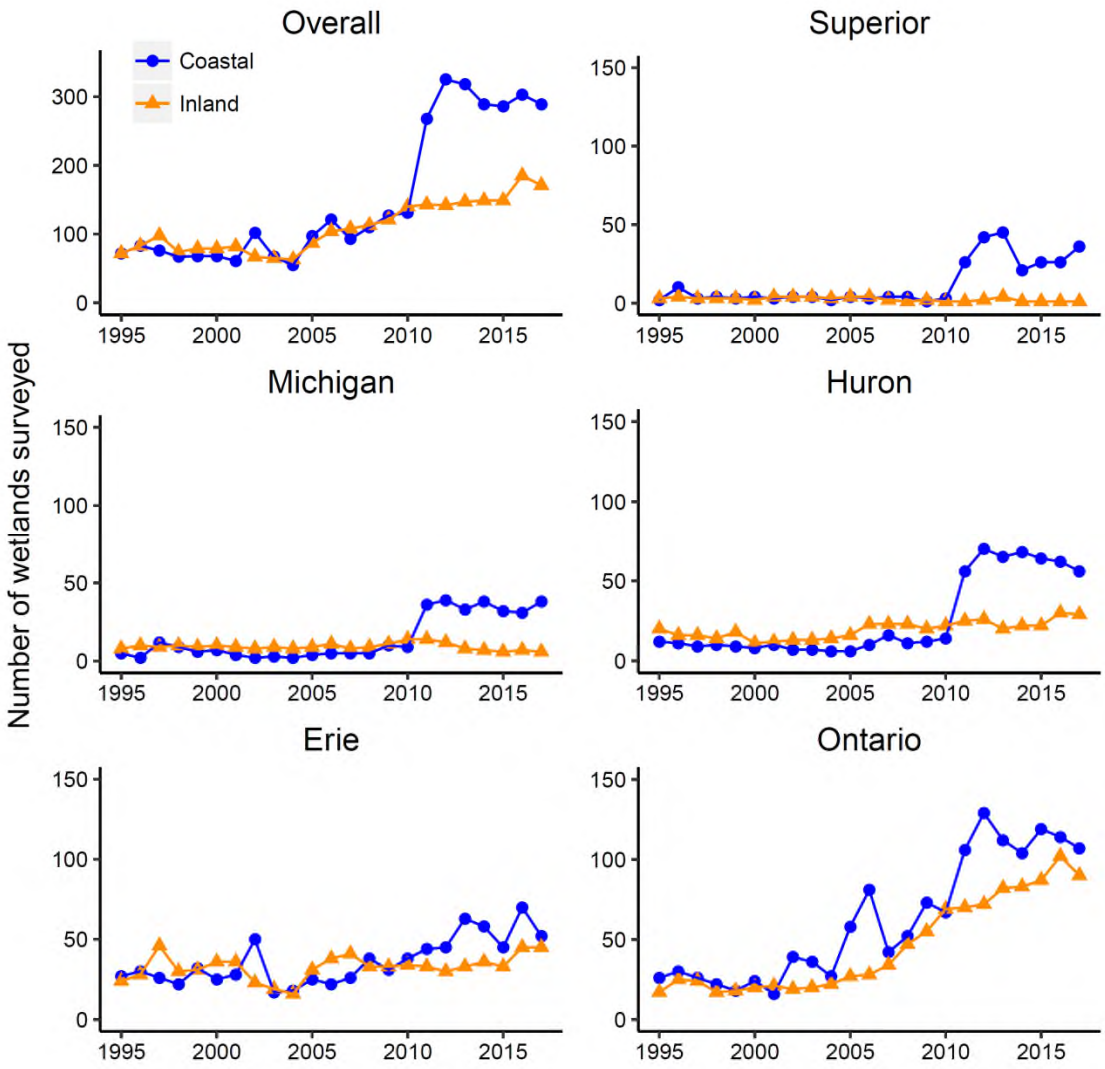


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Source: Great Lakes Marsh Monitoring Program, Great Lakes Environmental Indicator project, Great Lakes Coastal Wetland Monitoring Program

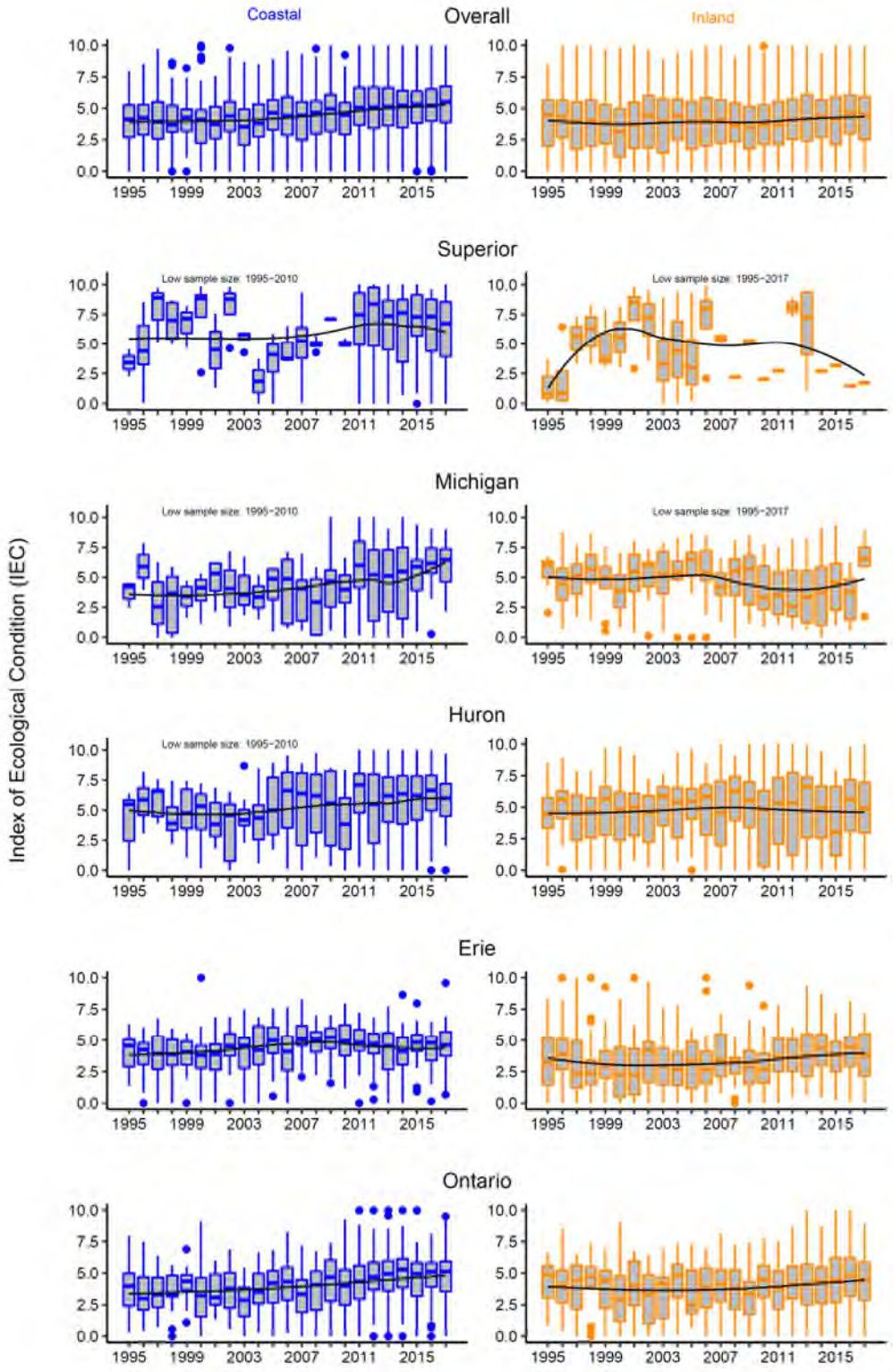


Figure 4. Temporal trends in the index of ecological condition (IEC) based on bird community data from 1995-2017 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal (blue) and inland (orange) wetlands. Shown are box-and-whisker plots of wetland-level means in 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



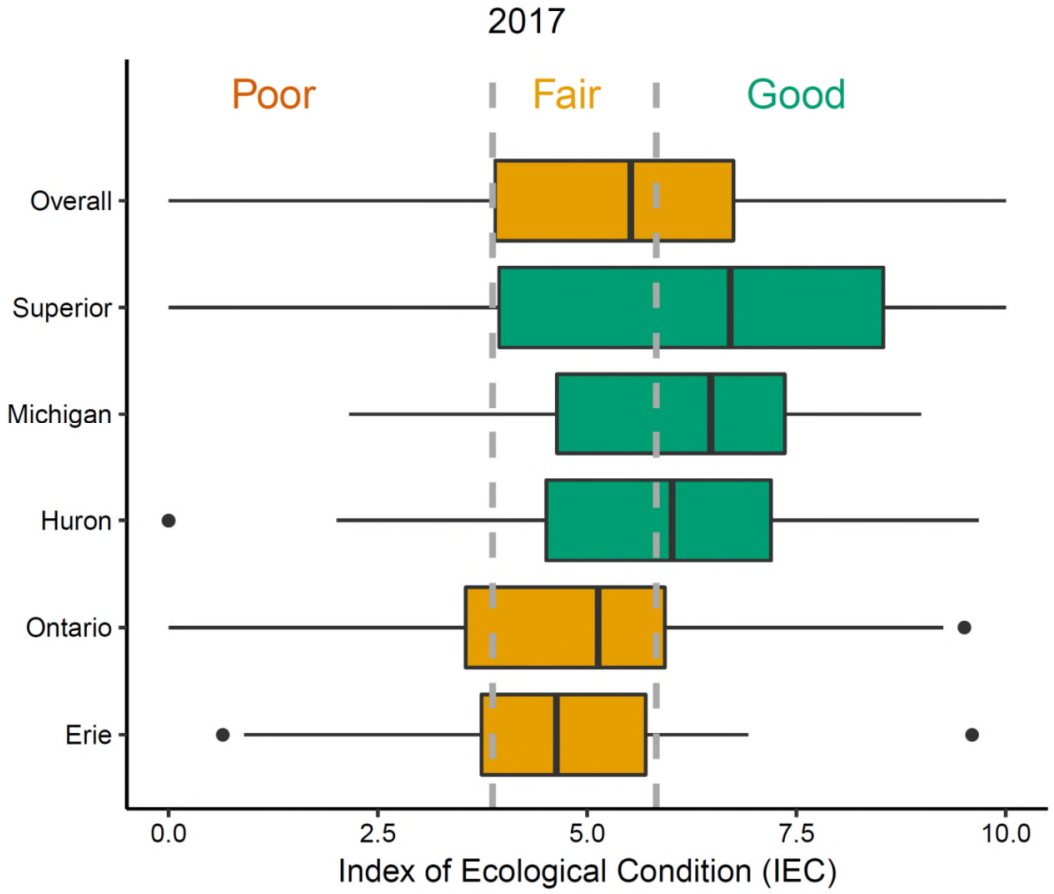


Figure 5. Distribution of the index of ecological condition (IEC) based on bird community data in 2017 throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal wetlands. Shown are box-and-whisker plots of wetland-level means. Whiskers represent 1.5 times the inter-quartile range; dots are outliers. Vertical dashed grey lines show defining values for Poor, Fair, and Good status.
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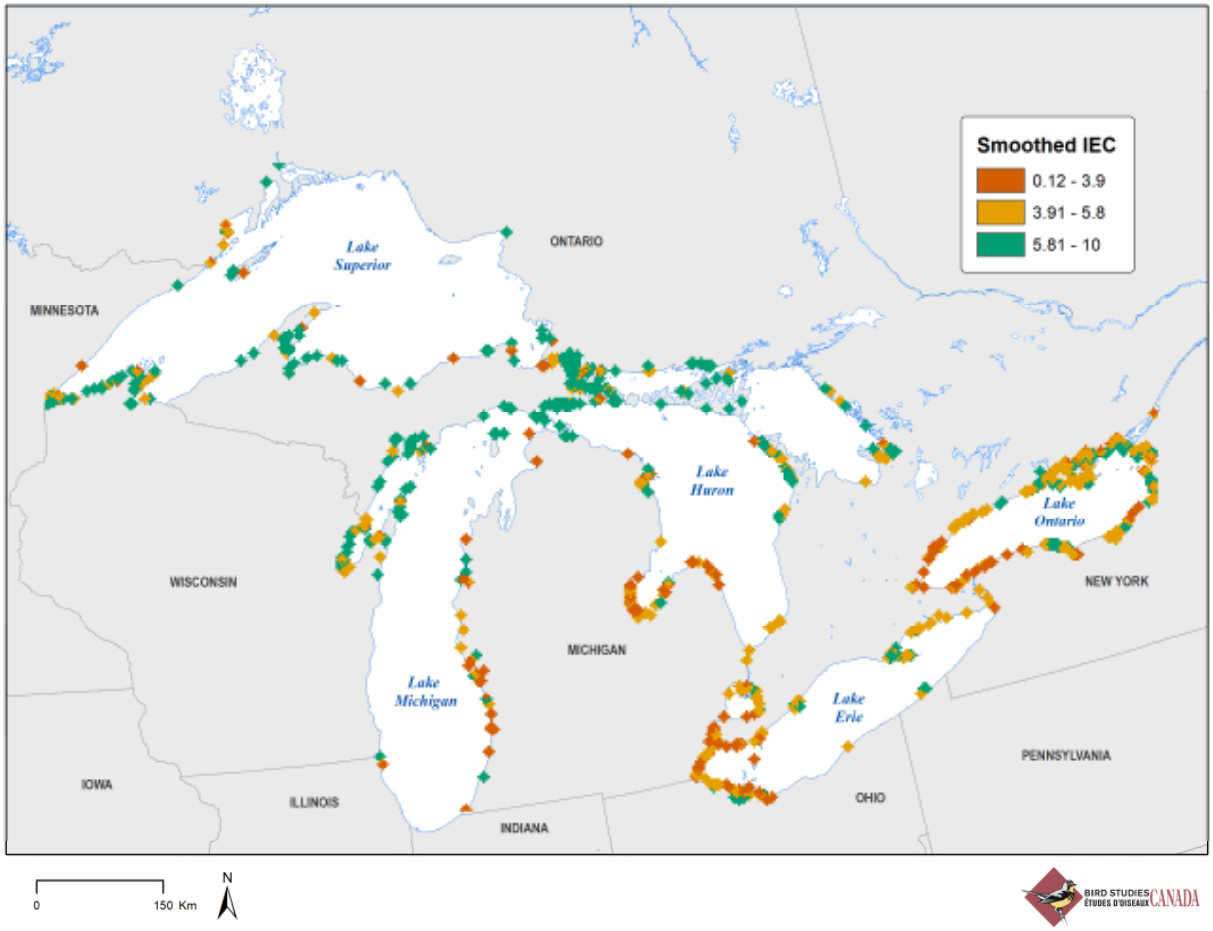


Figure 6. Index of ecological condition (IEC) throughout coastal wetlands of the Great Lakes based on bird community data from the past 5 years (2013-2017). Shown are smoothed wetland-level mean IEC values, averaged across years for wetlands that were sampled in multiple years. To smooth the data, the point-to-raster tool in ArcGIS was used to calculate mean values within 3000 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

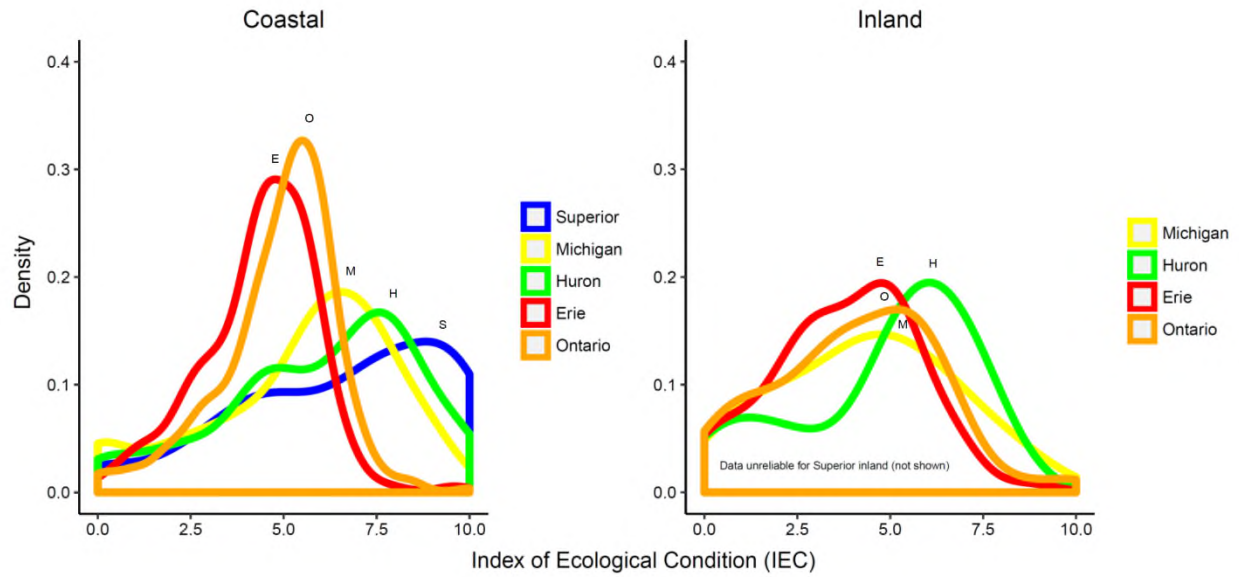


Figure 7. Density plots of the index of ecological condition (IEC) based on bird community data from the past 5 years (2013-2017) throughout the entire Great Lakes basin (overall) and each individual lake basin for coastal and inland wetlands. Shown are plots based on wetland-level mean IEC values, averaged 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 Plants**

Overall Assessment
Status: Fair
Trends:
10-Year Trend (2011-2017)*: Unchanging
Long-term Trend: Undetermined
Rationale: Based on scores of three plant community measures from the Coastal Wetland Monitoring inventory between 2011 and 2014 (Tables 1 and 2, Figures 1 and 2), status of the coastal wetland plant community in the Great Lakes is Fair. The three measures tell a similar story, although IBI scores (Albert 2008) are consistently higher than Mean C (Herman et al. 2001) and weighted Mean C (wC) (Bourdaghs et al. 2006) scores. On average, wetlands in Lakes Huron, Michigan, and Superior generally harbour Fair or Good wetland plant communities with some very high-quality sites and lower numbers of Poor sites. Wetlands in Lakes Erie and Ontario tend to be of more uniformly low quality, with only scattered high quality sites. Of the wetlands sampled in 2011-2015, 190 wetlands were resampled in 2016 and 2017. Overall there was a slight reduction in Mean C score, with higher scores for 65 wetlands, no change for 21 wetlands, and lower scores for 104 wetlands, with ratios of increase or decrease varying by Great Lake, as will be discussed below. * Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.**

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 (2011-2017)*: Unchanging
Long-term Trend: Undetermined
Rationale: Lakewide average values for the three plant community measures all fall in the 'Good' category. Over half of the surveyed wetland sites in Lake Superior have overall site scores categorized as Good. While there are low quality sites adjacent to urban centers and in other scattered locations, most wetlands in Lake Superior have good quality plant communities. The highest quality wetlands in Lake Superior tend to be barrier-protected poor fens (average mean C and wC >5), since many species in these wetlands are habitat specialists with high conservatism values. Since the last report, trends are based on 35 resampled wetlands, of which Mean C score dropped for 17 wetlands, increased for 14 wetlands, and remained the same for 4 wetlands. The overall drop was only 0.13 Mean C.
* Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Michigan
Status: Fair
10-Year Trend (2011-2017): Unchanging
Long-term Trend: Undetermined
Rationale: Among all Great Lakes, Lake Michigan has the widest distribution of sites across the gradients. On average, most wetland plant communities are considered having Fair condition, with the higher quality wetlands generally occurring in the northern part of the lake. Riverine wetlands have lower average scores, especially those in the south with extreme urban and agricultural nutrient enrichment, while open lacustrine and barrier wetlands farther north have higher scores associated with surrounding forest cover. Many wetlands in the Green Bay, WI region have experienced 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 2017 have resulted in erosion of wetland vegetation from the more exposed open lacustrine marshes. Since the last report, trends are based on 33 resampled wetlands, of which Mean C score dropped for 19 wetlands, increased for 11 wetlands, and remained the same for 3 wetlands. The overall drop was only 0.12 Mean C.* Note: A 7-year trend will be used for

the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend (2011-2017)*: Unchanging
Long-term Trend: Undetermined
Rationale: The overall status of Lake Huron wetlands is Fair based on Mean C and wC scores, and Good based on IBI scores. Wetlands in Lake Huron occur across a wide gradient in plant community condition, with some very poor and high quality sites and many good sites. Sites in the northern and eastern portion of Lake Huron tend to be of higher quality for barrier (protected), lacustrine, and riverine wetlands that reflect surrounding forest cover and management. 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*, particularly in the Saginaw Bay region. This long-term change was documented by observed changes between surveys conducted in the mid-1990s and those conducted between 2011-2015. 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. 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. 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. Overall, wetland quality in this lake is considered deteriorating. Since the last report, trends are based on 75 resampled wetlands, of which the IBI score dropped for 40 wetlands, increased for 25 wetlands, and remained the same for 10 wetlands. The overall IBI score drop was only 0.18. * Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend (2011-2017): Unchanging
Long-term Trend: Undetermined
Rationale: Wetlands of Lake Erie have plant communities of generally Poor status. Some high quality sites exist at Presque Isle, Pennsylvania and at 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 or lacustrine wetlands. Mean C scores are consistently higher than Weighted Mean C, indicating widespread dominance by species with low Conservatism values, including cattails and invasive species. A slight rise in IBI score may reflect response to restoration efforts at sites on Lake Erie and the Detroit River. Since the last report, trends are based on 20 resampled wetlands, of which the IBI score dropped for 8 wetlands, increased for 10 wetlands, and remained the same for 2 wetlands. The overall IBI score rise was only 0.17.* Note: A 7-year trend will be used for the shorter-term trend determination and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend (2011-2017): Unchanging
Long-term Trend: Undetermined
Rationale: The overall status of Lake Ontario's coastal wetlands is Fair. There are very few high quality coastal wetlands in Lake Ontario, whereas there are many wetlands of moderately low quality. Riverine wetlands have lower average quality than barrier or lacustrine wetlands. Substantially lower scores for Weighted Mean C compared to Mean C indicate Lake Ontario wetlands tend to be dominated by species with low Conservatism scores, including cattails and invasive species. There is a strong east to west gradient in condition, due largely from high levels of urbanization in the western portion of the basin. Since the last report, trends are based on 48 resampled wetlands, of which the IBI score dropped for 31 wetlands, increased for 13 wetlands, and remained the same for 4 wetlands. The overall IBI score drop was only 0.30.* Note: A 7-year trend will be used for the shorter-term trend determination

and serve as an estimate for the 10-year trend as only 7 years of data have been collected.

Status Assessment Definitions
Lake Assessment Scale for *C* and *wC*
Good: 5.0 and above
Fair: 3.0 - 4.9
Poor: 0.0 - 2.9
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Lake Assessment Scale for IBI
Good: 5.0 and above
Fair: 3.0 - 4.9
Poor: 0.0 - 2.9
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: Mean C, wC, and/or IBI scores per lake have increased by at least 0.5 unit.
Unchanging: Mean C, wC, and/or IBI scores per lake have experienced low amounts of natural variation within a range of -0.5 to + 0.5 units.
Deteriorating: Mean C, wC, and/or 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 three wetland plant quality measures (*C*, *wC*, 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 CWM project 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 basinwide Great Lakes Coastal Wetland Monitoring program (CWM) 2011-2015, 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.

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. Analysis of the statistical features and performance of the IBI are underway as a part of the CWM program.

Ecological Condition

Across the entire Great Lakes basin, the state of the wetland plant community is quite variable, ranging from Good to Poor 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 2014-2015, 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 5 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 reed (*Phragmites australis*), reed canary grass (*Phalaris arundinacea*), purple loosestrife (*Lythrum salicaria*), curly pondweed (*Potamogeton crispus*), Eurasian milfoil (*Myriophyllum spicatum*), frog bit (*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 reed 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 reed should be monitored.

One of the disturbing trends is the expansion of frog-bit, 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. 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 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).

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).

With climate change, some modelers have projected that water-level fluctuations will be more extreme, with higher highs and lower lows. 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 (*Phragmite*s, 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 frog-bit, 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. This expansion will likely continue to all of the remaining Great Lakes. In addition, our sampling has shown water chestnut 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 props.

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 Limitations

The characteristic presence and abundance of coastal wetland native plants has not been adequately documented across the Great Lakes basin, but most regional wetland types can be adequately described on the basis of existing studies. 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 Coastal Wetland Monitoring program was funded by the Great Lakes Restoration Initiative 2011-2015 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 2016-2020. 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 2011-2014 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 2014 on three broad hydrogeomorphic wetland types: barrier, lacustrine, and coastal wetlands that were characterized for each separate Great Lake. In subsequent analyses these types will be further divided into recognized 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 three measures of wetland plant quality including: 1. Mean Coefficient of Conservatism (C); 2.Weighted Mean Coefficient of Conservatism (wC); and 3. Vegetation Index of Biological Integrity (IBI). The Mean C approach is preferred by many, because it provides a nice, neat, easily computed number however, it provides little understanding of the overall diversity of the wetlands within the lake. In both Lake Michigan and Lake Huron, there is an extreme environmental gradient [climate and hydro-geomorphology] that are reflected in land use and vegetation response, and a single FAIR designation ignores that gradient. One number or condition cannot reflect these lakes. The IBI better demonstrates the breadth of types and conditions. However, for the purposes of this sub-indicator report, if calculation results fall into different assessment categories then the conservative score is used.

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 Lake-wide surveys of coastal wetland 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. 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. Their dense mats 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 then 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 the 2011-2015 (Uzarski et al. 2016) 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.

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List of Tables

Table 1. Lakewide means and 95% confidence intervals for three measures of Great Lakes coastal wetland plant community condition observed 2011-2014. Some sites with missing vegetation zones were not used in calculations for the vegetation IBI, resulting in slightly lower sample size. Mean C and wC scores are based on a maximum score of 10, while Veg IBI scores are based on a maximum score of 5. Vegetation IBI scores must be doubled to be equivalent of Mean C and wC scores.
Source: GLRI Coastal Wetland Monitoring Program, analysis by Nicholas Danz

Table 2. Condition class categories based on sub-indicator definitions for three measures of coastal wetland plant communities.
Source: GLRI Coastal Wetland Monitoring Program, analysis by Nicholas Danz

Table 3. Lakewide and wetland-type means and 95% confidence intervals for three measures of Great Lakes coastal wetland plant community condition observed 2011-2014. Some sites with missing vegetation zones were not used in calculations for the vegetation IBI, resulting in slightly lower sample size. Mean C and wC have a maximum score of 10, while Vegetation IBI has a maximum score of 5 and must be doubled to be the equivalent of Mean C and wC.
Source: GLRI Coastal Wetland Monitoring Program, analysis by Nicholas Danz

List of Figures

Figure 1. Frequency histogram of overall site Mean C (blue) and Weighted Mean C (red) values for 451 Great Lakes coastal wetland sites surveyed between 2011 and 2014.
Source: GLRI Coastal Wetland Monitoring Program, analysis by Nicholas Danz

Figure 2. Frequency histogram of overall site Vegetation IBI values for 415 Great Lakes coastal wetland sites surveyed between 2011 and 2014.
Source: GLRI Coastal Wetland Monitoring Program, analysis by Nicholas Danz

Last Updated

State of the Great Lakes 2019 Technical Report

		Mean C and wC					Veg IBI		Veg IBI x2
Lake	*n*	Mean C	95% CI	wC	95% CI	n	Veg IBI	95% CI	
Lake Erie	52	2.53	0.19	2.22	0.25	50	1.6	0.15	3.2
Lake Huron	162	4.33	0.17	4.21	0.19	140	3.0	0.15	6.0
Lake Michigan	65	3.57	0.26	3.46	0.30	61	2.9	0.20	5.8
Lake Ontario	107	3.02	0.13	2.53	0.16	104	1.9	0.10	3.8
Lake Superior	65	5.18	0.30	5.19	0.34	60	3.7	0.23	7.4

Table 1. Lakewide means and 95% confidence intervals for three measures of Great Lakes coastal wetland plant community condition observed 2011-2014. Some sites with missing vegetation zones were not used in calculations for the vegetation IBI, resulting in slightly lower sample size. Mean C and wC scores are based on a maximum score of 10, while Veg IBI scores are based on a maximum score of 5. Vegetation IBI scores must be doubled to be equivalent of Mean C and wC scores.
Source: Great Lakes Coastal Wetland Monitoring Program

	Measures			
Lake	Mean C	Weighted Mean C	Veg IBI	Overall
Lake Erie	Poor	Poor	Fair	Poor
Lake Huron	Fair	Fair	Good	Fair
Lake Michigan	Fair	Fair	Good	Fair
Lake Ontario	Fair	Poor	Fair	Fair
Lake Superior	Good	Good	Good	Good

Table 2. Condition class categories based on sub-indicator definitions for three measures of coastal wetland plant communities.
Source: Great Lakes Coastal Wetland Monitoring Program

		Mean C and wC					Veg IBI		
Lake	Hydrogeomorphic Type	*n*	Mean C	95% CI	wC	95% CI	*n*	IBI	95% CI
Erie	Barrier (protected)	10	2.61	0.25	2.34	0.44	5	1.78	0.49
	Lacustrine (coastal)	22	2.66	0.33	2.40	0.41	17	1.61	0.22
	Riverine	31	2.40	0.30	2.06	0.39	28	1.45	0.21
Huron	Barrier (protected)	16	4.60	0.70	4.58	0.76	12	3.46	0.57
	Lacustrine (coastal)	113	4.23	0.23	4.07	0.26	82	2.95	0.18
	Riverine	62	4.46	0.25	4.36	0.30	46	3.03	0.27
Michigan	Barrier (protected)	11	3.75	0.65	3.69	0.80	10	3.32	0.68
	Lacustrine (coastal)	37	3.74	0.39	3.67	0.42	30	2.88	0.27
	Riverine	26	3.25	0.39	3.07	0.47	21	2.67	0.28
Ontario	Barrier (protected)	27	3.39	0.41	2.95	0.48	23	1.99	0.27
	Lacustrine (coastal)	28	3.04	0.20	2.49	0.24	24	1.88	0.18
	Riverine	68	2.87	0.15	2.38	0.20	57	1.81	0.13
Superior	Barrier (protected)	17	6.29	0.55	6.48	0.55	15	4.35	0.29
	Lacustrine (coastal)	9	5.12	0.51	4.99	0.78	7	3.63	0.56
	Riverine	42	4.75	0.33	4.71	0.39	38	3.48	0.29

Table 3. Lakewide and wetland-type means and 95% confidence intervals for three measures of Great Lakes coastal wetland plant community condition observed 2011-2014. Some sites with missing vegetation zones were not used in calculations for the vegetation IBI, resulting in slightly lower sample size. Mean C and wC have a maximum score of 10, while Vegetation IBI has a maximum score of 5 and the value noted above must be doubled to be the equivalent of Mean C and wC.
Source: Great Lakes Coastal Wetland Monitoring Program

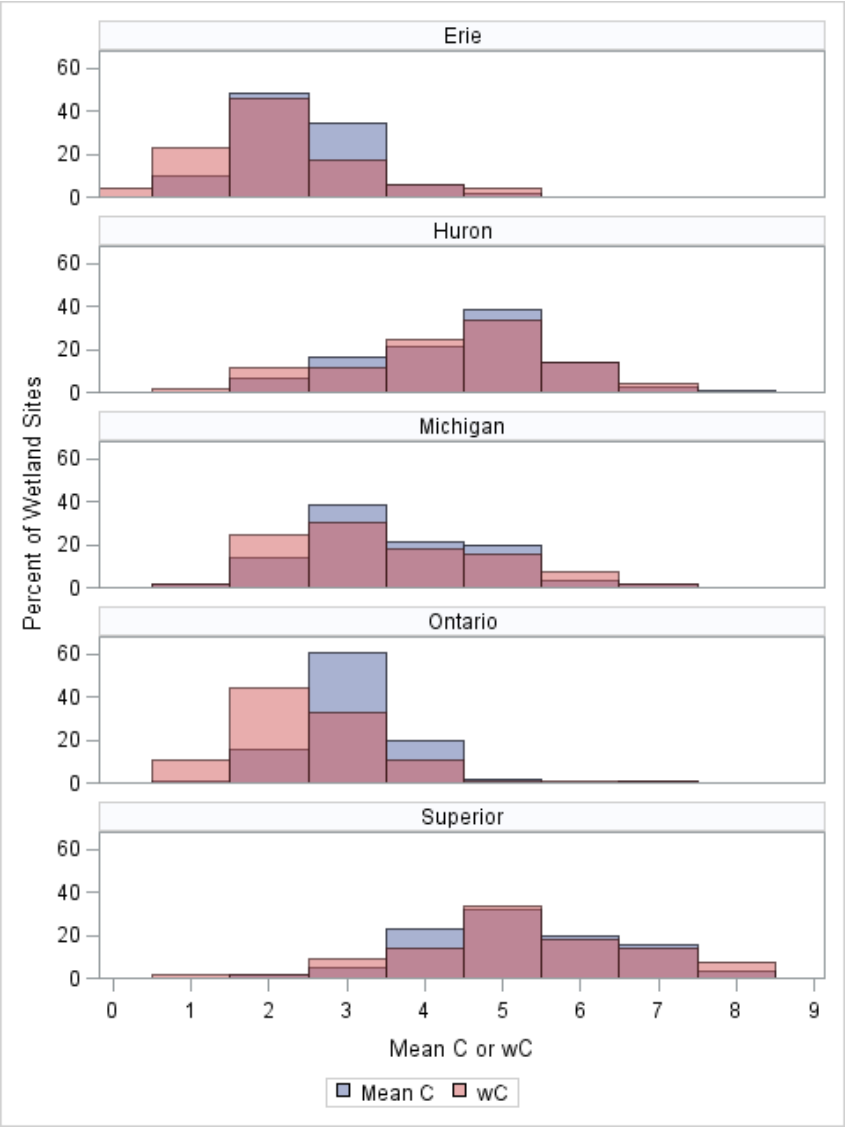


Figure 1. Frequency histogram of overall site Mean C (blue) and Weighted Mean C (red) values for 451 Great Lakes coastal wetland sites surveyed between 2011 and 2014. Lake Assessment Scale for Mean C and wC are Good: 5.0 and above; Fair: 3.0 - 4.9 and Poor: 0.0 - 2.9. Please note the difference in scale between Figure 1 and 2 on the x-axis.
Source: Great Lakes Coastal Wetland Monitoring Program

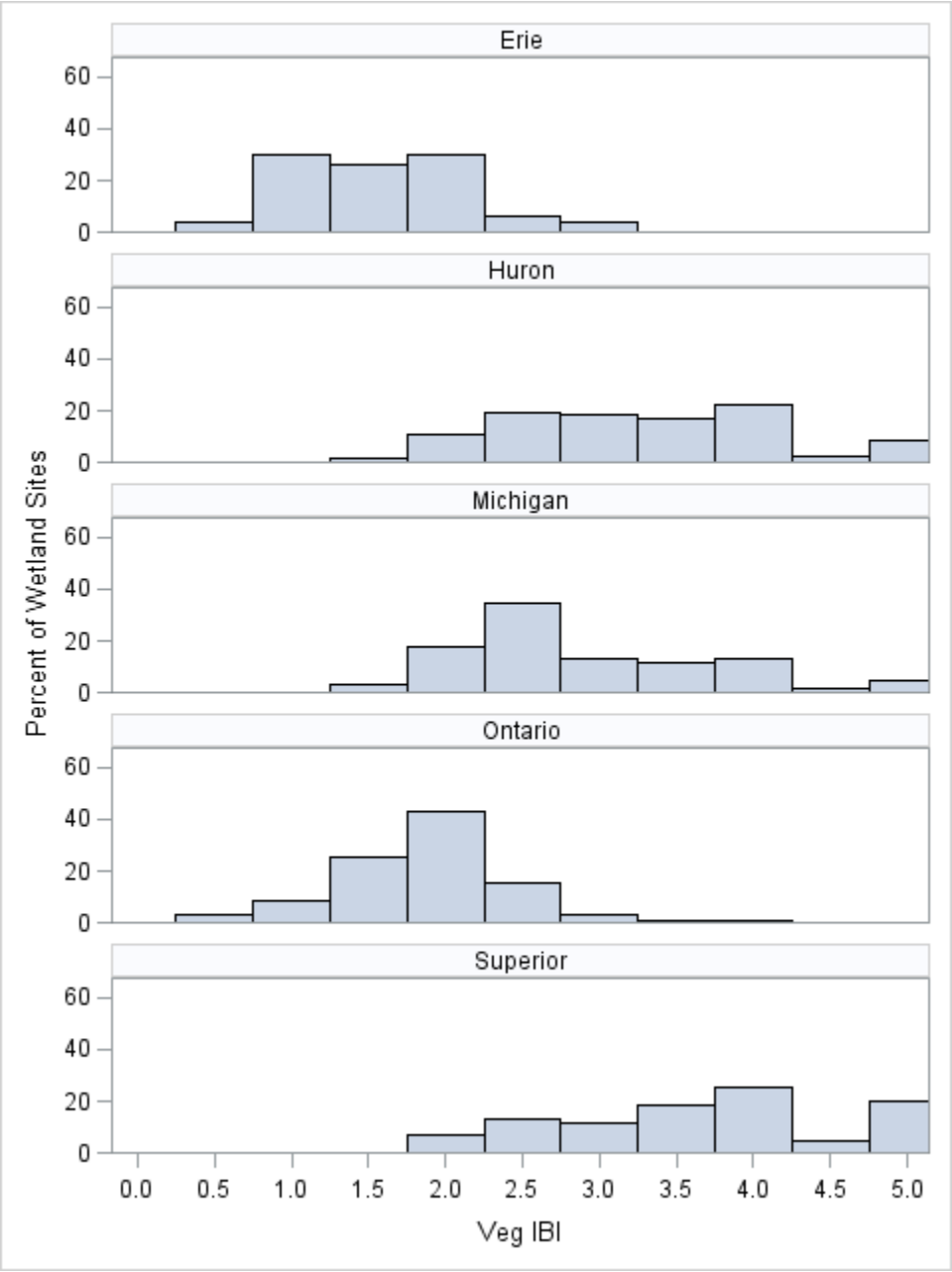


Figure 2. Frequency histogram of overall site Vegetation IBI values for 415 Great Lakes coastal wetland sites surveyed between 2011 and 2014. Lake Assessment Scale for IBI are Good: 5.0 and above; Fair: 3.0 - 4.9; and Poor: 0.0 - 2.9. Please note the difference in scale between Figure 1 and 2 on the x-axis.
Source: Great Lakes Coastal Wetland Monitoring Program

Sub-Indicator: Coastal Wetlands: Extent and Composition

Overall Assessment
Status: Undetermined
Trends:
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Mapping and estimation of the areal coverage of Great Lakes coastal wetlands was done in 2004. An update is underway but has not yet been completed. Landscape measures have been challenging due to data gaps and because coastal wetlands are extremely dynamic systems. Coastal wetlands migrate, disappear, and appear with changing water levels that can be due to are not necessarily related to anthropogenic disturbance making it difficult to determine the range of coastal wetlands that is acceptable. Measuring habitat gains is also very challenging with varying water levels. Because there has not been an update to the estimation of areal extent in almost 15 years, the status and trend are Undetermined.

Lake-by-Lake Assessment
Lake-by-lake assessments are not available for the same reason the basin-wide assessments are not available.

Status Assessment Definitions
Good: Maintain areal extent of wetlands within 1 %; less than 1% increase in degraded wetland (e.g. phragmites).
Fair: Maintain areal extent of wetlands within 5 %; less than 5% increase in degraded wetland (e.g. phragmites).
Poor: Maintain areal extent of wetlands within 10 %; less than 10% increase in degraded wetland (e.g. phragmites).
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions
Improving: Increasing areal extent of wetlands and improvement in wetland restoration (decrease in invasive plants and wetland degradation).
Unchanging: No change in areal extent of wetlands and no change in wetland quality.
Deteriorating: Decreasing areal extent of wetlands and increase in wetland degradation.
Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets
No net loss of coastal wetland area due to human actions and, in the future, a gain in coastal wetlands due to restoration activities, recognizing that a reference year needs to be selected and accurate inventory integrated into the analysis.

Sub-Indicator Purpose
- To assess the periodic changes in area (gains and losses) of coastal wetland types, taking into account natural lake level variations. Coastal wetlands provide critical breeding and migratory habitat for wildlife such as birds, mammals, reptiles, and amphibians. These habitats are also critical spawning and nursery areas for many fish species of ecologic and economic importance.

Ecosystem Objective
Maintain total areal extent of Great Lakes coastal wetlands, ensuring adequate representation of coastal wetland types across their historical range. 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.”

Measure

This sub-indicator will measure areal extent of coastal wetlands by hydrogeomorphic type as a range (e.g., dry year/low water level area versus wet year/ high water level area), for a specific time period based on data sources/imagery available. An existing baseline map circa 2010 of the binational coastal area was produced from multi-season (spring, summer and fall) satellite optical and L-band radar data with a minimum mapping unit of 0.2 ha (Bourgeau-Chavez *et al*. 2015) and overall accuracy of 94%. Change detection methods may be used to update this map on a 5 year incremental basis. This map delineates ecosystem type (i.e. emergent, shrub and forested wetland) as well wetland monocultures (*Typha, Phragmites, Schoenoplectus*) and peatland types (fens and bogs). In addition potential wetland stressors: upland and land use classes are mapped. Both the change in wetland extent and type may be monitored by updating this map. The bands found most important for wetland mapping were the thermal, NIR and L-band SAR and should be integrated into any map update to maintain the integrity and level of accuracy. Optical data alone may be used but woody wetlands in particular are not mapped as accurately with optical data alone (e.g. forested wetlands, scrub shrub, bogs, fens).

Current optical data from Landsat-8 and Sentinel-2 and L-band SAR data from Japan's PALSAR-2 and Argentina's SaoCom (to be launched in early 2016) will allow for an approximate 5 year update (http://earth.esa.int/workshops/cmasar_2003/papers/E27gira.pdf).

NASA's NISAR is planned for a 2020 launch (http://nisar.jpl.nasa.gov/). Together these sensors will provide continuity into the future. Note that PALSAR-2 is commercial and has a cost associated with it, while all other sources are free. NISAR L-band sensor will collect data every 2 weeks and will thus allow for monitoring of extent of inundation and hydroperiod.

Additional efforts may also help to support the assessment of the Coastal Wetlands: Extent and Composition sub-indicator in future reporting cycles including an updated Southern Ontario Land Resources Information System (SOLRIS) which will provide a more detailed inventory of all Great Lakes coastal ecosystems (Great Lakes Shoreline Ecosystem (GLSE) inventory). Through this project, the Ontario Ministry of Natural Resources and Forestry is classifying and mapping a 2 kilometer band along all Canadian Great Lakes to provide an inventory capturing both landscape-scale information for broader analyses and site-specific information describing coastal habitats. This is a complementary effort to Canada's work to assess the waters of the Great Lakes through the nearshore framework (discussed in the Lakewide Management Annex) and will inform a Canadian baseline survey of Great Lakes habitats in order to establish a bench-mark for making progress towards a target of net habitat gain. Mapping and classification methodologies will be made public to help practitioners focus conservation efforts.

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 nutrient and sediment loading, fragmentation, invasive species, shoreline alteration, and water level control (Albert and Simonson, 2004; Ingram and Potter, 2004).

An existing baseline map circa 2004 of the binational coastal wetland occurrence and general boundaries was produced from available data sources on wetland occurrence including the USFWS National Wetland Inventory, Michigan National Wetland Inventory, Ohio Wetlands inventory, Wisconsin DNR Wetlands Inventory, and best professional judgement (Figures 1, 2, and 3). There has not yet been a complete update to this map, so current areal extent and composition of coastal wetlands across the entire Great Lakes basin cannot be reported.

New data sets have been produced that allow the circa 2004 data set to be reexamined and refined, which will ultimately allow determination of a more current status and trend over time. For example, a multi-season (spring, summer and fall) satellite optical and L-band radar data with a minimum mapping unit of 0.2 ha (Bourgeau-Chavez *et al*. 2015) for wetland plant communities and other landuse classes was produced (Figure 4). This map delineates ecosystem type (i.e. emergent, shrub and forested wetland) as well wetland monocultures (*Typha, Phragmites, Schoenoplectus*) and peatland types (fens and bogs). In addition, upland and landuse classes, potential wetland stressors, are mapped. An overall accuracy of 94% was documented by this effort when the map was compared to vegetation types identified in field studies between 2008 and 2011. The bands found most important for wetland mapping were the thermal, NIR and L-band SAR and should be integrated into any map update to maintain the integrity and level

of accuracy. Optical data alone may be used but woody wetlands in particular are not mapped as accurately with optical data alone (e.g. forested wetlands, scrub shrub, bogs, fens). This map could be updated on an incremental basis, such as a five year cycle, using change detection methods.
Updating maps in a standardized way across the whole Great Lakes Basin is now planned, and there are efforts underway to use the 2008-2011 field study data set and update the circa 2004 coastal wetland data set in select geographic areas (e.g., Saginaw Bay to Western Lake Erie Basin – US side only, state of Michigan.)
It should be noted that the assessment in the State of the Great Lakes 2011, 2009, 2007 and 2005 reports was Fair (Mixed) and Deteriorating for this sub-indicator, based on historical data, 1981-1997.

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 nearshore and coastal habitat structure
- Precipitation Amounts – change in atmospheric temperature will potentially affect the amount of precipitation and the number of extreme storms in the Great Lakes region which will, in turn, affect coastal wetlands
- Water Levels – water level change has strong influences on Great Lakes habitat and biological communities associated with Coastal Wetlands. Water levels have a major influence on un-diked coastal wetlands and are basic to any analysis of wetland change trends
- Terrestrial Invasive Species – many terrestrial invaders are found in Great Lakes coastal wetlands and can displace native vegetation as they spread

This sub-indicator links directly to the other sub-indicators in the Habitats and Species indicator, particularly the other coastal wetlands-related sub-indicators.

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			
Clarifying Notes: The Data Quality assessment given here is copied from the State of the Great Lakes 2009 data quality assessment, which was based on the State of the Great Lakes 2005 report. This was completed because the majority of this report still refers to the SOGL 2005 report.				

Data Limitations
The total wetland area should be reported by lake basin and hydrogeomorphic type. The baseline status should be considered within a historical perspective. The monitoring must be conducted on a regular and ongoing basis over an entire Great Lakes water level cycle for meaningful baseline data. For each wetland type, tabular summaries and graphics could show the areal extent of hydrogeomorphic wetland types by lake basin as they change relative to water level and over time.

This sub-indicator needs to be evaluated in terms of both wetland quality and extent. While some wetlands may decrease in both area and quality due to the lack of water level fluctuation, as on Lake Ontario, the area of other wetlands could remain within the range determined by natural water level fluctuations, but be degraded by other factors, such as sedimentation, excessive nutrients, invasive species or land use pressures. When interpreting the data, the other coastal wetland sub-indicators that evaluate wetland quality need to be considered. Measurement

should be based upon total area of inventoried coastal wetlands where known. Where areal extent is not known, efforts should be focused on collecting that baseline data. Total change can be roughly determined on a lake basin basis and for scientifically-based sampling, priority sites should be established where regular ground-truthing facilitates a statistical analysis.

An overall view of wetland health can be derived by considering the 6 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, these coastal wetland sub-indicators related to fish, amphibians, birds, and plants cannot account for habitat gains and losses of coastal wetlands. For example, landscape measures are used to determine loss, transformation and restoration of wetland types experiencing varying degrees of anthropogenic disturbance. However, landscape measures have been challenging due to data gaps and because coastal wetlands are extremely dynamic systems; they migrate, disappear, and appear with changing water levels not necessarily related to anthropogenic disturbance.

Acknowledgments
Author of 2017 update: Kevin O'Donnell, US EPA Great Lakes National Program Office

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Krista Holmes, Canadian Wildlife Service, Environment Canada
Dennis Albert, Michigan Natural Features Inventory, Michigan State University Extension

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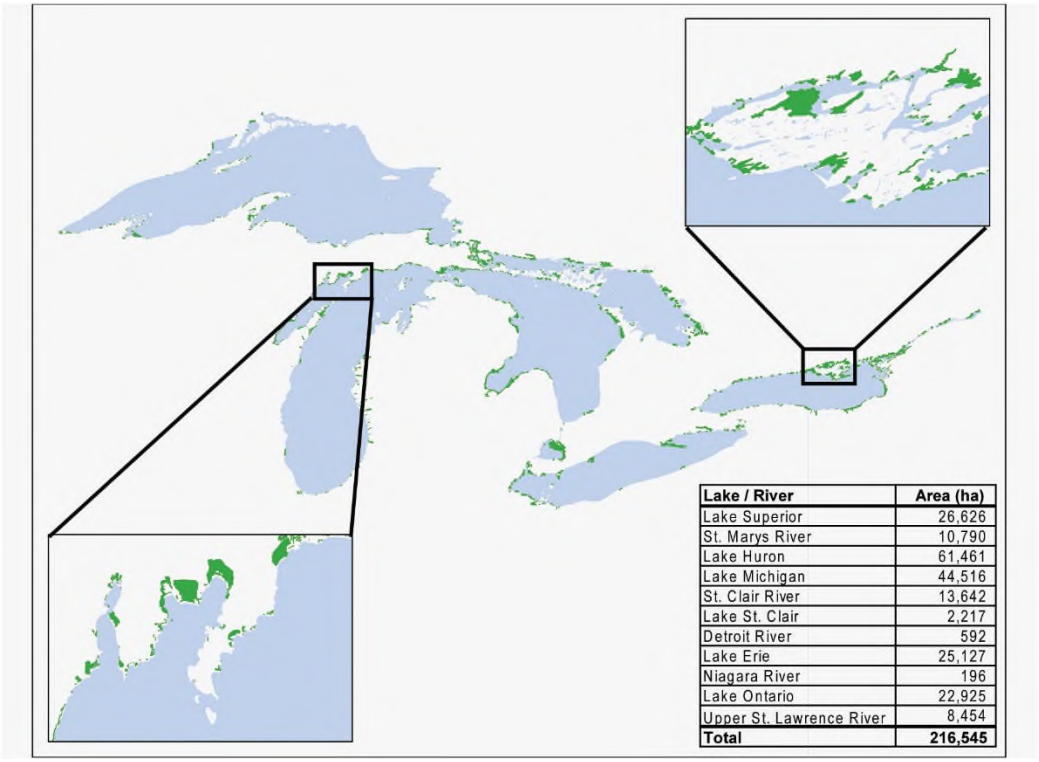
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List of Figures

Figure 1. Great Lakes coastal wetland distribution and total area by lake and river.
Source: Great Lakes Coastal Wetlands Consortium
Figure 2. Coastal wetland area by geomorphic type within lakes of the Great Lakes system.
Source: Great Lakes Coastal Wetlands Consortium
Figure 3. Coastal wetland area by geomorphic type within connecting rivers of the Great Lakes system.
Source: Great Lakes Coastal Wetlands Consortium
Figure 4: Wetlands and land use land cover (LULC) classes within a 10 km buffer of the Great Lakes coastline in both the United States and Canada.
Source: Bourgeau-Chavez, Laura. 2015

Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)



Lake / River	Area (ha)
Lake Superior	26,626
St. Marys River	10,790
Lake Huron	61,461
Lake Michigan	44,516
St. Clair River	13,642
Lake St. Clair	2,217
Detroit River	592
Lake Erie	25,127
Niagara River	196
Lake Ontario	22,925
Upper St. Lawrence River	8,454
Total	**216,545**

Figure 1. Great Lakes coastal wetland distribution and total area by lake and river.
Source: Great Lakes Coastal Wetlands Consortium, from SOGL 2007 report

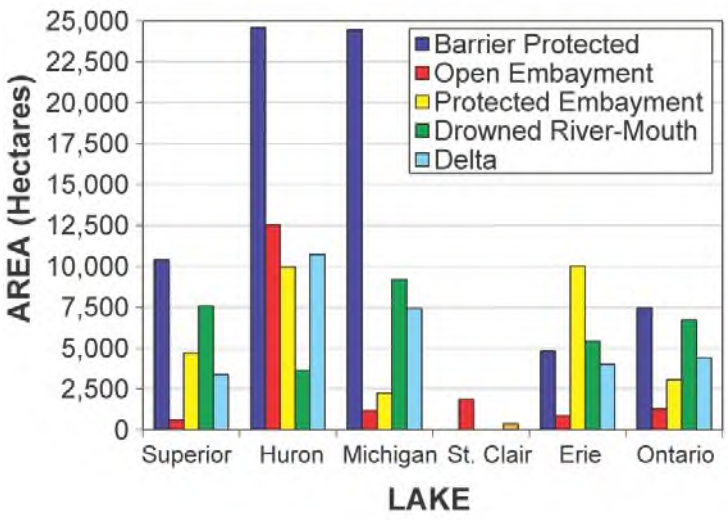


Figure 2. Coastal wetland area by geomorphic type within lakes of the Great Lakes system. Source: Great Lakes Coastal Wetlands Consortium, from SOGL 2007 report

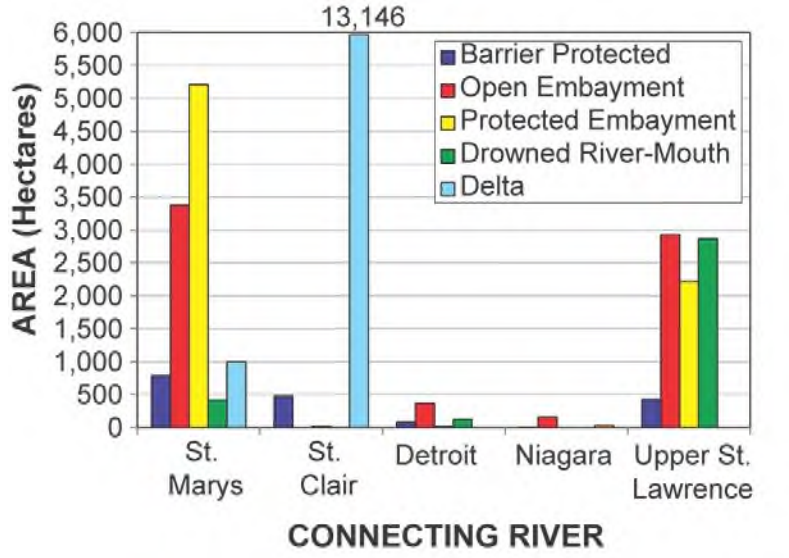


Figure 3. Coastal wetland area by geomorphic type within connecting rivers of the Great Lakes system. Source: Great Lakes Coastal Wetlands Consortium, from SOGL 2007 report

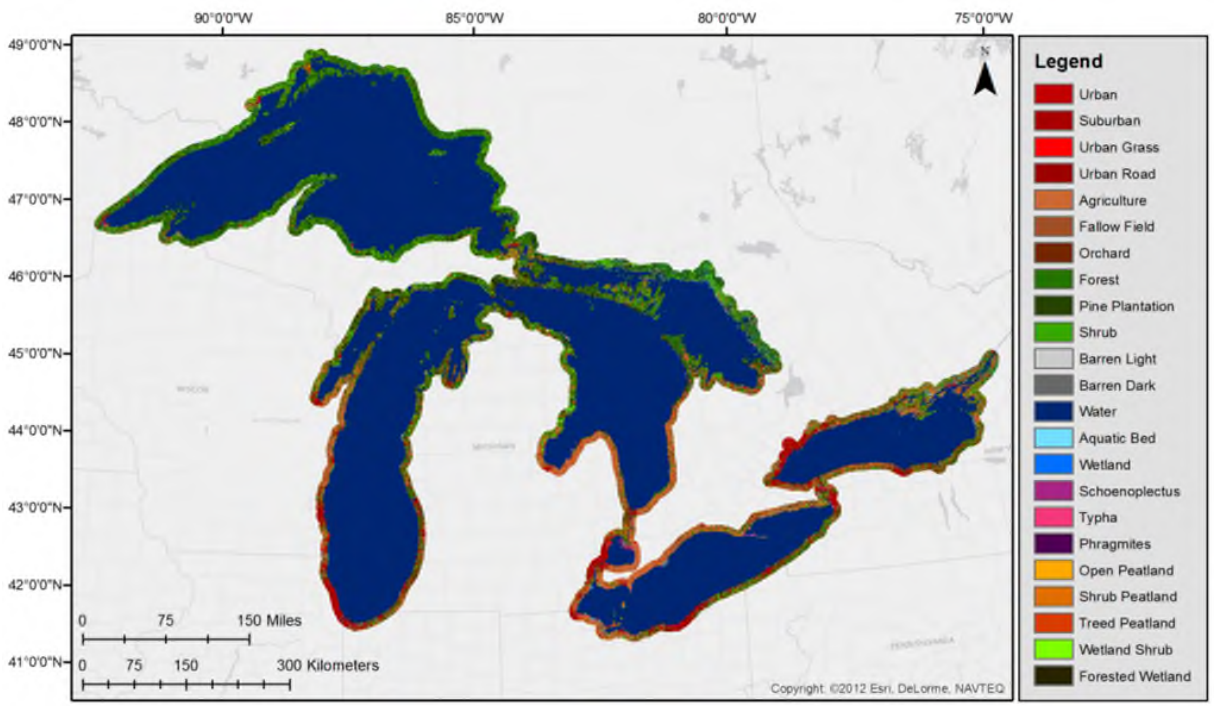


Figure 4: Wetlands and land use land cover (LULC) classes within a 10 km buffer of the Great Lakes coastline in both the United States and Canada.
Source: Bourgeau-Chavez, Laura. 2015

Sub-Indicator: Aquatic Habitat Connectivity

Overall Assessment

Status: Poor
Trends:
10-Year Trend: Improving
Long-term Trend (1950-2017): Undetermined
Rationale: Although there are many current efforts to remove dams and barriers, some new dams have been constructed since 1950, and the length of road network has increased substantially. There are currently over 250,000 barriers in the Great Lakes basin that can potentially inhibit fish movement. These range from large dams to perched culverts. As a result of barriers, less than 20% of tributary habitat in the basin is connected to the waters of the Great Lakes. The issue remains significant in comparison to the current pace of barrier removal, although new data to identify and optimize barrier removal are now available. Barrier removal is closely linked to some invasive aquatic species and the removal of many of larger dams remains stalled due to review processes, local conflict, and the presence of Sea Lamprey.

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: Improving
Long-term Trend (1950-2017): Undetermined
Rationale: Approximately 28% of tributary habitat in the Lake Superior basin is connected to the lake. Dams and barriers are identified as a high threat to migratory fishes (Lake Superior LAMP 2013) and are considered an impediment to the recovery of some fishes, such as Lake Sturgeon, Brook Trout and Walleye (Horns et al. 2003). There are several projects that have been completed or are exploring options to improve connectivity such as the Camp 43 dam on the Black Sturgeon River. A collaborative geo-database of inventoried connectivity barriers within the South Central Superior basin will be used to prioritize restoration for approximately 1,800 inventoried road-stream crossings and is an example of the efforts to address connectivity (U.S. Fish and Wildlife Service)

Lake Michigan
Status: Poor
10-Year Trend: Improving
Long-term Trend (1950-2017): Undetermined
Rationale: Approximately 14% of tributary habitat in the Lake Michigan basin is connected to the lake. Dams and barriers are ranked as a high threat to migratory fishes (Pearsall et. al 2012a). Several dam removal and mitigation projects have been initiated through the Great Lakes Restoration Initiative (e.g. Boardman River dam removal projects will connect over 250 km of stream habitat back to Lake Michigan - the dam closest to the river mouth will be modified to allow for fish passage while blocking access for Sea Lamprey).

Lake Huron (including tributaries of the St. Marys River)
Status: Poor
10-Year Trend: Improving
Long-term Trend (1950-2017): Undetermined
Rationale: Approximately 18% of tributary habitat in the Lake Huron basin is connected to the lake. Dams are ranked as a high threat to migratory fishes (Franks Taylor et al. 2010). Aquatic habitat connectivity varies in the basin. Franks Taylor et al. (2010) identified that Eastern Georgian Bay has sufficient access to spawning habitat to maintain fish populations while in Saginaw Bay access to spawning habitat is severely limiting fish populations. Construction of a rocky ramp in the Shiawassee River provides an example of a mitigation project that has improved connectivity for some fishes but has not fully restored conditions (Stoller et. al 2016).

Lake Erie (including tributaries of the St. Clair-Detroit River watershed)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1950-2017): Undetermined
Rationale: Approximately 26% of tributary habitat in the Lake Erie basin is connected to the lake. Dams and barriers are ranked as a medium threat to migratory fishes (Pearsall et. al 2012b). Several dam removal and mitigation projects have been initiated in the last few years through the Great Lakes Restoration Initiative (e.g. Ballville Dam on the Sandusky River will open up 35 km of river habitat for walleye).

Lake Ontario (including tributaries of the Niagara River and International section of the St. Lawrence River)
Status: Poor
10-Year Trend: Improving
Long-term Trend (1950-2017): Undetermined
Rationale:
Approximately 3% of tributary habitat in the Lake Ontario basin is connected to the lake. The Lake Ontario Biodiversity Conservation Strategy identified dams and barriers as high threat to the health of the lake (Lake Ontario Biodiversity Conservation Strategy Working Group, 2009). In addition to dams on Lake Ontario tributaries, the Moses -Saunders Power Dam on the St. Lawrence River impacts habitat connectivity, particularly for the migration of the American Eel (MacGregor et. al 2013). The Eel Passage Research Center was established in 2013 to address this issue. Several dam mitigation projects have been initiated including dam removal in the Duffins Creek watershed by the Toronto Region Conservation Authority to improve access for Atlantic Salmon and removal of the Hogansburg dam to restore connectivity in the St. Regis River. The amount of connected aquatic habitat was not calculated in 2017, and the status was assessed as “Fair” based on the Lake Ontario Biodiversity Conservation Strategy. This status was changed to “Poor” based on new data on aquatic habitat connectivity and not a genuine change in status.

Status Assessment Definitions
The following ranking categories are used to assess aquatic connectivity for this sub-indicator:

Good: >75% aquatic habitat connectivity, i.e. healthy level of connectivity, and access to spawning habitat is likely sufficient to maintain fish population
Fair: 25% to 75% aquatic habitat connectivity, i.e. stressed levels of connectivity, and access to spawning habitat is likely limiting population of some fish species
Poor: < 25% aquatic habitat connectivity, i.e. impaired connectivity, access to spawning habitat is likely to limit population of some fish species
Undetermined: % of impaired connectivity is insufficient to assess conditions of the ecosystem components or data are not available.

The current categories and thresholds are based on the results of biodiversity conservation strategies that were prepared for each Great Lake and used the Open Standards for Conservation.

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.
Note: changes based on new spatial information will not be considered in the trend assessment.

Endpoints and/or Targets

1. Number of key initiatives to improve aquatic habitat connectivity.
2. 25% increase in number of watersheds within a lake basin with an increase in aquatic habitat connectivity.
3. At least 75% of the total length of each type of stream is connected to each lake and are sufficient to maintain target populations of migratory fishes. (Pearsall et. al 2012b)
4. Tributary connectivity is maximized for migratory fish, while increased risk of aquatic invasive species spread and proliferation is minimized. (Pearsall et. al 2012b)

Sub-Indicator Purpose

- To determine the amount of accessible tributary habitat for migratory Great Lakes fishes;
- To summarize key initiatives to improve the connectivity of aquatic habitat; and
- To highlight some of the issues related to barrier mitigation.

Ecosystem Objective

Maintaining or increasing the aquatic habitat/connectivity for native fish would be considered desirable. Conversely, decreases in aquatic habitat connectivity would be considered undesirable.

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

In almost all cases, increases in aquatic connectivity would be considered desirable (barriers to Sea Lamprey are the exception). Conversely, decreases in aquatic connectivity would be considered undesirable. Although direct measures on 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 of data 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. This sub-indicator does not include the loss of lateral aquatic habitat connectivity (i.e. to floodplains) that has also been impacted by flow regulation, habitat conversion and hardening of river banks. Data to measure lateral aquatic habitat connectivity has not been developed for 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.

These calculations should be updated every five years (due to ongoing dam construction and/or removals), and can now be based on the data 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). The data on barriers can be used to analyze for the percentage of tributaries that are connected to each of the Great Lakes. A critical requirement will be periodic updates and validation of the dam inventory database.

This sub-indicator will measure tributary connectivity as the percent of mainstem channel length (naturally accessible) that is currently connected to the Great Lakes. It can be calculated for the entire Great Lake basin, the basin of each lake or by watershed.

Tributary Connectivity for the Great Lakes = $\frac{L_b}{L_m} \times 100$

L_b is the distance between the Great Lakes and the first barrier on the mainstem channel;
L_m is the total length of the mainstem channel.

Values range from 0 to 100. A value of 0.0 represents a main stem channel that is not connected to the receiving water body (no connectivity) and a value of 100 represents a main stem channel with unimpaired connectivity.

Ecological Condition

Dams have impacted over half of the world's drainage basins (Nilsson et. al 2005). Dams and barriers have been impacting the health of aquatic ecosystems in the Great Lakes basin for over a century and are limiting the recovery of some fish populations. In addition to limiting access of fishes to spawning and nursery habitats, loss of aquatic connectivity impacts nutrient flows, and riparian and coastal processes.
The construction of new dams and barriers on Great Lakes tributaries peaked over a century ago when water power was the primary energy source in the basin. Many of the larger dams were built in the 20th century for hydro-electric 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 habitat connectivity is defined for the purposes of this report as the direct connection between the Great Lakes and waterways that are used by migratory fishes. 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. Certain migratory fish species (e.g. Atlantic Salmon and Walleye) depend on unimpeded access to spawning habitats in streams. In many cases dams and other obstructions (e.g. perched culverts) prevent mature fish from reaching spawning habitat and thus compromise stock and species diversity, losses in annual recruitment and reduced production and harvests. For some fishes (e.g. Walleye, Lake Sturgeon) passage facilities will not mitigate these effects, because these species cannot jump. In addition to impacting the fishes that migrate from the Great Lakes into tributaries, many stream-dwelling species of fish (e.g. suckers and minnows) suffer discontinuity in their ranges because of barriers.

The installation and management of dams threatens the diversity 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). Less than 20% of tributary habitat in the basin is connected to the waters of the Great Lakes (Table 1, Figure 1). This loss of tributary habitat has resulted in significant declines in native fish populations in the lake, 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).

Aquatic habitat connectivity is complex, as some dams and barriers impede the movement of some fish species but not others. 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.

Linkages

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

- Aquatic Invasive Species – There are examples in all of the Great Lakes where dams and barriers, in some instances, are protecting the native stream assemblages from competition and physical disturbance of substrates from non-native salmonids (Bredin 2002). Hence, decisions about removal of dams and barriers in Lake Huron 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, Tubenose Goby, Viral Hemorrhagic Septicemia, and Sea Lamprey (see below). 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.
- Sea Lamprey – Dams and barriers currently limit the spread of some Great Lakes invaders. 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.
- Walleye – Loss of aquatic connectivity has contributed to the decline of fluvial populations.
- Water Quality in Tributaries – Barrier removal could improve water quality as natural flow patterns are restored and stream temperatures are reduced.
- 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.

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).

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 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. Spatial analysis of connectivity can be challenging if dams coordinates do not intersect with the hydrology layer. Road stream crossing can highlight potential barriers, but these need to be ground-truthed to assess their actual impact. Recent efforts to relicense 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 identification of new projects will require research and contact with agencies.

This 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.

Additional Information

Since the last indicator report, the University of Wisconsin has launched a website (Fishwerks) that maps barriers and provides a decision support tool for barrier removal. There are also an increasing number of 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).

Acknowledgments

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List of Tables

Table 1. Summary of the number of barriers, tributary length, and tributary connectivity in the Great Lakes. Source: Fishwerks

List of Figures

Figure 1. Tributary Connectivity in the Great Lakes (2018)
Source: Fishwerks

Last Updated

State of the Great Lakes 2019 Technical Report

Watershed	Number of Barriers	Total Tributary Length (km)	Connected Tributary Length (km)	Tributaries Connected (%)
All	102,676	236,315	45,499	19%
Lake Erie	29,176	39,175	10,191	26%
Lake Michigan	25,372	42,848	5,798	14%
Lake Superior	9,232	64,184	17,948	28%
Lake Huron	21,125	60,560	10,644	18%
Lake Ontario	17,771	29,548	919	3%
Canada	29,852	119,111	22,435	19%
United States	72,824	117,204	23,064	20%

Table 1. Summary of the number of barriers, tributary length, and tributary connectivity in the Great Lakes. Source: Fishwerks

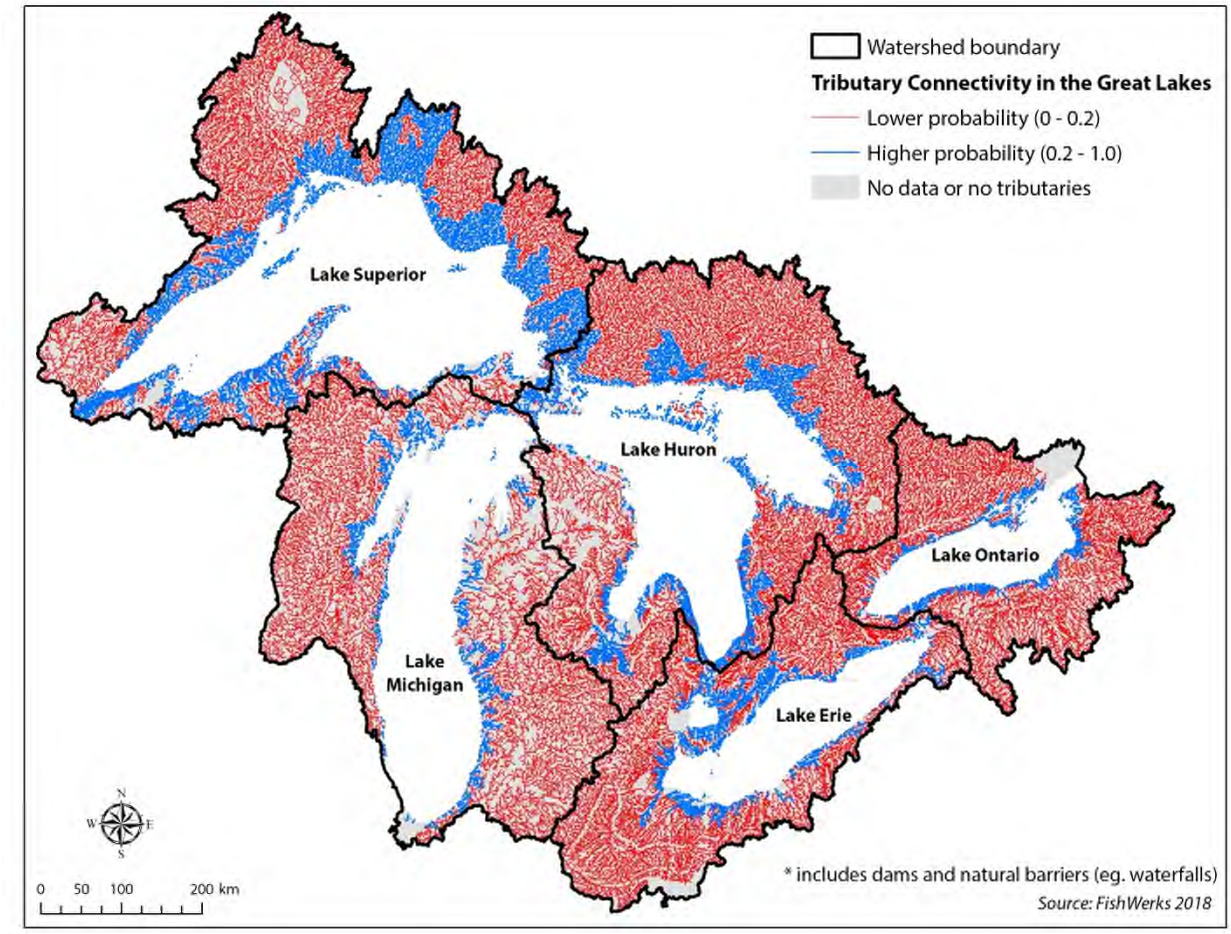


Figure 1. Tributary Connectivity in the Great Lake (2018)
Source: Fishwerks

Sub-Indicator: Phytoplankton

Open water

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Deteriorating
Long-term Trend (1950-2016): 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.

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 (1950-2016): Unchanging
Rationale: The lake has maintained a phytoplankton assemblage reflecting oligotrophic conditions. 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 (1950-2016): 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 (first year- last year): 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 (1950-2016): 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
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1950-2016): 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.

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
Lake-specific trophic status that reflects baseline conditions determined from paleolimnological analyses (oligotrophic conditions for Lakes Superior, Huron and Michigan; oligo-mesotrophic conditions for Lakes Erie and Ontario). Re-attaining historical phytoplankton assemblages (or "desirable" species) is not a realistic target because of the fundamental reorganization of the algal communities that are likely not recoverable under current physical, chemical and biological regimes. Target and/or endpoint interpretations could be based on assemblage structure, biovolume, or inferred phosphorus concentrations. To include these specifics in this section would be onerous, and would be better provided in the interpretations throughout the indicator report. General terms like "oligotrophic" are suitable in this section.

At this time 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 of Lake Huron may be interpreted as lower nutrient stress, but instead is a negative food web impact resulting from the invasion of profundal dreissenids.

Sub-Indicator Purpose
The purpose of this 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 and lower food webs. This includes inferring impacts from pollution, invasive non-native species and climate change.

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 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." 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 (e.g. Reavie et al. 2014 [EPA, twice yearly pelagic collections in all lakes], Conroy et al. 2005 [The Lake Erie Plankton Abundance Study]). Taxonomic identification is to the best detail possible (e.g., species) to identify atypical or non-native taxa. Abundances of taxa (e.g., blue-green algae or diatom blooms) will reflect conditions relative to known historical condition within a lake. Ideally, 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) assessments and may include bulk biomass measurements (e.g. chlorophyll *a*), biovolume (e.g. $\mu m^3/mL$) and assemblage composition. Paleolimnological data allow for historical inferences as far back as the 19th century, and such data were used to support the new "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 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 plants 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) and recent paleolimnological studies of fossil phytoplankton (e.g. Shaw Chraïbi et al. 2014, Sgro et al. 2018). Additional phytoplankton data have been collected by 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 significantly 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 decade, mainly indicated by cyanobacterial blooms that are occurring with greater frequency in the western basin of Lake Erie. 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.

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. 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 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.

In the western basin of Lake Erie, blooms of the nuisance algae *Microcystis* (among other cyanobacteria) have continued to occur (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.

Over the last decade in Lake Ontario spring chlorophyll levels have remained stable, but there is evidence of a slight summer chlorophyll increase (USEPA, unpublished 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 has been the case in Lake Erie.

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 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.

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			
Clarifying Notes: These data have been derived from many sources, including scientific literature, satellite data, and unpublished data.				

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.

The study of lower trophic levels and their use as indicators has been largely ignored in the Great Lakes, especially in 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, heterotrophic nanoflagellates, autotrophic picoplankton, and ciliates up to phytoplankton (nanoplankton and microplankton-netplankton). 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.

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.

The main purposes of this indicator are to: (1) measure biological responses of primary producers to changing water quality and invasive species abundance; (2) evaluate direct problems (e.g. blooms) associated with phytoplankton; (3) 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.

It is worth considering the incorporation of satellite-based estimates of phytoplankton biomass (as estimated by chlorophyll) to assess the status of offshore communities. Such an application will enable filling temporal gaps in the monitoring data. Further, temporal gaps should also be filled by year-round collections similar to those collected by the Lake Guardian, 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), but remote, automated sampling stations could play an important role in more comprehensive year-round sampling.

Acknowledgments

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List of Figures

Figure 1. Histograms of phytoplankton biovolume and community composition in the Great Lakes basins from 2001 through 2016. Spring and summer assemblages are provided from offshore, surface waters. Small numbers at the bottom of each bar indicate the number of samples averaged. Major noteworthy trends include: 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).
Source: U.S. Environmental Protection Agency, Great Lakes National Program Office. Modified from Reavie et al. (2014a)

Last Updated

State of the Great Lakes 2019 Technical Report

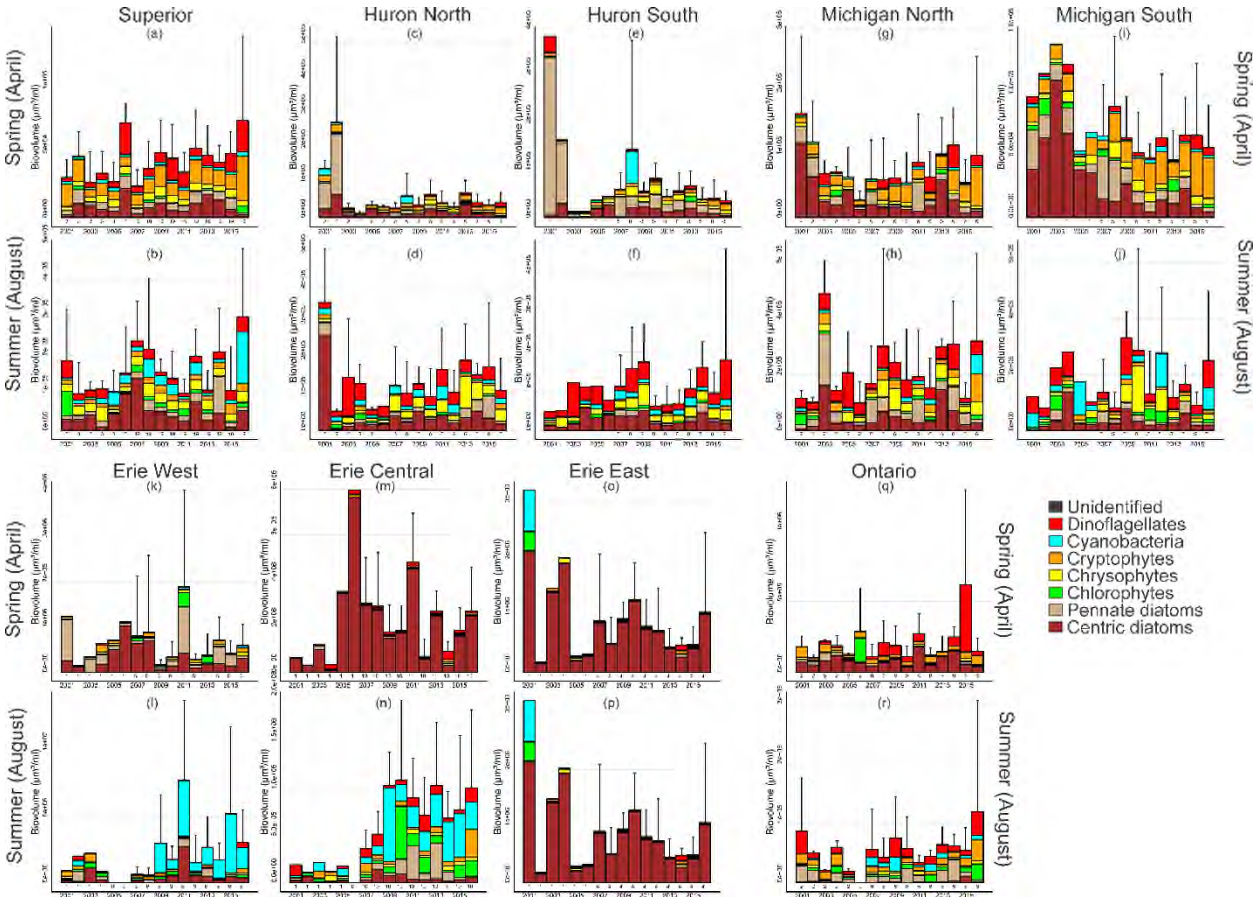


Figure 1. Histograms of phytoplankton biovolume and community composition in the Great Lakes basins from 2001 through 2016. Spring and summer assemblages are provided from offshore, surface waters. Small numbers at the bottom of each bar indicate the number of samples averaged. Major noteworthy trends include: 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).
Source: U.S. Environmental Protection Agency, Great Lakes National Program Office. Modified from Reavie et al. (2014a)

Sub-Indicator: Zooplankton
Open water

Overall Assessment
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1997-2016): Undetermined
Rationale: The oligotrophic zooplankton community of Lake Superior have been unchanging from 1997 to 2016. Lake Huron experienced a well-documented decrease in zooplankton biomass in 2003, particularly for cladocerans. Similar but less significant biomass decreases and community shifts occurred at this time for Lakes Michigan and Ontario. Lake Erie zooplankton biomass has been highly variable over the time series with no discernible trend.

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 (1997-2016): 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 (1997-2016): 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
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1997-2016): 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, however the long-term trend remains of concern.

Lake Erie
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1997-2016): Unchanging
Rationale: Highly variable biomass over the time series for all three basins. The shallow Western basin is dominated by cladocerans, while calanoid copepods are more prevalent in the deeper Central and Eastern Basins.

Lake Ontario
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1997-2016): 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 1997-2016. New data added in this report is for 2012-2016.

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}) having a target offshore zooplankton biomass of 3 g m^{-2}. Although Lake Erie has a similar TP target as Lake Ontario, a shallower habitat necessitates lower whole water column zooplankton biomass goals of 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 track 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 measure the trophic efficiency of the food web at transferring algal production to fish. Zooplankton biomass has often been used to explain deviations in the relationship of 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 abundant zooplankton tend to improve water quality and fish production capacity.

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 overall areal biomass (g dry weight/m^2). This value can be presented as summer values (July-August) or as averages over the growing season (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 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). 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).

Note that biomass or standing stock of zooplankton is not a measure of productivity (g dw m^{-2} yr^{-1}), which is the preferred metric for estimating potential fish production. Measures of productivity require understanding population turnover (productivity/biomass or P/B ratios). Zooplankton generation times can range from weeks (for rotifers) to months (for cladocerans and copepods) to years (mysids). Secondary production by zooplankton can be estimated using the egg-ratio method (Paloheimo 1976), but few zooplankton monitoring programs have sufficiently frequent and spatially extensive sampling to use that method. Alternative temperature and body size-based models developed to estimate P/B ratios for copepods (Shuter and Ing 1997) and cladocerans (Stockwell and Johannsson 1997) agree well with egg-ratio based methods but are also uncertain without comprehensive seasonal sampling (Johannsson and Bowen 2012).

More focused indices consider mean body size or community composition (relative % abundance, ratios of taxa, or ordination). Mills et al. (1987) proposed the use of zooplankton mean body size as an indicator of the balance between predatory fish and prey fish as fish visually select large zooplankton. Gannon and Stemberger (1978) proposed an often-cited ratio of the density of calanoid copepods and combined numbers of cyclopoid copepods and cladocerans; a modification of this index using the proportion of total zooplankton biomass that consists of calanoid copepods is being considered for inclusions in future reports. Such an index ranges from 0 (no calanoids) to 1 (all calanoids). Increasing representation of calanoids tracks oligotrophic settings as long as depth sampled does not change.

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 is now available for the period 1997-2016.

Sampling efforts intensify during Cooperative Science Monitoring Initiative (CSMI) years that rotate through a five-year cycle for each lake (e.g. Lake Ontario in 2018). Typically, these sampling programs add coordinated offshore zooplankton sampling by U.S. EPA GLNPO, US Geological Survey (USGS), NOAA Great Lakes Environmental Research Laboratory (GLERL), Department of Fisheries and Oceans Canada (DFO), and other agencies to achieve lakewide coverage for 3 to 5 time periods over a year.

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 in the zooplankton communities of Lakes Huron and Michigan, and to a lesser extent Lake Ontario, are consistent with reductions in nutrient levels, which have been seen in all three lakes, and could represent a consequence of nutrient reduction activities, perhaps compounded by effects of dreissenid mussels (Figures 2, 3 and 7). The reductions in cladocerans in the former two lakes, 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 Michigan and Huron have converged towards Lake Superior in terms of biomass levels and community composition (Barbiero et al. 2012). These track an increase in the importance of calanoid copepods that is consistent with Gannon and Stemberger (1978)'s oligotrophic index. 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). The updated time series shows 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 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* that may be related to low alewife abundance and a rise in invertebrate predation by predatory cladoceran *Bythotrephes* (Barbiero et al. 2014, Rudstam et al. 2015). In 2013, the community returned to more cyclopoid copepods and bosminids following a strong alewife year class in 2012 and a decline in *Bythotrephes*, illustrating the coupling between lower and higher trophic levels. A shift back to calanoids was seen in Lake Ontario in 2015-2016.

The zooplankton community of Lake Erie is taxonomically diverse and rich in native and non-native cladoceran species (Figures 4-6). 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 shallow habitat as well as a mesotrophic trophic state. Zooplankton biomass has been highly variable in all three basins with no discernible trend.

Linkages

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

- Other Habitat and Species sub-indicators (phytoplankton and benthos).
- Nutrients in Lakes (open water) – phosphorus levels regulate primary productivity by phytoplankton and thus food levels for zooplankton.
- Increased water clarity shifts primary production to deeper depths in the form of deep chlorophyll layers (DCL).
- The connection of the zooplankton sub-indicator to other trophic levels provides a test of the principle developed in marine settings that pelagic communities, on average, have approximately equal biomass in exponentially widening size classes (Sheldon et al. 1972). Material and energy flow up this size spectrum from bacteria and phytoplankton via zooplankton to fish with varying efficiency. Some of this production 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 as algae, negatively affecting water quality while little energy reaches top predators. For example, Lake Erie can have low zooplankton relative to phytoplankton biomass, and thus attain nuisance levels of phytoplankton.
- Climatic changes 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.

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 Limitations

Both U.S. and Canadian agencies conduct extensive 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. GLNPO Standard Operating Procedures (SOPs) for zooplankton have been updated recently with more suitable length-weight regressions, particularly for calanoid copepods.

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.

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.

The importance 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.

The importance of mysids as part of the zooplankton community and as prey of planktivorous fish needs to be assessed. Mysids can represent up to 30-40% 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). Existing monitoring programs are based on both net collection and hydroacoustic transects. U.S. EPA GLNPO has recently formalized their mysid sampling program with new net collection and analysis SOPs and the 2006 to 2016 data has been compiled, analyzed, and is now published (Jude et al. 2018).

Additional Information

Note that we use areal biomass (g m^{-2}) rather than volumetric (mg 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 areal units was done for the last report. 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 are thus "diluted" relative to shallower lakes that have less hypolimnion. Areal biomass is calculated by summing the zooplankton biomass found within one meter square of lake 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.

The length-weight coefficients have been updated for calanoid copepods based on recent studies to better reflect their contribution (Watkins et al. 2011, Burgess et al. 2015). This update leads to an increase in estimated calanoid biomass by a factor of 2 compared to previous State of the Great Lakes indicator reports and Bunnell et al. (2014).

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). A *Mysis* sub-indicator has been proposed for the Great Lakes suite as well which could be supported using the GLNPO time series. Such sub-indicators need further development. Nearshore measures for the Zooplankton sub-indicator have also been proposed beyond 2016. Several long-term nearshore biomonitoring programs that focus on zooplankton exist (e.g. Lake 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.

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. Invasive predatory cladocerans have been shown to have had a major impact on zooplankton community structure in the Great Lakes (Lehman 1991; Barbiero and Tuchman 2004; Warner et al. 2006). The four new non-native zooplankton species recently detected in Western Lake Erie have not rapidly expanded and potential impacts are not likely to be significant.

Some of the other measures which would help with the interpretation of the zooplankton data would include: total phosphorus, chlorophyll *a*, temperature, oxygen (seasonal depletion in upper hypolimnion including anoxia in central basin of Lake Erie), and, if possible, primary production and phytoplankton composition and biomass.

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Data Source:
U.S. EPA Great Lakes National Program Office
Elizabeth Hinchey-Malloy, Great Lakes National Program Office (U.S. EPA), Chicago, IL
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List of Figures

Figures 1-7. Areal biomass (g m^{-2}) calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-μm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

Last Updated

State of the Great Lakes 2019 Technical Report

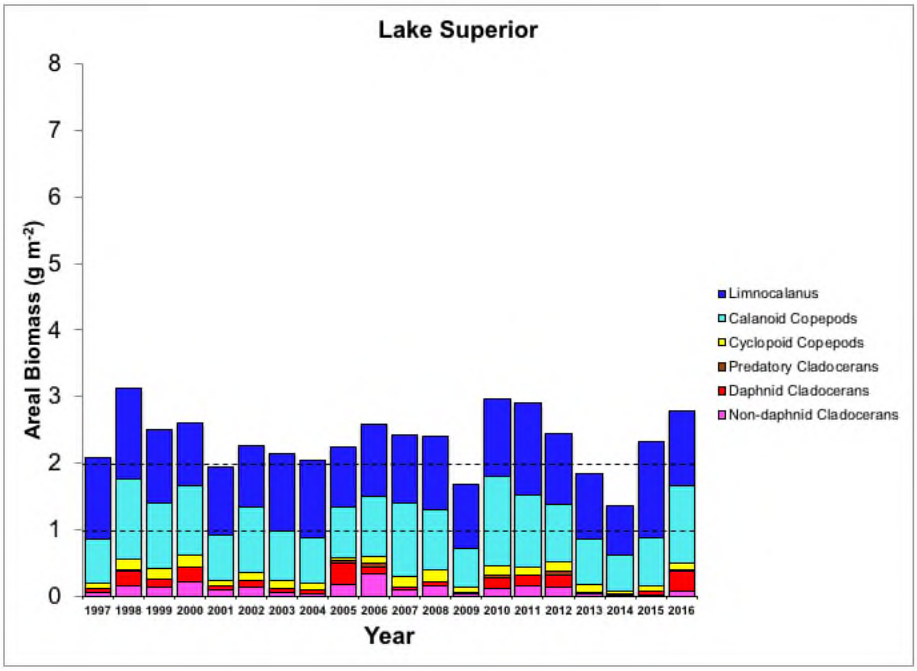


Figure 1. Areal biomass (g m^{-2}) for Lake Superior calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

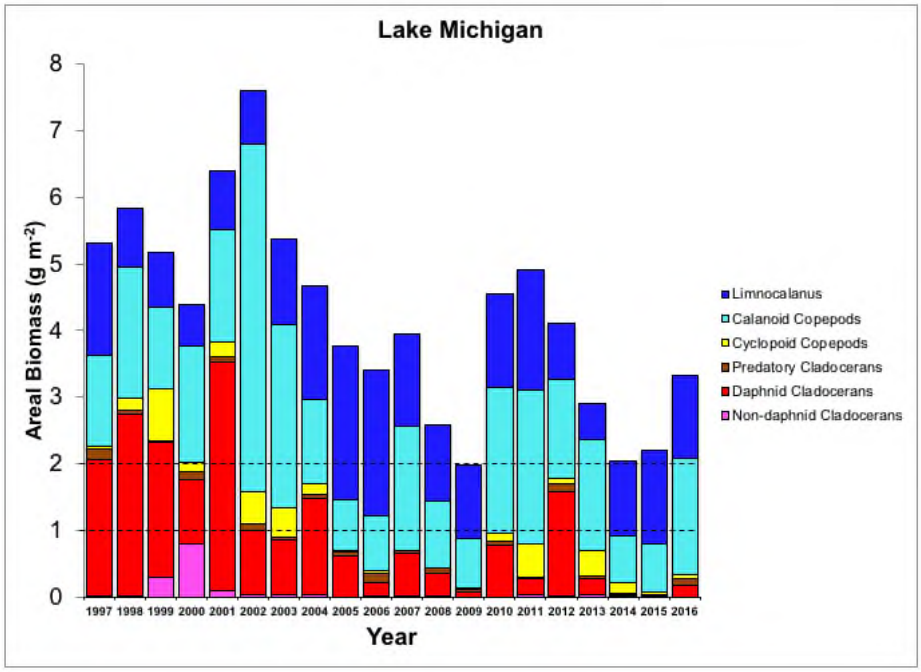


Figure 2. Areal biomass (g m^{-2}) for Lake Michigan calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

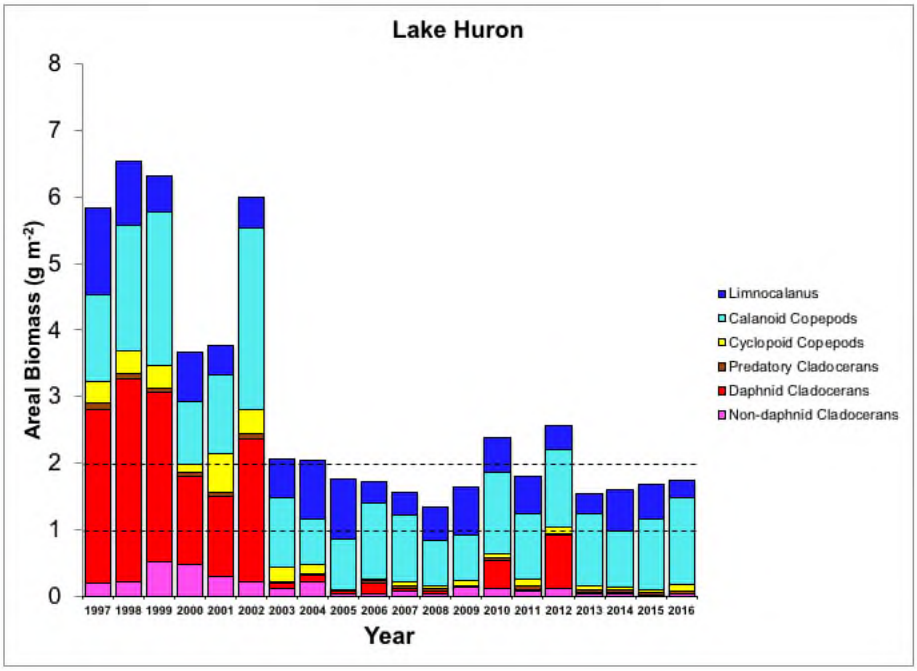


Figure 3. Areal biomass (g m^{-2}) for Lake Huron calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

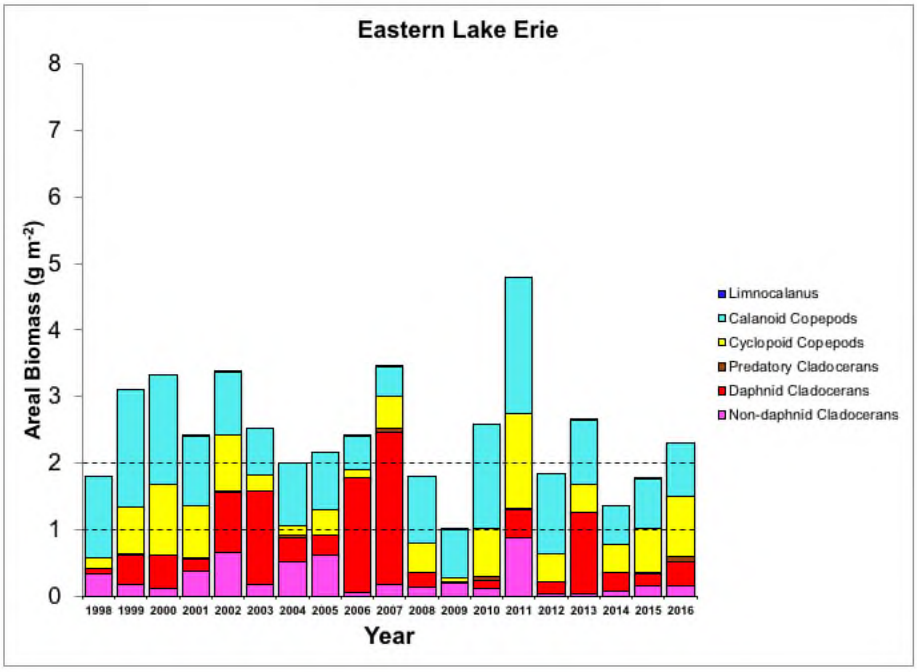


Figure 4. Areal biomass (g m^{-2}) for Eastern Lake Erie calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

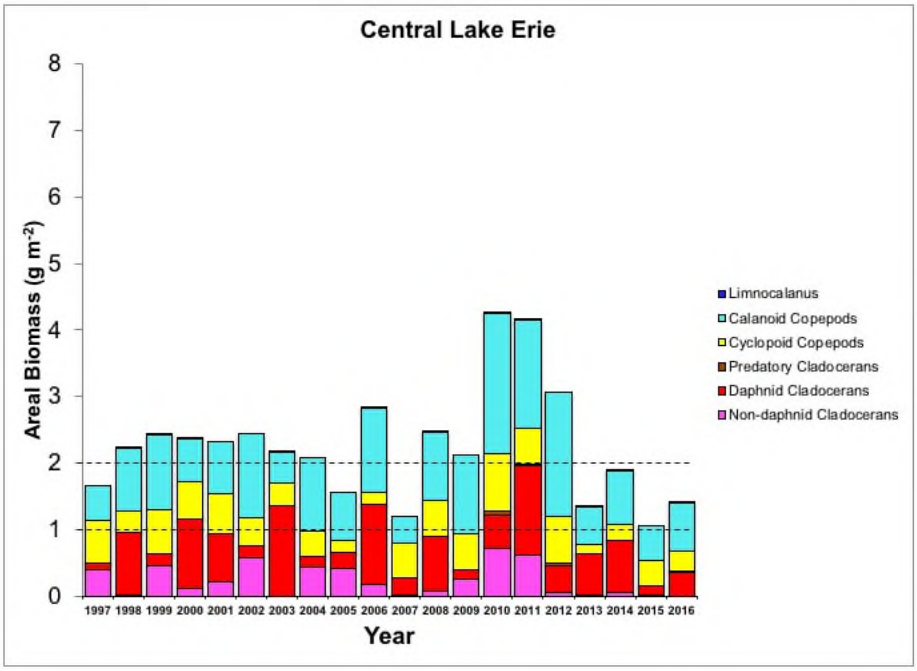


Figure 5. Areal biomass (g m^{-2}) for Central Lake Erie calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

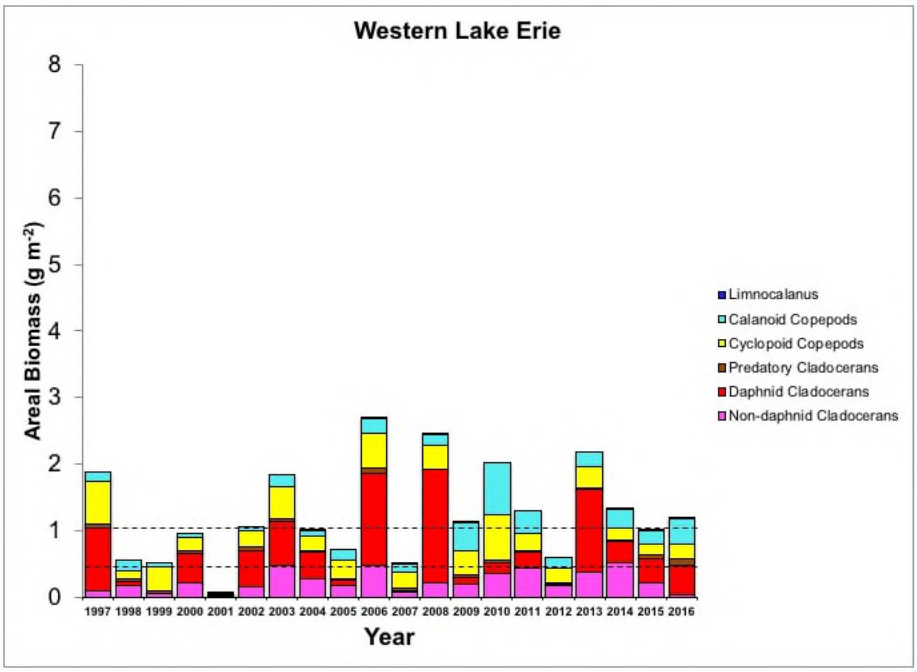


Figure 6. Areal biomass (g m^{-2}) for Western Lake Erie calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
Data Sources: U.S. EPA Great Lakes National Program Office, Cornell University

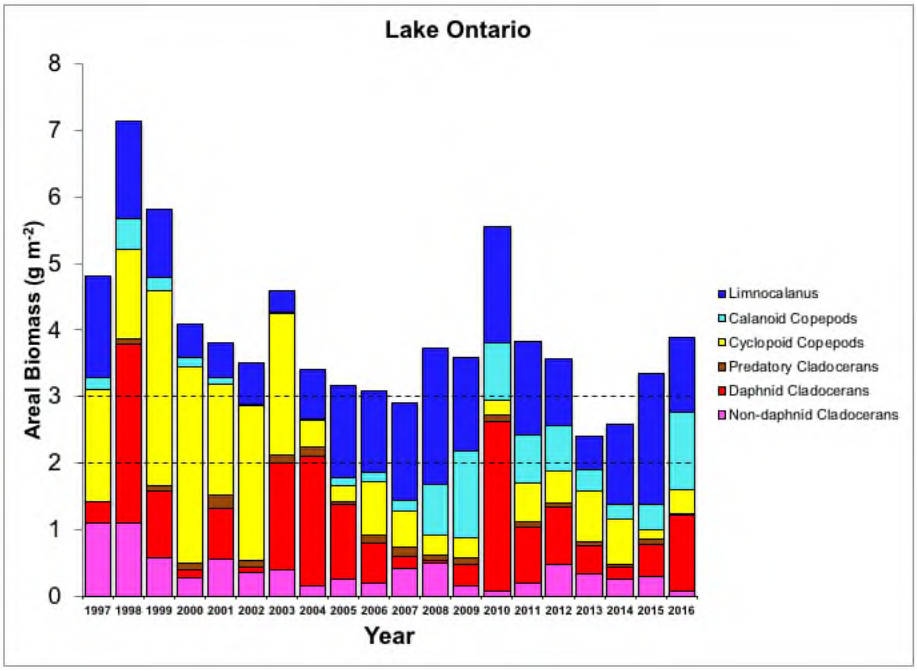


Figure 7. Areal biomass (g m^{-2}) for Lake Ontario calculated from U.S. EPA's GLNPO summer survey D100 tows (100 m or 2 m above bottom for shallower sites) 153-µm tows for each lake. Length-weight coefficients used are listed in EPA SOP LG 403. "Good" and "Poor" thresholds are identified by dashed lines for each figure.
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 (1997-2016): Unchanging
Rationale: Based on the Oligochaete Trophic Index (OTI) scores, both the status and long-term (1997 - 2016), 10-year (2007-2016) and short-term (2013-2016) trends in the trophic condition of the lakes are generally considered to be good and unchanging, except for Lake Erie and some shallow stations in lakes Ontario, Huron and Michigan where the long-term and 10-year trends are indicative of increased eutrophication and are assessed as poor. Overall, an increasing OTI score reflects increasing eutrophication or an increase in trophic condition.

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 (1997-2016): Unchanging
Rationale: All stations in Lake Superior were classified as oligotrophic, based on OTI scores, both long-term (1997-2016) and in recent years (2013-2016). No significant negative trends were found in the trophic condition of the lake both long-term and in the last 10 years. 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 (1997-2016): Unchanging
Rationale: Since 1997, almost all stations in northern and central Lake Michigan, as well as deep stations in the southern part of the lake, have had OTI scores below 0.6 indicating an oligotrophic condition. OTI scores > 1.0 (indicating “poor” condition) were reported in recent years at two nearshore stations in the southeastern part of the lake. Significant long-term trends of increasing eutrophication are evident at four nearshore stations, compared to two stations indicating similar trends in the previous report. The endpoint for this sub-indicator is to maintain an oligotrophic state in the open waters of Lake Michigan. Overall, no significant negative trends were found lake-wide in the trophic condition both long-term and in the last 10 years.

Lake Huron
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1997-2016): Unchanging
Rationale: Almost all stations in northern, southern and central Huron are oligotrophic, except for one mesotrophic station on the eastern shore, one eutrophic station in the southern basin, and one in Saginaw Bay. The trophic condition of the lake, as measured by OTI scores, has not changed significantly in the last 20 years. 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 (1997-2016): Deteriorating

Rationale: All stations in Lake Erie are eutrophic, and several have a long-term trend of increasing OTI scores. The highest OTI scores occur in the eastern basin. A significant negative trend was found in the trophic condition of the lake since 1997, similar to the previous report. 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 (1997-2016): 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 most of the nearshore stations are mesotrophic or eutrophic. Two stations changed their status from mesotrophic to eutrophic in 2013-2016 compared to the previous reporting cycle. Two stations in the western basin and one station in the eastern basin showed long-term trends of increasing eutrophication. Overall, no significant negative trends were found in the trophic condition of the lake either since 1997 or in the last 10 years.

Status Assessment Definitions
Good: OTI score of less than 0.6
Fair: OTI score of 0.6 to 1.0
Poor: OTI score of greater than 1.0
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 community indicative of oligotrophic conditions throughout the lakes, with the exception of the central and western basins of Lake Erie, where mesotrophic conditions are desired.

Improving: a decrease in the OTI indicating declining eutrophication
Unchanging: OTI score is not changing
Deteriorating: an increase in the OTI indicating increasing eutrophication
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, relative algal biomass, and algal species consistent with healthy aquatic ecosystems, in the open waters of Lakes Superior, Michigan, Huron and Ontario; 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.

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 and to infer health of the benthic community.

Ecosystem Objective
The Ecosystem Objective is that the benthic community in the Great Lakes should remain relatively constant over time and be comparable to unimpaired waters with similar depth and substrate. One estimate of benthic community status and lake trophic condition is based on the OTI, which uses oligochaete abundance, diversity, and individual species responses to organic enrichment to compute trophic status of a body of water.

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."

This sub-indicator will evaluate trophic conditions in the Great Lakes using oligochaete diversity, abundances, and the individual species responses to organic enrichment.

Measure
This sub-indicator (OTI, ECCC and U.S. EPA) will evaluate trophic conditions in the Great Lakes using oligochaete diversity, abundances, and the individual species responses to organic enrichment.

Calculation of the OTI

To evaluate trends in the benthic community of the Great Lakes, 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. 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 attached table) 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 will be drawn from the regular long-term benthic monitoring program of U.S. EPA Great Lakes National Program Office (GLNPO) which visits 59 stations annually within the five Great Lakes. Please note that beginning in 2016, the station ON64 was discontinued, thus the total number of stations from 2016 is 58. Less frequent but denser 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 (previously known as SOLEC) 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 58 permanent GLNPO sampling stations (Lakes Superior and Huron: 11 stations each, Michigan: 16, Erie and Ontario: 10 stations each) located mostly offshore (Figure 2). The trophic condition index 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 index ranges 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 1.0 suggest mesotrophic conditions. Scores approaching 3 indicate high densities of

oligochaetes dominated by the pollution tolerant tubificids including *Limnodrilus hoffmeisteri*.

A consistent difference in trophic conditions among and within the Great Lakes was found during the study period (1997– 2016) (Figure 1). Trophic condition was significantly inversely related to station depth ($r = –0.60$, $P < 0.001$), 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. The only significant lake-wide long-term trend of increasing trophic condition ($P = 0.02$) was observed in Lake Erie, where significant trends were found at almost half of the sampled stations. Localized increases in OTI scores over time were found at shallower stations in Lake Michigan, Lake Huron, and Lake Ontario (Figure 2).

All 11 stations in Lake Superior were oligotrophic based on OTI scores since 1997, and there were no significant long-term or 10-year trends in OTI scores at either a lake-wide scale nor at specific stations (Figure 3). The trend of eutrophication identified at one western station and the trend of oligotrophication identified at one western station in the previous report were not confirmed in the last four years.

Almost all stations in northern and central Lake Michigan, as well as deep stations in the southern part of the lake (13 of total 16) are oligotrophic (Figure 2). Two nearshore stations in southeastern Michigan (near the Grand and Kalamazoo River outlets) are eutrophic and both stations had significant long-term trends of increasing eutrophication ($P < 0.02$). During the last four years, two more nearshore stations in northwest Michigan showed signs of eutrophication (MI42 and MI52, $P < 0.03$). One station in Green Bay (MI49) showed the opposite trend of increasing oligotrophication (Figure 3).

Almost all stations in northern, southern and central Huron (8 of total 11) are oligotrophic; one station in the eastern basin is mesotrophic (Figure 2). This mesotrophic station (HU96, near the outlet of Saugeen River in Ontario, Canada) again showed an increasing eutrophication trend, as observed 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, and consequently the OTI score significantly increased as well ($P = 0.02$). Only two stations in southern Lake Huron are eutrophic, including one in Saginaw Bay, which was highly eutrophic in the late 1990s, improved to mesotrophic in 2002, but has become eutrophic again since 2007. Since the last reporting period, one more station in central basin (HU97) showed sign of eutrophication, while one station in northern Huron (HU48) is displaying a long-term trend toward oligotrophication.

The most eutrophic stations in Lake Erie were found in the eastern basin, and significant long-term trends of increased OTI scores were found here at two of five sampled stations, while one more nearshore station (ER 63) showed a decline in OTI score (Figure 3). We found significant trends of increasing trophic conditions since 1997 at two of the three stations in the central basin ($P < 0.025$) and basin-wide ($P = 0.004$, 10 stations total). In contrast, OTI scores for stations in the western basin have not changed significantly since 1997 (Figure 3). The average OTI score for the eastern basin (1.92 ± 0.50, mean ± standard deviation) exceeded those for both the western (1.39 ± 0.48) and the central basins (1.41 ± 0.35). The overall phytoplankton biomass in the lake has increased since the mid-1990s (Conroy et al. 2005b), potentially a result of the dramatic increase in dissolved reactive phosphorus loads from tributaries (Richards et al. 2010; Scavia et al., 2014). In addition, dreissenid populations declined in the central basin in the early 2000s (Patterson et al. 2005; Karatayev et al. 2014, 2018) most likely due to hypoxic events. Considering that the central and eastern basins are the main areas of long-term sediment and organic matter deposition in Lake Erie, the increase in basin- and lake-wide OTI scores may be indicative of increasing trophic condition of the lake.

Deepwater stations in Lake Ontario have remained oligotrophic throughout the 1997-2016 reporting period. In contrast, all the nearshore stations, especially along the southern shore, are mesotrophic or eutrophic (Figure 2). Three stations (out of 10 total) displayed a trend toward increasing eutrophication since 1997: two nearshore stations in the western basin (likely being affected in the southern shore by the outlet of the Niagara River, and on the northern shore by the Toronto metropolitan area) and one station in the eastern basin (Figure 3). Compared to the previous report, the OTI scores of nearshore stations increased during the 2013-2016 time period, and two stations have changed their status from mesotrophic to eutrophic. As a result, significant long-term trends of increasing OTI scores were found at two additional nearshore stations. However, the deepwater stations, for the most part, remain oligotrophic and unchanging over the long- and shorter-term trends.

Linkages

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

- Dreissenid Mussels – the relative abundance of non-native benthic species such as zebra and quagga mussels can dramatically change the structure of aquatic communities including the benthos, affect ecosystem functioning and 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, decreasing the amount of available food for some profundal taxa. 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 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 can be 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).
- *Diporeia* (open water) – *Diporeia* is a benthic macroinvertebrate in the cold, deep-water habitats of all the Great Lakes (except Lake Erie). It is an indicator of oligotrophic conditions, and is an important fish food item. Historically *Diporeia* has 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 not yet well-understood. While the abundance of *Diporeia* is not considered by the current index (OTI), a significant increase in organic enrichment may negatively affect *Diporeia*.
- Climate change – increasing 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 temperature. Both abiotic changes and biological responses in the lakes are, however, more complex, and changes in water chemistry, hydrology, water levels, etc. may be important as well for the performance 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, fishing pressure) 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

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			
Clarifying Notes: *The annual benthic monitoring program of U.S. EPA GLNPO has a limited number of stations.				

Data Limitations

The benthic monitoring program of U.S. EPA Great Lakes National Program Office (U.S. EPA GLNPO) is designed to efficiently track the long-term trends at a variety of locations across each Lake. Due to the variability and complexity of the benthic zone, the data cannot be used to definitively quantify oligochaete populations lakewide. Current monitoring has poor representation of nearshore areas. Complementing these annual surveys with a wider range of stations during Cooperative Science and Monitoring Initiative (CSMI) years and historical data will aid greatly in identifying trends in benthic community.

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. In addition, this index is based on only a handful of Oligochaeta species. The performance of the OTI was recently examined and an improved index (iOTI) based on a larger number of Oligochaeta species has been suggested (Burlakova et al., 2018a). In addition, the National Costal Condition Assessment (NCCA) is considering improvements to the Index for the nearshore zone where it fails to perform at a large number of sites due to lack of Oligochaeta. A better performing index should be selected and applied to the 2022 sub-indicator report to better classify the trophic condition of the lakes and the trends in benthic community.

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. Even though mussel biomass has been declining in the 30-90m depth zones in some of the lakes, dreissenids are still a dominant component of the benthos in 2013-2016, but are not considered by the OTI.

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). 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 (Higgins and Vander Zanden 2010; 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 but denser sampling of all Great Lakes occurs during the Cooperative Science and Monitoring Initiative intensive field years to follow changes in benthic community diversity and abundance. This sampling design will be used as a basis for intense sampling of all Great Lakes once every five years (within CSMI intensive field years). 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 will enable the estimation of lake-wide population changes of invasive and native indicator species (e.g., dreissenids, *Hexagenia*, *Diporeia*).

Values of the Benthos sub-indicator representative of the nearshore are very important, as nearshore benthos are more diverse and also are affected more quickly and more directly (than open-water benthos) to 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 Oligochaetes which are the basis of the current benthic index (OTI) used for State of the Great Lakes reporting and the GLNPO Biology Monitoring Program.

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List of Tables

Table 1. Trophic classifications for select mature lumbriculids and tubificids taken from Howmiller and Scott (1977), Milbrink (1983) with additions from Krieger (1984), Lauritsen et al. (1985). If Milbrink classifications differed from Howmiller and Scott, Howmiller and Scott was used.
Source: Riseng et al. 2014.

List of Figures

Figure 1. Scatterplot of the index scores for Milbrink's (1983) modified Oligochaete Trophic Index, applied to data from GLNPO's 1997 through 2016 summer surveys. Scores ranging from 0 to less than 0.6 indicate oligotrophic conditions (blue line); scores from 0.6 to 1.0 indicate mesotrophic conditions (black line); and scores greater than 1.0 indicate eutrophic conditions. Data points represent the average of triplicate samples taken at each sampling station; immature specimens were included in the analysis for calculation of overall density used to establish the coefficient *c* but only mature specimens were used to calculate the number belonging to each ecological group of oligochaetes (see attached description of index calculation).
Source: 1997-2016 U.S. EPA GLNPO benthic data collected from permanent stations.
Figure 2. Map of the Great Lakes showing the mean trophic condition at each sampling station calculated for 2013-2016. Trophic condition was based on the modified Oligochaete Trophic Index based on Milbrink (1983).
Source: 2013-2016 U.S. EPA GLNPO benthic data.
Figure 3. Maps of the Great Lakes showing stations with significant temporal trend in trophic condition between 1997 and 2016. 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: 1997-2016 U.S. EPA GLNPO benthic data.

Last Updated

State of the Great Lakes 2019 Technical Report

Trophic classifications for select mature lumbriculids and tubificids

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), Lauritsen et al. (1985). If Milbrink classifications differed from Howmiller and Scott, Howmiller and Scott was used.
Source: Riseng et al. 2014.

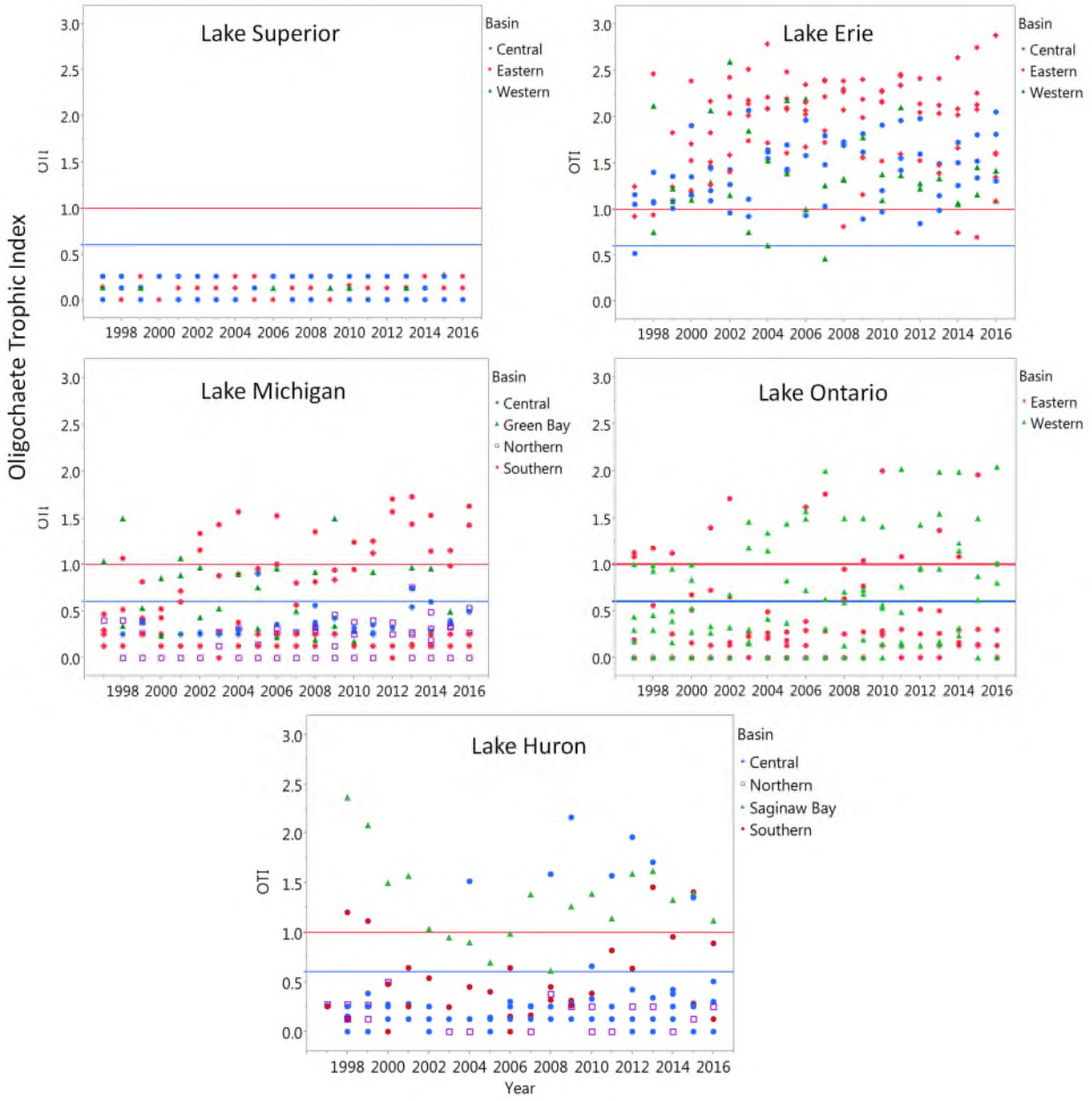


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Source: 1997-2016 U.S. EPA GLNPO benthic data collected from permanent stations.

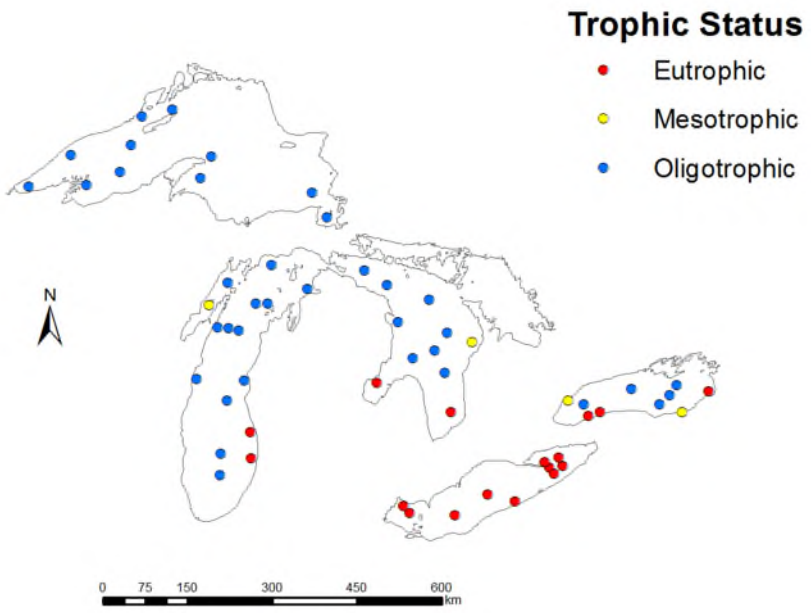


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Source: 2013-2016 U.S. EPA GLNPO benthic data.

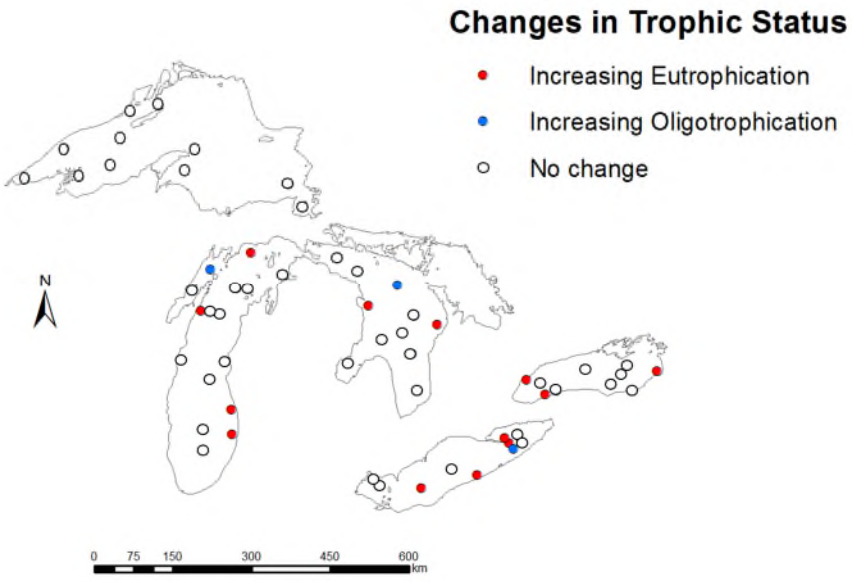


Figure 3. Maps of the Great Lakes showing stations with significant temporal trend in trophic condition between 1997 and 2016. Stations without significant changes in oligochaete trophic index with time ("no change", $P > 0.10$, linear regression) and with significant trends ("increasing eutrophication" or "oligotrophication", $P < 0.05$) are indicated.
Source: 1997-2016 U.S. EPA GLNPO benthic data.

Sub-Indicator: *Diporeia*

Open water

Overall Assessment
Status: Poor
Trends:
10-Year Trend: Deteriorating
Long-term Trend (1972-2017): Deteriorating
Rationale: *Diporeia spp.* was once the dominant benthic invertebrate in the Great Lakes, but their populations experienced rapid declines starting in the 1990's. 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, but populations remain at good levels. *Diporeia* is currently extremely rare in Lake Erie and has likely been extirpated. In all the lakes where *Diporeia* has declined, lower abundances first became apparent a few years after dreissenid mussels became established. The assessments included in this report are restricted to the main basins of each lake because *Diporeia*, a cold-water stenotherm, is not found in shallow-warm bays, basins, or connecting channels. The data presented here are mostly based on whole lake surveys conducted every 5 years, however, some data from frequent regional assessments are included as well.

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 (1973-2016): Unchanging
Rationale: Long term monitoring and studies of distribution patterns indicate substantial temporal variability, with a trend of decreasing density between 1994-2016 (Figure 1). However, the population is not considered to be "Deteriorating" in spite of the observed declines because the density of *Diporeia* both nearshore and offshore remain above the level recommended by the 1978 Great Lakes Water Quality Agreement. Further, *Diporeia* are consistently found throughout the lake and remain the dominant benthic organism (Mehler et al. 2018).

Lake Michigan
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1981-2017*):* Deteriorating
Rationale: *Diporeia* abundances continue to decline in Lake Michigan. A lakewide survey in 2015 indicated that *Diporeia* is still extremely rare at depths < 90 m (297 ft.) over the entire lake (Figure 2). At depths > 90 m, this taxa can still be found in "Fair" condition, but abundances were 58% lower compared to abundances found in 2005, and similar to 2010 levels (Figure 3). Recent annual surveys (2010-2017) conducted in just the southern region of Lake Michigan reveal *Diporeia* densities to be essentially absent <90m and low and variable >90m (Figures 4 and 5). There is only one site (depth = 129m) in southern Lake Michigan where *Diporeia* is consistently found.

Lake Huron
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1972-2017): Deteriorating
Rationale: *Diporeia* abundances continue to decline in Lake Huron. The most recent lakewide 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 is now < 1 m^{-2} at depths 31-90 m and 73.3 m^{-2} at depths > 90 m (Karatayev et al. in review). In 2017, the density of *Diporeia* at <90m in the North Channel was 170.1 m^{-2}, notably higher than that observed at comparable depths in the main basin (Karatayev et al. in review).

Lake Erie
Status: Poor
10-Year Trend: Unchanging
Long-term Trend (1978-2014): 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 (Barbiero et al. 2011).

Lake Ontario
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1994-2013): Deteriorating
Rationale: *Diporeia* abundances continue to decline in Lake Ontario (Figures 2 and 3). The 2013 lakewide survey in Lake Ontario produced only a single individual (site depth = 140m) out of the 45 sites sampled. Additional benthic monitoring by USGS has identified a deep site (95m) in western Lake Ontario with low but persistent numbers of *Diporeia* (B. Weidel, pers. comm.) Sample processing is underway for the 2018 lakewide survey, which will reveal if *Diporeia* are still constrained to a few refuge populations or are potentially approaching extirpation in Lake Ontario.

Status Assessment Definitions
Good: Densities of *Diporeia* remain above 220-320/m^2 in nearshore waters (<100m) and 30-160/m^2 in offshore waters (>100m; 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.
Fair: Densities of *Diporeia* remain above the GLWQA recommendations, but are only found at a few locations.
Poor: Densities of *Diporeia* are significantly below the GLWQA recommendations.
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
Unchanging: No change in *Diporeia* densities
Deteriorating: Decrease in *Diporeia* densities below GLWQA recommendations
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 <100m and >30-160/m^2 at depths >100m as per the 1978 GLWQA.

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/m2) of Diporeia in cold, deep-water habitats of the Great Lakes. The recent lake-wide trends presented here are based on extensive, whole lake surveys (nearshore and off-

shore) that are conducted in each of the lakes every 5 years as a part of the Coordinated Science and Monitoring Initiative (CSMI). Also presented are data collected during the annual NOAA/GLERL benthic survey of the southern basin of Lake Michigan.

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. 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 abundances.

For each lake with an updated metric, a figure with *Diporeia* metrics over time will be presented. For less frequent but more spatially-intense regional assessments, a figure giving metrics as described above or a figure giving density isopleths will be presented.

Ecological Condition

This glacial-marine relic was once the most abundant benthic organism in cold, offshore regions (greater than 30 m (98 ft) of each of the lakes. It was present, but less abundant in nearshore regions of the open lake basins, but 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; in particular by many forage fish species, which 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.

On a broad scale, abundances are directly related to the amount of food settling to the bottom, and population trends reflect the overall productivity of the ecosystem. Abundances can also vary somewhat relative to shifts in predation pressure from changing fish populations. In nearshore regions, this species is sensitive to local sources of pollution, but because of varying conditions such as temperature fluctuations, substrate heterogeneity, and wave-induced turbulence, it is difficult to assess population trends in this region.

Diporeia populations have undergone dramatic declines with no signs of recovery in all the lakes except Lake Superior (Figures 2 and 3). Based on the most recent surveys, *Diporeia* are present but continue to decline in Lakes Michigan and Huron, while it has likely been extirpated from Lake Erie and may be nearing extirpation in Lake Ontario. The population in Lake Superior is highly variable and had exhibited declines since 1994 and 2003 (Figure 1). However, their densities remain relatively high and *Diporeia* are found distributed throughout the lake. Initial declines among the lower Great Lakes were first observed within two to three years after zebra mussels (*Dreissena polymorpha*) or quagga mussel (*Dreissena bugensis*) first became established. Reasons for the negative response of *Diporeia* to these mussel species are not entirely clear. 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*. However, evidence suggests that the reason for the decline 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.

The Great Lakes National Program Office (GLNPO) monitoring program assesses abundances of *Diporeia* (and other benthos) in offshore regions of each of the lakes on an annual basis. In addition, extensive, whole lake surveys (nearshore and offshore) are conducted in each of the lakes every 5 years; also, NOAA/GLERL conducts a survey throughout the southern basin of Lake Michigan every year. 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.

Linkages

Linkages of this sub-indicator to other sub-indicators include:

- Benthos (open water) – *Diporeia* is still the dominant benthic species in Great Lakes deep-water habitats where dreissenid mussels are absent.
- 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*. 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.

Being a cold-water stenotherm, *Diporeia* could potentially decline in the nearshore zones of Lake Superior as water temperatures increase. Since *Diporeia* is already absent from the nearshore in the lower Great Lakes, no further declines are expected.

Assessing Data Quality

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 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 taxa, 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. Canadian data sources are limited due to changes in the monitoring programs within the Department of Fisheries and Oceans Canada. However, the CSMI whole-lake surveys include data from both U.S. and Canadian waters.

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 re-

sponses 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.

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 an understanding of exactly why *Diporeia* populations are declining, one may better predict what additional areas of the lakes are at risk. Also, by better understanding the cause, one can better assess the potential for population recovery if dreissenid populations significantly decline.

Acknowledgments
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B.C. Weidel, Great Lakes Science Center, U.S. Geological Survey, Lake Ontario Biological Station, Oswego, NY

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List of Figures

Figure 1. Mean density (number per square metre ± S.E.) of the amphipod *Diporeia* spp. From 25 stations in the U.S. nearshore waters of southern Lake Superior in 1994, 2000, 2003, and 2016.
Data Sources: Great Lakes Center, SUNY Buffalo; Mehler et al. 2018; Scharold et al. 2008.

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 – 2017. Data are from lake-wide surveys conducted mostly at 5-year intervals. Lake Michigan = triangles, dashed line (blue); Lake Huron = squares, dot-dash line (red); Lake Ontario = circles, solid line (black).
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Figure 3. Mean densities (number per square metre) of the amphipod *Diporeia* spp. from sites at > 90 m in lakes Michigan, Huron, and Ontario, 1995 - 2017. Data are from lake-wide surveys conducted mostly at 5-year intervals. Lake Michigan = triangles, dashed line (blue); Lake Huron = squares, dot-dash line (red); Lake Ontario = circles, solid line (black).
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Figure 4. *Diporeia* population density (number x 10^3 per square metre) declines in Lake Michigan, 1994/5 – 2015. Small red dots indicate location of sampling sites.
Data Source: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2014, 2017.

Figure 5. Mean densities (number per square metre) of the amphipod *Diporeia* spp. in southern Lake Michigan, reported by depth: < 30 m (squares, solid line); 31-90 m (triangles, long dashed line); and > 90 m (circles, short dashed line), 2010-2017. Note that the axis scale is greatly reduced compared to Figures 2 and 3.
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Figure 6. *Diporeia* population density (number x 10^3 per square metre) declines in Lake Huron, 2000 – 2017. Small red dots indicate location of sampling sites.
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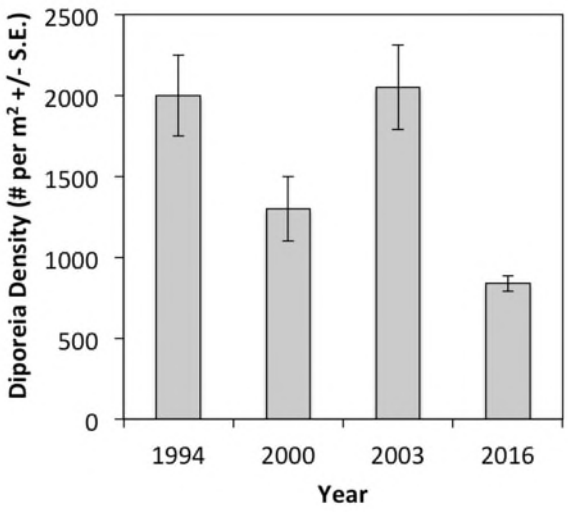


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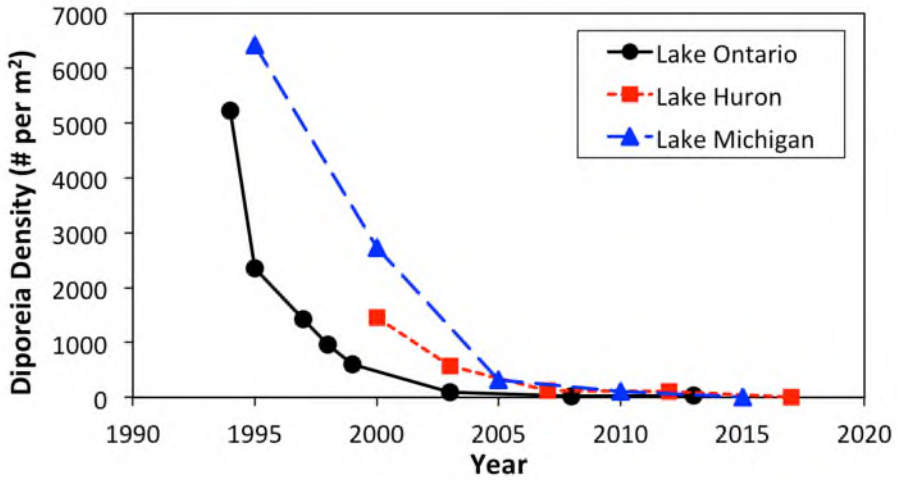


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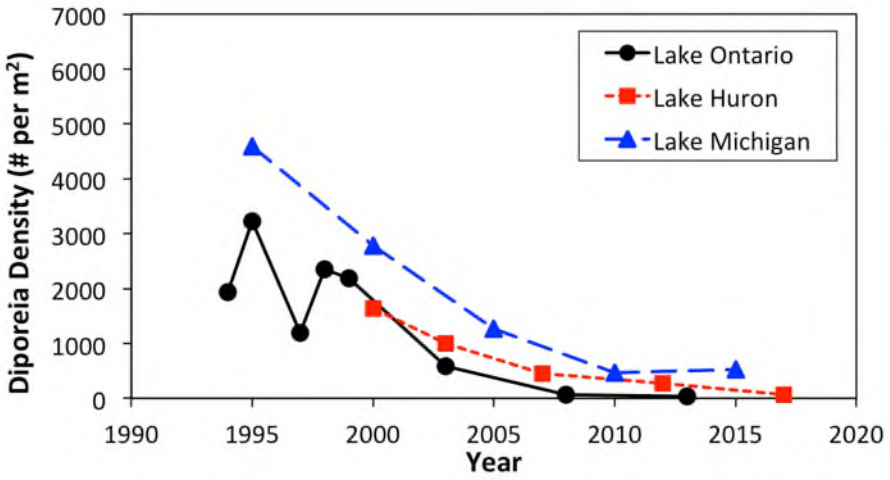


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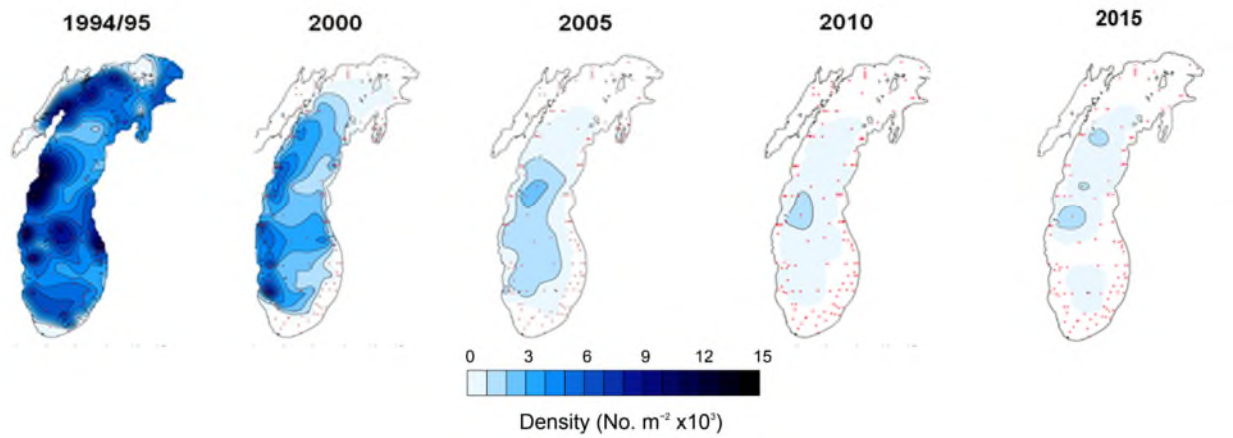


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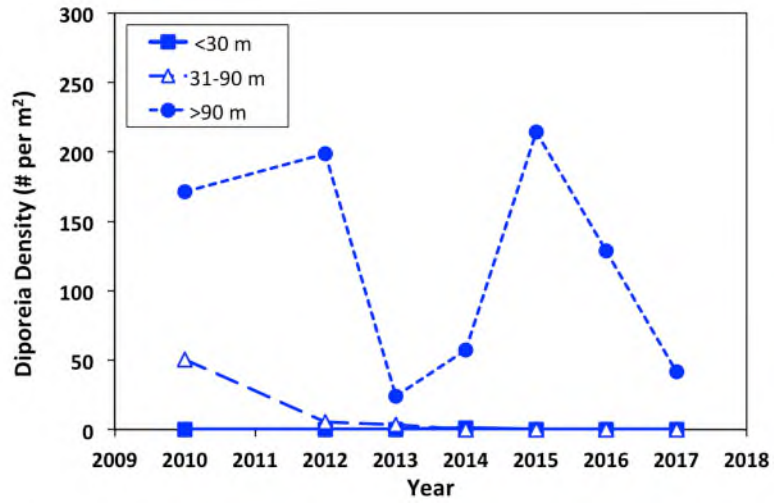


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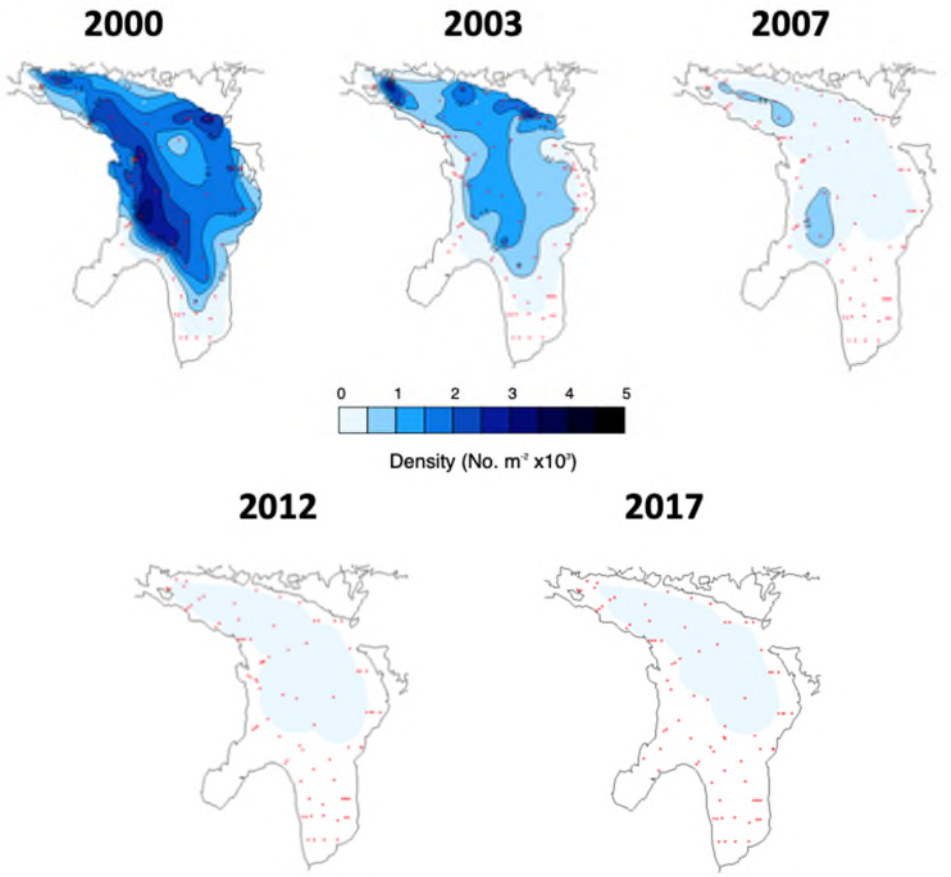


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Data Source: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2018; Karatayev et al. in review.

Sub-Indicator: Lake Sturgeon

Overall Assessment
Status: Poor
Trends:
10-Year Trend: Improving
Long-term Trend: Undetermined
Rationale: There are remnant populations of Lake Sturgeon in each basin of the Great Lakes, 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 some 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 in all lakes remains Poor compared to historical estimates. In many areas, habitat restoration is needed because spawning and rearing habitat has been destroyed or altered, or access to it has been blocked.

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

Lake-by-Lake Assessment
Lake Superior
Status: Poor
10-Year Trend: Improving
Long-term Trend: Improving
Rationale: Populations meet all rehabilitation criteria in three tributaries and most criteria in three other rivers. Reproduction occurs in at least 11 tributaries and in tributaries of Lake Nipigon. Abundance is increasing through natural reproduction and stocking.

Lake Michigan
Status: Poor
10-Year Trend: Improving
Long-term Trend: Undetermined
Rationale: Wild populations of Lake Sturgeon spawn annually and produce measurable recruitment in 8 tributaries to Lake Michigan; the Fox, Oconto, Peshtigo, Menominee, Manistee, Muskegon, Grand and Kalamazoo Rivers. Stocking from streamside rearing facilities is being used to reestablish populations in the Milwaukee, Kewaunee, Cedar and Whitefish Rivers and supplement wild recruitment in the Manistee and Kalamazoo Rivers. Catch rates of both wild and stocked juvenile sturgeon have increased in assessment gill nets in several areas since stocking began suggesting increased recruitment over the past 10 years. Several dams have been removed on the Milwaukee River and passage of sturgeon around the lower 2 dams on the Menominee River is ongoing.

Lake Huron (including St. Marys River)
Status: Poor
10-Year Trend: Improving
Long-term Trend: Undetermined
Rationale: Lake Sturgeon spawning occurs in eight tributaries, the Garden, Mississaugi, Moon, Musquash, Nottawasaga, Spanish and St. Marys River, as well as at the upper St. Clair River. 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 also taking place in the Saginaw River watershed, a tributary to Lake Huron. The contribution of these fish to the Lake Huron basin is being evaluated and will hopefully lead to increased catch per unit effort of juvenile Lake Sturgeon in the future.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Improving
Long-term Trend: Undetermined

Rationale: Lakewide incidental catches since 1992 indicate a possible improvement in their status in Lake Erie. Spawning occurs in the Detroit and St. Clair Rivers, connecting Lakes Huron and Erie and habitat restoration efforts in this system have created an additional six spawning locations over the last ten years. During the spring of 2017, spawning was also detected for the first time in Buffalo Harbor. Lake Sturgeon stocking began in the Maumee River the fall of 2018 which will hopefully lead to increased catch per unit effort of juvenile Lake Sturgeon in the Lake Erie basin.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Poor
10-Year Trend: Improving
Long-term Trend: 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 2016, 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. 2016) 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.

Note: The minimum number of historic Lake Sturgeon populations by lake was:

- Lake Superior: 21;
- 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 from last reporting cycle.
Unchanging: Number of self-sustaining populations of Lake Sturgeon remains unchanged and/or model-generated estimates of abundance remain unchanged and/or catch per unit effort of juvenile Lake Sturgeon from fishery independent surveys remain unchanged from last reporting cycle.
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 from last reporting cycle.
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 that 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 will be 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 of little consequence to the commercial fishery. Sturgeon are now 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 some of these populations are large in number (tens of 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).

<u>Lake Superior</u>
The fish community of Lake Superior remains relatively intact in comparison to the other Great Lakes (Bronte et al. 2003). Historic and current information indicate that at least 21 Lake Superior tributaries supported spawning Lake Sturgeon populations (Holey et al 2000; Quinlan 2007). Successful reproduction was confirmed in the St. Louis River in spring 2011 through capture of larval sturgeon. In the White River, Ontario successful spawning was

implied through the identification of a staging and spawning location (Ecclestone 2013). Lake Sturgeon currently reproduce in 11 Lake Superior tributaries. The Lake Sturgeon Rehabilitation Plan for Lake Superior (Auer 2003) serves as the guiding document for agency activities. Populations in the Sturgeon River, Michigan, Goulais River, Ontario, and Bad River, Wisconsin, meet rehabilitation plan criteria for self-sustaining populations (Auer 2003, Auer and Baker 2007, Quinlan 2007, Quinlan *et al*. 2010). Improvements in assessment techniques have provided better estimates of lakewide abundance (Auer and Baker 2007, Schram 2007, Schloesser *et al*. 2014). The estimated combined annual spawning run population size in the Bad and White Rivers, Wisconsin, was 844 individuals, 666 in the Bad River and 178 in the White River (Schloesser and Quinlan 2011). The estimated number of Lake Sturgeon in annual spawning runs in the Sturgeon River, Michigan range from 350 to 400 adults (Auer and Baker 2007). The abundance of juvenile Lake Sturgeon was estimated at 4,977 (95% CI 3,295-7,517) in Goulais Bay, eastern Lake Superior (Pratt *et al*. 2014). Lake Sturgeon abundance in Goulais Bay is the highest measured in Lake Superior (Schloesser *et al*. 2014). Stocking in the St. Louis River, Minnesota and Ontonagon River, Michigan have resulted in increases in abundance in localized areas. Genetic analysis has shown that Lake Sturgeon populations in most areas of Lake Superior, except eastern waters, are distinct from one another and significantly different from those in the other Great Lakes (Welsh et al. 2008).

Studies and assessments continue in embayments and nearshore waters associated with each of the 21 historic spawning tributaries. In 2011 and 2016, fishery agencies conducted coordinated lakewide Lake Sturgeon index surveys to evaluate trends in abundance and biological characteristics associated with all known current and historic Lake Sturgeon populations. A key study on the Kaministiquia River, Ontario, examined the effect of controlled flow regimes at Kakabeka Falls on the migratory behavior and reproductive response of Lake Sturgeon from 2002-2009 (Friday 2009). There remains a prohibition of commercial harvest of Lake Sturgeon in Lake Superior. Regulation of recreational and subsistence/home use harvest in Lake Superior varies by agency.

Despite progress, challenges remain. Spawning runs are absent in 10 of 21 historic spawning tributaries, and data gathered has provided evidence for only three populations to meet targets identified in the 2003 Rehabilitation Plan and the definition of a self-sustaining population in this document. Overall, Lake Sturgeon abundance remains a small fraction of historical abundance, estimated at 870,000 (Hay-Chmielewski and Whelan 1997). The improving 10-year and long-term trend assessments are based on increasing abundance through natural reproduction and stocking.

Lake Michigan

Remnant populations currently are known to spawn in waters of at least nine tributaries having unimpeded connections to Lake Michigan (Schneeberger *et al.* 2005, Clapp *et al.* 2012, Hayes and Caroffino 2012). 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, M. Donofrio, WDNR and 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 (Harris *et al*. 2017, Hayes and Caroffino 2012, Elliott and Gunderman 2008, Baker 2006). Successful reproduction has been documented in all eight of these rivers, and age 0 juveniles can be captured regularly in many of these river systems. Recruitment estimates in the Peshtigo River indicate that in some years, several hundred fall recruits are produced from that system (Caroffino *et al.* 2007), and research and assessment of the Manistee and Muskegon Rivers populations indicate significant recruitment from these systems in most years (LRBOI 2016, Harris *et al*. 2017). In addition, abundance of spawners in some rivers appears to have increased in the last two decades, and catch rates of wild juvenile fish in assessment gill nets in Lake Michigan has increased significantly over the past 10 years, indicating that increased recruitment has been occurring for several years in some rivers. Adult sturgeon also have been occasionally observed in the lower Manistique, Cedar, Milwaukee, Boardman, and St. Joseph Rivers during spawning times but spawning populations are not known in these rivers or if they exist, abundance is considered low (< 25 adults, Hayes and Caroffino 2012, B. Eggold, 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 upper Menominee River also supports two populations which are separated from each other and from the lower Menominee River population by several dams. These populations also support a closely regulated hook and line fishery in the fall of each year.

The coordinated use of streamside facilities to culture sturgeon for reintroduction and rehabilitation stocking continues to be an important component of 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 efforts to use streamside facilities to

rear and stock 4-6 month old fingerling Lake Sturgeon into 6 rivers (Milwaukee, Kewaunee, Cedar, Whitefish, Manistee and Kalamazoo). Methods follow recommendations of 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 40,000 fall fingerling age-0 Lake Sturgeon have been stocked using these methods. Multiple year classes of juvenile Lake Sturgeon have been captured in assessment gillnets fished in Lake Michigan waters adjacent to stocked rivers, and catch rates have increased since stocking began (Troy Zorn, Michigan DNR, unpublished data; Tom Burzinski, Wisconsin DNR, unpublished data). The intent is to continue these stocking efforts for at least 20 years to establish sustainable founding populations of at least 750 mature adults in each river.

Stocking has also occurred in the upper Menominee River for many years and in portions of the Winnebago system. Though limited recreational harvest is allowed in both the upper Menominee River and the Winnebago system, no harvest is allowed from other Lake Michigan tributaries or from Lake Michigan. Habitat evaluations have been conducted in many sturgeon tributaries within the Lake Michigan basin (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 2 dams on the Menominee River since 2015.

Sturgeon populations in Lake Michigan continue to sustain themselves at a small fraction of their historical abundance. An optimistic estimate of the lakewide adult abundance is less than 10,000 fish, well below 1% of the most conservative estimates of historic abundance (Hay-Chmielewski and Whelan 1997, Haxton *et al.* 2014). While all 9 of the extant populations appear to be self-sustaining, only the Winnebago, Menominee, and Peshtigo River populations are 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.

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. Spawning was documented for the first time in the St. Marys River during the summer of 2018 (E. Roseman, USGS, personal communication) and continues to occur at the mouth of the St. Clair River in southern Lake Huron. 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. 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 an estimated population near 15,000 individuals (U.S. FWS, unpublished data). Research in the Saginaw River watershed in 2005 – 2007 indicated that Lake Sturgeon are no longer spawning in that watershed, although sufficient spawning habitat does exist below the Dow Dam (Midland, MI) on the Tittabawassee River, and below the Hamilton Dam (Flint, MI) on the Flint River (Boase 2007). Also, creation of rock ramps at the Chesaning Dam (Chesaning, MI) on the Shiawassee River and Frankenmuth Dam (Frankenmuth, MI) on the Cass River now allows Lake Sturgeon passage and access to approximately 40 miles (64 kilometers) and 73 miles (117 kilometres), respectively above each former dam site. In the fall of 2017, Lake Sturgeon stocking began in the Saginaw River watershed and is expected to continue for 25 years to rebuild the population in this system. 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.

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 over 7,000 sturgeon tagged in Saginaw Bay, southern Lake Huron, St. Mary's River, Georgian Bay and the North Channel, with relatively large stocks of 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 order to better assess basin-wide juvenile abundance in Lake Huron, eleven tributaries were sampled in 2012 and 2013 using a protocol successful in capturing juvenile Lake Sturgeon in Lake Superior (Schloesser *et al.* 2014). Nine of the Lake Huron tributaries were sampled in the province of Ontario and two in the state of Michigan. Juvenile Lake Sturgeon were captured at four of these tributaries including the Blind, Echo, Serpent, and Spanish Rivers all located in the North Channel. The development of a juvenile index to assess the status of Lake Sturgeon in Lake Huron continues to be of interest to management agencies.

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 four 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 (Hondorp *et al.* 2015).

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, St. Clair, Spanish, and Mississaugi Rivers) exist in Canadian waters. The improving 10-year trend assessment is based on a new stocking program initiated in the Saginaw River and improved river connectivity.

Lake Erie

Spawning has been identified at eight locations in the connecting waters between Lakes Huron and Erie (Caswell *et al.* 2004; Manny and Kennedy 2002; Roseman *et al.* 2011) and was also detected in Buffalo Harbor during the spring of 2017 (Neuenhoff *et al.* 2018). Three new spawning sites have been identified in the St. Clair River resulting from artificial reef restoration projects aimed at removing the loss of fish and wildlife habitat beneficial use impairment (BUI) in this river (E. Roseman, USGS, personal communication). Tag recovery data and telemetry research indicate that a robust Lake Sturgeon stock of approximately 11,000 fish reside in the North Channel of the St. Clair River and Lake St. Clair (Thomas and Haas 2002, Chiotti *et al.* 2013). The spawning population in the upper St. Clair River near Port Huron, Michigan contains one of the largest populations of Lake Sturgeon in the Great Lakes with an estimated population near 15,000 individuals (U.S. FWS, unpublished data). The North Channel of the St. Clair River, Anchor Bay in Lake St. Clair, the Detroit River (East of Fighting Island), and the western basin of Lake Erie have been identified as nursery areas as indicated by consistent catches in commercial and survey fishing gears. 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 done in the upper Niagara River to confirm to these reports. In the central and eastern basins of Lake Erie, Lake Sturgeon are scarcer with only occasional catches of sub-adult or adult Lake Sturgeon in commercial and research fishing nets. Lake Sturgeon stocking began in the Maumee River the fall of 2018 which will hopefully lead to increased catch per unit effort of juvenile Lake Sturgeon in the Lake Erie basin. An observed concentration of sturgeon in the spring of 2009 and subsequent sampling through 2015 in Buffalo Harbor has yielded maturing and sexually mature adult and sub-adult Lake Sturgeon. Sidescan sonar imagery for a roughly seven square mile (18 square kilometres) section of Buffalo Harbor has been collected to develop a categorical habitat map intended to identify potential sturgeon spawning habitat.

As noted above, 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 four 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.* 2015). In Buffalo Harbor, a total of 19 Lake Sturgeon were implanted with acoustic transmitters, nine of which were equipped with satellite transmitters, in the spring of 2015. To date GLATOS has provided nearly five million detections for sturgeon acoustically tagged in Buffalo Harbor.

In an effort to assess basin-wide juvenile abundance in Lake Erie, 14 tributaries were sampled in 2013 and 2014 using a protocol successful in capturing juvenile Lake Sturgeon in Lake Superior (Schloesser *et al.* 2014). A total of 176 nets were set and a total of 15 Lake Sturgeon were captured, all in the St. Clair – Detroit River System.

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 historic 15 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 a new stocking program initiated in the Maumee River, an increase in spawning locations in the SCDRS and Buffalo Harbor, and improved river connectivity.

Lake Ontario/ Upper St. Lawrence River

The numbers of mature sturgeon are not well quantified for most of the spawning areas surrounding Lake Ontario; however, data is available to assess Lake Sturgeon status. 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 (D. Gorsky, personal communication). Also, numbers of 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 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).

Between 1993 and 2016, NYSDEC in collaboration with U.S. FWS, have stocked 123,045 (0 to 14,047 fish per year) 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, 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 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 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 appears to be 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 sexual 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.

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 maybe be present (O'Connor et al. 2016). Sea lamprey wounding rates and potential impacts to juvenile and adult Lake Sturgeon should continue to be monitored (Dobiesz et al. 2018).
- 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 (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			
Clarifying Notes: Since the status assessment is highly dependent upon the number of self-sustaining populations within each lake basin, the source of the data for the historical population status is currently being assessed for Lakes Huron and Erie. For some of the Great Lakes, the 4. "Geographic coverage and scale of data" may not be appropriate.				

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 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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List of Figures

Figure 1. Historic Lake Sturgeon harvest from each of the Great Lakes.
Source: Baldwin *et al*. 1979
Figure 2. Historic distribution of Lake Sturgeon.
Source: Zollweg *et al*. 2003
Figure 3. Lake Sturgeon population status in Lake Superior, 2012.
Source: Lake Superior Lake Sturgeon Work Group, Great Lakes Fishery Commission
Figure 4. Lake Sturgeon population status in Lake Michigan, 2012.
Source: Lake Michigan Lake Sturgeon Task Group, Great Lakes Fishery Commission
Figure 5. Lake Sturgeon population status in Lake Huron, 2012.
Source: Lake Huron Lake Sturgeon Task Group, Great Lakes Fishery Commission
Figure 6. Lake Sturgeon population status in Lake Erie, 2012.
Source: Lake Erie Lake Sturgeon Working Group, Great Lakes Fishery Commission
Figure 7. Lake Sturgeon population status in Lake Ontario, Ottawa River and the St. Lawrence River, 2012.
Source: New York Lake Sturgeon Working Group, and Tim Haxton, OMNRF

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State of the Great Lakes 2019 Technical Report

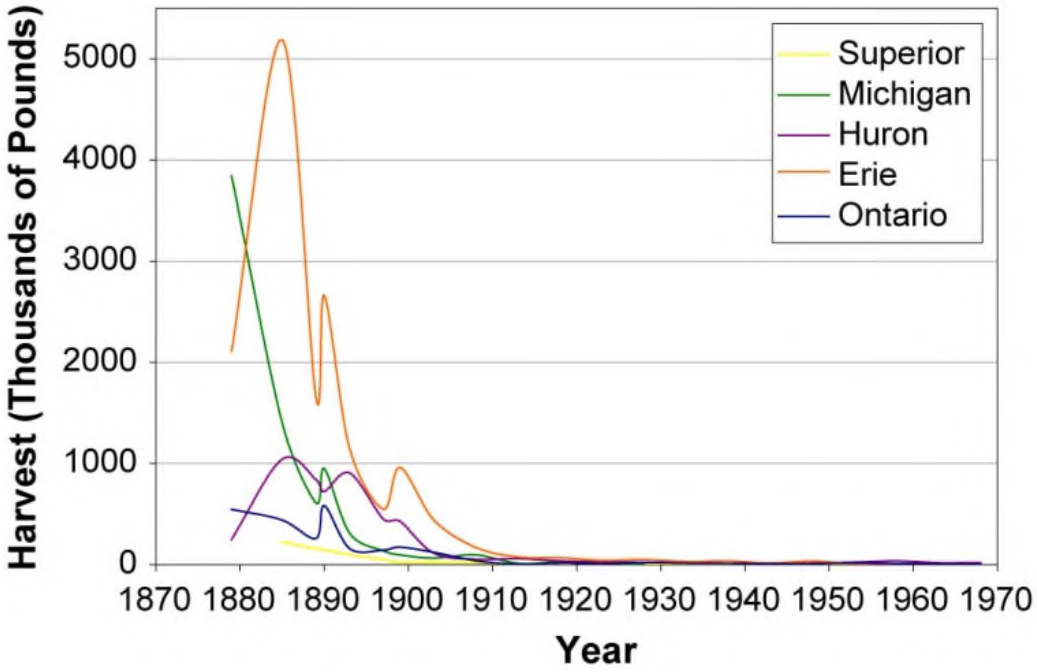


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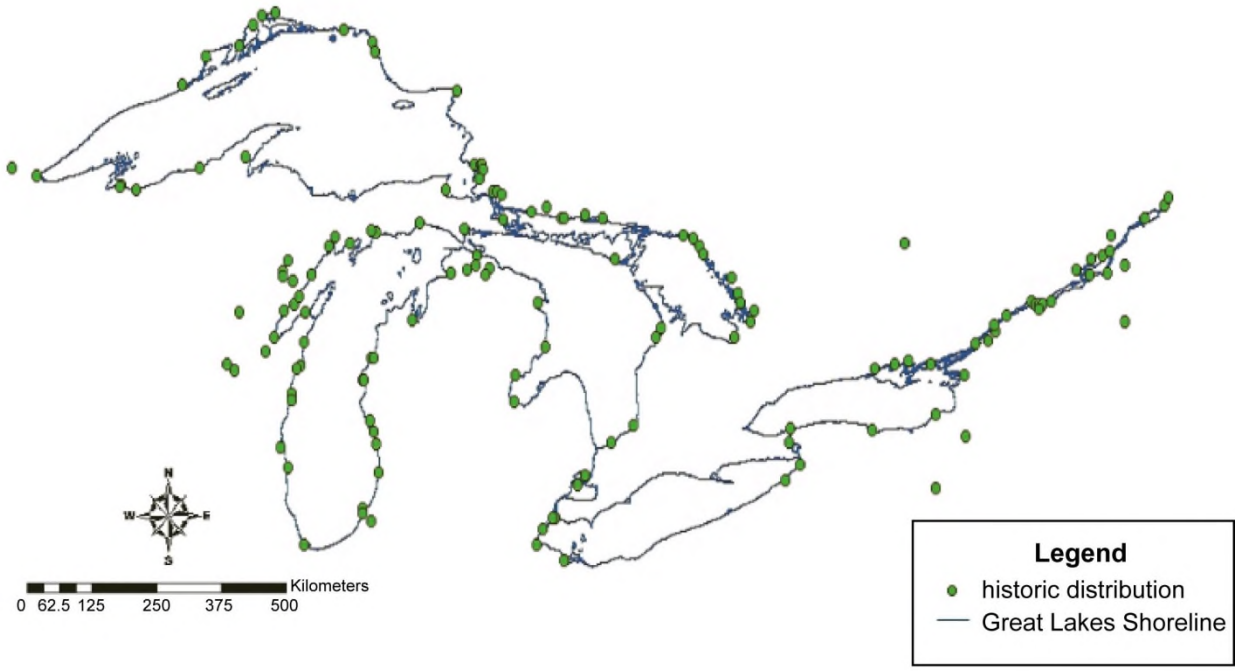


Figure 2. Historic distribution of Lake Sturgeon.
Source: Zollweg *et al*. 2003

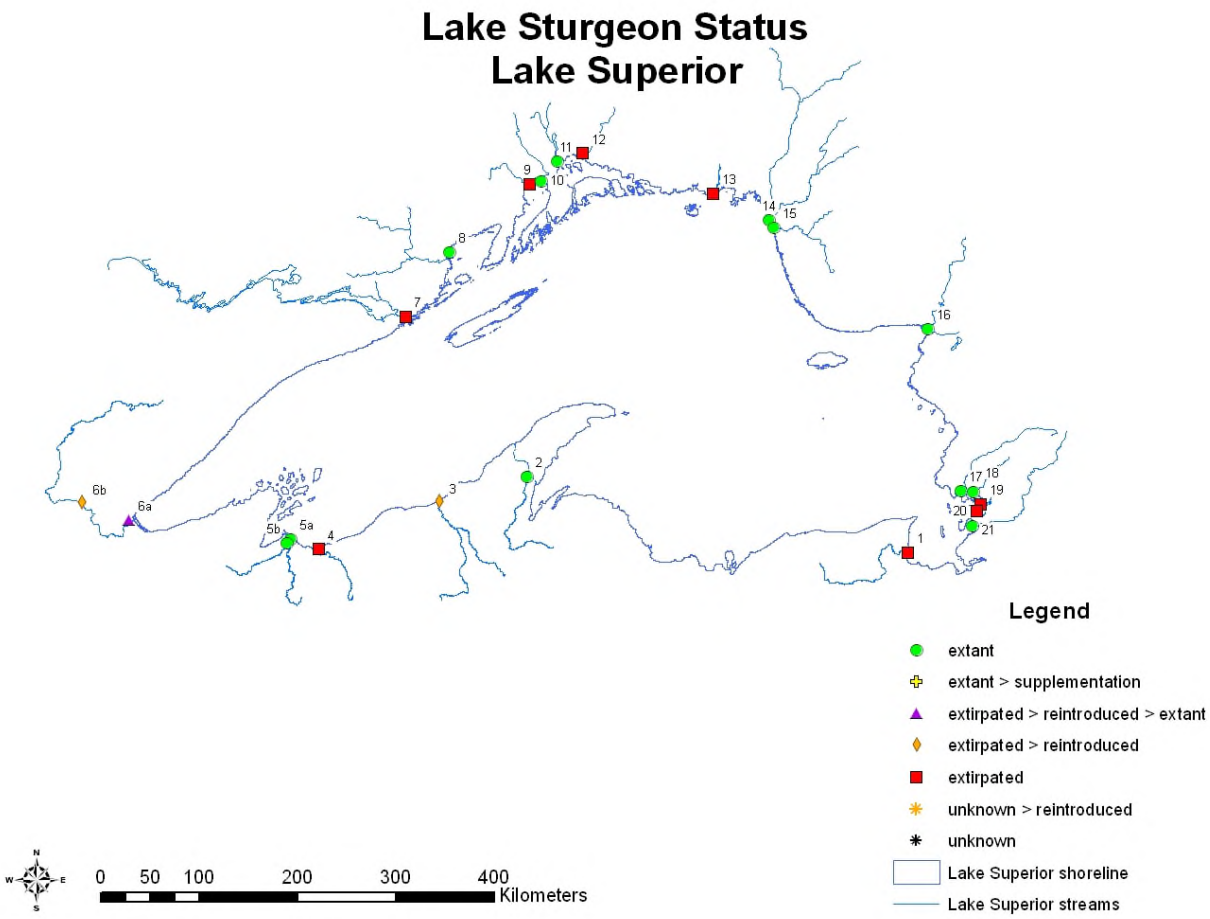


Figure 3. Lake Sturgeon population status in Lake Superior, 2012.
Source: Lake Superior Lake Sturgeon Working Group, Great Lakes Fishery Commission

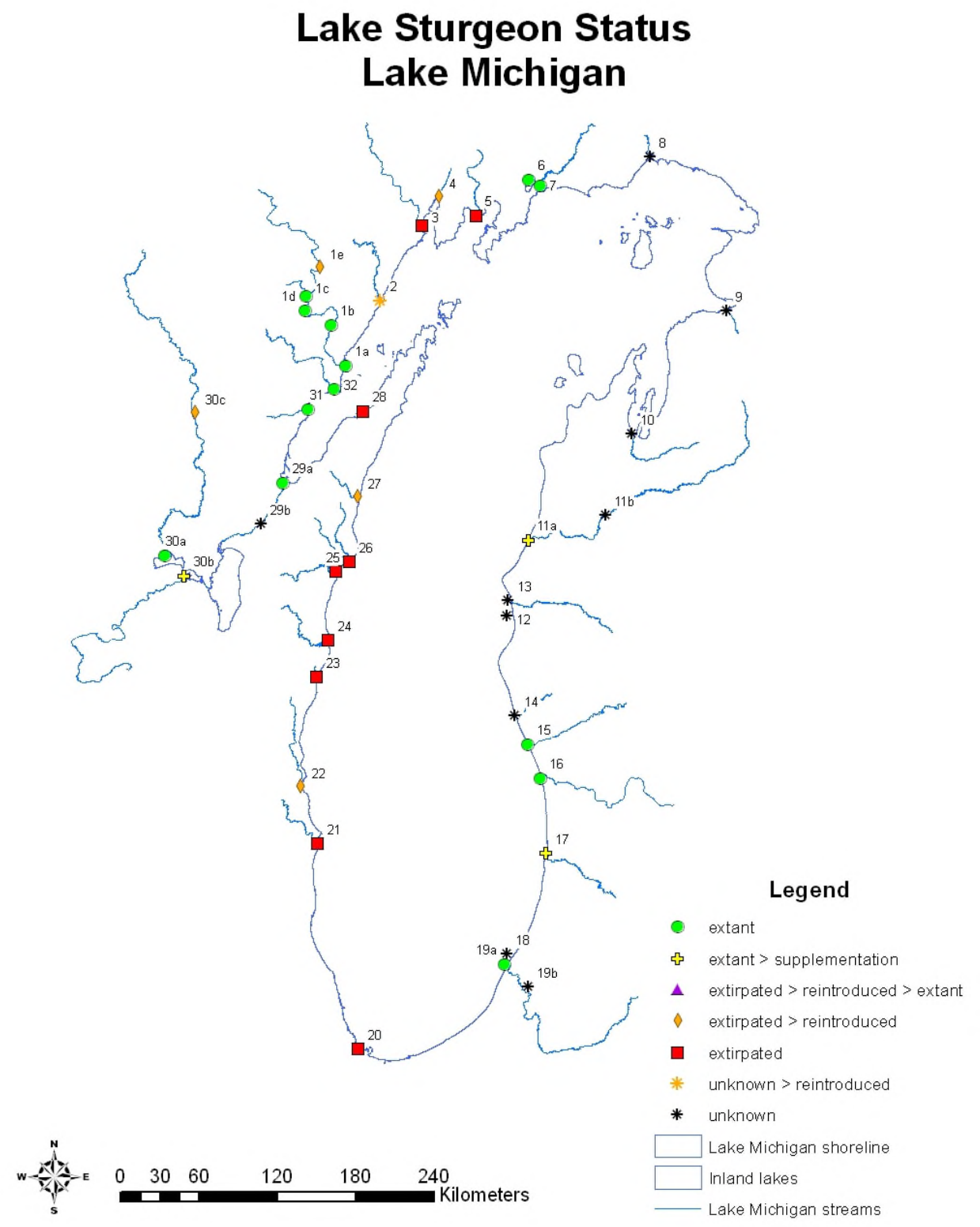


Figure 4. Lake Sturgeon population status in Lake Michigan, 2012.
Source: Lake Michigan Lake Sturgeon Task Group, Great Lakes Fishery Commission

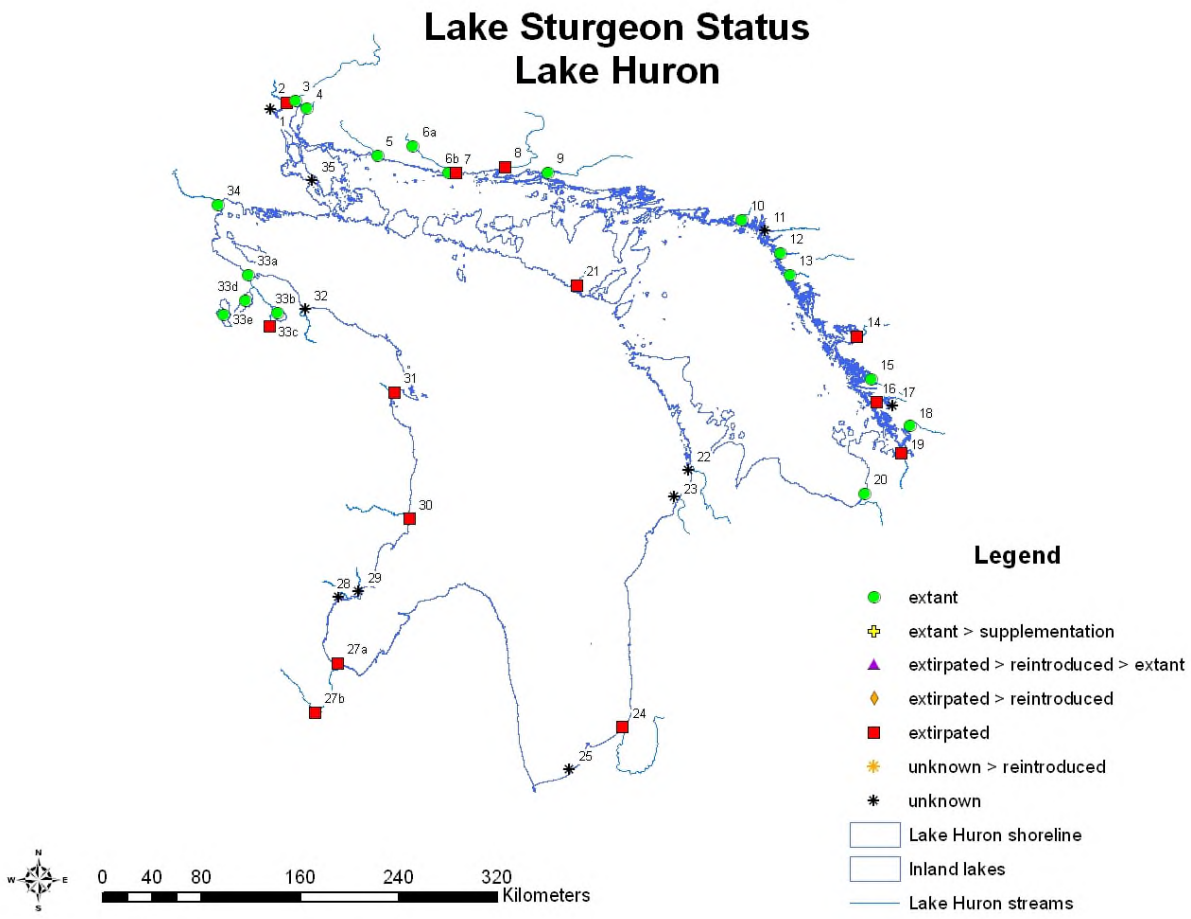


Figure 5. Lake Sturgeon population status in Lake Huron, 2012.
Source: Lake Huron Lake Sturgeon Working Group, Great Lakes Fishery Commission

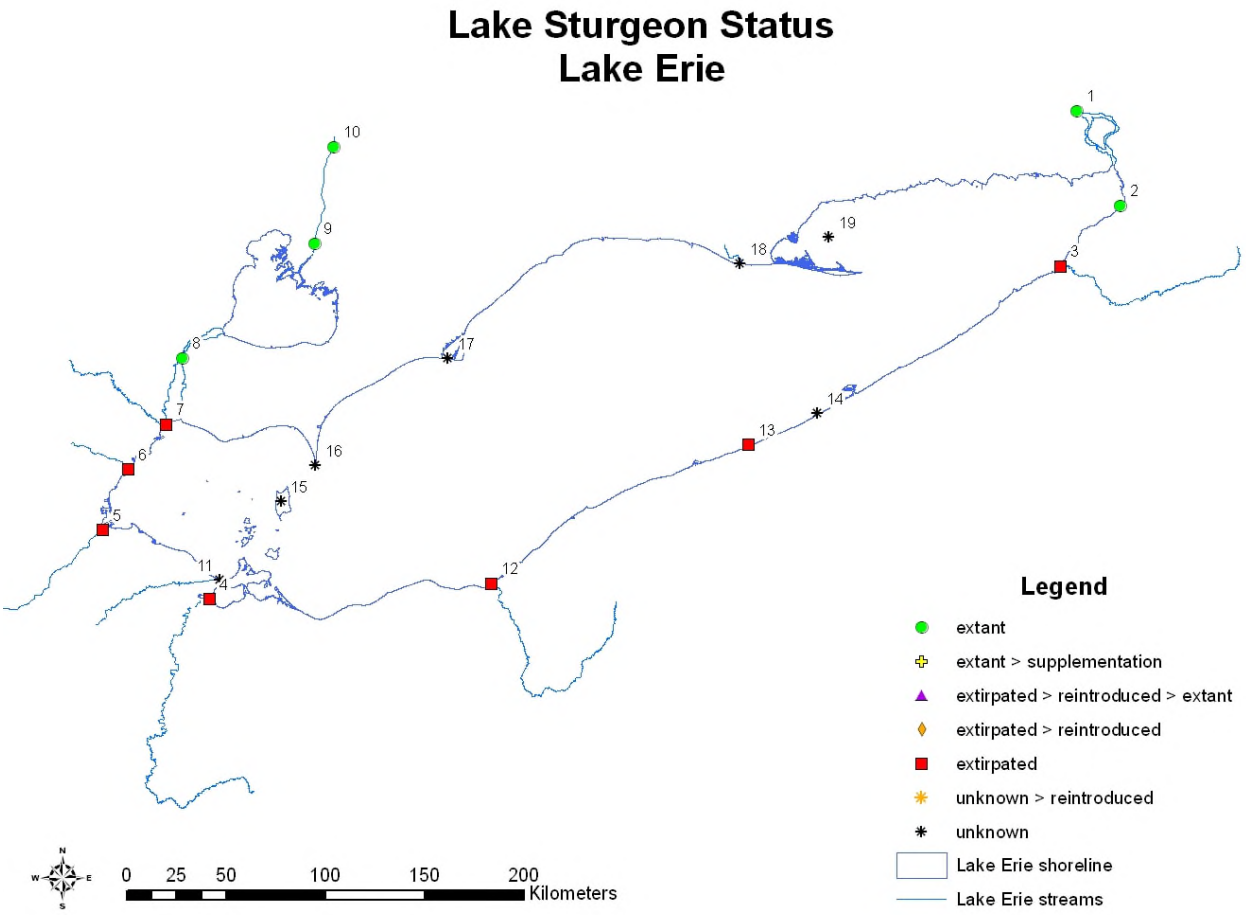


Figure 6. Lake Sturgeon population status in Lake Erie, 2012.
Source: Lake Erie Lake Sturgeon Working Group, Great Lakes Fishery Commission

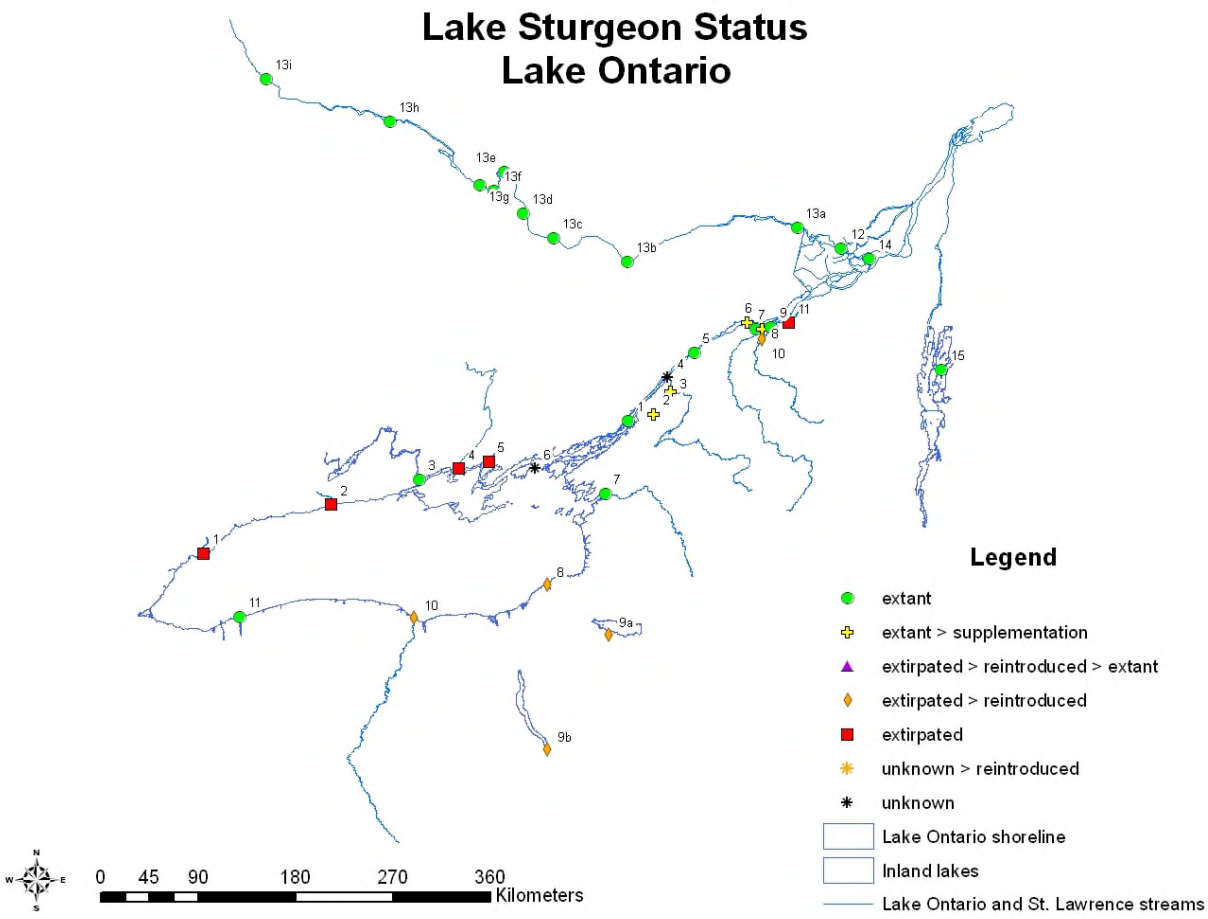


Figure 7. Lake Sturgeon population status in Lake Ontario, Ottawa River and the St. Lawrence River, 2012. Source: New York Lake Sturgeon Working Group, and Tim Haxton, OMNRF

Sub-Indicator: Prey Fish
Open water

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Unchanging
Long-term Trend (1973-2017): Undetermined
Rationale: Great Lakes prey fish community status remains 'Fair' based on diversity and percent native species, but individual lake status varied. Both diversity and percent native metrics were classified as 'Good' in Lake Superior, but based on both metrics, the status is 'Poor' overall in Lake Ontario (Table 1). Lakes Huron and Michigan were both 'Fair' (Table 1). In Lake Erie, diversity remained 'Fair', but the proportion native species shifted to 'Poor', resulting in an overall conservative classification of 'Poor' (Table 1). Four of the five lakes had the same overall status as the previous reporting period, but Lake Erie shifted from 'Fair' to 'Poor'.

At the ten-year timescale, lake-specific trends were 'Unchanging' in three lakes and 'Deteriorating' in two lakes (Table 2). The trend for all lakes was therefore categorized as 'Unchanging'. It is important to recognize six of the ten individual prey fish metrics did not trend up or down over the past ten years (Table 2). In Lake Erie, diversity and percent native were both 'Deteriorating' and the Lake Michigan diversity was noted as 'Deteriorating'. The only 'Improving' trend was observed in the Lake Ontario where the percent of native species significantly increased from two to four percent of the total catch due to increased relative abundance of native Deepwater Sculpin (Weidel et al., 2017b).

Long-term prey fish trends varied substantially with categorizations of 'Improving', 'Deteriorating', 'Undetermined' and two lakes were 'Unchanging' (Table 2). Because many of the long-term trends were in opposite directions the overall classification for the long-term trend was 'Undetermined'. In Lake Superior, both metrics have 'Improving', in Lake Michigan diversity is 'Improving', and in Lake Huron the percent native metric is 'Improving' as non-native Alewife declined, and the relative importance of native Bloater increased. Alternatively, Lake Ontario long-term trends are 'Deteriorating' as the proportion of Alewife in catches has increased. No long-term trends were detected in either of the Lake Erie metrics.

Prey fish community changes are driven by changing ecosystem conditions including productivity changes, fluctuating predator composition and density, increasing water clarity, increasing water temperatures, and non-native species effects. While these driving factors are changing in similar directions across the region, because lakes are unique in their nutrient concentrations, morphometry, hydrology, and fish communities, the prey fish communities in each lake respond differently to changes in ecosystem drivers (Figure 1).

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 (1978- 2017): Improving
Rationale: Lake Superior is recognized as the least anthropogenically-altered Great Lake, and its prey fish community metrics reflect that more pristine state. Native species dominate the relatively diverse prey fish community and both metrics classify the community as 'Good', like the previous reporting period (Table 1). Unlike the other Great Lakes that have a variety of non-native prey fish, Rainbow Smelt are the only non-native species that substantially contributes to the Lake Superior bottom trawl catch, however their abundance has declined (Vinson et al., 2017). Over the past ten years Lake Superior prey fish metrics have been 'Unchanging', but since 1978, the diversity index

and proportion native have increased resulting in an 'Improving' long-term trend (Table 2). While the metrics evaluated herein suggested trends commensurate with stated objectives, there are concerns over population declines of important species like Cisco that serve as prey fish and also support commercial fisheries (Pratt et al., 2011). Recently, the Cisco population has experienced lower than average recruitment (Vinson et al., 2017) and elevated mortality (Pratt et al., 2011).

Lake Michigan
Status: Fair
10-Year Trend: Deteriorating
Long-term Trend (1973- 2017): Unchanging
Rationale: The overall Lake Michigan prey fish community is categorized as 'Fair', with individual indicators as 'Good' (native species) and 'Fair' (diversity). This is slightly improved from previous period's status ('Fair, 'Fair'). Michigan's prey fish diversity has fluctuated but is 'Improving' over the whole time series (Table 1 and 2). Interestingly, in the past ten years, diversity has shown 'Deteriorating' trends as trawl catches have been more dominated by a single species, Bloater (Bunnell et al., 2017, Table 2).The percent native species has fluctuated over time, but no trend was apparent over the past 10 years or the whole time series (Figure 1). The observed fluctuations in the percent native is driven in part from substantial Bloater year class variability and variable Round Goby catches (Figure 1). While prey fish diversity has been "Improving" over the longer time scale, the lack of directional change in the percent native metrics results in an 'Unchanging' long-term trend (Table 2).

Lake Huron
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1976-2017): Undetermined
Rationale: The Lake Huron prey fish community continues to be categorized as 'Fair', but the percent native species was 'Improving' over the long-term scale (Tables 1 and 2). The prey fish community was dominated by non-native Alewife and Rainbow Smelt in the 1970s through 2000, but Alewife declined in the early 2000s and the community shifted to be dominated by native Bloater, similar to Lake Michigan (Bunnell et al., 2017; Riley et al., 2017). Interestingly, this shift to a more-native community has also resulted in a diversity decline, resulting in an overall long-term trend of 'Undetermined' (Table 2). At the 10-year scale the fish community was 'Unchanging' (Table 2). Long-term, consistent observations on prey fishes in the St Mary's River were not available for comparison.

Lake Erie
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1990-2017): Unchanging
Rationale: The Lake Erie prey fish community status was conservatively classified as 'Poor' since the percent native was characterized as 'Poor' and the diversity was classified as 'Fair' (Table 1). Both the diversity and percent native indices significantly declined in the past 10 years. Much of that change was due to declines in Emerald Shiners, Yellow Perch, Spottail Shiner and Trout-perch. As the native species declined, Rainbow Smelt, Gizzard Shad and White Perch made up a larger proportion of a less-diverse prey fish community (Figure 1). At the long-term scale, both metrics were 'Unchanging' (Table 2). As the most productive Great Lake with the most diverse habitats Lake Erie supports some of the Great Lake's largest commercial and sport fisheries. Designations here in are based on metrics that can be applied a lake-wide scale across all Great Lakes and may not accurately reflect regional prey fish population trends. Long-term, consistent observations on prey fishes in the St Clair-Detroit River were not available for comparison.

Lake Ontario
Status: Poor
10-Year Trend: Unchanging
Long-term Trend (1978 - 2017): Deteriorating
Rationale: Lake Ontario prey fish percent native was 'Poor' and the diversity was 'Fair', resulting in a 'Poor' overall status (Table 1). Native species status was poor, but the 10-year trend was 'Improving' with native species component increasing from two to four percent between reporting periods (Table 2). The increase was due the growing population of Deepwater Sculpin, a species once thought extirpated, which has come back on its own over the past 20 years (Weidel et al., 2017b). At the 10-year time scale Lake Ontario prey fish trends were 'Unchanging'

since percent native 'Improving' but diversity remained 'Unchanging' (Table 2). Over the entire time series both metrics were classified as 'Deteriorating' (Table 2, Figure 1).

Status Assessment Definitions
Good: The percentage 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 percentage 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 percentage 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 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'.

Trend Assessment Definitions
Trends at 10-year and long-term timescales are quantified using a linear model 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 time in both metrics is positive, p-value of 0.10 or less.
Unchanging: Neither slopes are significant, below a p-value of 0.10.
Deteriorating: The slopes for time in both metrics is negative with a p-value of 0.10 or less.
Undetermined: If data is not sufficient, or if the metric trends differ (e.g. 'Improving' and 'Deteriorating')

Endpoints and/or Targets
Endpoints or targets for prey fish are based on the lake-specific Fish Community Objectives, which are developed independently in each lake, by the respective Lake Committees within a management framework overseen by the Great Lakes Fishery Commission (GLFC). This sub-indicator uses two metrics that quantify prey fish community diversity and percentage of native species, which are commonly mentioned elements among the lake's different Fish Community Objectives.

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 – "M*aintain a diversity of planktivore (prey) species at population levels matched to primary production an d 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: Forage Fish Objective – *"Maintain a diversity of forage fishes to support terminal predators and to sustain human use"* (Ryan et al. 2003).

Lake Ontario: Offshore Pelagic Zone Objective- "*Increase prey-fish diversity – maintain and restore a diverse prey-fish community that includes Alewife, Lake Herring, Rainbow Smelt, Emerald Shiner, and Threespine Stickleback. Status and trend indicators are 1) maintaining or increasing populations and increasing species diversity of the pe-*

lagic prey fish community including introduced species (Alewife, Rainbow Smelt) and selected native prey fish species (Threespine Stickleback, Emerald Shiner and Lake Herring); and 2) increasing spawning populations of Lake Herring in the Bay of Quinte, Hamilton Harbor, and Chaumont Bay"(Stewart et al., 2017).

Sub-Indicator Purpose
This sub-indicator quantifies 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.

Ecosystem Objective
Maintain diverse prey fish populations to support healthy, productive predator populations and maintain or increase native prey fish populations as stated in Fish Community Goals and Objectives (FCGO) for each lake (noted above). This sub-indicator also relates to the 1997 Strategic Great Lakes Fisheries Management Plan Common Goal Statement for the Great Lakes Fisheries Agencies. 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
This sub-indicator relies on time series of two metrics that describe the prey fish communities. One metric quantifies prey fish community diversity using the Shannon diversity index and compares mean values during different time periods to the maximum value in the time series. The other metric quantifies the proportion of native prey fish in the total prey fish catch. Data used to calculate those metrics are from bottom trawl surveys that have been generally conducted annually and consistently across time, are appropriate for catching a range of prey fish species. In all lakes, metrics are based on biomass from bottom trawl catches, except for Lake Erie where data are represented by density. 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.

Ecological Condition
Lake Superior, Status: Good, 10-year trend: Unchanging
Native species dominate the relatively diverse Lake Superior prey fish community (Figure 1). Unlike the other Great Lakes that have a variety of non-native prey fish, Rainbow Smelt are the only non-native species that substantially contributes to the Lake Superior prey fish community. Changes in the metrics describing the Lake Superior prey fishes are primarily driven by fluctuations in the coregonid populations and Rainbow Smelt which are known to exhibit variable year class strength (Vinson et al., 2017). The recently observed declines in abundance of these prey fish populations may be the result of lower than average year class strength in combination with predation from siscowet Lake Trout (Vinson et al., 2017).

Lake Michigan, Status: Fair, 10-year trend: Deteriorating
Based on the metrics considered in this sub-indicator, Lake Michigan prey fish status remains "Fair", however trends suggest the community is changing in ways that are inconsistent with the stated fish community objectives (Tables 1 and 2). The sharp decline in proportion of native species during the previous reporting period was primarily driven by decreased proportions of Bloater and increased proportions of non-native Round Goby (Figure 1). Recently, declines in Lake Michigan prey fish abundance (primarily Alewife) have caused resource management to reduce native and sport fish stocking levels in an effort to reduce predation on prey fish populations and maintain sport fisheries (Lake Michigan Committee, 2016; Tsehaye et al., 2014). While dreissenid mussels are often implicated in food web productivity reductions, rigorous evaluation finds mineral nutrient concentrations and loads from the Lake Michigan watersheds along with climate, are more likely driving productivity dynamics (Rowe et al., 2017; Warner and Lesht, 2015). Lake Michigan spring phosphorus concentrations (~ 3-4 µg/L) are well below the Great Lakes Water Quality target of 7.5 µg/L (Dove and Chapra, 2015) and these low phosphorus concentrations are consistent with prey fish productivity declines.

Lake Huron *Status: Fair, 10-year trend: Unchanging*
Across the period of observation, the Lake Huron prey fish catches suggest that lake's community has been the most

dynamic in the Great Lakes (Figure 1). When trawl surveys began in the 1970s, the prey fish catch was dominated by non-native Alewife and Rainbow Smelt (Riley and Adams, 2010). In the early 2000s the community shifted to one dominated by native Bloater after Alewife populations declined (Dunlop and Riley, 2013; Riley and Adams, 2010). This change 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). Interestingly, this shift towards a more native community has also resulted in an overall decline in prey fish diversity as measured by the index used in this analysis (Figure 1). The diversity decline is also partly driven by the decline of Deepwater Sculpin in bottom trawls. This native benthic species historically comprised approximately 5% of the community biomass but has declined to 1% of the total (Riley et al., 2017).

Lake Erie *Status: Poor, 10-year trend: Deteriorating*
The Lake Erie prey fish community has experienced substantial changes in the past 10 years (Figure 1). Both the diversity and percent native metric significantly declined due to declining catches of native species such as Emerald Shiners, Yellow Perch, Spottail Shiner and Trout-perch (Wiemer et al., 2018). Above-average reproductive success of Walleye has been observed in the past eight years such that Lake Erie predator abundance is currently high and could account for the observed trends in Lake Erie prey fishes. Another possible factor influencing native prey fish declines involves human changes to river hydrology. For instance hydrologic changes in the Niagara River have been implicated in Emerald Shiner population declines (Allen et al., 2017). Unfortunately, river-oriented prey fish time series that are comparable to lake-based surveys are not currently available.

Lake Ontario, Status: Poor, 10-year trend: Unchanging
The Lake Ontario prey fish community continues to be dominated by Alewife, which support a diverse sport fishery (Lantry and Eckert, 2018). The dominance of a single species ultimately results in low and relatively unchanging time series of prey fish diversity and the percent of native species. Since the 1970's the proportion of Alewife in the catches have increased from 50-65% to more recently over 96% of the prey fish community. This change was primarily driven by a decrease in the relative abundance of non-native Rainbow Smelt. Benthic prey fishes were dominated by native Slimy Sculpin when surveys began (1970s) but that community is now primarily composed of the rebounding native Deepwater Sculpin and non-native Round Goby. Increasing Deepwater Sculpin abundance over the past 10 years has resulted in slight improvements to percent native and diversity metrics in Lake Ontario. Lake Ontario management is actively attempting to restore native species and increase prey fish diversity. Lake Michigan Bloater have been reared and stocked in Lake Ontario since 2012. In addition, experiments have stocked progeny from Cisco collected in eastern Lake Ontario embayments into southern shore embayments where the species was believed to historically spawn.

Linkages
Prey fishes are the dominant fish biomass in Great Lakes food webs (Cox and Kitchell, 2004; Rogers et al., 2014; Stewart and Sprules, 2011). This fish community is closely linked to Great Lakes invertebrates and primary producers, which can drive prey fish abundance and community composition. Similarly, as the prey of stocked and naturally-reproduced fish and avian predators, prey fish are closely linked to higher trophic levels in Great Lakes food webs. Physical process such as water temperature, water clarity, and geomorphology influence prey fish behavior, growth dynamics, and spawning success.

- Nutrients inputs and internally recycled nutrients are the primary driver of Great Lakes primary productivity and therefore are also a dominate driver of prey fish biomass and community composition. Nutrient levels often dictate the density of invertebrates that support prey fish populations. Climate also influences productivity at given nutrient levels by changing the volume or depth of the warm surface layers within the lakes (Rowe et al., 2017; Warner and Lesht, 2015).
- Zooplankton are the primary food of pelagic prey fishes
- Benthic invertebrates are primary food of some prey fish populations and are especially important in nearshore habitats.
- *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 fish (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)
- Water temperature influence prey fish energetics and behavior that in turn influence prey fish abundance and community dynamics
- Water levels can influence the availability of different habitats which link to prey fishes by changing spawning and nursery habitat availability.
- Lake Trout and walleye are two historically important native predators of Great Lakes prey fish. Lake Trout, Walleye and the other Great Lakes piscivores are a dominate driver of prey fish biomass and community composition.

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 Limitations

Inconsistency in prey fish survey methods complicates metric tabulation and potentially biases the interpretation of prey fish status and trends. Prey fish survey designs differ because the five lakes vary in size, bathymetry, substrate type, and fish communities. Additionally, the information needs differ among the respective lake management agencies. Constraining prey fish observations to a single collection method (bottom trawls) helps to maintain consistency in our comparisons, but differences in trawl survey designs may still influence our results. 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 may be important to consider when interpreting these results.

Bottom trawl surveys are often thought to be most effective at measuring relative changes of a species over time in a lake, but they may not catch all species in equal proportion to their abundance in the environment. Specifically, pelagic species may be underestimated by bottom trawling relative more demersal, or bottom oriented species (Yule et al., 2008). For example in Lake Superior, acoustic surveys estimated substantially greater biomass of pelagic *Coregonus kiyi* and *Coregonus artedii*, than bottom trawls estimates from the same year (Yule et al., 2013). Similarly in Lake Ontario acoustic and midwater trawl observations suggest Cisco were a larger part of the fish community than bottom trawls indicated. (Holden et al., 2017a; Weidel et al., 2017a). Alewife may also be similarly underrepresented in bottom trawl catches relative to their abundance in the prey fish community. In Lake Michigan, fall-collected bottom trawl Alewife biomasses are lower than summer-collected acoustic estimates from the same year (Bunnell et al., 2017; Warner et al., 2017). Both bottom trawl and acoustic Alewife estimates are lower than Alewife biomass estimates based on the consumptive demand of Lake Michigan Chinook Salmon (Tsehaye et al., 2014). In Lake Ontario multiple trawl surveys are conducted and spring, Alewife biomass estimates are consistently ten times higher than fall-based estimates (Holden et al., 2017b; Weidel et al., 2017a). Bottom trawl surveys have provided critical long-term information on Great Lakes prey fish dynamics. Efforts to understand potential biases in community metrics based on all available surveys will be critical for improving cross-basin prey fish comparisons such as those presented in this report.

The classification metrics used in this report were chosen because they were commonly referenced among the different lake Fish Community Objectives. Categorizing Great Lake's prey fish community dynamics into single metrics is at best imperfect and incomplete but serves as a starting point from which to understand differences among the lakes. For instance, we use the Shannon index to quantify "diversity", however there many different indices that quantify diversity and while the Shannon index is often used it is not without bias (Hurlbert, 1971; Jost, 2006). In addition, theoretical or widely-agreed upon thresholds for what constitutes a prey fish community metric as 'good', 'fair', or 'poor' do not exist. Categorizing prey fish communities is also subjective based on individual's preferences and the different ecosystem services the prey communities provide. For instance, in some of the lakes non-native Alewife dominate the fish community and likely reduce the reproductive success of native prey fishes (Fitzsimons et al., 2007; Madenjian et al., 2008). Those same Alewife can however support thriving, diverse sport fishery and associated local communities (Lantry and Eckert, 2018).

Neither prey fish abundance nor predator-balance are used as a graded metrics in this report, however actions are often influenced by these metrics. In response to prey fish biomass and density declines such as those illustrated in Figure 2, predator stocking levels have been altered by resource management agencies across the basin (Great Lakes Fishery Commission Lake Michigan Committee, 2012; Great Lakes Fishery Commission Lake Ontario Committee, 2016; Lake Michigan Committee, 2016; New York State Department of Environmental Conservation, 2018). Ecosystem ecology illustrates that prey fish biomass is strongly dependent on lake nutrient concentration and piscivores fish abundance (Carpenter et al., 2001; Downing and Plante, 1993). Within the Great Lakes, resource management has reduced nutrient inputs and restored piscivorous fish populations. Dove and Chapra (2015) recently quantified the consistent declines in phosphorus concentration across most of the Great Lakes and stocking continues to be a frequently used management tool (http://www.glfc.org/fishstocking/). These actions have contributed to prey fish biomass declines, confounding the utility of abundance metrics for the prey fish sub-indicator.

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List of Tables

Table 1. Status of Great Lakes prey fish communities in the current period (2015-2017) and previous reporting period (2010-2014) based on the percent native and the Shannon diversity index. To attain as status of 'good', 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. The overall status is conservatively categorized as the lower of the two metrics.
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.

List of Figures

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
Figure 2. Prey fish biomass or density trends based on annual bottom trawl surveys.
Source: Data primarily derive from bottom trawl surveys conducted by U.S. federal and state and Canadian provincial agencies

Last Updated:
State of the Great Lakes 2019 Technical Report

Lake	Percent Native				Diversity Index				Overall
	Current	Previous	Max.	Status	Current	Previous	Max.	Status	Status
Superior	79%	86%	96%	Good	2.0	2.0	2.2	Good	Good
Michigan	85%	28%	91%	Good	1.1	1.3	1.7	Fair	Fair
Huron	69%	75%	90%	Fair	0.8	0.8	1.6	Fair	Fair
Erie	9%	34%	67%	Poor	1.3	1.6	2.1	Fair	Poor
Ontario	4%	2%	28%	Poor	0.4	0.3	1.0	Fair	Poor

Table 1. Status of Great Lakes prey fish communities in the current period (2015-2017) and previous reporting period (2010-2014) based on the percent native and the Shannon diversity index. To attain as status of 'good', 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. The overall status is conservatively categorized as the lower of the two metrics.

Lake	Indicator	Whole Time Series		Last 10 Years	
		Years	Trend	Years	Trend
Superior	Diversity Index	1978-2017	Improving	2008-2017	Unchanging
	Percent Native	1978-2017	Improving	2008-2017	Unchanging
Michigan	Diversity Index	1973-2017	Improving	2008-2017	Deteriorating
	Percent Native	1973-2017	Unchanging	2008-2017	Unchanging
Huron	Diversity Index	1976-2017	Deteriorating	2007-2017	Unchanging
	Percent Native	1976-2017	Improving	2007-2017	Unchanging
Erie	Diversity Index	1990-2017	Unchanging	2008-2017	Deteriorating
	Percent Native	1990-2017	Unchanging	2008-2017	Deteriorating
Ontario	Diversity Index	1978-2017	Deteriorating	2008-2017	Unchanging
	Percent Native	1978-2017	Deteriorating	2008-2017	Improving

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.

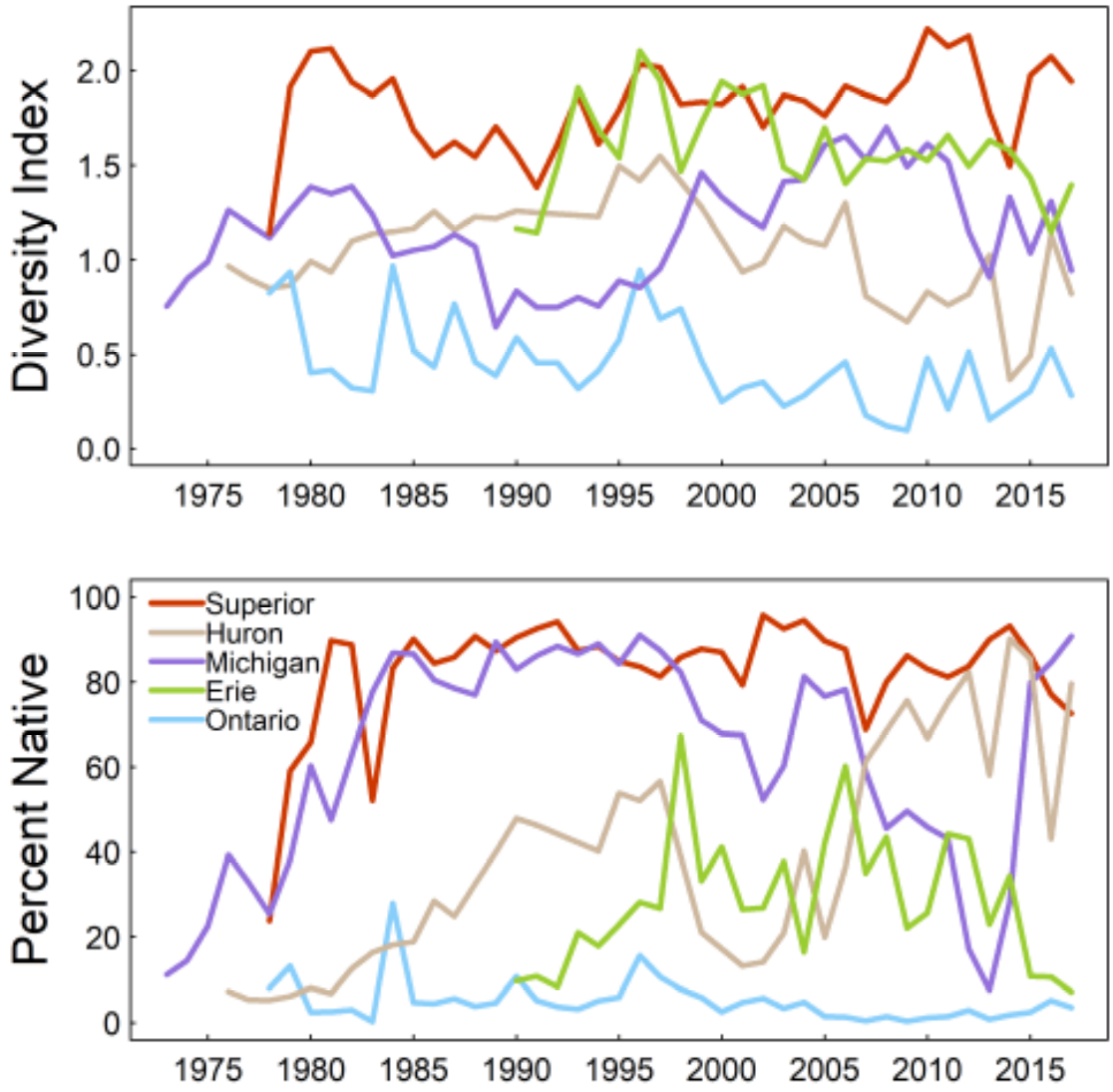


Figure 1. Shannon Diversity index values and proportion of native species of Great Lakes prey fish communities. Source: Data primarily derive from bottom trawl surveys conducted by U.S. federal and state and Canadian provincial agencies

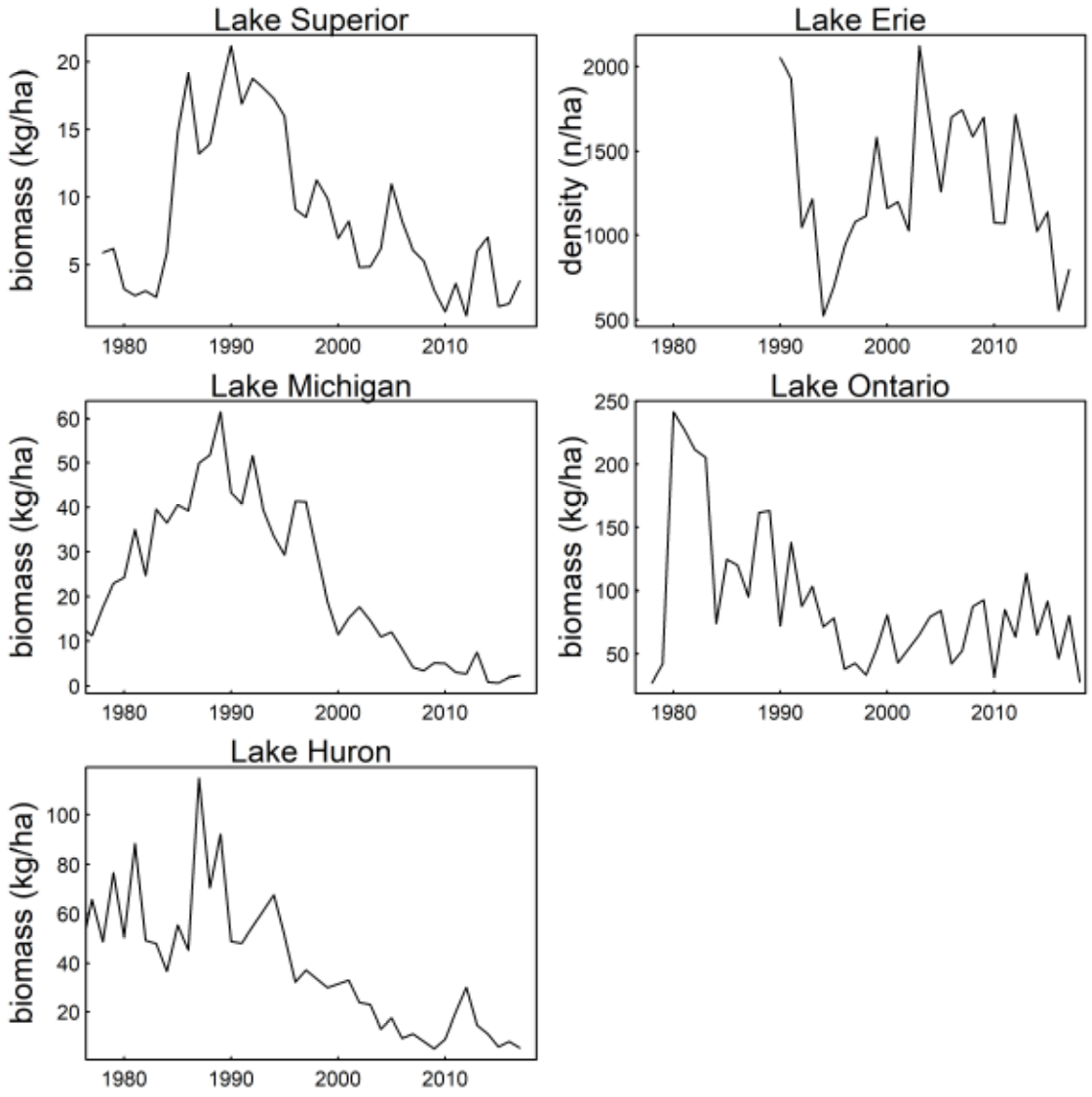


Figure 2. Prey fish biomass or density trends based on annual bottom trawl surveys.

Source: Data primarily derive from bottom trawl surveys conducted by U.S. federal and state and Canadian provincial agencies

Sub-Indicator: Lake Trout

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Improving
Long-term Trend (1975-2017): Improving
Rationale: Sustained and ubiquitous natural reproduction is present only in Lakes Superior and Huron, is minimal but promising in Lakes Michigan and Ontario, and absent in Lake Erie. Stocking, harvest regulation, and Sea Lamprey control are the main tools for continued restoration. High predator consumption, combined with oligotrophication, is reducing alewives to their lowest levels ever measured, especially in Lakes Michigan and Huron. Fewer alewives may be releasing Lake Trout from the bottlenecks of thiamine deficiency and fry predation both perceived to be caused by alewives, which may account for increases in wild recruitment. Lake trout, being opportunistic feeders, have started to consume round gobies, a high-density nearshore subsidy, that has likely increased survival of stocked and wild recruits alike.

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 (1975-2017): Improving
Rationale: Since 1975, wild Lake Trout abundance has increased five-fold in Lake Superior and in the last 10 years, Lake Trout abundance has increased by 10%. Natural reproduction of both nearshore (lean) and offshore (siscowet) populations is widespread and supports nearly all populations. Most stocking has been discontinued because rehabilitation targets have been achieved. In southeastern waters, stocking has ceased because of excessive commercial fishery exploitation. Sea Lamprey mortality has been increasing. In 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 (1998-2017): Improving
Rationale: Lake-wide, Lake Trout densities are relatively stable but at 40% of target levels. Some natural reproduction being detected in southern Lake Michigan, and in populations experiencing lower mortality, older age compositions and higher parental densities compared to areas with little reproduction. These modest gains, while promising, are inconsequential and significant recruitment of wild fish to the general population remains to be seen. In northern Lake Michigan, survival of hatchery-reared fish to the parental stock was poor due to high Sea Lamprey induced mortality but that has been partly remediated with increased control efforts. Fishing mortality is now an important stressor and is contributing to the inadequate density and age structure of the parental stock. Recent increases in juvenile densities are attributed to increased stocking. Most agencies are committed to rehabilitation but oligotrophication is threatening the future of the legacy salmon/alewife fishery and that, combined with angler pressure is, challenging these commitments.

Lake Huron (including St. Marys River)
Status: Good
10-Year Trend: Improving
Long-term Trend (1977-2017): Improving
Rationale: Annual catch from the assessment survey and fisheries have changed from juvenile dominance to adult dominance. Adults stocks increased since the 1990s, peaked at 2007 and appeared to be stable until 2013, although

overall recruitment largely declined since the early 2000s as the increases in wild recruitment did not fully offset the decline in post-release survival of hatchery yearlings. Relative abundance of wild fish in assessments have increased rapidly in recent years, and wild Lake Trout have contributed more than 40% of total population in U.S. waters and more than 80% of total catch from Canadian waters of the main basin of Lake Huron. Similar trends are apparent in the North Channel, while Georgian Bay continues to show lower levels of wild reproduction, with some recent improvements. Stocking of hatchery-reared fish has been reduced in response to wild recruitment and declines in post-release survival of hatchery yearlings.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1992-2017): Improving
Rationale: Increased stocking levels in recent years combined with an emphasis on 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 now comprised of over ten cohorts. Stocking has recently expanded to include all basins in the lake. Sea Lamprey predation continues to be an issue, and no natural reproduction has been detected although relative abundance has increased, which is a precursor to wild recruitment. All agencies remain committed to further rehabilitation and conservation.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1983-2017): 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 has increased.

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
Self-sustaining, naturally reproducing populations that support fisheries is the goal of Lake Trout rehabilitation as established by the Fish Community Objectives (FCOs) and Goals drafted by the Great Lakes Fishery Commission (GLFC). 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 can be based on the values provided in the Great Lakes Fishery Commission (GLFC)'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, as specified in this guide.
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.

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, 1995; Ebener, 1998; OMNR, 1996).

Lake Erie: Ultimate goal – 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 2020, 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 CPE 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 CPE is based on the maximum adult CPE values in New York's waters of Lake Erie during the 1990s. CPEs 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.
Objective 1 – Increase abundance of stocked Lake Trout to a level allowing for significant natural reproduction (CUE of mature females > 4000g greater than 2.0 and 1.1 fish per standard assessment gill net set 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 strains that show the best combination of low post-stocking, juvenile, and adult mortality.
Management Strategy 5: Emphasize 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 fish
- To infer the control measures on fishing and Sea Lamprey predation through the age structure and abundance of mature fish.
- 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.

Self-sustaining populations maintained 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 natural reproduction on the x-axis and year on the y-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 by 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 the stocking and harvest control of Lake Trout. Success to achieve population targets and ultimately to self-sustaining naturally reproducing populations has been mixed among the lakes. Concern over predatory consumption of declining alewife populations has reduced the stocking rates of Lake Trout in Lakes Michigan and Ontario, which may hamper rehabilitation potential.

Since the last report, more wild recruitment is being seen in Lakes Michigan, Huron, and Ontario and Sea lamprey numbers are reduced basin wide. Sustained modest recruitment has continued in southern Lake Michigan and populations are stable but below density targets. Parental stocks seem adequate in most areas with the exception of northern Lake Michigan. In Lakes Erie and Ontario, status and trends are the same as reported in 2017 and in Lake Ontario elevated levels of natural reproduction have continued during 2014-2017. Lake Superior saw a slight improvement in abundance since the last report.

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 for 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 survived in Lake Superior and a few lean Lake Trout populations 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 habitat loss in the Great Lakes, its considered minimal and not a significant impediment to restoration.

Status of Lake Trout

Lake Superior

Wild lean Lake Trout populations have 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. Furthermore, Lake Trout stocking has been deferred in this area because of uncontrolled mortality. In eastern Wisconsin and western Michigan waters, recovered Lake Trout stocks have been impacted by excessive fishery harvest. Recognizing this issue, management actions 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 is 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.

Currently, wild lean Lake Trout abundance has declined in some areas in recent years, but remains higher on average than the prior State of the Great Lakes reporting period. 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). 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 or increasing.

Lake Michigan

Stocking continues in all jurisdictions but has recently been reduced in nearshore waters of Wisconsin waters 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 were recovered in assessment surveys (Lake Trout Task Group 2018) than previously, which indicates that natural reproduction has increased but overall remains low. Areas (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. Increase stocking in the north should lead to increases in overall densities but only if fishing mortality is controlled. Lake Trout appear to be able to cope with the changing forage base with declining alewives being less important in their diets and being replaced by more abundant round gobies. There is currently pressure to maintain Chinook salmon stocking numbers, which has resulted in the stocking of less Lake Trout.

Recommendations to advance recovery include minimizing adult Lake Trout mortality from fishing and 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 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 abundance increased rapidly during the 1990s, peaked in 2007, and did not show a large lake wide decline until 2014, although the 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. In northern Lake Huron, adult abundance has been stable, reflecting a balance among increases in wild recruitment, declines in recruitment of hatchery-reared fish, and substantial reduction in adult mortality

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. Alewives are suspected to have adverse effects on reproduction via Thiamine Deficiency Syndrome and predation on Lake Trout eggs and fry. 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. Survey relative abundance of wild adults has continued to increase since 2010. 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 in Georgian Bay remains lower at 20 to 30%.

Lake Erie

Directed efforts to restore Lake Trout 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.

More recently, 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 now comprised of over ten cohorts. Sea Lamprey continue to be a major issue with adult abundance remaining well above targets despite increased 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 detected in Lake Erie. 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.

Lake Ontario

Prior to 1996, Lake Trout were monitored with a targeted binational Lake Trout netting program. Since 1996, in Canadian waters Lake Trout targets 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. 2018). 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 2008-2014 were followed by a 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 from no stocking in 2012. 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-dressenid era.

Although the abundance of adults reached a peak in 1986, appearance of naturally reproduced 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 1993-2013, a failure to achieve self-sustaining stocks has been attributed to: the dense populations of alewives 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). Recent predictions for restoration have improved with: the recovery of the adult stock from the collapse experienced in 2005-2007; the reappearance of deepwater sculpin, which have steadily increased during 2002-2014. (Lantry et al. 2007, Weidel et al. 2015); the international efforts to re-establish cisco and bloater; and with the inclusion of round gobies in Lake Trout diets (Diertrich et al. 2006; Rush et al. 2012). Signs of improving conditions for natural reproduction were substantiated during 2014-2017 with assessment catches of wild age-1 and -2 Lake Trout rising sharply to a level nearly 10 fold greater than the mean for the previous twenty years (1994-2013). Small catches of wild young-of-year Lake Trout were also observed in Kingston Basin trawls during this period.

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 thaiminase that lowers 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 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 are bentho-pelagic on their foraging behavior and shown to be more thermally tolerant than previously realized (Sellers et al. 1998), is uncertain.

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 Limitations

The sub-indicator is of greatest value in assessing ecosystem health in the oligotrophic, open-water portions of Lake Superior. 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 catches, which may be less robust. Alone, yield goals 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 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 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 slate 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 like stocking can complicate interpretation of their status, the successes of these species are indicative of progress toward the goals of the GLWQA.

Reporting frequency should be every five years. 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, 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 - 2017. 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 non-peered 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 2019 Technical Report

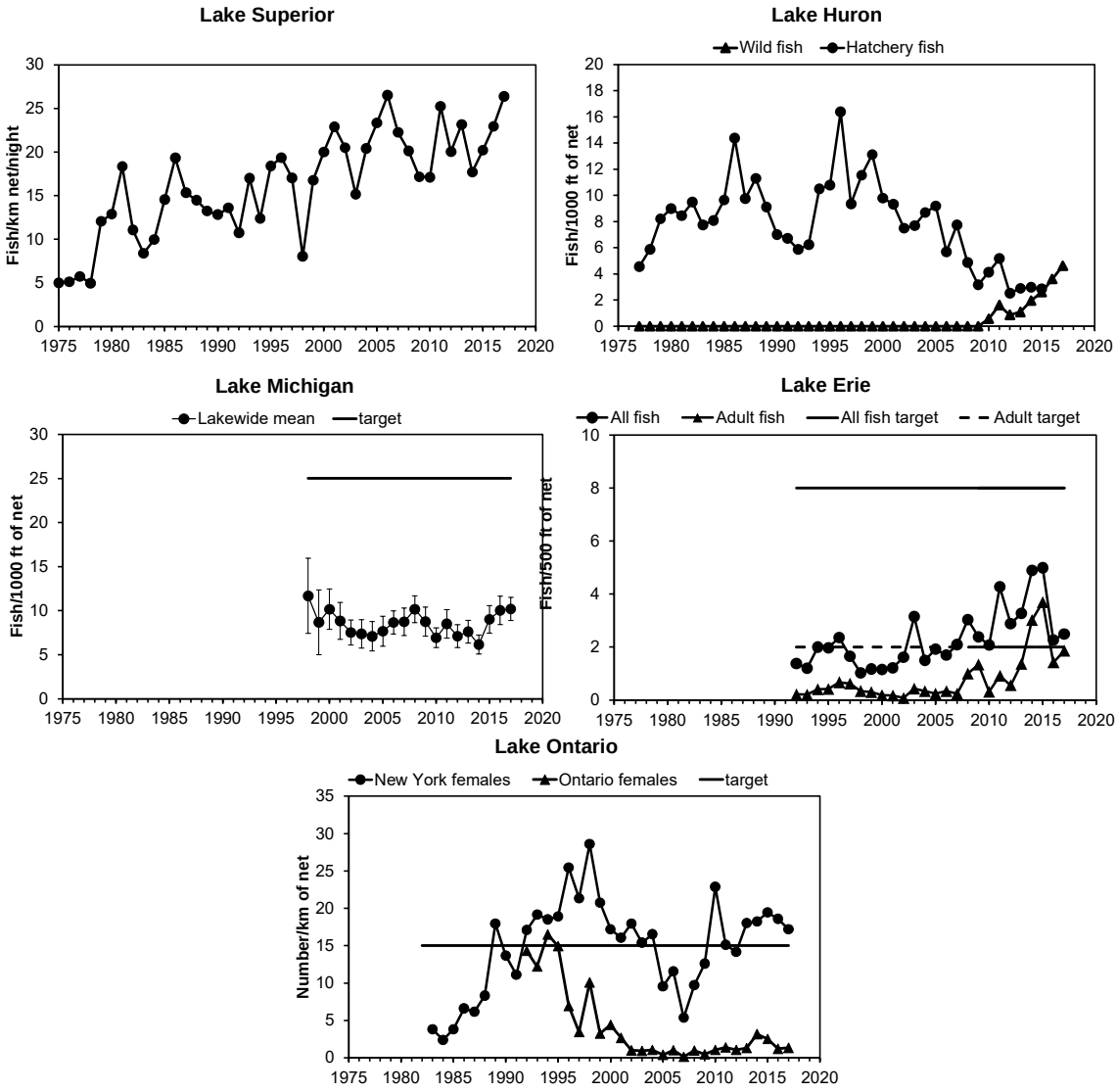


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Sub-Indicator: Walleye

Overall Assessment
Status: Good
Trends:
10-Year Trend: Unchanging
Long-term Trend (1975-2017): Improving
Rationale: Walleye populations in the Great Lakes are quite variable, although the overall status is that populations are Good. Walleye were among the many Great Lakes fishes that experienced population collapse during the 1900s. However, improving water quality, declines in invasive species (e.g., alewife), and sustainable fisheries management during the last 40 years have allowed Walleye to rebound in many locations (i.e., Lake Erie, and parts of Lakes Huron, Ontario and Superior) that now support fisheries of varying sizes (Figure 1). However, Walleye have struggled in other areas (e.g., Lake Michigan) possibly due to changes in productivity caused by dreissenid mussels. Recruitment trends in each Great Lake's localized sub-populations (i.e., river, embayment, basin) continue to play a large part in the overall health of Walleye populations.

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: Improving
Long-term Trend (1980-2017): Improving
Rationale: Walleye population assessment efforts have increased throughout Lake Superior over the reporting period; although data gaps still exist. Index surveys in historically relevant areas such as Black Bay, Nipigon Bay/River, ON, and the St. Louis River, MN/WI have generally shown Walleye populations to have remained stable or improved over the time series. This includes increases in relative abundance (Black Bay), consistent recruitment (all areas) and relatively low mortality (Black Bay and Nipigon Bay). Efforts have been made throughout the lake to address management concerns for Walleye populations including 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 (1985-2017): Unchanging
Rationale: Walleye populations are highly variable across Lake Michigan. For instance, creel survey estimates of Walleye catch and harvest have shown increasing numbers of Walleye are being caught and harvested by anglers in Wisconsin waters of Green Bay, with young-of-year assessments showing above average young-of-year (YOY) production in nine of ten years during 2008-2017. However, similar data from Michigan waters of Green Bay demonstrated that annual Walleye harvest has declined by about 75% from 2008 to 2017, with little evidence for strong natural recruitment. Similarly, natural reproduction has been largely absent outside of Green Bay, although the majority of abundance and harvest historically came from Green Bay. Despite the variability in recruitment, harvest levels have largely been at or above those as defined by Lake Michigan's Fish Community Goals and Objectives during 2014-2017.

Lake Huron (including St. Marys River)
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1971- 2017): Improving
Rationale: The Saginaw Bay Walleye recovery is well-documented and is the subject of a revised management plan similar to that used on Lake Erie, where harvest regulations are revised annually based on population metrics derived from annual fisheries-dependent and fisheries-independent surveys, along with a statistical catch-at-age model.

Walleye populations in Michigan and to a lesser extent in Ontario waters are subject to annual surveys, stock assessment, and modeling. This monitoring and reporting provides very clear indications of status and trends. The St. Marys fish community is assessed periodically as well. In Ontario waters, the commercial Walleye catch is monitored and quotas are adjusted annually. Independent fisheries assessment of most Walleye stocks has been intermittent but has been more systematic and standardized since 2015. In Saginaw Bay, recreational harvest achieved a new record in the 2017 Fishing Year (April 2017 through March 2018) at 360,000 walleyes. Examination of sustainability reference points such as growth rates, Spawning Stock Biomass levels, exploitation rates and mortality rates indicated that the fishery was still within the boundaries of sustainability. Concurrently, the SCAA model indicated that 1.7 million new age-2 walleyes recruited to the population that same year. Since the last report, small scale change in Saginaw Bay appears to trace back to a non-static stock/recruitment relationship. As the population came to peak abundance after initial resurgence of reproductive success, recruitment has declined some. This is viewed as a predictable adjustment to a newly recovered population. In the southern main basin of Lake Huron, in the Ontario waters, there is evidence of a steep increase in Walleye abundance in the Grand Bend area (in 2017). This abundance will be monitored to see if it is the start of some new expansion or a one-year artifact. In the St. Marys River, fishery-independent surveys and creel surveys together indicate a generally stable Walleye population and fishery over time although there was some decline in harvest in 2017. That population is supplementally stocked and it is not fully clear the degree to which that population and fishery depend on stocking but is the subject of a ten-year evaluation that is coming to conclusion soon. Walleye are also found along the nearshore waters of much of Lake Huron usually oriented around river mouths and embayments but are less well-studied.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1975-2017): Unchanging
Rationale: Over the 40-year time-series, Walleye abundance estimates in Lake Erie have fluctuated widely around the average of 48.8 million fish, with no significant improving or deteriorating trend (WTG 2018). In the last 10 years (2007-2017), the estimated Walleye abundance in Lake Erie declined from 130 million fish in 2005, which was the highest abundance estimate in the 40-year time-series, as the 2003 cohort aged. During this time, Walleye fisheries were predominately supported by the 2003 and 2007 cohorts, with minor contributions from the 2010 and 2011 cohorts. Recent increases in abundance during 2016 and 2017 were driven by the strong 2014 and 2015 cohorts. The 2014, 2015 and 2017 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 (including Niagara River and International section of the St. Lawrence River)
Status: Good
10-Year Trend: Unchanging
Long-term Trend (1992-2017): 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 Bay of Quinte and in Ontario and 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. Since the last sub-indicator report, Lake Ontario Walleye status has been reassessed from Fair to Good. This change reflects on-going stability in the Walleye population and realistic expectations of Walleye abundance within the current Lake Ontario ecosystem.

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 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. 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 than absolute values within any year.

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 2007-2017. 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 may occur.
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 2007-2017. 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 2007-2017. 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 yielded.

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 Walleyes/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 2003).
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 TAC area
2. Maintain both 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.

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) 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 dominance 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 has been closed.

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 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 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. Being co-evolved with the rest of the fish community and the natural ecosystem of the Great Lakes, Walleye represent the natural biodiversity of the lakes. They have been subjected to the full slate of other 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 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 coolwater 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 in Lakes Michigan, Ontario or Superior because of the limited data available.

Lake Superior

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 recreational Walleye fisheries in the Lake Superior basin, with an estimated population size of 50,000 adult Walleye (Olson et al, 2018). 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

Walleye populations are highly variable across Lake Michigan. For instance, creel survey estimates of Walleye catch and harvest have shown increasing numbers of Walleye are being caught and harvested by anglers in Wisconsin waters of Green Bay, with young-of-year assessments since 2008 showing above average young of year (YOY) production in nine of ten years during 2008-2017. However, similar data from Michigan waters of Green Bay demonstrated that annual Walleye harvest has declined by about 75% from 2008 to 2017, with little evidence for strong natural recruitment. Similarly, natural reproduction has been largely absent outside of Green Bay. In southern Lake Michigan, near Milwaukee, creel surveys have indicated that Walleye catch and harvest have declined since the last report likely due to the cessation of stocking. Although YOY Walleye have been detected in electroshocking surveys, recruitment to the fishery has been minimal. The Walleye population in Wisconsin waters of southern Lake Michigan has decreased since the early 1990s, but Walleye numbers have remained steady recently. Similarly, angler harvest estimates of walleyes for the Michigan waters of the main basin of Lake Michigan have remained at relatively low levels since the 1990s, with harvest estimates of fewer than 500 walleyes per year in seven of the last 10 years.

Lake Huron

Much of the trends of Walleyes 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. Walleye abundance has contracted some in the bay as it has adjusted to a new equilibrium at its revered state. Coming to a stable abundance that is somewhat lesser than the initial resurgence, similar to abundance trends exhibited in Lake Erie in the 1980s as it recovered. Fishery managers in Michigan have liberalized harvest regulations for the recreational fishery within the bay and that has resulted in a new record harvest by 2017 of nearly 350,000 Walleyes. The intent was to more fully utilize the recovered Walleye population and to try and improve survival of the depressed Yellow Perch population within the bay. Managers are working with researchers to keep harvest policies set within limits of sustainability while achieving full utilization of the stock.

Spatially, Ontario's Walleye populations in Lake Huron are expansive and diverse. The mixed stock aggregation in the southern main basin comprised of dynamic inputs from western Lake Erie and Saginaw Bay, 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 with a mix of some showing signs of improved recruitment while most found at depressed levels of abundance. Walleyes are harvested recreationally, commercially, and by Indigenous fisheries. The exact extent of the mixing of some of these reproductive sources is difficult to assess and determine impacts of exploitation on localized stocks. Assessment netting provides trend information on localized

reproductive stocks at specific spawning tributaries. Increased efforts to mark and tag spawning aggregations of Walleye will provide insights into home ranges and stock mixing in nearshore and offshore waters.

Overall the lake wide trend appears to be unchanging. The overall status of the Lake Huron Walleye population and fisheries has to be characterized as Good given the recovery of the Saginaw Bay stock, although there is further improvement desired, particularly in Ontario waters. Generally, yield across all sources has not fully achieved this historic average or the Fish Community Objective of 0.7 million kilograms/year, but may be beginning to approach that.

Lake Erie

Since 2011, the annual Total Allowable Catch (TAC, or fishery quota) set for the west and central basins of Lake Erie has been gradually increasing (no TAC is set for the east basin), resulting in increased Walleye harvest in both the sport and commercial fisheries. The commercial harvest has annually exceeded the 4 million pound management objective identified in the Walleye Management Plan (Kayle et al. 2015). In 2017, the projected spawner biomass was estimated at 37.6 million kilograms, well above the 12 million kilogram limit reference point of 20% of the unfished spawner biomass (WTG 2017). Across Lake Erie, the annual sport fishing effort remains below the long-term mean, but has been trending upwards since 2011 (WTG 2018). There had been a downwards trend in the sport fishing catch rates since 2014 but in 2017 this rebounded to levels above the long-term average. In 2016, the catch rate in all management units was below the long-term average and the Walleye management objective of 0.4 Walleye/hour. However, in 2017, catch rates in all management units increased to levels near or above the long-term averages and above the Walleye management objective. Commercial effort across the lake had been trending upwards since 2010 while lake-wide catch rates had been trending downwards during the same period. In 2015, this downward trend in catch rates switched and is now trending upwards with catch rates in 2017 exceeding the long-term averages in all management units.

Lake Erie Walleye fisheries remained largely dependent on the 2003 and 2007 cohorts, with some contributions from the 2010 and 2011 cohorts until 2015 (WTG 2018). Until 2015, the mean age of Walleye in the sport and commercial harvest continued to rise with the average age in the commercial harvest peaking at age-7 and the sport harvest peaking at age-6 in that year. Three years of moderate to strong recruitment in 2014, 2015 and 2017 shifted the Walleye recruitment trend in recent years. In 2017, age-2 (2015 cohort) and age-3 (2014 cohort) fish dominated both the commercial and sport fisheries, and the mean age dropped in the commercial fishery to age-4 and in the sport fishery to age-3. The 2017 cohort is also considered to be of moderate strength and 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.

East basin Walleye stocks remain difficult to independently evaluate largely because migration of west and central basin Walleye overwhelm trends of east basin stocks when using traditional assessment approaches (Kayle et al. 2015). Current research is working to evaluate the variability in the migration rates of fish among Lake Erie's basin to help in this evaluation. In the east basin, despite a relatively stable trend in effort, harvest rates in the sport fishery has been increasing since 2009. Similar to west and central basin fisheries, in 2017, age-2 (2015 cohort) and age-3 (2014 cohort) fish significantly contributed to the east basin fishery (WTG 2018). Unlike the other regions of the lake, age-7 and older fish are still an important component of the east basin fishery. The mean age of Walleye in the sport harvest continued to rise until 2016 with the average age peaking at age-8 in that year. In 2017, this mean age of Walleye in east basin dropped to age-5. Sport and commercial catch rates in the east end of the lake currently remain above the long-term mean.

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 areas, and it is in northeastern Lake Ontario that the largest walleye population, fishery, and assessment focus occur. The Bay of Quinte provides the greatest quantity of Walleye spawning and nursery area 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 Bay of Quinte recreational fishing Walleye harvest averaged about 87,000 lb (39,463 kg) annually over the 2008-2017 decade. A small Walleye commercial fishery (less than 50,000 lb or 22,680 kg total annual quota) averaged about 25,000 lb (11,340 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 Prey Fish

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, 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 have 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	LE, LH, LM, LO,	LS		
Data are from a known, reliable and respected generator of data and are traceable to original sources.	LE, LH, LM, LO,	LS		
Geographic coverage and scale of data are appropriate to the Great Lakes basin	LE, LM, LO	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	LE, LM, LO	LH, LS,		

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 aperiodic assessment of the shared St. Marys River and there 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).

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 fisheries (recreational, commercial, or tribal) in some years for all of 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.

Acknowledgments

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Information Sources

Kayle, K., K. Oldenburg, C. Murray, J. Francis, and J. Markham. 2015. Lake Erie Walleye Management Plan 2015-2019. Great Lakes Fishery Commission, http://www.glfc.org/pubs/lake_committees/erie/LEC_docs/position_statements/walleye_managment_plan.pdf
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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, 1975–2017. Fish Community Goals and Objectives are: Lake Michigan, 100-200 metric tonnes; Lake Huron, 700 metric tonnes; Lake Erie, sustainable harvest in all basins; Lake Ontario, maintain early 1990s populations and expand populations into favorable habitats. Source: Chippewa Ottawa Resource Authority, Michigan Department of Natural Resources, Minnesota Department of Natural Resources. New York State Department of Environmental Conservation, Ontario Ministry of Natural Resources, Ohio Department of Natural Resources, Pennsylvania Fish and Boat Commission, Wisconsin Department of Natural Resources

Last Updated

State of the Great Lakes 2019 Technical Report

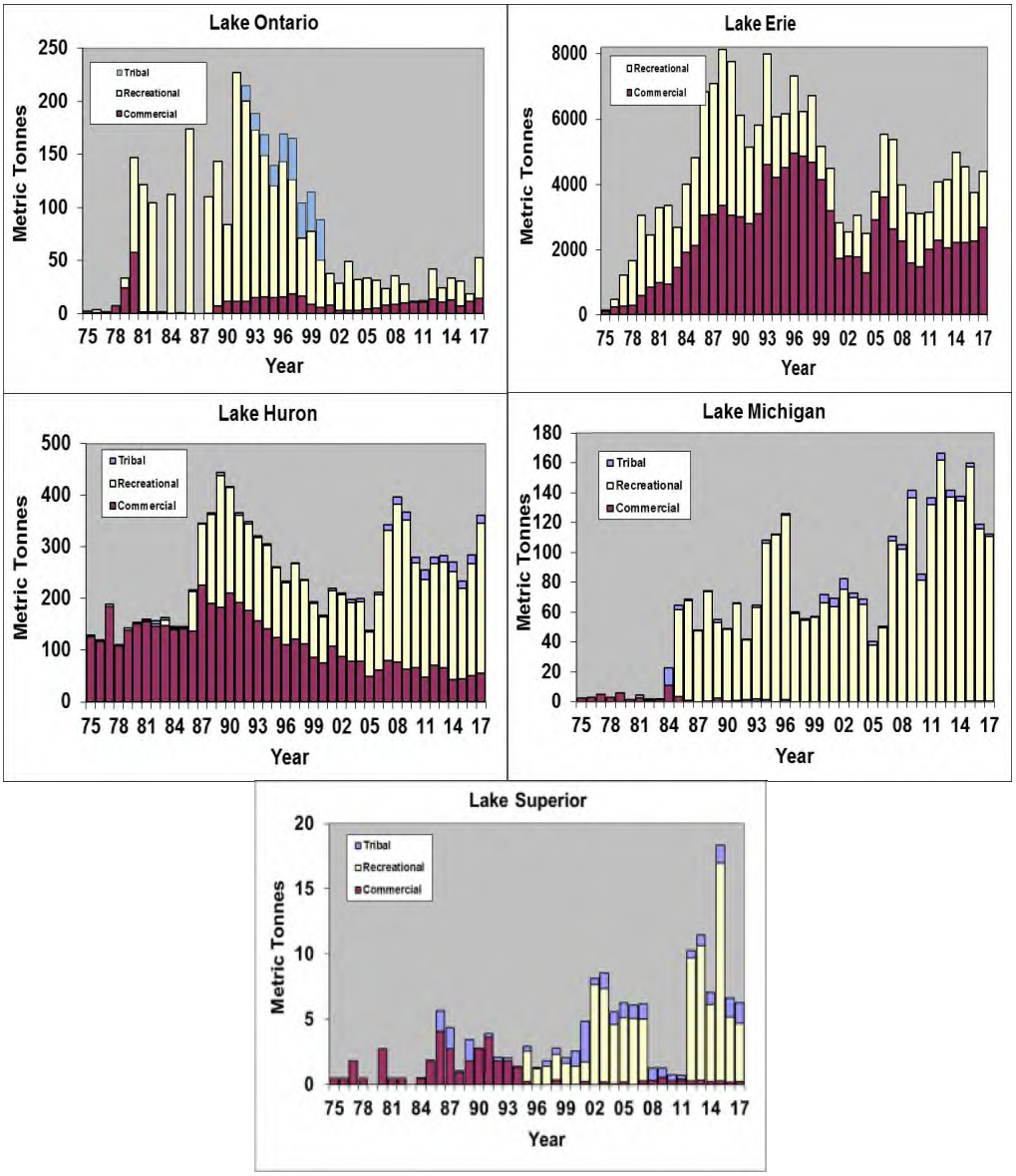


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Source: Chippewa Ottawa Resource Authority, Michigan Department of Natural Resources, Minnesota Department of Natural Resources. New York State Department of Environmental Conservation, Ontario Ministry of Natural Resources, Ohio Department of Natural Resources, Pennsylvania Fish and Boat Commission, Wisconsin Department of Natural Resources

Sub-Indicator: Fish-Eating and Colonial Nesting Waterbirds

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Unchanging
Long-term Trend (1976-2009): Not provided
Rationale: Colonial nesting waterbirds function as apex predators in freshwater systems, deriving a large proportion of their diet from fish and other aquatic prey and providing an important linkage between aquatic and terrestrial habitats. Eleven focal species of colonial waterbirds breed at sites (predominantly islands) distributed across all of the Great Lakes: Herring, Ring-billed and Great Black-backed gulls, Caspian, Common, Forster's and Black terns, Great Blue Herons, Great Egrets, Black-crowned Night-Herons and Double-crested Cormorants. Four species are less numerous now than when systematic monitoring began (1976-80; Great Blue Heron, Black-crowned Night-Heron, Herring Gull, Common Tern), although rates of decline have slowed for all species over the last decade. Twenty-year (1989-91 to 2007-09) population trends for six of eight species have been assessed as stable. Great Blue Herons exhibited a moderate 20-year decline (-40%). Double-crested Cormorant nests increased 385% since 1989-91, although the rate of increase has slowed over the last decade (a 30% increase since 1997-99).

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: Unchanging
Long-term Trend (1976-2009): Not provided
Rationale: Two species are less numerous now than when systematic monitoring began (1976-80; Great Blue Heron, Common Tern), although rates of decline have slowed for both species over the last decade. Since 1989-91, one species has exhibited a stable trend (Common Terns), two have undergone moderate declines (Great Blue Heron, Herring Gull), one species has had a large decline (Ring-billed Gulls), and one a large increase (cormorants). Unable to calculate trends for night-herons or Caspian Terns and egrets have never nested on Lake Superior.

Lake Michigan
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1976-2009): Not provided
Rationale: Two species are less numerous now than when systematic monitoring began (1976-80; Black-crowned Night-Heron, Common Tern), although rates of decline have slowed for both species over the last decade. Twenty-year populations trends: two species have experienced large declines (Common Tern, Great Blue Heron), one had a moderate decline (Black-crowned Night-Heron), three species were stable (Herring Gull, Ring-billed Gull, Caspian Tern) and one species exhibited a large (> six-fold) increase (cormorants). Unable to calculate a trend for egrets.

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1976-2009): Not provided
Rationale: Five species are less numerous now than when systematic monitoring began (1976-80; Great Blue Heron, Herring Gull, Ring-billed Gull, Common Tern, Caspian Tern); rates of decline have slowed for all species, except Great Blue Heron, over the last decade. From the 1989-91 period to 2009, one species has undergone a large decline (Great Blue Heron), one had a moderate decline (Caspian Tern), three species were stable (Herring Gull, Ring-billed Gull, Common Tern), one species had a moderate increase (Black-crowned Night-Heron) and two exhibited large increases (cormorants, 2.5 times greater; egrets, 7.8 times greater).

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1976-2009): Not provided
Rationale: Three species are less numerous now than when systematic monitoring began (1976-80; Great Blue Heron, Black-crowned Night-Heron, Common Tern); rates of decline have slowed for all these species over the last decade. Since 1989-91, three species exhibited a moderate decline (Great Egret, Black-crowned Night-Heron, Herring Gull), three species were stable (Great Blue Heron, Ring-billed Gull, Common Tern) and one species had a large increase (cormorants, 7.5 times greater). Unable to calculate a trend for Caspian Terns as they colonized Lake Erie during the past decade and there is not enough data.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1976-2009): Not provided
Rationale: One species is less numerous now than when systematic monitoring began (1976-80; Common Tern); the rate of decline has increased over the last decade. Since 1989-91, two species exhibited a moderate decline (Ring-billed Gull, Common Tern), three species were stable (Great Blue Heron, Black-crowned Night-Heron, Herring Gull) and two species had a large increase (cormorants, 2.3 times greater; Caspian Tern, 1.7 times greater). Unable to calculate a trend for Great Egret as they colonized Lake Ontario during the 1997-99 census and there is not enough data.

Status Assessment Definitions
Status assessment is dependent upon the species as population objectives differ; for instance, population size that is not appropriate would refer to both hyper-abundant species (e.g. Double-crested Cormorants, Ring-billed Gulls; populations are considered too large) or declining species (e.g. Common Terns) with populations that are considered too low or rates of decline considered too high.

Good: The sizes of waterbird populations are considered appropriate and there has been a net gain in nesting habitat.
Fair: The sizes of waterbird populations are considered close to appropriate and there has been no net loss in nesting habitat.
Poor: The sizes of waterbird populations are considered inappropriate and there has been a net loss in nesting habitat.
Undetermined: Data are unavailable or are insufficient to assess status.

Trend Assessment Definitions
Robust estimates of abundance and distribution, and an assessment of change in these population metrics over long (since monitoring began in the 1970s) and short (since the last survey period) time scales, by discrete water bodies and for the Great Lakes as a whole are available to report. The current survey design for the Great Lakes region is considered to have a high reliability (i.e. high precision, low survey bias and high geographic coverage). Estimates of population vital rates (e.g. productivity) for the Great Lakes have not been collected systematically in either space or time, but may be adequate for general comparisons with other parts of the breeding range (i.e. data from reference sites), for some parameters / species.

Due to the current census frequency, population status can only be assessed every ten years. Short- and long-term trends will be assessed according to the following scale (or equivalent annual rate of change for shorter time frames), which has been widely applied to avian population trend assessment in Canada (see Environment Canada's "*Status of Birds in Canada*" website; https://wildlife-species.canada.ca/bird-status/index-eng.aspx?sY=2014&sL=e):

- Large decline: ≥50% decline over 20 years
- Moderate decline: ≥25% to <50% decline over 20 years
- Stable: <25% decline to <33% increase over 20 years
- Moderate increase: ≥33% to <100% increase over 20 years
- Large increase: ≥100% increase over 20 years

Improving:
(1) populations of species experiencing long-term declines are increasing (either moderate or large)

(2) populations of hyper-abundant species (cormorants and Ring-billed Gulls) are decreasing (either moderate or large)

Unchanging:
(1) populations of species experiencing long-term decline remain stable
(2) populations of hyper-abundant species remain stable
(3) populations for species considered to be at appropriate levels remain stable

Deteriorating:
(1) populations of species experiencing long-term declines continue to decrease (either moderate or large)
(2) populations of hyper-abundant species are increasing (either moderate or large)
(3) populations for species considered to be at appropriate levels are decreasing

Undetermined: Data are unavailable or are insufficient to assess trends.

Endpoints and/or Targets

- Healthy, self-sustaining populations of each waterbird species.
  - Populations of stable or declining species remain stable or increase, respectively
  - Populations of hyper-abundant species (cormorants and Ring-billed Gulls) either remain stable or decline
- Critical island breeding habitat is conserved

There are no specific population objectives for these species, other than within a few Great Lakes Areas of Concern (e.g. Hamilton Harbour, ON).

One conservation goal to consider as an endpoint in the future is to increase the number of historically important breeding sites that have protected status (very few at present).

Sub-Indicator Purpose

- Assessment of ecosystem health by examining long-term trends in the abundance and distribution of colonial waterbird populations breeding on the Great Lakes.
- The sub-indicator tracks changes in the number of breeding pairs (nests), breeding colonies, and populations of eleven species of fish-eating birds since the mid-1970s, at multiple geographic scales
- Secondarily, some ecological endpoints will be assessed for representative colonial waterbird species at selected sites on the Great Lakes.

The purpose of this sub-indicator, “Fish-Eating and Colonial-Nesting Birds”, is to assess ecosystem health by examining 10-year and long-term trends in the abundance and distribution of colonial waterbird populations breeding on the Great Lakes. Previously, this sub-indicator was reported in conjunction with the historic State of the Great Lakes “Toxic Chemicals in Herring Gull Eggs” sub-indicator, which described trends in chemical contaminants found in the eggs of fish-eating, colonial waterbirds (mainly using the non-migratory Herring Gull as a sentinel species). These sub-indicators have now been separated to report on progress towards two separate objectives under the Great Lakes Water Quality Agreement. This sub-indicator tracks changes in the number of breeding pairs (nests), breeding colonies, and populations of eleven species of fish-eating birds since the mid-1970s, at multiple geographic scales, and secondarily, will assess some ecological endpoints in representative colonial waterbird species at selected sites on the Great Lakes.

Ecosystem Objective

Conservation of critical island breeding habitat, and maintenance of self-sustaining populations (i.e. no further declines in abundance or reductions in distribution) of each of the eleven waterbird species that comprise that avian community. Fish-eating, colonial waterbirds are distributed across all five Great Lakes, their connecting channels, and the St. Lawrence River, both in Canadian and US waters.

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

Eleven focal species of colonial waterbirds breed at sites (predominantly islands) distributed across all of the Great Lakes: Herring, Ring-billed and Great Black-backed gulls, Caspian, Common, Forster’s and Black terns, Great Blue

Herons, Great Egrets, Black-crowned Night-Herons and Double-crested Cormorants. A complete census of all waterbird colonies on the Great Lakes, their connecting channels and the St. Lawrence River (up to 1 km inland from shorelines) has been conducted, jointly, by the Canadian Wildlife Service and the U.S. Fish and Wildlife Service, approximately every 10 years, since the mid-1970s (four complete census periods; the most recent was completed in 2009; the next comparable survey is planned for 2020). Survey timing and methodologies were coordinated between Canada and the USA. Measures include:

1. Nest counts of colonial waterbird species across all water bodies and connecting channels at relevant temporal and spatial scales:
    i. Annual: Counts for Herring Gull (13 focal colonies distributed across the Great Lakes) and Double-crested Cormorant (Lake Ontario and the St. Lawrence River to Cornwall, ON) since the late 1970s. Methods are consistent with 'decadal' survey efforts.
    ii. 10-year: All breeding sites for the eleven focal colonial waterbird species.

2. Periodic measurement of waterbird demographic parameters known to be directly or indirectly impacted by environmental stressors, including (but not limited to): clutch size, egg volume, hatching and fledging success, natal and breeding site fidelity, age at first breeding and age-specific survivorship. Additional monitoring considerations include: avian disease surveillance (e.g. botulism type E) and studies tracking adults through the full annual cycle to establish connectivity between breeding and wintering areas.

Ecological Condition

Fish-eating, colonial waterbirds are distributed across all five Great Lakes, their connecting channels, and the St. Lawrence River, both in Canadian and US waters. Colonial waterbirds function as apex predators in freshwater systems, and provide an important linkage between aquatic and terrestrial habitats. As a guild, waterbirds derive a large proportion of their diet from fish and other aquatic prey (species range from obligate piscivores to having a mix of aquatic and terrestrial prey). On the Great Lakes, waterbird species differ in the foraging strategies they employ, and thus, differ in the aquatic habitats and trophic levels they utilize (e.g. surface feeders or pursuit divers in open water; sit-and-wait predators in littoral zones and wetlands, surface feeders in wetlands). Another life-history trait that waterbirds share is that they nest in dense aggregations (i.e. colonially) and almost exclusively on islands (except for Forster's and Black terns, which nest in wetlands). As such, they can also serve as an important indicator of change in status of this unique habitat within the Great Lakes system.

Changes in waterbird population abundance, distribution and demography can reflect changes in ecosystem trophic structure and/or island or wetland nesting habitat and, therefore, are important metrics for assessing the health of a variety of Great Lakes ecosystem components. Inter-specific differences in foraging and nesting strategies make it possible to assess and integrate trend information across a variety of temporal, spatial and ecosystem scales. Declining waterbird populations (number of breeding pairs or nests) or vital rates (hatching success, fledging success, mortality rates, etc.) can be indicators of local environmental stress. The Great Lakes-wide population of colonial waterbirds has been censused jointly, by the Canadian Wildlife Service and the U.S. Fish and Wildlife Service since the 1970s, approximately every 10 years. Four "decadal" censuses have been conducted to date: in the 1970s, 1980s, 1990s and 2000s. For this sub-indicator, population change (over the last 20 years) is defined as: large decline = ≥50% decline; moderate decline = ≥25% to <50% decline; stable: <25% decline to <33% increase; moderate increase: ≥33% to <100% increase; large increase: ≥100% increase. Briefly, in the long-term (1976-2009), these censuses have shown that the breeding numbers of four species have undergone large increases: Double-crested Cormorants, Great Egrets, Ring-billed Gulls and Caspian Terns (population growth has slowed since the 2nd census for the latter two species; Figure 1).

Declining numbers of Caspian Terns in North America was of concern in the 1960s. Their decline can be attributed to reduced waste as a food source, due to better landfill practices, declining forage fish populations in the Great Lakes, predation (i.e. coyotes, foxes), and competition for nest sites with Ring-Billed Gulls. Declining numbers are particularly notable in Lake Michigan and Lake Huron. Since the 1980s, the numbers of breeding pairs have significantly increased in the Great Lakes (Wires et al, 2000). This increase could be a result of remediation efforts. For example, the Hamilton Harbour Remedial Action Plan supports management techniques to maintain species diversity in the Hamilton Harbour. One of the actions is to prevent Ring-billed Gulls from nesting on Caspian Tern habitat (Zanchetta et al., 2016).

Three species, Great Blue Heron, Herring Gull and Great Black-backed Gull (GBBG, an uncommon breeder on the Great Lakes, trend not shown), exhibited a period of population growth followed by a decline; current numbers of

breeding pairs for these species are similar to 1970s levels and are considered stable. In contrast, Common Terns and Black-crowned Night-Herons have both undergone long-term declines, although the rates of decline have slowed over the last decade and these populations are currently considered stable. For the six species that have undergone declines since the 1989-91 census (Figure 1; GBBG trend not shown), continued monitoring will determine whether these populations have in fact stabilized or if there is evidence for concern. Currently, drivers such as habitat change and loss, changes in trophic structure and abundance of prey (Hebert et al. 2008, 2009), reduced access to alternate sources of food (for gulls, due to changes in agricultural and waste disposal practices), inter-specific competition for nesting space (e.g. increased pressure from hyper-abundant species such as cormorants and Ring-billed Gulls) and stressors in overwintering areas likely play a larger role in regulating waterbird populations than contaminant-related impairments.

Linkages

Waterbird population trends and breeding habitat are indicators in some of the AOCs, and are used for delisting criteria.

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

- **Toxic Chemicals in Herring Gull Eggs –** Toxic chemicals like pesticides (PCBs and DDEs) can cause eggshell thinning in Herring Gull eggs, leading to a high rate of egg breakage (EC, 1995). This can lead to reproductive failure in Herring Gulls. Since the 1970s, concentrations of contaminants in Herring Gull eggs have decreased and have stabilized in recent years. However, contaminants are still found in certain areas of the Great Lakes. In particular, legacy contaminants persist in some areas of the Great Lakes which has led to contaminants like dioxins, furans, PCBs and metals being found in eggs (ECCC and USEPA, 2018). The effects of these contaminants include deformities, reproduction problems, and restrictions on fish consumption in the area (ECCC and USEPA, 2018).
- **Toxic Chemicals in Whole Fish –** Given their position on the trophic level food web, waterbirds are vulnerable to consuming toxic chemicals found in prey. These contaminants include pesticides, polychlorinated biphenyls, oil hydrocarbons, and heavy metals. Contaminants can accumulate in the tissue or gills of whole fish, and causes acute toxicity in waterbirds once they ingest these organisms (FDA, 2011).
- **Water Levels –** High water levels can disrupt the habitat of birds, rending their environment susceptible to the introduction of invasive species and nutrient loading, as well as the loss of biodiversity (Zohary and Ostrovsky, 2011). Increased water levels have also led to habitat encroachment in many bird populations (Ferguson, 2014).
- **Climate Change –** Increases in temperature, rainfall, fire, and storms cause changes in egg laying patterns, migration times, behaviour, disruption in habitat, and increased extinction rates in fish and colonial waterbird populations (Wormworth and Mallon, 2006). Inferences on the effects of climate change on population trends are beyond the scope of this sub-indicator. In addition, birds could be moving out of the Great Lakes region (i.e. a shift in distribution) in response to climate-related effects, with no net change in abundance at larger spatial scales.

This sub-indicator also links directly to the other sub-indicators in the Habitat and Species indicator, particularly:

- **Lake Sturgeon, Lake Trout, Walleye, and Prey Fish –** Fish-eating and colonial waterbirds can impact the fish populations of Lake Sturgeon, Lake Trout, Walleye, and Prey Fish in some areas of the Great Lakes. Fish anglers and fish-eating birds are competing to capture fish for food (EC, 1995). Cormorants demand a large food supply; and there have been instances where their large population size has depleted and reduced local fish populations (EC, 1995).

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 Limitations

- Most waterbird species are migratory. Changes in population status or trends could reflect environmental or anthropogenic stressors experienced during the non-breeding period (or cumulative effects over the full annual cycle, inside and outside of the Great Lakes region).
- Data are collected at 10-year intervals, which is longer than the reporting cycle for State of the Great Lakes Reporting.

Additional Information

The current monitoring scheme for colonial waterbirds (monitor all colony sites across the Great Lakes every 10 years) is costly, requires a large commitment of personnel time and takes 2-3 years to complete. To reduce overall survey costs and enable surveys to be conducted at shorter intervals (to improve inference of population trends), there has been an impetus to shift to a sampling framework (monitor fewer, focal colonies more often and base trend assessments on this sampling approach). This change in survey design has already been implemented in the USA, and currently is being developed for Canada. It is important to note, that yearly population estimates indicate that the long-term trends are probably captured with decadal colonial waterbird surveys, but the dynamic nature of waterbird populations, such as the Breeding Double-crested Cormorant, is not. Research suggests that cormorants are dynamic birds with frequent behavioural changes and variation in population size which may warrant yearly monitoring (Seefelt, 2012).

Bald eagle (or other birds of prey) nest counts or number of nestlings produced per year could be considered as a measure in future reports. If these data are available, they would be collected by provincial wildlife agencies.

Acknowledgments

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List of Figures

Figure 1. Population trends for the entire Great Lakes region (black line) and by water body (coloured lines, see legend) for eight species of colonial waterbirds census used during four 'decadal' periods, 1976-2009.
Sources: Canadian Wildlife Service- Ontario Region, Environment and Climate Change Canada, Burlington, ON; Cuthbert and Wires (2013).

Last Updated

State of the Great Lakes 2019 Technical Report

(no new data for SOGL 2019)

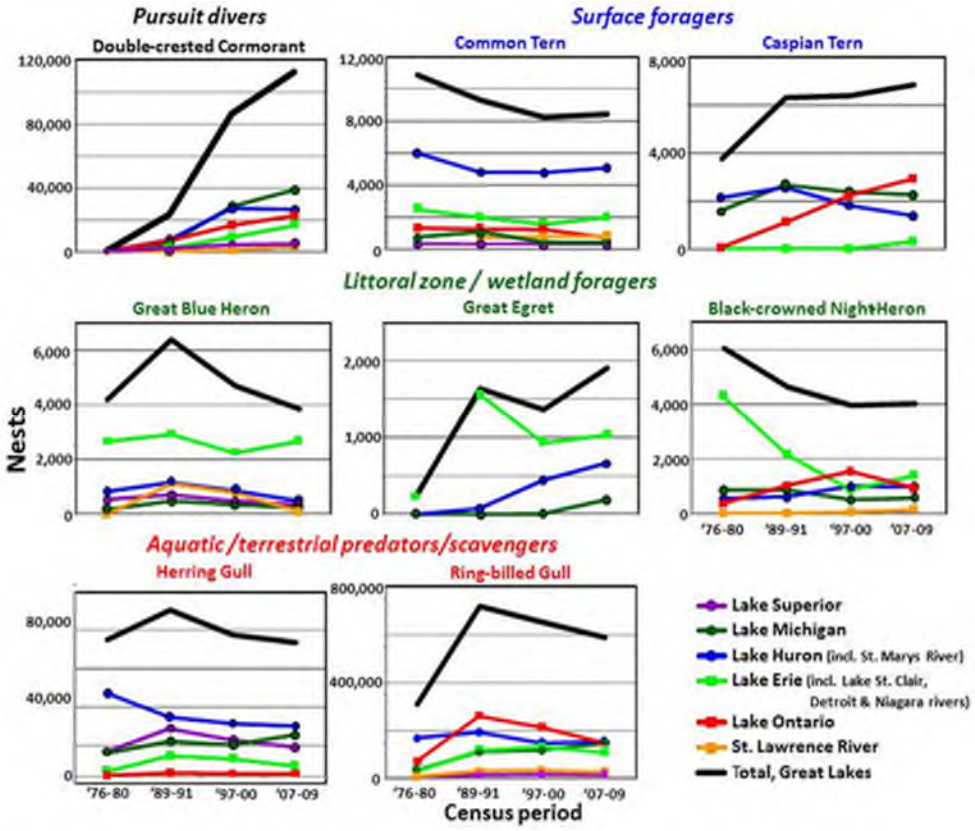


Figure 1. Population trends for the entire Great Lakes region (black line) and by water body (coloured lines, see legend) for eight species of colonial waterbirds census used during four 'decadal' periods, 1976-2009.
Sources: Canadian Wildlife Service- Ontario Region, Environment and Climate Change Canada, Burlington, ON; Cuthbert and Wires (2013).

Sub-Indicator: Nutrients in Lakes

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Deteriorating
Long-term Trend (~1970 - 2017): 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). Nearshore, symptoms of nutrient enrichment persist in some locations. In Lake Erie, objectives are frequently exceeded, and conditions are Deteriorating. Only in Lake Superior is the offshore objective being met and conditions acceptable

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 (1970-2017): 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: Deteriorating
Long-term Trend (1983-2017): Deteriorating
Rationale: Offshore phosphorus concentrations are continuing to decrease below objectives. Concentrations have fallen to low levels and may be negatively affecting lake productivity based on the historic food web and is therefore assessed as Fair. In some nearshore areas, elevated phosphorus is observed and may be supporting nuisance algae growth. New information is included for Green Bay.

Lake Huron
Status: Fair
10-Year Trend: Deteriorating
Long-term Trend (1970-2017): Deteriorating
Rationale: Offshore phosphorus concentrations are continuing to decrease to values that are well below objectives. Concentrations may be too low to support a healthy level of lake productivity based on the historic food web and is therefore assessed as Fair. In some nearshore areas, elevated nutrients may be contributing to nuisance algae growth.

Lake Erie
Status: Poor
10-Year Trend: Unchanging
Long-term Trend (1970-2017): Unchanging
Rationale: Total phosphorus objectives continue to be exceeded. Although high values are most frequently elevated 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 past 35 years. Overall trend was previously reported to be “Deteriorating” due to conditions possibly leading to increased algal growth. New information is included for Lake St. Clair.

Lake Ontario
Status: Fair
10-Year Trend: Deteriorating
Long-term Trend (1970-2017): Deteriorating
Rationale: Offshore phosphorus concentrations are continuing to decrease to levels too low to support healthy offshore lake productivity based on the historic food web and is therefore assessed as Fair. 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: Metrics do not indicate a clear overall trend.

Lake Ecosystem Objectives for Nutrients in Lakes are identified in Annex 4 of the Great Lakes Water Quality Agreement (specifically Lake Ecosystem Objectives #5 and #6).

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:

Basin	Total Phosphorus (µgP/L)	Chlorophyll *a* (µgChl*a*/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

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.

New loading targets established for Western and Central Lake Erie will be utilized in future sub-indicator reports to support Great Lakes assessments.

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; 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.

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. There are no objectives for nearshore nutrient concentrations; Provincial and/or State nearshore nutrient objectives will be considered in this report as provisional benchmarks only.

The establishment of Substance Objectives for phosphorus concentrations and loading targets take into account the bioavailability of phosphorus (and seasonality); therefore, status and trends of the bioavailable phosphorus fraction (soluble reactive phosphorus) and seasonal information are provided in this report where possible.

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). 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) to be able to provide a clearer signal of lake changes.

Data are collected in the spring and summer of each year (U.S. EPA) or selected years (ECCC) in each Great Lake. Stations are located in both nearshore and offshore regions; however, not all nearshore regions are captured in the federal programs. Additional nearshore data (e.g., collected by local agencies) may be available that have not been integrated into this sub-indicator, but there is greater variability of conditions in the nearshore.

Ecological Condition
Current Status
The current status of spring total phosphorus concentrations in 2016-17 is shown graphically in Figure 1. Recall the

current GLWQA objectives (Table 1) apply to offshore waters only. It is noted that these objectives are achieved in the offshore of all of the Great Lakes with the exception of Lake Erie. In Lake Erie, objectives have recently been exceeded at about 40% of stations in the central and eastern basins, and at about 70% of stations in the western basin 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 or 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 ecological offshore planktonic communities based on the historic food web. For this reason, these lakes are all designated as having a Fair or Good status. In some cases, phosphorus concentrations are less than one-half of objectives. While offshore regions are nutrient deficient, elevated concentrations are observed in some nearshore regions of all the lakes with the exception of Lake Superior. 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 to 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 – 2017) 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 significantly interannual variability, it is difficult to discern meaningful 10-year trends in any of the Great Lakes except Lake Ontario where a continuing decline in total phosphorus concentrations is indicated.

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 to represent the lake and 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.

The long-term trends of offshore soluble reactive phosphorus (not shown) are similar to those for total phosphorus, but the overall values are lower. In Lake Ontario in particular, the ratio of SRP:TP also shows a marked decline over time. In the 1970s, about 70% of the total phosphorus in the offshore comprised the soluble reactive fraction. Since about 2000, the ratio has averaged only 20 to 30%. Because SRP is the most readily available form of phosphorus, these trends together indicate a shortage of available phosphorus in offshore regions of the lake, as well as potentially rapid uptake in an ultraoligotrophic (nutrient deficient) environment.

Trends of spring nitrogen levels are represented as nitrate (NO_3) in Figure 9. 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. 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 (Maccoux et al., 2016). The most recent loadings estimates available show that targets for the other Great Lakes are only occasionally exceeded and that there are no significant temporal trends since the

1980s with the exception of declines noted for Lakes Ontario and Huron (Dolan and Chapra 2012). The lack of sufficient and appropriate information upon which to directly compute loads remains a significant challenge in 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, 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.

Other Spatial Scales – Nearshore Regions

This sub-indicator report uses recent offshore total phosphorus (TP) concentrations to compare with objectives, for which the objectives were intended. These offshore waters also best indicate long-term trends because, in contrast to shallower, nearshore waters, they are less influenced by local pollutant discharges. There is a need to identify the status and trends of nutrients in nearshore waters; however, there are no objectives for nearshore nutrient concentrations. Additional agencies may be conducting work that can feed into this sub-indicator component. In this update, newly acquired information from the southern portion of Green Bay and from Lake St Clair have been incorporated into the sub-indicator and these help fill in the picture about spring total phosphorus concentrations at a basin-wide scale (Figure 1).

Incorporating nearshore information is important because some regions of the Great Lakes are experiencing nuisance algae problems. The extent of the algae problem seems to be of similar magnitude as was experienced in the 1970s (GLWI 2005), despite significantly lower phosphorus loads and in-lake concentrations since that time. In Lake Michigan, growth of the benthic alga *Cladophora* remains a problem, making some beaches un-swimmable (Bootsma et al. 2015). *Cladophora* blooms appear to be most extensive in eastern Lake Erie, while the western Lake Erie basin is also plagued by the more toxic *Microcystis* algal blooms (Stumpf et al. 2012). In Lake Huron, the benthic alga Chara is flourishing on the east side and additional algal species are associated with fouling issues in the lake; here, *Cladophora* is not the sole contributor (E.T. Howell, personal communication). In Lake Ontario, nearshore regions on both the south and north shores routinely experience nuisance benthic algae blooms.

The causes of the nearshore algae resurgence are not clear. For example, in Lake Erie, loadings of phosphorus exhibit high inter-annual variability but have decreased since the 1970s and show no temporal trend since the late 1980s (Maccoux et al. 2016). The invasion and proliferation of non-indigenous mussels (Dreissena spp.) have likely altered nutrient dynamics, simultaneously depleting offshore nutrients and elevating concentrations in nearshore regions, causing an imbalance especially in Lakes Ontario, Michigan and Huron. Lake Erie is an exception, where phosphorus concentrations are above objectives throughout the western basin and much of the central basin and

there is no evidence of a significant decline. Symptoms of nearshore eutrophication (elevated nutrients) are observed.

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, increasing the rate of nutrient uptake and therefore reducing in-lake concentrations.
- 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 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 recently conducted and are ongoing.

A fuller picture of the status of nearshore nutrients would be obtained by including additional Provincial, State and other more local quality-assured sources of data.

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.

In the future additional data collected by other State and Provincial environmental agencies should be included as feasible in order to report more fully on nearshore nutrient status and trends on a basin-wide scale. This will require data integration and further consideration of interagency laboratory comparability.

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 in Lake Erie. Additional monitoring or improved monitoring coordination and integration may be warranted to improve our ability to detect meaningful change in such a dynamic system. For the other Great Lakes, the establishment of loading reductions are still being discussed. 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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Contributors: E.T. Howell and N. Diep, Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment and Climate Change. Sarah Bartlett, NewWater (Green Bay Metropolitan Sewerage District).

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List of Tables
Table 1. Interim Substance Objectives for Spring Total Phosphorus and Summer Chlorophyll a Concentrations, with resultant Trophic State

List of Figures
Figure 1. Spatial distribution of total phosphorus (µg/L) in the Great Lakes based on lake-wide cruises conducted during the spring 2016 and 2017**.** Sampling stations are shown as black dots.
Data sources: Environment and Climate Change Canada, United States Environmental Protection Agency and New-Water (Green Bay Metropolitan Sewerage District).
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.
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Figure 5. Federal (US and Canada) long-term record of offshore, spring (April - May) total phosphorus in Lake Erie (µg/L). The central basin of Lake Erie is used to represent the overall condition of the lake. 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. Note the scale difference for Lake Erie.
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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. Note the scale difference for Lake Erie.
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 western 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. Note the scale difference for Lake Erie.
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. Note the scale difference for Lake Ontario.
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).
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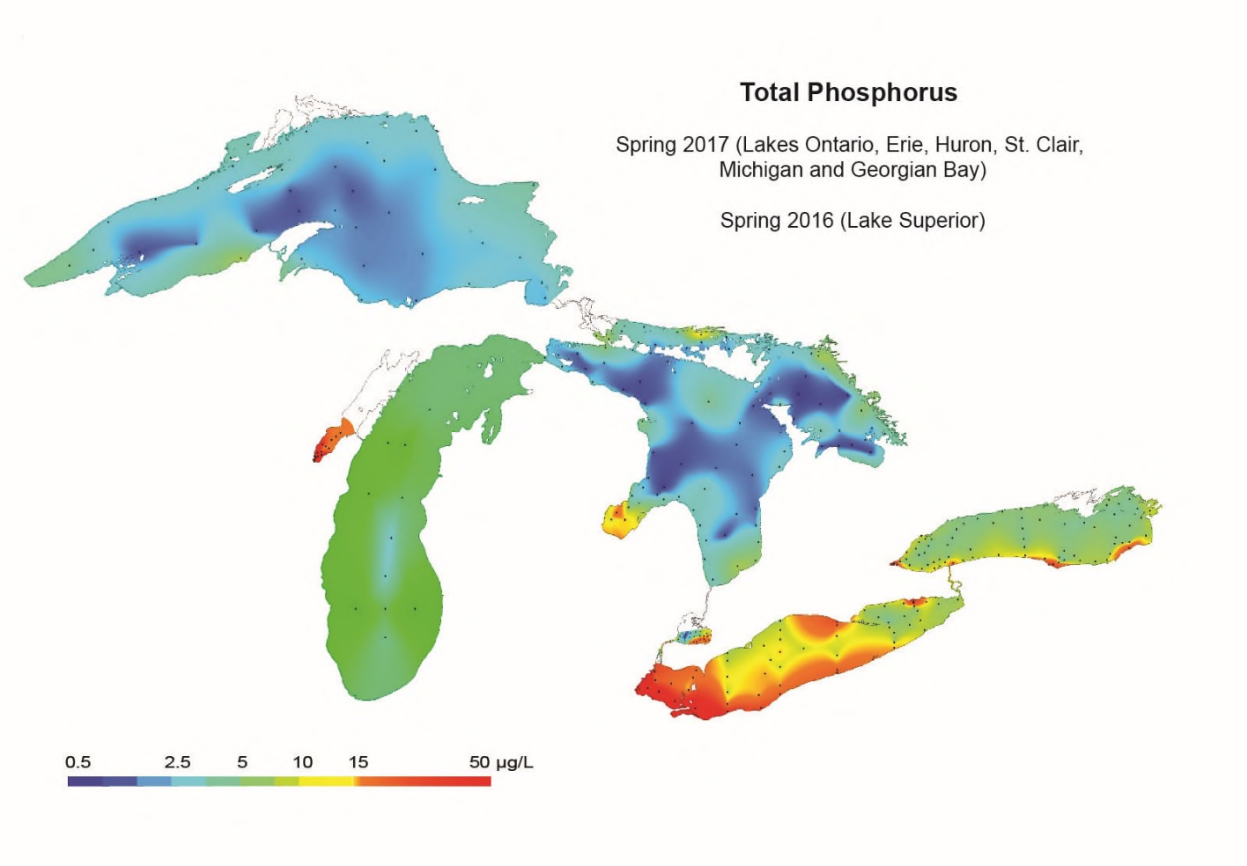


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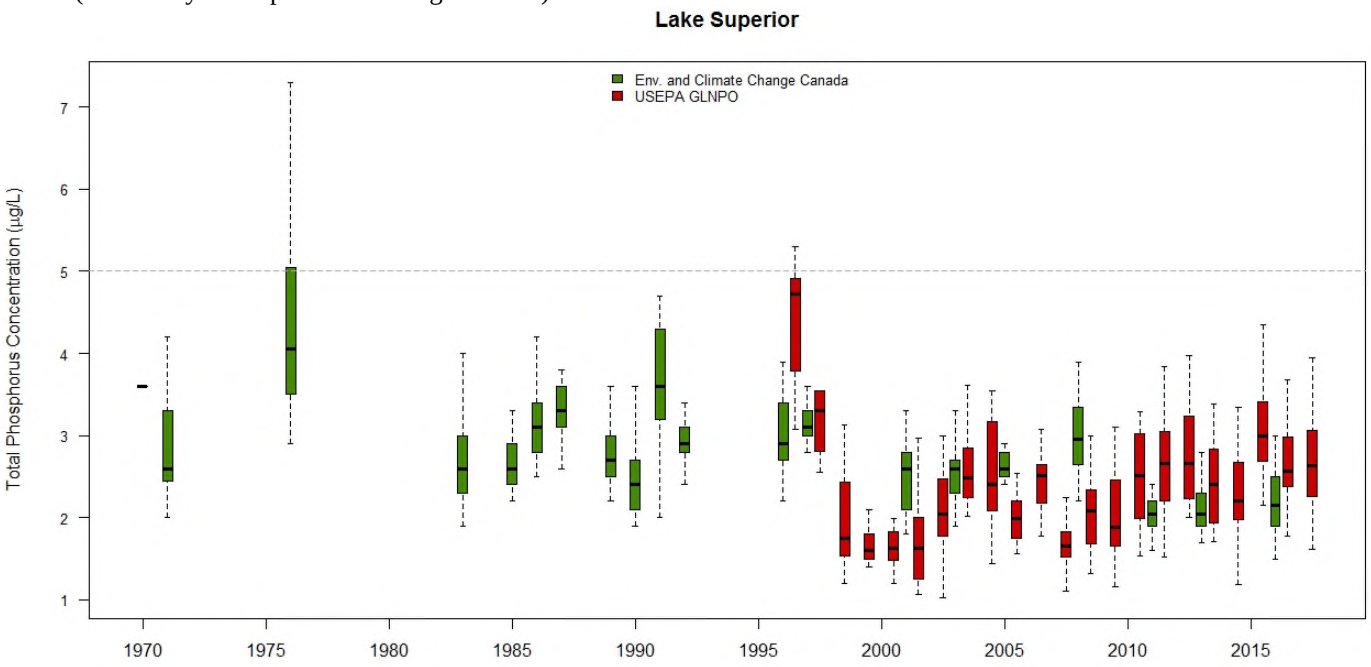


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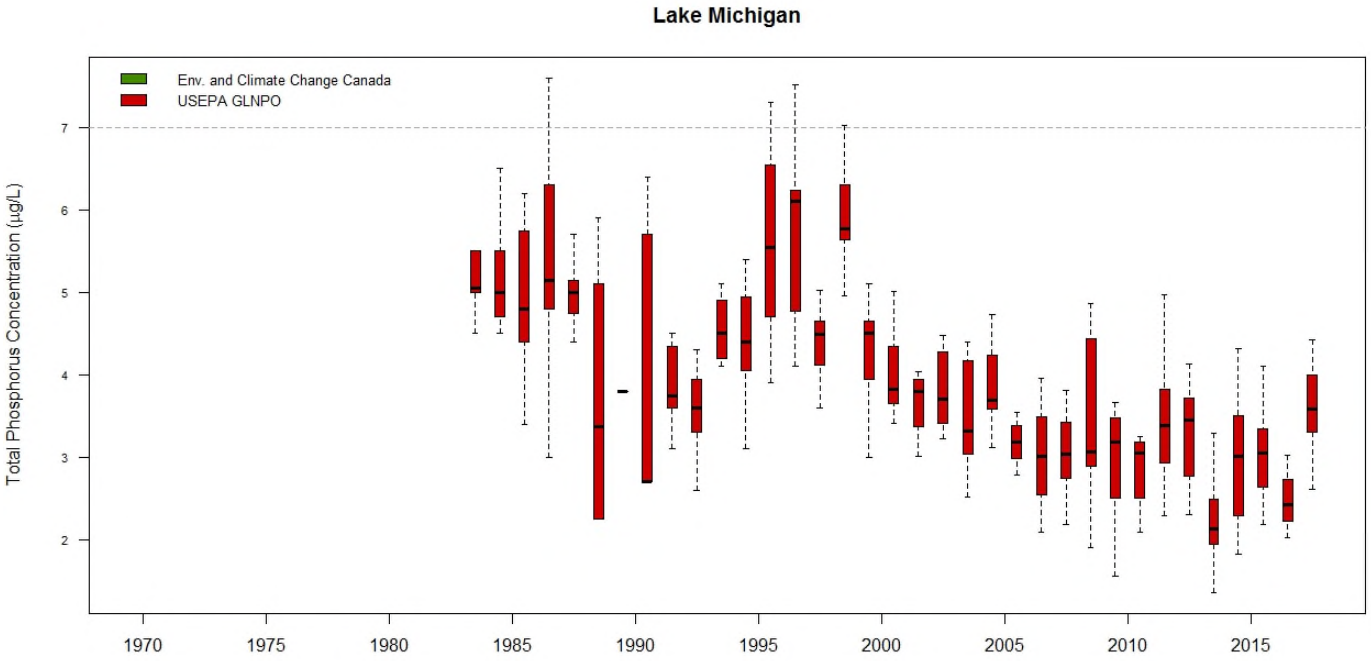


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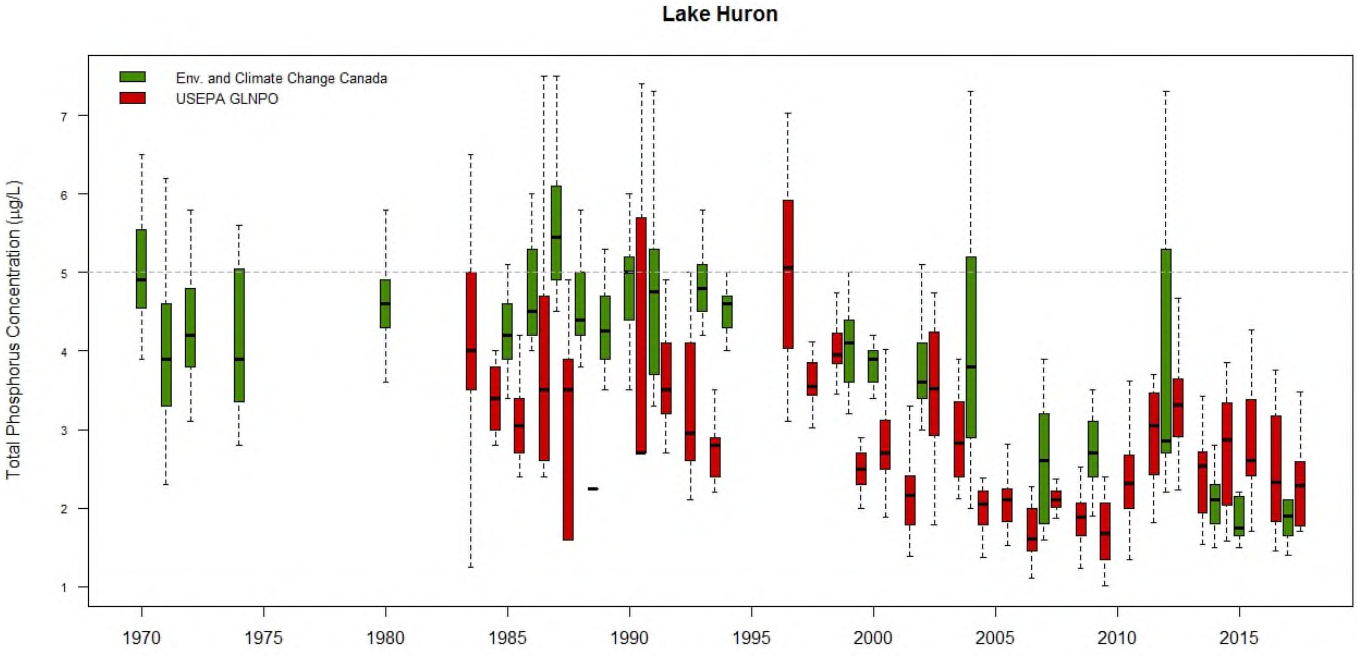


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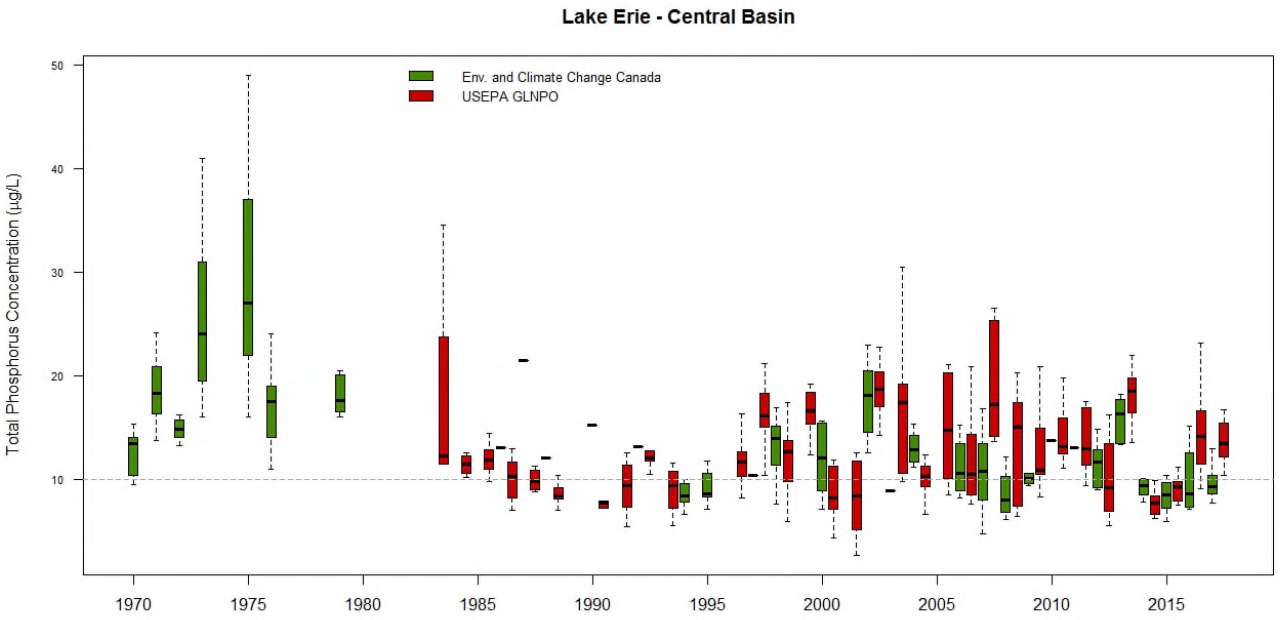


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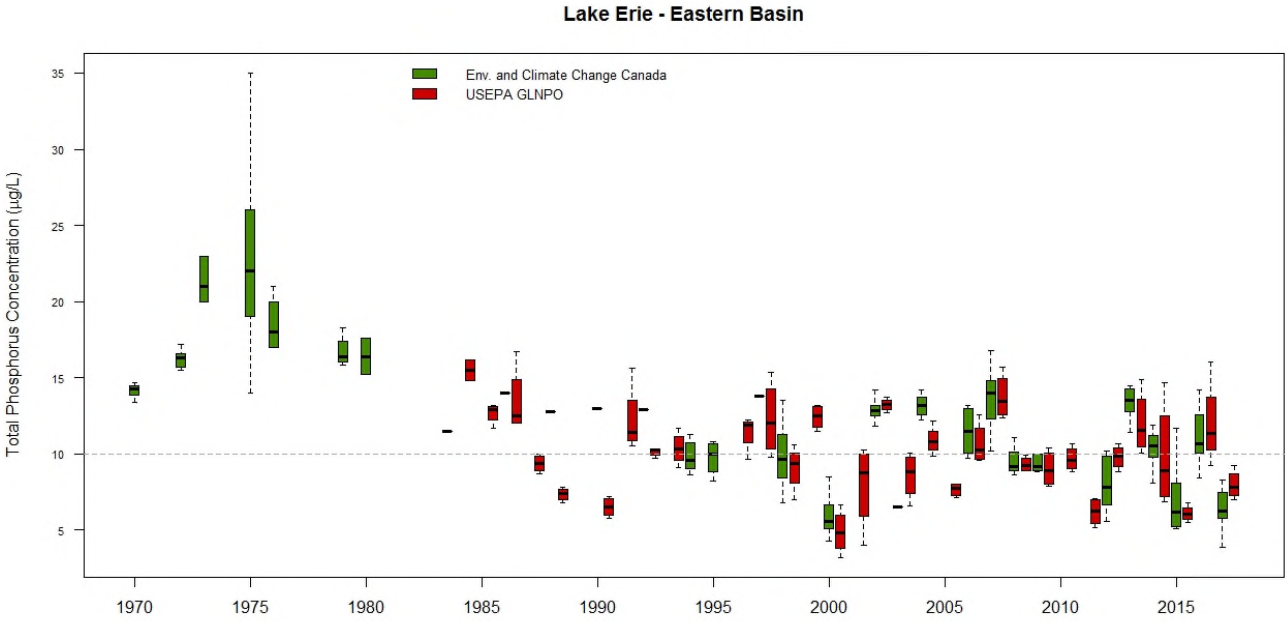


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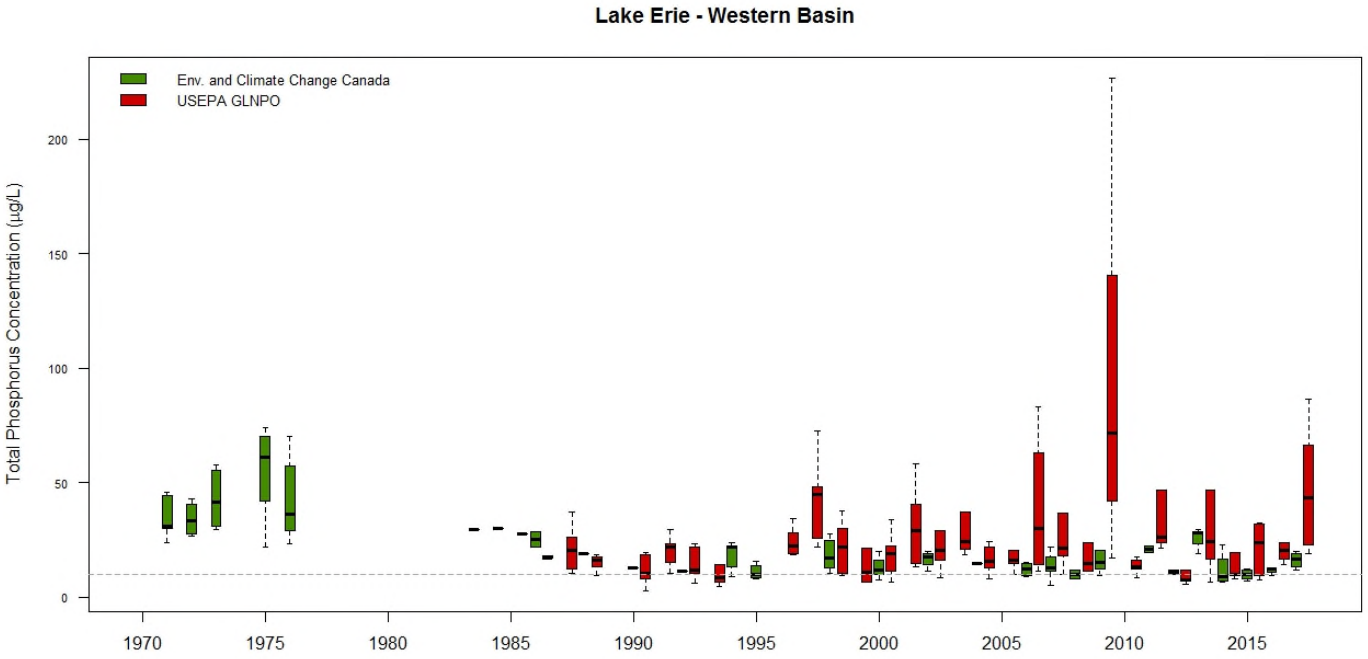


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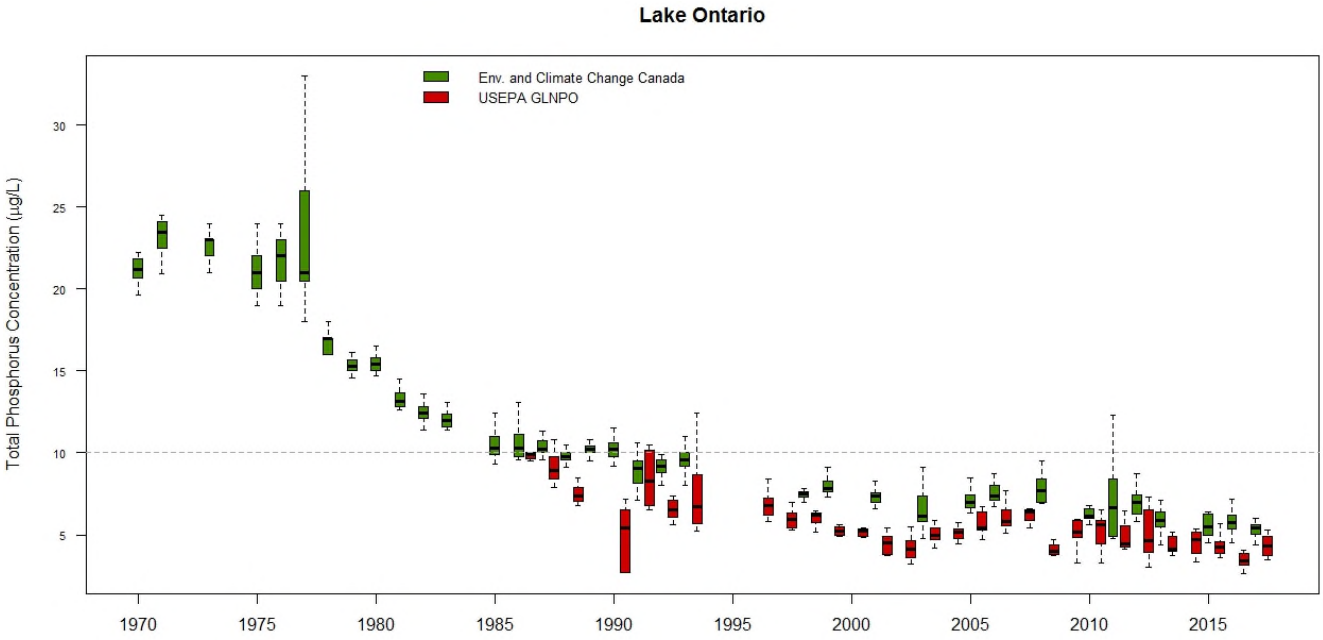


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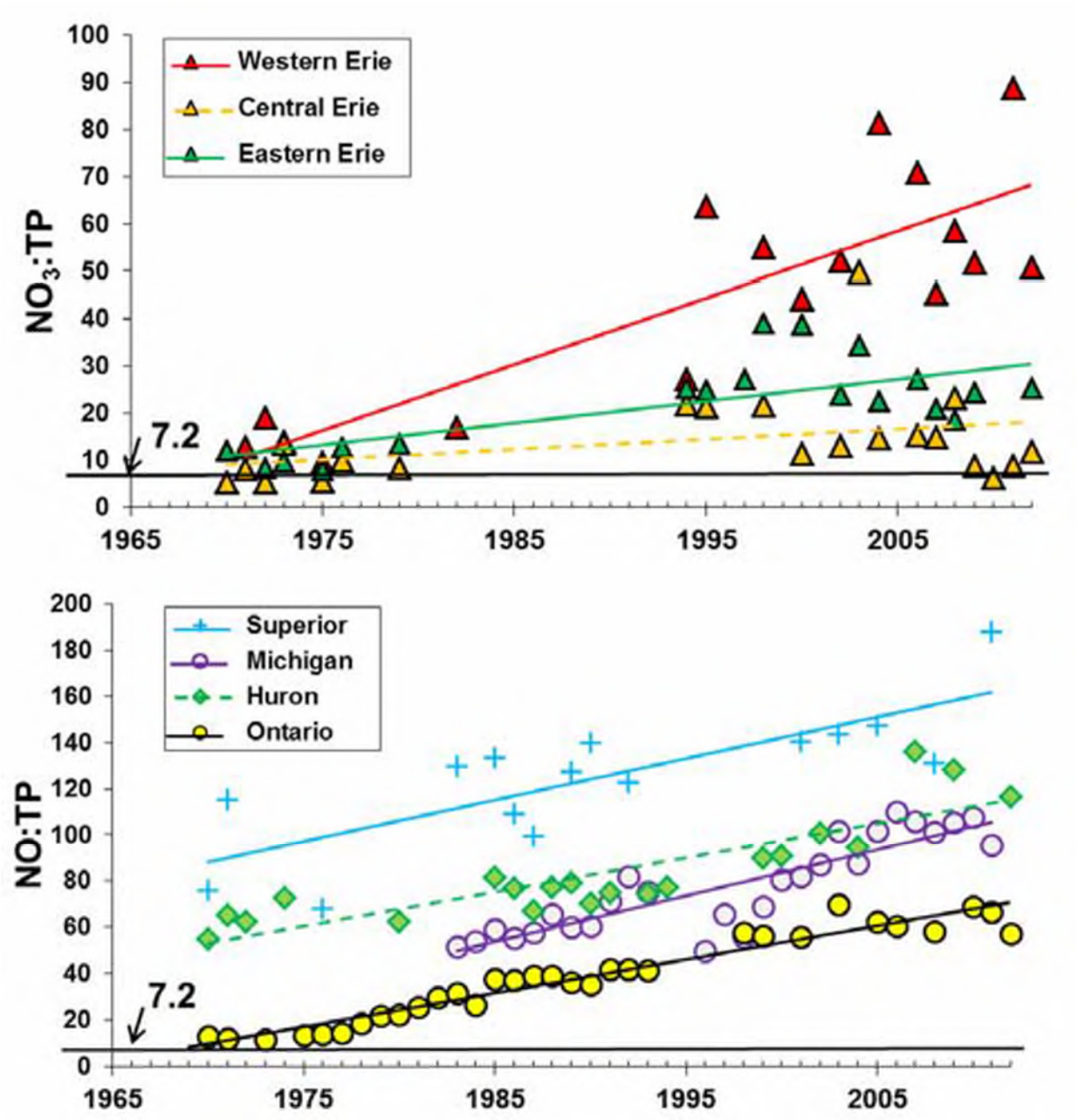


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Data source: Environment and Climate Change Canada and U.S. Environmental Protection Agency

Sub-Indicator: Harmful Algal Blooms
Nearshore

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Undetermined
Long-term Trend (2001-2017): Undetermined
Rationale: There is little systematic monitoring outside of Lake Erie and Lake Ontario to enable a rigorous evaluation of harmful algal blooms (HABs) in the Great Lakes. HABs have become a major issue for the western basin of Lake Erie and to a significant but lesser degree in some eutrophic inshore embayments in Lakes Michigan, Huron, Ontario and St. Clair. Based on available data and professional judgement, the overall status of the Great Lakes in deep offshore waters is generally Good. However, nearshore areas, which is the focus of this sub-indicator, are assessed as Fair and the trend is Deteriorating in some embayments and shallower basins. The overall trend for the Great Lakes is noted as Undetermined for the 10-year period and long-term, due to a lack of spatial coverage in data.

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: Undetermined
Long-term Trend: Undetermined
Rationale: There is little systematic monitoring for HABs (toxic and nuisance) in Lake Superior; however this waterbody is dominated by pico-cyanobacteria that are less likely to produce toxins compared to the larger cyanobacteria that typically dominate more harmful blooms in the Great Lakes.

Lake Michigan
Status: Fair
10-Year Trend: Undetermined
Long-term Trend (2001-2017): Undetermined
Rationale: Offshore waters are generally good but cyanobacteria blooms have been reported in some coastal regions and eutrophic embayments such as Green Bay, Muskegon Bay and in many of the drowned river mouths along the western shore. Nuisance algal blooms and beach fouling by *Cladophora* remains a problem for many of the beaches and nearshore regions.

Lake Huron
Status: Fair
10-Year Trend: Undetermined
Long-term Trend (2001-2017): Undetermined
Rationale: Lake Huron is generally limited by low nutrients in most areas, but experiences toxic and nuisance blooms in some nearshore areas, notably 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 2000 Lake Erie report (Brittain et al. 2000, Vanderploeg et al, 2001).

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (2002-2017): Deteriorating
Rationale: Lake Erie continues to experience toxic and nuisance cyanobacteria blooms throughout the western basin. Blooms in 2011, 2013, 2014, 2015, and 2017 were ranked as Severe in a number of categories (see page 5), and the 2014 event caused the closure of the City of Toledo water supply system. These blooms often expand into the central basin, and have resulted in loss of economic and ecosystem services provided by the lake. Southwest nearshore

areas experience benthic proliferation of the cyanobacteria *Lyngbya* which has been documented elsewhere as a potential source of toxins. Sandusky Bay experiences proliferations of the toxic cyanobacterium *Planktothrix*, as early as May and as late as October with microcystin concentrations observed to exceed recreational guidelines. *Cladophora* remains a nuisance along the northeast shoreline of Lake Erie. Lake St. Clair offshore sites have a low plankton biomass representative of upstream Lake Huron assemblages. HABs with low toxicity were observed at some inshore sites and benthic *Lyngbya* proliferation has been observed on the southeast shoreline. Information on benthic algal abundance is sparse in the other connecting channels like the Detroit River.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend (2007-2017): Deteriorating
Long-term Trend: Undetermined
Rationale: Offshore waters remain good with very little bloom-forming cyanobacterial abundance. Similar to other lakes, increased monitoring has led to more reports of toxicity. Toxicity was first reported in 2007; since that time, toxic and nuisance planktonic blooms have been reported in some embayments on the New York side (Sodus Bay, Port Bay Henderson Bay) and continue to occur in Hamilton Harbour and the Bay of Quinte on the Canadian side where a history of poor water quality has led to the complete closure of at least one of the beaches. New York State has dramatically increased their monitoring activities since 2011 which has led to increased reports of beach closures due to HABs. Nearshore waters continue to experience nuisance algal blooms of *Cladophora*. Monitoring for HABs rarely occurs in riverine systems in Lake Ontario but the occurrence of pelagic blooms is expected to be low due to the higher flow conditions. Benthic and attached algae are an increasing issue in the St. Lawrence River and have been associated with toxicity though the extent of this problem is currently unknown. Information on benthic algal abundance is sparse in the other connecting channels like the Niagara River.

Status Assessment Definitions
Average number of sample sites meeting HABs index
Good: There are no Moderate or Severe ratings in any of the four metrics
Fair: There is one or more Moderate rating in any of the four metrics
Poor: There is one or more Severe rating in any of the four metrics
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.
See Measure section for description of metrics.

Trend Assessment Definitions
Improving: HABs index shows a change (>20%) toward lower frequency of occurrences.
Unchanging: HABs index shows no change (annual variance of +/- 10%).
Deteriorating: HABs index shows a change (>20%) toward higher frequency of occurrences.
Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

Endpoints and/or Targets
The appropriate endpoint is that all lakes should not have any HAB occurrences in offshore waters exceeding 30 meters in depth and up to one occurrence not exceeding 2 weeks in length in nearshore waters or embayments. For the purpose of this target, the central and western basins of Lake Erie would be considered a nearshore water or embayment.

Sub-Indicator Purpose
The purpose of this sub-indicator is to assess potential harm to human health, livestock, 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 human, 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 growth of algae and cyanobacteria that interfere with aquatic ecosystem health, or human use of the ecosystem.”

Measure

Data have historically not been available upon which to assess basin-wide status and lake-specific trends. With increased monitoring, the development of additional remote sensing tools and methods to measure toxicity, the knowledge base has increased concurrently in recent years. Currently, a variety of different sources of information help to inform this assessment; in the future, it will be refined and reassessed. A HABs Index (described below), calculated using up to five different metrics, is currently in development, with the final result being the total of all the individual metric values, each rated according to the following scale. Although the knowledge-base has improved, gaps remain in applying the metrics to many regions and insufficient time series data are available yet to rigorously assess trends.

Table 1. Harmful Algal Bloom Index Rating

Rating	**Metric 1a total Chl-a* µg/L**	**Metric 1b Chl-a (µg/L) with > 60% cyanobacteria** (% biomass or % chl-a)**	**Metric 2 total microcystins µg/L**	**Metric 3 and 4 Taste & Odor reports or beach postings**
Good	<20	< 10	< 1.5 µg/L; < 30 µg/g dwt	None
Moderate	20-30	10-20	1.5-20 µg/L;30-300 µg/g dwt	1-2 occurrences
Severe	> 30	> 20	> 20 µg/L; > 300 µg/g dwt	Three or more

*based on Carlson, 1977
** based on Downing et al. 2001

Metric 1: Pelagic algal biomass dominated by cyanobacteria[i]. This metric will be considered moderate for algal abundance if the total chlorophyll-a content of the water exceeds 20 µg/L (or > 10 µg/L with > 60% cyanobacteria[ii]) or severe for algal abundance if total chl-a exceeds 30 µg/L[iii] (or > 20 µg/L with > 60% cyanobacteria). The resulting index is the number of samples collected at a given site or within a lake or basin between June-November that rate positive for algal abundance. Total algal biomass can be assessed using remote sensing estimates of chlorophyll-a.

Metric 2: Pelagic cHAB toxicity. This metric uses one specific class of cyanobacteria toxin, microcystins, measured from samples collected at a monitoring site between June-November. Protocols to measure microcystins are not specified[iv]. The index is based on the occurrence of one or more samples at the site where total microcystin concentrations in the water column[v] exceed either 1.5 µg/L (moderate) or 20 µg/L (severe)[vi].

Metric 3: Pelagic algal biomass with offensive malodour or taste. This metric will be positive if the water sample (i) has measured levels of the two most common cyanobacterial taste-odour compounds (geosmin and/or 2-MIB) at levels exceeding 15 ng/L or (ii) has a level of malodour or taste consistent with algal/cyanobacterial sources (i.e. not derived from chlorine etc.) that is deemed to be unacceptable to a sensory screening.

Metric 4: Benthic HAB resulting in beach closures or similar negative impact on the shoreline (e.g. lost tourist revenue possibly due to toxins or malodours, as evaluated above). The Benthic HAB index will be positive if government or similar regulatory agency issues a limited use warning due to accumulation of algal material on the shoreline. As above, the resulting index is the number of samples collected between June-November that rate positive.

Metric 5 (optional): Pelagic or benthic algal blooms with other harmful effects measured using quantitative or qualitative criteria; these are usually site-specific (e.g. closure of cooling water intakes, lost property value, anoxia, fish kills etc.) The resulting index is number of documented events between June-November that rate positive for one or more of these harmful effects.

- This sub-indicator uses a HABs Index which can yield one of the three ratings (see Table 1, below):

  1=Good; no occurrences of chl-a > 20 µg/L, or >10 µg/L with >60% cyanobacterial dominance, no taste and odor events or beach closures due to HABs or cHABs, and all total microcystin concentrations are below 1.5 µg/L[vii] or 30 µg / g[viii] dry weight.

2=Moderate; One or more observations with microcystin concentrations between 1.5 – 20 µg/L (or between 30 -300 µg/gram dry weight), OR the occurrence of one or more observations with chlorophyll-a between 20-30 µg/L for pelagic samples and moderate cyanobacterial dominance (20-80%), one or more algal/cyanobacterial related taste and odor event, OR 1-2 beach (or other) advisories generated by an appropriate agency due to high levels of algal or cyanobacterial biomass.

3=Severe; The occurrence of one or more observations with microcystin concentrations > 20 µg/L or > 300 µg/gram dry weight, OR the occurrence of one or more observations that have chlorophyll-a > 30 µg/L for pelagic samples and dominance (> ~80%) of the biota by cyanobacteria or are associated with a taste and odor event, OR three or more limited-use beach or other advisory generated by an appropriate agency due to high levels of algal or cyanobacterial biomass.

As there are few monitoring programs specifically targeting HABs, the HAB index will draw from information from a variety of sources. This includes data from biweekly samples (or more frequently during high risk periods) taken at high risk and reference monitoring site**s** from discreet surface (0.5-1m) samples, euphotic zone integrated samples, autonomous monitoring systems such as buoys or remotely operated vehicles, or appropriate satellite imagery indicating chlorophyll or cyanobacteria biomass (e.g. Stumpf et al. 2012; Binding et al. 2018) obtained between June and November. The sampling period should be adapted to local conditions and previous records of harmful and nuisance algae seasonality, and may be expanded to include additional time periods (e.g. winter (under ice) sampling) if future information warrants this change.

Ecological Condition

Background

Harmful algal blooms for this report include: i) cyanobacteria-based harmful algal blooms (cHABs): e.g. blooms that are documented to contain cyanobacterial toxins or are dominated by cyanobacteria species with the genetic potential to produce toxins, and ii) non-toxic or poorly described toxigenic, nuisance algal blooms (NABs) e.g. episodes of high algal/cyanobacterial biomass that, while not documented to contain toxins, disrupt ecosystem services provided by the water body.

Harmful cyanobacterial and/or algal blooms (HABs) are typically a global issue in eutrophic waters with high nutrient loadings. HABs can be differentiated from 'non-harmful' (i.e. nuisance) blooms by their impacts on water quality and the associated 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 toxin. HABs and NABs can have detrimental impacts on ecosystem services provided by the lake and negatively impact aesthetics or recreational use of the water body. Prior to remediation in the late 1970s, HABs and NABs were a major problem in many offshore and nearshore areas in the Great Lakes (e.g. Watson et al. 2008) 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). This progress changed in 2000 with the identification of the toxins produced by blooms of *Microcystis* in western Lake Erie (Brittain et al. 2000). Because toxin production was not generally recognized as a threat to the Great Lakes in the 1970s, there are no historical data on their occurrence prior to 2000. It is now recognized that many genera of bloom-forming cyanobacteria contain both toxic and non-toxic species and within species there are toxic and non-toxic strains. Differentiation between the toxic and non-toxic counterparts therefore is not possible at the level of light microscopy. Current management approaches which target planktonic (subsurface) chl-a as a measure of total algal biomass and productivity may be using a poor metric for these events. Newer analytical techniques specifically targeting the pigments produced by cyanobacteria (phycocyanin and phycoerythrin) provide a better measure of cyanobacteria biomass, but again cannot differentiate between toxic and non-toxic species. Recognizing this issue, many agencies now specifically test for the hepatotoxin microcystins, a class of related compounds produced by toxic members of the genus *Microcystis*, *Planktothrix* and *Dolichospermum* (syn. *Anabaena*) and extensive monitoring and bloom forecasting programs now exist for Lake Erie, Lake St. Clair and some of the embayments in lakes Huron and Ontario (e.g. http://www.glerl.noaa.gov/res/HABs_and_Hypoxia/habsMon.html; https://www.hamilton.ca/parks-recreation/parks-trails-and-beaches/beach-water-quality-in-hamilton) .

Most efforts are focused on visible HABs caused by planktonic toxic cyanobacteria, but HABs also can be caused by benthic/littoral macroalgae. These benthic mats, along with planktonic outbreaks, have shown an apparent resurgence in the late 1970s and early 1980s, particularly in the lower Great Lakes. Because these events are often episodic, and vary seasonally and inter-annually in severity and spatial coverage, it is difficult to implement appropriate research, monitoring and management programs, particularly in large and complex waterbodies such as the Great Lakes where sampling is often subject to weather and vessel access. These blooms are not restricted to the lakes themselves and have been reported in major embayments, tributaries and connecting channels.

Most algal blooms in the Great Lakes are reported in the nearshore areas, which are most prone to shoreline development issues, greater influx of nutrients and to some extent, increased public awareness. The size of nearshore zones varies from approximately 1-10% in Superior to 60-90% in Erie, as does the influence of physical and climatic factors (runoff, erosion, thermal bar formation, upwelling/down-welling, alongshore/nearshore/offshore currents, circulation patterns, surface/ground water inputs, lake level regulation, ice formation, etc). As a result, the nearshore zones are highly dynamic, and there is significant spatiotemporal variance in the areas supporting littoral and planktonic communities and offshore-nearshore material exchange.

Key aspects of HABs

These are summarized in detail in Watson and Boyer (2008), but some key points are below:

- HABs cause significant economic harm. Annual estimates vary, but range up to annual 4.6 billion USD/year in the USA including monitoring, fisheries, tourism, public health & advisory, lost revenue and property value (Anderson et al. 2000). For the Lake Erie basin alone, a recent report estimated that the major 2011 HAB event cost approximately 71 million USD; the smaller 2014 HAB event cost approximately 65 million USD (Bingham et al. 2015). Lost benefits of 1.3-2.2 billion dollars are predicted over the next 30 years if no management action is taken to control the blooms, a cost which could be reduced by 60-75% if remedial action is taken (Bingham et al. 2015; Smith et al. 2015).
- Not all HABs resemble green paint or pea soup. They are caused by many species, and vary in colour from green to red and brown. Algal blooms do not always appear as surface scums, and can be difficult to identify or anticipate. Some blooms are mixed through the water column, or grow in deep water layers, under ice or as benthic/attached mats.
- Cyanobacteria produce many toxins which fall into three major categories, based on their activity: liver toxins (hepatotoxins), neurotoxins and dermal toxins/irritants. These toxins vary greatly in their chemical properties, stability and toxicity. The hepatotoxic microcystins (MCs) are the most stable and prevalent across the Great Lakes. These toxins can persist in the water column after a bloom has died and disappeared. Many cyanobacteria and several classes of algae produce volatile organic compounds (VOCs) that cause unpleasant taste and odour in drinking water supplies, but measures of toxins, taste and odour, visible blooms, cyanobacteria and algal biomass, and chl-a may or may not be correlated. Toxins are odourless and colourless and are often poorly related to malodorous VOCs, which are derived from different biochemical pathways. Both classes of compounds are produced by a diversity of cyanobacteria and algal taxa, and vary in cell production with environmental conditions and growth stage, both among and within species.
- The term ‘algal bloom’ is a non-quantitative descriptor for visible increases in free-floating or attached algal/cyanobacterial density, often manifested as scums, mats or water colour. (see e.g. Watson and Molot 2012). Blooms are difficult to define, measure and predict. Blooms can show rapid changes in their spatial location and abundance. With calm conditions (or overnight), buoyancy-regulating cyanobacteria can float to the surface and be carried large distances by wind/waves. These may wash onshore, creating patches of very high toxin levels along beaches. Variations in sampling and analytical methods can lead to inconsistencies in the reported levels of these compounds.
- Fluorescence-based, cell counts and other abundance measures (e.g. molecular, biochemical) are often poorly correlated with each other and actual cell biomass, due to wide variance in pigment content, photoacclimation and cell composition. Taxonomic identification of many of the responsible species is complex, leading to differences between analysts.
- Harmful algal blooms are differentiated as having harmful socioeconomic or ecological effects, and may be caused by algal/cyanobacteria species belonging to different taxonomic groups. The greatest concern is with HABs caused by cyanobacterial species with the capacity to produce one or more toxins (neurotoxins, hepatotoxins or dermal toxins/irritants). Currently cyanobacteria are the only known sources of algal toxins in inland waters that directly affect humans. Detrimental health effects from benthic algal accumulations on the shore are more difficult to quantify but may result in significant socioeconomic and ecological damage.

Current State of HABs in the Great Lakes

Although a HABs Index has been developed for this sub-indicator, there are not enough data available to use this index to assess status and trends. There are few long-term data or rigorous monitoring programs in place outside of Lake Erie and Lake Ontario, and only a qualitative assessment of the current status in each lake can be made at this time. Recent efforts to use satellite imagery for measuring and quantifying HABs including NABs are increasing (e.g. Stumpf et al. 2012, Binding et al. 2018), and offer one potential approach for future assessments.

Lake Superior: There is very little quantitative and current information on HABs in Lake Superior. Severe HAB outbreaks have not been documented recently in this lake and the offshore waters are generally dominated by non-toxic pico-cyanobacteria. Algal biomass, especially for potentially toxic cyanobacterial species remains mostly at low levels, although there may be some local impairment near shoreline development. Localized, low toxicity blooms have been observed in the tributaries across the Keweenaw Peninsula

Lake Michigan: 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, where there has also been an increase in hypoxia (e.g. de Stasio et al. 2014). Satellite-derived chlorophyll-a imagery in 2017 classified algal biomass as Severe in the southern-most sections of Green Bay (Figure 3). Nuisance algal blooms and beach fouling by *Cladophora* remains an issue for many of the beaches and nearshore regions, especially along the western shoreline and in the area of Sleeping Bear Dunes.

Lake Huron: Lake Huron is generally oligotrophic throughout the offshore and many nearshore regions, but experiences potentially toxic cHABs in some embayments, notably Saginaw Bay which develops toxic summer outbreaks of *Microcystis aeruginosa* (see http://www.glerl.noaa.gov/res/HABs_and_Hypoxia/SBMicrocystin.html). Moderate to Severe algal biomass was detected in 2017 satellite imagery within Saginaw Bay, particularly along the southern shores. Blooms have also been reported in Sturgeon Bay (Georgian Bay) since the early 2000s and are largely dominated by Nitrogen-fixing cyanobacteria (Vershoor et al, 2017). The genetic potential for microcystin production exists but levels of these toxins have been low/undetectable (Duggan and Chambers 2017).

St. Clair River/Lake St. Clair/Detroit River: Seasonal sampling along the south shore from the Thames River to the outflow of the Detroit River into Lake Erie showed high microcystin levels near the Thames mouth, from blooms dominated by *Microcystis* (Davis et al. 2014). *Lyngbya* mats were reported in 2015 along the eastern shoreline (Vijayavel et al. 2013). NASA and NOAA Coast Watch satellite imagery showed extensive algal blooms again in 2015 that covered much of the southern areas of Lake St. Clair, but no toxicity monitoring was performed. Environment and Climate Change Canada satellite imagery from 2017 detected Moderate to Severe algal biomass in the south east of Lake St. Clair. In 2016 and 2017, a major monitoring initiative on the Canadian side found no detectable toxicity (2016) or very low toxicity (2017). Nearshore sites vulnerable to HABs have been recently incorporated into the NOAA/GLERL/ECCC tracking and forecasting system; see http://www.glerl.noaa.gov/res/HABs_and_Hypoxia/STCMicrocystin.html.

Lake Erie: Lake Erie is the most heavily impaired by planktonic cHABs, particularly over the last few years when satellite images of extensive surface blooms of *Microcystis* and other cHAB species such as *Dolichospermum* have been widely posted (e.g. NOAA; http://coastwatch.glerl.noaa.gov). Imagery from the 2017 bloom showed the majority of the western basin (with the exception of the Detroit River plume) to have severe algal biomass conditions. Sandusky Bay, as well as intermittent points along the south shores of the central basin were also classified as Severe. Currently, highly resolved data on chlorophyll and toxin levels in the west basin are available online (http://www.glerl.noaa.gov/res/HABs_and_Hypoxia/WLEMicrocystin.html) along with a 'HAB tracker' and weekly HAB bulletins posted (http://www2.nccos.noaa.gov/coast/lakeerie/bulletin/bulletin_current.pdf).

Data indicates a high year-to-year variability in bloom intensity, coverage, and timing with a general deterioration in overall status since about 2007 (Figure 1). During eight (8) out of the last ten years, there has been a significant harmful algal bloom in the western basin of Lake Erie. Shifts in the physical/chemical/ biological regimes (e.g. Michalek et al. 2013; Watson et al. 2016) are evident – notably in the western basin. Overall, satellite imagery indicates an increase in the severity of cyanobacteria blooms in the western basin (Figure 1) and some nearshore areas of the north shore (Point Pelee, Rondeau Bay, Long Point), and a general decline or no change in overall chl-a and total and/or eutrophic species biomass in the offshore regions of the central and eastern basins (based on water testing). Immense surface blooms (>20 km^2) are now annually recorded in the western basin of Lake Erie near the Maumee and Sandusky Rivers (e.g. Stumpf et al. 2012; Michalek et al. 2013; Steffen et al. 2014; Watson et al. 2016). Microcystins are the most common cyanobacterial toxins measured in Lake Erie. Data from 2000–2004 measured a wide

range in microcystin levels from detection limits (in 2002) to >20 µg/L (in 2003). Water quality data from 2017 indicated microcystin concentrations exceeding recreational guidelines between August and October. Toxicity is not restricted to the western basin and has been reported in Sandusky Harbor, Presque Isle and in Long Point Bay. Neurotoxins (anatoxin–a, saxitoxin, neosaxitoxin) occurred at or near detection limits in the open lake waters. Samples collected throughout the lake since 2003 show the greatest proportion of samples with detectable MC levels from the western basin, although only a small fraction of these samples exceed the drinking water guidelines of 1.5 µg/L and even fewer exceed the recreational contact level of 20 µg/L.

Wind driven material from west basin blooms intermittently impair northern shorelines and offshore in the central basin (e.g. Figure 2) – although some of these events may be of local origin e.g. near Point Pelee. Blooms are frequently dominated by potentially toxic non-nitrogen-fixers, notably *Microcystis* and *Planktothrix* spp., suggesting increased nitrogen loading or dreissenid activity, although significant blooms of nitrogen-fixers (*Dolichospermum* and *Aphanizomenon)* also occur in both western and eastern basins (Allinger and Reavie 2013). Severe impairments by thick mats of the cyanobacterium *Lyngbya wollei* reported in the mouth of the Maumee River between 2006-2009 appear to have abated (Western Lake Erie Waterkeeper Association unpublished). However, extensive mats of attached green algae, notably *Cladophora* are showing high abundance along some northern shorelines (see *Cladophora* sub-indicator).

Most impairment occurs at shorelines and beaches and can be manifested as fish/bird kills. Lyngbya toxins (inflammatory/vesicatory and tumour-promoting) were not detected in the mats of *Lyngbya wollei* proliferating in the Maumee and Detroit rivers. Geosmin and 2-methylisoborneol (MIB) occur in several areas of the lake (Kutovaya and Watson 2014) and are likely the cause of annual musty-muddy odour problems in drinking water in supplies in the western basin (e.g. Toledo). Significant odour is also produced by extensive rotting mats of shoreline attached algae. In 2014, a *Microcystis* bloom in the western basin of Lake Erie near the Collins Park Water Treatment facility serving the City of Toledo resulted in levels of microcystin toxin in the finished drinking water in excess of 2.5 µg/L, which is significantly higher than the 1.0 µg/L drinking water guideline. This resulted in the City of Toledo being placed under an emergency drinking water decree and severely disrupted city services for nearly 500,000 residents for a period of 56 hours.

Lake Ontario: Blooms of cyanobacteria and related impairments (toxins, shoreline fouling, taste and odour) occur on an annual basis in some nearshore areas, notably Areas of Concern (AOCs) in Lake Ontario. Outbreaks of high MC levels associated with cyanobacteria blooms have been recorded most years in Hamilton Harbour, Bay of Quinte, Oswego Harbor and the southern shore embayments of New York (Watson and Boyer 2008, Perri et al. 2015). Beach closures related to toxic cHABs occur annually in Hamilton Harbour, where the City of Hamilton in 2014, has established a systematic beach monitoring program which includes microcystin toxin testing. Several beaches were also closed on the New York side due to the visual presence of a cyanobacterial bloom. New York State Department of Health and Ontario Public Health Units do not currently use toxicity testing to determine public beach closures.

Spatiotemporal dynamics of MCs in the Bay of Quinte, Hamilton Harbour, Oswego Harbor (now delisted), Sodus Bay, and the Rochester embayments indicate periods of severe impairment of nearshore sites by windblown accumulations of toxic material, where MC levels can reach levels in excess of 500 µg/L (Watson et al. 2009). MC and toxigenic *Microcystis* are also commonly found in many of the nearshore regions and embayments that span the northern coast of New York State (Perri et al. 2015). While MCs are certainly the toxin of most concern in Lake Ontario, recent surveys indicate the widespread occurrence of low concentrations of anatoxin-a in nearshore embayments (Boyer 2007, Perri et al 2015). The organism responsible for anatoxin-a production is currently unidentified. Cylindrospermopsins have not been detected (New York State Department of Conservation Citizens Statewide Lake Assessment Program).

Connecting Channels

There are few studies or reports of HABs in connecting channels although a number of papers report significant blooms in tributaries to Lake Erie (Maumee, Sandusky; e.g. Kutovaya et al. 2012; Davis et al. 2015). Toxin-producing *Microcystis* blooms have been reported recently in the Detroit River, likely derived from upstream blooms in Lake St. Clair (Davis et al. 2014) although MC levels reported to date have been below the World Health Organization guideline. Anatoxin-a has been recently reported in the Thames River at levels near detection and Planktothrix is suspected to be responsible (MOECC pers. comm. 2017). Toxin-producing (saxitoxin analogues) and taste and odour producing *Lyngbya* has been reported in the St. Lawrence River (e.g. Lajeunesse et al. 2012) and embayments of Lake Ontario and Lake Erie (New York State Department of Conservation pers. comm. 2017), along

with frequent impairments of drinking water and shoreline odour produced by benthic and epiphytic cyanobacteria (Watson et al. 2008).

Linkages

Current knowledge indicates that climate is likely to exacerbate the magnitude and frequency of harmful algal blooms, mainly because of warming water temperatures (O'Neil et al., 2012). Additional linkages include (but are not limited to):

- *Tributary Flashiness, Precipitation Amounts, Water Levels, Surface Water Temperature and Ice Cover* – Timing and magnitude of pulses of intense nutrient loading, higher water temperatures and reduced ice cover may promote earlier and more extensive blooms; changes in water column stratification/stability may favour large buoyancy-regulating cHABs and the development of bottom hypoxia, exacerbating the bloom-promoting release of bioavailable nutrients from the sediments.
- *Nutrients in Lakes* (driving HABs and cHABs development).
- *Beach Advisories* - cHABs and NABs are an increasing cause of these advisories[ix], although the provincial recreational beach water quality guideline uses *E.coli* concentrations to determine the status of beach water quality.
- *Cladophora* (and other NABS e.g. *Lyngbya*)
- *Invasive species*, notably *Dreissenid Mussels* that increase water clarity and recycle bioavailable nutrients in the inshore regions
- *Treated Drinking Water* – HABs are known to affect drinking water with taste and odour compounds and toxins (cyanobacteria) as well as decrease the treatment capacity due to interference from high biomass.
- *Phytoplankton and Zooplankton* (HABs result in dramatic decreases in biodiversity, integrity and stability of these groups).

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 Limitations

The sources of data are varied and in many cases, use different sampling and analytical methods. Monitoring in the lower lakes is generally Good but monitoring in the upper lakes (Michigan, Huron and Superior) is sparse and largely reactive to observations and reports. Increasingly, the data are validated and quality controlled by recognized agencies e.g. National Oceanic and Atmospheric Administration-Great Lakes Environmental Research Laboratory, State University of New York, Environment and Climate Change Canada and U.S. Geological Survey.

1. Trend analysis is limited by the fact that there are few long-term data collected on cHABs and more specifically, toxins, in the Great Lakes. In addition, there are few regular, systematic monitoring programs, although some agencies have recently established sampling programs for cHABs and related water quality data (e.g. nutrients) in severely impaired areas such as the west basin of Lake Erie – which can bias lake-wide assessments where non-impaired areas are infrequently monitored[x]. Other alternative data sources have considerable potential, such as the increasing use of genomic tools to track the composition and functional potential of these events in concert with limnological conditions to elucidate drivers. Widely available source-water measurements of chl-a, cell numbers, microcystin and taste and odour compounds are also that are increasingly collected by drinking water utilities (e.g. Ontario Drinking Water Surveillance Program[xi]).

2. This sub-indicator is extremely sensitive to sampling frequency. Intensive sampling in one area (e.g. western

basin of Lake Erie) increases the likelihood of a Severe rating relative to regions that have more limited sampling efforts for cHABs (e.g. open waters of eastern Lake Erie, Lake Michigan, Superior, Huron and Ontario). Sampling regimes in the Great Lakes are often sparse (both temporally and spatially) and are likely to miss spatial and temporal peaks in cyanobacterial/algal abundance. Informed use of satellite imagery is encouraged to optimize sampling schedules and capture bloom events that may be otherwise missed. cHABs are generally episodic, subject to significant transport by wind and water movement and often not captured by low intensity fixed position sampling. Shoreline scums often represent a concentrated sample and as such, can be more indicative of wind and weather events than of cHAB density in the water column. Samples from these sites should be normalized to wet weight or chlorophyll, to mitigate bias under these conditions.

3. Differences in sampling regimes and analytical protocols (e.g. surface or integrated sampling; taxa enumeration; toxin analyses) used by individual studies affect the ability to compare data and evaluate long-term trends in toxins and bloom occurrences.

4. Public attention is most often focussed on shoreline scums or algal material visible at the surface, particularly for inland waters where many reported blooms are caused by attached macroalgae *(Cladophora, Lyngbya*) or large, buoyancy-regulating cyanobacteria. These buoyancy-regulating taxa can produce rapid surface accumulations from populations through the mixed layer or deep living/benthic populations. Concentrated surface scums appear/disappear and migrate rapidly with changes in vertical mixing, currents and wind activity. These can produce rapid changes in toxin levels along a waterfront or cover extensive areas in large lakes, and are difficult to sample, quantify or predict. Beach and shoreline sampling programmes require multiple sub-sites to capture this spatial/temporal variance in risk and impairment, which are poorly represented by basin-wide seasonal means.

5. The cHAB metric currently focuses on a single class of cyanobacterial toxins (microcystins) that have been shown to be the most abundant in the Great Lakes region (Boyer 2007). Other toxins such as anatoxin-a, cylindrospermopsin and the paralytic shellfish toxins, such as saxitoxin, may be important at specific sites. These toxins may need to be considered as new information becomes available.

6. Satellite observations of HABs are limited by cloud cover and the spatial, temporal and spectral resolution of the satellite sensors. Satellites typically detect only surface algal biomass so imagery should be interpreted with knowledge of the vertical distribution of algal cells as determined by wind mixing and species composition. While satellite imagery provides a good estimate of the extent of a surface bloom, it is not yet possible to determine if the bloom is harmful (produces toxins or taste and odor compounds) based on the satellite imagery.

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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List of Figures

Figure 1. NOAA-GLERL Bloom severity index for Western Lake Erie, 2002-2017, based on the amount of biomass over the peak 30-days.
Source: NOAA-GLERL Experimental Harmful Algal Bloom Bulletin;
https://www.glerl.noaa.gov//res/HABs_and_Hypoxia/lakeErieHABArchive/bulletin_2015-027.pdf

Figure 2. (a) Sentinel-3 OLCI true colour composite of bloom conditions on September 23rd 2017 (b) derived chlorophyll-a at the peak bloom severity (presented as a 14 day rolling average in order to limit bias introduced by cloud cover) (c) number of satellite observations where chlorophyll-a >20μg/L (i.e. moderate or severe biomass) between

June and October. Number of cloud free observations per pixel was approximately 80 over this time period, and (d) time series of ECCC bloom severity index (the product of bloom spatial extent and average chlorophyll-content) derived from the OLCI satellite sensor for the 2017 bloom season, marking the start and peak bloom conditions.
Figure 3. Classification of Great Lakes satellite-derived chlorophyll-a according to metric 1a in Table 1. One or more detected instance of chlorophyll-a 20-30 µg/L (Moderate) and >30 µg/L (Severe) during April-October imagery.

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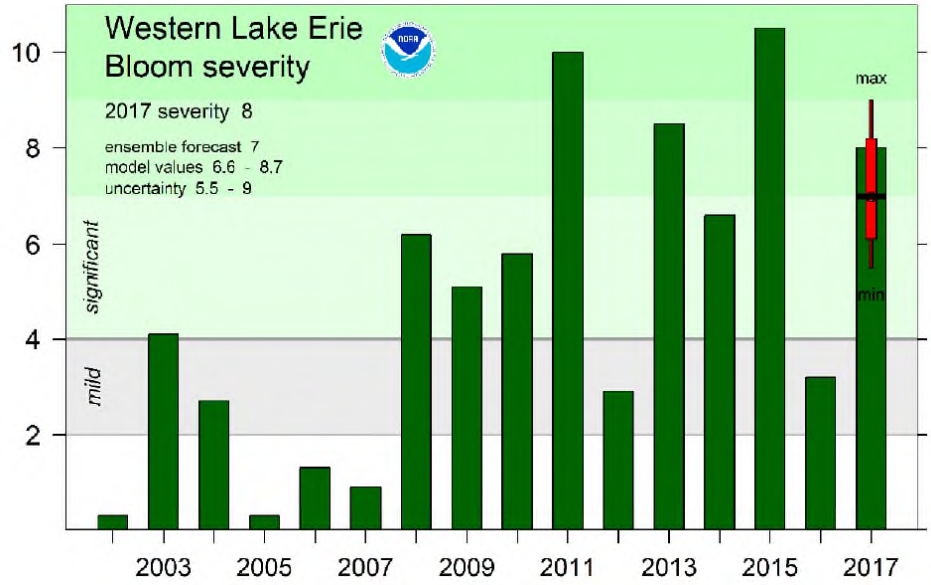


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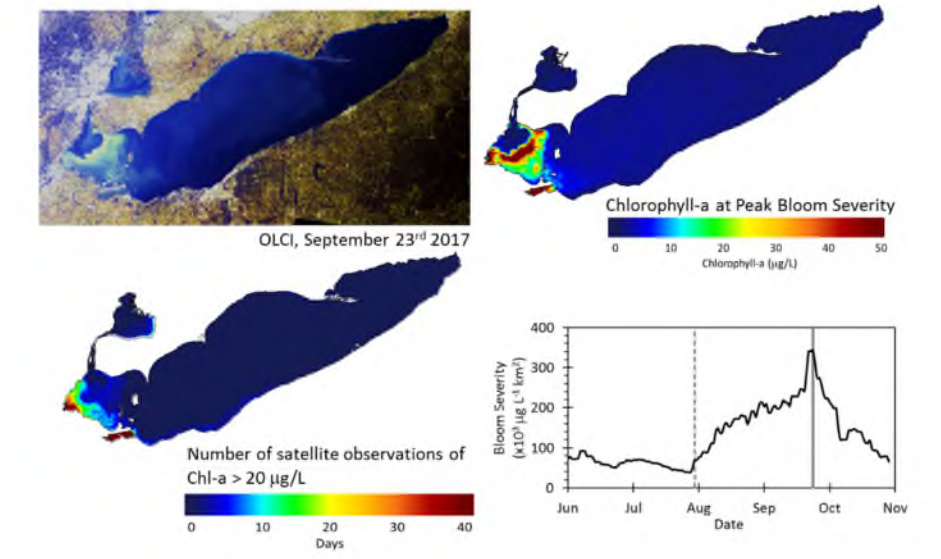


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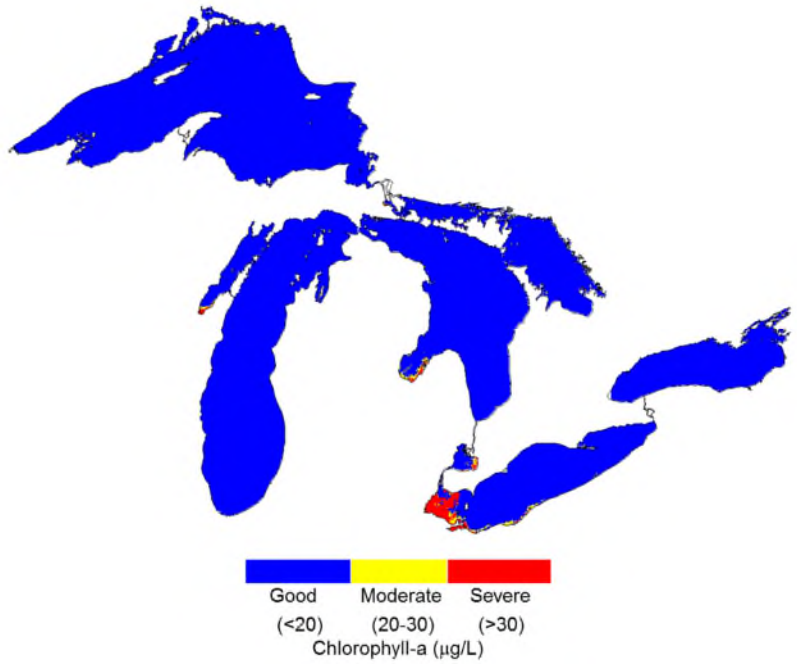


Figure 3. Classification of Great Lakes satellite-derived chlorophyll-a according to metric 1a in Table 1. One or more detected instance of chlorophyll-a 20-30 µg/L (Moderate) and >30 µg/L (Severe) during April-October imagery.

[i] Rating scale for chl-a is based on Carlson 1977 trophic index (oligotrophic, mesotrophic, eutrophic states), and for plankton biomass is based on Downing et al. (2001)

[ii] This metric focuses only on those species with a likely potential to produce toxins. Dominance of the bloom by cyanobacteria can be established by one of several methods including (a) qualitative (or quantitative) assessment from a microscope inspection that indicates a high (>80%) abundance of cyanobacteria, (2) *in-vivo* measurers using suitable probes that can distinguish between cyanobacteria-specific chlorophyll and total chlorophyll, (e.g. YSI®, bbe-Moldaenke® AOA or FluoroProbe or Walz® PhytoPAM), or satellite imagery that can resolve between cyano-bacteria-chlorophyll/biomass and total algal chlorophyll or biomass.

High *potential* to produce microcystins is established by either (1) visual examination of one or more samples and the identification that the bloom contains significant numbers of certain species of *Microcystis* (*M. wesenbergii* is not known to be toxic). Note that most species have both toxic and non-toxic strains which can occur together or separately. *Planktothrix/ Oscillatoria, Aphanizomenon, Limnoraphis* or *Anabaena/Dolichospermum* dominated blooms have a lower likelihood of producing microcystins and toxicity needs to be assessed using direct toxicity measurements or PCR (see below) or Metric 1. Molecular techniques such as the polymerase chain reaction (PCR) can be used to indicate the presence of the biosynthetic operon (*mcy*) and potential for microcystin biosynthesis but the presence of this operon by itself without a significant cyanobacterial biomass or toxin production is not sufficient indication of HAB impairment.

[iii] Total chlorophyll-a concentrations can be measured using any of several different methods including extracted chlorophyll-a EPA method 445.0, in-vivo based measures using a suitably calibrated water quality sonde, or by appropriate satellite imagery. A level of 30 µg/L was chosen based on prior indicators (e.g. Carlson, 1977) and is supported by current research that shows chlorophyll concentrations greater than 30 µg/L have a 90% chance of containing the *mcy* genes (Hotto 2007).

[iv] This is a performance based metric and the method(s) used to measure microcystins are not specified. It may include particulate or dissolved toxins as measured by enzyme-linked immunosorbent assays (ELISA), protein phosphatase inhibition assays (PPIA), liquid chromatography coupled with mass spectroscopy (LCMS or LC-MS/MS) or other suitable technique.

[v] This represents a change from considering a single microcystin congener (microcystin-LR) to measuring multiple microcystin congeners. Several of the currently used methods (ELISA) do not distinguish between congeners. Many of the other microcystin congeners (MC-RR, MC-LA) have similar toxicities to microcystin-LR and can occurred in water samples in the absence of microcystin LR. This change recognizes that these toxins can also pose a health risk. Here we recommend using the combined total value of all toxins measured.

[vi] Propose 1.5 µg/L as the lower (i.e. non-impaired) threshold to be consistent with the US EPA and Health Canada adult drinking water advisory levels. We propose 20 µg/L as the upper threshold for the following reasons:

- Currently the WHO proposes the use of 10 µg/L for low level risk and 20 µg/L for moderate-to-high level of risk.
- In Canada, 20 µg/L is the current recreational guideline set by Health Canada and some of the provinces.
- A guideline value of 4 µg/L proposed by USEPA is based on ingestion of water during swimming in the bloom. This value may be overly protective given recreational uses of Great Lakes waters often include activities such as boating, sailing or fishing which do not involve extensive water contact.

[vii] Propose changing the value here from 1.0 (WHO drinking water advisory level) to 1.5 µg/L (USEPA and ECCC adult drinking water advisory levels) to be in-line with current regulations adopted by the two countries.

[viii] Provisional value, based on unpublished data (may be subject to revision)

[ix] Currently based on bacterial counts; we suggest that it could include measures of both HAB and fecal indicators

[x] e.g. tributary data, Heidelberg University http://www.heidelberg.edu/academiclife/distinctive/ncwqr/data; NOAA-GLERL HABs and Hypoxia data, http://www.glerl.noaa.gov/res/HABs_and_Hypoxia/

[xi] http://www.ontario.ca/data/drinking-water-surveillance-program

Sub-Indicator: *Cladophora*

Overall Assessment
Status: Poor
Trends:
10-Year Trend: Undetermined
Long-term Trend (1995-2017): 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 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.

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 (based on anecdotal information): Unchanging
Rationale: Fouling of shorelines by *Cladophora* has not historically been an issue in Lake Superior, nor have nuisance levels of C*ladophora* been observed at the lakebed, 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 (2006-2018): Undetermined
Rationale: *Cladophora* grows at nuisance levels over large areas of the Lake Michigan nearshore zone. *Cladophora* monitoring has been conducted at two primary sites: offshore of Atwater Beach (north of Milwaukee); and Good Harbor Bay (in the Sleeping Bear Dunes National Lakeshore region). Maximum *Cladophora* biomass at both of these sites varies significantly from year-to-year. While there is some evidence that biomass near Atwater Beach may have declined between 2006 and 2016, biomass in Good Harbor Bay was consistently high from 2010 to 2018. Prior to 2006 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
Status: Fair
10-Year Trend: Undetermined
Long-term Trend (2006 - 2015): Undetermined
Rationale: *Cladophora* biomass approaches nuisance levels in localized areas which coincide with points of nutrient input on the Canadian shores of the main basin. Field-based monitoring has been limited but suggests that *Cladophora* biomass is generally low and below nuisance levels. Nonetheless, periodic fouling of shorelines by mixtures of algae comprised of macroalgae (Charophytes and *Cladophora*) and diatom periphyton is common-place. *Cladophora* is not found at macroscopically visible levels in the nearshore of eastern Georgian Bay nor has it been reported to foul shoreline in Georgian Bay.

Lake Erie
Status: Poor
10-Year Trend: Unchanging

Long-term Trend (1995-2017): Undetermined
Rationale: *Cladophora* remains broadly distributed along much of the north shore of the eastern basin, as well as offshore shoals with sufficient hard substrate for *Cladophora* attachment. Monitoring of 4 – 5 transects between Port Dover, 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 for this high growth. There is no indication of trends getting better or worse from 2007-2017 in Lake Erie as inter-annual 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 2001 as exploratory monitoring and research and in response to regional concerns.

Lake Ontario
Status: Poor
10-Year Trend: Undetermined
Long-term Trend (1998-2017): Undetermined
Rationale: *Cladophora* is widely distributed in Lake Ontario. Biomass routinely exceeds nuisance conditions where hard substrate dominates the nearshore lake bottom. Surveys from recent years indicate nuisance conditions both in the vicinity of point source inputs, and also in regions remote from any known sources. 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: In-lake biomass <50g DW/m^2
Fair: In-lake biomass >50g and < 75g DW/m^2
Poor: In-lake biomass >75g DW/m^2
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

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

Endpoints and/or Targets
In 1982, Canale and Auer defined nuisance conditions of *Cladophora* growth as a maximum standing crop >50g DW/m^2. 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 negative beach fouling 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 negative (beach fouling) effects. The current target has no spatial scale and the area affected may need to be considered when determining if *Cladophora* biomass has reached nuisance levels. Additional work would be 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 spatial and temporal trends in biomass of *Cladophora* in the Great Lakes. Data can be used to infer the availability of *Cladophora* to be transported to the lake shore where it may foul beaches and clog water intakes, as well as its potential contribution to other negative impacts such as avian botulism.

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. 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 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 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 will report the biomass of *Cladophora* as standing crop in grams dry weight (DW)/m^2. In addition to biomass, *Cladophora* tissue phosphorus content is also a useful measure. However, its use as an indicator must take light availability into account, as tissue P content is strongly influenced by both dissolved P concentration and irradiance. With the apparent role of dreissenid mussels as an immediate nutrient source for *Cladophora*, measurements of dreissenid density are also useful. Stations for preliminary sentinel monitoring have been selected in Lakes Erie and Ontario. These were chosen with an aim to be representative of either broader scale or more local environmental conditions; as such, the results will have to be considered in the context of their light availability (i.e. depth and turbidity) and their 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.

Long-term monitoring of *Cladophora* is currently being conducted within the context of academic research programs at two locations in Lake Michigan (near Milwaukee and Sleeping Bear Dunes National Lakeshore). Annual monitoring of *Cladophora* at sites in eastern Lake Erie has been conducted since 2012 and on the north shore of Lake Ontario since 2017 by Environment and Climate Change Canada. The USEPA and USGS initiated ongoing monitoring in the four lower lakes beginning in 2018. Quantitative sampling of *Cladophora*, typically by divers, is logistically challenging and method-sensitive. Hence there may be a need for further standardization of methods among lakes and among institutions.

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 algae 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 pollution, these blooms elicited public outcry and were identified as an emerging issue under the 1978 Great Lakes Water Quality Agreement. Targeted research in the late 1970s generally concluded that phosphorus (P) load reductions being 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 control programs in reducing *Cladophora* growth, but systematic monitoring was not conducted. By the mid-1990s, reports of shore fouling began to appear in Lake Erie (Howell 1998) and by the early 2000s, had extended to Lakes Ontario (DeJong 2000, Malkin et al. 2008) and Michigan (Bootsma et al. 2005). With the recent resurgence of the nearshore algal problem in some areas and with other changes in the ecosystem, 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 pathogens associated with rafts of decaying organic matter along the lake shore (Higgins et al., 2008, Peller et al. 2014, Ishii et al. 2006, Chun et al. 2017). The ecological impacts of excessive *Cladophora* growth and biomass are not as well understood, but may nonetheless be important. *Cladophora* is generally considered to be a poor food resource for grazers (Dodds and Gudder 1992), thus expansive standing crops may represent a substantial (albeit perhaps temporary) nutrient sink over much of the growing season (Higgins et al. 2005). 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) which 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 Michigan, as well as the northern shore of eastern Lake Erie. In Lake Huron, reports of excessive biomass are generally restricted to isolated locations along the south-eastern shore of Lake Huron (Figure 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 one location about 7 km north of Milwaukee Harbor since 2006. These measurements indicate that peak biomass varies from year-to-year ranging from a high of 268 g m^{-2} in 2008 to a low of 38 g m^{-2} in 2014. Highest biomass levels were observed between 2006 and 2011. Since 2012, peak biomass levels have been more moderate, but there continue to be problems with fouling of beaches and water intakes. The 10-year record suggests that there may be a trend toward lower peak summer biomass, but the time series is not long enough to confirm whether this is a real trend or simply inter-annual variation. *Cladophora* biomass and phosphorus content have also been monitored on a shallow reef in Good Harbor Bay, near Sleeping Bear Dunes National Lakeshore, since 2010. Between 2010 and 2017, biomass has peaked at or above 200 g DW/m^2 in most years, and phosphorus content has frequently been above 0.6 ug/mg, which is considered the minimum threshold for *Cladophora* growth (Fig. 2). 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.

Lake Huron

Cladophora biomass can reach nuisance conditions in the vicinity of local nutrient inputs in isolated regions along the south-eastern shore. 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 WWTP discharge) and Kincardine ON (affected by a small inflowing agricultural drain). Biomass at Goderich was 46 g m^{-2} and 49 g m^{-2} respectively, while at Kincardine biomass 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). The spatial extent of growth at these locations was limited. Over broader stretches of the eastern shoreline, *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 (Figure 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 linked to episodic fouling of beaches with decaying organic matter.

Lake Erie

In Lake Erie, *Cladophora* has reached 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. 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 Environment and Climate Change Canada (Figure 3). Inter-annual variation in peak biomass is high across these sites and displays no temporal trend (Figure 3). *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 can experience high algal biomass. Valipour et al (2016) modeled that upwelling of offshore waters can result in significant Soluble Reactive Phosphorus (SRP) supplies to the nearshore in some years, potentially sustaining regional *Cladophora* growth independent of local sources.

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. General features of *Cladophora*, confirmed in more recent surveys by the Ontario Ministry of the Environment and Climate Change and Environment and Climate Change Canada in 2012-2013, 2015, and 2017-2018, include high surface coverage to the point of blanketing hard substrate, strong attenuation of biomass with depth but persisting cover to depths > 10-20 m, typically co-occurring with high dreissenid mussel cover, and frequently 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. High short-term and spatial variability in biomass levels make inferences about spatial and temporal patterns of variation 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 P 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 soluble reactive phosphorus (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, due both to their ability to clear the water column (and hence increase *in situ* irradiance) by removing particulate material, and their recycling of 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 2015 *Cladophora* biomass in Good Harbor Bay (near Sleeping Bear Dunes) in Lake Michigan, where there are no major tributary sources of nutrients, peaked at 186 g DW m^{-2}, while peak biomass several kilometers north of Milwaukee Harbor, which is a major nutrient source, was 38 g DW m^{-2}. However, in other regions, (i.e. Lake Ontario) there is evidence that local nutrient inputs do indeed have a local influence on *Cladophora* biomass (Higgins et al. 2012).

Biomass and Phosphorus status as indicators

Monitoring of in-lake biomass has been and remains a favoured 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 term periods (hours to days) in response to environmental conditions (i.e. wind and wave action, turbidity, nutrient supply, thermal regime). This generally leads to significant spatial and temporal variability in attached biomass at a given point in time (e.g., Figure 3). Comparisons of point-in-time measurements of biomass across spatial and temporal gradients may be misleading without appropriate consideration of environmental conditions and context.

Approaches for monitoring *Cladophora* were reviewed in a 2017 *Cladophora* report. These 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 they 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 P content (or P status) of algal filaments has long been considered a useful metric for assessing the status of *Cladophora* and the potential for P 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 P that a) removes uncertainty in P 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 P 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 P saturated, values between 1.6 and 0.6 mg g^{-1} (0.16 – 0.06 %) are P limited, and values below 0.06 % are critically limiting and insufficient to sustain net positive growth rates. However, P limitation cannot be inferred from Q_P alone for the simple reason that other factors can affect internal P levels. Q_P may be an inexact indicator of P availability, as P 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 P availability 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 be unable to dilute internal P 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 P 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. In the same survey, *Cladophora* P content was found to vary more than 3 fold and *Cladophora* biomass was negatively correlated with Q_P, suggesting that Q_P alone is not a good indicator of P availability and growth potential. Similar observations were documented in eastern Lake Erie from 2012-2017, 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 (Figure 4). Water quality monitoring conducted in 2013 at the mouth of the Welland Canal in Lake Ontario demonstrate 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.

There is uncertainty about the degree to which high levels of biomass on the lakebed manifest as shore fouling. In a broad sense, shore fouling concerns map to 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 limitations, the presence of excessive biomass at a given location is likely to indicate the potential for shore fouling and other negative impacts.

Monitoring

The lack of a 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 was struck in support of the GLWQA Annex 4 (nutrients) and this workgroup developed a research plan for Great Lakes *Cladophora* modeling, research and monitoring. The Parties are currently implementing monitoring plans. Since 2012 sentinel site monitoring has been conducted at priority nearshore sites in Lake Erie by ECCC and collaboratively with the Ontario Ministry of Environment, Conservation and Parks since 2017 in Lake Ontario. Starting in 2018, the USGS is implementing monitoring in the lower four Great Lakes.

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 submerged aquatic vegetation (SAV) were conducted in 2010 and 2012 via remotely sensed images. 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 AUV and ROV-mounted optical sensors for determining *Cladophora* biomass and distribution. 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
- 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, 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.

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. This effort has resulted in a data set that is more amenable for calibrating and validating a *Cladophora* growth model. 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.

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 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 are required in order to permit reporting of regional or lakewide 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, it was relatively widespread, but in moderately eutrophic 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). The application of models that simulate *Cladophora* growth in response to light, temperature and dissolved phosphorus concentration (Tomlinson et al. 2010), 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 nutrient excretion by mussels also playing a role (Auer et al. 2010; Bootsma and Liao 2014). 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 dreissenids. 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 decreases. It is anticipated that biomass of *Cladophora* will decline as nutrient concentrations decrease and algae become stressed. Phosphorus control strategies have worked in the past at 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. In some cases, nuisance benthic algae can also consist of other taxa, including the green alga *Chara* and *Spirogyra*. 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 refers 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. For the Eastern Basin, a target has not been recommended to address nuisance algae (*Cladophora*) at this time. 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 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 this might suggest that the frequency and magnitude of accumulation on the shoreline is a more useful measure, this 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 P 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 these factors be increasingly understood.

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List of Figures

Figure 1. Locations within the Great Lakes where *Cladophora* has been reported since the year 2000. Empty circles indicate biomass below nuisance threshold of 50 g m^{-2} DW while filled circles indicate biomass above the nuisance threshold. Inset panel denotes higher resolution regions of eastern Georgian Bay where Ontario Ministry of Environment and Climate Change monitoring in 2014 took place.
Data sources: Lake Ontario – Malkin et al. 2008, S. Malkin unpubl.data, Higgins et al. 2012, D. Depew – unpubl. data, Lake Huron – Barton et al. 2013, d.Depew unpubl.data, T. Howell, unpubl. data, Lake Erie – Environment and Climate Change Canada unpubl. data, Higgins et al,. 2005, Lake Michigan – Garrison et al.2008, Tomlinson et al. 2010, H. Bootsma unpubl. data, Dayton et al. 2014.
Figure 2. Time series of *Cladophora* biomass and tissue phosphorus content at a depth of 10 m on the Good Harbor Bay reef, Sleeping Bear Dunes National Lakeshore, Lake Michigan.
Source: H. Bootsma, unpubl. data.
Figure 3. Mean and standard error of *Cladophora* biomass at Grand River sites ranging from 3 metre to 18 metre in depth. Data are presented for July from 2012-2017.

Source: Environment and Climate Change Canada, unpubl.data.
Figure 4. Plot of attached biomass and Q_P for sites in eastern Lake Erie for the same stations in Figure 3.
Source: Environment and Climate Change Canada, unpubl. data.

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State of the Great Lakes 2019 Technical Report

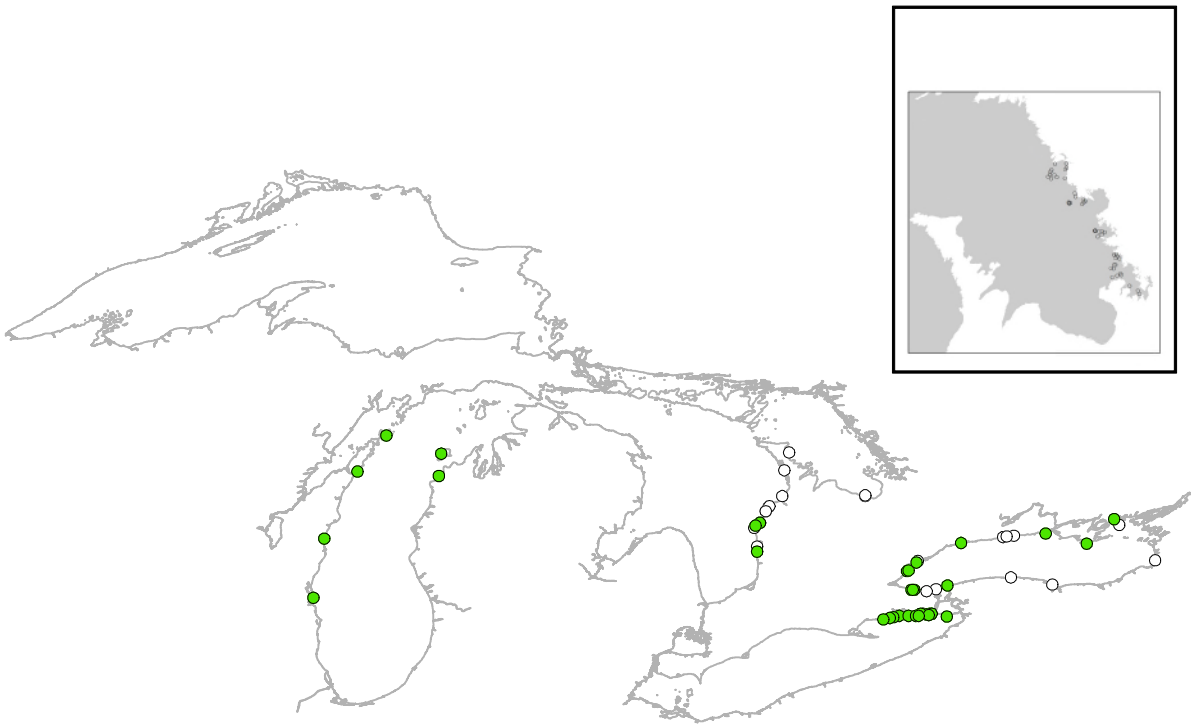


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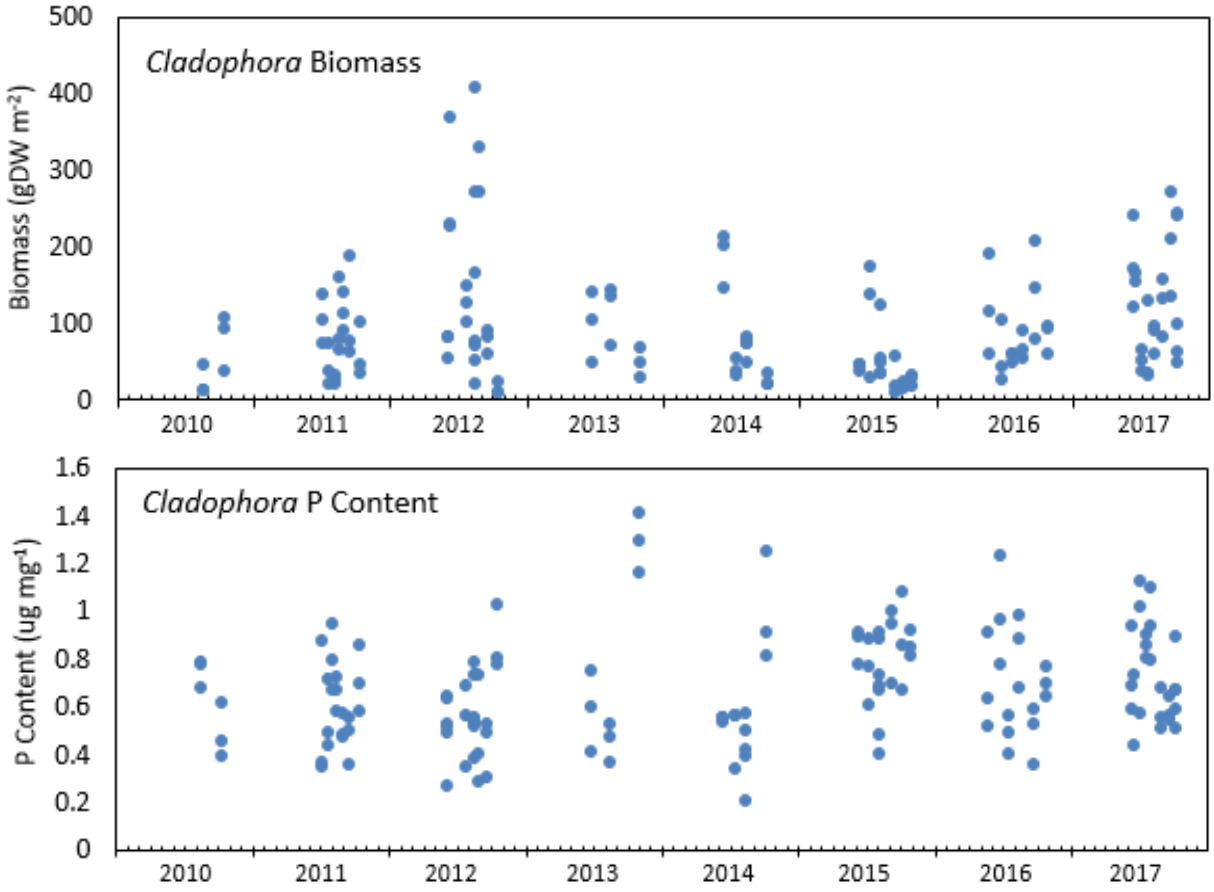


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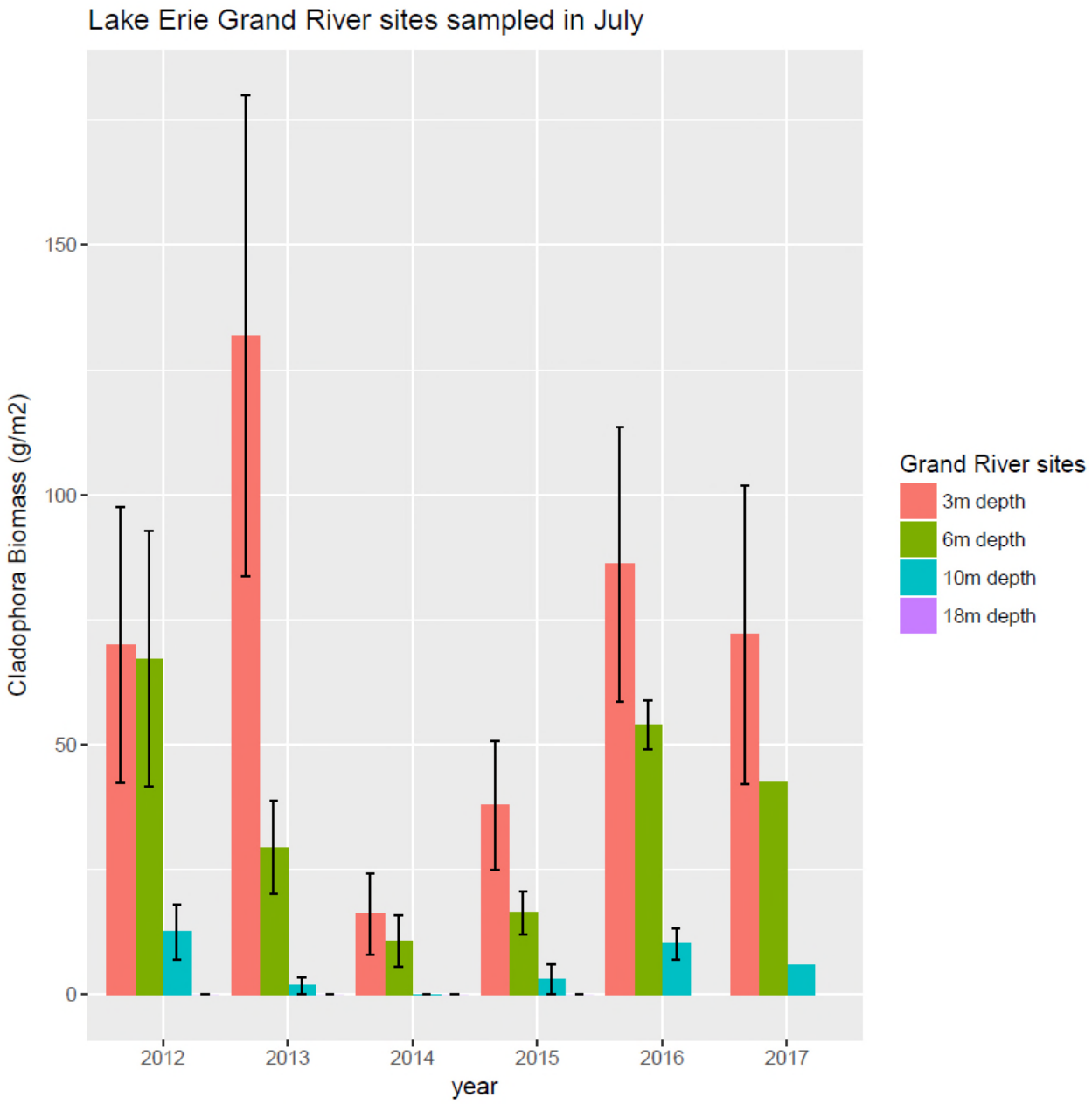


Figure 3. Mean and standard error of *Cladophora* biomass in the eastern basin of Lake Erie, near the mouth of the Grand River. Data are presented for July from 2012-2017.
Source: Environment and Climate Change Canada, unpubl.data.

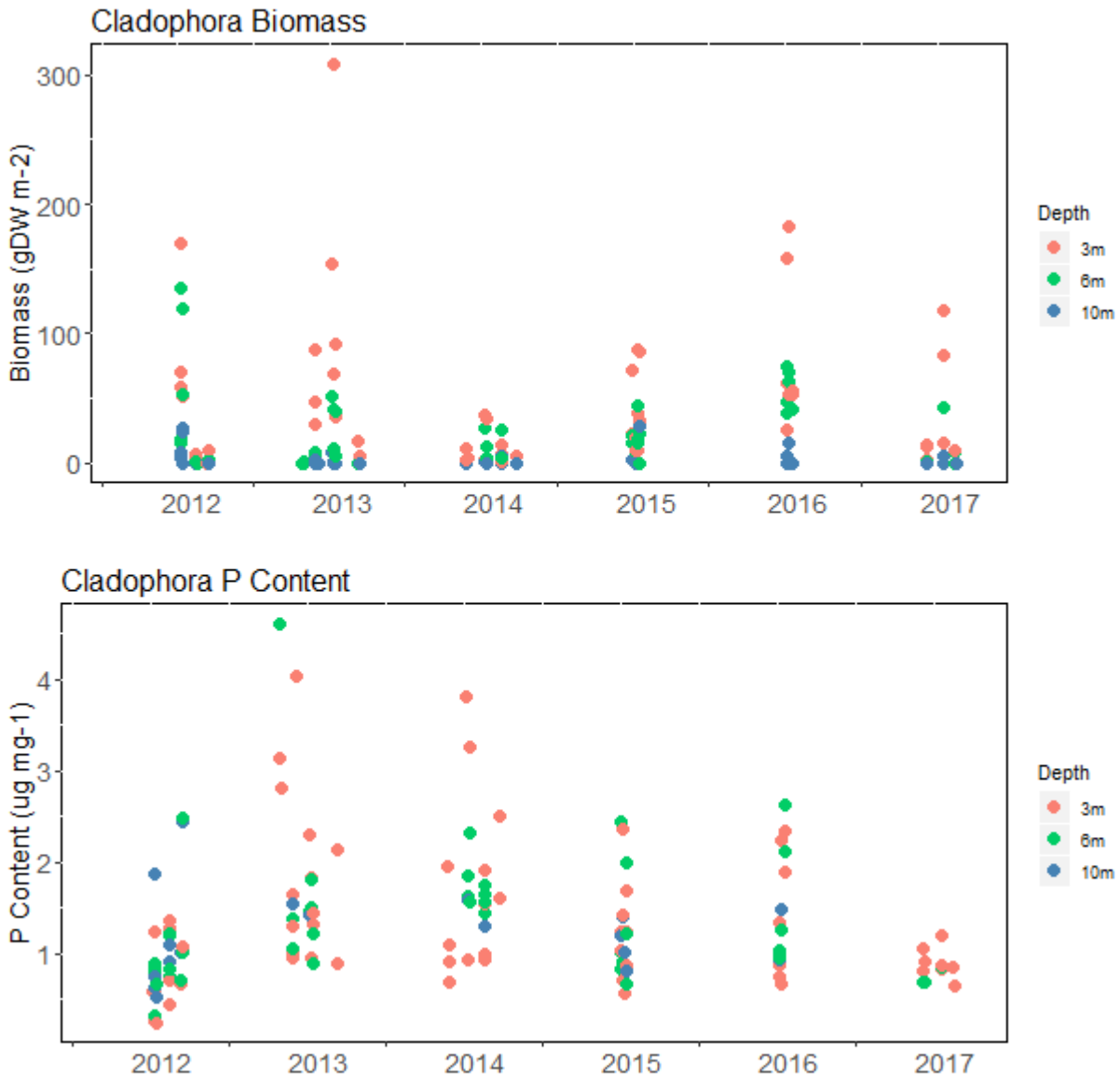


Figure 4. Time series of *Cladophora* biomass and tissue phosphorus content in eastern Lake Erie near the mouth of the Grand River (same sites as Figure 3).
Source: Environment and Climate Change Canada, unpubl. data.

Sub-Indicator: Water Quality in Tributaries

Overall Assessment – Canadian Only
Status: Fair
Trends:
10-Year Trend: Unchanging
Long-term Trend: Undetermined
Rationale: The WQI was computed for 73 Canadian tributaries to the Great Lakes. 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 73 tributaries (WQIavg=65, WQIrange=10-100). Broken down by category, 25% of the tributaries were categorized as having Good water quality, 55% as Fair, and 20% as Poor.

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

Lake-by-Lake Assessment
Lake Superior
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: The average WQI score for 4 tributaries (down from 9 in the previous State of the Great Lakes (SOGL) report) was 77/100. WQI scores ranged from 72 to 85 (Fair to Good). There were only a few sites monitored and all four sites were clustered near Thunder Bay, therefore it is difficult to assign an overall lake assessment value based on so few tributaries. In 2017, the average WQI value was 78/100 based on 9 sites and WQI scores ranged from 65 to 100.

Lake Michigan
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: No tributaries to Lake Michigan are monitored by the Ontario Provincial Water Quality Monitoring Network (PWQMN).

Lake Huron
Status: Fair
10-Year Trend: Unchanging
Long-term Trend: Undetermined
Rationale: The average WQI score for 20 tributaries was 79/100. WQI scores ranged from 50 to 100 (Fair to Good). In 2017, the average WQI value for 28 tributaries was 81/100.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Unchanging
Long-term Trend: Undetermined
Rationale: The average WQI score for 15 tributaries was 44/100. WQI scores ranged from 10 to 76 (Poor to Fair). In 2017, the average WQI value for 18 tributaries was 43/100.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend: Undetermined
Rationale: The average WQI score for 34 tributaries was 65/100. WQI scores ranged from 25 to 100 (Poor to Good). In 2017, the average WQI value for 31 tributaries in the Lake Ontario basin was 65/100 and for the St. Lawrence River was 73/100 for 6 tributaries and assessed as Fair.

Status Assessment Definitions
The calculated values fit into five categories that describe water quality conditions as used by the 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 and 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" and 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
WQI calculations for Ontario tributaries to the Great Lakes were computed using monitoring data from the PWQMN (https://www.ontario.ca/data/provincial-stream-water-quality-monitoring-network). The WQI may be calculated independently for U.S. tributaries if data are available.

The Water Quality Index (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. Only data for the period 2013-2016 (OMOE, 2013) were used to determine the status in this report. While the previous SOGL report included 92 tributaries (including stations where older data were used), this report only includes those tributaries with data for the most recent period 2013-2016 (n=73).

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. Subsequent SOGL reports will report trends in water quality using more robust methods.

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://www.ontario.ca/data/provincial-stream-water-quality-monitoring-network).

Status of Water Quality in the Great Lakes Tributaries

The WQI was computed for 73 Canadian tributaries to the Great Lakes. The overall water quality status of tributaries to the Great Lakes can be described as Fair (WQIavg=65, WQIrange=10-100). Broken down by category: 25% of the tributaries were categorized as having Good water quality, 55% as Fair, and 20% were Poor (Figures 1 and 2).

Good water quality was found in certain tributaries to Lakes Huron and Ontario, and the St. Lawrence River. Poor water quality was found in certain tributaries of Lakes Erie and Ontario. The WQI scores ranged from 10 (Sturgeon River, Lake Erie) to 100 (Sauble River, Lake Huron and 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}=79, WQI_{range}=50-100, n=20) and Ontario and St. Lawrence River (WQI_{avg}=65, WQI_{range}=25-100, n=34) can be described as having Fair water quality. Tributaries to Lake Erie (WQI_{avg}=44, WQI_{range}= 10-76, n=15) were categorized as having Poor water quality. The description 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. While Lake Huron has changed from "Good" to "Fair", the WQI value itself has only decreased by two which is why the trend was still reported as Unchanging since the last report, irrespective of the change in status.

Linkages

The WQI values for the 73 tributaries show a statistically significant negative correlation (Spearman's Rank Correlation; $p<0.001$) with percent of the watershed occupied by human land uses (Figure 3). 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 the 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.

However, 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.

This sub-indicator also links directly to the other sub-indicators in the Nutrients and Algae indicator

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 Limitations

The Index 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 Index 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 73 in this 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.

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.

For this Water Quality in Tributaries report, the WQI has been computed only for Canadian tributaries. The application of the WQI to assess water quality in U.S. tributaries to the Great Lakes depends on the availability of monitoring data. An anticipated challenge is that WQI results are not directly comparable between jurisdictions where different water quality parameters and criteria are used.

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.

This current Water Quality in Tributaries report is a status update from 2017 (ECCC and USEPA, 2017). 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 73 tributaries whereas scores were computed for 92 tributaries used previously (ECCC and USEPA, 2017). In the Lake Ontario basin, the furthest site downstream on the Napanee River (used in previous SOGL reports) was not actively sampled for this time period, so a station further upstream was used to calculate the WQI.

Acknowledgments
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Contributors: Georgina Kaltenecker, Kimberly Summers, and Aaron Todd, Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment, Conservation and Parks, Toronto, ON, and Ontario's Conservation Authorities

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List of Tables

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}-$CaCO_3$ 3 µg L^{-1} at water hardness of 120-180 mg L^{-1}-$CaCO_3$ 4 µg L^{-1} at water hardness of >180 mg L^{-1}-$CaCO_3$	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

Sources: CCME = Water quality guidelines for the protection of aquatic life (CCME 2011*a*); OMOE = Interim provincial water quality objective (OMOE 1994).

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List of Figures

Figure 1. CCME Water Quality Index (WQI) values for 73 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. 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

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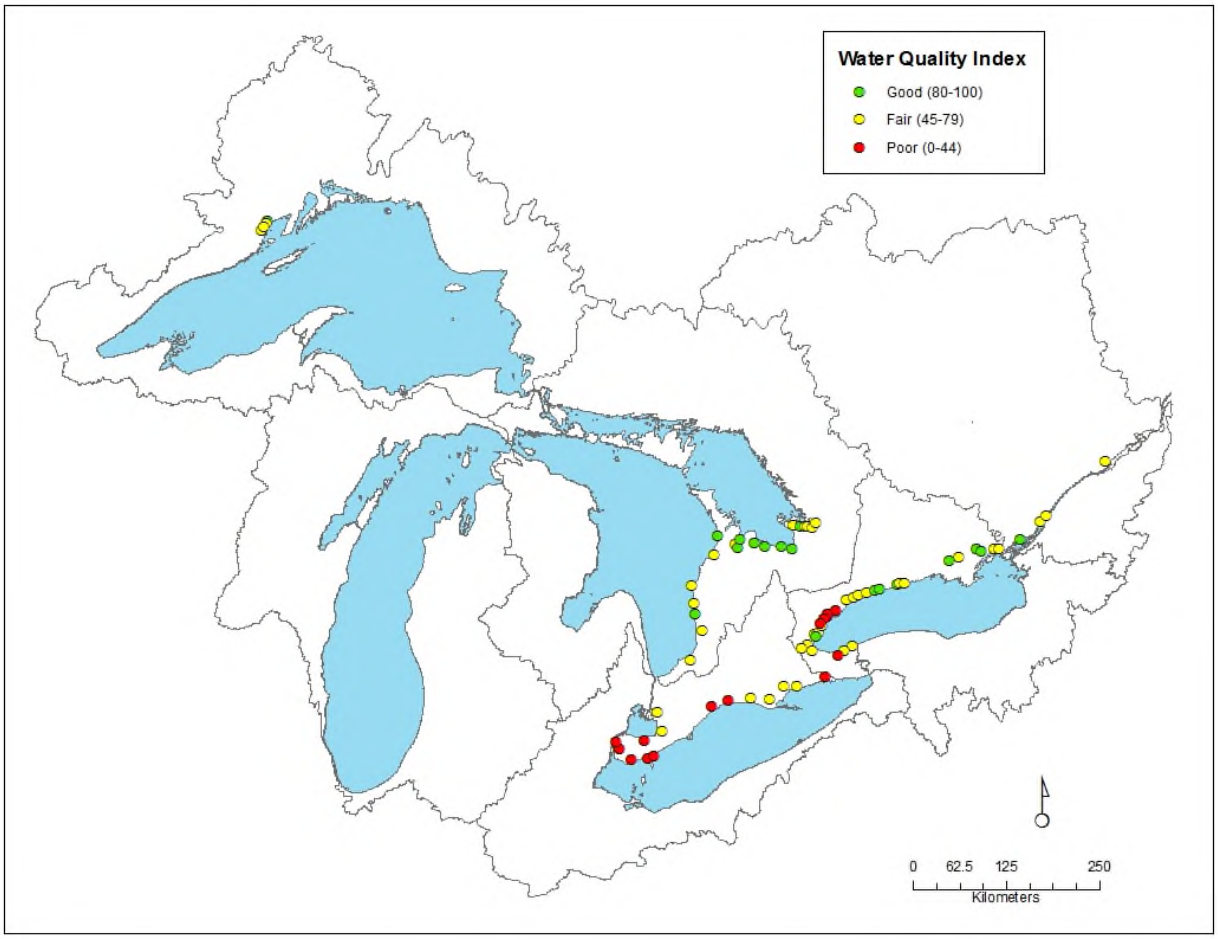


Figure 1. CCME Water Quality Index (WQI) values for 73 Canadian tributaries to the Great Lakes.
Source: Ontario Ministry of the Environment, Conservation and Parks

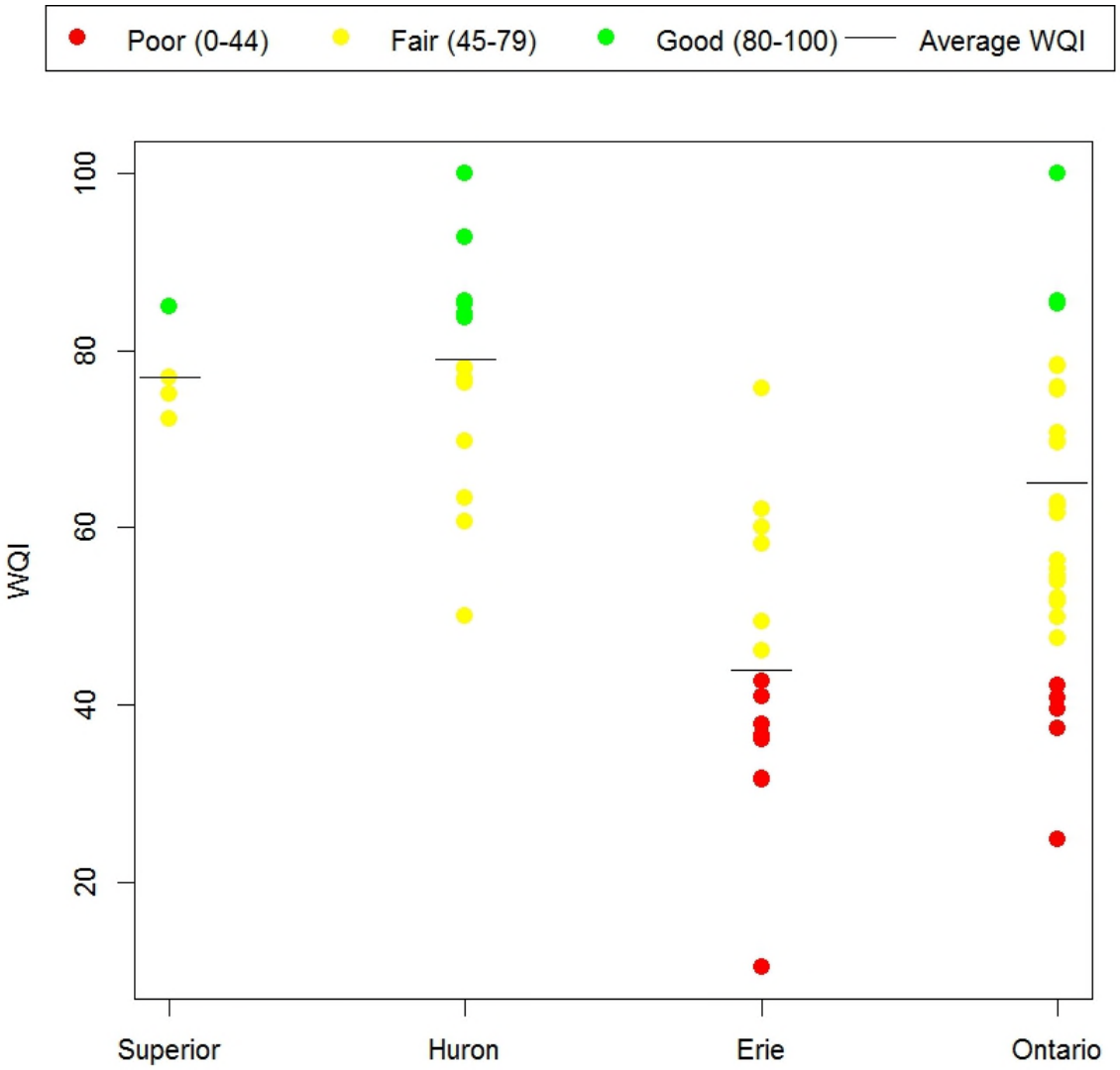


Figure 2. CCME Water Quality Index (WQI) values for Canadian Great Lakes tributaries by lake basin. Source: Ontario Ministry of the Environment, Conservation and Parks

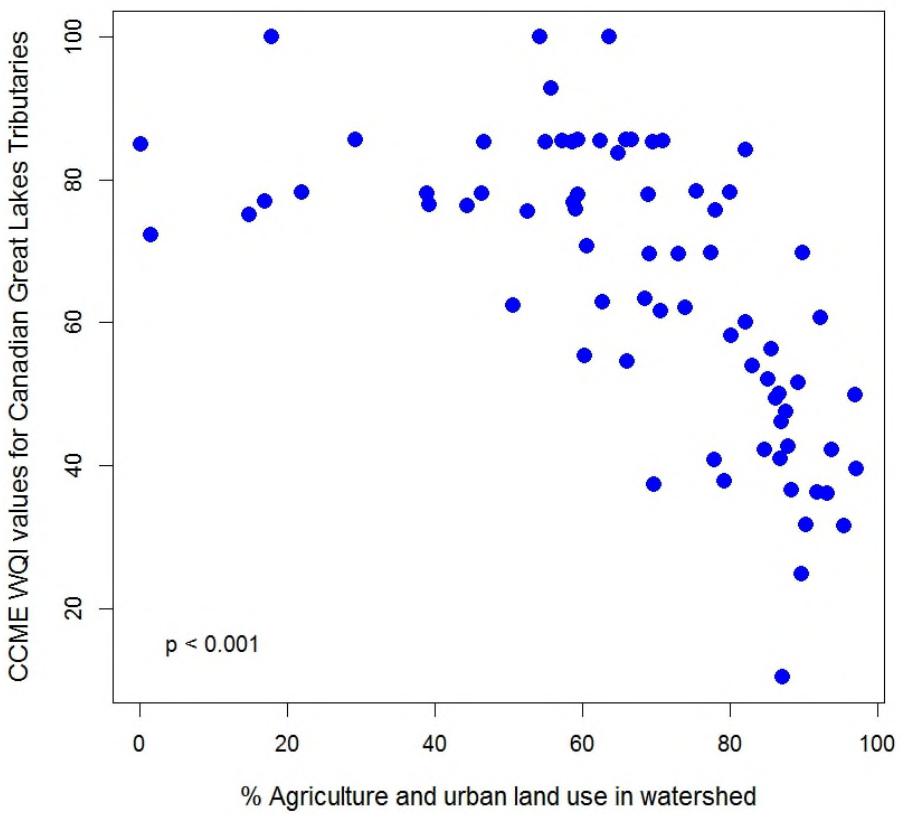


Figure 3. CCME Water Quality Index (WQI) values for Canadian Great Lakes tributaries (n=73) versus percent agriculture and urban land uses
Source: Ontario Ministry of the Environment, Conservation and Parks

Sub-Indicator: Rate of Invasion of Aquatic Non-Indigenous Species

Overall Assessment
Status: Good
Trends:
10-Year Trend: Undetermined
Long-term Trend (1989-2018): Improving
Rationale: In the past decade (2009-2018), three new aquatic non-indigenous (ANS) specie have been established in the Great Lakes basin (*Thermocyclops crassus* in 2014, *Diaphanosoma* 2015, and *Mesocyclops pehpeiensis* in 2016) (Figure 1). *Thermocyclops crassus* and *Diaphanosoma fluviatile* are currently assigned as an unknown vector while *Mesocyclops pehpeiensis* is assigned as a 'hitchhiker with organisms in trade because it has previously been found in North America associated with aquarium and water garden plants and in greenhouse ponds. For the current decade, this introduction results in a rate of 0.3 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), indicating that the long-term trend improved dramatically with the implementation of the regulations for NOBOB (No Ballast on Board) vessels in 2006. Within the last decade (Figure 3) the invasion rate has not changed significantly.

Lake-by-Lake Assessment
Individual lake basin assessments were not prepared for this report.

Status Assessment Definitions
Good: Current decadal invasion rate is significantly lower than the pre-1950 average invasion rate.
Fair: Current decadal invasion rate is not significantly different from the pre-1950 baseline invasion rate.
Poor: Current decadal invasion rate is significantly higher than the pre-1950 average invasion rate.
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions
Improving: Decreasing rate of introduction
Unchanging: No change in rate of introduction
Deteriorating: Increasing rate of introduction
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 the number of new introductions of 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 invasion 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 invasion as a measure of progress towards the prevention of establishment of new non-indigenous species and control of inter-basin spread.

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

The previous report used cumulative number of established non-indigenous species as a single sub-indicator for invasive species (Figure 1). The original report has been separated into RATE (measuring progress toward prevention goals) and IMPACT (measuring progress towards control/eradication goals). Impact is assessed in a separate sub-indicator in the State of the Great Lakes suite. In previous reports, data was presented in this format seen in Figure 1. However, the SLOPE of this graph (invasion rate per year) is the more relevant metric now that rate and impact are separated. This figure is included because it (a) allows direct comparison to previous 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 (e.g., tropical and marine species such as flounder, alligator, and piranha) are introduced species, but not established. These introduced species are excluded from this analysis because reports are not believed to be comprehensive.

Decadal rates of invasion are calculated and used as a direct metric. Both lake-by-lake and basin-wide spatial scales are implemented for these measurements. Partitioning by vector (e.g. ballast water, deliberate release, canal, etc.) can only be done for the invasions coming from outside the Great Lakes – there is not consistent data attributing species movements between lakes to particular vectors, thus all introductions from other points within the Great Lakes basin is included in the 'dispersed' vector (even though secondary human-mediated vectors such as inter-lake shipping, recreational boating or trade may be facilitating this 'dispersal'). Inter-basin spread as a vector for reaching the individual lake basins is called out separately from the primary vectors (i.e., when the introduction is from outside the Great Lakes). For example, a new introduction to Lake Superior includes species coming from the lower lakes. The individual lake metrics thus also encompass spread between lakes. This is the more meaningful parameter from the perspective of the 'health' of the particular Lake.

The proposed new measure for RATE of Introduction is the rate (species per year) of new aquatic non-indigenous species found in the Great Lakes. Currently, no species have been successfully eradicated from an entire basin, but eradication rate should be considered as an accompanying measure when relevant. This report uses the binational standardized data from the Great Lakes Aquatic Non-indigenous Species Information System (GLANSIS) and other standardized data if and when appropriate.

Ecological Condition

Background

The National Oceanic and Atmospheric Administration (NOAA) currently reports a total of 187 established Great Lakes ANS (plus at least 19 species native to some part of the basin which have expanded their ranges to other parts). 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 overall assessment (e.g., Rusty crayfish is indigenous to the Great Lakes) but ARE included in the individual lake assessments where relevant (e.g., Rusty crayfish is non-indigenous to Lake Superior).

In the Great Lakes, transoceanic ships (including solid ballast, packing materials, ballast water and ballast residuals) have been the primary invasion vector responsible for 44% of the total established ANS. Historically, deliberate introductions (stocking fish and agricultural/horticultural plants) have also been a significant vector (23%). Other notable vectors include hitchhikers with organisms in trade (9%), aquarium releases (4%), canals (4%), escapes from culture (4%), bait (1.6%) – despite best efforts to attribute invaders 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. *Thermocyclops crassus (*introduced in 2014) and *Diaphanosoma fluviatile* (introduced 2015) are currently assigned as an unknown vector, but considered unlikely to have invaded via transoceanic ballast while *Mesocyclops pehpeiensis* (introduced 2016) is assigned as a hitchhiker with aquatic plants. Ballast water regulation appears to have been largely successful in preventing new introductions from this vector – there have been no new introductions attributed to this vector in the last decade (2009-2018); in comparison there were 6 introductions attributed to this vector in the previous decade (1999-2008) and 14 in the decade prior to that (1989-1998).

Even with the successes of ballast management regulations in preventing new invasions 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 only 37 of the 187 non-indigenous species established in the Great Lakes (20%) have spread to all 5 lakes – 80% of the ANS in the Great Lakes thus still pose a significant risk for additional spread. Over the historic record, Lake Superior has received 92% of its new invaders from the other 4 lakes, Lake Huron 91%, Lake Michigan 72%, Lake Erie 62%, and Lake Ontario 58%. The significance of inter-basin transport to Lake Superior and Lake Huron non-indigenous species numbers suggests that continued inter-basin transport is responsible for the slower response of these systems (status of Lakes Superior and Huron remains Poor with regard to this indicator) to the successes of ballast management programs in preventing new introductions from foreign sources. Ballast water movement between lakes within the basin, which is not currently regulated, may pose a relatively high risk of spreading ANS (Casas-Monroy et al. 2014). Movement of non-indigenous species into Lakes Ontario and Erie also appears to be slowing, which may reflect success of control efforts such as angler education and boat wash programs.

Second to shipping, deliberate release, transfer, and escape have introduced ANS into the Great Lakes. Of particular concern are private sector activities related to aquaria, garden ponds, baitfish, and live food fish markets.

Evidence indicates that newly invading species may benefit from the presence of previously established invaders. For example, *Echinogammarus ischnus* (amphipod) have thrived in the presence of previously established zebra (*Dreissena polymorpha*) and quagga mussels (*Dreissena bugensis*). 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.

Nearly a million Asian carp, including bighead and black carp, are sold annually at fish markets within the Great Lakes basin. Until recently, most of these fish were sold live. All eight Great Lakes states and the province of Ontario now have some restriction on the sale of live Asian 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, Quebec, and Ontario, as identified by Alexander (2003). 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., Asian 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). As of June 2018, 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 the mid-1990s. Some evidence indicates that grass carp may be reproducing in the Sandusky and Maumee Rivers (tributaries to western Lake Erie), but as of December 2018 the evidence is inconclusive as to whether this species is established. Grass carp, bighead carp, silver carp, and black carp are 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 three new species have been discovered since 2006 However, species introduced in the previous decades continue to spread – each of the Great Lakes has seen new species become established in its waters in the last decade (ranging from 3 new species for Ontario to 11 for Lake Superior). Lakes Huron and Superior remain particularly susceptible to species spreading from initial populations established in the lower Great Lakes which is causing their status to continue as 'poor'.

In Lake Superior, eleven (11) new non-indigenous species have been found in Lake Superior in the last decade (2009-2018) (Figure 4). These species are all secondary invasions from populations established first in the lower lakes that have spread to Lake Superior. This gives a rate of 1.1 aquatic non-native species per year that are newly established in Lake Superior (Figure 5), which remains significantly higher than the pre-1950 (1873-1949) average rate (0.3 species per year) and not significantly lower than the previous two decades. Within the last decade (Figure 6) invasion rate has not changed significantly.

In Lake Michigan, five (5) new non-indigenous species have been found in Lake Michigan in the last decade (2009-2018) (Figure 7). These species are secondary invasions from populations established first in the other lakes that have spread to Lake Michigan. This is a rate of 0.5 new species per year for the last decade (Figure 8), which is significantly lower than the previous 2 decades, but not significantly higher than the pre-1950 (1838-1949) historic rate (0.3 species per year). Within the last decade, (Figure 9) invasion rate has declined significantly.

In Lake Huron, seven (7) new non-indigenous species have been found in Lake Huron in the last decade (2009-2018). These species are secondary invasions from populations established first in the other lakes and that have spread to Lake Huron (Figure 10). This gives a rate for the current decade of 0.7 new species per year (Figure 11); which is significantly lower than the previous 2 decades but remains significantly higher than the pre-1950 (1850-1949) historic record (0.2 species per year). Within the last decade (Figure 12) invasion rate has not changed significantly.

In Lake Erie, four (4) new non-indigenous species have been found in Lake Erie and the St. Clair-Detroit River Ecosystem (SCDRE) in the last decade (2009-2018). *Thermocyclops crassus (2014), Diaphanosoma fluviatilis (2015), and Mesocyclops pehpeiensis (2016)* all represent first reports for the Great Lakes basin. *Mesocyclops pehpeiensis* is presumed to have been introduced as a hitchhiker on aquatic plants, the vector of introduction for the other two species is unknown (Figure 13). Thus the current decade has an average rate of 0.4 new species per year (Figure 14) which is significantly lower than the previous 2 decades, but not significantly lower than the pre-1950 (1839-1949) average (0.5 species per year). Within the last decade, (Figure 15) invasion rate has declined significantly.

In Lake Ontario, three (3) new non-indigenous species have been found in Lake Ontario in the last decade (2009-

2018), all represent secondary invasions of species established first in other parts of the basin (Figure 16). The current invasion rate (0.3 new species per year (Figure 17) is significantly lower than the previous 2 decades but not significantly lower than the pre-1950 (1843-1949) average. Within the last decade (Figure 18) invasion rate has not changed significantly. No data were included for the St. Lawrence River in this assessment. Within the last decade, (Figure 12) invasion rate has not changed significantly.

Linkages

- This sub-indicator also links directly to the other sub-indicators in the Invasive Species category, particularly Impacts of AIS Introductions, Sea Lamprey and Dreissenid Mussels. Note that some emergent wetland plants are included both in this report as aquatic species as well being included in the terrestrial invasive species sub-indicator.
- 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 ANS and 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 arrival of new invaders. 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 invasion may increase if positive interactions involving established ANS or native species facilitate the establishment of new ANS.

Citizen Science and 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 and other Citizen Sciences 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 Limitations

There is often a delay between the time of introduction for a new aquatic invasive species and the time in which the new ANS is discovered. This temporal consequence has implications for the rate of new introductions of ANS to the Great Lakes. Monitoring effort also influences the temporal lag.

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 'earliest dates' used for the rate calculations at the Great Lakes basin scale nor at individual lake basin scales. Lack of equivalent historic Canadian data may contribute to time lags.

Additional Information

Aquatic Non-native 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 invasion. Healthy, intact ecosystems tend to be less vulnerable to invasion while degraded ecosystems tend to be more readily invaded. Previous introductions of non-indigenous species may facilitate the establishment of new ones.

This report draws attention to the various vectors of invasion 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 invasions.

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 19 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.

Acknowledgements

Rochelle Sturtevant, GLANSIS Program Manager, Michigan Sea Grant, NOAA GLERL, 4840 South State Road, Ann Arbor, MI, 48108. 734-741-2287
Contributors: Kristen Holeck
El Lower, CIGLR/GLANSIS assisted with data prep.

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List of Figure

Figure 1. Cumulative establishment of aquatic non-indigenous species by vector in the Great Lakes Basin. 187 total species as of 2018.
Source: GLANSIS

Figure 2. Decadal invasion rate. Solid line indicates the pre-1950 average.
Source: GLANSIS

Figure 3. Running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

Figure 4. Cumulative establishment in Lake Superior by vector. This includes species non-native to Lake Superior that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Superior, regardless of the intra-basin vector. 97 species have become established in Lake Superior as of 2018. OIT = Organisms in trade.
Source: GLANSIS

Figure 5. Decadal rate of new non-indigenous species establishing in Lake Superior (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

Figure 6. Lake Superior running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

Figure 7. Cumulative establishment in Lake Michigan by vector. This includes species non-native to Lake Michigan that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Michigan, regardless of the intra-basin vector. 127 species have become established in Lake Michigan as of 2018.
Source: GLANSIS

Figure 8. Decadal rate of new non-indigenous species establishing in Lake Michigan (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

Figure 9. Lake Michigan running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

Figure 10. Cumulative establishment in Lake Huron by vector. This includes species non-native to Lake Huron (including the St. Marys River) that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Huron, regardless of the intra-basin vector. 110 species have become established in Lake Huron as of 2018.
Source: GLANSIS

Figure 11. Decadal rate of new non-indigenous species establishing in Lake Huron (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS
Figure 12. Lake Huron running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS
Figure 13. Cumulative establishment in Lake Erie by vector. This includes species non-native to Lake Erie (including the Huron-Erie Corridor) that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Erie, regardless of the intra-basin vector. 151 species have become established in Lake Erie as of 2018.
Source: GLANSIS
Figure 14. Decadal rate of new non-indigenous species establishing in Lake Erie (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS
Figure 15. Lake Erie running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS
Figure 16. Cumulative establishment in Lake Ontario by vector. This includes species non-native to Lake Ontario (including the Niagara River) that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Ontario, regardless of the intra-basin vector. 111 species have become established in Lake Ontario as of 2018.
Source: GLANSIS
Figure 17. Decadal rate of new non-indigenous species establishing in Lake Ontario (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS
Figure 18. Lake Ontario running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

Last Updated
State of the Great Lakes 2019 Technical Report

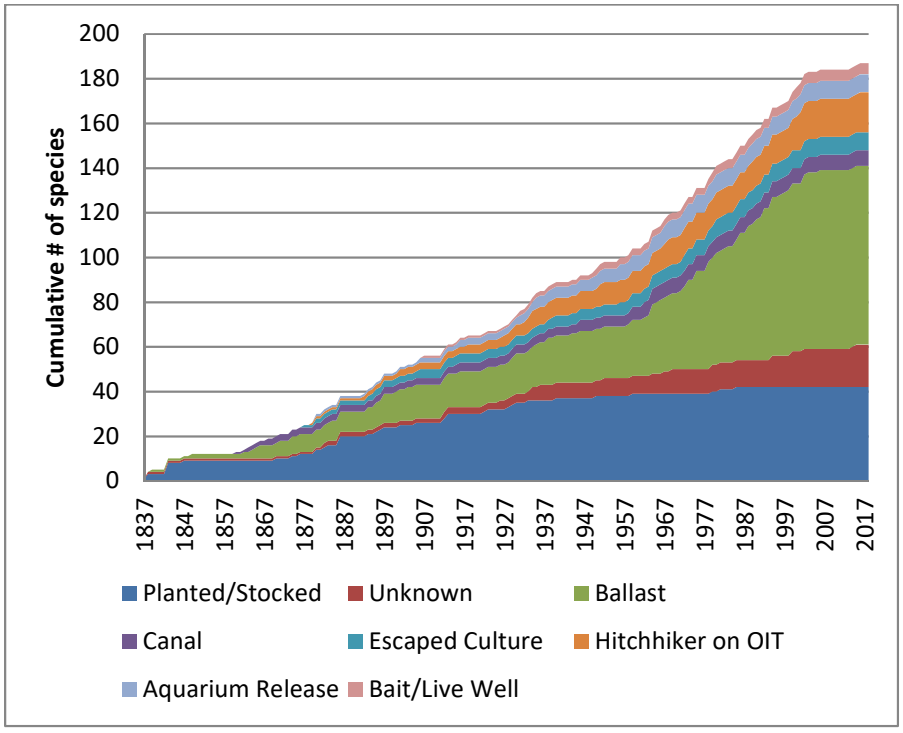


Figure 1. Cumulative establishment of aquatic non-indigenous species by vector in the Great Lakes basin. 187 total species as of 2018.
Source: GLANSIS

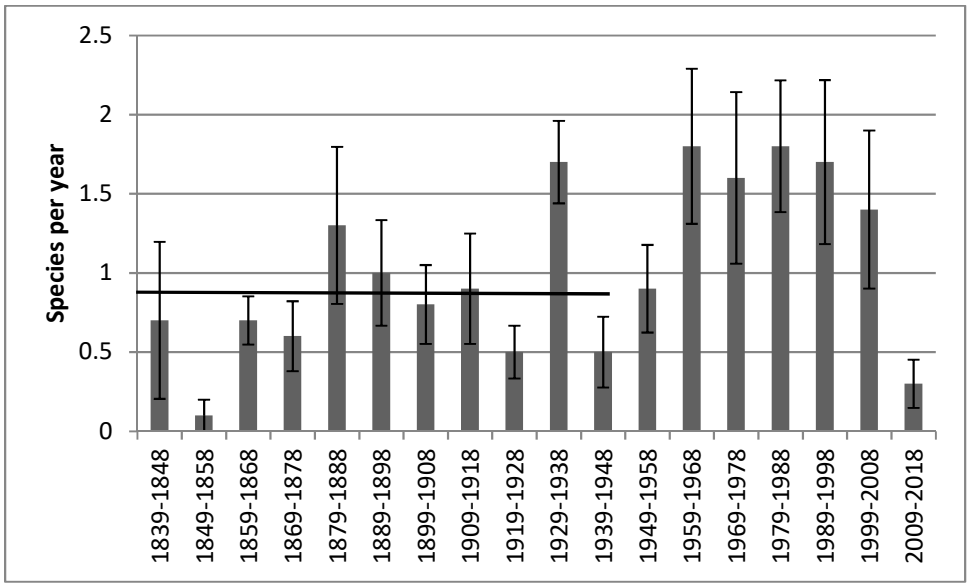


Figure 2. Decadal invasion rate. Solid line indicates the pre-1950 average.
Source: GLANSIS

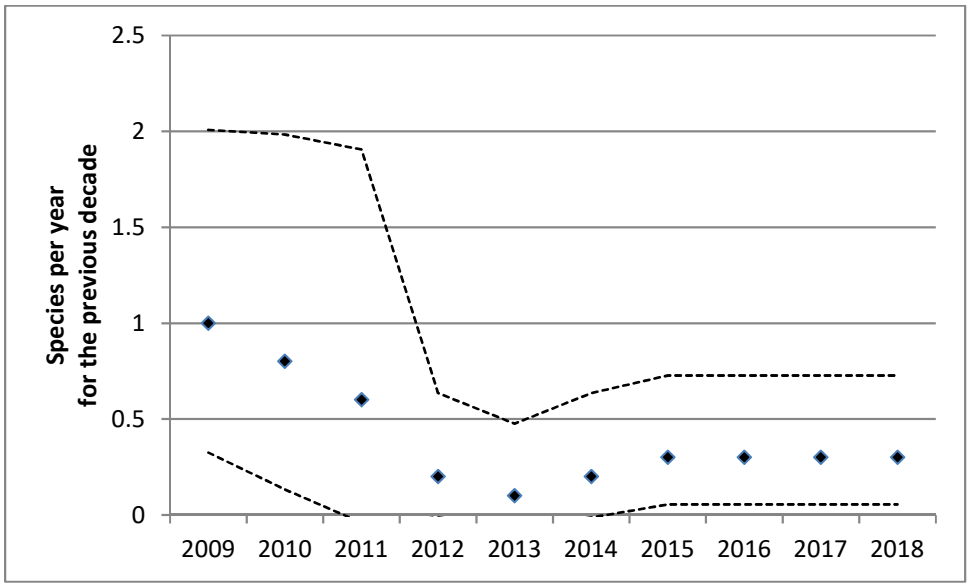


Figure 3. Running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

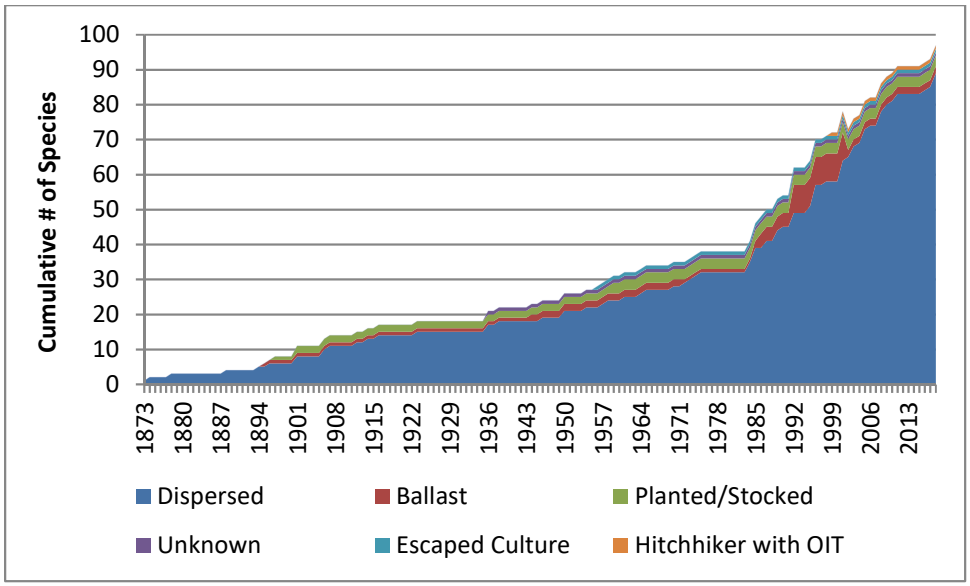


Figure 4. Cumulative establishment in Lake Superior by vector. This includes species non-native to Lake Superior that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Superior, regardless of the intra-basin vector. 97 species have become established in Lake Superior as of 2018. OIT = Organisms in trade.
Source: GLANSIS

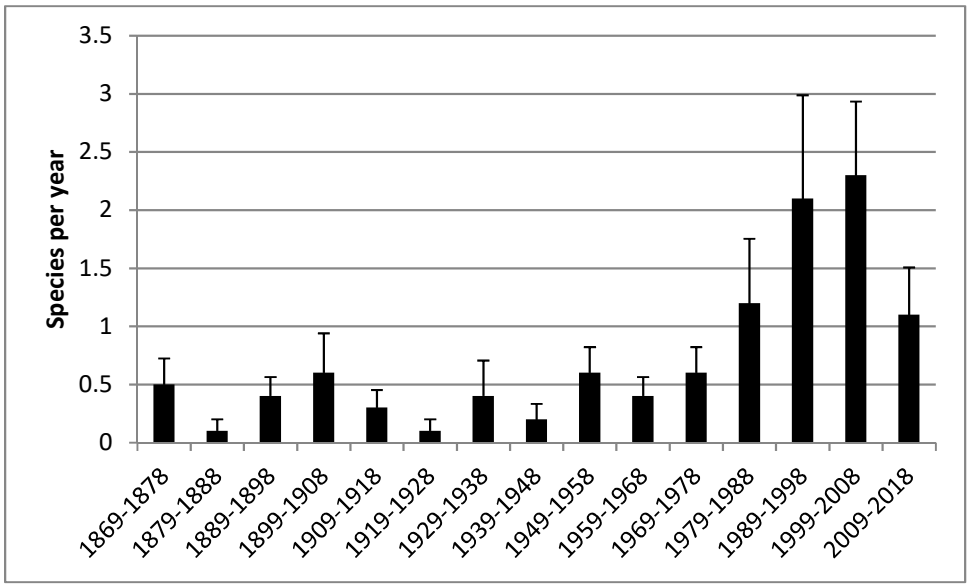


Figure 5. Decadal rate of new non-indigenous species establishing in Lake Superior (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

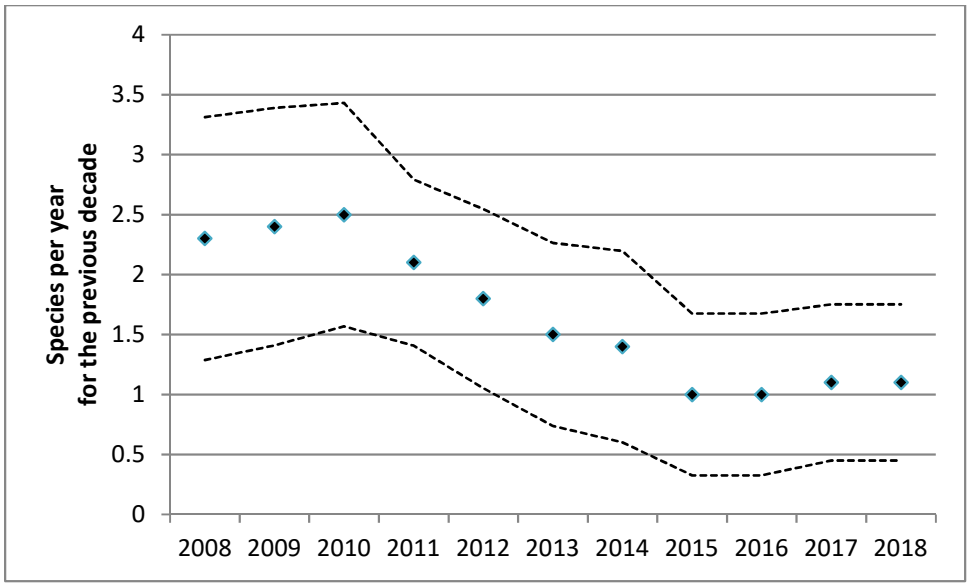


Figure 6. Lake Superior running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

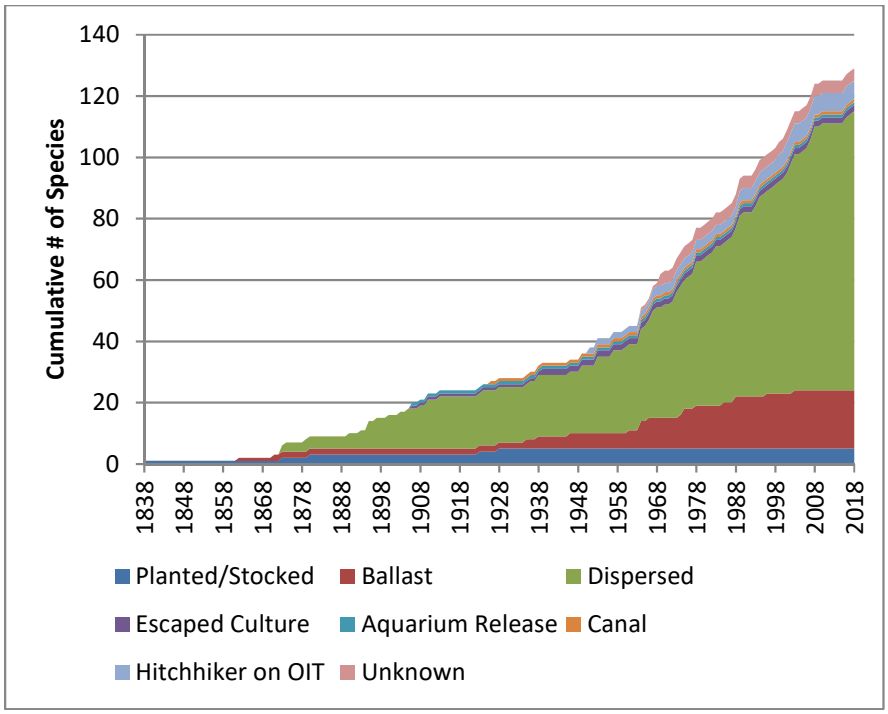


Figure 7. Cumulative establishment in Lake Michigan by vector. This includes species non-native to Lake Michigan that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Michigan, regardless of the intra-basin vector. 127 species have become established in Lake Michigan as of 2018.
Source: GLANSIS

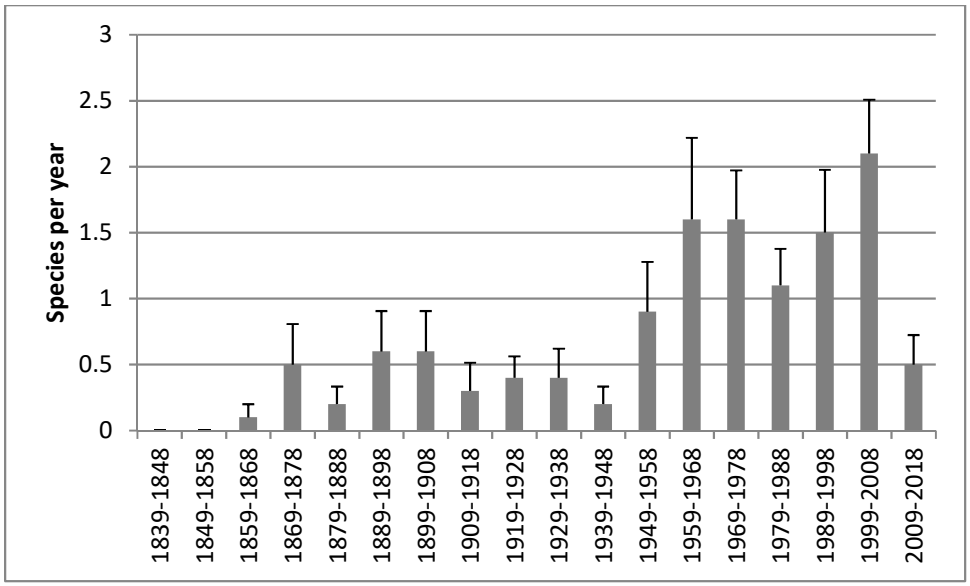


Figure 8. Decadal rate of new non-indigenous species establishing in Lake Michigan (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

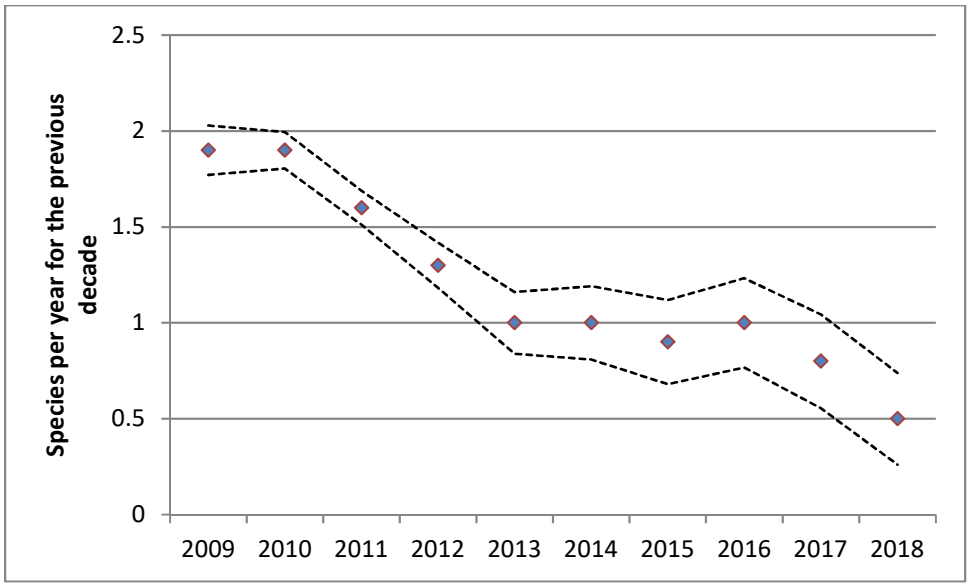


Figure 9. Lake Michigan running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

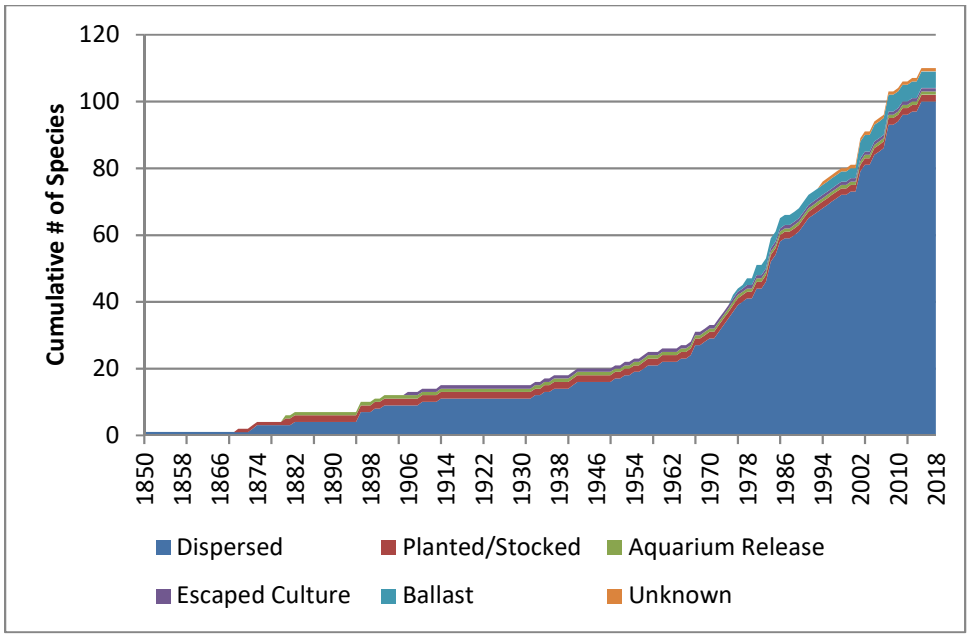


Figure 10. Cumulative establishment in Lake Huron by vector. This includes species non-native to Lake Huron (including the St. Marys River) that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Huron, regardless of the intra-basin vector. 110 species have become established in Lake Huron as of 2018.
Source: GLANSIS

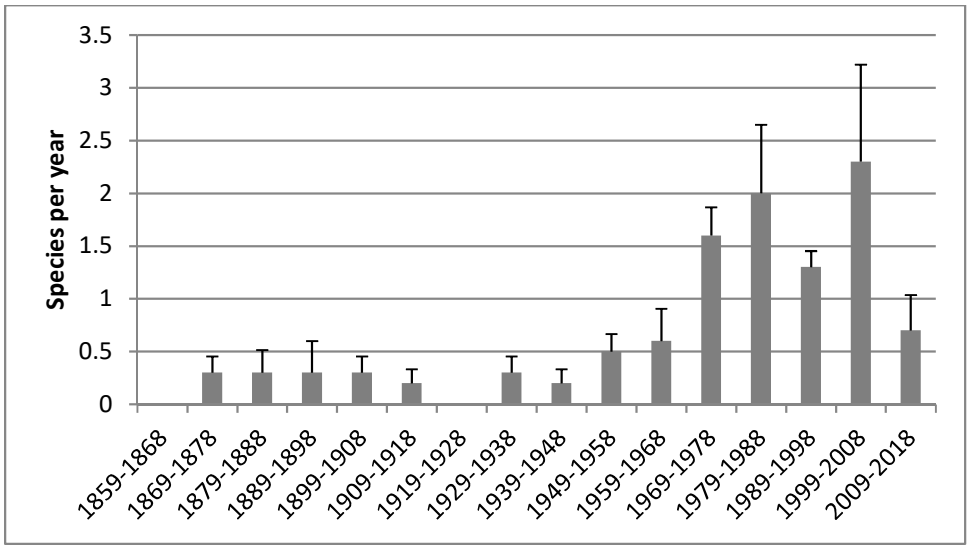


Figure 11. Decadal rate of new non-indigenous species establishing in Lake Huron (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

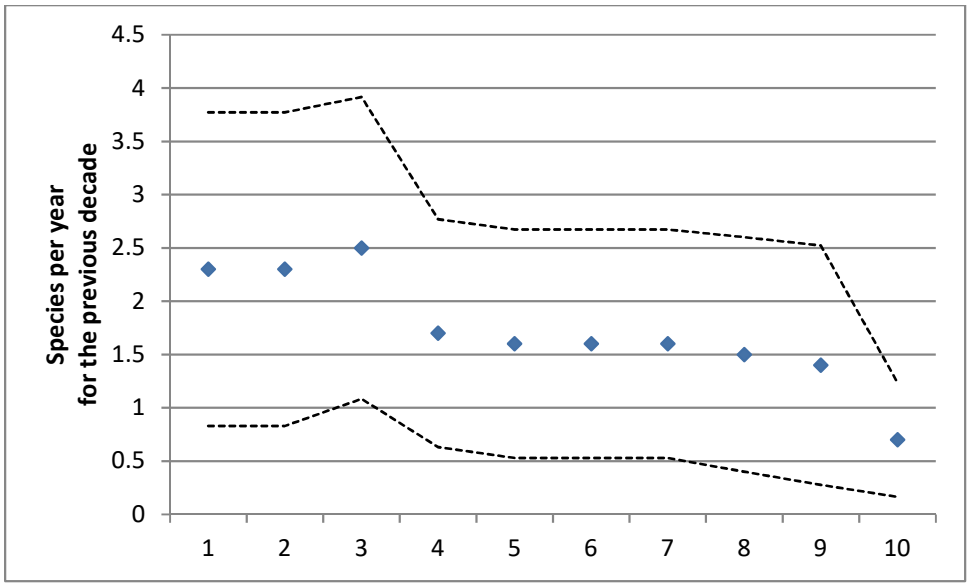


Figure 12. Lake Huron running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

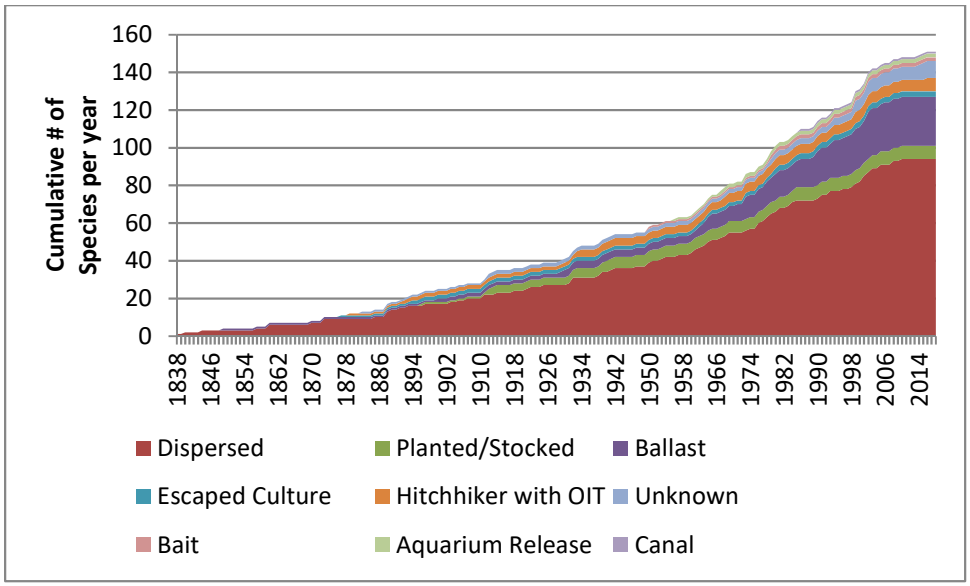


Figure 13. Cumulative establishment in Lake Erie by vector. This includes species non-native to Lake Erie (including the Huron-Erie Corridor) that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Erie, regardless of the intra-basin vector. 151 species have become established in Lake Erie as of 2018.
Source: GLANSIS

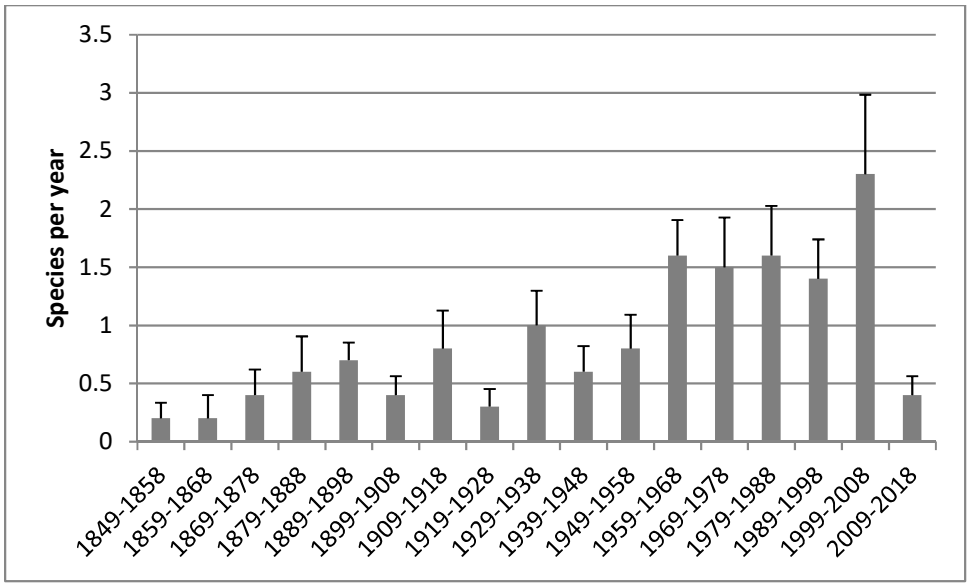


Figure 14. Decadal rate of new non-indigenous species establishing in Lake Erie (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

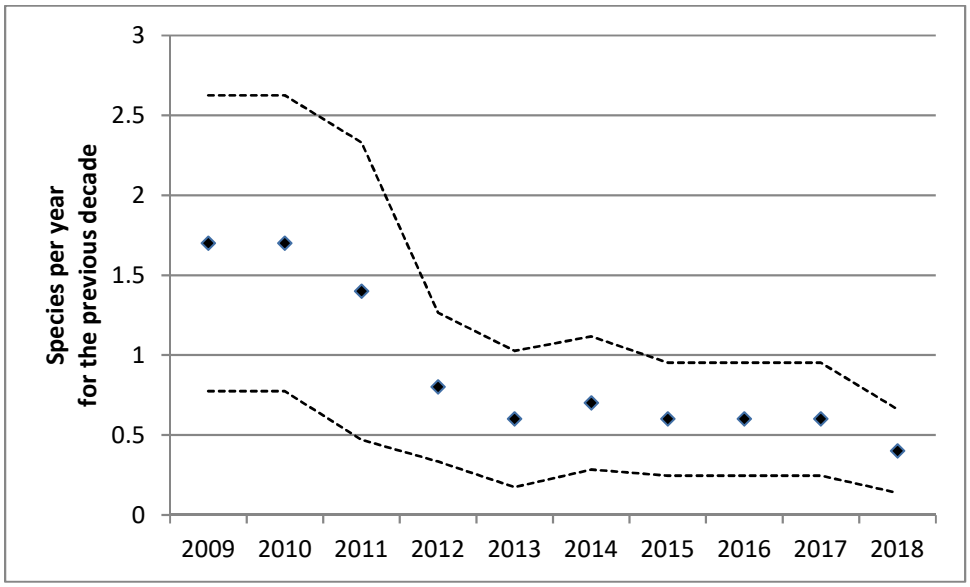


Figure 15. Lake Erie running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

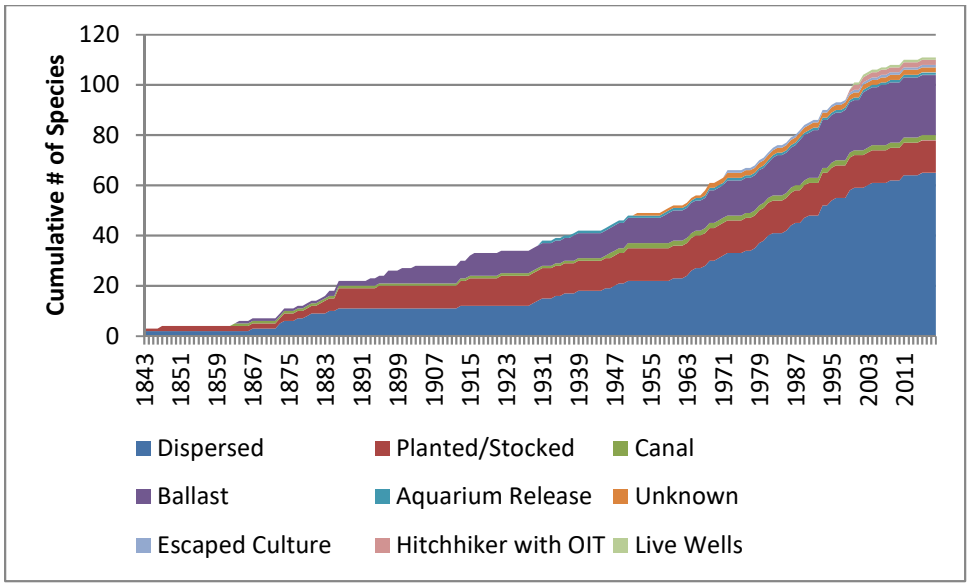


Figure 16. Cumulative establishment in Lake Ontario by vector. This includes species non-native to Lake Ontario (including the Niagara River) that may be native to other portions of the Great Lakes basin. Dispersed includes all species that were first established elsewhere in the Great Lakes basin before spreading to Lake Ontario, regardless of the intra-basin vector. 111 species have become established in Lake Ontario as of 2018.
Source: GLANSIS

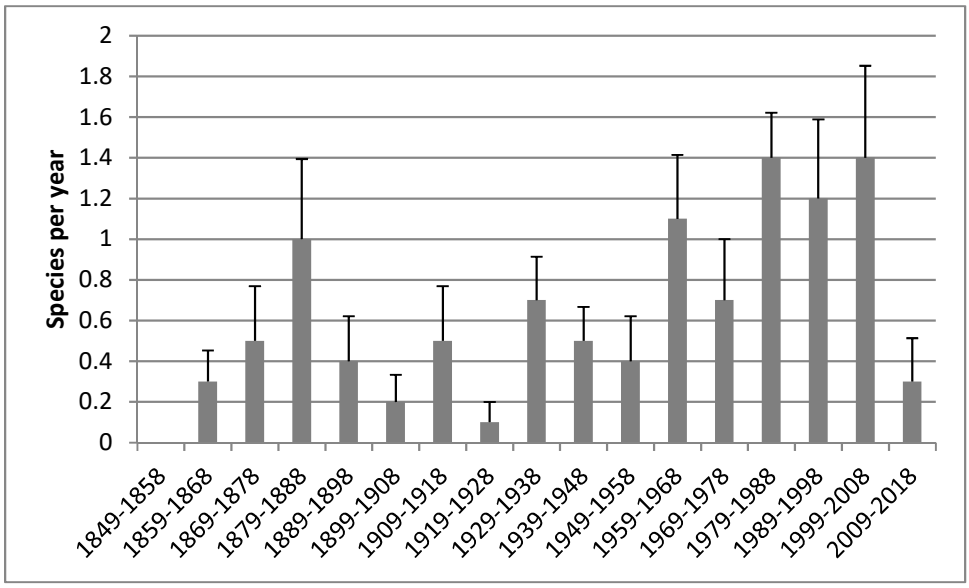


Figure 17. Decadal rate of new non-indigenous species establishing in Lake Ontario (regardless of origin). Solid line indicates the pre-1950 average.
Source: GLANSIS

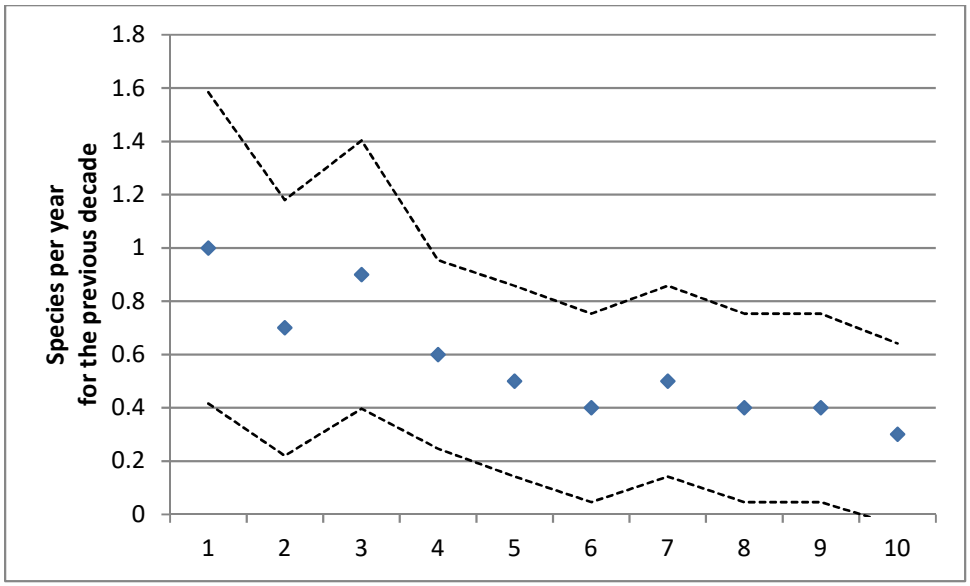


Figure 18. Lake Ontario running decadal invasion rate for 2009-2018 with 95% confidence intervals.
Source: GLANSIS

Sub-Indicator: Impacts of Aquatic Invasive Species (AIS)

Overall Assessment
Status: Poor
Trends:
10-Year Trend: Deteriorating
Long-term Trend (1837-2017): Deteriorating
Rationale: Total number of aquatic non-indigenous species (ANS) in the Great Lakes has stabilized over the last decade, but this number remains over double the pre-1950 level. While total non-indigenous species correlates well to the total number of invasive species in the system, this sub-indicator does not adequately capture impact because not all non-indigenous species have significant negative impacts. Of the 187 non-indigenous species that have become established in the Great Lakes and the 19 range expansion species (native to part of the Great Lakes but non-indigenous to other parts), 64 (31%) have been assessed as having significant negative environmental or socioeconomic impact to date (NOAA 2014 and GLANSIS). *Thermocyclops crassus*, *Mesocyclops pehpeiensis* and *Djaphanosoma fluviatile* are three non-indigenous species that have become established in the last decade, but the potential impacts remain unknown so they are not included in this impact assessment. The filamentous alga, *Ulva flexuosa*, introduced 2003, is the most recently introduced species to demonstrate significant (moderate) environmental and socio-economic impact. Even this number of invasive species alone fails to adequately capture total impact because not all non-indigenous species have equal impacts. Impact is a function of the number of invasive species, the spread of these species (number of watersheds impacted) –and the strength and diversity of each specie's effects on the native system and economy of the region. Even in the absence of new introductions, the impact of these species to the Great Lakes basin continues to increase as high impact species first introduced more than a decade ago continue to expand their ranges into additional lakes. Overall Impact Factor (IF) for the Great Lakes (a function of the number of species, the number of lake basins to which each has spread, and the species-specific impact factors) has increased 7% in the last decade and has tripled since the opening of the Seaway in 1950.

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

Lake-by-Lake Assessment
Lake Superior
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1867-2017): Deteriorating
Rationale: The overall impact factor for US waters of Lake Superior (a function of the number of species, the number of subwatersheds to which each has spread, and the species-specific impact factors) has increased more than 16-fold since 1950, leading us to assess the current status as poor and deteriorating. Five species that have proven environmental or socio-economic impacts have expanded into Lake Superior in the last decade from the lower lakes – *Heterosporis sutherlandae*, VHS, *Bithynia tentaculata*, Quagga mussels and *Typha angustifolia*. In addition to these five species, 16 other species have also expanded their ranges within the Lake Superior basin during this period. Species-specific impact factors range from 2 (Eurasian ruffe) to 55 (Zebra mussels) and new subwatersheds impacted range from 1 (*Potamogeton crispus*) to 7 (*Phragmites australis australis).* Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Superior basin has increased 22% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Superior and across the Lake Superior basin.

Lake Michigan
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1838-2017): Deteriorating
Rationale: The overall impact factor for Lake Michigan (a function of the number of species, the number of subwatersheds to which each has spread, and the species-specific impact factors) has increased 13-fold since 1950,

leading us to assess the current status as poor and deteriorating. Two species that have proven environmental or socio-economic impacts have expanded into Lake Michigan in the last decade from the other Great Lakes – VHS and Starry stonewort. In addition to these two species, 39 other species have expanded their ranges within the Lake Michigan basin during this period. Species-specific impact factors range from 2 (Marsh thistle) to 55 (Zebra mussels) and new subwatersheds impacted range from 1 (Common carp) to 22 (*Phragmites australis australis*). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Michigan basin has increased 39% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Michigan and across the Lake Michigan basin.

Lake Huron (including St. Marys River)
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1870-2017): Deteriorating
Rationale: The overall impact factor for U.S. waters of Lake Huron (a function of the number of species, the number of subwatersheds to which each has spread, and the species-specific impact factors) has increased more than 16-fold since 1950, leading us to assess the current status as poor and deteriorating. Five species that have proven environmental or socio-economic impacts have expanded into Lake Huron from the other Great Lakes in the last decade – *Phragmites australis australis, Hydrocharis morsus-ranae, Najas minor, Didymosphenia geminata* and Starry stonewort. In addition to these five species, 23 other species have expanded their ranges within the Lake Huron basin during this period. Species-specific impact factors range from 2 (Eurasian ruffe) to 55 (Zebra mussels) and new subwatersheds impacted range from 1 (New Zealand mudsnail) to 10 (*Phragmites australis australis*). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Huron basin has increased 51% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Huron and across the Lake Huron basin.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1838-2017): Deteriorating
Rationale: The overall impact factor for U.S. waters of Lake Erie (a function of the number of species, the number of subwatersheds to which each has spread, and the species-specific impact factors) has increased nearly 12-fold since 1950, leading us to assess the current status as poor and deteriorating. Two species that have proven environmental or socio-economic impacts have expanded into Lake Erie from the other Great Lakes in the last decade – *Persicaria maculosa* and *Phragmites australis australis*. In addition to these two species, 35 other species have expanded their ranges within the Lake Erie basin during this period. Species-specific impact factors range from 2 (Redear sunfish) to 45 (Quagga mussels) and new subwatersheds impacted range from 1 (Carolina fanwort) to 20 (*Phragmites australis australis*). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Erie basin has increased 31% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Erie and across the Lake Erie basin.

Lake Ontario (including Niagara River)
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1843-2017): Deteriorating
Rationale: The overall impact factor for U.S. waters of Lake Ontario (a function of the number of species, the number of subwatersheds to which each has spread, and the species-specific impact factors) has increased 6-fold since 1950, leading us to assess the current status as poor and deteriorating. One specie that has proven environmental or socio-economic impact has expanded into Lake Ontario from the other Great Lakes in the last decade- *Butomus umbellatus*. In addition to this one specie, 31 other species have expanded their ranges within the Lake Ontario basin during this period. Species-specific impact factors range from 2 (Garden loosestrife) to 55 (Zebra mussels) and new subwatersheds impacted range from 1 (Carolina fanwort) to 7 (Reed canary grass). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Ontario basin has increased 20% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Ontario and across the Lake Ontario basin. No data are included to assess the International Section of the St. Lawrence River.

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-Native (ANS) and aquatic invasive species (AIS) in the Laurentian Great Lakes.

This sub-indicator will also qualify the AIS as pressure (impact) 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. Fundamental to this goal is to control existing, and 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
The previous report used cumulative number of non-indigenous species as a single sub-indicator. With this report RATE (measuring progress toward prevention goals) and IMPACT (measuring progress towards control/eradication goals) were separated. This sub-indicator is intended to capture key facets of the impact of invasive species.
For this new sub-indicator, a multi-pronged semi-quantitative indicator of species-specific impact is used (Sturtevant et al 2014) combined with spread over time (impact applied as the species invades each new lake or watershed). The Great Lakes have 5 sub-basins (lakes), so the maximum contribution of any species to the index is 5 times its species-specific impact factor. Meanwhile, the individual lake basins are calculated on watersheds (HUC8) – these vary from 10 for Lake Ontario to 33 for Lake Michigan. Using only the spread between lakes, and not sub-watersheds, for the overall assessment was deliberately done to avoid 'double counting' within this sub-indicator. 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 invasive (impactful) and 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* from Erie to Superior have the same weight – one species moving into the lake. In this assessment, the movement of zebra mussel from Michigan to Superior scores 55 and the movement of *Thermocyclops* from Erie to Superior scores zero (*Thermocyclops* impact is currently assessed at unknown/zero).

Ecological Condition
Background
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 basinwide efforts such as that of Great Lakes Fishery Commission's sea lamprey control program, with an annual budget of about $18 million, and local responses, such as the $1,040-$26,000 cost per acre of Eurasian watermilfoil removal (Rosaen et al. 2012). Economic impacts from Dreissenid Mussel control and monitoring are estimated at $1.2 million annually per power plant, $1.97 million for removal of 400 cubic yards at a paper plant, and $480,000-$540,000 annually at a water treatment plant (Rosaen et al. 2012).

At least 31% of the non-indigenous species found in the Great Lakes can be considered to be invasive in that they have high to moderate environmental and/or socioeconomic impact as assessed using NOAA's method (Sturtevant et al 2014) –See Figure 1 (red + pink portions of the graph). Approximately one-third of the species are assessed as having little to no impact (green portion of the graph) – while not native, these species are not causing harm. While there is some time-lag following introduction during which the impact of a species is unknown, 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 to properly assess their environmental impact). If the species which are currently unable to be fully assessed (due to lack of data – scored 'Unknown' – grey portion of the graph) follow the trends of the assessed species the proportion of non-indigenous species which are harmful may be closer to 60%. For previous reports, the total number of non-indigenous species (whole graph) were used as the indicator for impact. For this report, ONLY the species which can be demonstrated to be invasive by the NOAA assessment (red + pink portion of the graph) are considered in looking at impact status and trends.

This sub-indicator is primarily intended to assess progress towards control of AIS. Control of the spread of invasive species into new watersheds is a critical component of management efforts. Using 1950 (just prior to opening of the St. Lawrence Seaway) as a benchmark, it can be seen that nearly every watershed in the region has experienced an increase in the number of invasive species within its boundaries (Figure 3). By the NOAA index, no new invasive species have entered the Great Lakes since 2003 (*Hemimysis anomala* which was introduced 2006 and *Thermocyclops crassus* introduced in 2014 are still scored as unknown impact, not invasive). Yet the species introduced prior continue to expand their ranges resulting in an increase to their overall impact.

This sub-indicator was devised to look specifically at the cumulative impact of invasive species as they spread across the lakes, weighted by the diversity and strength of known impact of the individual species (Figure 2). Even in the absence of new introductions, as high and moderate impact invasive species enter new lake basins, the cumulative impact of these species increases. The year 1950 (just prior to the opening of the St. Lawrence Seaway and the inflection point at which the rate of introduction began to increase logarithmically) is used as the benchmark for comparison of the status of the system. The overall cumulative impact of invasive species on the Great Lakes has tripled since 1950 by our assessment. This increase is the result of the invasion of 22 new species from beyond the Great Lakes taken together with movement between lake basins of the 42 species that had been introduced prior to 1950. In the last decade, even in the absence of new invaders from outside the Great Lakes, a 7% increase can be seen in the impact of invasive species as they spread to additional lake basins within the region.

While spread is a critical driver in the changes, cumulative impact shows that the strength and diversity of impacts of particular species are also important. The top 10 invasive species – Zebra mussel, Quagga mussel, Alewife, Sea lamprey, *Phragmites*, White perch, Eurasian watermilfoil, BKD, VHS and Common carp – are responsible for approximately 60% of the overall cumulative impact, https://www.glerl.noaa.gov/pubs/tech_reports/glerl-161/tm-161.pdf). The difference between zebra mussel and quagga mussel (2nd highest score) is based on the greater diversity of impacts of the nearshore zebra mussels. Adding the spread data to this to create cumulative impact may also weight nearshore species heavier as the nearshore is divided more finely (with many HUC8 watersheds along the coast) than the offshore zone (1-3 HUC8 watersheds comprise the offshore waters of each lake).

The same basic method to assess the impact of aquatic invasive species in US waters of each of the lake basins was applied. Because the number of watersheds within each lake-basin vary, the index is scaled differently and cannot be used to compare between lakes, only over time within a lake-basin.

More than 82 non-indigenous species have become established in Lake Superior (Figure 4), with 38 of these (46%) exhibiting moderate to high environmental or socioeconomic impact. Twenty-eight (28) of these invasive species entered Lake Superior after 1950 and 5 of those in the last decade. The overall impact factor for US waters of Lake Superior has increased more than 16-fold since 1950 (Figure 5). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Superior basin has

increased 22% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Superior and across the Lake Superior basin.

More than 96 non-indigenous species have become established in Lake Michigan (Figure 6), with 44 of these (46%) exhibiting moderate to high environmental or socioeconomic impact. 21of these invasive species entered Lake Michigan after 1950 but none of those in the last decade. The overall impact factor for Lake Michigan has increased more than 12-fold since 1950 (Figure 7). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Michigan basin has increased 39% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Michigan and across the Lake Michigan basin.

More than 94 non-indigenous species have become established in Lake Huron (Figure 8), with 40 of these (43%) exhibiting moderate to high environmental or socioeconomic impact. Thirty-two (32) of these invasive species entered Lake Huron after 1950 and 4 of those in the last decade. The overall impact factor for US waters of Lake Huron has increased more than 16-fold since 1950 (Figure 9). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Huron basin has increased 51% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Huron and across the Lake Huron basin.

More than 148 non-indigenous species have become established in Lake Erie (Figure 10), with 52 of these (35%) exhibiting moderate to high environmental or socioeconomic impact. Twenty-seven (27) of these invasive species entered Lake Erie after 1950 but only 1 of those in the last decade. The overall impact factor for US waters of Lake Erie has increased nearly 12-fold since 1950 (Figure 11). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Erie basin has increased 31% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Erie and across the Lake Erie basin.

More than 104 non-indigenous species have become established in Lake Ontario (Figure 12), with 43 of these (42%) exhibiting moderate to high environmental or socioeconomic impact. Twenty-one (21) of these invasive species entered Lake Ontario after 1950 but none of those in the last decade. The overall impact factor for US waters of Lake Ontario has increased more than 6-fold since 1950 (Figure 13). Even in the absence of new invaders from beyond the Great Lakes, within the last decade the cumulative impact of these species to the Lake Ontario basin has increased 20% as invasive species first introduced more than a decade ago continue to expand their ranges into Lake Ontario and across the Lake Ontario basin.

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 Invasion of ANS, Sea Lamprey and Dreissenid Mussels – evidence indicates that newly invading species may benefit from the presence of previously established invaders.

Invasion Meltdown: Evidence indicates that newly invading species may benefit from the presence of previously established invaders. That is, the presence of one ANS may facilitate the establishment or population growth of another (Ricciardi 2001). For example, the Sea Lamprey (*Petromyzon marinus*) may have created 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*) 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. 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.

Secondary Shifts in Native Populations: ANS 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. ANS 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 non-indigenous 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 (S. Mackey, Habitat Solutions NA, pers. comm.).
Climate Change: Climate change may also facilitate the northward migration of species as well as altering habitat in a way that favors some invaders over natives or alters their impacts. 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). 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 19 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 Limitations
The state of scientific knowledge remains insufficient to determine the impact of nearly half (49%) of the established non-indigenous and range-expanding species. By limiting this indicator to only those species with proven significant impact, the indicator is a conservative estimate of impact.

Significant effort has been made to quality control historic data for the earliest date of each species for the Great Lakes basin as well as individual lakes. A similar effort is underway, but not yet complete, for the subwatershed (HUC 8) scale. Data for spread of species at the subwatershed (HUC 8) scale is currently available only for the US. Thus the indicators for the individual lake basins are somewhat biased toward the US. Data are comparable for the overall assessment, but Canadian data are lacking for the individual lake assessments. Nonetheless, this remains a conservative indicator of impact.

There is often a delay between the time of introduction for a new aquatic invasive species and the time in which the new species is discovered. This temporal consequence has implications for the cumulative number of invasive species to the Great Lakes, to each lake basin and to HUC 8 watersheds. Monitoring effort also influences the temporal lag.

Because impact factor is calculated as a constant for each species, this indicator fails to take into account 'naturalization' processes in which a species 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.

Cumulative impact factor includes a component of spread at the subwatershed (HUC8) scale. It does not take into account population numbers/density within the subwatershed. Thus cumulative impact factor 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 captured by this indicator. This indicator thus fails to capture successes of programs like 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. It is possible to see a change in 'number of reported species' (what can actually 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). 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.

Aquatic Non-native 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.

Acknowledgements

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Acknowledgments:
Erika Lower, CIGLR – assisted with the data preparation
Kaye LaFond, NOAA GLERL – assisted with GIS mapping

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List of Figures

Figure 1. Cumulative non-indigenous species by impact category – Great Lakes basin.
Data Source: GLANSIS 2018
Figure 2. Cumulative impact of invasive species to the Great Lakes.
Data Source: GLANSIS 2018
Figure 3. Distribution of non-indigenous species, 1950 and current.
Data Source: GLANSIS 2018
Figure 4. Cumulative non-indigenous species by impact category – Lake Superior.
Data Source: GLANSIS 2018
Figure 5. Cumulative impact of invasive species to Lake Superior.
Data Source: GLANSIS 2018
Figure 6. Cumulative non-indigenous species by impact category – Lake Michigan.
Data Source: GLANSIS 2018
Figure 7. Cumulative impact of invasive species to Lake Michigan.
Data Source: GLANSIS 2018
Figure 8. Cumulative non-indigenous species by impact category – Lake Huron.
Data Source: GLANSIS 2018
Figure 9. Cumulative impact of invasive species to Lake Huron.
Data Source: GLANSIS 2018
Figure 10. Cumulative non-indigenous species by impact category – Lake Erie.
Data Source: GLANSIS 2018
Figure 11. Cumulative impact of invasive species to Lake Erie.
Data Source: GLANSIS 2018
Figure 12. Cumulative non-indigenous species by impact category – Lake Ontario.
Data Source: GLANSIS 2018
Figure 13. Cumulative impact of invasive species to Lake Ontario.
Data Source: GLANSIS 2018

Last Updated

State of the Great Lakes 2019 Technical Report

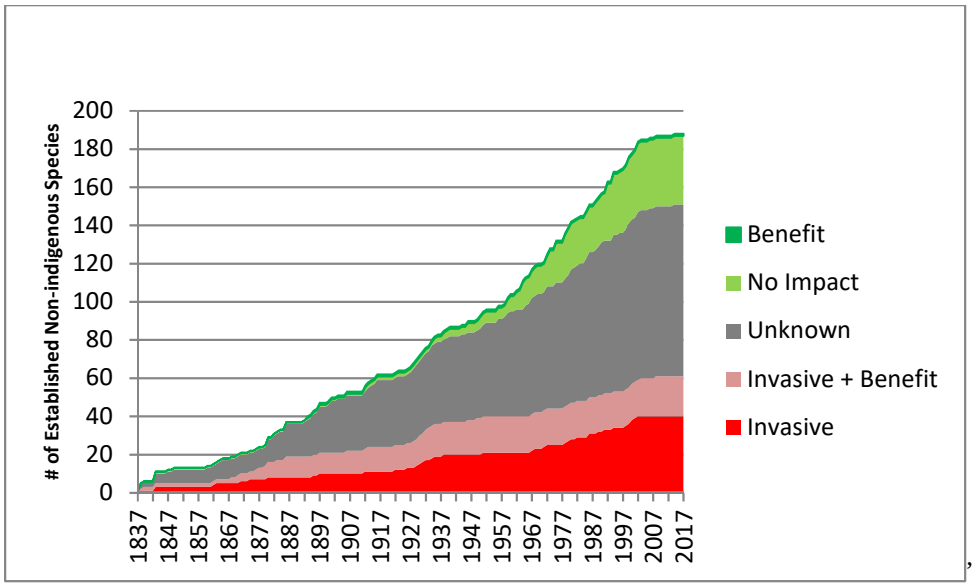


Figure 1. Cumulative non-indigenous species by impact category – Great Lakes basin.
Data Source: GLANSIS 2018

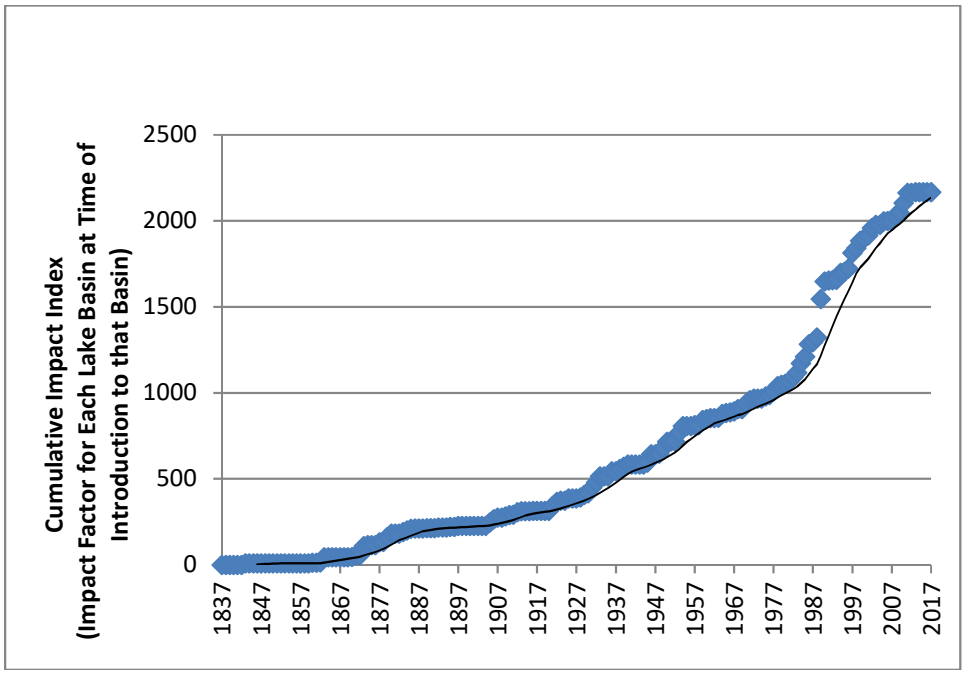


Figure 2. Cumulative impact of invasive species to the Great Lakes (binational).
Data Source: GLANSIS 2018

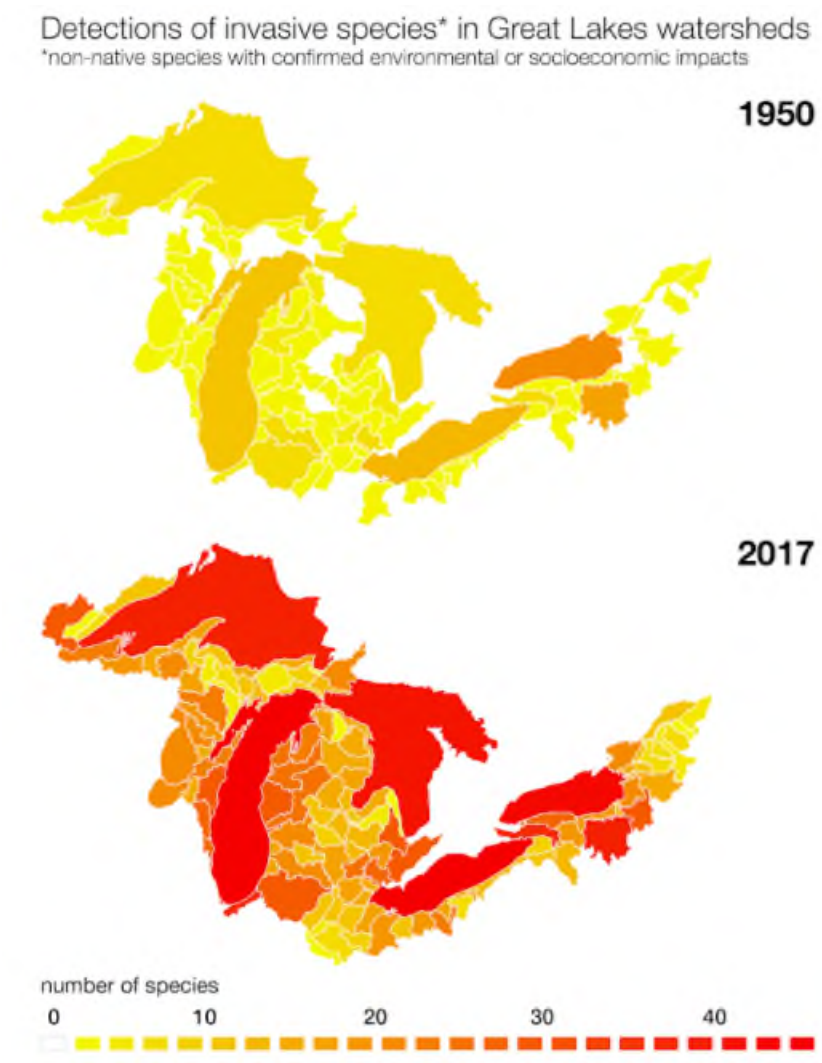


Figure 3. Distribution of non-indigenous species, 1950 and current.
Data Source: GLANSIS 2018

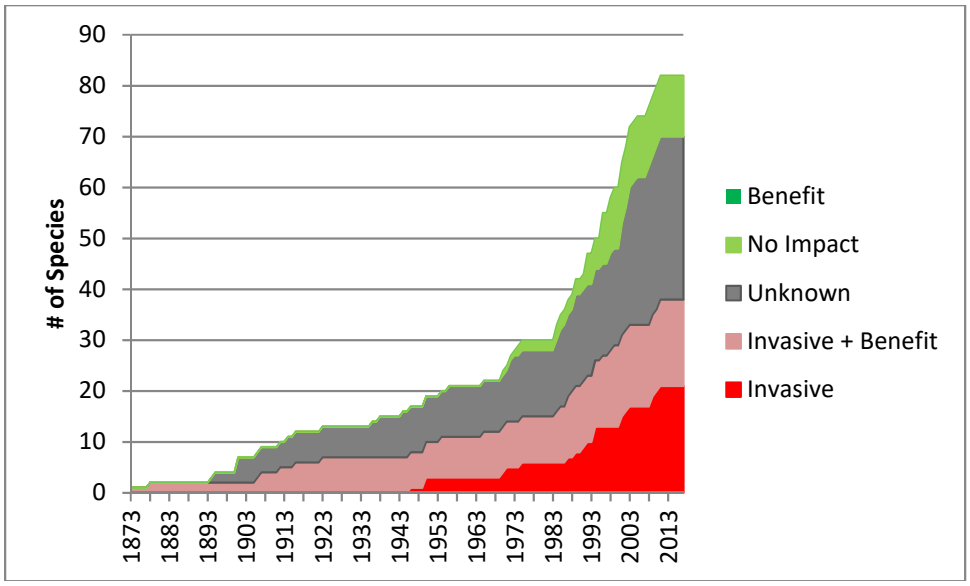


Figure 4. Cumulative non-indigenous species by impact category – Lake Superior.
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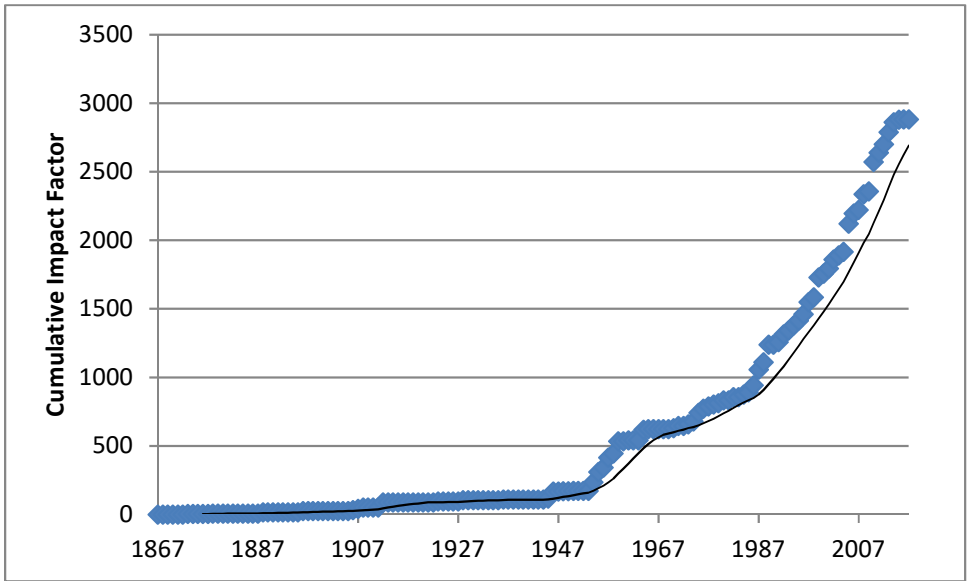


Figure 5. Cumulative impact of invasive species to Lake Superior (US).
Data Source: GLANSIS 2018

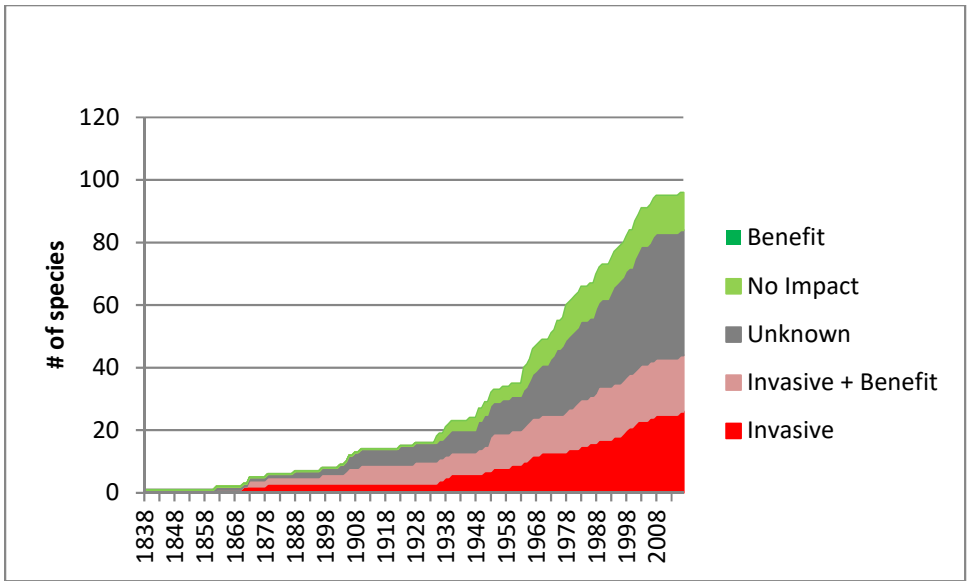


Figure 6. Cumulative non-indigenous species by impact category – Lake Michigan.
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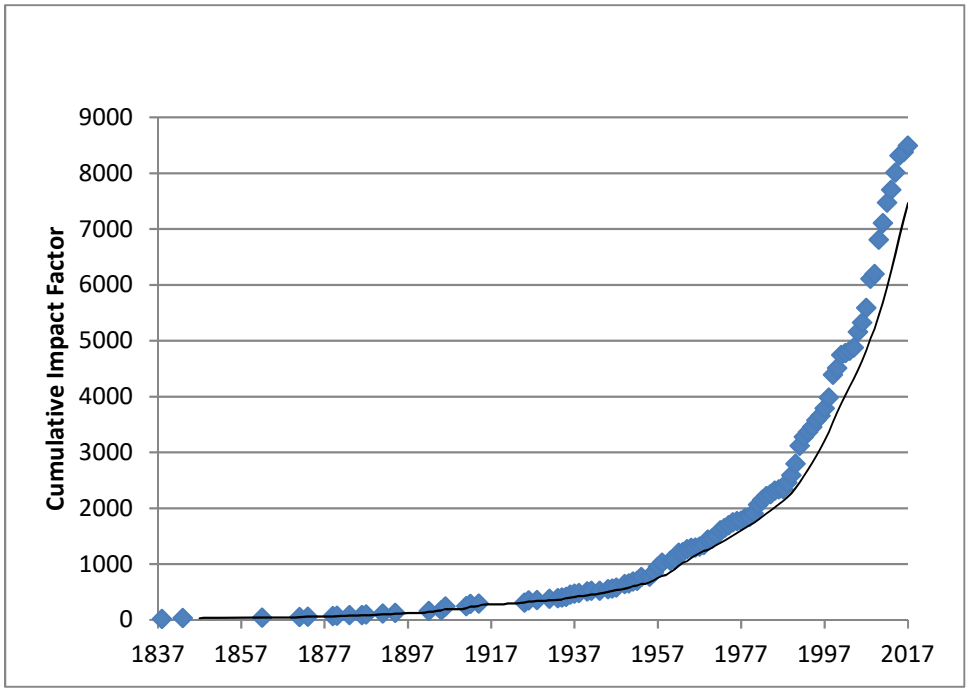


Figure 7. Cumulative impact of invasive species to Lake Michigan (US).
Data Source: GLANSIS 2018

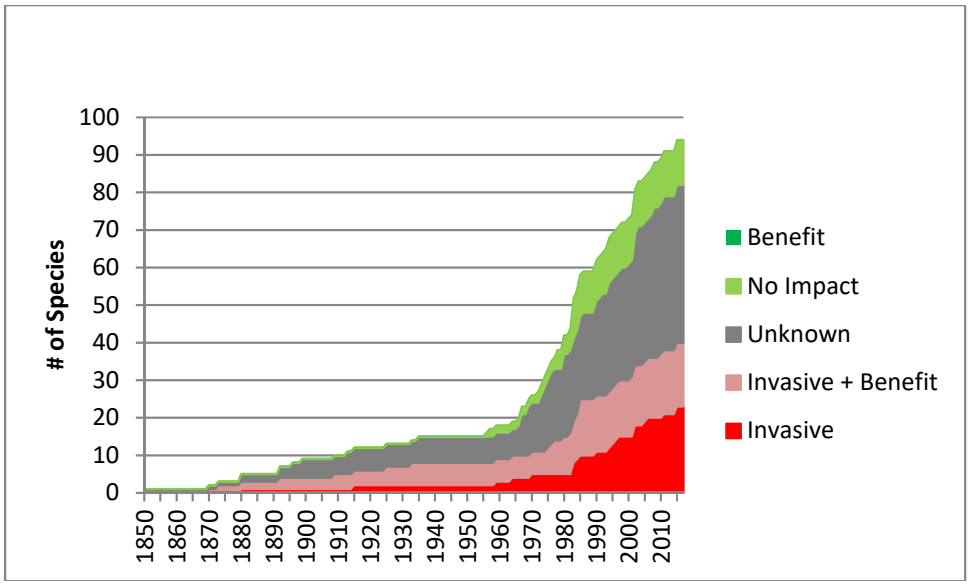


Figure 8. Cumulative non-indigenous species by impact category – Lake Huron.
Data Source: GLANSIS 2018

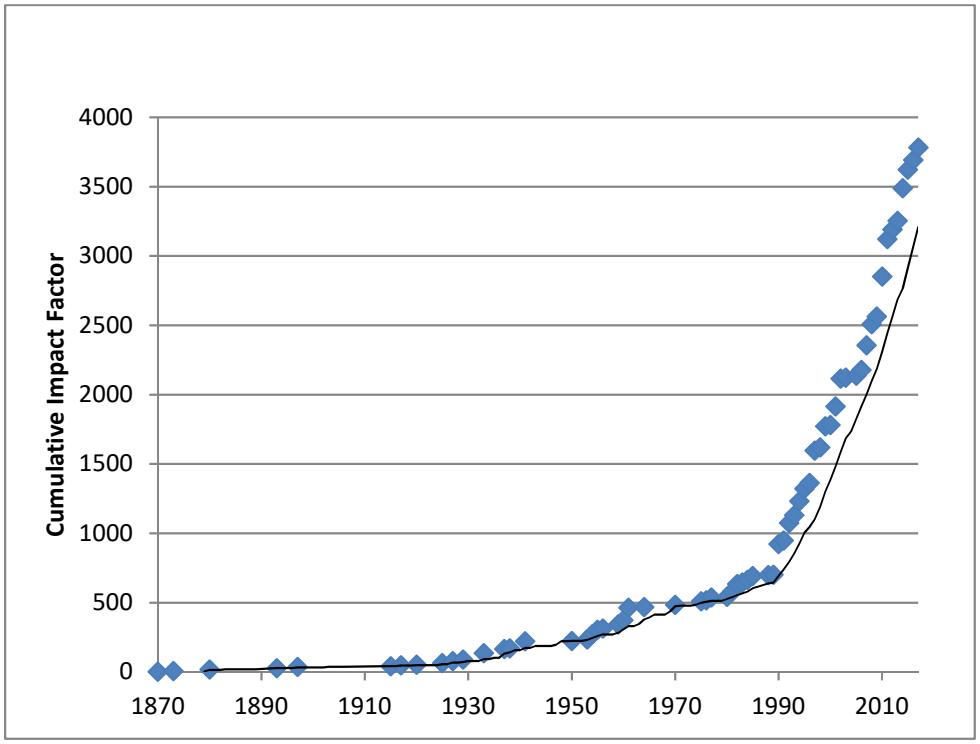


Figure 9. Cumulative impact of Invasive species to Lake Huron (US).
Data Source: GLANSIS 2018

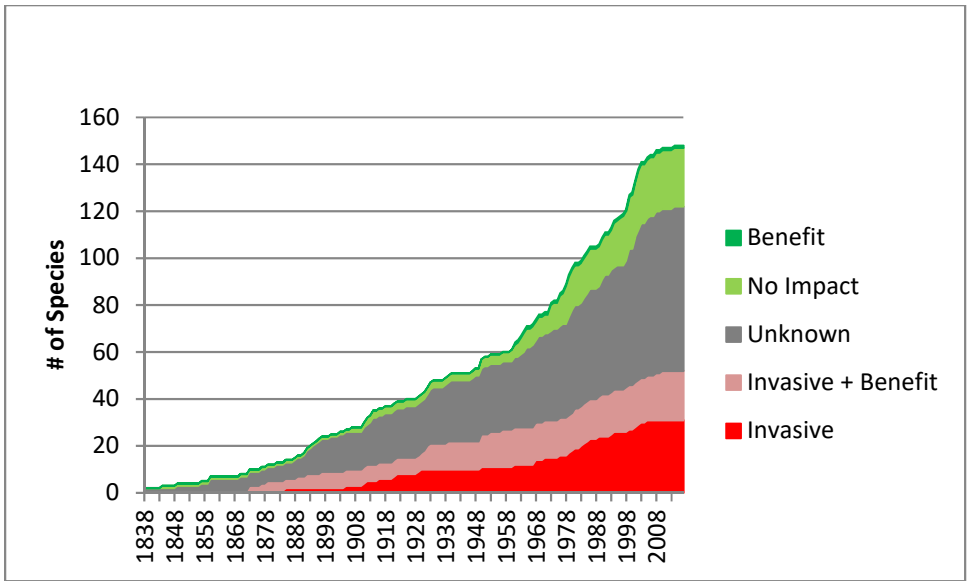


Figure 10. Cumulative non-indigenous species by impact category – Lake Erie.
Data Source: GLANSIS 2018

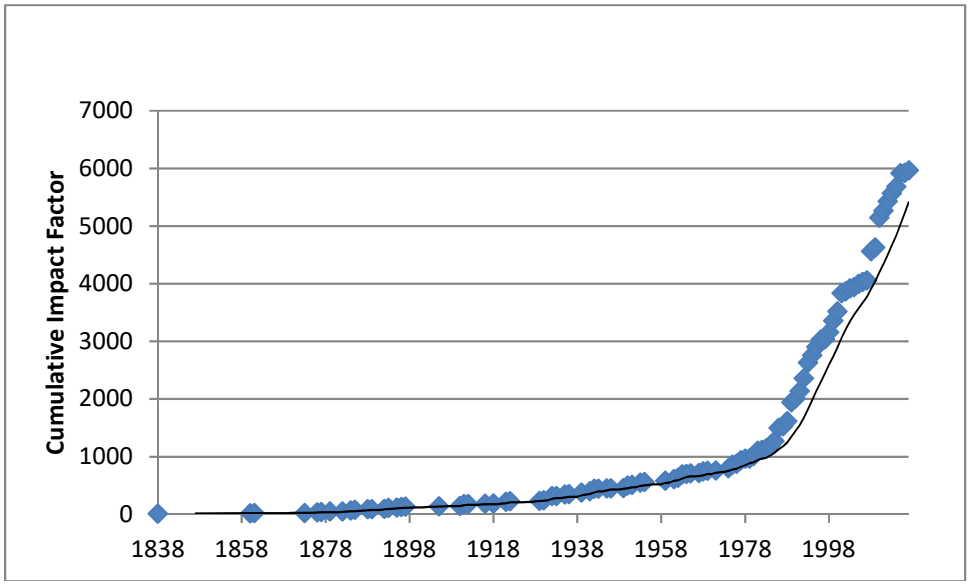


Figure 11. Cumulative impact of invasive species to Lake Erie (US).
Data Source: GLANSIS 2018

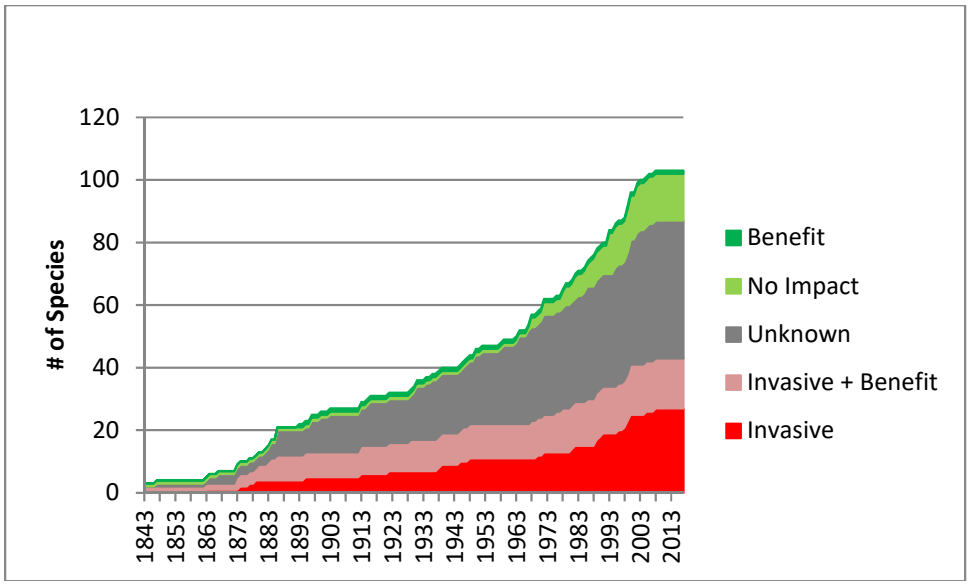


Figure 12. Cumulative non-indigenous species by impact category – Lake Ontario.
Data Source: GLANSIS 2018

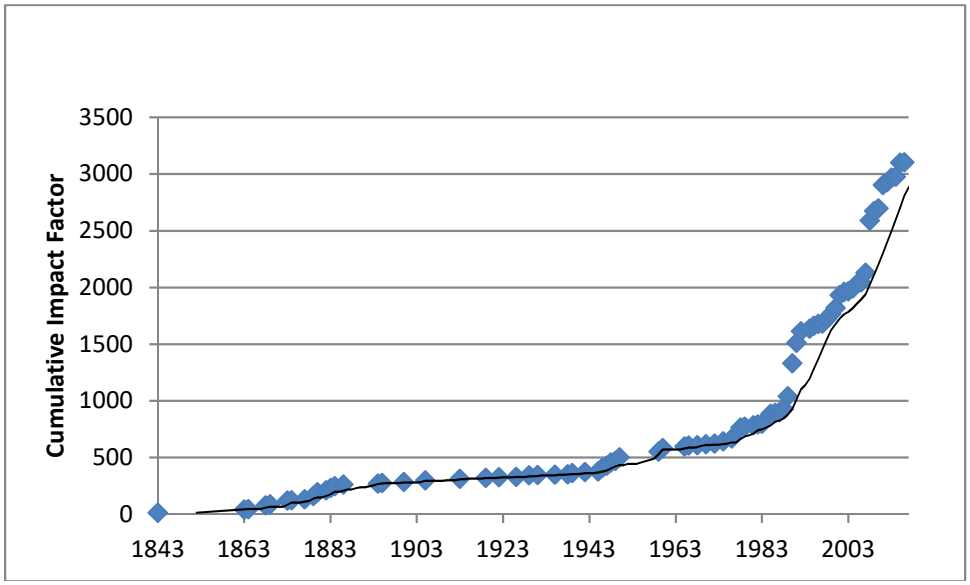


Figure 13. Cumulative impact of invasive species to Lake Ontario (US).
Data Source: GLANSIS 2018

Sub-Indicator: Sea Lamprey

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Improving
Long-term Trend (1985-2017): 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, index estimates of adult Sea Lamprey abundance are meeting targets in Lakes Michigan and Ontario and are above targets in Lakes Superior, Huron, and Erie. The basin-wide 10-year and long-term adult Sea Lamprey abundance trend is declining. More suppression is needed to bring Sea Lamprey populations to targets in all lakes.

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: Unchanging
Long-term Trend (1986-2017): Unchanging
Rationale: Index estimates of adult Sea Lamprey abundance are above target and are unchanging.

Lake Michigan
Status: Good
10-Year Trend: Improving
Long-term Trend (1995-2017): Improving
Rationale: Index estimates of adult Sea Lamprey abundance are meeting the target and have declined since 1995.

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1985-2017): Improving
Rationale: Index estimates of adult Sea Lamprey abundance are above target, but at a relatively low level, and have been declining since 1985.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend: Improving
Long-term Trend (1991-2017): Unchanging
Rationale: Index estimates of adult Sea Lamprey abundance are above target, but have declined since 2012

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Good
10-Year Trend: Improving
Long-term Trend (1987-2017): Unchanging
Rationale: Index estimates of adult Sea Lamprey abundance are meeting the target and have declined since 2006.

Status Assessment Definitions
Good: adult Sea Lamprey index is below target
Fair: adult Sea Lamprey index is above target, but decreasing
Poor: adult Sea Lamprey index is above target and increasing

Trend Assessment Definitions
Improving: adult Sea Lamprey index shows a change toward more 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 to 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 an index stream is removed.

Lake Superior: suppress Sea Lampreys to population levels that cause only insignificant mortality on adult Lake Trout. The calculated adult Sea Lamprey index target in Lake Superior is 9,700 Sea Lampreys.

Lake Michigan: suppress the Sea Lamprey to allow the achievement of other fish-community objectives. The calculated adult Sea Lamprey index target in Lake Michigan is 24,900 Sea Lampreys.

Lake Huron: reduce Sea Lamprey abundance to allow the achievement of other fish-community objectives. The calculated adult Sea Lamprey index target in Lake Huron is 24,100 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 calculated adult Sea Lamprey index target in Lake Erie is 3,000 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 calculated adult Sea Lamprey index target in Lake Ontario is 11,400 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. 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 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 numbers of adult Sea Lampreys migrating into each index stream are estimated with traps using a modified Schaeffer mark/recapture method explained by Mullett et al. (2003). Indices of adult Sea Lamprey abundance are updated on an annual basis.

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 mark/recapture estimates 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. Recently, the GLFC changed their adult Sea Lamprey monitoring protocols moving away from the spawner model in favor of 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. This method to calculate the indices has been applied to historic data so that the results can be compared to the new data.

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 attack smaller fish like cisco, Walleye, and Yellow Perch (GLFC). 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.

Index estimates 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.

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 as additional information to show some of the impact Sea Lamprey have on Great Lakes fish, specifically Lake Trout.

Lake Superior

In Lake Superior, the adult index estimate is above the target and has been unchanging since 2008. Sources of Sea Lampreys that are of concern include the Bad River and lentic (estuaries, bays, and slower moving tributaries) populations in the Kaministiquia, Nipigon, Gravel, and Batchawana rivers where populations are sparsely distributed and granular Bayluscide treatment is less effective than conventional TFM applications. 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. Nevertheless, the adult index estimate has increased since the last report (2017). This increase is associated with an unusually high index estimate from the Brule River and is responsible for contributing more than 40% of the lake-wide index. The reasons for the recent increase are not yet fully understood, but could be a consequence of operational changes at the trap site.

Lake Michigan

In Lake Michigan, the adult index estimate is meeting the target. 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 with additional treatments. In

addition, the Manistique River has been treated seven times since 2003. Reductions in Sea Lamprey abundance during the past ten years are a result of increased lampricide treatment efforts.

Lake Huron
In Lake Huron, the adult index estimate is above the target, but has been decreasing since 2008. 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. Additionally, 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 adult index estimate is above the target, but has been decreasing since 2012 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. The adult Sea Lamprey index has yet to meet the target as expected and has been highly variable, but remains unchanged since the last reporting period (2017). The reasons for the relatively high index is not fully understood. 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 adult index estimate is meeting the target. 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 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, Steelhead, Walleye, Whitefish, 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 Lamprey to colonize new locations is increased with improved aquatic habitat connectivity through the removal of dams and improved water quality. 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. As examples, 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, it is likely only a matter of time before Sea Lamprey reproduction is documented in the Detroit River.

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.

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 Limitations

- 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 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 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., wounds or scars, 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 used to interpret this sub-indicator. These experimental observations are limited and verification of lethality in the field will improve the interpretation of this sub-indicator.

Additional Information

Increases in lampricide treatments have reduced index estimates of adult Sea Lamprey abundance to within target ranges in two 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/prey 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 index estimates at targets.

Acknowledgments

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List of Figures

Figure 1. Streams selected for inclusion in the index of adult Sea Lamprey abundance.
Source: Great Lakes Fishery Commission

Figure 2. Index estimates of adult Sea Lamprey abundance 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. Number of A1-A3 Sea Lamprey wounds per 100 Lake Trout > 532 mm (Superior, Huron, Michigan, and Erie) and number of 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

State of the Great Lakes 2019 Technical Report

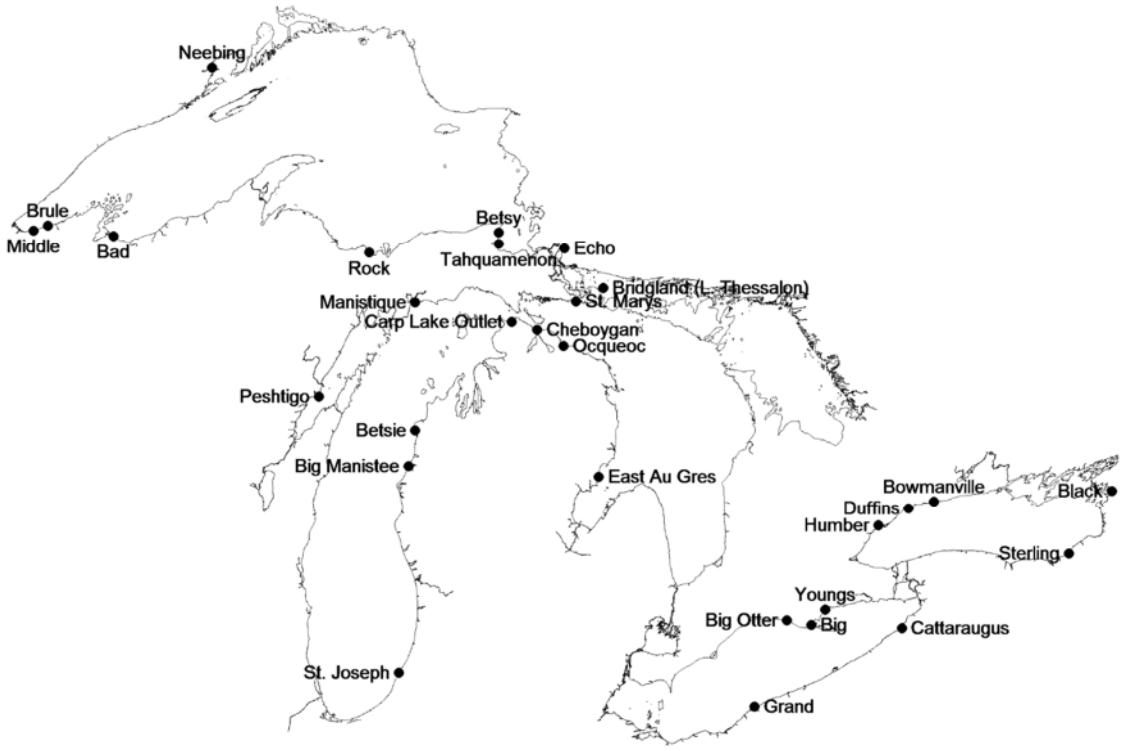


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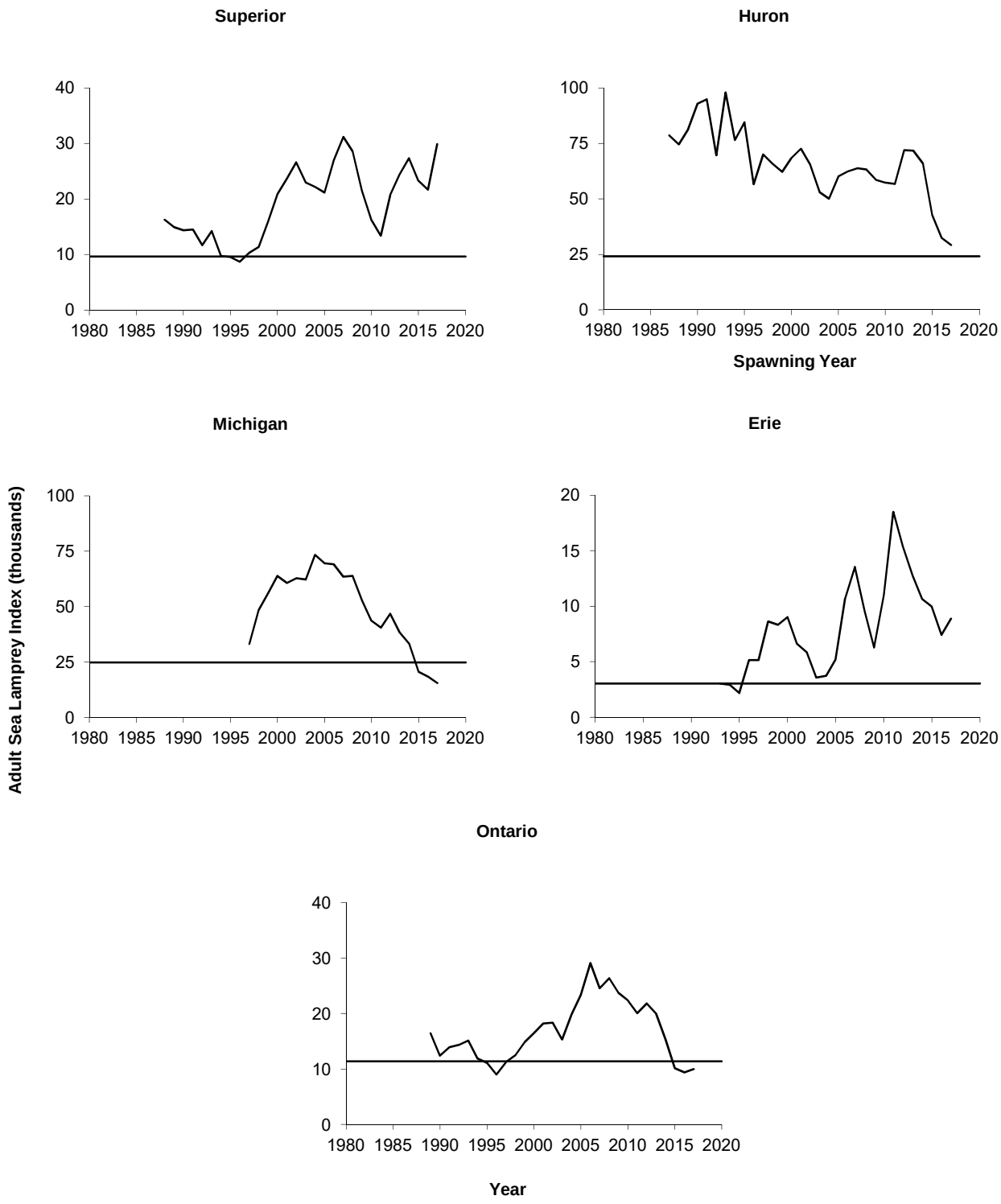


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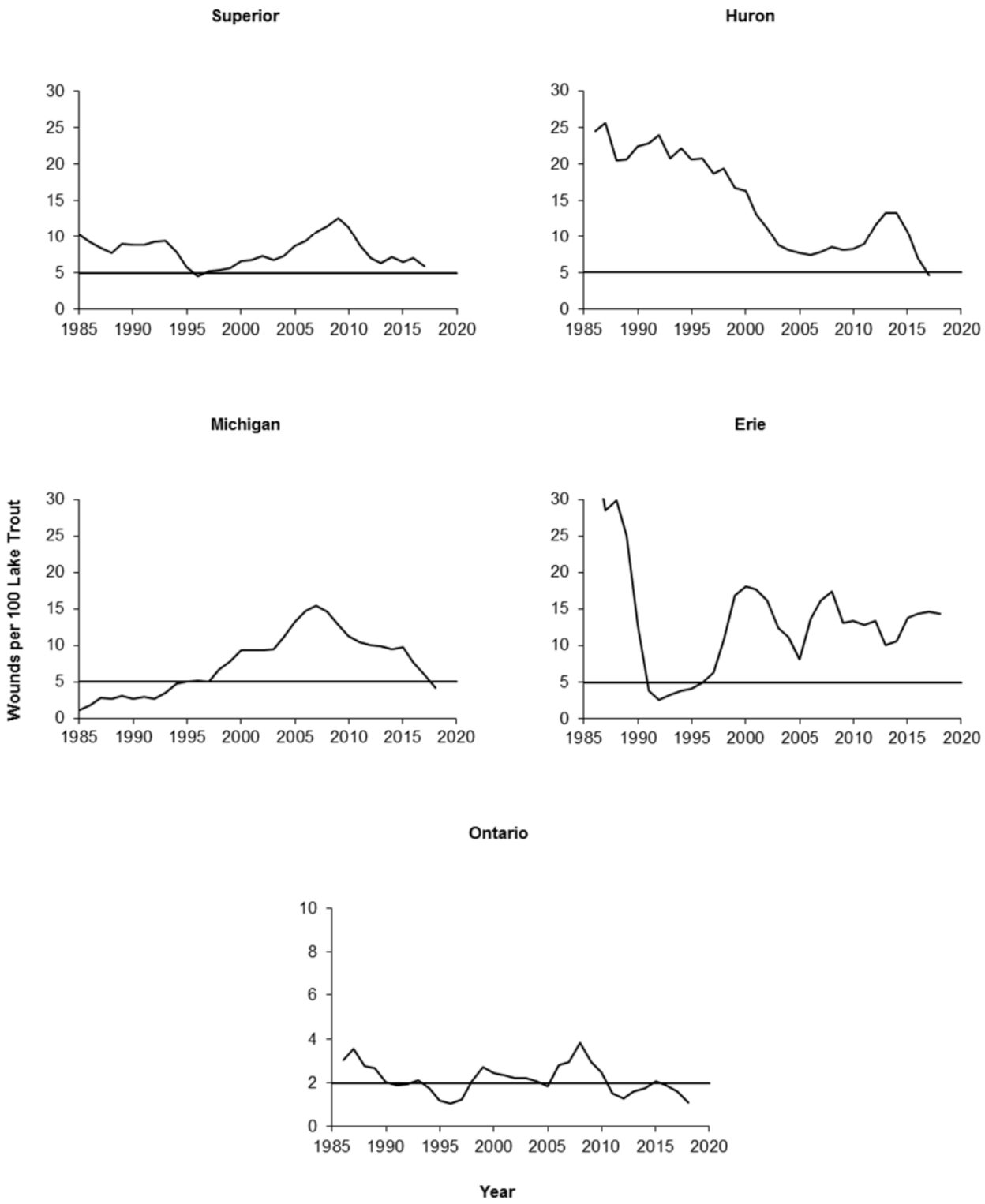


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Source: Great Lakes Fishery Commission

Sub-Indicator: Dreissenid Mussels

Overall Assessment
Status: Poor
Trends:
10-Year Trend: Deteriorating
Long-term Trend (1989-2016): Deteriorating
Rationale: The status of dreissenid mussel populations varies by water depth and particular lake region. In general, populations 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 Lake Erie have been relatively stable at moderate levels for the past decade, while populations in Lake Superior remain at low levels.

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 (1989-2016): 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. Other discovered invasions include 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). The broad spatial coverage but low densities of these introductions indicate that Lake Superior is likely just marginally habitable for dreissenids. Low calcium and other water quality characteristics pose a challenge to spawning success and veliger survival (Trebitz et al. 2019).

Lake Michigan
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1994-2016): 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 1-3), biomass held fairly steady (Figure 2). This observation is a result of an increase in the mean mussel size. Dreissenid density and biomass continued to increase at depths > 90 m (Figures 1-3). 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 survey will take place in 2020.

Lake Huron
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (2000-2012): 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 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 1 and 4), notably to the point that the deep-water dreissenid density in Lake Huron has caught up with that in Lakes Michigan and Ontario (Figure

1, lower panel). The mussels in Georgian Bay are found at overall lower densities than in the Main Basin, but have followed similar depth-specific changes over time. While mussels were not present in North Channel at the sites sampled in 2007 and 2012, a few mussels appeared in the 2017 survey.

Lake Erie
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (1992-2014): Improving
Rationale: An update on trends in dreissenid populations in Lake Erie was provided by Karatayev et al. (2014) and a Coordinated Science and Monitoring Initiative report (Burlakova et al. 2017). Overall, lake-wide densities of dreissenids were much lower in 2014 compared to the peaks in 1993 (zebras) and 1998 (quaggas). The three basins of Lake Erie exhibit different dreissenid mussel trends. Quagga and zebra mussels still coexist in the western basin, but zebra mussels are largely absent in the rest of the lake (Figure 5). In the western basin, zebra mussels have steadily declined while quagga mussels increased in density since the beginning of the surveys in 1992 (Figure 5). In 2014, the increase in quagga density is likely due to smaller mussels from recent spawning (Burlakova et al. 2017). Populations in the western basin are likely limited because of poor food quality (cyanophytes, inorganic particulates) and occasional hypoxia. In the central basin, which experiences regular seasonal hypoxia, zebras have become virtually absent and quagga mussels have declined 5-fold since their peak in 2002. The eastern basin consistently supports the largest population of *Dreissena* (primarily quaggas), though there have been declines since a peak in 2002. The size structure of the eastern basin mussels indicates a lack of hypoxia events and consistent recruitment at most depths (Burlakova et al. 2017).

Lake Ontario
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1994-2013): Deteriorating
Rationale: According to the 2013 lake-wide survey of dreissenid populations in Lake Ontario, quagga mussel densities at 31-90 m were lower than in 2008 and 2003 (Figure 1). Densities at this depth interval appear to have peaked in 2003. Despite declines in density, biomass at depths of 31-90 m increased from 19.3 g m^{-2} in 2008 to 31.2 g m^{-2} in 2013. However, the population at > 90 m still seems to be expanding as densities at these deep depths in 2013 were the highest ever recorded (Figure 1). This observation can be attributed to the greater mean size of mussels in 2013. No zebra mussels were collected in 2013. Sample processing from the 2018 lake-wide survey is in progress, the results of which will provide further insight into these depth-specific trends.

Status Assessment Definitions
Since there are no known methods to eliminate mussels, these status assessments should be consistent with the proposed endpoint. That is, a stable population that is lower than a prior time period. Hence, population abundances can be better predicted in given areas.

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 yet been determined. A proposed endpoint of zero dreissenids is unrealistic. A working qualitative endpoint is the point in time and space in which a dreissenid population becomes stable, or varies within a given range. Such an endpoint will allow for 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 altering nutrient and energy cycling, promoting nuisance algal blooms and benthic algae, and negatively impacting native species of invertebrates and fish. 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 and, when available, biomass of dreissenid mussels in near to offshore benthic habitats of the Great Lakes. The recent lake-wide trends reported for Lakes Michigan, Huron, Ontario, and Erie are based on Cooperative Science and Monitoring Initiative (CSMI) data collected in 2015, 2017, 2013, and 2014, respectively. Also presented are data collected during the annual NOAA/GLERL benthic survey of the southern basin of Lake Michigan. Data are usually summarized by depth zone because dreissenid mussel dynamics are strongly influenced by depth.

The U.S. EPA Great Lakes National Program Office monitoring program assesses density of *Dreissena* (and other benthos) in offshore regions of each of the lakes on an annual basis, but those data are not presented in this report.

Monitoring programs collect data on density, biomass, size-frequency distributions, and length-weight relationships for dreissenid mussels. 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. 2017, Karateyev et al. in review). 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. 2017 for Lake Michigan and Karatayev et al. in review for Lake Huron).

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 *Dreissena r. bugensis*.

Ecological Condition

Dreissenid populations in the Great Lakes are presently in various stages of change. In many offshore regions, populations are increasing, but in some nearshore regions populations seem to be stable or declining. 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, 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 regime, 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 *D. r. bugensis* (quagga mussels) over *D. polymorpha* (zebra mussels) because *D. r. bugensis* has

higher energetic efficiency and can persist in areas with lower food availability (Wilson et al. 2006) and tolerate colder temperatures.

Recent research connects the observed distribution of *Dreissena* in Lake Erie to hypoxia events (Karatayev et al. 2018). 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 can be sufficient to cause mass mortality and thereby reduce *Dreissena* longevity. In contrast, the eastern basin does not experience hypoxia and contains the most stable mussel populations in Lake Erie. *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 – the filtering and nutrient excretion activities of dreissenids may lead to increased frequency, distribution and severity of both inshore (attached/benthic) and offshore algal blooms and favour the predominance of cyanobacteria.
- Phytoplankton – the abundance and composition of phytoplankton has dramatically changed in areas of the Great Lakes where dreissenids have become abundant.
- Impacts of Aquatic Invasive Species – *Dreissena* is 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.

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 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. The data presented here are primarily 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 limited to the main basins of the lakes and exclude the connecting channels. There are few, if any, regular monitoring programs in the connecting channels and, even so, physical factors such substrate variability, current patterns, etc. do not provide the best conditions to assess temporal trends in populations. Canadian data sources are limited due to changes in the monitoring programs within the Department of Fisheries and Oceans Canada. However, the CSMI whole-lake surveys include data from both U.S. and Canadian waters.

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.

Ideally, the dreissenid population should be assessed over most lake regions, particularly in nearshore regions of the open lakes and in shallow bays and basins every 1-2 years. However, this will require a considerable amount of resources and support. Presently, the U.S. EPA GLNPO monitoring program assesses density of *Dreissena* (and other benthos) in offshore regions of each of the lakes on an annual basis. In addition, extensive, whole lake surveys (nearshore and offshore) are conducted in each of the lakes every 5 years as a part of the Coordinated Science and Monitoring Initiative, for which both Canadian and U.S. waters are sampled. Also, NOAA/GLERL conducts a survey throughout the southern basin of Lake Michigan every year. Given other programs that assess dreissenid populations in specific locales as related to different program goals, it seems that these present monitoring programs are adequate to detect population changes.

Acknowledgments

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List of Figures

Figure 1. 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.
Data Sources: Great Lakes Environmental Research Lab, NOAA; Watkins et al. 2007; Birkett et al. 2015; Nalepa et al. 2014, 2017, 2018

Figure 2. Long-term trends of total *Dreissena* in Lake Michigan in 1994/1995, 2000, 2005, 2010, and 2015. Values given are lake-wide means (± SE) at four depth intervals: < 30 m (black, circles), 31-50 m (red, triangles), 51-90 m (blue, squares), and > 90 m (green, diamonds). Upper panel = density (number per square meter); lower panel = biomass (ash-free dry weight of mussel tissue in grams per square metre).

Data Source: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2017
Figure 3. Densities (number per square metre) of zebra (upper panel) and quagga (lower panel) mussels in Lake Michigan from 1994/5-2015.
Data Sources: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2014, 2017
Figure 4. Densities (number per square metre) of zebra (upper panel) and quagga (lower panel) mussels in Lake Huron from 2000-2017.
Data Source: Great Lakes Environmental Research Lab, NOAA; Nalepa et al. 2018; Karatayev et al. in review
Figure 5. Long-term dynamics of zebra mussels (*Dreissena polymorpha*; red lines) and quagga mussels (*D. r. bugensis*; black lines) density (number per square metre) in Lake Erie (means ± SE). Data represent a variety of non-randomly selected sampling stations and differing sampling methods over the years. Note the differences in scale among the panels.
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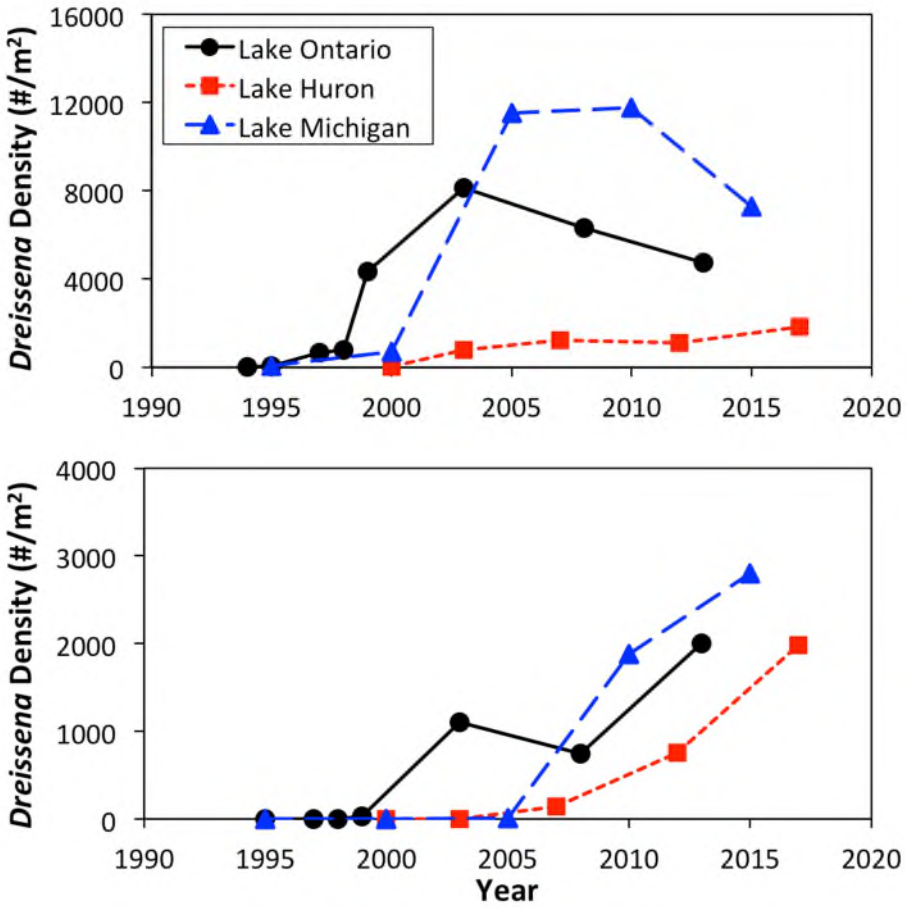


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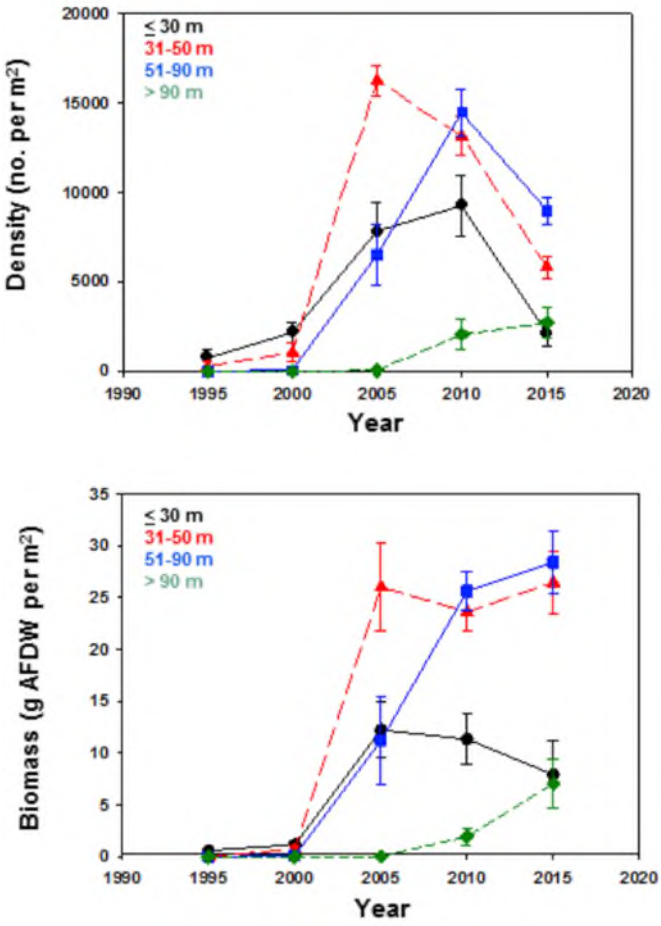


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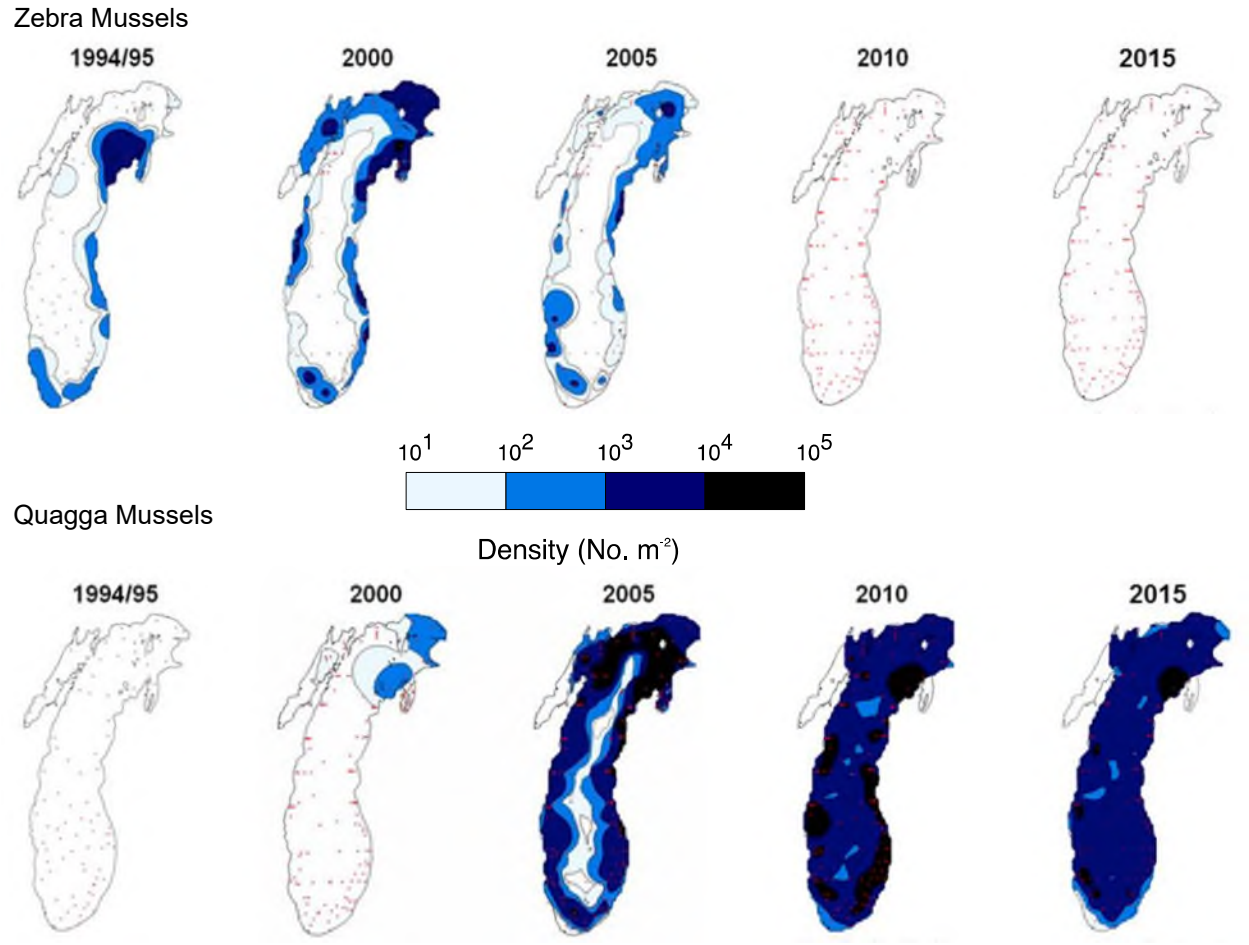


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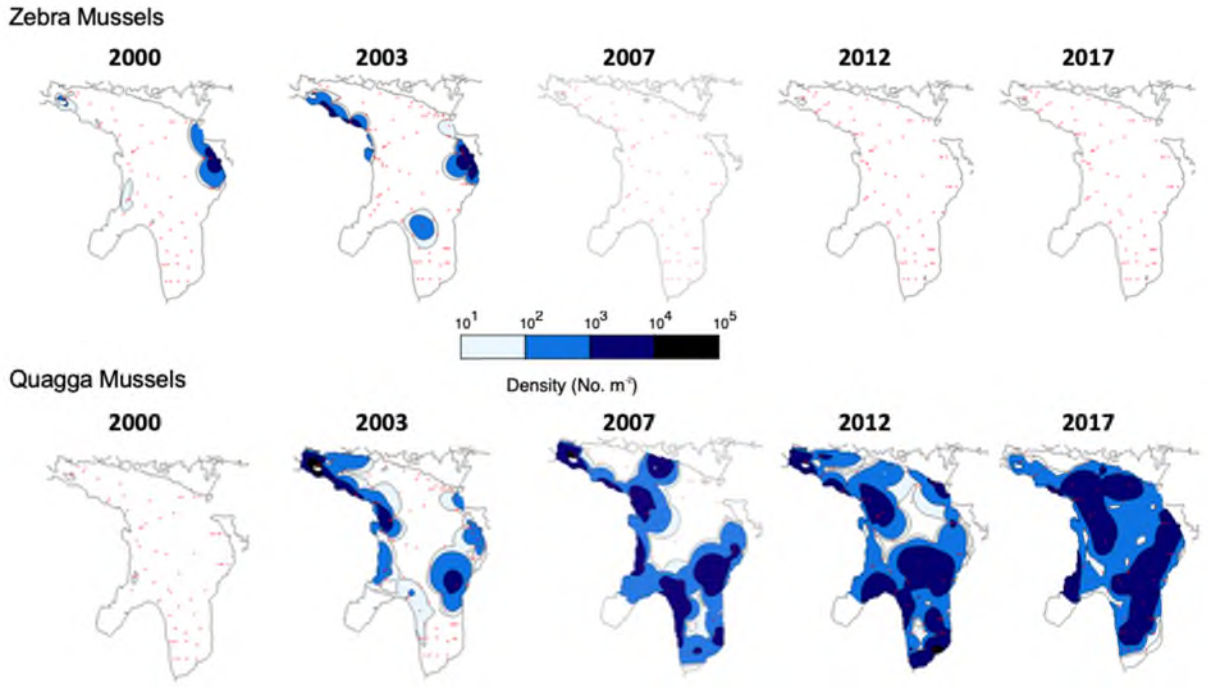


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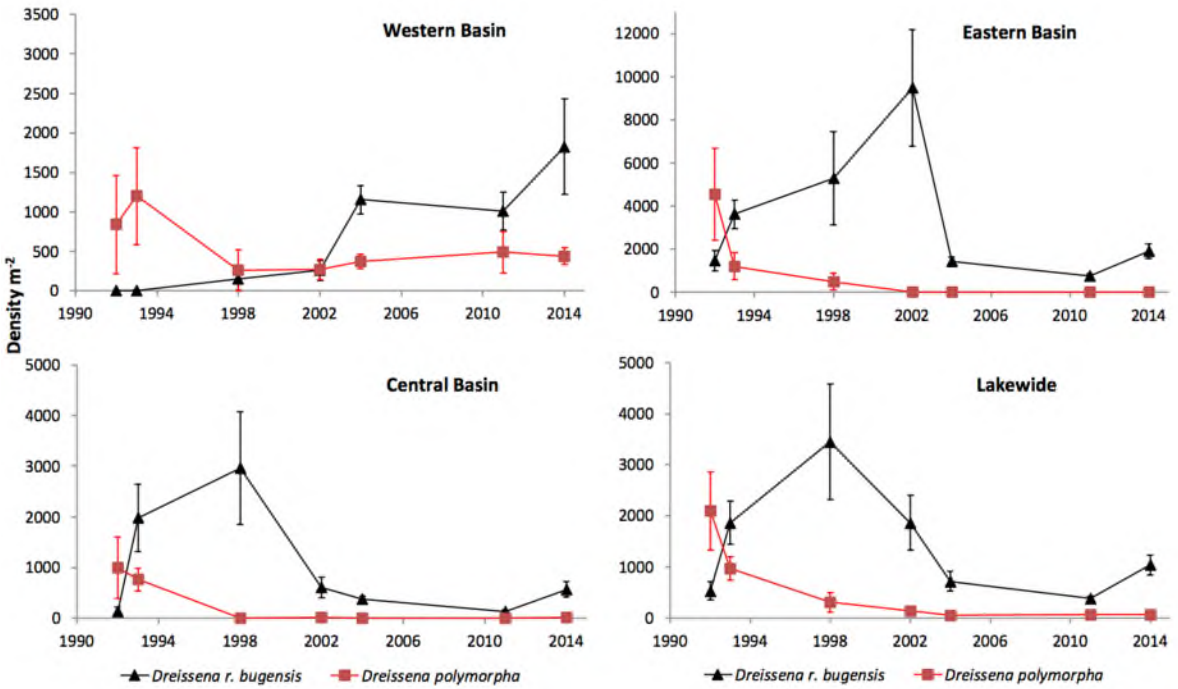


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Data Source: Great Lakes Center, SUNY Buffalo; Burlakova et al. 2017 (and references therein)

Sub-Indicator: Terrestrial Invasive Species

Overall Assessment
Status: Poor
Trends:
10-Year Trend: Deteriorating
Long-term Trend (1948-2015): Deteriorating
Rationale: Based on the five species of interest which have impacts on water quality and/or coastal habitat – Asian longhorned beetle, emerald ash borer, garlic mustard, *Phragmites* and purple loosestrife --- terrestrial invasive species are having significant negative impacts and continue to spread throughout the Great Lakes basin ecosystem. Assessments of spread are based on limited information. Due to the variety of different governmental jurisdictions and the mix of public and private land ownership, it is challenging to assess the introduction and spread of terrestrial invasive species. New internet-based technologies, including the Early Detection and Distribution Mapping System (EDDMapS) (http://www.eddmaps.org/), allow land managers and private citizens to voluntarily share information. EDDMapS provides some limited spatial data that helps track the spread of terrestrial invasive species. A variety of Great Lakes jurisdictions have implemented control programs and restoration efforts in recent years which is expected to reduce the rate of introduction and spread of terrestrial invasive species.
Note: not all species have been monitored over the full period noted.

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: Deteriorating
Long-term Trend (1948-2015): Deteriorating
Rationale: The impact of the five species of interest has been less significant in the Lake Superior basin than in the other Great Lake basins. The limited amount of impact and number of introductions of the five species may be attributed to the fact that the basin has few major population centres, which reduces the potential for anthropogenic movement of terrestrial invasive species, and a colder climate. Nevertheless, the threat of the five terrestrial invasive species remains high and warming temperatures due to climate change may increase the habitat range for invasive species. Garlic mustard and purple loosestrife threaten to further spread their ranges in the basin. There have also been a few sites south of Lake Superior with confirmed emerald ash borer infestations and strict regulation is important to limit its spread. Furthermore, data from the Early Detection and Distribution Mapping System (EDDMapS) suggest that since 2003, there has been an increase in *Phragmites* observations in the lake basin. No infestations of the Asian longhorned beetle have been reported in the Lake Superior basin.

Lake Michigan
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1948-2015): Deteriorating
Rationale: The five species of interest continue to have considerable negative impacts on the Lake Michigan basin ecosystem. It appears that emerald ash borer, garlic mustard and *Phragmites* continue to spread through the Lake Michigan basin. In the Lake Michigan basin, over 6,000 hectares of monotypic *Phragmites* stands were detected by satellite imagery in 2010 (Bourgeau-Chavez et al, 2013). The vast range of *Phragmites* is likely impacting the quality of both wetland and riparian habitat. However, the Asian longhorned beetle was declared eradicated from the Chicago, Illinois area after 10 years of eradication efforts beginning in 1998. The magnitude of purple loosestrife infestations has also been successfully limited by biocontrol programs, but they are not capable of complete eradication.

Lake Huron (including St. Marys River)
Status: Poor
10-Year Trend: Deteriorating

Long-term Trend (1948-2015): Deteriorating
Rationale: Emerald ash borer, garlic mustard and *Phragmites* are having significant negative impacts on the Lake Huron basin. The emerald ash borer exists in numerous locations along the south shore of Lake Huron. Based on volunteered geographic information (VGI) observations from the Early Detection and Distribution Mapping System, garlic mustard and purple loosestrife have been spreading in the basin. Though it is difficult to discern the magnitude of infestations based on VGI data (an observation could represent one plant or hundreds of plants), it provides insight into potential distribution and spread of the two plant invasive species. In the U.S. Lake Huron basin, over 10,000 hectares of dense *Phragmites* stands were detected by radar imagery in 2010. The extensive range of *Phragmites* likely impacts the habitat and populations of wildlife. No infestations of the Asian longhorned beetle have been reported in the Lake Huron basin.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (1948-2015): Deteriorating
Rationale: Garlic mustard continues to have considerable negative impact on the Lake Erie basin ecosystem. Furthermore, the impacts of emerald ash borer on forests in southwestern Ontario have been particularly devastating; from 2004-2012, over 66,000 hectares of forests in the Aylmer and Guelph Ministry of Natural Resource districts have experienced moderate-to-severe defoliation and decline. Quarantine areas exist throughout the Lake Erie basin and education and eradication campaigns have been crucial in slowing the spread of the emerald ash borer. *Phragmites* has also had considerable negative impact on the U.S. Lake Erie basin; more than 8,200 hectares of dense *Phragmites* stands were detected by satellite imagery in 2010. A study by the Canadian Wildlife Service of Environment and Climate Change Canada suggests that *Phragmites* continued to spread in the Canadian Lake Huron-Erie corridor in the areas around the St. Clair River, Lake St. Clair and Detroit River from 2006-2010. Meanwhile, the extent and severity of purple loosestrife infestations has been controlled by two leaf-eating beetles, *Galerucella calmariensis* and *Galerucella pusilla*, which feed exclusively on this wetland perennial. No infestations of the Asian longhorned beetle have been reported in the Lake Erie basin.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Poor
10-Year Trend: Deteriorating
Long-term Trend (first year- last year): Deteriorating
Rationale: Garlic mustard and Phragmites have continued to negatively impact the Lake Ontario basin ecosystem and the extent of these 2 species has spread across this basin. By comparison, purple loosestrife has been effectively controlled by the two leaf-eating beetles (noted above) in the Lake Ontario basin. This perennial plant may continue to spread through the basin, but the beetles can limit the severity of infestations. In the Lake Ontario basin, emerald ash borer was detected in the Niagara Region in 2012. While its impacts have not been as severe as the Lake Erie basin, there are large areas in the region that are experiencing moderate-to-severe decline and mortality in ash trees. The emerald ash borer has the ability to spread quickly and negatively impact forest ecosystems. For the Asian longhorned beetle, two infestation areas exist in the basin; the infestation in the Toronto-Vaughan area was declared eradicated. The other infestation in Toronto-Mississauga is under quarantine since 2013 and the pest will be declared eradicated if there are no detections after 5 years of surveys.

Status Assessment Definitions
Good: The selected 5 terrestrial invasives are having minimal negative impacts on the ecosystem
Fair: The selected 5 terrestrial invasives are having some impacts on the ecosystem
Poor: The selected 5 terrestrial invasives are having considerable negative impacts on the ecosystem
Undetermined: Data are not available or are insufficient to assess condition of the ecosystem components

Trend Assessment Definitions
Improving: No new introductions or spread of the 5 selected species of interest
Unchanging: No significant changes in the introductions of the 5 selected species of interest
Deteriorating: New introductions or spread of the 5 selected species of interest
Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend

Endpoints and/or Targets
The target for this sub-indicator is to lower the impact and geographic extent of 5 terrestrial invasive species in the Great Lakes basin and to reduce the rate of new introductions. It is also important to establish quantifiable target

measures that assess whether the five selected terrestrial invasive species populations are being reduced and prevented from continually encroaching in the Great Lakes through best management practices, control and treatment methodologies and effective policy implementation.

Sub-Indicator Purpose

- The purpose of this sub-indicator is to assess the presence, number, and distribution of five selected terrestrial invasive species (TIS) in the Laurentian Great Lakes watershed, and to understand the means by which these species are introduced and persist.
- It is also to 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 five selected terrestrial invasive species: Asian longhorned beetle, emerald ash borer, garlic mustard, *Phragmites* (Common Reed) and purple loosestrife, in the Great Lakes because they can negatively impact the biodiversity, habitat, chemical loads, nutrient cycling, and hydrogeology of terrestrial and other ecosystems within the Great Lakes watershed.

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 measurement for this sub-indicator will be a lake-by-lake assessment of the status and trends of 5 species with considerable negative impact on the ecosystem – Asian longhorned beetle, emerald ash borer, garlic mustard, *Phragmites*, and purple loosestrife. Government reports will be primarily relied upon to assess status and trends of the selected terrestrial invasive species. Other qualitative sources, such as journal articles, will be used to assess impacts and spread of the selected invasive species on the Great Lakes basin. Some spatial data from EDDMapS will also be used to help support the status and trend assessments for garlic mustard, *Phragmites* and purple loosestrife. Negative impacts of the five selected species on the Great Lakes ecosystem are described later in the document.

Ecological Condition

The proliferation of terrestrial invasive species in the Great Lakes basin has occurred as an unintended consequence of global trade and movement of people. As the movement of goods and people continues to grow, a greater number of species are transported from their native ranges to introduced ranges. Though not all non-native species are a threat to their introduced range, a small number of these non-native species are invasive and have the ability to significantly disturb ecosystems (Canadian Food Inspection Agency, 2005; Great Lakes Regional Collaboration Strategy, 2005). Since the1800s, there has been a dramatic increase in the number of invasive non-native plant species introduced into Canada (Canadian Food Inspection Agency, 2005; Figure 2). This trend is also supported by data analyzed from the World Wildlife Fund's Invasive Species database, which indicates that there has been a 55% increase in the number of terrestrial invasive species introductions in the Great Lakes basin from 1900 to 2000 (World Wildlife Fund 2003).

The status and trends of terrestrial invasive species will be assessed based on the impacts and distribution of the Asian longhorned beetle, emerald ash borer, garlic mustard, *Phragmites* and purple loosestrife. These species were selected because of their significant and widespread impact on the Great Lakes basin. It must be noted that though the species selected would predominantly result in poor and deteriorating assessments, there is an opportunity to realize improvement through limiting their spread and impact.

Asian Longhorned Beetle

The Asian longhorned beetle (Figure 1), discovered in the Toronto area in 2003, (*Anoplophora glabripennis*) kills hardwood trees which include maples, elms, willows and birches, threatening to devastate forests in the Great Lakes basin and beyond. Forests play a key role in stabilizing soil and limiting the amount of sediment-bound pollutants into receiving waters (Turner & Rabalais 2003).These forests also protect water quality as well as the habitats of a number of native species (The Nature Conservancy).

Asian longhorned beetles (ALB) are native to China and Korea and have been discovered in Ontario and Illinois. The preferred hosts of ALB are maple trees, but they have been known to infest and kill other hardwood trees, such as poplars, willows, birches, horse chestnuts and elms. The ALB has no natural predators in North America, therefore they pose a substantial threat to millions of trees. The United States Department of Agriculture (2006) estimates that the ALB could have a potential market economic impact of more than $41 billion in the United States. However, the intangible economic losses associated with ALB, such as ecosystem service and aesthetic losses, are estimated to have a greater impact than market economic losses (Animal and Plant Health Inspection Service [APHIS], 2015). In North America, extensive eradication measures have been introduced to ensure that the number and spread of ALB is limited. Treatment options include strict regulation of quarantine areas and tree removal. Trees are typically burned or sent through a wood chipper which kills larvae, pupae, or adult beetles in the bark (Government of Ontario, 2018). In Ontario, susceptible and infested trees located in the two identified infestation sites were removed; one of the infestations was declared eradicated as no beetles or infested trees were discovered after 5 years of surveys. The other infestation site is being monitored to ensure that further eradication measures are implemented if warranted. Since 2008, the ALB has been eradicated from Illinois. Outside the Great Lakes basin, efforts continue in nearby southern Ohio and New York to limit the spread of the ALB.

Emerald Ash Borer

The emerald ash borer (EAB) (Figure 1), was first discovered in North America in the Detroit-Windsor area in the early 2000s (Canadian Food Inspection Agency, 2014). It is now found throughout the Great Lakes basin). These wood-boring pests are believed to have arrived from East Asia in wood shipping containers. EAB feed on green, red, white, black and blue ash and are responsible for the destruction of millions of ash trees across Ontario and all eight Great Lakes States. The loss of native ash trees, which are an important component of stream-side habitats, can lead to increases in storm water runoff, resulting in an increase in the erosion of soil and sediment into streams. The loss of trees can also cause an increase in water temperature due to increased solar exposure (Natural Resources Canada, 2014). Soil erosion into streams is particularly concerning because pollutants are bound to sediment, altering the stream's chemical properties and increased nutrient loading (Hewlett, North, Johansson, Vandergucht & Hudson 2014). Excessive nutrient loading can also disrupt the aquatic food web (Hewlett et al. 2014).

Treatment, removal, and replacement of emerald ash borer are costly and expensive. Like many of the invasive plants in the Great Lakes, emerald ash borer has caused significant economic losses to Canadian municipalities (National Resources Canada, 2018). In Ontario, Canadian Forest Service scientists estimate that $2 billion over a 30-year horizon will be required to remove and replace trees (Natural Resources Canada [NRCAN], 2015). High mortality rates are typical once an infestation occurs; after 6 years of initial infestation, roughly 99% of ash trees are killed in the woodlot (NRCAN, 2015). In 2001, Toronto had approximately 860,000 ash trees throughout the city. In 2016, approximately 9,500 viable trees remain. The effects of EAB on the Great Lakes ecosystem are wide-ranging, particularly in areas that are dominated by ash trees. It is also expected that urban areas will be affected since ash trees are often planted for their quick-growing nature. These forests also protect water quality as well as the habitats of a number of native species (The Nature Conservancy). The emerald ash borer is significantly impacting the Lake Erie ecosystem (Figure 3); it is estimated that more than 65,000 acres across southern Ontario have experienced moderate-to-severe decline and mortality in ash trees (Figure 4). The impacts of EAB have also been having a severe impact in the Lake Huron basin in the area surrounding Sarnia, Ontario (Figure 3). Areas west of London, Ontario have been particularly affected and there is concern that the emerald ash borer will continue to spread east into the Lake Ontario basin and north into the Lake Superior basin. The Canadian Council of Forest Ministers [CCFM] (2015) predicts that EAB will spread to Thunder Bay, Ontario and into other parts of Northern Ontario due to the lack of biological prevention tools and regulation. However, the rate of spread for EAB in Northern Ontario will be slower than in the south because of cooler climatic conditions (CCFM 2015). Quarantine areas, strict regulation, education programs and removal of ash trees in infested areas are some important measures to limit the spread of emerald ash borer in the Great Lakes basin and beyond.

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 for 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 about the EAB. It gathers data on the invasive EAB from across North America. The network is maintained by USDA Forest Service and Michigan State University. The EAB Information network site provides information on EAB, tracks detections of EAB (Figure 5), and suggests preventative measures to reduce EAB spread (Emerald Ash Borer Information Network, 2018).

Garlic Mustard

Garlic mustard *(Alliaria petiolata*) is considered one of the most invasive non-native species in Canada as it out-competes native plants and disrupts natural underwood growth (The Lake Huron Centre for Coastal Conservation [LHCCC]).

Garlic mustard (Figure 1) was likely introduced to North America from Europe in the late 19th century for culinary or medicinal purposes. The invasive nature of garlic mustard results in altered forest composition since garlic mustard can control the nutrient supply in soil, making it difficult for tree seedlings to germinate (Rodgers, Stinson & Finzi, 2008). 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. Garlic mustard also reduces the amount of native plant pollen, seeds, and fruits available for wildlife (Anderson, 2012). This invasive species is also toxic to the larvae of some butterflies, which results in a reduction of plant pollination (Lake Huron Centre for Coastal Conservation). The habitats for ground-nesting birds are also affected by the litter layer created by garlic mustard on the forest floor. The habitat for salamanders and other forest floor dwelling animals is also negatively affected by garlic mustard (Anderson, 2012). Tracking the distribution of garlic mustard is an important step in eradication efforts as it highlights areas that require management. The Early Detection & Distribution Mapping System (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 that there has been a spread of garlic mustard in the Great Lakes basin as it has now been observed across Ontario and in all eight Great Lakes States (Figure 7). The Greater Toronto Area and southern and western Michigan appear to have a number of garlic mustard observations. Over time, the range of garlic mustard in Ontario has spread to the northern shores of Lake Superior. It is predicted that garlic mustard will continue its spread across North America as it possesses a specific combination of traits, making it a successful competitor in a number of different 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).

Phragmites

A European strain of Common Reed (*Phragmites australis*) (Figure 1) was introduced to North America roughly 200 years ago.

Two varieties of *Phragmites* exist in the Great Lakes basin, the native subspecies (*americanus*) and the invasive subspecies (*australis*). *Phragmites australis* form dense stands in roadside ditches, along the water's edge and in wetlands, decreasing biodiversity by choking out native plant species. In 2005, Agriculture and Agri-food Canada declared that *Phragmites australis* (herein *Phragmites)* was the worst invasive plant species in Canada. Invasive *Phragmites* is responsible for changes in the hydrologic cycle, alterations to the nutrient cycle as well as losses to biodiversity and habitat (Ontario Ministry of Natural Resources [MNR], 2011). Further, *Phragmites* is able to affect the habitat of migrating and breeding birds because birds prefer open-water pools, and more ground cover for nesting (Robichaud and Rooney, 2017). The spread of *Phragmites* occurs quickly as it can grow up to 4 centimetres a day vertically and can establish root systems that measure several metres (MNR, 2011). The rhizomes of its roots release toxins that inhibit the growth of native species, resulting in the formation of a dense monoculture. Invasive *Phragmites* seeds are easily transported by the wind, water or birds and can rapidly colonize disturbed environments. Once it is established in an area, multiple management controls are typically required for eradication because of their large root systems (MNR, 2011). Based on data analyzed from EDDMapS, Lake Ontario and Lake Erie have the most observations of *Phragmites* in the Great Lakes basin. In Ontario, this perennial grass has begun to migrate north to Georgian Bay and Lake Superior. It appears that over a period of 67 years beginning in 1948, the distribution of observations has expanded to multiple locations in Ontario and into five of the eight Great Lakes States (Figure 8). The presence of *Phragmites* across the Great Lakes basin has been supported by research undertaken by a team at the Michigan Tech Research Institute, led by Laura Bourgeau-Chavez. The locations of mature stands of invasive *Phragmites* on the U.S. side of the Great Lakes basin were mapped using satellite imagery (Figure 6). Data collected in 2008-2010 was used to detect invasive *Phragmites* that dominated 90% of 0.2 hectare mapping units. Significant stands were mapped in Lake Huron (10,395 ha), Lake Erie (8,233 ha) and Lake Michigan (6,002 ha), while little to none were mapped in the Lake Ontario and Lake Superior basins (Bourgeau-Chavez 2011). It should be noted that the imagery can only detect large, dense stands of *Phragmites*. It can also be difficult for the researchers to determine whether the imagery depicted *Phragmites* as other monotypic aquatic plants. The overall accuracy of the basin-wide map was 87%, illustrating that imagery is an effective means to detect the presence of large, mono-typic invasive *Phragmites* stands in the Great Lakes basin. Wilcox et al. (2010) investigated the change in plant communities on the northern shores of Lake Erie in Long Point, Ontario and found

that areas of predominantly cattails and marsh were replaced by *Phragmites*. Long Point is noted as an important staging area for waterfowl, which may be negatively impacted by dense monocultures of *Phragmites* (Wilcox et al. 2010).

Purple Loosestrife (Lythrum salicarica)
Purple loosestrife (Figure 1) is a wetland plant native to Europe and Asia that was brought to North America in the early 19th century. The invasion of purple loosestrife and *Phragmites* in the Great Lakes alters the structure and function of wetland ecosystems (Keil & Hickman 2015). Healthy wetlands provide ecosystem services that are significant to the Great Lakes basin including filtering nutrients that stimulate algal growth and limiting eutrophication in lakes and tributaries (Zedler & Kercher 2005). Furthermore, wetlands provide fish spawning habitat and help store large quantities of carbon in their soils (Zedler & Kercher 2005). Purple loosestrife has the capability of hybridizing with native loosestrife species and thereafter depleting the native species gene pool (Centre for Invasive, n.d). Purple loosestrife spreads extensively to wetlands and disturbed environments due to its small, easily-transported seeds. Purple loosestrife weaves thick mats of roots that cover vast areas, impacting the quality of habitat for birds, insects and other plants (Government of Ontario, 2012). Furthermore, purple loosestrife threatens wetland ecosystems by altering water levels and reducing food sources for both aquatic and terrestrial native species (Thompson, Stuckey & Thompson, 1987). The invasive plant has spread across the Great Lakes-St. Lawrence River basin (Warne, 2016).

According to data collected by EDDMapS, purple loosestrife is present across Ontario and in all eight Great Lakes States. It appears that beginning in 1900, purple loosestrife has expanded its range over a 115-year period as it is now ubiquitous along the shorelines of all five Great Lakes (Figure 9). However, there is an effective control measure to combat the spread of purple loosestrife, which has been the use of their natural predators, *Galerucella calmariensis* and *G. pusilla* beetles. Multiple studies were carried out to ensure that the use of *Galerucella calmariensis* and *G. pusilla* would not impact native species (Michigan Sea Grant). It was determined that these particular varieties of beetle only target purple loosestrife, making it a viable biocontrol option. They have been used at multiple sites in the Great Lakes basin and can significantly reduce purple loosestrife populations (Government of Ontario, 2012). In 2006, the Ontario Federation of Anglers and Hunters reported that this perennial invasive has been successfully controlled by *Neogalerucella spp*. beetles (formerly of the genus Galerucella) in more than 80% of the 300 control sites located in Ontario. It must be noted that the beetles can only reduce purple loosestrife to manageable populations and are not capable of complete eradication. Other means such as pulling and mowing, burning, flooding, and chemicals have also been used to mitigate the non-native plant (Warne, 2016).

Linkages
Climate change may expand the current habitat ranges of terrestrial invasive species as temperatures warm and growing seasons become longer (Clements & DiTommaso, 2012). Based on spatial data from EDDMapS, it appears that invasive *Phragmites* has begun to move north into the Lake Superior basin, perhaps as a result of warmer temperatures. Smith et al. (2012) have described the need to better study the impacts of climate change on terrestrial invasive species and stressed the need to bridge the gap between policy and science.

Warmer temperatures can lead to emerald ash borer expanding its range. Emerald ash borer has a lower survival rate in extreme cold temperatures. With an extended lifespan, the invasive species can eat its way through more ash trees because it does not have to face extreme cold temperatures (Cuddington et al.). Purple loosestrife is also believed to be able to expand under warmer temperatures as well as the plant is believed to be very adaptive under a variety of weather conditions (Colautti, 2013). Asian longhorned beetle can also experience a longer lifespan under warmer temperatures, which will increase their impact on forests (Lindsay, 2016).

The invasion patterns of garlic mustard may change under different climate conditions. This is because garlic mustard thrives better under cooler climates. Thereby, it's predicted that while garlic mustard may decline in areas that are warmer, there may be spread in areas that are relatively cooler (Potter, 2017). In addition, *Phragmites* is able to readily acclimate to rising temperatures, specifically areas where warmer weather has caused the freshwater wetland to become wetter (Lavoie, 2015).

- Forest Cover – Forest cover can be negatively impacted by the Asian longhorned beetle and the emerald ash borer as they kill trees which can result in increases in the amount of runoff and sediment-bound pollutants into the Great Lakes and its tributaries (Turner & Rabalais 2003). Forests also play a key role in carbon sequestration by absorbing and removing greenhouse gas emissions (Natural Resources Canada 2015).

- Coastal Wetland sub-indicators (Coastal Wetland Invertebrates, Coastal Wetland Fish, Coastal Wetland Plants, Coastal Wetland Amphibians, Coastal Wetland Birds, Coastal Wetlands: Extent and Composition) –The invasion of purple loosestrife and *Phragmites* in the Great Lakes can alter the structure and function of coastal wetland ecosystems (Keil & Hickman 2015). Wetlands provide ecosystem services that are significant to the Great Lakes basin including filtering nutrients that stimulate algal growth and limiting eutrophication in lakes and tributaries (Zedler & Kercher 2005). Furthermore, wetlands are unique habitats for plants and animals and help store large quantities of carbon in their soils (Zedler & Kercher 2005).

- Water Levels – *Phragmites* has a rapid growth rate, which can cause lower water levels because water is transpired faster than it would be with native vegetation. Also, *Phragmites* has a slow decomposition rate. This means much of their plant material is left behind in the wetland across the seasons which can lead to higher rates of sediment deposition. The combination of lower water levels and increased sediment produces drier wetlands. Drier wetlands struggle to reduce erosion, filter contaminants, and absorb excess storm water, and also affect the livelihood of fish and wildlife habitat (Ontario Phragmites, 2018).

- Nutrients in Lakes – Research suggests that nutrient enrichment in wetlands can help *Phragmites* have a competitive advantage over native species, in particular, *Melaleuca ericifolia*. Phragmites has rhizosphere chemistry in nutrient enriched fields which enhances its capability to invade the ecosystem. High concentrations of nitrogen, phosphorous, dissolved organic carbon, and phenolic contents have been found in *Phragmites* invaded areas verses non-invaded areas (Volesky et al. 2018).

This indicator also links directly to the other sub-indicators in the Invasive Species category.

Citizen Science and Other Bodies of Knowledge

This report is supported by information collected from Early Detection and Distribution Mapping System (EDDMaps). EDDMapS combines data from other databases and organizations as well as volunteer observations to create a national network of invasive species distribution data that is shared with educators, land managers, conservation biologists, and beyond. As of October 2018, EDDMapS has over 4.5 million records.

EDDMaps collects data volunteered geographic information from citizens who report sightings of terrestrial invasive species. Citizens are encouraged to continue to report their sightings so there are more data available to help support the monitoring of the introduction and spread 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 Limitations

It is difficult to quantify the impacts of the selected terrestrial invasive species due to the number of sources used to assess status and trends. Furthermore, the sources used may not refer specifically to the condition of the species in a particular lake basin, but rather to the localized impacts of a species in a lake basin. Consequently, the assessments are based on the cumulative impacts of specific areas or regions within the lake basin, and may not perfectly represent the conditions of the basin as a whole.

The timeline of terrestrial invasive species introductions is contingent upon the discovery of and identification of said species. Therefore there is an inherent time-lag between the physical arrival and establishment of new species and the discovery, identification, and incorporation of new species into data sets. Furthermore, the identification of terrestrial invasive species is often related to first seeing the negative impact of an invader after it has been introduced to the area and the amount of financial resources expended to investigate, monitor and research. Some species are also inherently more identifiable than others.

Tracking the abundance of all TIS over time would be an ideal indicator, but the data requirements to do this would be unrealistic. Using surrogates, such as Asian longhorned beetle, emerald ash borer, *Phragmites*, purple loosestrife and garlic mustard, is a good approach, but the surrogates used must be representative. The species that were selected, and the approach to track and report on this information, is based on the species having the greatest impact on ecosystem health in the Great Lakes region based on academic literature and government publications. Ontario's Invasive Species Awareness Program, the Canadian Food Inspection Agency, Nature Conservancy of Canada and the State of Ontario's Biodiversity 2010 report were good resources for the five species selected.

EDDMapS is a web-based mapping system for documenting invasive species distribution. Launched in 2005 by the Center for Invasive Species and Ecosystem Health at the University of Georgia, it was originally designed as a tool for state Exotic Pest Plant Councils to develop more complete distribution data of invasive species. EDDMapS Ontario (http://www.eddmaps.org/ontario/), which collaborates with the U.S. Forest Service, National Park Service, U.S. Fish and Wildlife Service, U.S. Army Corps of Engineers and U.S. Department of Agriculture, possesses a repository of spatial data for 3/5 selected invasive species that will be used to analyze changes in their spatial distribution. The data is validated by one of their partner organizations to ensure its accuracy. Because the data is volunteered geographic information, the maps may not depict a complete portrait of the spatial distribution of terrestrial invasive species in the Great Lakes basin.

There is a possibility of a disparity in observations in terrestrial invasive species across the region since 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 data only reflects observations that were made and does not reflect treatment options that have been applied; for instance, a stand of *Phragmites* may have been eradicated after the observation was submitted to EDDMapS. An observation may also represent one plant or hundreds of plants. EDDMapS does however provide some spatial data that helps ecosystem managers to track the spread of terrestrial invasive species and to identify areas that require greater intervention. Reporting on species based solely on data availability is not ideal, but is necessary as spatial and time trend data on TIS are limited in the basin. While the current assessment and trend for this sub-indicator is Poor and Deteriorating in 2017 and 2019 because of how the species of interest were selected, i.e. based on data availability and species that were already having negative impacts, there is however the opportunity to then see improvements in conditions and trends over time. If management and mitigation measures are able to reduce the impacts, distribution and spread of these species going forward, these efforts will be seen through future updates to this sub-indicator report. Future iterations of this report will aim to include an expanded list of species.

It is also important to note that agencies have increased their level of public education and outreach for the time period shown in the species-specific figures and thus the public is far better informed about invasive species. The development of tools such as EDDMapS and associated applications has made it much easier for the public to report observations of invasives. While there is evidence that garlic mustard and others is increasing on the landscape, the frequency of reports and the distribution of the reports may have considerably outpaced the actual spread of the species. EDDMapS is likely the best information available and it is a great tool, but the limitations should be carefully considered and explained so that the information is not misrepresented – especially for tracking spread and trends.

Additional Information

This report was supported by data from both EDDMapS and qualitative information from government reports, non-governmental agencies and journal articles.

Terrestrial invasive species have the ability to alter the species composition of the Great Lakes basin ecosystems, resulting in altered watershed dynamics and decreased biodiversity (Canadian Food Inspection Agency, 2005; Great Lakes Regional Collaboration Strategy, 2005). This sub-indicator includes a timeline of introductions of non-native plant species to Canada from 1600-2000 (Figure 2). This sub-indicator will also be supported by data from the

World Wildlife Fund's invasive species database, which documents invasive species by taxonomic group from 1600-2000.

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. As a consequence of high population density and high-volume transportation of goods, the Great Lakes basin receives a high number of invasive species. Every time a new species is brought into the basin there is a chance that it will escape to local ecosystems where it may become invasive.

Although only a relatively few non-native species become established as invasive, the introduction of invasive species is recognized as one of the greatest threats to the biodiversity and natural resources of this region, second only to habitat destruction (Great Lakes Regional Collaboration Strategy 2005; Canadian Food Inspection Agency 2005; Dextrase and Mandrak 2006).

As noted above, there are a number of Canadian and U.S. agencies that possess TIS-related data. The Great Lakes *Phragmites* Collaborative (GLPC), for example, is a regional partnership established to improve communication and collaboration and lead to more coordinated, efficient and strategic approaches to non-native *Phragmites* management, restoration and research across the Great Lakes basin in both the United States and Canada.

In Canada, the Ontario Invasive Species Awareness Program, the Food Inspection Agency, and the Nature Conservancy of Canada provide data on terrestrial invasive species. In the U.S., the USDA Forest Service, US Geological Service, US Fish and Wildlife Service, state agencies such as Michigan Natural Features Inventory as well as citizen driven data are important sources of information. A data source for basin-specific terrestrial invasive species is the organization of locally led Cooperative Weed Management Areas (CWMA) in the U.S. Great Lakes basin.

It is difficult to fully appreciate the status and spread of terrestrial invasive species in the Great Lakes region 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.

Other information could be considered for inclusion in future sub-indicator reports, including:

- TIS Hotspots – plot location of first occurrence for each TIS spatially and determine hotspots that need focused attention.
- Look at well-studied areas that are revisited every so often (e.g. forest monitoring plots at a small scale or parks with monitoring programs if a broader geography is desired).
- Look at 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.
- Include remote sensing data for reporting on the status and trends of *Phragmites*.

Acknowledgments
Author: Lindsay Wong, Environment and Climate Change Canada
Contributors: Jenique Kennedy, Environment and Climate Change Canada
Stacey Cherwaty-Pergentile, Environment and Climate Change Canada
Julia Koh Hatcher Ayyoub, Environment and Climate Change Canada (map support)

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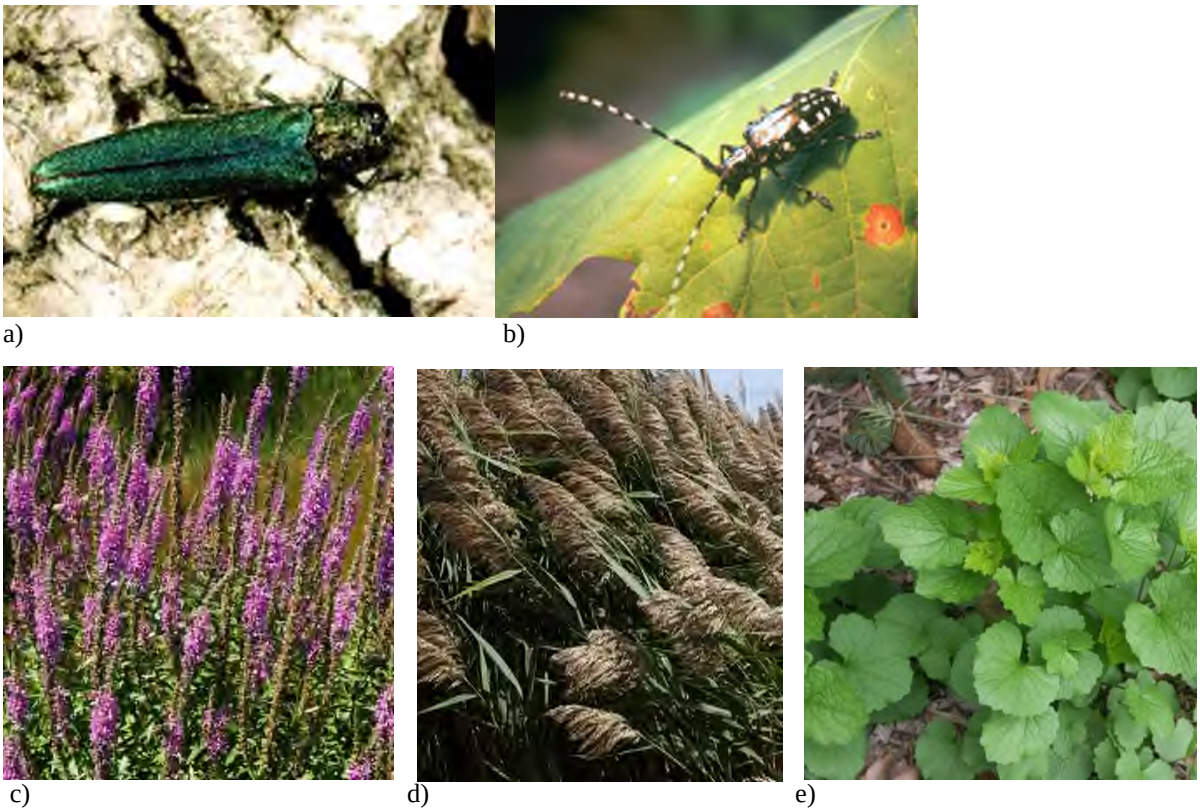
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List of Figures

Figure 1. Photos of terrestrial invasive species include in this report, a) Emerald ash borer b) Asian longhorned beetle, c) Purple loosestrife, d) *Phragmites*, e) Garlic mustard.
Sources: a) Emerald Ash Borer Information Network (2018), b) Natural Resources Canada (2018), c) Great Lakes Sierra Club blog, d) Getty Images e) Great Lakes Gazette.
Figure 2. Cumulative number of invasive non-native plant species introduced into Canada from 1600 to 2005 – Estimated.
Source: Canadian Food Inspection Agency
Figure 3. Areas where emerald ash borer has caused moderate-to-severe decline and mortality to ash species.
Source: Ontario Ministry of Natural Resources
Figure 4. Cumulative amount of area where emerald ash borer has caused moderate-to-severe decline and mortality to ash species.
Source: Ontario Ministry of Natural Resources
Figure 5: Initial county emerald ash borer detections in North America (October 2018)
Source: United States Department of Agriculture
Figure 6. Potential distribution of invasive *Phragmites* in the U.S. Great Lakes basin, 2008-2010.
Source: Bourgeau-Chavez et al.
Figure 7. Garlic mustard observations in the Great Lakes basin (2002-2015).
Source: EDDMapS
Figure 8. *Phragmites* observations in the Great Lakes basin (1948-2015).
Source: EDDMapS
Figure 9. Purple loosestrife observations in the Great Lakes basin (1900-2015).
Source: EDDMaps

Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)



a) b) c) d) e)

Figure 1. Photos of terrestrial invasive species include in this report. a) Emerald ash borer b) Asian longhorned beetle c) Purple loosestrife d) *Phragmites* e) Garlic mustard.
Sources: a) US Department of Agriculture, b) US Department of Agriculture, c) Michigan Sea Grant, d) Michigan Sea Grant e) Great Lakes Gazette

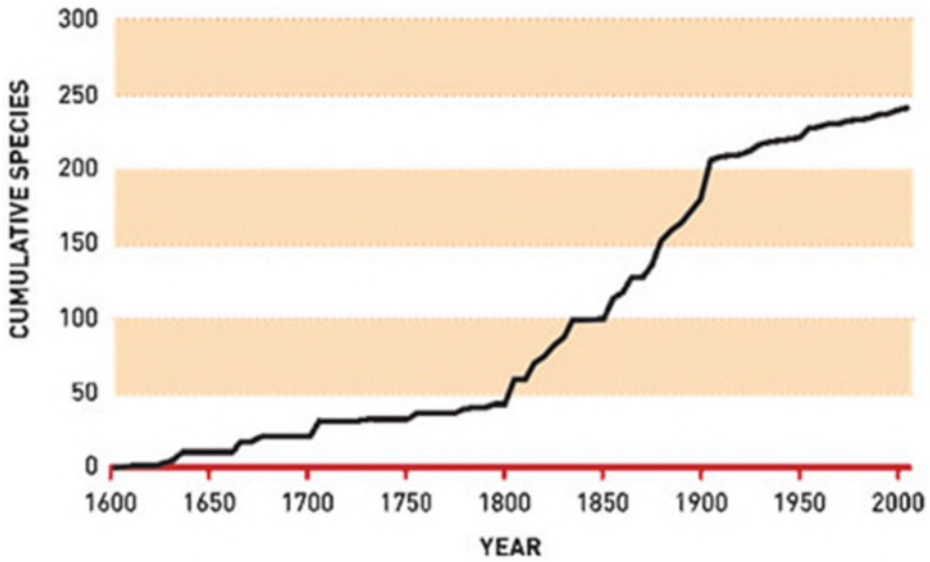


Figure 2. Cumulative number of invasive non-native plant species introduced into Canada from 1600 to 2005 – Estimated.
Source: Canadian Food Inspection Agency

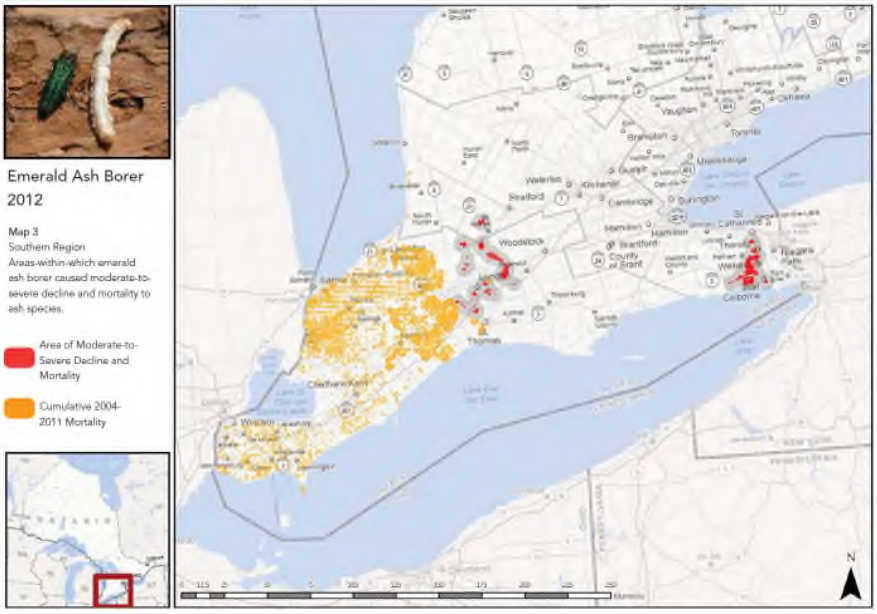


Figure 3. Areas where emerald ash borer has caused moderate-to-severe decline and mortality to ash species.
Source: Ontario Ministry of Natural Resources

Area Summary

Total area-within-which emerald ash borer caused moderate-to-severe defoliation and decline 2004 - 2012 by MNR District (area in hectares)

District	2004-2007	2009	2010	2011	2012	Total
Southern						
Aylmer	7,122	11,734	19,436	24,324	1,410	64,026
Guelph	0	0	0	177	2,812	2,989
Kemptville	0	0	0	628	328	956
Provincial Total	7,122	11,734	19,436	25,129	4,550	67,971

**Area calculations for 2009 include damage that occurred in 2008*

Cumulative area-within-which emerald ash borer caused moderate-to-severe decline and mortality 2004-2012 (area in hectares)



Figure 4. Cumulative amount of area in specific regions of the province of Ontario where emerald ash borer has caused moderate-to-severe decline and mortality to ash species.
Source: Ontario Ministry of Natural Resources

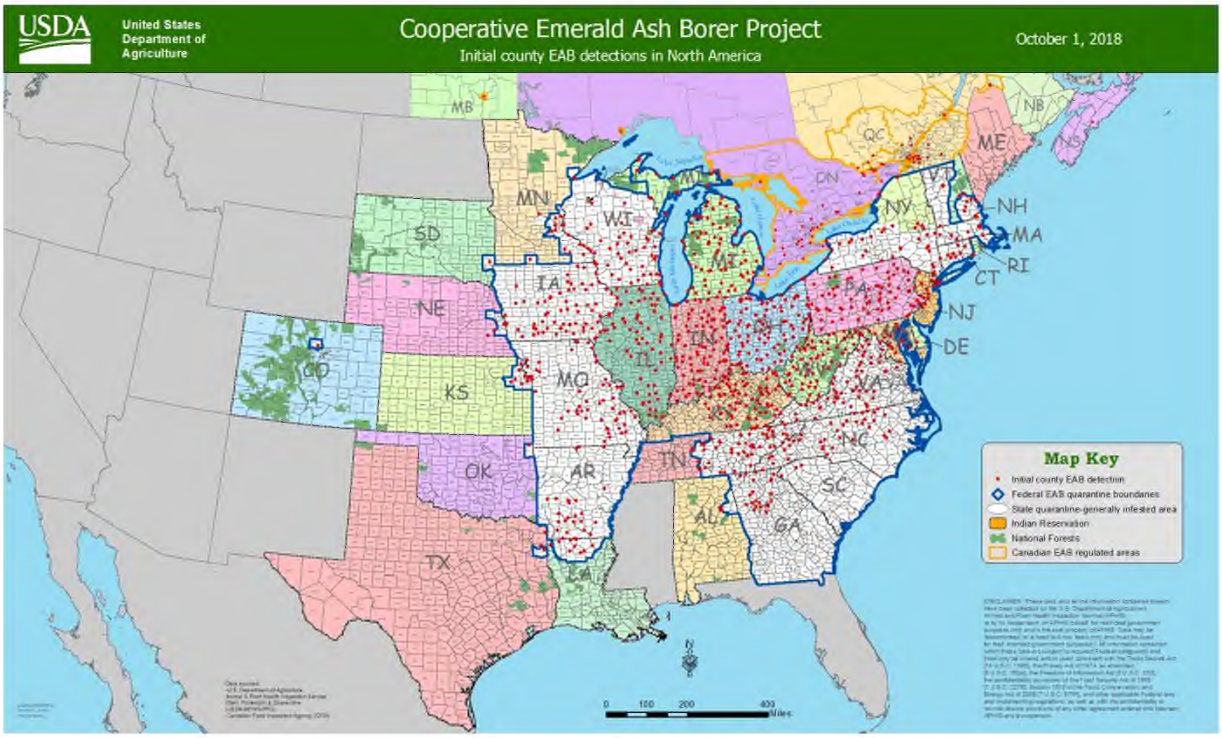


Figure 5. Initial county emerald ash borer detections in North America (October 2018)
Source: United States Department of Agriculture



Figure 6. Potential distribution of invasive *Phragmites* (may include dense, mono-typic stands of other wetland plants) in the U.S. Great Lakes basin using remotely sensed data, 2008-2010.
Source: Bourgeau-Chavez et al.

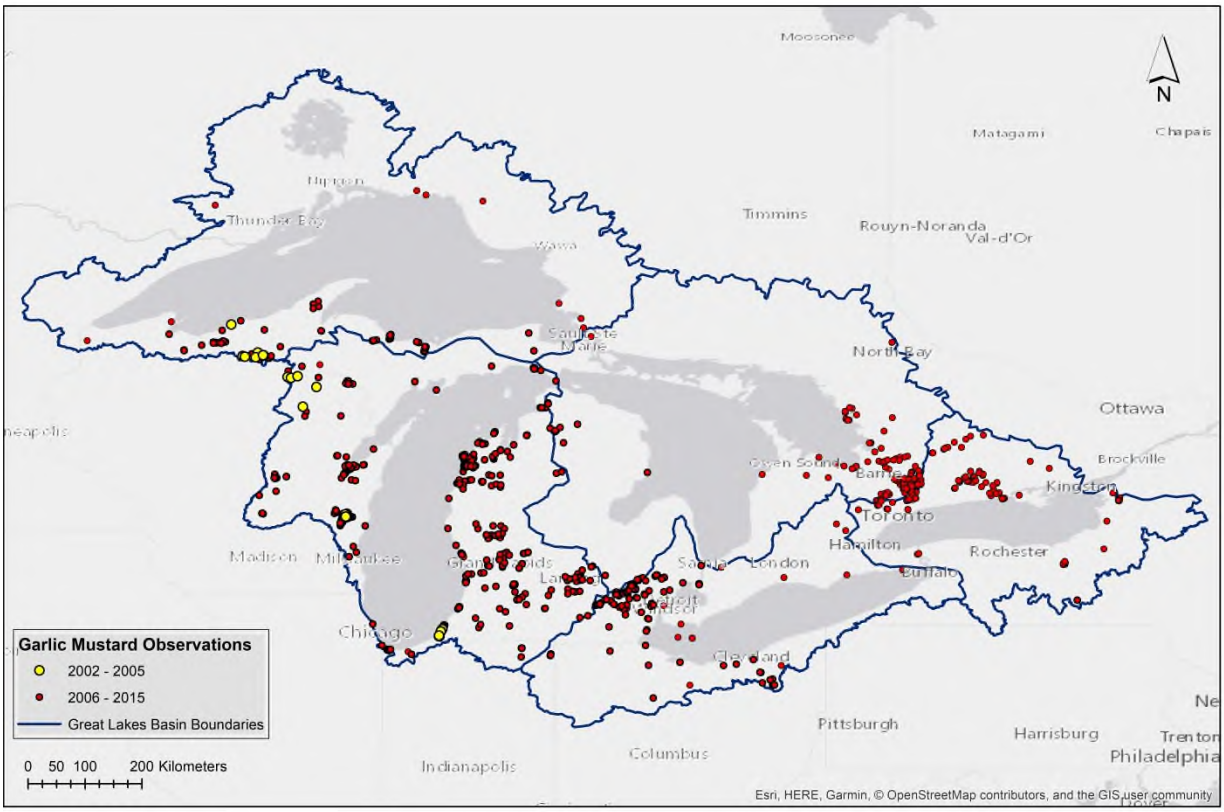


Figure 7. Garlic mustard observations in the Great Lakes basin (2002-2015).
Source: EDDMapS

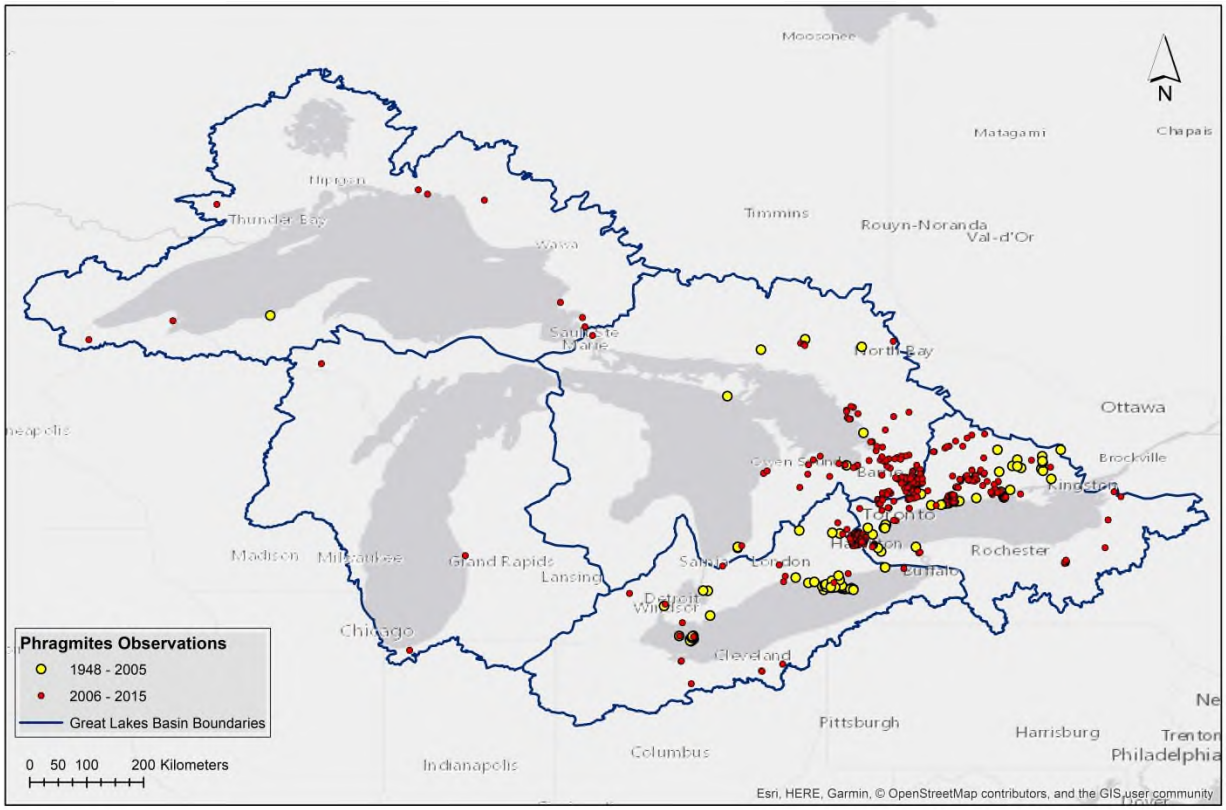


Figure 8. *Phragmites* observations in the Great Lakes basin (1948-2015).
Source: EDDMapS

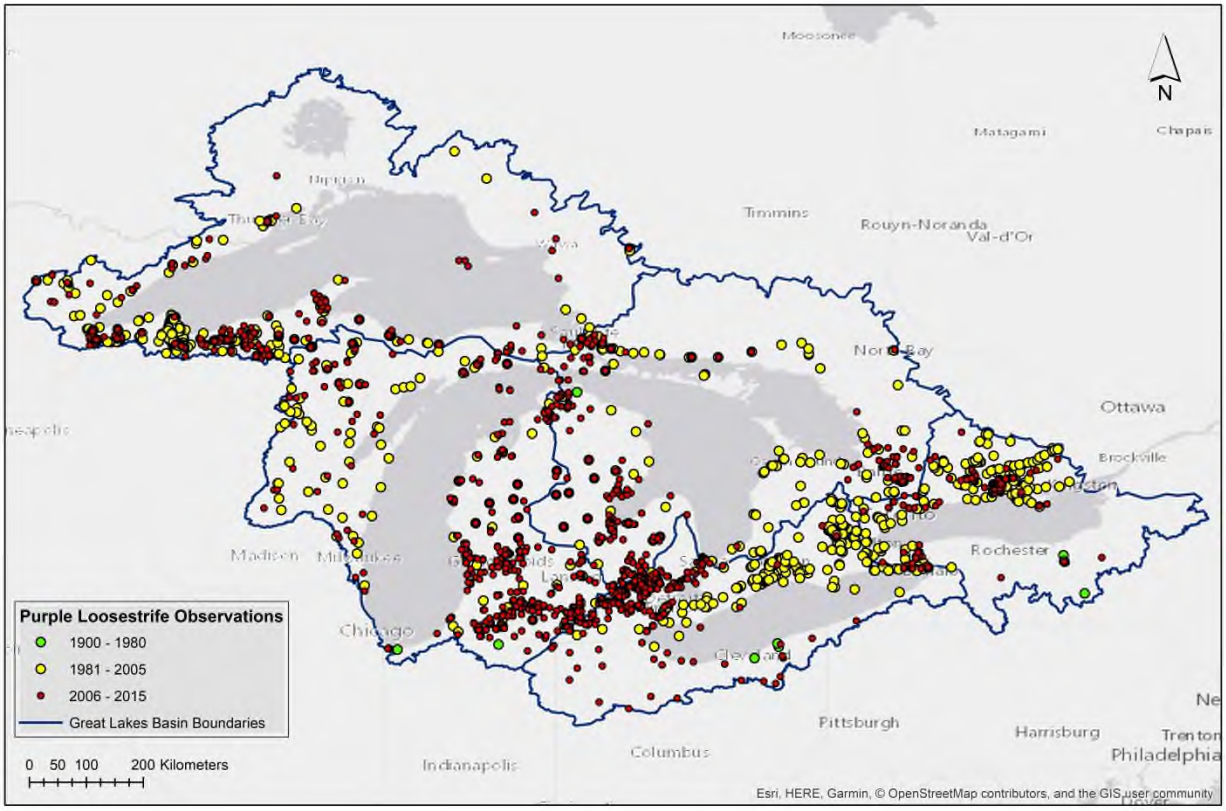


Figure 9. Purple loosestrife observations in the Great Lakes basin (1900-2015).
Source: EDDMapS

Sub-Indicator: Groundwater Quality

Regional-scale Assessment of Nitrate and Chloride Concentrations

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: The overall status of groundwater quality, based on current knowledge and data in the Great Lakes basin, is assessed as "Fair". Of the 670 monitoring wells in the basin that were included, the groundwater quality was assessed to be "Poor" in 203 (30%), "Fair" in 173 (26%), and "Good" in 294 (44%). Given that no single category captured more than 50% of wells, the overall assessment is "Fair" (following the criteria as explained on page 3). Caution must be used when interpreting and applying this overall assessment, due to the large spatial gaps in the data, particularly in the northern and central portions of the basin where hundreds of kilometres/miles exist between wells in the Lake Superior basin, for example. In large areas of the basin where groundwater quality data are missing (Figure 1), the status of groundwater quality should be considered "Undetermined". The overall trend in groundwater quality in the basin is "Undetermined" for two reasons: (1) a lack of available long-term sample analysis for many of the monitoring locations; (2) a statistical analysis of the available data has not yet been completed. However, as noted in this report, trends of increasing (or upward trends in) chloride and nitrate concentrations in groundwater have been reported previously for various watersheds within the basin.

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

Lake-by-Lake Assessment
Lake Superior Basin
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Data (22 wells) are insufficient for assessing overall groundwater quality in the Lake Superior basin (Figure 2).

Lake Michigan Basin
Status: Fair
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: The status assessment as "Fair" should only be considered valid within the western portion of the Lake Michigan basin, given that this is where almost all of the available data were collected (Figure 3). Of the 136 wells that were assessed, the groundwater quality was "Poor" in 64 (47%), "Fair" in 29 (21%), and "Good" in 43 (32%). On the eastern side of Lake Michigan (Figure 3), the status should be considered as "Undetermined". Trend analysis was not part of the assessment completed in 2016-17, but is anticipated to be a component of future assessments.

Lake Huron Basin
Status: Good
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: The status assessment as "Good" should only be considered valid within the southeast portion of the Lake Huron basin (Ontario) for which data were available (Figure 4). Of the 77 wells that were assessed, the groundwater quality was "Poor" in 14 (18%), "Fair" in 16 (21%), and "Good" in 47 (61%). Where data gaps exist (Figure 4), the status should be considered as undetermined. Trend analysis was not part of the assessment completed in 2016-17, but is anticipated to be a component of future assessments.

Lake Erie Basin (including St. Clair-Detroit River Ecosystem)
Status: Fair
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: The status assessment as "Fair" should only be considered valid within the areas of the basin for which data were available (Figure 5). Of the 177 wells that were assessed, the groundwater quality was "Poor" in 50 (28%), "Fair" in 49 (28%), and "Good" in 78 (44%). Where data gaps exist (Figure 5), the status should be considered as undetermined. Trend analysis was not part of the assessment completed in 2016-17, but is anticipated to be a component of future assessments.

Lake Ontario Basin
Status: Fair
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Of the 258 wells that were assessed, the groundwater quality was "Poor" in 74 (29%), "Fair" in 78 (30%), and "Good" in 106 (41%) (Figure 6). Trend analysis was not part of the assessment completed in 2016-17, but is anticipated to be a component of future assessments.

Status Assessment Definitions
The following assessment criteria will initially be applied to each monitoring location (usually a single well), and then extrapolated to each Great Lake drainage basin (watershed).

Step 1
For each monitoring location/well, the groundwater quality is assessed, on the basis of concentrations of chloride (Cl^-) and nitrate (N/L NO_3^-) as being:

> *Good*: less than or equal to (≤) 0.8 mg N/L NO_3^- AND ≤ 30 mg/L (Cl^-)
> *Fair*: greater than (>) 0.8 BUT less than (<) 3 mg N/L NO_3^- 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^-)

In this approach, the distinction between "Fair" and "Poor" is based on the above water quality guideline criteria (120 mg/L for chloride; 3 mg N/L for nitrate) and 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). These "25% of guideline" criteria provide an interim, protective approach for this sub-indicator assessment, based on judgement rather than directly on previously established criteria. They may be modified in future, if sufficient support for alternative criteria becomes available.

Step 2
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.

Existing monitoring wells are usually located to address local groundwater issues (quantity or quality), and a regional context is often not considered. However, for this sub-indicator, the objective is to provide a regional assessment of groundwater quality based on the selected parameters of interest. Thus, preference will be given to selection of monitoring well networks (and potentially also individual wells) that have been placed to provide information about regional-scale ambient groundwater conditions. Statistical methods along with geographic information system (GIS) approaches could be used 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.

It will be necessary to exercise professional judgement in determining whether there is sufficient information to assess individual lake drainage basin, or only portions of each, and to assess the entire Great Lakes Basin. Where sufficient data are available (individual lake basin or entire Great Lakes basin) to provide a groundwater assessment, the following criteria will be applied:

The overall groundwater quality in the basin is assessed as follows:

- Good: If the percentage of wells assessed as "Good" is more than 50 (>50), THEN the basin is assessed as "Good".

- Poor: If the percentage of wells assessed as “Poor” is more than 50 (>50), THEN the basin is assessed as “Poor”.
- Fair: If the percentage of wells assessed as “Fair” is more than 50 (>50), then the basin is assessed as “Fair”
- If the basin is not assessed as “Good”, “Fair”, or “Poor” by the above definitions, THEN by default the basin is assessed as “Fair”.
- Undetermined: Data are not available or are insufficient to assess conditions of the ecosystem.

Tables and ternary diagrams will be used to plot the overall percentages of wells in each assessment category for each lake drainage basin, to show how the groundwater quality sub-indicator compares from lake basin to lake basin.

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
Only wells that are included in ongoing formal monitoring programs will be used for the trend assessments. The following criteria will be applied:

Improving: Concentrations of nitrate and chloride are decreasing (or at least one is decreasing if the other constituent is unchanging).
Unchanging: Concentrations of nitrate and chloride are not changing.
Deteriorating: Concentrations of nitrate and chloride are increasing (or at least one is increasing if the other constituent is unchanging).
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. This will be based on 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 (not in 2019). For 2019, general trends in peer-reviewed articles and journals are referenced where appropriate at a national or Great Lakes scale.

In February 2018, in collaboration with the Ontario Ministry of the Environment, Conservation and Parks (MECP), Environment and Climate Change Canada (ECCC) initiated a project to support future development of the required trend assessments for this sub-indicator. Specifically, ECCC contracted Kilgour & Associates Ltd. (Ottawa, Ontario) 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.

Endpoints and/or Targets
Various standards and guidelines have been established by jurisdictions that have regulatory responsibilities in the Great Lakes basin (GLB) (Table 1).

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 protection of water quality for aquatic life. Under the Great Lakes Water Quality Agreement, the sub-indicator and indicators are used to report on water quality and aquatic ecosystem health and as such targets for the protection of aquatic life will be used for assessment purposes. Therefore, 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 Great Lakes Water Quality Agreement (GLWQA), which states that the waters of the Great Lakes should be "free from the harmful impact of contaminated groundwater."

Measure
This sub-indicator will utilize available measurements of the dissolved concentrations of two relevant (see below) water-quality constituents in groundwater in the GLB: nitrate and chloride. The data will be 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; and the Ontario Ministry of the Environment, Conservation and Parks (MECP), Ontario Geological Survey, in coordination with Conservation Ontario (various Conservation Authorities), as well as various municipalities.

It is important to note that for this sub-indicator, the available data are derived from existing monitoring well networks, which were established in earlier programs, using various criteria; these networks were not established for the purposes of this sub-indicator. In the 2019 status assessment, as for the previous (2016-17) assessment, existing data from USGS and MECP's monitoring programs (using this report's selection criteria) for the period 2000-2015 were included (either dedicated monitoring wells or other private or public water supply wells in the GLB). For wells that have multiple data for this period (2000-2015), the average concentrations were used.

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 and as resources allow for quality control and quality assurance to be completed for the 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 (by providing data in areas where they are currently missing).

Monitoring data collected from known groundwater contamination investigations (for example, studies related to plume investigations) will be 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 a lake and Great Lakes scale. The work and resources to include an assessment of all known contaminated groundwater wells/sites in the Great Lakes basin would be extremely significant. Work under the Groundwater Annex 8 of the GLWQA will consider this type of work in future reporting.

The definition of "shallow groundwater" is arbitrary; for the purposes of this sub-indicator, shallow groundwater is defined as groundwater that is 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. 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.

The recent surveillance of groundwater quality in southern Ontario by the Ontario Geological Survey was completed in 2014 (http://www.mndm.gov.on.ca/en/mines-and-minerals/applications/ogsearth/ambient-groundwater-geochemistry). Because this surveillance is not part of an ongoing monitoring program and wells monitored as part of this study are not intended to be sampled for this purpose again, the data from that project have not been included in the groundwater quality assessment report for 2019.

Ecological Condition

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. Nitrate is mainly from agricultural practices and chloride is mainly an urban contaminant as a result of de-icing road salt.

Elevated concentrations of nitrate in water have been shown to have detrimental effects on aquatic organisms and aquatic ecosystems (e.g., direct toxicity and increasing the 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;
- As anions, they are both extremely mobile (soluble) in water, including the subsurface environment;
- They are stable contaminants that do not have much physical or chemical interaction with the material they flow through;
- Chloride in particular, is persistent – chloride is not subject to attenuation in the subsurface by processes such as biodegradation, sorption or precipitation, and therefore may have an adverse effect on the water quality of streams, rivers and lakes in the GLB;
- Although nitrate is potentially reduced or eliminated by denitrification in some subsurface environments, nitrate also may have an adverse effect on surface-water quality in the GLB; and
- Even though some natural sources of these compounds exist in the environment (e.g., geological), nitrate and chloride are considered as general indicators of anthropogenic impact to aquatic systems.

As noted in a recent report on "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)".

This sub-indicator regional-scale assessment was based on measurements (2000-2015) of the dissolved concentrations of two water quality constituents in groundwater in the GLB, nitrate and chloride, as part of ongoing monitoring of groundwater quality. For this initial assessment, the data were obtained from groundwater monitoring networks maintained by (1) the U.S. Geological Survey (USGS) and its partners, and (2) the Ontario Ministry of the Environment, Conservation and Parks (MECP) and its partners.

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%).

Only "shallow" groundwater samples (collected from wells screened at depths less than (<) 40 metre) were included in this assessment, given that shallow groundwater is the most interactive with the rest of the hydrologic system, including surface water 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 spatial distribution of data used in this assessment was uneven (Figures 1-6). Given that there were very few

data points for the Lake Superior basin, the groundwater quality in this basin was assessed as "undetermined." The Lake Michigan basin was assessed using data from only the western portion of the basin given the lack of data on the east side. Likewise, the Lake Huron basin was assessed using data from the southeast portion of the basin, given that there were only two scattered data points throughout the northern and western portions of this basin. In the Lake Erie basin, data were concentrated in the north, resulting in large areas (especially the southwest portion) where groundwater quality data were limited. The Lake Ontario basin had sufficient, distributed data and thus has no caveats to note on the whole basin assessment for this lake.

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), and a tendency for groundwater quality to range from "Fair" to "Good" in the less developed areas (e.g., Huron and Superior basins) (Figure 1). For example, although only 22 data points (wells) were available for the Lake Superior basin, 90.9% of these (20) had "Good" groundwater quality.

For the Canadian monitoring wells included in this study, statistical tests indicated no significant difference (probability value less than 0.05) between the depths of the wells and the concentrations of nitrate and chloride (Figure 8). The lack of correlation may reflect differences in the settings of the well sites (e.g., differences in land uses and in nitrate loadings from surface, and differences in subsurface conditions such as permeability of geologic units and in intensity of microbial activity).

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, in the western portion of the Lake Michigan basin, many of the wells have excessive nitrate concentrations resulting in the overall assessment of "Poor" water quality. But, if only the chloride concentrations were considered, many of these wells would have been assessed as having "Fair" to "Good" groundwater quality (data not shown). 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 noted on page 2 of this report, it is informative to analyze both contaminants together, in particular, as it provides a fairly 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. For example, in a recent national study in the United States, which included the Great Lakes basin, DeSimone et al. (2014) stated that "concentrations of .. chloride, and (or) nitrate in groundwater increased in two-thirds of groundwater well networks that were sampled at 10-year intervals between the early 1990s and 2010" (Figure 9). Similarly, on the Canadian side of the Great Lakes Basin, Sawyer (2009) reported that "increasing concentrations of nitrate and chloride are obvious" in groundwater throughout the Grand River watershed. 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).

<u>Chloride</u>

One of the early studies to document chloride trends in the Great Lakes basin was Bubeck et al. (1971), who reported that "salt used for deicing streets near Rochester, New York, had increased the concentration of chloride in Irondequoit Bay at least fivefold over two decades."

Thomas (2000a) investigated groundwater quality in the Detroit metropolitan area, and found that "young, shallow waters…. had significantly higher median concentrations of ….chloride ….than older, deeper waters." Based on analysis of chloride/bromide ratios, Thomas (2000a) concluded that the elevated salinities were "due to human activities rather than natural factors, such as upward migration of brine."

Kelly and Wilson (2008) reported that the majority of shallow public supply wells in some counties in northeastern Illinois have had increasing chloride concentrations since the 1960s. The increases were attributed primarily to "road salt runoff." DeSimone et al. (2014) reported that in the glacial aquifer system, which extends across the northern United States, including the Great Lakes basin, "chloride concentrations were highest in shallow groundwater beneath urban areas, reflecting the use of deicing salt and the many other manmade sources of chloride in urban and suburban areas." Similarly, Mullaney et al. (2009) also reported evidence for increasing chloride

concentrations in streams in urbanized and urbanizing areas of the United States, including the Great Lakes basin.

In addition, the USGS National Water-Quality Assessment (NAWQA) has evaluated various well networks across the U.S. 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).

On the Canadian side of the Great Lakes basin, Howard and Beck (1993) reported that background concentrations of chloride in groundwater in glacial deposits in southern Ontario were in the range of 15–20 mg/L, but chloride concentrations were as great as 700 mg/L in domestic wells, as great as 2,840 mg/L in urban springs, and as great as 13,700 mg/L in pore waters extracted from beneath shallow urban areas. The potential sources of the chloride included road salts, landfill leachates, agricultural fertilizers, and saline bedrock waters. There was "extensive association of high chloride concentrations with urbanization in Metropolitan Toronto" (Howard and Beck, 1993). Bowen and Hinton (1998) reported that long-term monitoring of surface water in the Greater Toronto area showed a "gradual increase in chloride concentrations" and that "detailed baseflow water chemistry surveys….confirm that lower chloride concentrations occur predominantly in the rural portions of the watersheds." Similarly, Williams et al. (2000) concluded that chloride contamination of groundwater in the Greater Toronto Area was strongly related to urbanization and use of road de-icing salt.

In a more detailed study of a watershed in Metropolitan Toronto, Howard and Haynes (1993) estimated that 55% of the deicing salt applied each winter to roads, highways, and parking lots entered "temporary storage in shallow sub-surface waters." Howard and Haynes (1993) predicted that if salt application was maintained at the same rate, the average steady-state chloride concentrations in groundwater discharging as springs in the basin would exceed 400 mg/L, possibly within 20 years. In a follow-up study of the same watershed (20 years later), Perera et al. (2013) reported that chloride concentrations in baseflow ranged widely, peaking at 500–600 mg/L in late spring, and then declining to around 250–300 mg/L. This was evidence that "a component of the groundwater, elevated in salinity due to the prior season's salting activity, moves…. rapidly to the stream via relatively shallow, preferential flow zones within the aquifer". Perera et al. (2013) reported that if "current road salt application rates are continued, late summer baseflow chloride concentrations will reach around 505 mg/L, almost double present levels," and would be above the drinking water guideline and the CCME (2012) aquatic chronic toxicity guideline. Similarly, Eyles and Meriano (2010) reported that in an urbanized watershed at Pickering, Ontario (Lake Ontario basin), 52% of the deicing salt applied "accumulates in groundwater where it continues to be released as brackish baseflow to creeks in summer."

By 1998, the Ontario Ministry of the Environment (1998) reported that a high percentage of 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 the 2009 "State of the Great Lakes" report, Sawyer et al. (2009) noted that chloride levels in groundwater in the Grand River watershed of Ontario "can be linked to urban growth and its associated land uses." Sawyer et al. (2009) reported that increasing chloride concentrations have been observed in most municipal wells in the Grand River watershed, and that this increase has been attributed to winter deicing of roads with sodium chloride. Similarly, at Barrie, Ontario, one of the municipal wells has an upward trend in chloride concentrations that has become a drinking water issue (South Georgian Bay-Lake Simcoe Source Protection Committee., 2015). Another example is in the town of Orangeville, Ontario, where increasing chloride concentrations were documented in 5 of the 12 municipal supply wells for the 1982–2012 period (Credit Valley Conservation Authority, 2015).

Various Conservation Authorities in Ontario have also reported chloride trends in groundwater monitoring wells. For example, the Credit Valley Conservation Authority (2012) reported statistically significant increasing chloride concentration trends in five of the six Provincial Groundwater Monitoring Wells within its watershed, while the sixth had a statistically significant decreasing chloride trend. Another example is a report by the Central Lake Ontario Conservation Authority (2015), which indicated that, for the period 2002–2008, chloride trends were evident in five of sixteen monitoring wells in the Central Lake Ontario Source Protection Area, two of which showed increasing chloride levels, while the other three wells had decreasing chloride levels.

Nitrate

Nitrogen is an essential nutrient for plants and animals. Nitrogen 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 water systems.

Nitrate is highly soluble in water and weakly absorbed by soil particles so it easily infiltrates through the soil profile and subsequently enters the groundwater system. The ability of nitrate to move through the soil can depend on the biological activity in the soil, the type of soil, and on the concentration of nitrate in the infiltrating water (Mikolajkow, 2003).

A study of groundwater quality in agricultural areas of Ontario was funded by Agriculture Canada in 1991. Approximately 1,300 domestic farm wells were sampled in 1991 and 1992 and the groundwater samples were analyzed for nitrate, total and fecal coliforms, and several pesticides (Rudolph and Goss, 1993). Nitrate concentrations exceeded the drinking water quality standard of 10 mg N/L in samples from 15% of the domestic farm wells. The occurrence and concentration of nitrate in groundwater were found to be associated with the following:

- Most of the nitrate contaminated wells were shallow dug or bored wells;
- The nitrate concentrations tended to be higher in areas where the soils had high permeability;
- The nitrate concentrations were consistent at the same location and did not show a seasonal variation; and
- The nitrate concentrations decreased linearly with depth.

Similarly, a survey in Ontario in the late 1990s showed 14% of drinking water supply wells on farms had nitrate concentrations above the drinking water quality limit (Goss et al. 1998). Nitrate concentrations in groundwater are often elevated in urban and agricultural areas (Dubrovsky et al. 2010; IJC, 2010).

Sawyer et al. (2009) noted a linkage between "increased agricultural activity and groundwater contamination and its impact on surface water quality." Some elevated nitrate concentrations are linked to agricultural practices, but some may also be linked to "rural communities with a high density of septic systems that leach nutrients to the subsurface."

In a study of nitrate concentrations in groundwater in an agricultural region within the western Lake Erie basin, Thomas (2000b found that 37% of the samples had elevated nitrate concentrations that indicated human effects (e.g., fertilizer, manure, septic systems), and that 7% of the samples had nitrate concentrations that exceeded the U.S. Environmental Protection Agency's Maximum Contaminant Level of 10 mg N/L (U.S. Environmental Protection Agency, 2015).

Similar to chloride, increasing nitrate concentrations in some of the municipal supply wells in the Town of Orangeville have also been seen from 1982-2012 (Credit Valley Conservation Authority, 2015). The Central Lake Ontario Conservation Authority (2015) reported that, for the period 2002–2008, nitrate trends were evident in two of sixteen monitoring wells in the Central Lake Ontario Source Protection Area; one showing increasing nitrate concentrations and the other showing decreasing concentrations. Likewise, the USGS NAWQA program found increasing trends in nitrate in the Lake Erie and Lake Michigan basins (Lindsey et al., 2016).

Potential for Future Incorporation of Additional Data from other Sources to Address Gaps in Monitoring Networks
In February 2017 Environment and Climate Change Canada contracted Waters Environmental Geosciences Ltd. (2017) 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 that fit the overall criteria (excepting that they were not part of the provincial groundwater monitoring network). This included nine additional wells in the Lake Superior basin and 22 additional 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 have not been included in the current status assessment, but they are available for possible inclusion in future assessments.

Due to the sparse distribution of the data from the additional wells identified by Waters Environmental Geosciences Ltd. (2017), 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 sources that could be used in 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 is required, as a follow up to the above project. Future 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.

Another potential source of additional data for status assessment in the Ontario portion of the Great Lakes basin is the Ontario Geological Survey's ongoing Ambient Groundwater Geochemistry program. The extensive dataset for southern Ontario (more than 2300 samples collected from 2007-2014) has been released (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 are focused in the North Bay area, which includes a portion of the Lake Huron basin (https://www.mndm.gov.on.ca/sites/default/files/article_files/ontario_geoscience_mapping_projects_2018.pdf). The Ambient Groundwater Geochemistry data could be used for the status assessment, which would address some of the spatial data gaps that were 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.

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 groundwater, 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 the Water Quality Index (WQI), which is calculated using a total of eight parameters, including both chloride and nitrate concentrations.
- 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.
- Watershed Stressors – 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 areas of more intense land use, including urban and agricultural land use. 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 should be given to 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 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 Limitations

The networks of monitoring wells that were used for this assessment differ on the U.S. 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 analyses used for the U.S. and Canadian data also differed, this is not likely to have had a substantial effect on the outcome of the assessment.

This sub-indicator takes into account 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 natural chemicals (e.g., petroleum hydrocarbons and arsenic), synthetic chemicals (e.g., organic pesticides and pharmaceuticals), or other substances, such as pathogenic microorganisms. Also, the impact of some 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 was 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 naturally high chloride concentrations that are controlled by hydrogeological/geochemical conditions. One possible approach would be to look at chloride/bromide (Cl/Br) ratios, which hold some promise in terms of helping to distinguish naturally occurring chloride from anthropogenic chloride (e.g., derived from de-icing salt) (e.g., Thomas, 2000a; Panno et al. 2006; Katz et al., 2011). Zones with naturally high chloride in groundwater might be particularly vulnerable to 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 is needed. 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

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 feed, 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 focusses 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).

However, further adoption of these practices is required, being mindful that it can take many years for changes in in land management practices to positively impact shallow groundwater quality.

Future Assessments at the Sub-watershed Level

In future assessments, where sufficient data are available, the basic areal (geographic) unit of observation for this sub-indicator may be sub-watersheds within the Great Lakes basin. However, due to large gaps in spatial distribution of currently available groundwater quality data (USGS and MECP monitoring networks) this component will not be included in the 2017 status report.

For inclusion of a sub-watershed-level assessment in future sub-indicator reports, the following tasks would need to be completed: (1) augmentation of the shallow groundwater quality dataset to fill in gaps (spatial distribution); (2) a consistent way to identify sub-watersheds in both Canada and the United States will have to be selected. One approach would be to use the existing "Tertiary watersheds" defined for the Ontario side of the Great Lakes basin, and approximately equivalent sub-catchments on the U.S. side of the basin. Another approach would be to use the sub-catchments defined for application of the Watershed Stressor sub-indicator for the Great Lakes (e.g., Hollenhorst et al. 2007)

As for the assessment of lake drainage basins, the density of data (chloride and nitrate concentrations) that will be considered adequate for sub-watershed groundwater quality assessment will likely vary within the Great Lakes basin. It will be necessary to exercise professional judgement in assessing whether sufficient information is available for the sub-watershed and whole basin assessments. Such judgement could consider several sources of information, as discussed above for the assessment of lake drainage basins.

Application to the entire Great Lakes Basin

When sufficient data are available (based on professional judgement), the same approach as used for the lake watersheds and sub-watersheds (above) will be 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.

Other considerations for future assessments include:

- Determining the statistical approach to use in order to identify trends for the Great Lakes basin.
- Considering the Watershed Stressor and Water Quality in Tributaries sub-indicator assessments to see if the patterns of wells with degraded water quality mirror the patterns of watershed stressors (e.g. land use). A sub-watershed assessment may be useful in the future to link to land use activities in various parts of the Great Lakes basin.
- Considering the inclusion of other constituents to assess groundwater quality in future reports.
- Considering the inclusion and value of other chloride and nitrate surveillance data which are not part of ongoing monitoring programs in future reporting cycle to address possible data gaps.
- Considering the inclusion of groundwater quality metrics in the Treated Drinking Water sub-indicator report in the future.
- 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.

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 U.S., including areas in the Great Lakes basin (Lindsey et al., 2016). How the contaminated

groundwater impacts and interacts with the water of the Great Lakes, in particular in the nearshore zone, requires a better understanding.

Acknowledgments
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List of Tables

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 GLB.
Table 2.Summary of well data assessments for each Great Lake.
Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

List of Figures

Figure 1. Groundwater quality status in Great Lakes basin based on nitrate and chloride concentrations in shallow groundwater (based on measurements from 2000-2015 for wells ≤ 40 m below ground). A total of 670 monitoring wells in the basin were included in the analysis with groundwater quality being assessed as "Good" in 294 (√), "Fair" in 173 (+), and "Poor" in 203 (x). Symbols indicate the results for individual monitoring wells, and shaded areas indicate the results for each lake basin.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey
Figure 2. Assessment results for the groundwater quality sub-indicator for the Lake Superior basin (based on measurements from 2000-2015 for wells < 40 m below ground). Symbols indicate the results for individual monitoring wells. The U.S. data plot very close together near the international border, having the appearance of one (overlapped) symbol.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey
Figure 3. Assessment results for the groundwater quality sub-indicator for the Lake Michigan basin (based on measurements from 2000-2015 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells.
Source of data: U.S. Geological Survey
Figure 4. Assessment results for the groundwater quality sub-indicator for the Lake Huron basin (based on measurements from 2000-2015 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 and U.S. Geological Survey
Figure 5. Assessment results for the groundwater quality sub-indicator for the Lake Erie basin (based on measurements from 2000-2015 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 and U.S. Geological Survey
Figure 6. Assessment results for the groundwater quality sub-indicator for the Lake Ontario basin (based on measurements from 2000-2015 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 and U.S. Geological Survey
Figure 7. Ternary diagram showing the lake- by- lake groundwater quality assessments. Lake Superior is assessed as "Undetermined" and therefore is not included on this ternary diagram.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey
Figure 8. Plots of nitrate and chloride concentrations versus depth for the Canadian monitoring wells included in this study. (Pearson correlations for these: nitrate vs depth: -0.092, p = 0.146; chloride vs depth: 0.038, p = 0.543, i.e., not significant at the 0.05 level).
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Figure 9. Maps illustrating decadal changes (from early 1990s to 2010) in chloride and nitrate concentrations in groundwater in the United States, including increasing chloride and nitrate concentrations in the vicinity of the Great Lakes.
Source: DeSimone et al. (2014)

Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

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 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.

		"Poor" groundwater quality		"Fair" groundwater quality		"Good" groundwater quality	
Canadian data	**Total # wells**	**# wells**	**%**	**# wells**	**%**	**# wells**	**%**
Superior	9	1	11%	1	11%	7	78%
Huron	75	12	16%	16	21%	47	63%
Erie	54	15	28%	8	15%	31	57%
Ontario	114	35	31%	34	30%	45	39%
All basins	252	63	25%	59	23%	130	52%
		"Poor" groundwater quality		**"Fair" groundwater quality**		**"Good" groundwater quality**	
US data	**Total # wells**	**# wells**	**%**	**# wells**	**%**	**# wells**	**%**
Superior	13	0	0%		0%	13	100%
Michigan	136	64	47%	29	21%	43	32%
Huron	2	2	100%	0	0%	0	0%
Erie	123	35	28%	41	33%	47	38%
Ontario	144	39	27%	44	31%	61	42%
All basins	418	140	33%	114	27%	164	39%
		"Poor" groundwater quality		**"Fair" groundwater quality**		**"Good" groundwater quality**	
Binational data	**Total # wells**	**# wells**	**%**	**# wells**	**%**	**# wells**	**%**
Superior	22	1	5%	1	5%	20	91%
Michigan	136	64	47%	29	21%	43	32%
Huron	77	14	18%	16	21%	47	61%
Erie	177	50	28%	49	28%	78	44%
Ontario	258	74	29%	78	30%	106	41%
Entire Great Lakes Basin	670	203	30%	173	26%	294	44%

Table 2. Summary of well data assessments for each Great Lake.
Source: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

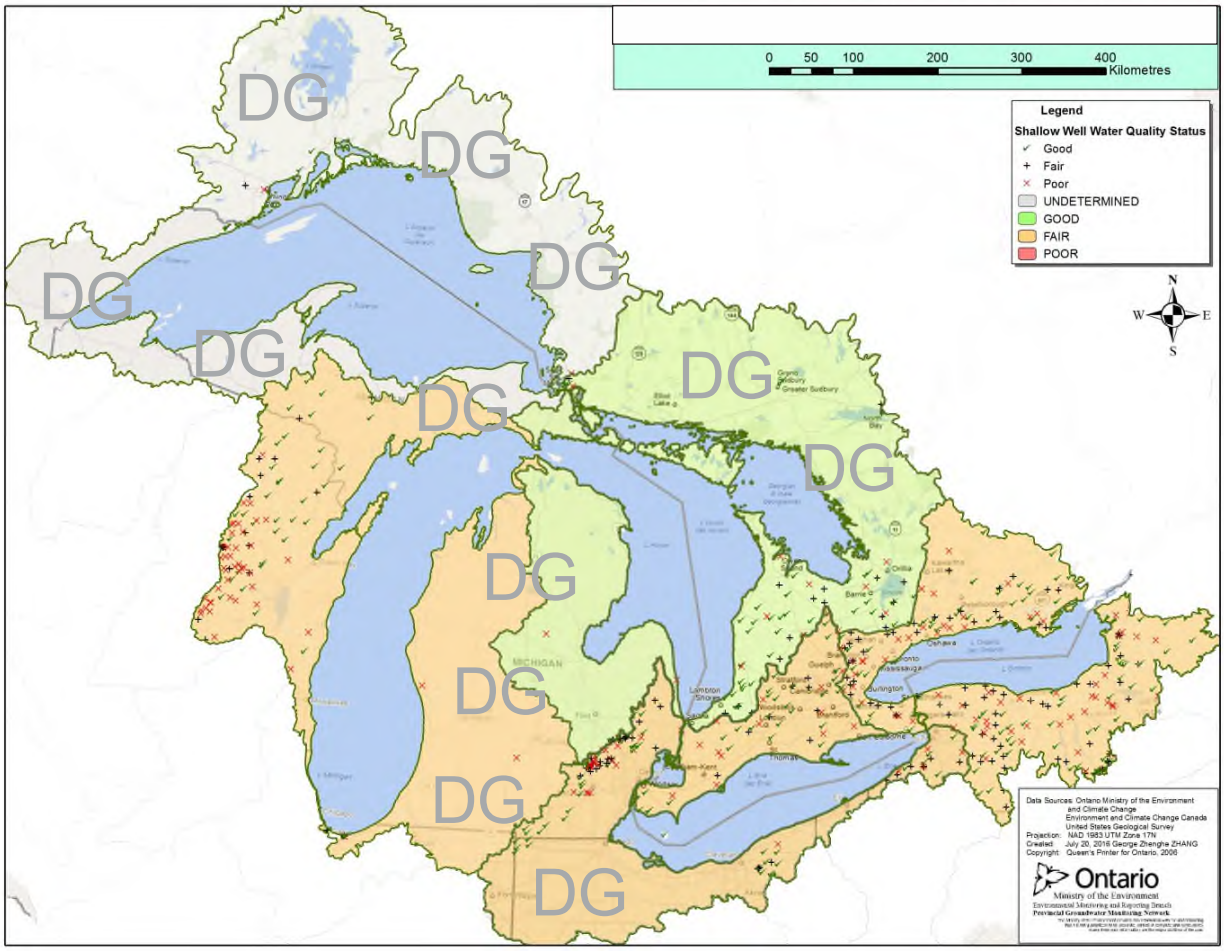


Figure 1. Groundwater quality status in Great Lakes basin based on nitrate and chloride concentrations in shallow groundwater (based on measurements from 2000-2015 for wells ≤ 40 m below ground). A total of 670 monitoring wells in the basin were included in the analysis with groundwater quality being assessed as "Good" in 294 (√), "Fair" in 173 (+), and "Poor" (x) in 203. Symbols indicate the results for individual monitoring wells, and shaded areas indicate the results for each lake basin. Subareas marked "DG" have data gaps, thus an Undetermined status. Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

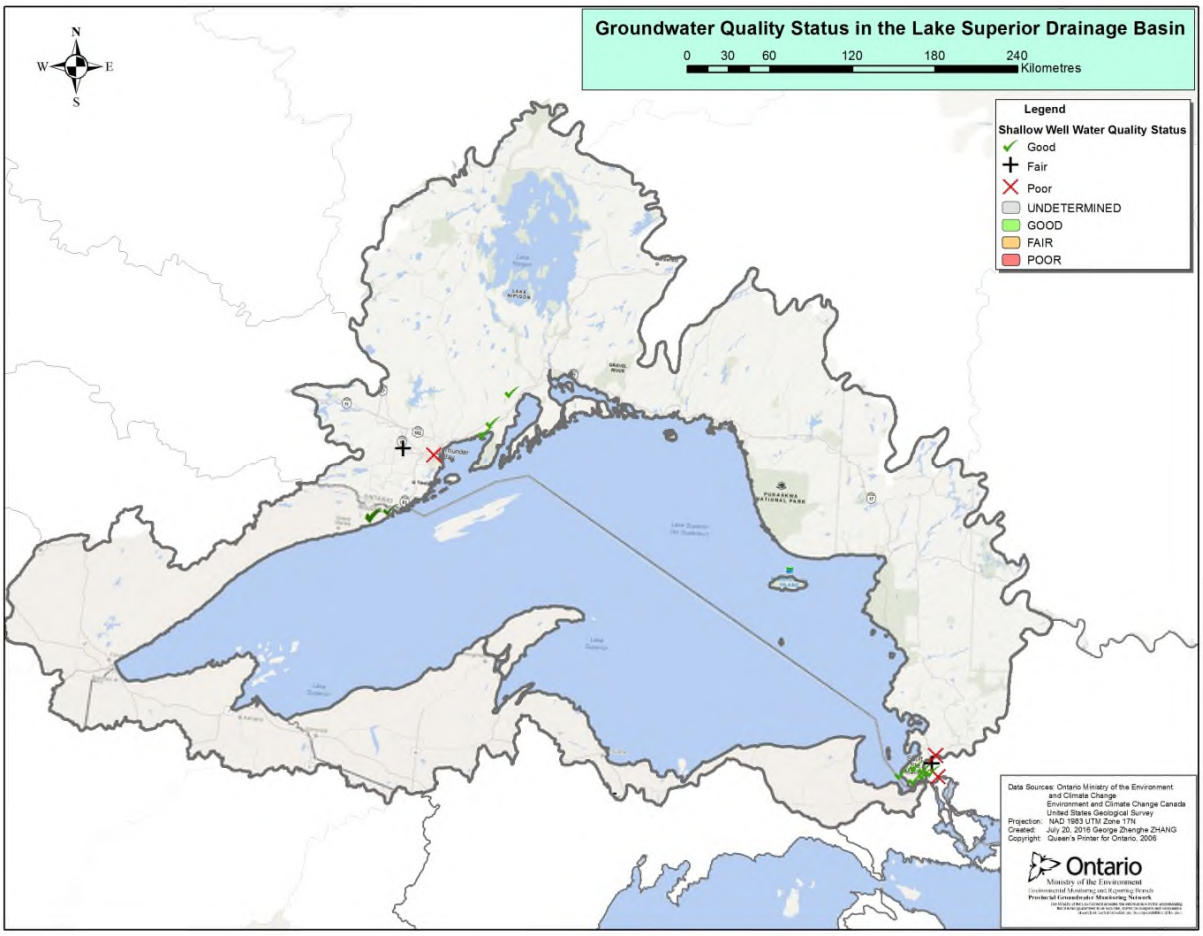


Figure 2. Assessment results for the groundwater quality sub-indicator for the Lake Superior basin (based on measurements from 2000-2015 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. The U.S. data plot very close together near the international border, having the appearance of one (overlapped) symbol. Areas with no data within 50 km will be shown as Undetermined, the other small areas may be given a specific status.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

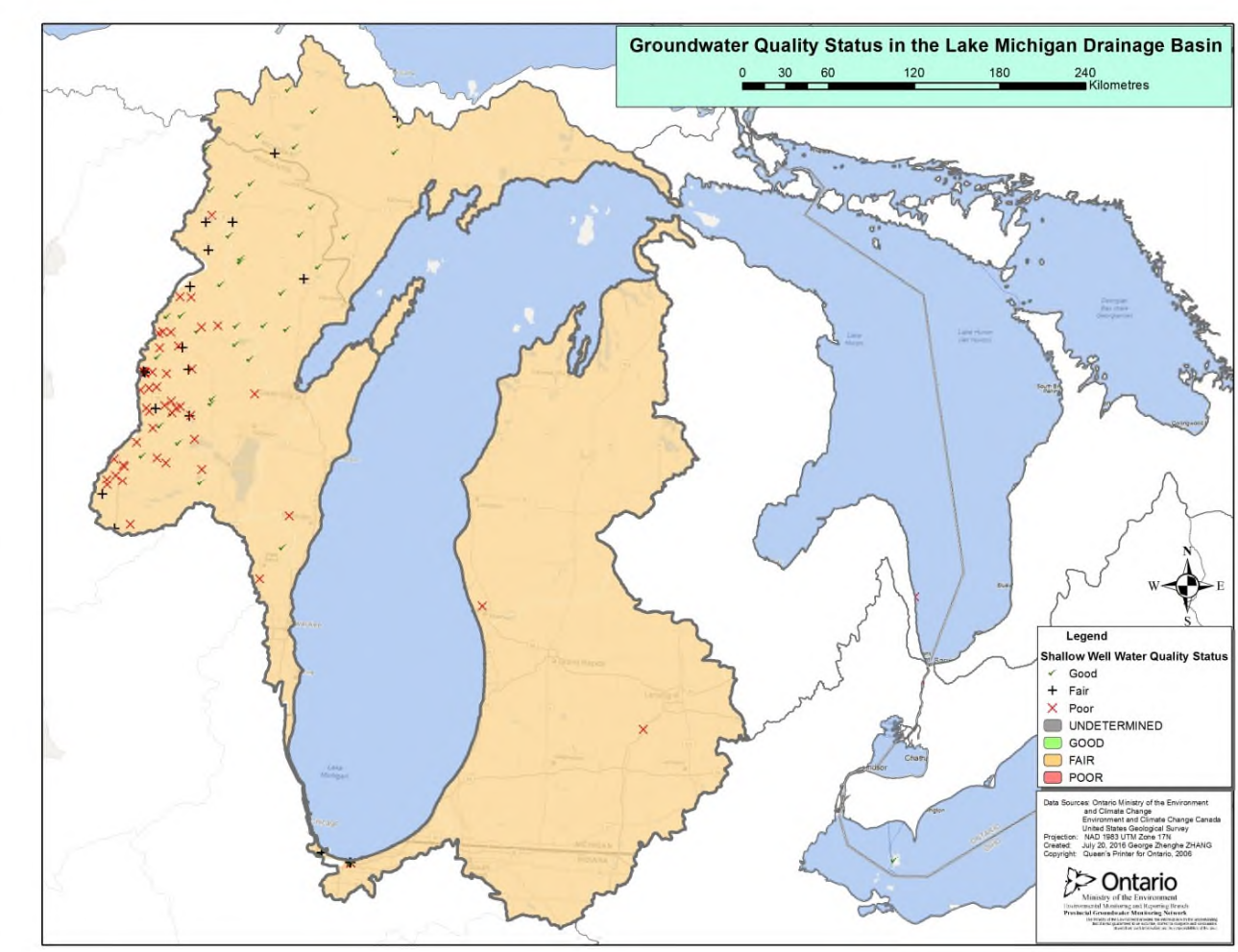


Figure 3. Assessment results for the groundwater quality sub-indicator for the Lake Michigan basin (based on measurements from 2000-2015 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Subareas with data gaps > 50 km will be shown as Undetermined status.
Source of data: U.S. Geological Survey

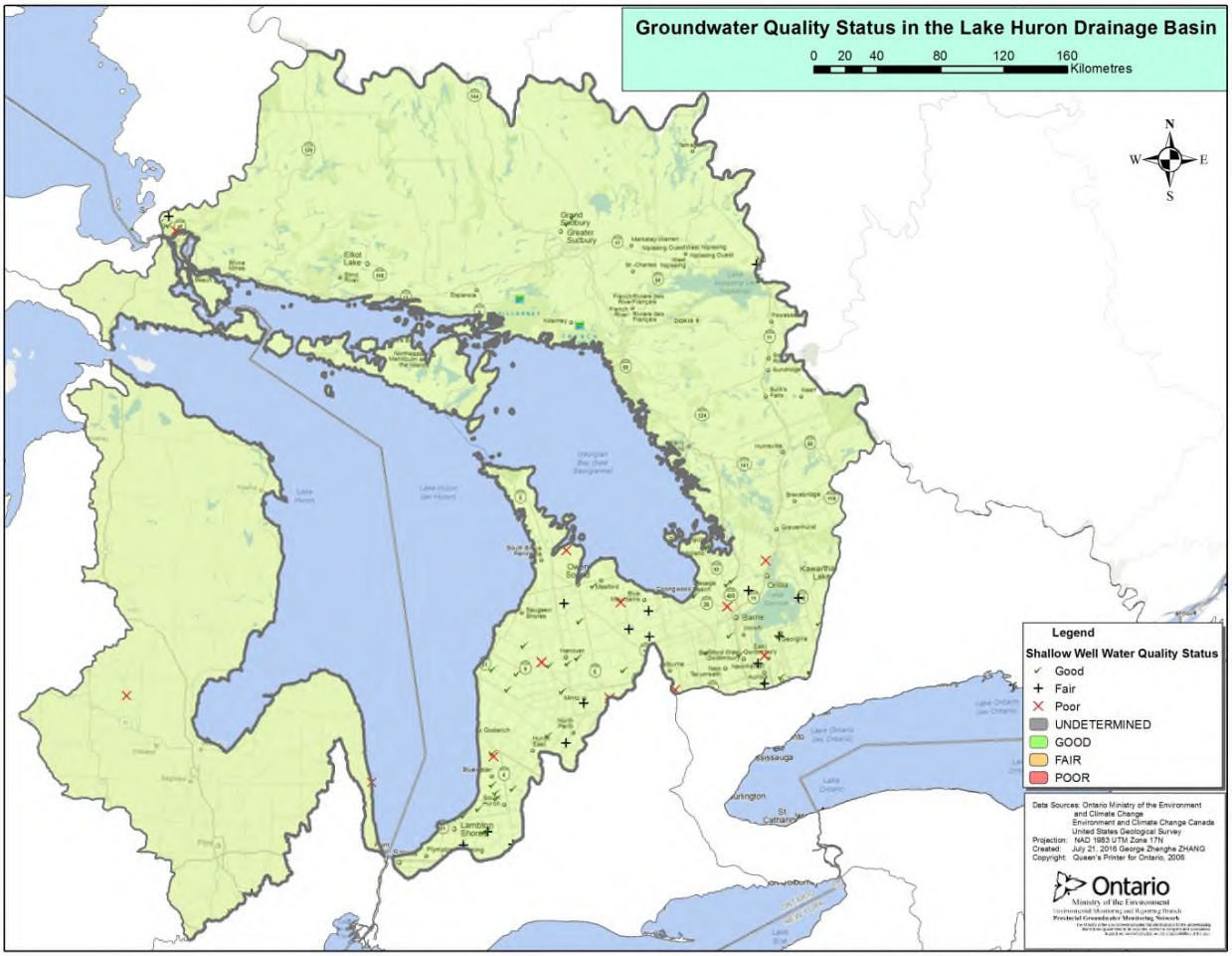


Figure 4. Assessment results for the groundwater quality sub-indicator for the Lake Huron basin (based on measurements from 2000-2015 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Subareas with data gaps > 50 km will be shown as Undetermined status.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

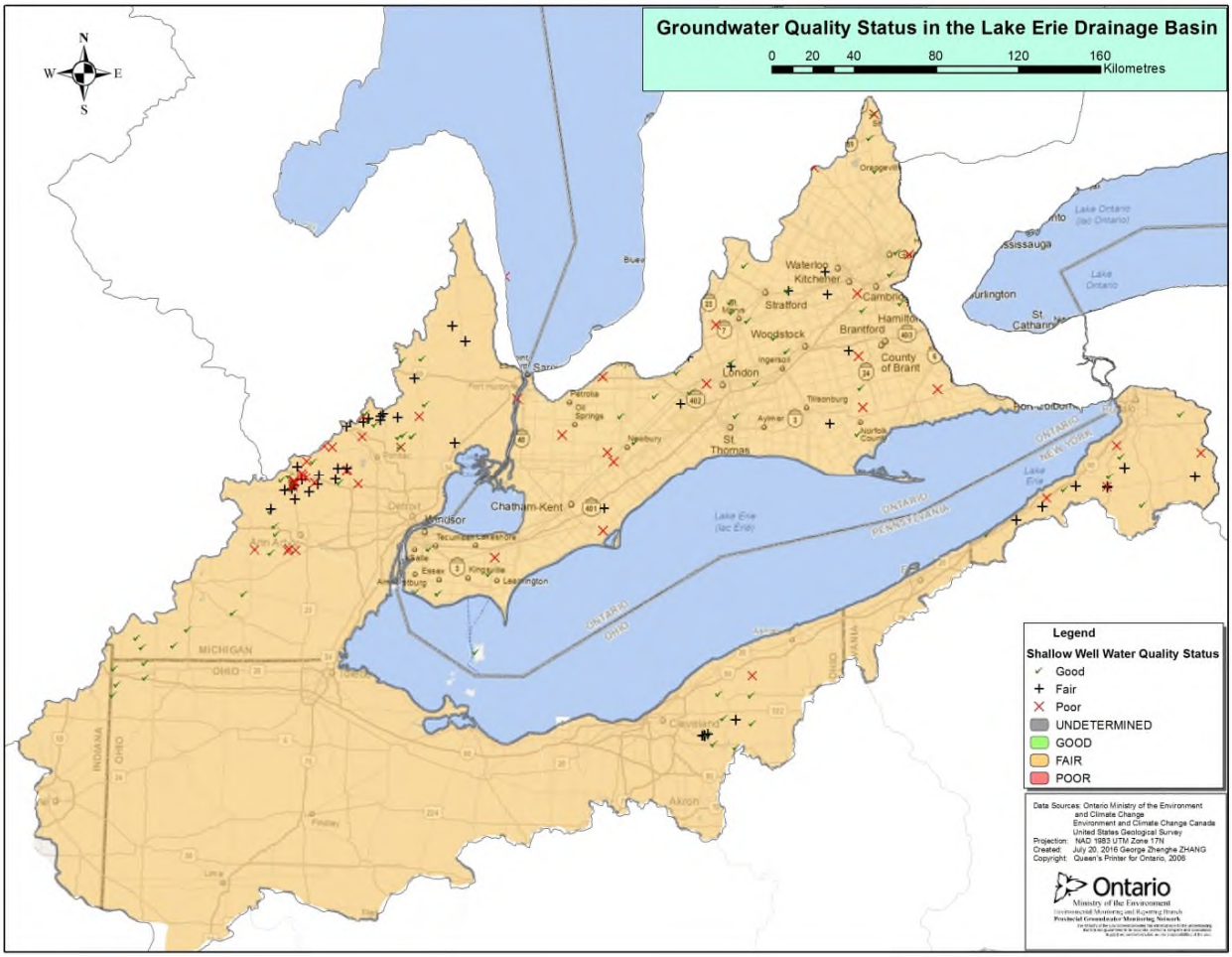


Figure 5. Assessment results for the groundwater quality sub-indicator for the Lake Erie basin (based on measurements from 2000-2015 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Subareas with data gaps > 50 km will be shown as Undetermined status.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

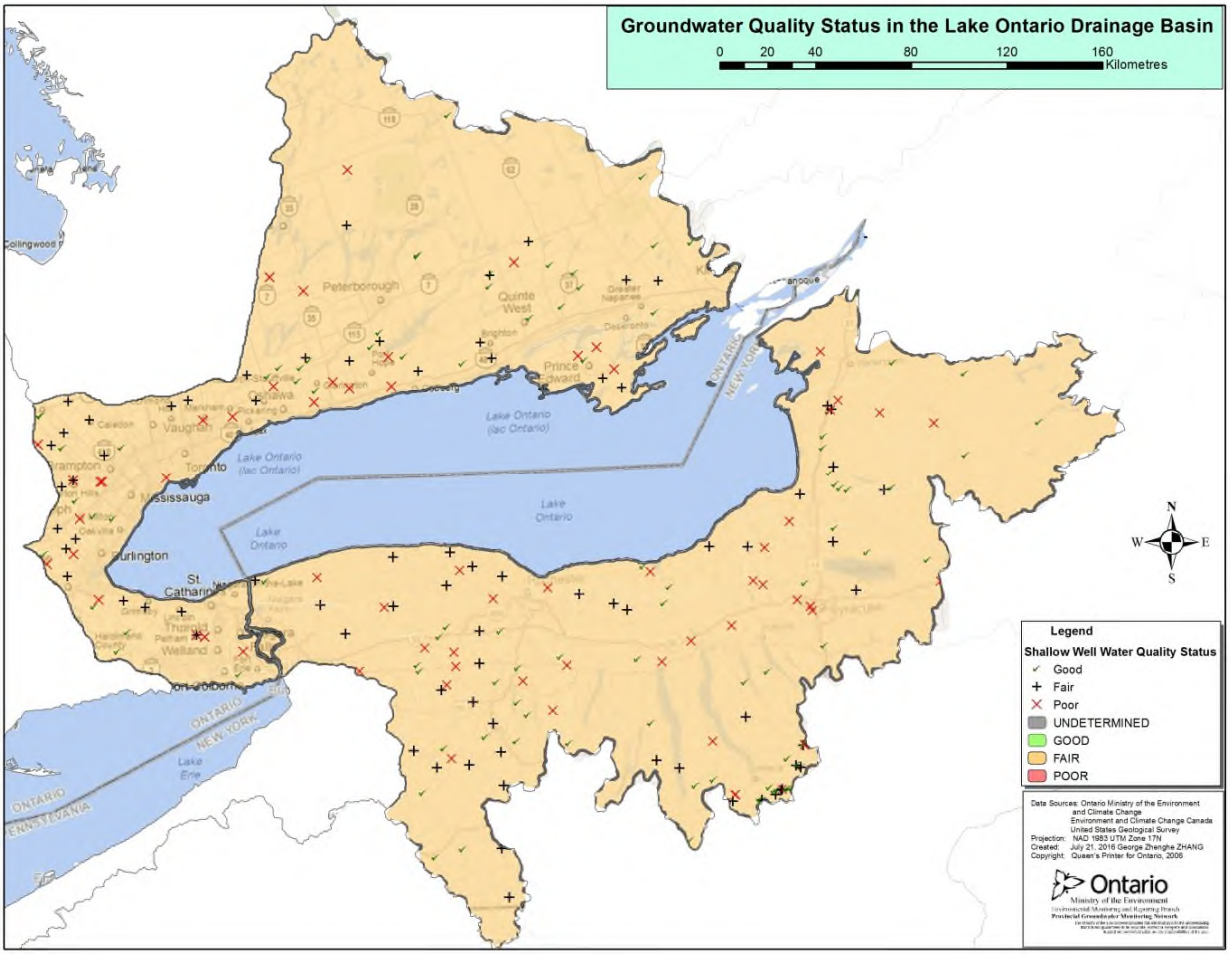


Figure 6. Assessment results for the groundwater quality sub-indicator for the Lake Ontario basin (based on measurements from 2000-2015 for wells ≤ 40 m below ground). Symbols indicate the results for individual monitoring wells. Subareas with data gaps > 50 km will be shown as Undetermined status.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

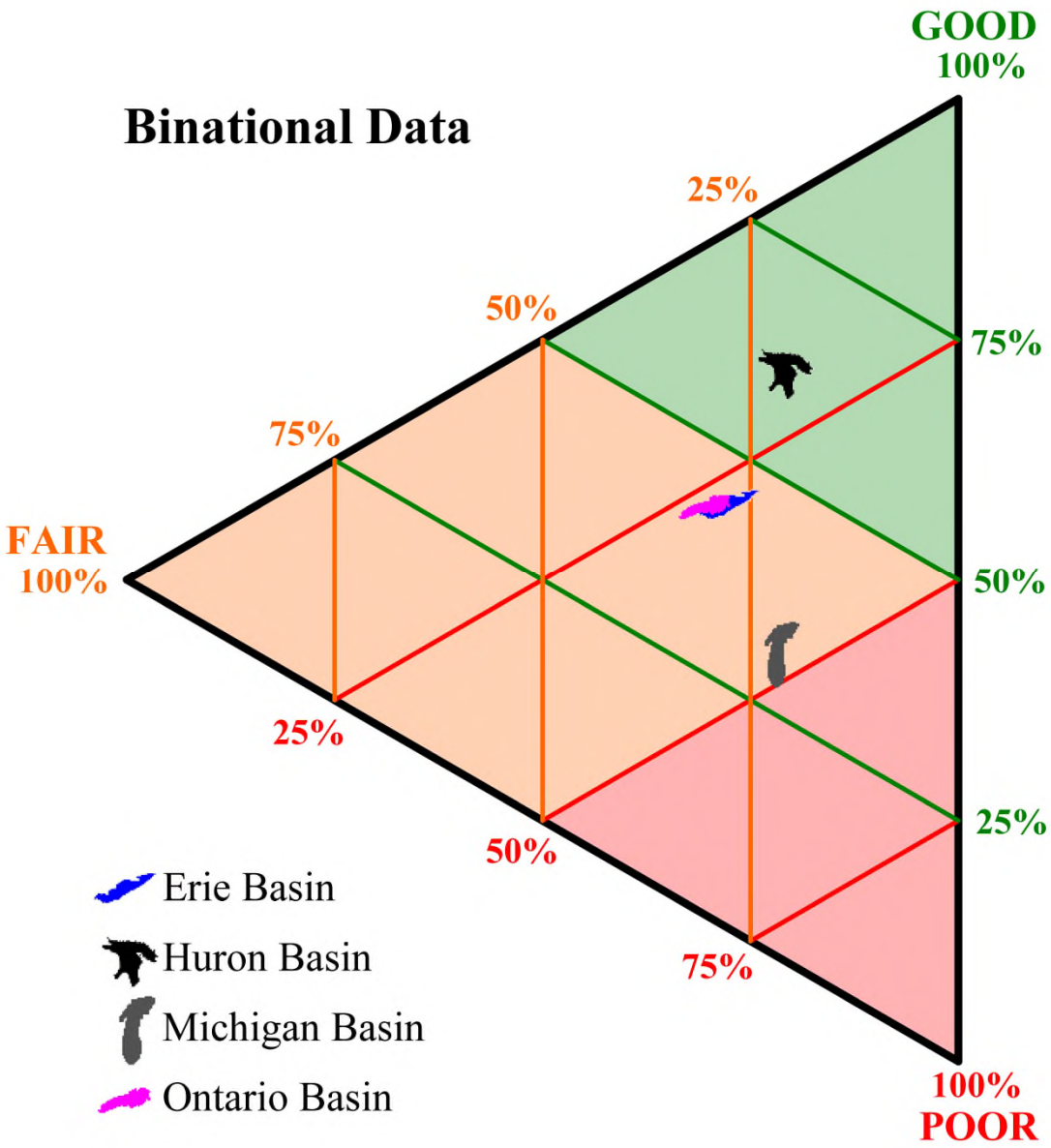


Figure 7. Ternary diagram showing the lake- by- lake groundwater quality assessments. Lake Superior is assessed as "Undetermined" and therefore is not included on this ternary diagram.
Source of data: Ontario Ministry of the Environment, Conservation and Parks and U.S. Geological Survey

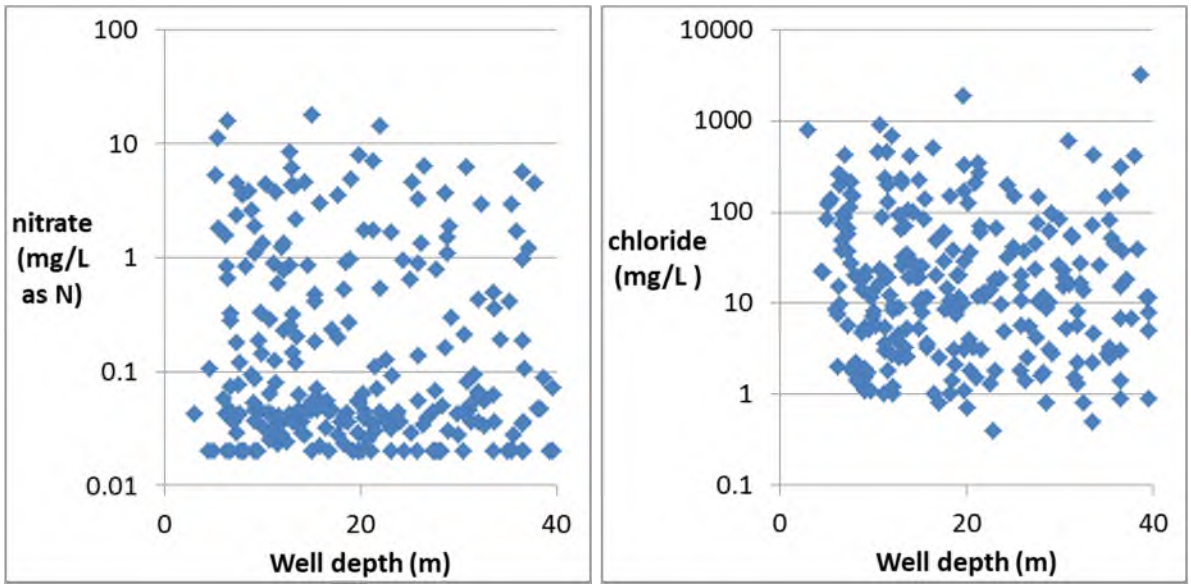


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Source of data: Ontario Ministry of the Environment, Conservation and Parks

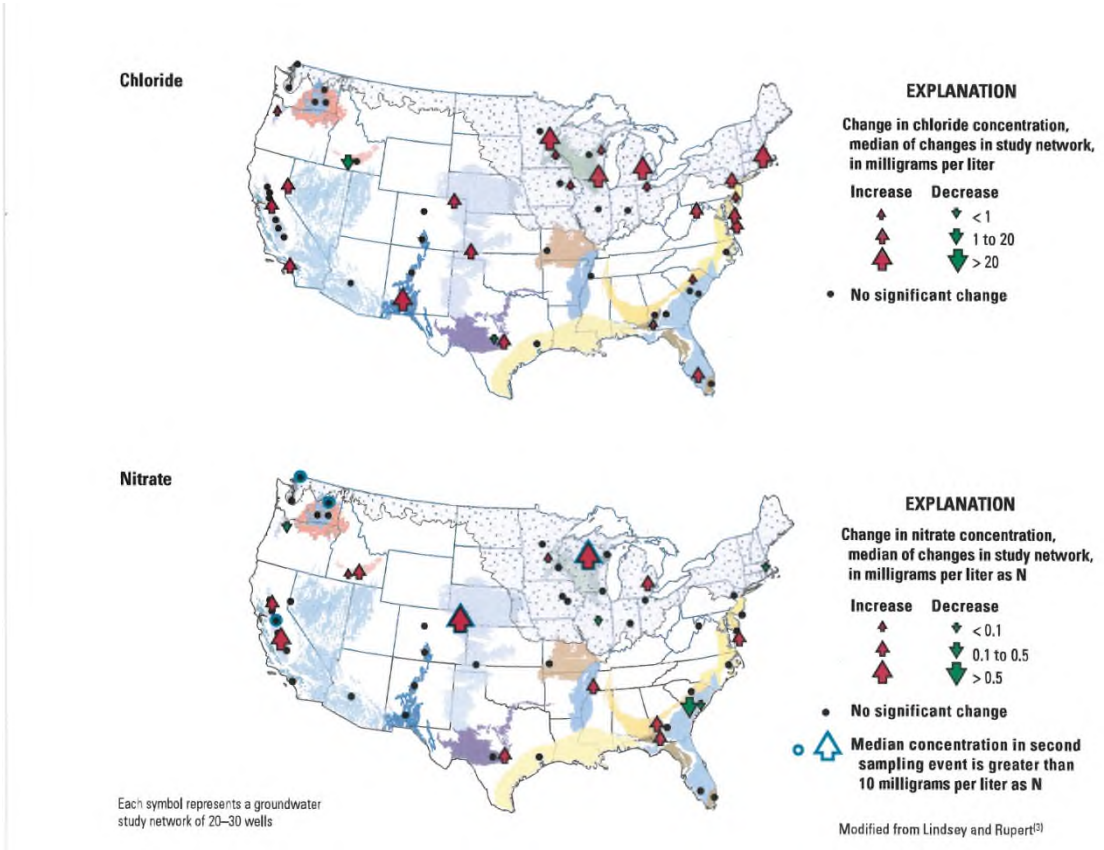


Figure 9. Maps illustrating decadal changes (from early 1990s to 2010) in chloride and nitrate concentrations in groundwater in the United States, including increasing chloride and nitrate concentrations in the vicinity of the Great Lakes.
Source: DeSimone et al. (2014)

Sub-Indicator: Forest Cover

Riparian Zone

Overall Assessment
Status: Fair
Trends:
10-Year Trend (2006-2011 for U.S. and 2002-2011 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 (61 – 73%) and low in the Lake Erie basin (31%) based on satellite imagery. Trends in forested cover (2006 – 2011 in U.S. and 2002 – 2011 in Canada) in riparian zone are showing unchanging conditions in the Lake Superior, Michigan and Huron basins, small decrease in the Ontario basin (-1.7%) and an increase in the Erie basin (+4.5%). 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 land area within the Lake Superior basin (93%), a moderate amount in the Lake Michigan, Huron and Ontario basins (48 - 65%) and low in the Lake Erie basin (19%) based on satellite imagery. Trends in forest cover across the lake basins are very similar to the riparian zone assessments, showing unchanging conditions in the Superior and Huron basins. However, losses in forest cover were seen in Lakes Michigan (-1.2%), Erie (-1.2%) with the largest losses in the Ontario basin (-3.9%).

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 (2006-2011 for U.S. and 2002-2011 for Canada): Unchanging
Long-term Trend: Undetermined
Rationale: Riparian zones in the Lake Superior basin have high forest cover and these waters are likely to be well protected. 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 (2006-2011 for U.S. and 2002-2011 for Canada): Unchanging
Long-term Trend: Undetermined
Rationale: Northerly 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 (2006-2011 for U.S. and 2002-2011 for Canada): Unchanging
Long-term Trend: Undetermined
Rationale: Northerly 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 Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend (2006-2011 for U.S. and 2002-2011 for Canada): Improving
Long-term Trend: Undetermined
Rationale: A low proportion of forest cover in riparian zones suggests heightened threat to water quality and ecosystem integrity. However, the trend (between 2002 and 2011) is improving on the Canadian-side of the basin. This basin also has low forest cover, which has declined over the 2002 to 2011 period on the Canadian-side of the basin despite an increase in the riparian forest cover. 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 (2006-2011 for U.S. and 2002-2011 for Canada): Deteriorating
Long-term Trend: Undetermined
Rationale: There is a moderate level of forest cover in riparian zones in this basin which suggests there a moderate risk to water quality and ecosystem integrity. Similarly, most watersheds in the Lake Ontario basin have moderate forest cover, which has declined over the 2002-2011 period on the Canadian-side of the basin. These data suggest there is a potential for water quality problems and risks to ecological integrity due to forest cover, particularly in Canada where losses have been larger while the U.S. has remained unchanged.

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

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 $\leq \pm$ 1%
Deteriorating: decreasing % of forest cover within riparian zones and changing if < - 1%.

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 the 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 watersheds 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 wetlands. Classified remote sensing data will 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 $\leq \pm$ 1% and changing if $>\pm$ 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 (63%), Huron (73%), and Ontario (61%). Only 31% 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 2011 and for the Canadian portions of the basins from 2002 and 2011. Trend analysis showed that riparian forest is unchanging for northern basins, a small increase in Erie (4.5%) and small decrease in Ontario (-1.7%) basins. Changes were small in the U.S. (<1.1%) and larger in Ontario (range from -3.3% to +16.3%) (Table 2). These trends should be interpreted with some caution, realizing the short span of time in which they are calculated (5 years for U.S. and 10 years for Canada). A longer record (>20 years) 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, 93% 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 49% (Michigan), 65% (Huron), 19% (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 U.S. portion of all lake basins, there is a trend of unchanging (Erie, Ontario) or small declines (Superior, Huron, Michigan), whereas in Canadian basins there are unchanging (Superior, Huron) to some larger declines (Erie, Ontario) in forest cover (Table 2).

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 pollutant, loadings to the tributaries and lakes and help to improve the negative effects 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 Limitations

- Percent forested data in riparian zones for lake basins are available, however, the data may not indicate if the expansion in forest area is occurring in riparian zones.
- 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 (USFWS, FIADB). Raw data available online at: http://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. 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 Crown land forest management units of 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 bulk of this assessment is 2002 imagery with 2004-2008 updates based on actual harvest, disturbance and renewal layers. A new and more detailed landcover product for southern Ontario will be available later in 2018 will be available for the next reporting cycle. A new U.S. national land cover dataset (NLCD 2016) is scheduled for release in late December 2018. It complements (updates) the 2011 data used for the forest cover in riparian zone for the U.S. portion of sub-indicator report. That will result in 7 spatially and temporally consistent products between 2001 and 2016 (i.e., 2001, 2004, 2006, 2008, 2011, 2013, and 2016).

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 FIADB) is 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 compile and calculate indicators given the different sources of data will be 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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Contributors: Charles Perry and Dale Gormanson, USDA Forest Service, 1992 Folwell Avenue, St. Paul, MN 55108

Information Sources

Fry, J., Xian, G., Jin, S., Dewitz, J., Homer, C., Yang, L., Barnes, C., Herold, N., and Wickham, J., 2011. Completion of the 2006 National Land Cover Database for the Conterminous United States, PE&RS, Vol. 77(9):858-864. Data Link: http://www.mrlc.gov/nlcd2006.php (accessed October 10, 2011).

Homer, C.G., Dewitz, J.A., Yang, L., Jin, S., Danielson, P., Xian, G., Coulston, J., Herold, N.D., Wickham, J.D., and Megown, K., 2015, Completion of the 2011 National Land Cover Database for the conterminous United States-Representing a decade of land cover change information. Photogrammetric Engineering and Remote Sensing, v. 81, no. 5, p. 345-354. Data link: http://www.mrlc.gov/nlcd11_data.php (accessed 27 October 2015).

List of Tables

Table 1. Percent of forest cover in riparian zones and percent change by basin for U.S. (2006 and 2011) and Canada (2002 and 2011) 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), 2011 (Homer et al. 2015) and Ontario Landcover 2002 and SOLRIS 2002 (OMNRF 2006, Forest Sustainability and Information Section, unpublished data) and Landcover 2008 and SOLRIS 2011(OMNRF 2015, Forest Sustainability and Information Section, unpublished data)
Table 2. Percentage of forest cover and percent change by lake basin for U.S. (2006 and 2011) and Canada (2002 and 2011) 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), 2011 (Homer et al. 2015) and Ontario Landcover 2002 and SOLRIS 2002 (OMNRF 2006, Forest Sustainability and Information Section, unpublished data) and Landcover 2008 and SOLRIS 2011(OMNRF 2015, Forest Sustainability and Information Section, unpublished data)

List of Figures

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) and treed wetlands.
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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) and treed wetlands.
Source: U.S. National Land Cover Database NLCD 2011 (Homer et al. 2015) and Ontario Landcover 2008 and SOLRIS 2011(OMNRF 2015, 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) and treed wetlands.
Source: U.S. National Land Cover Database NLCD 2011 (Homer et al. 2015) and Ontario Landcover 2008 and SOLRIS 2011(OMNRF 2015, 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 2011 (Homer et al. 2015) and Ontario Landcover 2008 and SOLRIS 2011(OMNRF 2015, Forest Sustainability and Information Section, unpublished data)

Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

U.S. Basin	Riparian Forest 2006 (ha)	Riparian Forest 2011 (ha)	% Change in Riparian Forest	Amount of Riparian Forest 2011	Class Value
Superior	172,927	171,014	-1.1%	85.9%	Good
Michigan	270,484	268,988	-0.6%	62.5%	Fair
Huron	93,021	92,367	-0.7%	56.4%	Fair
Erie	95,593	95,421	-0.2%	35.1%	Poor
Ontario	95,857	96,807	1.0%	58.0%	Fair
Total:	727,882	724,597	-0.5%	58.8%	Fair

Canada Basin	Riparian Forest 2002 (ha)	Riparian Forest 2011 (ha)	% Change in Riparian Forest	Amount of Riparian Forest 2011	Class Value
Superior	619,980	626,200	1.0%	98.8%	Good
Michigan					
Huron	607,694	611,857	0.7%	75.8%	Fair
Erie	37,571	43,689	16.3%	23.8%	Poor
Ontario	163,564	158,216	-3.3%	62.1%	Fair
Total:	1,428,808	1,439,963	0.8%	76.6%	Fair

Great Lake Basin	Riparian Forest 2006/02 (ha)	Riparian Forest 2011/11 (ha)	% Change in Riparian Forest	Amount of Riparian Forest 2011	Class Value
Superior	792,907	797,214	0.5%	95.7%	Good
Michigan	270,484	268,988	-0.6%	62.5%	Fair
Huron	700,715	704,224	0.5%	72.5%	Fair
Erie	133,164	139,110	4.5%	30.5%	Poor
Ontario	259,421	255,023	-1.7%	60.5%	Fair
Total:	2,156,690	2,164,560	0.4%	69.6%	Fair

Table 1. Percent of forest cover in riparian zones and percent change by basin for U.S. (2006 and 2011) and Canada (2002 and 2011) 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), 2011 (Homer et al. 2015) and Ontario Land-cover 2002 and SOLRIS 2002 (OMNRF 2006, Forest Sustainability and Information Section, unpublished data) and Landcover 2008 and SOLRIS 2011(OMNRF 2015, Forest Sustainability and Information Section, unpublished data)

U.S. Basin	All Forest 2006 (ha)	All Forest 2011 (ha)	% Change in Forest	Amount of Forest 2011	Class Value
Superior	3,539,252	3,483,919	-1.6%	83.5%	Good
Michigan	5,577,078	5,507,977	-1.2%	48.9%	Fair
Huron	2,048,628	2,006,615	-2.1%	49.8%	Fair
Erie	1,107,959	1,100,254	-0.7%	20.7%	Poor
Ontario	1,533,078	1,537,099	0.3%	46.1%	Fair
Total:	13,805,995	13,635,864	-1.2%	48.5%	Fair

Canada Basin	All Forest 2002 (ha)	All Forest 2011 (ha)	% Change in Forest	Amount of Forest 2011	Class Value
Superior	7,038,011	7,037,552	0.0%	98.9%	Good
Michigan					
Huron	6,278,642	6,289,194	0.2%	72.3%	Good
Erie	296,517	287,027	-3.2%	14.2%	Poor
Ontario	1,330,982	1,215,674	-8.7%	49.3%	Fair
Total:	14,944,151	14,829,448	-0.8%	73.0%	Good

Great Lake Basin	All Forest 2006/02 (ha)	All Forest 2011/11 (ha)	% Change in Forest	Amount of Forest 2011	Class Value
Superior	10,577,263	10,521,471	-0.5%	93.2%	Good
Michigan	5,577,078	5,507,977	-1.2%	48.9%	Fair
Huron	8,327,270	8,295,809	-0.4%	65.2%	Good
Erie	1,404,476	1,387,281	-1.2%	18.9%	Poor
Ontario	2,864,060	2,752,773	-3.9%	47.5%	Fair
Total:	28,750,146	28,465,312	-1.0%	58.8%	Fair

Table 2. Percentage of forest cover and percent change by lake basin for U.S. (2006 and 2011) and Canada (2002 and 2011) 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), 2011 (Homer et al. 2015) and Ontario Land-cover 2002 and SOLRIS 2002 (OMNRF 2006, Forest Sustainability and Information Section, unpublished data) and Landcover 2008 and SOLRIS 2011(OMNRF 2015, Forest Sustainability and Information Section, unpublished data)

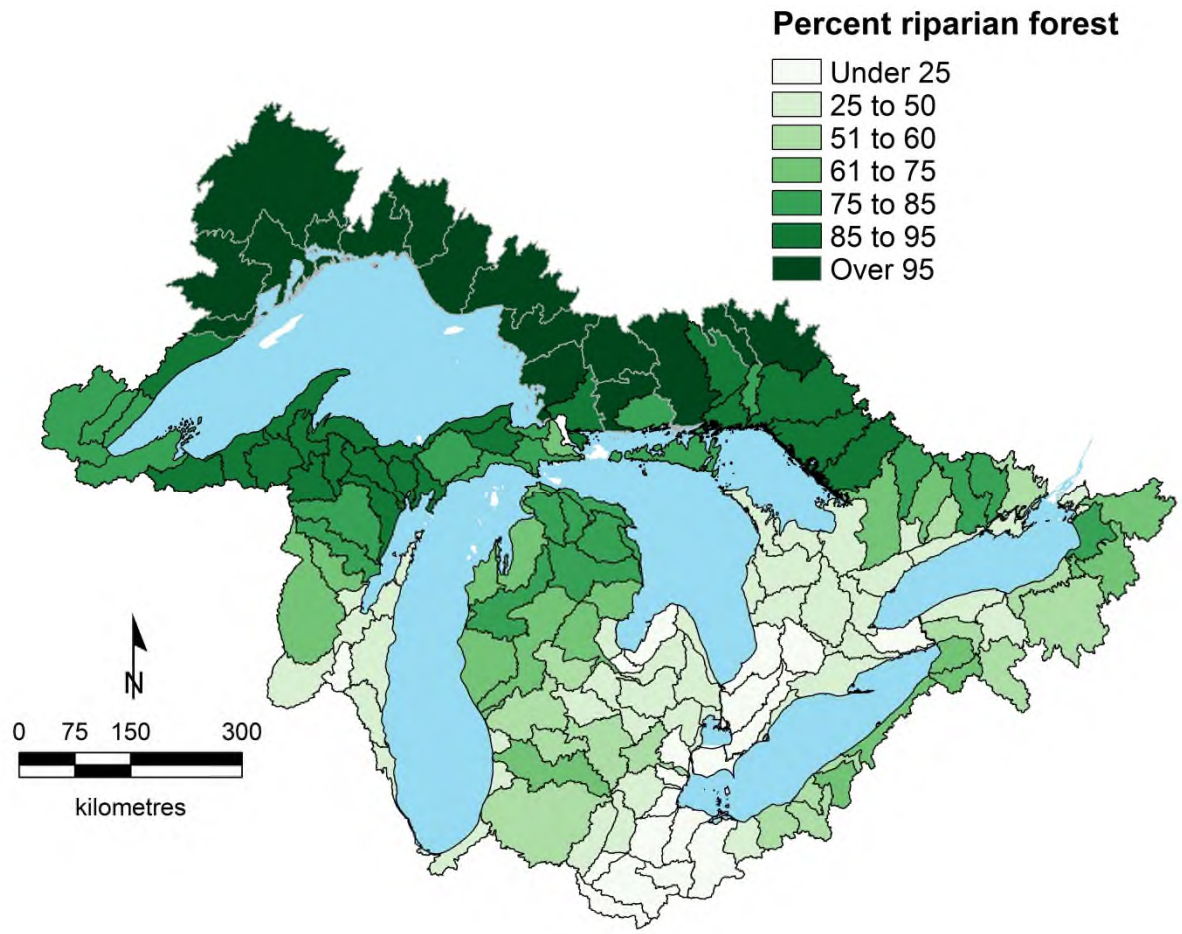


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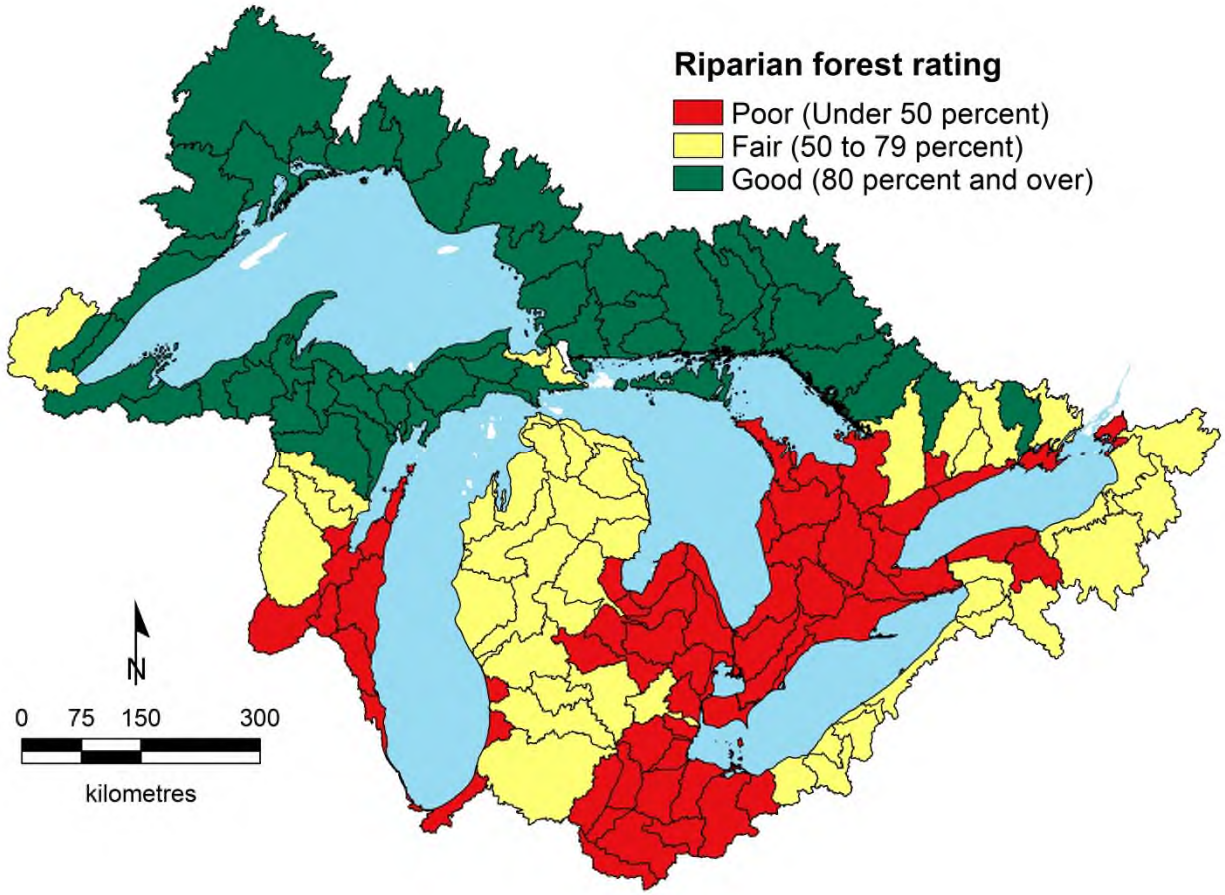


Figure 2. Forest cover within riparian zone (30 m buffer around water bodies) rating for tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed) and treed wetlands.
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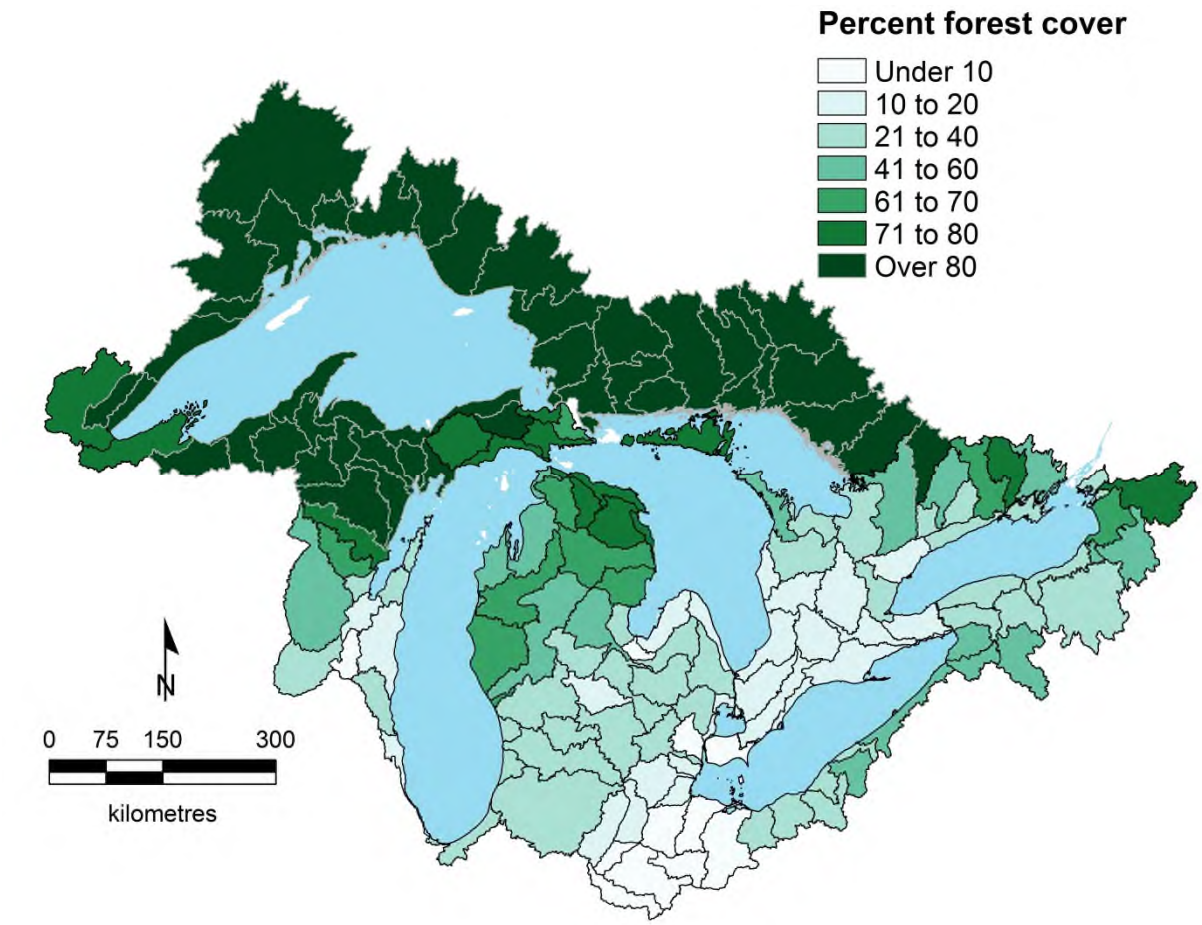


Figure 3. Percentage of forest cover in tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Forest cover was estimated from satellite imagery and includes a variety of forest types (i.e. deciduous, conifer, mixed) and treed wetlands.
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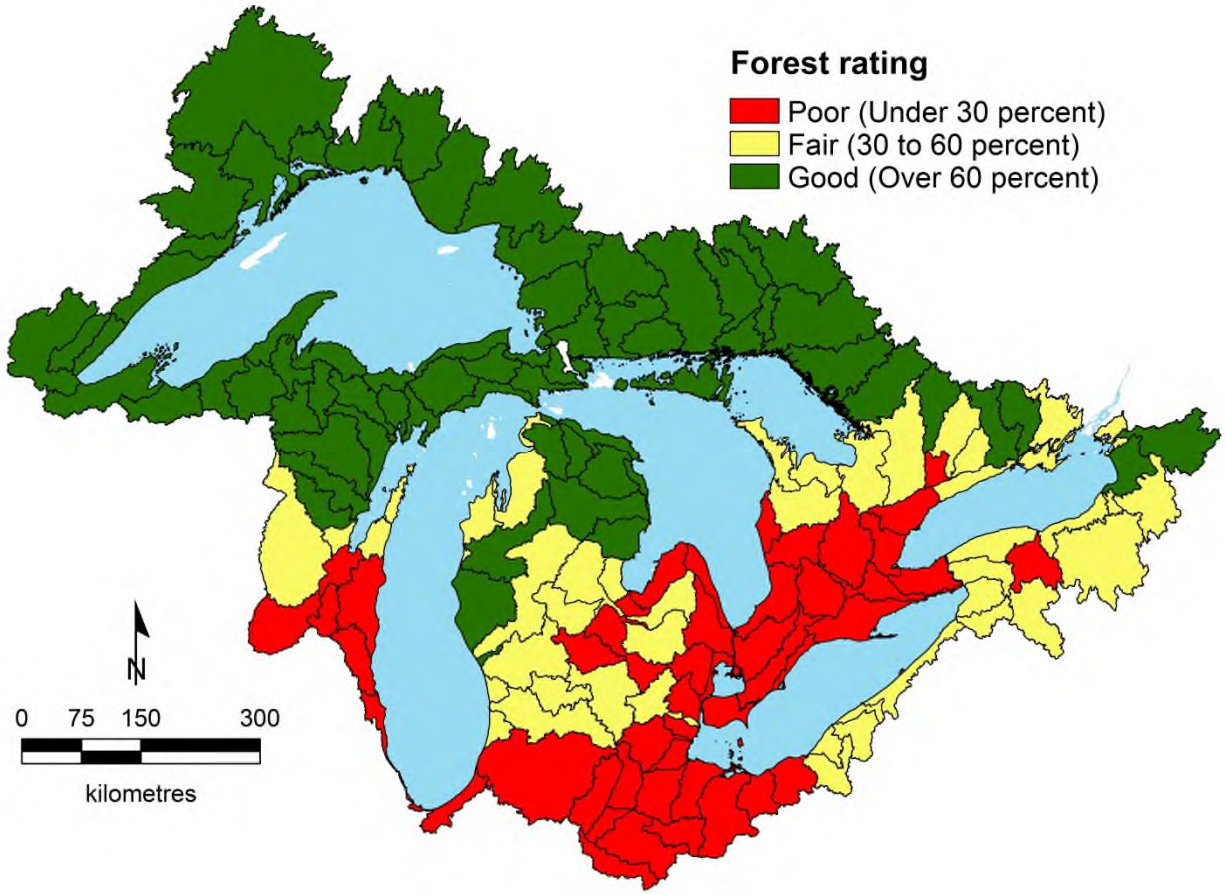


Figure 4. Forest cover rating in tertiary watersheds (HUC8 in U.S. and 4 digit in Ontario) of the Great Lakes. Source: U.S. National Land Cover Database NLCD 2011 (Homer et al. 2015) and Ontario Landcover 2008 and SOLRIS 2011(OMNRF 2015, Forest Sustainability and Information Section, unpublished data)

Sub-Indicator: Land Cover

Overall Assessment
Status: Fair
Trends:
10-Year Trend: Unchanging
Long-term Trend (2001-2011): Undetermined
Rationale: With 50% of the basin in agricultural or developed land use, the status by definition is "Fair" however, this percentage is straddling the Fair – Poor threshold. Based on the lake-by-lake assessments below, the overall sub-indicator assessment will remain as "Fair" for this reporting cycle. Across the entire basin, between 2001 and 2011, there was a net conversion of 393 km^2 from natural land cover to developed land cover. This decrease in natural land cover constituted 0.05% of the assessed land area (see explanation of the geographic extent considered under "Ecological Condition), resulting in a determination of "unchanging". Note that since the previous report there has been no basin-wide update of land cover that allows us to determine trends beyond 2011. Basin-wide land use/land change data has not been updated since the last report.

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

Lake Superior
Status: Good
10-Year Trend: Unchanging
Long-term Trend (2001-2011): Undetermined
Rationale: Land cover change was assessed only on the U.S. side of the basin as there were no 2011 era land use data available for Canada in the Lake Superior basin. Forest land in the U.S Lake Superior basin decreased by approximately 400 km^2 or 0.93% of the watershed, but this conversion was predominately to grass/shrub land cover, which increased by 384 km^2 or 0.089%. With 93% natural land cover, the status is "Good" and the trend is "Unchanging". Land use/land change data has not been updated since the last report.

Lake Michigan
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (2001-2011): Undetermined
Rationale: Forest land cover decreased by 600 km^2 and developed land increased by 450 km^2. However the predominant transition of forest was to grass/shrub, whereas the increase in developed land came from conversion of agricultural land. For this reason the trend is "Unchanging". With 42% of the watershed in agriculture and 11% in developed land, the status is "Fair". Land use/land change data has not been updated since the last report.

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (2001-2011): Undetermined
Rationale: Land cover data were unavailable for the Canadian portion of the basin outside of the SOLRIS coverage. Forested land cover decreased by 450 km^2, with much of this converting to grass/shrub. Developed land increased by 117 km^2 but agricultural lands decreased by 90 km^2. With essentially no net change between developed and natural land covers, the trend is unchanging. With 42% of the watershed in agriculture and 8% in developed land, the status is "Fair". Land use/land change data has not been updated since the last report.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend: Unchanging
Long-term Trend (2001-2011): Undetermined
Rationale: Land cover data were unavailable for the Canadian portion of the basin outside of the SOLRIS coverage. Lake Erie's largest land use change was an 458 km^2 increase in developed land, largely due to the conversion of

agricultural land, which decreased by almost 300 km^2. Forest land decreased by 225 km^2, primarily by conversion to developed land or agriculture. With 62% of the watershed in agriculture and 17 % in developed land, the status is characterized as Poor and the trend is Unchanging. Land use/land change data has not been updated since the last report.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Fair
10-Year Trend: Unchanging
Long-term Trend (2001-2011): Undetermined
Rationale: Land cover data were unavailable for the Canadian portion of the basin outside of the SOLRIS coverage. The largest land use change was a 300 km^2 increase in developed land, due to conversion from agricultural land, and to a lesser degree, forestland. Forest land cover decreased overall by 100 km^2. With 42% of the watershed in agriculture and 11% in developed land, the status is characterized as Fair and the trend is Unchanging. Land use/land change data has not been updated since the last report.

Status Assessment Definitions
The following status classifications have been employed, which are most applicable to watersheds at multiple spatial scales (Table 1). It should be noted that land use is not the direct cause of particular water quality or habitat condition. Degradation or maintenance of water quality is associated with variables that change as a result of land use. Consequently, 'status' should be evaluated in terms of 'risk of degradation' (low, medium, or high), which would be analogous to State of the Great Lakes' (previously known as SOLEC's) "Good", "Fair", and "Poor" condition classes. The following table summarizes a provisional status assessment for the different land use classes at the watershed scale based on literature values. For reporting purposes a consistent watershed unit should be identified. It is recommended that the Great Lakes Hydrography Dataset developed under the Great lakes Aquatic Habitat Framework (GLAHF) be used for future reporting (Forsyth et al., 2016).

Trend Assessment Definitions
Since land use data are updated at frequencies of five or more years, land use trend metrics cannot currently be updated on a three-year cycle. In December 2018 however, US National Land Cover Database 2016 will be released. This database will have several advantages over previous ones – it will harmonize land cover and impervious surface products for 7 product dates between 2001 and 2016 including 2001, 2004, 2006, 2008, 2011, 2013, and 2016, making it amenable to a three-year cycle. It will also have improved polygon rendering to enhance the accuracy of multi-year comparisons.

In Canada, the Ontario Land Cover Compilation V2.0 (OLCC) is a rationalized land cover dataset that combines three separate land cover databases (Far North Land Cover v1.4, Southern Ontario Land Resource Information System (SOLRIS) v1.2, and the Provincial Land Cover 2000 Edition). From the metadata abstract: the OLCC is "… a standardized product with a pixel resolution of 15 metres, a coordinate system of Ontario Lambert Conformal Conic and a class structure of 29 land cover classes. However, the standardized classification has been accomplished at the expense of the more detailed class structures contained within the source land cover products. Where possible, the original land cover products should be used for analysis. V2.0 is an update to OLCC v1.0. In this version the Far North Land Cover component has been updated and extends farther south into the Area of Undertaking and into Manitoba than the previous version." (*https://www.javacoeapp.lrc.gov.on.ca/geonetwork/srv/en/main.home*). The three datasets compiled in the OLCC are of different vintages, confounding trend assessments.

As noted in the previous report, the amount of developed land cover is not likely to decline substantially through time. Therefore, the trend target should reflect the desire to achieve "no net loss" of wetland and forest cover types, and where possible, a decrease in the area of impervious surface cover or use of ameliorating management practices (Table 2).

A benchmark year of 2001 is recommended, representing the earliest year for which reliable and consistent data across the US-Canada border are available. When new land cover data are available for the Great Lakes Basin in Ontario, a basin-wide change analysis can be conducted (Table 2).

At the lake and basin levels, trends can be expressed in terms of a 'transition matrix' that tracks the change in land use from one type to another (by percentage from 2001 or the most recent assessment period, whichever is available; Table 3). Positive percentages in any of the green-shaded cells are classified as improving trends. Positive percentages in any of the red-shaded cells are classified as deteriorating trends. Scores in the grey-shaded cells are neutral

trends. Desirable trends for land change at the lake or basin level include no net loss of wetlands, grasslands or forest and/or no change or a reduction from natural lands to developed or agricultural lands. Undesirable trends would be a reduction from natural lands to developed or agricultural lands (Table 2).

Endpoints and/or Targets
In the absence of an established target that reflects regional conditions, a provisional operational benchmark can be used (e.g., percentage of land in each class in 2001; Table 1). Yet, these data don't reflect specific land management practices, which can mitigate the effects of land use on water quality and the Great Lakes ecosystem (Table 3).

The target should reflect the desire to achieve "no net loss" of 'natural' land covers (i.e., wetland and forest cover types), and where possible, a decrease in the area of impervious surface cover, or use of ameliorating management practices. No specific absolute target values are available at this time that reflect regional goals; however, literature supports the notion that water quality degradation (resulting in degraded biological communities) is observed when between 2% and 10% of a watershed is covered by impervious surfaces (Morse et al. 2003, Paul and Meyer 2001), and when row crop agriculture in a watershed exceeds between 20-60% (depending on regional conditions, crop types, management types, and watershed size (Wang et al. 1997, Cuffney et al. 2000, Donohue et al 2006). Strongly coherent biological thresholds were observed for coastal wetlands across multiple assemblages, verifying the ranges of these thresholds (Kovalenko et al. 2014).

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 directly reflective of the health of the Great Lakes ecosystem

Ecosystem Objective
Sustainable development is a generally accepted land use goal for the Great Lakes Basin.

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 the current percentages (estimated from areal coverage (km^2) of various types of land cover and the degree of change over time relative to a benchmark set in 2001. Land cover types may include but are not exclusive to the following: area of water, developed land, agriculture, forest (deciduous, evergreen, mixed), shrub / scrub, grassland, and wetland (emergent, forested). Due to differences in the definitions of more refined classes between U.S. and Canadian datasets, these classes represent the lowest common denominator for a seamless map product. As data products are improved through time, it will be possible to detect trends in more refined land covers (e.g., types of agriculture, types of natural lands). It is recommended that land use and land cover estimates be based on resolution of 30-m (e.g. Landsat) pixels, which is standard for several satellite-based products, and has already been used for the Great Lakes basin. Landsat is the basic input for CCAP, Vegetation Change Tracker (VCT) and LandTrendr, all of which do land use change analyses.

Temporally comparable (i.e., 2001-2002 era) datasets derived from U.S. National Land Cover Dataset (NLCD) and Ontario Land Cover Compilation v.2.0 were merged into a single binational land cover product by the Great Lakes Aquatic Habitat Classification project (Wang et al. 2015) (GLAHF; http://ifr.snre.umich.edu/projects/glahf), by cross-walking land cover classes to a common denominator. A more contemporary product was created utilizing 2011 (NLCD) and 2012 (SOLRIS v2.0) data. The northern portion of the basin in Canada, however, lacks newer data and thus is mapped using the 2000 era data (PLO).

Ecological Condition
For the previous analysis, a common land cover classification was developed to allow an integrated comparison of land use in both Canada and the U.S. This involved integrating the detailed but distinct classifications of the U.S. system (24 land use classes as delineated by Wolter et al. 2006) with the Canadian system (The Ontario Ministry of Natural Resources' Ontario Provincial Land Cover, consisting of 27 (in 1990) or 28 (in 2000) classes). The resulting unified assessment consisted of six land classes (Developed, Agriculture, Grassland/ Shrubland, Forest, Wetland,

and Water (Ciborowski et al. 2011)). Using this common land cover classification for the year 2000, the total and proportional amounts of each land cover class was calculated by lake and across the Great Lakes Basin.

In the present assessment, temporally comparable (i.e., 2000-2002 era) datasets derived from U.S. National Land Cover Dataset (NLCD) and Ontario Land Cover Compilation v.2.0 were merged into a single binational land cover product by the Great Lakes Aquatic Habitat Framework project (GLAHF; *http://ifr.snre.umich.edu*) (Wang et al. 2015). Subsequently, a more contemporary product was created utilizing 2011 NLCD (Homer et al. 2015) and 2012 (SOLRIS v2.0) data. The SOLRIS land cover dataset however, is not a complete coverage of the Canadian side of the Great Lakes Basin - it excludes approximately 175,000 km^2 of the largely forested northern regions of the Lake Superior and Lake Huron watersheds. This represents 34% of the land area within the Great Lakes basin watershed. It is expected that outside of forest harvest activities, this region has experience relatively little land use change. The assessments of land cover change presented below however, only reflect that portion of the basin where 2001 and 2011 data are directly comparable. An updated US NLCD classification will become available in December 2018.

Over the extent of our study area there was a net conversion of 393 km^2 from natural to human-modified land cover. This change came largely at the expense of forest land, which decreased by 1780 km^2. The area of the basin in agricultural land use also decreased by 948 km^2. Increases were seen in the amount of developed land (1341 km^2) and grass/shrub land cover (1257 km^2). Rates of land use change provide an important integrated indicator of the degree and location of both loss and gain of natural lands, representing increases and reductions in the risks of degradation. These latest analyses (2000-2010) reflect a growing trend of increasing developed lands, at the expense of both agricultural and forest lands.

As might be expected, the large variations in land cover across the Great Lakes noted in the previous report has remained constant, with the Lake Superior basin continuing to be predominately forested (Figure 1) and Lake Erie predominantly agricultural (Figure 4). Forest and Agricultural land uses are more evenly distributed in lakes Michigan and Ontario (Figures 2 and 5). This large variation in land use among lakes reflects the underlying climatic and soil gradients across the Great Lakes Basin that have historically constrained the conversion of the native vegetation (forest or grassland) to agricultural land use.

The distribution of land use classes for each Great Lake is shown in Figures 1-5. The greatest change toward human-modified land use uses on both an absolute and percentage basis occurred in the Lake Erie basin, with a net change of 165 km^2. This change was entirely due to increases in developed lands, which increased by 458 km, largely due to the conversion of agricultural (-292 km^2) and forested (-225 km^2) lands. Similar changes occurred in the Lake Ontario basin, which saw a 298 km^2 increase in developed land, again due to loss of agricultural land (-226 km^2) and forest land (-106 km^2). In fact, with the exception of Lake Superior, all lakes experienced declines in agricultural land and increases in developed lands (Table 4). The row totals in Table 4 show the total area in a land use class in 2001, whereas the column totals show the total area by class in 2010. The barren land class was too uncommon to show in the figures but included in Table 4 for completeness.

Other Spatial Scales

This sub-indicator pertains primarily to risk of degradation of the coastal margins and nearshore waters. The importance of land use condition (especially as a source of nutrients and contaminants) declines with increasing distance away from the coastal margin since substances are typically transported by the water contributed by tributaries.

The importance of land use condition (especially as a source of nutrients and contaminants) is greatest at shorelines and coastal margins, and declines with increasing distance away from the shore since substances are typically transported by the water contributed by tributaries. Natural land cover is an indicator of good conditions because it incorporates nutrients into biomass and slows the rate of water runoff into the lakes, together with materials (sediments, pollutants) that the water transports.

Linkages

The link between land use and the climate is complex. First, land cover, as shaped by land use practices, affects the global concentration of greenhouse gases. Second, while land use change is an important driver of climate change, a changing climate can lead to changes in land use and land cover. For example, farmers may shift from their customary crops to crops that will yield higher economic return under changing climatic conditions -- and higher temperatures can affect vegetation cover as well as water needed for irrigation. The understanding of the interactions between climate and land use change is improving but continued scientific investigation is needed.

This sub-indicator also relates to the Forest Cover sub-indicator, and indirectly to Tributary Flashiness, which is influenced by conversion to human-modified land covers.

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 Limitations

The most significant data limitation is the lack of temporal concordance among different land classification products. This confounds analyses of both status and trends across the Great Lakes basin. New satellites, such as the European Space Agency's Sentinel program or Digital Globe's satellite-based stereo imagery have the potential to produce much more highly resolved and frequent imagery. Coupled with advanced land use classification routines such as Collect Earth Online (*https://southern-blues-dev.appspot.com/*), these data limitations will likely be overcome in the next monitoring cycle.

The land cover sub-indicator provides a measure of land use change, but not of the change in quality of the land use. For example, conversion of a highly manipulated, chemical-intensive agricultural area to an urban area, particularly one that is well-planned and utilizes environmental and resource conservation management plans, may result in less stress to the ecosystem; conversion of a natural forest to an intensively managed forest may similarly result in a risk to the receiving waters. In contrast, urban development on excavated land, landfill or other contaminated sites may also be positive changes. Lastly, arguably the best measure of the impact of urban land use is the amount of impervious surface, while the proportion of agricultural land that is tile-drained is an important measure of the impact of agricultural land. However, impervious surface cover extent has yet been mapped across Ontario, and maps of tile drainage are not widely available in either the U.S. or Canada. When such data become available, it would be advisable to use them for future trend assessments.

Finally, the detection of restoration from non-natural to natural would be very difficult to determine using remote sense information, since the scale of individual restorations would not likely be detected at the resolution of ordinary satellite imagery. Further, features such as bare rock (natural) are difficult to distinguish from "parking lot" or "big box store roof except at very high spatial resolution (e.g., 3-m pixels). The error rate of such an analysis would greatly exceed the number of actual hectares of restored area.

Clarifying notes regarding geographic coverage in the previous report – the SOLRIS land cover dataset is not a complete coverage of the Canadian side of the Great Lakes Basin. It excludes the largely forested northern regions of the Lake Superior and Lake Huron watersheds, north of N 45.88334 and west of W 83.10000.

Additional Information

As natural lands become converted to agricultural or urban uses, the ecosystem products and services provided by those cover types such as timber, water storage and purification, wildlife habitat, carbon storage, recreation, and aesthetic beauty, etc. are changed. The loss of natural habitat, particularly forest, can also have profound economic impact on communities that rely on the forest for food and economic development. Furthermore, conversion of natural land cover to urban and agricultural lands leads to increased runoff and associated increased inputs of sediments, nutrients, and contaminants to inland waters and the Great Lakes (Seilheimer et al., 2013; Wolter et al., 2006). High rates of land conversion place stress on the natural ecosystem and are typically associated with inefficient land uses, such as those resulting from urban sprawl. Other things being constant, high conversion rates are associated with

rapid rates of urban sprawl, which is economically inefficient and displaces either natural land that serves other biological purposes in the ecosystem or agriculture, which in turn may take the place of natural land. The conventional pattern of land conversion has been for urban growth to displace agricultural lands, which, in turn, expand into remaining natural land types. Urban development also encroaches on natural lands, but often displaces agricultural lands on the outskirts of urban areas. Growth in human population and resource consumption are drivers for more development, which displace both agricultural and natural lands. Other things being constant, high conversion rates are associated with rapid rates of development. The spatial pattern of land use conversion affects wildlife habitat and associated wildlife populations and communities. For example, 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. 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 due to the increased number of stakeholders that must be involved in land management decisions (Pijanowski and Robinson, 2011).

Note: There are coarser-scale land cover and use data available for Canada, e.g., "NACP Forest Age Maps at 1-km Resolution for Canada (2004) and the U.S.A. (2006)" (http://daac.ornl.gov/NACP/guides/NA_Tree_Age.html)

Some additional categories in the future could be:

Major change in natural land cover types (%). If the magnitudes are defined in terms of known thresholds, this index could be defined in terms of particular cells in Table 3 as either positive or negative indicators. Based on Table 2, the expectation is that positive values across these cells would generally lead to improving conditions while negative values would lead to deteriorating conditions. This metric would be measureable with current tools and could be tied to risk of degradation.

Major change to anthropogenic non-urban or industrial land use (restorable) (%). If the magnitudes are defined in terms of known thresholds, this index could be defined in terms of particular cells in Table 3 as either a positive or negative indicator. Based on Table 2, the expectation is that positive values across these cells would generally lead to deteriorating conditions. Small changes (in area) may not be captured with prevailing 30-m resolution data, therefore, most increases in impervious surface cover are likely to remain undetected. With more highly resolved data (e.g., 10-m or less resolution) in urban areas this may be a feasible in the future.

Issues with data registration and classification criteria between 1992 and 2000-era data precluded meaningful land cover change analysis in the 2011 report, as noted in Ciborowski 2011.

Acknowledgments

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Members of the Great Lakes Environmental Indicators project - Gerald L. Niemi (Senior PI; NRRI, University of Minnesota Duluth), Nicholas P. Danz (University of Wisconsin – Superior), and Thomas Hollenhorst (U.S. EPA, Mid-Continent Ecology Division National Health and Environmental Effects Research Laboratory, Duluth, MN 55804) contributed to formation of the research group that identified the need for this database. The State of the Great Lakes (previously known as SOLEC) coordinators Rob Hyde, Nancy Stadler-Salt, Stacey Cherwaty-Pergentile (Environment and Climate Change Canada, Burlington, ON), and Paul Horvatin and Karen Rodriguez (U.S. EPA GLNPO, Chicago, IL) provided the impetus for developing the concept paper on land cover that allowed assessment of the status of land cover within the Great Lakes basin, and to infer the potential impact (risk of degrada-

tion) of land cover and land cover change on Great Lakes ecosystem health. Catherine Riseng was principal investigator for the Great Lakes Aquatic Habitat Framework that developed and now hosts the Great Lakes Hydrography Dataset.

The project on which these data were based was originally funded by the U.S. Environmental Protection Agency Science to Achieve Results (STAR) Estuarine and Great Lakes (EaGLe) program through funding to the Great Lakes Environmental Indicators (GLEI) and Reference Condition projects (U.S. EPA Agreements EPA/R-8286750 and EPA/R-82877701, respectively). More recent work was supported by the second stage GLEI-II Indicator Testing and Refinement project funded by a Great Lakes Restoration Initiative grant from the U.S. Environmental Protection Agency Great Lakes National Program Office to Lucinda.B. Johnson (GL-00E00623-0). Although the research described in this work has been partly funded by the U.S. EPA, it has not been subjected to the agency's required peer and policy review and therefore does not necessarily reflect the views of the agency and no official endorsement should be inferred. The GLAHF project was funded by the Great Lakes Fisheries Trust.

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Data Sources

The integrated and reclassified NLCD 2011 and SOLRIS 2012 land use/land cover data were obtained from the Great Lakes Aquatic Habitat Framework; *http://ifr.snre.umich.edu*

The following credits for land cover circa 2000 and 2010 were posted with on the GLAPH metadata page:

- National Land Cover Dataset, 2001 *v11 http://www.mrlc.gov/nlcd01_data.php*;
- The 2000 Provincial Land cover Ontario PLO *https://www.javacoeapp.lrc.gov.on.ca/geonetwork/srv/en/main.home;*

- 2000 Southern Ontario Land Resource Information System (SOLRIS) v 1.2 *https://www.javacoe-app.lrc.gov.on.ca/geonetwork/srv/en/main.home;*
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List of Tables

Table 1. Provisional thresholds in area of land cover (by watershed) representing degrees of risk of degradation of water/habitat quality of receiving waters. Such measures can be used in the absence of trend data. Percentages of natural lands are not shown due to the absence of literature showing thresholds of these cover types associated with particular risk or condition.
*will be derived from existing literature

Table 2. Predicted trends in water quality due to changes in land use/land cover (by watershed).

Table 3. Proposed template of a matrix summarizing the change in land cover type (percent) from a particular class (listed as rows) to a particular class (listed as columns). Cells would be populated with data from a change detection analysis.

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Table 4. Transitions among land use/land cover classes between 2001 (rows) and 2011 (columns); values are area in square kilometres.
Data Source: 2011 NLCD and 2012 SOLRIS; integrated classification by Wang et al. 2015; regions north of the SOLRIS demarcation line represent 2001-era data

List of Figures

Figure 1. Distribution of land use/land cover across the Lake Superior basin.
Source: GLAHF 2001 are an integration of the National Land Cover Dataset (NLCD) and the Ontario Land Cover Compilation v 2.0 data from 2001, whereas GLAPH 2011 incorporate 2011 NLCD and 2012 SOLRIS data (Wang et al. 2015); the GLAPH 2011 dataset does not cover the area north of the demarcation line

Figure 2. Distribution of land use/land cover across the Lake Michigan basin in 2011.
Source: 2011 NLCD and 2012 SOLRIS; integrated classification by Wang et al. 2015; regions north of the SOLRIS demarcation line represent 2001-era data

Figure 3. Distribution of land use/land cover across the Lake Huron basin.
Source: GLAHF 2001 are an integration of the National Land Cover Dataset (NLCD) and the Ontario Land Cover Compilation v 2.0 data from 2001, whereas GLAPH 2011 incorporate 2011 NLCD and 2012 SOLRIS data (Wang et al. 2015); the GLAPH 2011 dataset does not cover the area north of the demarcation line

Figure 4. Distribution of land use/land cover across the Lake Erie basin in 2011.
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Figure 5. Distribution of land use/land cover across the Lake Ontario basin.
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Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

	Risk of Degraded Water/Habitat Quality		
Land Use Class	Low (‘good’)	Moderate (‘fair’)	High (‘poor’)
Developed Land	<6%	6-27%	>27%
Agricultural/Rural	<20%*	20-50%*	>50%

Table 1. Provisional thresholds in area of land cover (by watershed) representing degrees of risk of degradation of water/habitat quality of receiving waters. Such measures can be used in the absence of trend data. Percentages of natural lands are not shown due to the absence of literature showing thresholds of these cover types associated with particular risk or condition.
*will be derived from existing literature

	Predicted Trends in Land Use and Land Cover		
Land Use Class	Decrease from Previous	No change	Increase from Previous
Developed Land	Improving	Unchanged	Deteriorating
Agricultural/Rural	Improving	Unchanged	Deteriorating
Forest	Deteriorating	Unchanged	Improving
Grassland	Deteriorating	Unchanged	Improving
Wetland	Deteriorating	Unchanged	Improving

Table 2. Predicted trends in water quality due to changes in land use/land cover (by watershed).

		Class at the end of a reporting period				
		Developed	**Agricultural**	**Forest**	**Grassland**	**Wetland**
Class at the beginning of a report-ing period	**Developed**					
	Agricultural					
	Forest					
	Grassland					
	Wetland					

Table 3. Proposed template of a matrix summarizing the change in land cover type (percent) from a particular class (listed as rows) to a particular class (listed as columns). Cells would be populated with data from a change detection analysis.

Lake Erie	Agriculture	Barren	Developed	Forest	Grass/Shrub	Wetland	Water	Totals
Agriculture	46314	38	487	192	8	95	18	**47151**
Barren	23	172	8	1	3	2	4	**213**
Developed	214	1	12234	23	0	13	3	**12488**
Forest	200	7	149	10320	55	123	7	**10860**
Grass/Shrub	4	2	27	5	886	2	1	**927**
Wetland	99	2	38	92	2	3939	6	**4178**
Water	4	1	3	2	0	16	615	**641**
Totals	**46858**	**222**	**12945**	**10635**	**955**	**4190**	**653**	**76457**

Lake Huron	Agriculture	Barren	Developed	Forest	Grass/Shrub	Wetland	Water	Totals
Agriculture	26284	34	270	276	25	122	13	**27024**
Barren	31	175	4	1	5	4	5	**227**
Developed	190	1	4727	48	0	21	3	**4990**
Forest	287	10	71	16682	474	172	9	**17706**
Grass/Shrub	11	3	4	95	2538	1	1	**2652**
Wetland	124	5	29	147	5	10690	10	**11010**
Water	7	3	2	5	0	18	852	**888**
Totals	**26934**	**231**	**5107**	**17255**	**3048**	**11029**	**893**	**64497**

Lake Michigan	Agriculture	Barren	Developed	Forest	Grass/Shrub	Wetland	Water	Totals
Agriculture	37112	38	308	11	34	15	14	**37532**
Barren	15	379	7	0	3	1	10	**415**
Developed	0	0	11486	0	0	0	0	**11486**
Forest	33	17	61	34731	768	7	4	**35621**
Grass/Shrub	25	9	46	281	4581	5	2	**4949**
Wetland	3	5	24	1	8	23086	10	**23138**
Water	1	6	2	1	1	12	3448	**3470**
Totals	**37189**	**453**	**11934**	**35024**	**5395**	**23127**	**3488**	**116611**

Lake Ontario	Agriculture	Barren	Developed	Forest	Grass/Shrub	Wetland	Water	Totals
Agriculture	22617	23	497	429	10	193	15	**23783**
Barren	32	110	5	1	1	1	1	**151**
Developed	263	2	5393	79	0	35	5	**5777**
Forest	442	7	120	16077	43	305	16	**17010**
Grass/Shrub	6	3	8	28	2102	6	3	**2156**
Wetland	186	3	49	275	3	6217	12	**6745**
Water	9	2	4	15	0	27	1216	**1272**
Totals	**23556**	**149**	**6075**	**16904**	**2159**	**6783**	**1267**	**56893**

Lake Superior	Agriculture	Barren	Developed	Forest	Grass/Shrub	Wetland	Water	Totals
Agriculture	1411	1	2	2	5	1	0	**1420**
Barren	0	153	1	0	6	1	5	**167**
Developed	0	0	1535	0	0	0	0	**1535**
Forest	10	13	12	25303	895	7	1	**26241**
Grass/Shrub	2	6	3	529	472	2	0	**1015**
Wetland	1	3	3	4	20	11346	1	**11378**
Water	0	10	0	0	3	5	1379	**1397**
Totals	**1423**	**188**	**1556**	**25839**	**1400**	**11361**	**1387**	**43153**

Table 4. Changes in area of land use/land cover classes between 2001 and 2011. Row totals show the total area in a land use class in 2001, column totals show the total area by class in 2010. Values are area in square kilometres. Data Source: 2011 NLCD and 2012 SOLRIS; integrated classification by Wang et al. 2015; regions north of the SOLRIS demarcation line represent 2001-era data.

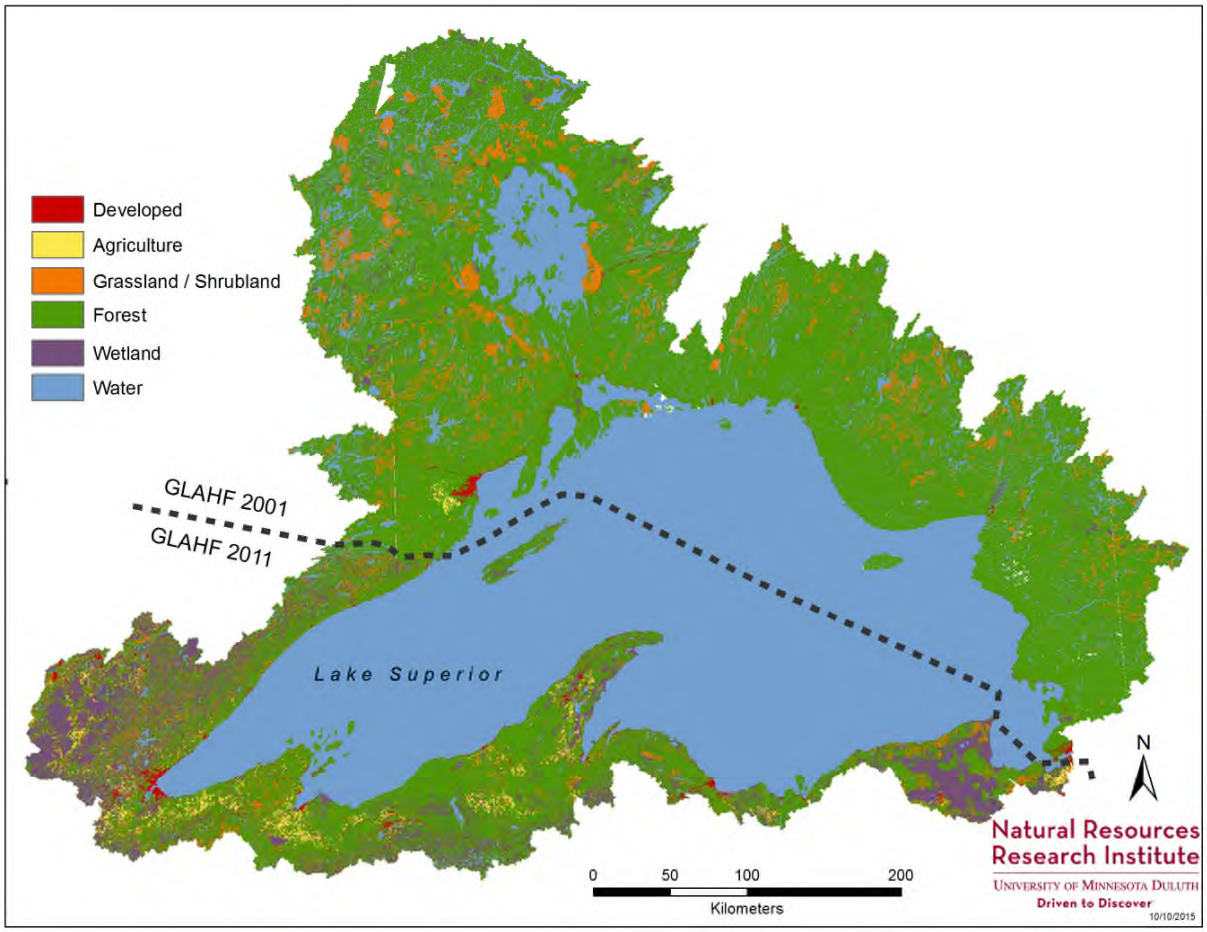


Figure 1. Distribution of land use/land cover across the Lake Superior basin.
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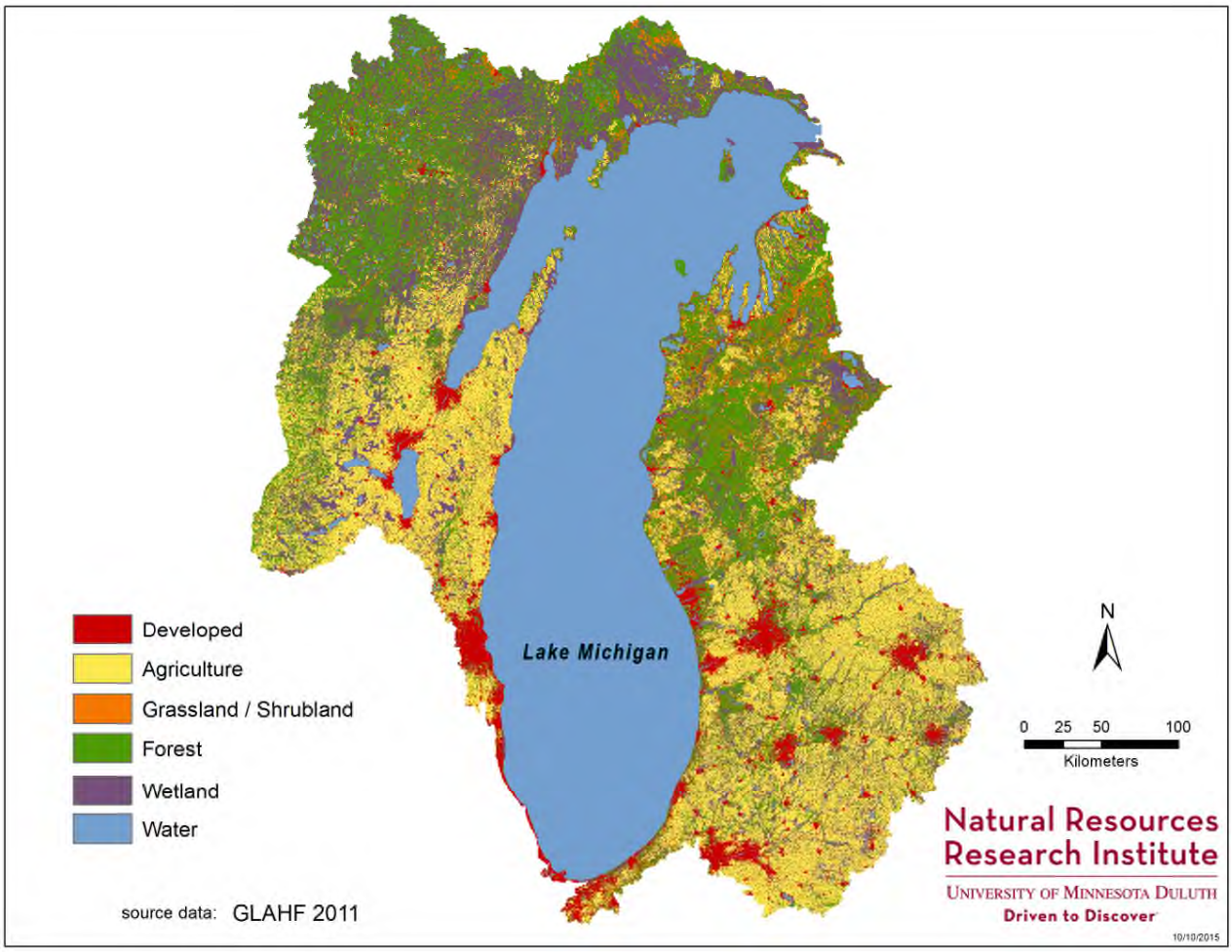


Figure 2. Distribution of land use/land cover across the Lake Michigan basin in 2011.
Source: 2011 NLCD and 2012 SOLRIS; integrated classification by Wang et al. 2015; regions north of the SOLRIS demarcation line represent 2001-era data

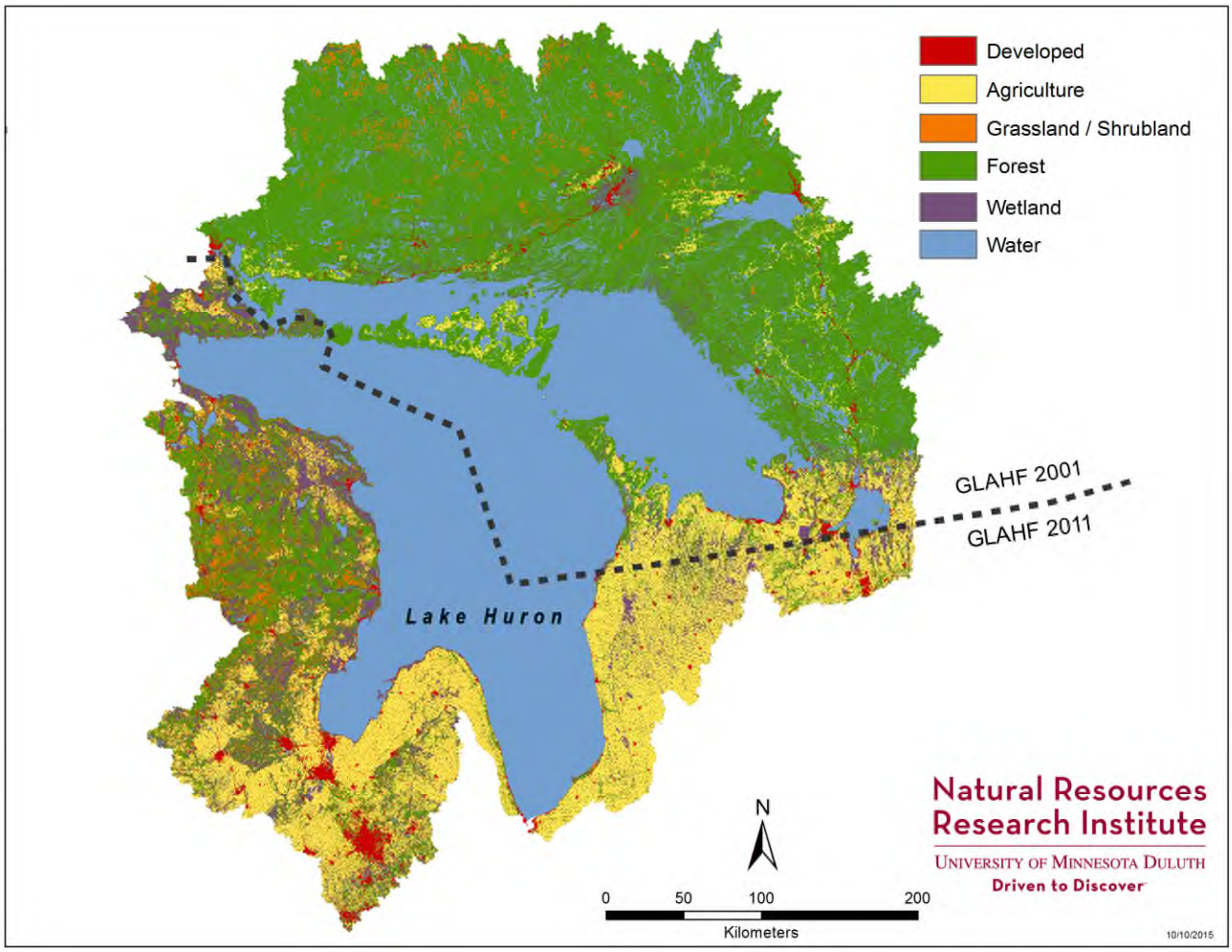


Figure 3. Distribution of land use/land cover across the Lake Huron basin.
Source: GLAHF 2001 are an integration of the National Land Cover Dataset (NLCD) and the Ontario Land Cover Compilation v 2.0 data from 2001, whereas GLAPH 2011 incorporate 2011 NLCD and 2012 SOLRIS data (Wang et al. 2015); the GLAPH 2011 dataset does not cover the area north of the demarcation line

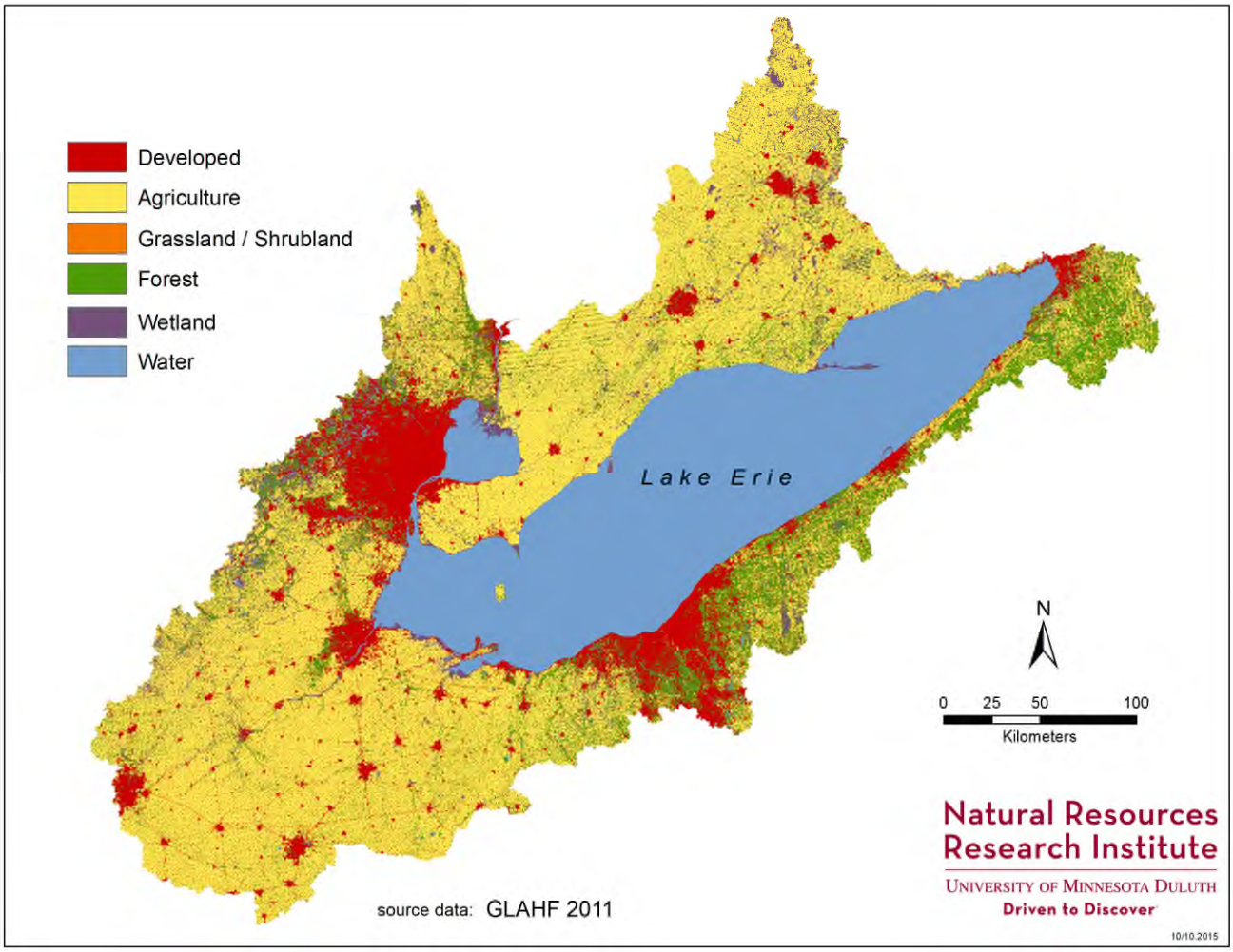


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Source: 2011 NLCD and 2012 SOLRIS; integrated classification by Wang et al. 2015; regions north of the SOLRIS demarcation line represent 2001-era data)

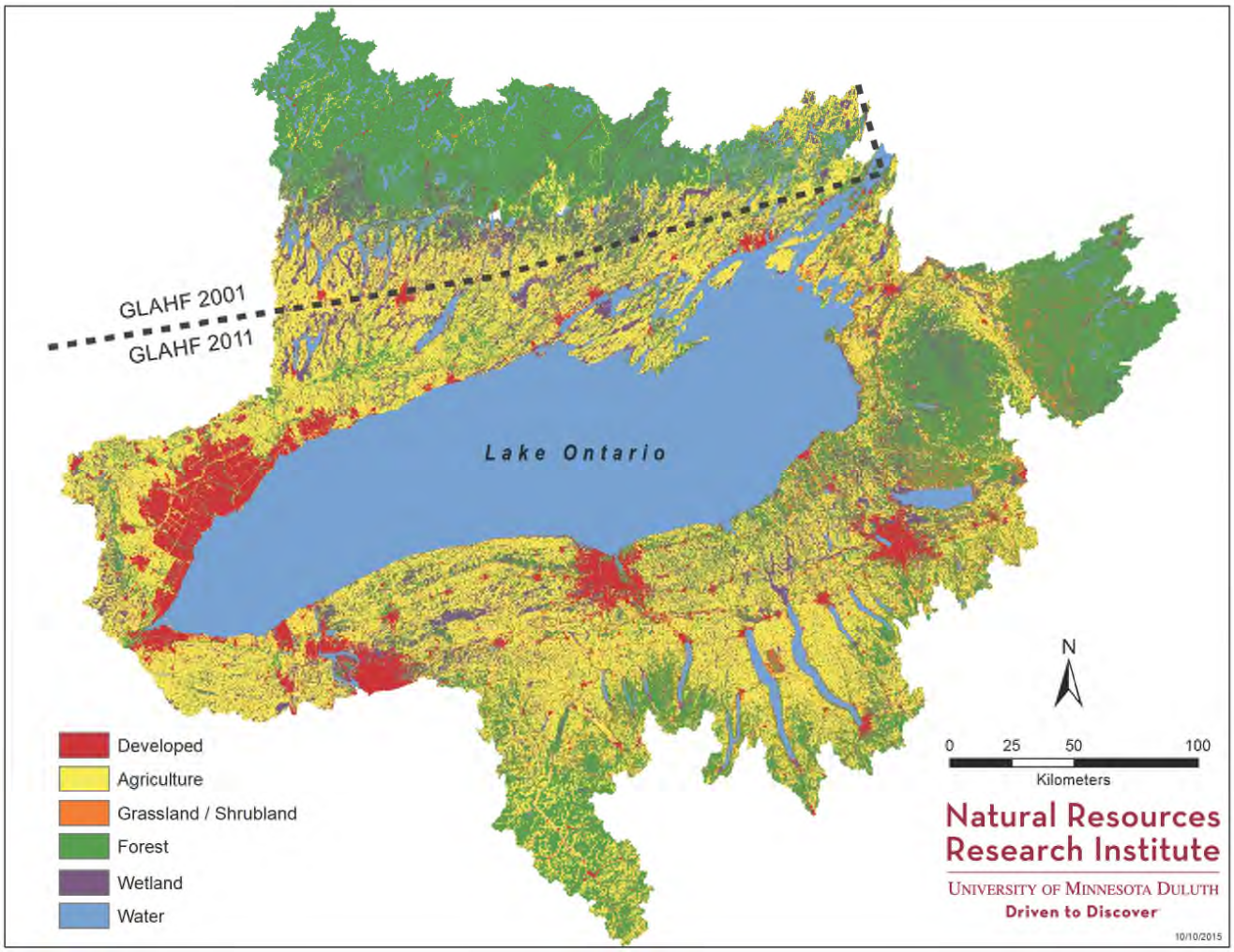


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Sub-Indicator: Hardened Shorelines

Overall Assessment
Status: **Undetermined**
Trends:
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: An overall assessment is not possible as information allowing a direct comparison to previous hardened shoreline sub-indicator status and trend are only available for the Lake Ontario shoreline from the 2017 reporting cycle. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for at least a partial (U.S. portion) basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle.

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

Lake-by-Lake Assessment
Lake Superior
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Lake Superior is not being assessed for the current reporting cycle. Available data does not allow a direct comparison to previous hardened shoreline sub-indicators. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for a basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle.

Lake Michigan
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Lake Michigan is not being assessed for the current reporting cycle. Available data does not allow a direct comparison to previous hardened shoreline sub-indicators. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for a basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle.

Lake Huron (including St. Marys River)
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Lake Huron is not being assessed for the current reporting cycle. Available data does not allow a direct comparison to previous hardened shoreline sub-indicators. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for a basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle.

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Undetermined
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: Lake Erie is not being assessed for the current reporting cycle. Available data does not allow a direct comparison to previous hardened shoreline sub-indicators. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for a basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle.

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Poor
10-Year Trend (2001/2002 and 2015)*: Deteriorating
Long-term Trend: Undetermined
Rationale: Updated (2015) shoreline classification datasets for Lake Ontario, not including connecting channels, indicate that approximately 68.5% of the shoreline reaches are in the minor protection or no protection category which is below the poor threshold of 70%. In other words, Lake Ontario has approximately 30% of its shoreline in a heavily or moderately hardened state/condition. The long term trend of Lake Ontario appears to be deteriorating; however, while the short term trend also appears to be deteriorating, there is some uncertainty in the data which could also suggest that the short term trend may be unchanging. While the percent of shoreline in the no protection category was comparable to the previous State of the Great Lakes report update (2001-2002), reductions in the unclassified category were offset by increases in the minor protection, moderate protection, and heavy protection categories suggesting a potential trend towards increased overall shoreline hardening in some areas. However, the redistribution of the proportions of classified shoreline types may be attributed to the increased availability of higher resolution aerial photographs than what were available during the more recent review of this sub-indicator. This allows for a more detailed delineation of the shoreline to be performed. There is uncertainty in the trend analysis due to variations in input datasets as discussed further below. This assessment does not include the Niagara or St. Lawrence Rivers.

Lake Ontario is not being updated for the current 2019 reporting cycle. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for a basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle. * Note: Although not a typical trend determination, the 2001/2002 and 2015 data analysis will be used as a proxy for the 10-year trend determination and serve as an estimate for this shorter term period.

Status Assessment Definitions
Good: >80% of the shoreline reaches have minor to no protection; in other words, 20% or less of the shoreline is moderately or highly protected
Fair: 70-80% of the shoreline reaches have minor to no protection; in other words, 20-30% of the shoreline is moderately or highly protected.
Poor: < 70% of the shoreline reaches have minor to no protection; in other words, 30% or more of the shoreline is moderately or highly protected.

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.

In future reports, trend determination will be based on no net increase in the percent of shoreline in the highly protected and moderately protected categories. 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. 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 shore protections, such as sheet piling, rip rap, and other erosion control shore protection structures.
- To infer the potential harm to aquatic-dependent life, water quality, and natural processes from conditions created by shore protections.

Ecosystem Objective
Shoreline conditions should be healthy to support aquatic and terrestrial plant and animal life, including the rarest 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.

Measure
The amount (kilometres) of shoreline that has been hardened through construction of sheet piling, rip rap and other erosion control shore protection structures. Shoreline reaches are categorized using descriptions from the baseline shoreline classification dataset and include highly protected (70-100% hardened), moderately protected (40-70% hardened), minor protection (15-40% hardened), no protection (< 15% hardened), non-structural protection, and unclassified.

Note: measure does not include artificial coastal structures that are extend out into the waters, such as jetties, groins, breakwaters, piers, etc.

Ecological Condition
An interagency effort is currently in process to generate a baseline shoreline dataset for the United States portions of the remaining Great Lakes. It is expected that this dataset will allow for an assessment of change to hardened shorelines on the U.S. portion of the Great Lakes for the next (post-2019) reporting cycle. The effort will utilize the methodology from the 2017 reporting cycle. Due to the scope of this analysis, there will be no updates to this sub-indicator for the 2019 reporting cycle. In order to obtain a complete basin wide assessment of changes to the hardened shorelines, an effort must be made to digitize and create a baseline dataset for the Canadian portion of the Great Lakes. Again, due to the scope of this effort, a full assessment will not be available until after the 2019 reporting cycle.

As stated in the 2017 report, there is limited documentation on specific shoreline hardening objectives, particularly at the basin wide and lake wide scales. The proposed endpoint values for a hardened shoreline status assessment provide a descriptive point of reference using the baseline Great Lakes (previously known as SOLEC) estimates of the extent and intensity of shoreline hardening. Various environmental services can be impacted by shoreline hardening including changes or reductions in aquatic habitat, alterations in sediment transport, and changes in nearshore groundwater-lake interactions (see Province of Ontario, 2001). There are a variety of challenges in defining appropriate target values regarding shoreline hardening. In particular, a refined assessment should reflect the differing quality and quantity of environmental services being provided (or not provided) by differing shoreline locations (e.g. pollution filtration, fish habitat, etc.) and weight the necessity and amount of the shoreline services required to achieve established ecosystem goals relative to the extent and impact of various shoreline hardening activities. However, the ecological services provided by natural shorelines and the impacts of hardened shorelines are difficult to measure as they often relate to many complex, long-term, and interdependent ecological processes (such as pollution filtration and sediment transport), in addition to more immediate and observable effects such as habitat and habitat loss. There are also variations in the extent to which certain types of shoreline hardening activities actually impact various ecological services based on the age, quality, and design characteristics of the shoreline structures. 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 relative sensitivity to shoreline hardening on each lake. The selected category ranges account for the fact that some shoreline hardening already exists on the Great Lakes and is likely to be maintained into the future. The trend assessment captures the relative change in the percent of shoreline with >40% hardening.

For the purpose of the 2017 report and assessment, an overall undetermined reference value has been selected for the basin wide assessment due to the lack of a standardized dataset on many of the lakes that can be directly compared to the baseline conditions established for the State of the Great Lakes (previously known as SOLEC) hardened shoreline sub-indicator. Where updated datasets do exist, they tend to be limited in geographic scope (i.e. they do not cover a full lake basin) or there are issues in matching the existing hardened shoreline assessment categories. The baseline conditions, as represented in the 2009 and 2011 Great Lakes/SOLEC hardened shoreline indicator reports, are provided in Table 1 for reference.

Lake Ontario does have a full dataset that was compared with the baseline conditions identified in previous State of the Great Lakes reporting based on NOAA 1997 data. This dataset was developed in 2001 and 2002 to support the

International Joint Commission's (IJC's) International Lake Ontario – St. Lawrence River Regulation Study. A similar methodology was utilized to classify the full U.S. and Canadian Lake Ontario shoreline based on the type and extent of shoreline hardening (see Stewart, 2002) with the results summarized in the Flood and Erosion Prediction System (FEPS) database (see Baird, 2005). The dataset was used to model water level impacts on shoreline structure lifespan and as a result, there are small gaps where direct comparisons to the baseline data set are difficult. In particular, there were some instances where the percent of very low quality shoreline structures was not identified as they were not included in the water level impact modeling. In the case of the Great Lakes comparison, these areas were identified within the unclassified category, even though there was likely some shoreline hardening occurring. It should also be noted that the updated Lake Ontario classification dataset utilized a higher resolution shoreline delineation than was used in the baseline conditions identified in previous State of the Great Lakes reporting. As a result, the classified shoreline extent is greater for the updated dataset. Finally, the updated dataset estimates the percent hardened shoreline using standard 1 km reaches along the full shoreline whereas the baseline dataset categorized reaches of variable (and generally greater) length.

To assess potential changes in the Lake Ontario shoreline since the 2011 State of the Great Lakes report, the U.S. Army Corps of Engineers (USACE) reviewed all existing geospatial data for the Lake Ontario shoreline and determined that while a delineation of the New York State side of Lake Ontario was performed in 2012, there has been no recent delineation of the Canadian side of Lake Ontario since the 2001 and 2005 analysis. Therefore, the USACE developed an updated shoreline dataset of the entire Lake Ontario Shoreline using data from two sources. The United States shoreline was delineated and classified in August 2012 by AECOM in association with the New York State Office of General Services (NYSOGS) and the New York State Department of Environmental Conservation (NYSDEC) for the IJC's Lake Ontario - St. Lawrence River water level study. Each feature within the data layer produced by AECOM represents a reach of shoreline of differing classification. Classification reaches were not defined according to a set unit of measure. The existing NYSOGS/NYSDEC shoreline polyline was modified to most accurately represent the actual shoreline boundary based on 2010-2012 Bing Maps aerial imagery. The shoreline was then split into classifications according to the type of shoreline based on review of 2012 oblique imagery produced by the USACE. The Canadian shoreline was also digitized and classified by the USACE-Buffalo District in 2015 using the AECOM classification scheme. Specifically, each feature within the data layer was created to represent a reach of shoreline based on predefined categories of shoreline types. Similar to the AECOM data format, the shoreline reaches defined by the USACE were not initially defined by the 1 km reach standard used in the 2002/2005 dataset. Following the AECOM methodology, the USACE delineated the Canadian shoreline of Lake Ontario to most accurately represent the shoreline based on a review of 2010-2012 imagery depicted on Bing Maps and ESRI world imagery aerial basemaps. Both the Bing Maps and ESRI imagery use photographs from various sources including federal, state and local entities using satellites and aerial photography. The classifications were assigned by shoreline type based on an additional review of imagery that was accessed via Google Earth Pro and oblique imagery from Bing Maps (Pictometry). The imagery found in Google Earth Pro included high resolution aerial photos from multiple sources taken between May 2015 and September 2015, and the Bing Maps oblique imagery was taken from 2007-2015. To create the final contiguous 2015 shoreline dataset, the AECOM delineation of the New York State shoreline was merged with the dataset created by the USACE. The merged dataset was then copied and divided into 1 km reaches. In order to determine the percent of each shoreline type within the 1 km reaches, a statistical analysis (tabulate intersection) was performed using ArcGIS. The analysis compared the predefined 1 km reaches with the classifications that were determined from the AECOM/USACE delineation of the shoreline. The resulting output included shoreline classification, length, and percentage of each type in each of the 1,988 reaches that were included in the 1 km shoreline reach dataset.

Table 2 provides the length of shoreline in the baseline, 2001-2002 datasets, and 2015 datasets, along with the percent of shoreline within the various percent hardening categories for Lake Ontario. The percent of shoreline within the moderately (40 to 70% hardened) and major (>70% hardened) categories increased by 0.9 and 1.0 %, respectively while the percent of the shoreline within the minor (15 to 40% hardened) increased 6.8% and no protection category (<15% hardened) was reduced by 1.3%. The extent of shoreline in the minor and low protection categories is still below the poor threshold established and resulted in the poor status classification. The results suggest that there has been a slight increase in the amount of shoreline hardening since the 2001-2002 dataset was established and a deteriorating trend was identified. 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. Figure 1 provides maps of the baseline Lake Ontario shoreline hardening categorization and the 2001-2002 Lake Ontario data, and Figure 2 shows the updated 2015 Lake Ontario data.
Data for the connecting channels were not included in this assessment due to a lack of data to compare to on the short term. In the baseline data the connecting channels or rivers were included as separate entities for comparison. In the 2011

classification they were not given a classification nor were they compared. When we classified Lake Ontario we, simply put, started where the 2012 AECOM dataset left off in New York State, which did not include the Niagara River or the St. Lawrence.

Linkages

The hardening of shorelines can result in the loss of habitat, further erosion of unprotected properties adjacent to the structure, water quality degradation and the interruption of natural shoreline processes including reduced sediment transport. The hardened shoreline 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 – Fish spawning and feeding habitat associated with coastal wetlands can both be accentuated or diminished based on the physical modification to the shoreline and the effects it may have on coastal and nearshore processes, as well as effects on habitat structure along the Great Lakes shoreline. These data will help to assess where both beneficial and unfavorable impacts occur.

Watersheds Impacts – this is directly related to changes in land cover climactic dynamism in areas with increased or increasing amounts of anthropogenic shoreline modification that diminish littoral drift and impact regional sediment management.

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 Limitations

1. There is documentation prepared as part of the IJCs International Lake Ontario – St. Lawrence River Study (see Stewart, 2002). The classification itself was undertaken by private contractors with considerable experience in shoreline classification procedures. However, there is no formal validation methodology for undertaking this type of shoreline classification. The 2015 data documentation was prepared by the USACE and includes documentation provided by AECOM
2. The classification itself was undertaken by private contractors and USACE employees with considerable experience in shoreline classification and aerial photography interpretation procedures
3. The geographic scale for the updated information only covers Lake Ontario and cannot be used for Great Lakes basin wide assessments
4. The procedure for identifying hardened shorelines was applied consistently on both the Canadian and U.S. shorelines of Lake Ontario. 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 and quality of input imagery used for individual shoreline reaches are not identified.
5. The variation in reach length and detail of shoreline delineation between the baseline dataset ,the 2001-2002 Lake Ontario data, and the 2015 Lake Ontario data result in uncertainty in the overall status and trends analysis regarding hardened shorelines

Data collection for this sub-indicator could include estimates based on aerial photography and limited field studies, with a focus on Areas of Concern and sites identified from the 1992 IJC data where shoreline hardening appears to be increasing. As existing information is collected and evaluated recommendations for cost effective data collection (consistent with

trend comparability wherever possible) will be evaluated and included in the report. Compatibility issues are expected between the U.S. and Canada and potentially also between various States and Provinces.

There exists a lack of a standardized dataset on many of the lakes that can be directly compared to the baseline conditions established for the Great Lakes/SOLEC hardened shoreline sub-indicator. For the 2016-2017 reporting cycle, an update will only be provided for the Lake Ontario basin.

There was uncertainty when trying to make a direct comparison between the different datasets for Lake Ontario. The shoreline reach categorizations are defined differently in all three datasets. However, the closest comparison can be made between the 2001-2002 data and the 2015 data since these data use the fixed 1 km shoreline reaches. The large increase in the minor protection and decrease in unclassified categories could be the result of the availability of higher resolution aerial imagery. In addition, it is possible that the difference in shoreline length could be due to variation in lake water levels during the period that aerial images are taken and the increased availability of higher resolution aerial photographs than those used in the 2001-2002 data. The most recent Lake Ontario dataset used the 2001-2002 shoreline delineation as a guide marker, but followed the shoreline present in the aerial imagery in order to obtain an accurate depiction of shoreline for comparison. 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. However, it should be recognized that direct comparisons between datasets will be highly uncertain without using a common baseline shoreline delineation and comparable reach lengths. Finally, as stated in the 2011 State of the Great Lakes report, the baseline dataset is not clear on the transition between percent protected categories. For example, a shoreline reach that is 70% hardened could fall within either the 40% to 70% category or the 70% to 100% category. More explicit transitions were used for the categorization of the updated datasets.

Additional Information

The degree of negative impact to aquatic life in the nearshore will vary depending on the design of the protection and on the antecedent conditions. Some types of hardened shoreline induce more severe impacts than do others. Existing information will be gathered into a publicly available single resource to update the current 1992 IJC data standard for reporting and as a tool refining future potential actions. A classification scheme that reflects the degree of impacts from different types of shore protection should be developed based on a literature review. A “change matrix” will be developed for this sub-indicator for use in future status assessments to track changes in hardened shorelines from one type to another.

There are opportunities for future updates to the hardened shorelines sub-indicator. An interagency effort is currently in process to generate a baseline shoreline dataset for the remaining Great Lakes. It is expected that this dataset will allow for at least a partial (U.S. portion) basin wide assessment of change to hardened shorelines for the next (post-2019) reporting cycle. Updated high resolution aerial imagery exists for much of the Great Lakes shoreline and oblique imagery was collected in 2012 for the U.S. shoreline of the Great Lakes. This information will make it possible to duplicate the Lake Ontario effort across the other Great Lakes to create new datasets of the shoreline and update any existing reach delineations, shoreline classifications, and the percent of shoreline hardening. Efforts to create new or update existing datasets should ensure that classification methodologies are similar to past efforts (e.g. as used for the updated Lake Ontario shoreline classification) and standardized reach delineations are utilized. Consideration will be given to including all anthropogenic features that are not currently included in the dataset in an updated basin wide dataset. When the basin wide dataset is completed in the future following the basic procedures used for the 2015 Lake Ontario dataset, this new dataset will be used as the baseline moving forward. This will allow for the use of a measure that would compare the ratio of human modified shoreline to the total length of shoreline in the Great Lakes basin. This will allow for comparison on a lake-by-lake basis, as well as provide an easy to understand overview for the entire basin.

Acknowledgments

Authors: Anthony Friona, U.S. Army Corps of Engineers, ERDC (2018)

E. Pirschel and T. Crockett, U.S. Army Corps of Engineers, Buffalo District (2018)

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List of Tables

Table 1. Baseline Great Lakes hardened shoreline classification used in the 2009 and 2011 State of the Great Lakes Hardened Shoreline indicator report assessments. Original data is from NOAA, 1997.
Source: National Oceanic and Atmospheric Administration (1997)

Table 2. Comparison of baseline Great Lakes hardened shoreline classification (using 1997 data), 2011 hardened shoreline classification (using 2002-2005 data), and updated hardened shoreline classification for Lake Ontario using 2015 data.
Source: Baseline data from National Oceanic and Atmospheric Administration (1997), 2001-2002 Lake Ontario data from Stewart (2002) and Baird (2005), and the 2015 updated data from AECOM (2012) and the United States Army Corps of Engineers – Buffalo District (2015)

List of Figures

Figure 1. Maps of baseline Great Lakes/SOLEC hardened shoreline classification (top figure) and updated (2001-2002) hardened shoreline classification for Lake Ontario (bottom figure).
Source: Baseline Great Lakes/SOLEC data from National Oceanic and Atmospheric Administration (1997) and updated Lake Ontario Data from Stewart (2002) and Baird (2005)

Figure 2. Map of the 2015 Lake Ontario Hardened shoreline classification update.
Source: New York shoreline data from AECOM (2012) and Canadian shoreline data from the United States Army Corps of Engineers – Buffalo District (2015)

Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

Baseline Great Lakes hardened shoreline classification

Lake/ Connecting Channel	Heavily Protected (%) (>70%	Moderately Protected (%) (40-70%	Minor Protection (%) (15-40%	No Protection (%) (<15%	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	1.0	4.5	91.6	1.1	0.3	6366
St. Clair River	69.3	24.9	2.1	3.6	0.0	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	8.0	22.2	0.0	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	0.9	184
Lake Ontario	10.2	6.3	18.6	57.2	0.0	6.2	1772
St. Lawrence	12.6	9.3	17.2	54.7	0.0	6.2	2571

Table 1. Baseline Great Lakes hardened shoreline classification used in the 2009 and 2011 State of the Great Lakes Hardened Shoreline indicator report assessments. Original data is from NOAA, 1997.
Source: National Oceanic and Atmospheric Administration (1997)

Comparison of baseline Great Lakes hardened shoreline classification and updated classification

	Baseline Classification	2011 Lake Ontario Classification	2015 Lake Ontario Classification
Length of Shoreline Categorized (km)	1772.0	2444.3	1988.0
1. Heavily Protected (%)(>70% hardened)	10.2	20.0	21.0
2. Moderately Protected (%)(40-70%)	6.3	8.0	8.9
3. Minor Protection (%) (15-40% protected)	18.6	5.7	12.5
4. No Protection (%) (<15% protected)	57.2	57.3	56.0
5. Non-structural Protection (%)	0.0	0.1	0.0
6. Unclassified (%)	6.2	8.8	1.6

Table 2. Comparison of baseline Great Lakes hardened shoreline classification (using 1997 data), 2011 hardened shoreline classification (using 2002-2005 data), and updated hardened shoreline classification for Lake Ontario using 2015 data.
Source: Baseline data from National Oceanic and Atmospheric Administration (1997), updated Lake Ontario data from Stewart (2002) and Baird (2005), and updated Lake Ontario data from U.S. Army Corps of Engineers (2015).

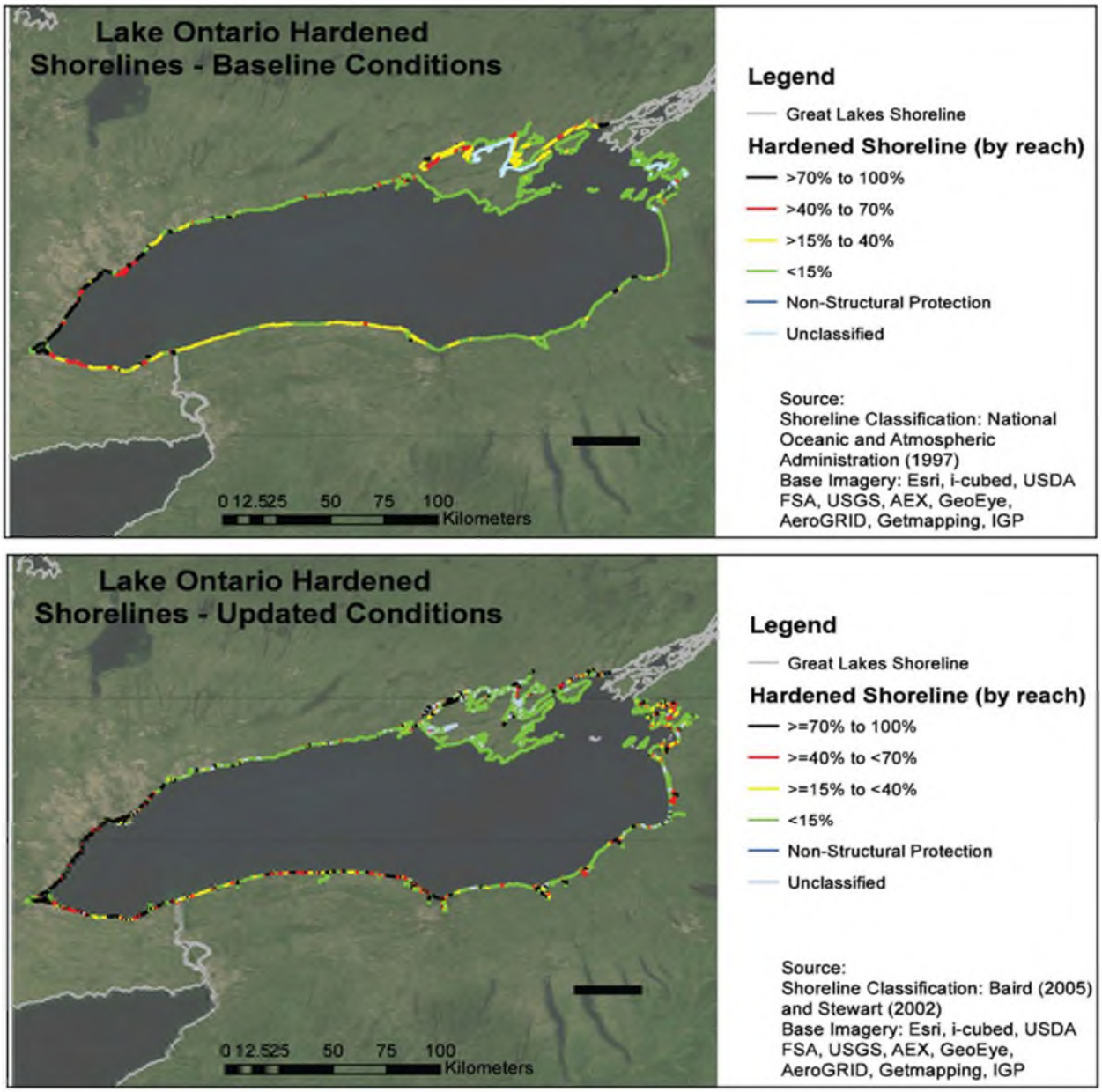


Figure 1. Maps of Baseline Great Lakes hardened shoreline classification (top figure) and updated (2001-2002) hardened shoreline classification for Lake Ontario (bottom figure).
Source: Baseline Great Lakes data from National Oceanic and Atmospheric Administration (1997) and updated Lake Ontario data from Stewart (2002) and Baird (2005)

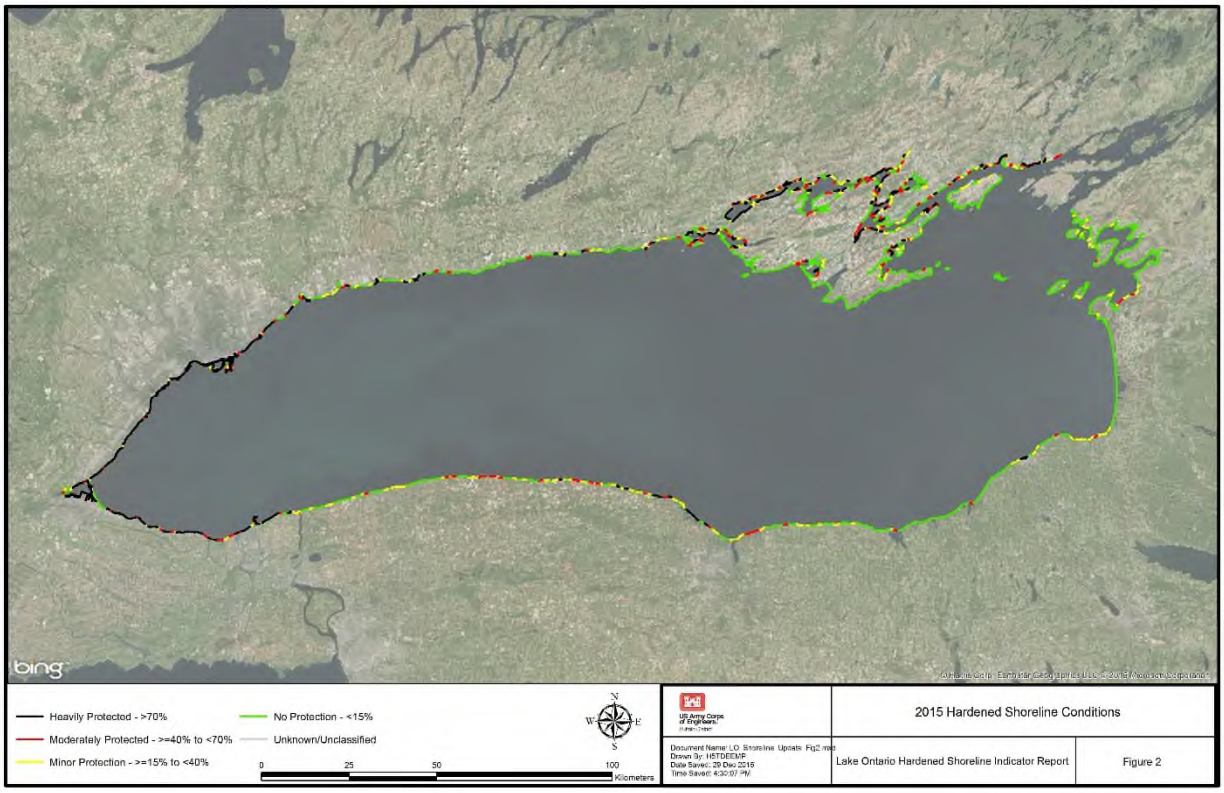


Figure 2. Map of the 2015 Lake Ontario Hardened shoreline classification update.
Source: New York Shoreline data from AECOM (2012) and Canadian shoreline data from the United States Army Corps of Engineers – Buffalo District (2015)

Sub-Indicator: Watershed Stressors

Overall Assessment
Status: Fair
Trends:
10-Year Trend (2000-2010): Unchanging
Long-term Trend: Undetermined
Rationale: The status can also be described as mixed, because "Good" condition is found in 19% of the watersheds in the basin, "Fair" condition is found in 60% of the basin's watersheds, and 21% of the basin is in "Poor" condition, see Tables 1, 2.

When the combined stresses of population density, road density, urban development, and agricultural development are considered based on percent of watersheds instead of percent area, two of the five Great Lakes (Erie and Ontario) have a status of "Poor", Lake Michigan is "Fair", and lakes Huron and Superior are "Good" (Table 1a; Figure 1). Consequently, the status of the Great Lakes basin overall is operationally defined as "Fair" (Table 1a; Figure 1), since the majority of its watersheds are in fair condition. When the operational definition of condition is based on the percent area, (Table 1b), the majority of the basin's area is in "Fair" condition. Across the basin, roads were ubiquitous and represent the largest source of potential risk of degradation in largely undeveloped areas (Figure 7).
These stressors together represent the majority of the variation described by anthropogenic stressors (agricultural; land use; human population/development; shoreline modification).

Basin wide, condition category shifts were relatively rare, with only 81 of 5,583 watersheds changing condition categories from 2000 to 2010. Changes from "Fair" (generally) to "Poor" classes were most common (38 watershed transitions), followed by transitions from "Poor" to "Fair" condition (26), and "Good" to "Fair" (12). Due to the small number of watershed transitions, which together represent only 1.4% of the total number of watersheds, and 0.13% percent of the basin area, the 10-year trend is listed as "Unchanging".

See the Additional Information section for further explanation of data interpretation issues.

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

Lake-by-Lake Assessment
Note: impacts from watersheds draining into connecting channels are assigned to the downstream lake.

Lake Superior
Status: Good
10-Year Trend (2000-2010): Unchanging
Long-term Trend: Undetermined
Rationale: Of the 1,534 watersheds in the Lake Superior basin, 595 were classified as "Good" (38.8%), 917 were classified as "Fair" (59.8%), and only 22 were classified as "Poor" (1.4%) (Table 2, Figure 9). The interval from 2000 to 2010 saw a very minor shift in the condition of Lake Superior's watersheds, with a change of 0.6% of total watershed numbers (10 of 1,534) from the "Good" to "Fair" category. Five watersheds transitioned from "Fair" to "Good". No watersheds shifted from "Fair" to "Poor". This suggests that conditions are largely unchanged (Table 2; Figure 8). A portion of the Lake Superior basin in Canada did not have 2010 era land use data; to derive an estimate of change the assumption was made that there was no change in percent agricultural or developed land. The basin-wide trend is therefore to be regarded as a conservative estimate. Lake Superior has the lowest percentage of agricultural land in the basin, the lowest road density and lowest population density. This basin was second in terms of the number of watersheds in the lowest quintile for percent developed land, behind Lake Huron due to the large number of small undeveloped island watersheds in Lake Huron (Table 3; Figures 2-7).

Lake Michigan
Status: Fair
10-Year Trend (2000-2010): Unchanging

Long-term Trend: Undetermined
Rationale: Lake Michigan's 629 watersheds were classified predominantly as "Fair" (83.5%); 16.1% were classified as "Poor" and less than 1% were scored as "Good" (Table 2; Figure 10). Lake Michigan was unremarkable in terms of the distribution of each of the component stressors with one exception. Few condition transitions were noted for Lake Michigan. Trends for Lake Michigan are based on complete data sets for the basin and therefore represent the best available estimates. It was notable that the Lake Michigan basin had the lowest number of watersheds in the lowest quintile in terms of road density, suggesting that few road-less areas remain within that basin (Table 3; Figures 2-7).

Lake Huron (including St. Marys River)
Status: Fair
10-Year Trend (2000-2010): Unchanging
Long-term Trend: Undetermined
Rationale: Lake Huron's condition can best be described as "Fair" since over 50% of its watersheds fall into the "Fair" category (Table 2; Figure 11) although it almost meets the criteria for "Poor" and it does meet the criteria for "Good". Six of the nine watersheds transitioned from "Fair" to the "Poor" category, with an additional three watersheds changing from "Poor" to "Fair". The proportion of watersheds that changed assessment status was minimal relative to the total number of watersheds in the Lake Huron basin (1,646); therefore, the 10-year trend is "Unchanging" (Figure 8). A portion of the Lake Huron basin in Canada did not have 2010 era land use data; therefore, the assumption was made that there was no change in percent agricultural or developed land. These trends are therefore to be regarded as conservative estimates of change. The Lake Huron basin has the highest number of watersheds in the lowest quintile for percent developed land (due to the large number of very small watersheds and islands with low percentages of development), and the second lowest in terms of percent agriculture, road density and population density (Table 3; Figures 2-7).

Lake Erie (including St. Clair-Detroit River Ecosystem)
Status: Poor
10-Year Trend (2000-2010): Unchanging
Long-term Trend: Undetermined
Rationale: Lake Erie's condition is rated as "Poor" because almost 50% of the watersheds (410 of 854) were classified as being in "Poor" condition, 47.5% were in "Fair" condition, and only 4.4% of watersheds were in "Good" condition (Table 2; Figure 12). Although there were 13 watersheds that transitions into the "Poor" category, this number represents a small proportion (less than 2%) of the watersheds in the Lake Erie basin (Figure 8). Ten-year trends for Lake Erie are based on complete data sets for the basin and therefore represent the best available estimates. The Lake Erie basin had the highest number of watersheds in the upper quintile of the distribution for all four stressor components (Table 3; Figure 2-7).

Lake Ontario (including Niagara River and International section of the St. Lawrence River)
Status: Poor
10-Year Trend (2000-2010): Unchanging
Long-term Trend: Undetermined
Rationale: Most (66.2%) of Lake Ontario's watersheds fall into the "Fair" category, but approximately 32% fall in the "Poor" category. The operational definition based on the 20_{th} percentile criteria puts the Lake Ontario basin into the "Poor" category, but the lake could also well be described as mixed (Figure 13).Transitions in Lake Ontario included 13 watersheds moving into the "Poor" category (1.5% of watersheds) and 15 moving into the "Fair" category from "Poor" (Figure 8). Ten-year trends for Lake Ontario are based on complete data sets for the basin and therefore represent the best available estimates. The latter represents the highest number of positive transitions in the overall basin. The Lake Ontario basin has the second highest number of watersheds in the upper quintile of the distribution for all four component stressors, behind Lake Erie (Table 3; Figures 2-7).

Status Assessment Definitions
To apply this assessment and score to the Great Lakes basin, one determines the maximum range of AgDev scores across the entire basin to derive the cut-off values between Good/Fair and Fair/Poor. These cut-offs are applied to watersheds within each lake to determine the percentage of watersheds in the lake that are in each of the Good, Fair, and Poor categories (see Table 1 of report).
Good: low risk of deterioration. Conditions for a lake are "Good" if > 20% of the watersheds are in the "Good" condition.

Fair: medium risk of deterioration. Fewer than 20% of lake's watersheds or watershed areas were in 'Poor' Condition, and fewer than 20% were also in the "Good".
Poor: high risk of deterioration. Markedly more than 20% of the lake's watersheds were in the "Poor" condition.
Undetermined: data are not available or are insufficient to assess condition of the ecosystem components.

Trend Assessment Definitions
Lake-specific and basin-wide trends in the component stressor scores and the AgDev index may be assessed by re-calculating the component stressors at appropriate intervals as new data are released. The most recent National Land Cover Database which includes land use/cover data from the U.S, side of the basin were released in 2011; the Ontario Land Cover Compilation dataset includes data from 2005 – 2011 (see Data limitations section below).

Improving: conditions will be indicated by an increase in the number of watersheds and/or the cumulative area of the watersheds classified as 'low risk' (= Good condition); or a decrease in the number of watersheds and/or the cumulative area of the watersheds classified as 'high risk' (= Poor condition).
Unchanging: no change in the number of watersheds and/or cumulative area of the watersheds classified as either 'low risk' or 'high risk'.
Deteriorating: conditions will be indicated by a decrease in the number of watersheds and/or the cumulative area of the watersheds classified as 'low risk' (= Good condition); or an increase in the number of watersheds and/or the cumulative area of the watershed classified as 'high risk' (= Poor condition).
Undetermined: Metrics do not indicate a clear overall trend, or data are not available to report on a trend.

When fully vetted, biological condition thresholds can be applied to define condition (see **Additional Information**, below).

Endpoints and/or Targets
The calculated scores are classified as "Poor", "Fair" or "Good" according to the risk that the stress will cause deteriorating aquatic habitat or water quality, or degraded biological communities. Subsequently, the status of each Great Lake is assessed according to the proportion of the lake's watersheds that fall into each risk category.

Sub-Indicator Purpose
- Assess the relative degree of stress derived from watersheds on the environmental quality of the Great Lakes;
- Infer potential risk of harm from impacts of human activities in watersheds on water quality, habitat, biota, and natural processes.

Ecosystem Objective
The combined effects of watershed stressors should not result in the impairment of the physical, biological, or chemical integrity of the Great Lakes as reflected in Annexes 2 (improve water quality), 4 (manage nutrients), and 7 (protect species and their habitats) of the 2012 Water Quality Agreement.

Measure
This sub-indicator measures condition and, with updated data, can be recalculated to determine changes between time steps. Times available for comparison are defined by the collection and release dates of synoptic datasets such as remotely sensed land cover and human population data. The 2011 iteration of this sub-indicator (ECCC and USEPA, 2011) did not make any temporal comparisons because suitable data from multiple dates were not available. This sub-indicator was updated using available data for all of the U.S. side and a portion of the Canadian side of the basin.

In 2015, a new set of basin wide watersheds (the spatial reference frame for this sub-indicator) became available (Forsyth et al. *2015*, Great Lakes Aquatic Habitat Framework; GLAHF); these will likely be a widely-used frame of reference for basin-wide analyses in the future. To establish a baseline metric using the same spatial units this sub-indicator has been re-calculated for the circa 2000 condition, superseding the SOGL 2011 version. Updated land use data for the U.S. are expected to be released in December 2018; the 2020 census data will be released in December 2020. The most recent population census in Canada was conducted in 2016; it is not known when Land Use data encompassing the Great Lakes basin will be updated (see Data Limitations section below).

This sub-indicator uses a combined agriculture + development stress index (AgDev) to calculate scores for individual Great Lakes watersheds using a consistent scale of resolution among reporting periods. This stress score is

adapted from a peer-reviewed methodology previously applied to the Great Lakes basin (Host et al 2011) and revised by Johnson et al. 2015). The index is based on standardized scores of data that represent key manifestations of human activity in the watersheds that are a potential risk to the Great Lakes ecosystem health. Stressors making up the index include road density, population density, agricultural land cover, and developed land cover (modified from Host et al. 2011). These stressors together represent the majority of the variation described by five anthropogenic stressors (agricultural/chemical loadings; land use; atmospheric deposition; human population / development; shoreline modification) quantified by Danz et al. (2005).

This revised index differs from the State of the Great Lakes 2011 version in two ways: 1) by eliminating the point source data (which was found to have numerous quality issues), and 2) revising the metric calculation (Johnson et al., 2015). In addition, for 2016, anthropogenic stress was summarized for GLAHF (Wang et al. 2015) watersheds on the U.S. and Canadian sides of the Great Lakes basin (a binational effort to develop a consistent set of drainage units for the basin; Forsyth et al. 2015).

The composite index was generated as follows: first, an index of agriculture stress (Ag) was based on the areal percentage of land in agricultural estimated from the 2011 National Land Cover Dataset (NLCD) and Ontario Land Cover Compilation v.2.0. Next, stress attributed to development was characterized based on the areal percentage of urban land use, human population density (U.S. Census Bureau and Statistics Canada) and road density (TIGER 2000 and 2010, U.S. and NRN 2nd and 7th edition Canada). These variables were scaled to range between values of 0.0 – 1.0 based on the range of data across the Great Lakes basin (not including the St. Lawrence River watersheds). Following methods in Host et al. (2005), the maximum of the three scaled (0-1) values for each watershed was used as the development index (MaxRel Dev). To combine the agriculture and development values for a watershed, we calculated a Euclidean distance from the graph origin (0,0) graph to the x, y coordinates of the Ag and MaxRel Dev Index scores (AgDev; Figure 1). The resulting metric is called AgDev, and supersedes the former Combined Watershed Stress Index (State of the Great Lakes 2011). To ensure consistency of reporting, the AgDev index calculation based on circa 2000 data was provided, as well as 2010 (Figure 14, Tables 2, 3).

Ecological Condition

The relative amount of stress imposed by four measures of human activity on the land within the 529,679 km^2 area of the Great Lakes basin was assessed for each of the 5593 watersheds surrounding the Great Lakes (as generated from an ArcHydro GIS analysis (Forsyth et al. 2015)).

In the absence of biological data against which to calibrate the stressor scores, authors designated the 20th percentile of the distribution of stress scores for each variable and the AgDev index as the criterion for classifying a watershed as “Good” versus “Fair”. Authors designated the 80th percentile the distribution as the boundary between “Fair” and “Poor”. Watersheds classified as “Good” pose minimal risk of degradation of the biological community in Great Lakes aquatic receiving habitats. Watersheds classified as “Poor” are at greatest risk of having degraded Great Lakes communities. The cutoff values representing the 20th percentile and 80th percentiles for the Great Lakes components and AgDev scores are listed in the legend of Table 1. Status assessments for the 2010 era data are made relative to these values for 2000 era data.

Other Spatial Scales

The data shown are benchmarked to 2000 era AgDev scores that were derived based on the 20th and 80th percentiles of the AgDev distribution across the entire Great Lakes basin (Table 1). The same process was applied individually to each lake to determine the relative condition of watersheds within each Lake. Table 3 depicts the 2000 and 2010 era distributions and transitions for individual lakes.

The components of the WSI (AgDev) are tabulated and scored for the land bordering each Great Lake rather than for the lakes themselves. However, there is strong evidence that the effects of land-based stress are manifested in habitats most closely associated with each watershed. Niemi et al. (2007), Peterson et al. (2007) and Yurista and Kelly (2009) found that the correlation between land-based stress and waterborne nutrients was highest for tributary streams and coastal wetlands. Although the correlation becomes weaker with increasing distance from shore, the correlation remains statistically significant in water 10 meter deep or more. The greater the stress, the greater the risk of degradation of biological features in the lakes themselves. These relationships have recently been qualitatively scored and shown in lakewide and basinwide maps as ‘threats’ (or risk of degradation) by Allan et al. (2012).

Linkages

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

- Aquatic Habitat Connectivity – The number of dams and barriers is an important factor in assessing watershed stress but is not directly incorporated into the stress metric.
- Coastal Wetlands: Extent and Composition – Human activity influences the extent and quality of coastal wetlands. Composition is largely determined by geomorphic factors including exposure to wave action and geologic setting; human disturbances such as water diversion, and shoreline alteration have a profound impact on the type and quality of coastal wetlands, influencing sediment and organic matter deposition and ultimately, habitat structure and function.
- Water Quality in Tributaries – Water quality is strongly linked to both human activities as well as weather and climate, with precipitation patterns serving as a driver of nutrient and sediment loading. In particular, increasing frequency and intensity of storms is responsible for enhanced nutrient and sediment loads from agricultural and urban landscapes. In addition to degraded water quality from enhanced stormwater runoff, urban buildout can also be linked to warming surface waters, due to runoff from heated surfaces.
  Water quality is regulated through permits of point sources and through implementation of best management practices for nonpoint source pollution. Such practices mitigate the overall effects of watershed stressors on receiving waters of the Great Lakes.
- Human Population – These data are integral to the calculation of the Watershed Stress index.
- This Watershed Stress sub-indicator also links directly to the other indicators in the Watershed Impacts and Climate Trends indicator, particularly Land Cover, since those land cover data are integral to the calculation of the Watershed Stress Index component metrics (i.e., Ag and MaxRelDev).

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 Limitations

Land use data are integral to the calculation of this sub-indicator. Land cover mapping is conducted periodically by the Canadian (Federal or Provincial) and U.S. governments. Development of a basin-wide land cover map requires that classification schemes be cross-walked to reflect common criteria for each class. The basin-wide land use and land cover classification reflects the lowest common denominator (coarsest level) for each land cover class (see https://www.glahf.org/ for metadata describing how the binational land cover map was created). Subsequent iterations of the Watershed Stress Index will rely on the updated NLCD on the U.S. side of the basin (based on 2016 imagery), and the most recent Ontario Land Cover Compilation (described below).

For Canada, land use and land cover data were derived from a composite data product, the Ontario Land Cover Compilation (OLCC) which is a regionalized land cover product for the province of Ontario that combines three separate land cover databases (Far North Land Cover v1.4, SOLRIS v1.2, and the Provincial Land Cover 2000 Edition). In southern Ontario the data are derived from the SOLRIS (Southern Ontario Land Resource Information System) program, while a separate data set was created for far Northern Ontario (Far North Land Cover, 2005-2011). The SOLRIS land cover dataset (v2, 2009-2011) covers southern Ontario and excludes the largely forested northern regions of the Lake Superior and Lake Huron watersheds, north of N 45.88334 and west of W 83.10000. Trends in each of the component measures were calculated for the entire basin, but agricultural and developed lands in the gap area in Ontario were held constant at the 2000 era values. Therefore, the trend assessments for Lake Superior and Lake Huron represent conservative estimates of change in risk of watershed stress.

Additional Information

The components and total AgDev score have been determined for every Great Lakes sub-watershed based on data from 2000-2011. The locations at greatest risk of significant biological loss (those approaching the boundaries) and those with greatest potential for restoration (sites with stress scores only slightly higher than the boundaries) can be identified using biologically-based thresholds, as well as the quantile approach presented here. These are the locations where investment in protection or restoration should most likely to succeed. Johnson et al. (2015) present a map of risk, based on biologically based thresholds derived from Kovalenko et al. (2014).

This revised version of the Watershed Stressor sub-indicator improves on the 2011 version in two ways: land use data are derived exclusively from data derived from government sources (e.g., land use from NLCD and SOLRIS), and the watershed framework (GLAHF) is based on a binational effort. In contrast, the 2011 version used land use derived from a variety of sources some with known classification flaws. The new Watershed Stressor sub-indicator (AgDev) should, therefore, be a repeatable metric that can be used for tracking trends in the future. Special note should be made regarding assumptions of change; many transitions were found to occur in very small coastal watersheds. These are especially susceptible to changes in area as a result of water level change, and therefore, interpretation of condition transitions should be made with caution. Changes in road network data (e.g., TIGER in the U.S.), for example, caused 34 small watersheds on Isle Royale in Lake Superior to appear to have changed from "Good" to "Fair" condition (these were omitted from the calculations of transitions). In addition, land use data derived from remote sensing (e.g., NLCD, SOLRIS) are not 100% accurate, and classifications of bare ground (i.e., exposed bedrock, quarries, sand flats, etc.) are easily confused with spectral signatures of impervious surfaces. Thus, areas along the coast can be subject to misclassification. The cutoff values between good/fair and fair/poor were determined based on the count of watersheds across the entire basin, rather than on area. Because of the very large variation in watershed sizes the cumulative distribution of area precluded the identification of reasonable cutoffs due to large gaps into which particular targets (i.e., 20^{th} and 80^{th} percentiles) were likely to fall.

More Trend Information

Development and calibration of the current sub-indicator and its previous iterations (Ciborowski et al. 2011; Host et al. 2011) were accomplished by researchers funded by competitive grants, rather than as part of agency programs. The ability to report on changes will also depend on the provision of updated and cross-walked binational land cover mapping data.

In the near future, the thresholds for classifying watersheds into risk categories where low = Good, medium = Fair, or high = Poor condition could be based on a TITAN analysis (Baker and King 2010) of taxon-specific data, and summarized independently by stressor type and region (as per Kovalenko et al 2014). TITAN identifies points along a stressor gradient where the changes in species composition occur, in terms of either the loss of sensitive species or an increase in abundance of tolerant species (Baker and King 2010). Change points will be calculated for each of five taxa – birds, diatoms, fish, macroinvertebrates and wetland vegetation - along the three stressor components (Ag, Dev, and AgDev) based on data collected as part of the Great Lakes Environmental Indicator project (Niemi et al. 2007). The change point itself is the point along the stressor gradient where the maximum change in species composition occurs. A bootstrapping technique is used to estimate uncertainty around these change points, creating an empirical equivalent of confidence intervals. The 0.05 and 0.95 intervals can be calculated around the change point and the lower (0.05) interval can then be used as a cutoff or threshold value for the "Good" condition, and the tolerant species change point for the "Poor" threshold.

Acknowledgments

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luth, MN 55804) contributed to formation of the research group that identified the need for this database. Dr. Catherine Riseng and members of the Great Lakes Habitat Framework (GLAHF) team delineated the watersheds for which the stressor data were summarized.

The project on which these data were based was originally funded by the U.S. Environmental Protection Agency Science to Achieve Results (STAR) Estuarine and Great Lakes (EaGLe) program through funding to the Great Lakes Environmental Indicators (GLEI) and Reference Condition projects (U.S. EPA Agreements EPA/R-8286750 and EPA/R-82877701, respectively). More recent work was supported by the second stage GLEI-II Indicator Testing and Refinement project funded by a Great Lakes Restoration Initiative grant from the U.S. Environmental Protection Agency Great Lakes National Program Office to L.B. Johnson et al. (GL-00E00623-0). Although the research described in this work has been partly funded by the U.S. EPA, it has not been subjected to the agency's required peer and policy review and therefore does not necessarily reflect the views of the agency and no official endorsement should be inferred. The Great Lakes Aquatic Habitat Framework project was funded by the Great Lakes Fisheries Trust.

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List of Tables

Table 1a. Summary of the status of each lake for ~2010, based on the basin-wide normalized AgDev score, which is applied to each lake. Shown are the number and percent of watersheds in the lowest (Good) and highest (Poor) quintiles. Data are summarized for 5584 watersheds delineated by the Great Lakes Aquatic Habitat Framework (GLAHF) project (Wang et al. 2015). Watersheds stress index (AgDev Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: NRRI

Table 1b. Summary of condition based on % area within each condition class for ~2000 era AgDev scores and ~2010 era AgDev scores.
Source: NRRI

Table 2. Number and percent of watersheds within each Great Lake basin, and assigned condition category based on the criteria set forth in the sub-indicator report. Transitions from condition categories represent loos or gain of the number of watersheds within a condition category from the period 2000 to 2010. Note that due to lack of SOLRIS land use data from a portion of western Ontario, the transition is believed to be conservative.
Source: NRRI. (See accompanying text and Land Cover sub-indicator for more information.)

Table 3. Summary of component stressors for 2000 and 2010 era AgDev scores, including: road density, population density, percent development, Max-Rel Development (= relative maximum of road density, percent development, population density), and percent agricultural land. Cutoff values for condition classes are derived based on the distribution of each stressor across the entire Great Lakes basin. Shown below are number and percent of watersheds in Good, Fair and Poor condition during each time period (00 = 2000; 10 = 2010). Boundary cutoffs were derived for each variable.
Source: NRRI

List of Figures

Figure 1. Condition rankings for the Great Lakes basin circa ~2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin. Constituents of the Ag-Dev score include: percent agricultural land, % developed land, population density, and road density. Data are summarized for 5,593 watersheds across the Great Lakes basin draining to the Great Lakes (see Forsyth et al. *in review*). Watersheds within the St. Lawrence Seaway were not included in the AgDev calculations as they were extreme outliers. Condition classes, however, were assigned to those watersheds based on the normalized scale for the rest of the basin. See text for an explanation of the index calculation.
Source: NRRI

Figure 2. AgDev combined stress score for the Great Lakes basin based on circa ~2010 era data. Color classes based on even distribution across 7 bins.
Source: NRRI

Figure 3. Percent agricultural land for the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins. Note grey area represents a data gap in the Canadian land use data set for this time period.
Source: NRRI

Figure 4. MaxRel Development for the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins. Note grey area represents a data gap in the Canadian land use data set for this time period.
Source: NRRI

Figure 5. Percent developed land for the Great Lakes basin, 2010. Note the grey area represents a data gap in the Canadian land use data set for this time period. Color classes based on even distribution across 7 bins.
Source: NRRI

Figure 6. Population density across the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins.
Source: NRRI
Figure 7. Road density across the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins.
Source: NRRI
Figure 8. Change in condition from circa ~ 2000 to ~ 2010. Note that there was a data gap in the Ontario land cover data set for the 2010 time period. Change in condition was based on the assumption of 'no change' in agriculture and developed lands for those watersheds (gray), changes shown are driven by population or road density changes. In addition, changes to 34 watersheds on Isle Royale are not shown, as they represent non-existent roads present in the 2010 TIGER dataset, but absent in the 2000 version.
Source: NRRI
Figure 9. Condition rankings for Lake Superior. circa ~2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: NRRI
Figure 10. Condition rankings for Lake Michigan watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673. Source: NRRI.
Figure 11. Condition rankings for Lake Huron watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: NRRI
Figure 12. Condition rankings for Lake Erie watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: NRRI
Figure 13. Condition rankings for Lake Ontario watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Note that watersheds in the St. Lawrence River system were not included in the calculations of the AgDev score, but condition classes are shown for those watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: NRRI
Figure 14. Distribution of Great Lakes watersheds along agricultural and development stressor gradients. Colors are quintiles from low to high stress. The AgDev index is the Euclidean distance from the origin to any given point.
Source: NRRI

Last Updated
State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

Lake	Number of Watersheds	Number Watersheds 'GOOD'	% Watersheds 'GOOD'	Number Watersheds 'FAIR'	% Watersheds 'FAIR'	Number Watersheds 'POOR'	% Watersheds 'POOR'	Condition Designation 2010
Superior	1,534	595	38.8	917*	59.8	22	1.4	Good
Michigan	629	3	0.5	525	83.5	101	16.1	Fair
Huron	1,646	431	26.2	899	54.6	316	19.2	Fair**
Erie	854	38	4.4	406	47.5	410	48.0	Poor
Ontario	930	18	1.9	616	66.2	296	31.8	Poor

* 34 Isle Royale watersheds not included.

** See Rationale for this designation in Lake-by-Lake Assessment section above.

Table 1a. Summary of the status of each lake for ~2010, based on the basin-wide normalized AgDev score, which is applied to each lake. Shown are the number and percent of watersheds in the lowest (Good) and highest (Poor) quintiles. Data are summarized for 5584 watersheds delineated by the Great Lakes Aquatic Habitat Framework (GLAHF) project (Wang et al. 2015). Watersheds stress index (AgDev Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: T. Brown, NRRI

Lake	Number of Watersheds	Total Area (km^2)	Watershed area 'GOOD'	% Watershed area 'GOOD'	Watershed area 'FAIR'	% Watershed area 'FAIR'	Watershed area 'POOR'	% Watershed area 'POOR'	Condition Designation 2010
Superior	1,534	141,151	85,745	60.7	55,304	39.2	102	0.1	Good
Michigan	629	116,610	2	0.0	112,065	96.1	4,543	3.9	Fair
Huron	1,646	133,294	17,109	12.8	105,800	79.4	10,384	7.8	Fair
Erie	854	76,607	60	0.1	22,585	29.5	53,962	70.4	Poor
Ontario	930	80,268	2	0.0	75,765	94.4	4,501	5.6	Fair

Table 1b. Summary of condition based on % area within each condition class for ~2010 era AgDev scores. These data are shown for contrast only, as cutoff values derived from watershed areas produce spurious results due to large gaps in the cumulative frequency distribution of watershed areas. See author's notes for further information.
Source: T. Brown NRRI

Lake	Condition	Number Watersheds 2000	% Watersheds 2000	Number Watersheds 2010	% Watersheds 2010	Loss 2000-2010	Gain 2000-2010
All Lakes	Good	1092	19.5	1085	19.5	12	5
	Fair	3368	60.2	3363	60.1	43	38
	Poor	1133	20.3	1145	20.5	26	38
Lake Superior	Good	602	39.2	595	38.8	10	3
	Fair	910	59.3	917	59.8	3	10
	Poor	22	1.4	22	1.4	0	0
Lake Michigan	Good	3	0.5	3	0.5	0	0
	Fair	528	83.9	525	83.5	6	3
	Poor	98	15.6	101	16.1	3	6
Lake Huron	Good	431	26.2	431	26.2	0	0
	Fair	902	54.8	899	54.6	6	3
	Poor	313	19	316	19.2	3	6
Lake Erie	Good	40	4.7	38	4.4	2	0
	Fair	412	48.2	406	47.5	13	7
	Poor	402	47.1	410	48.0	5	13
Lake Ontario	Good	16	1.7	18	1.9	0	2
	Fair	616	66.2	616	66.2	15	15
	Poor	298	32	296	31.8	15	13

Table 2. Number and percent of watersheds within each Great Lake basin, and assigned condition category based on the criteria set forth in the sub-indicator description. Transitions from condition categories represent loss or gain of the number of watersheds within a condition category from the period 2000 to 2010 (’00, ’10). Note that due to lack of SOLRIS land use data from a portion of western Ontario, the transition is believed to be conservative.
Source: T. BROWN, NRRI (See accompanying text and Land Cover sub-indicator for more information.)

Lake	# WshedGood '00	# WshedFair '00	# WshedPoor '00	# WshedGood '10	# WshedFair '10	# WshedPoor '10	% WshedGood '00	% WshedFair '00	% WshedPoor '00	% WshedGood '10	% WshedFair '10	% WshedPoor '10
Percent Agriculture (cutoff values: good to fair = 0%, fair to poor = 53.942%)												
All	2054	2414	1125	2067	2422	1104	36.7	43.2	20.1	37	43.3	19.7
Superior	1126	407	1	1135	398	1	73.4	26.5	0.1	74	25.9	0.1
Michi-gan	137	387	105	140	389	100	21.8	61.5	16.7	22.3	61.8	15.9
Huron	595	682	369	597	682	367	36.1	41.4	22.4	36.3	41.4	22.3
Erie	140	376	338	141	380	333	16.4	44	39.6	16.5	44.5	39
Ontario	56	562	312	54	573	303	6	60.4	33.5	5.8	61.6	32.6
Road Density (cutoff values: good to fair = 0.104 km/km2, fair to poor = 7.778 km/km2)												
All	1102	3351	1140	1045	3256	1292	19.7	59.9	20.4	18.7	58.2	23.1
Superior	512	887	135	471	907	156	33.4	57.8	8.8	30.7	59.1	10.2
Michi-gan	6	431	192	5	400	224	1	68.5	30.5	0.8	63.6	35.6
Huron	441	984	221	437	962	247	26.8	59.8	13.4	26.5	58.4	15
Erie	84	487	283	74	447	333	9.8	57	33.1	8.7	52.3	39
Ontario	59	562	309	58	540	332	6.3	60.4	33.2	6.2	58.1	35.7
Population Density (cutoff values: good to fair = 1.557 people/km2, fair to poor = 62.104 people/km2)												
All	1083	3399	1111	1252	3235	1106	19.4	60.8	19.9	22.4	57.8	19.8
Superior	747	705	82	810	640	84	48.7	46	5.3	52.8	41.7	5.5
Michi-gan	13	456	160	23	442	164	2.1	72.5	25.4	3.7	70.3	26.1
Huron	311	1221	114	356	1184	106	18.9	74.2	6.9	21.6	71.9	6.4
Erie	4	460	390	53	411	390	0.5	53.9	45.7	6.2	48.1	45.7
Ontario	8	557	365	10	558	362	0.9	59.9	39.2	1.1	60	38.9
Percent Developed (cutoff values: good to fair = 0%, fair to poor = 17.284%)												
All	1395	3065	1133	1391	3045	1157	24.9	54.8	20.3	24.9	54.4	20.7
Superior	564	841	129	564	841	129	36.8	54.8	8.4	36.8	54.8	8.4
Michi-gan	5	479	145	5	474	150	0.8	76.2	23.1	0.8	75.4	23.8
Huron	698	750	198	697	750	199	42.4	45.6	12	42.3	45.6	12.1
Erie	74	411	369	70	410	374	8.7	48.1	43.2	8.2	48	43.8
Ontario	54	584	292	55	570	305	5.8	62.8	31.4	5.9	61.3	32.8
MaxRel(roads, population, %developed) (cutoff values: good to fair = 0.006, fair to poor = 0.212)												

Lake	# WshedGood '00	# WshedFair '00	# WshedPoor '00	# WshedGood '10	# WshedFair '10	# WshedPoor '10	% WshedGood '00	% WshedFair '00	% WshedPoor '00	% WshedGood '10	% WshedFair '10	% WshedPoor '10
All	1095	3359	1139	1046	3337	1210	19.6	60.1	20.4	18.7	59.7	21.6
Superior	555	846	133	511	886	137	36.2	55.1	8.7	33.3	57.8	8.9
Michi-gan	4	470	155	4	454	171	0.6	74.7	24.6	0.6	72.2	27.2
Huron	447	990	209	448	976	222	27.2	60.1	12.7	27.2	59.3	13.5
Erie	53	467	334	46	461	347	6.2	54.7	39.1	5.4	54	40.6
Ontario	36	586	308	37	560	333	3.9	63	33.1	4	60.2	35.8
AgDev (cutoff values: good to fair = 0.012, fair to poor = 0.673)												
All	1091	3367	1135	1050	3398	1145	19.5	60.2	20.3	18.8	60.8	20.5
Superior	601	911	22	560	952	22	39.2	59.4	1.4	36.5	62.1	1.4
Michi-gan	3	526	100	3	525	101	0.5	83.6	15.9	0.5	83.5	16.1
Huron	431	902	313	431	899	316	26.2	54.8	19	26.2	54.6	19.2
Erie	40	412	402	38	406	410	4.7	48.2	47.1	4.4	47.5	48
Ontario	16	616	298	18	616	296	1.7	66.2	32	1.9	66.2	31.8

Table 3. Summary of component stressors for 2000 and 2010 era AgDev scores, including: road density, population density, percent development, Max-Rel Development (= relative maximum of road density, percent development, population density), and percent agricultural land. Cutoff values for condition classes are derived based on the distribution of each stressor across the entire Great Lakes Basin. Shown below are number and percent of watersheds in Good, Fair and Poor condition during each time period (00 = 2000; 10 = 2010). Boundary cutoffs were derived for each variable. Note: very low or zero 'good to fair' transition values reflect the large percentages (sometimes in excess of 20%) of the watersheds in the basin with very low or zero development and / or agriculture levels.
Source: T. Brown, NRRINRRI

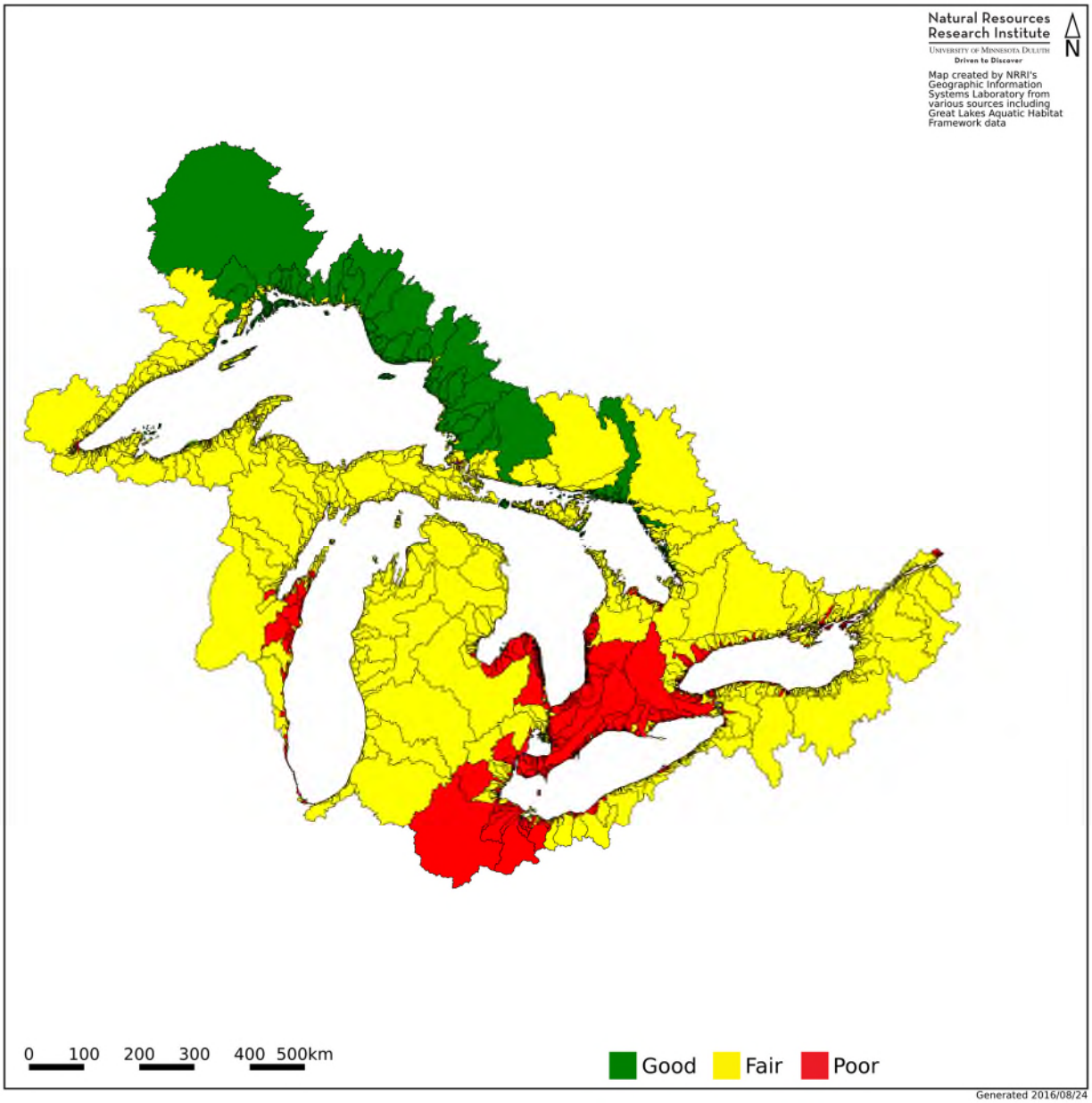


Figure 1. Condition rankings for the Great Lakes basin circa ~2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin. Constituents of the Ag-Dev score include: percent agricultural land, % developed land, population density, and road density. Data are summarized for 5,593 watersheds across the Great Lakes basin draining to the Great Lakes (see Forsyth et al. *in review*). Watersheds within the St. Lawrence Seaway were not included in the AgDev calculations as they were extreme outliers. Condition classes, however, were assigned to those watersheds based on the normalized scale for the rest of the basin. See text for an explanation of the index calculation.
Source: T. Brown. NRRI

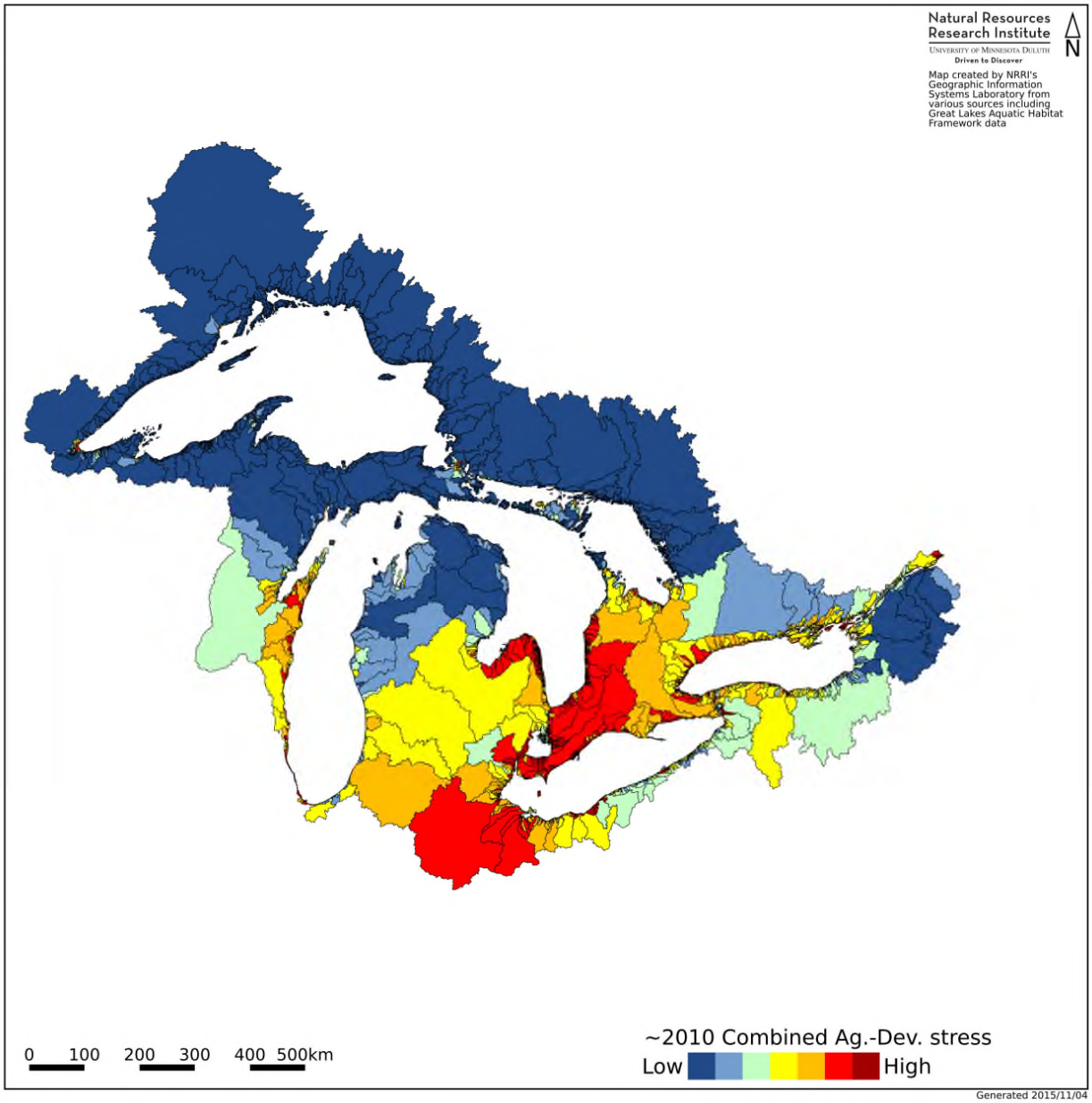


Figure 2. AgDev combined stress score for the Great Lakes basin based on circa ~2010 era data. Color classes based on even distribution across 7 bins.
Source: T. Brown, NRRI

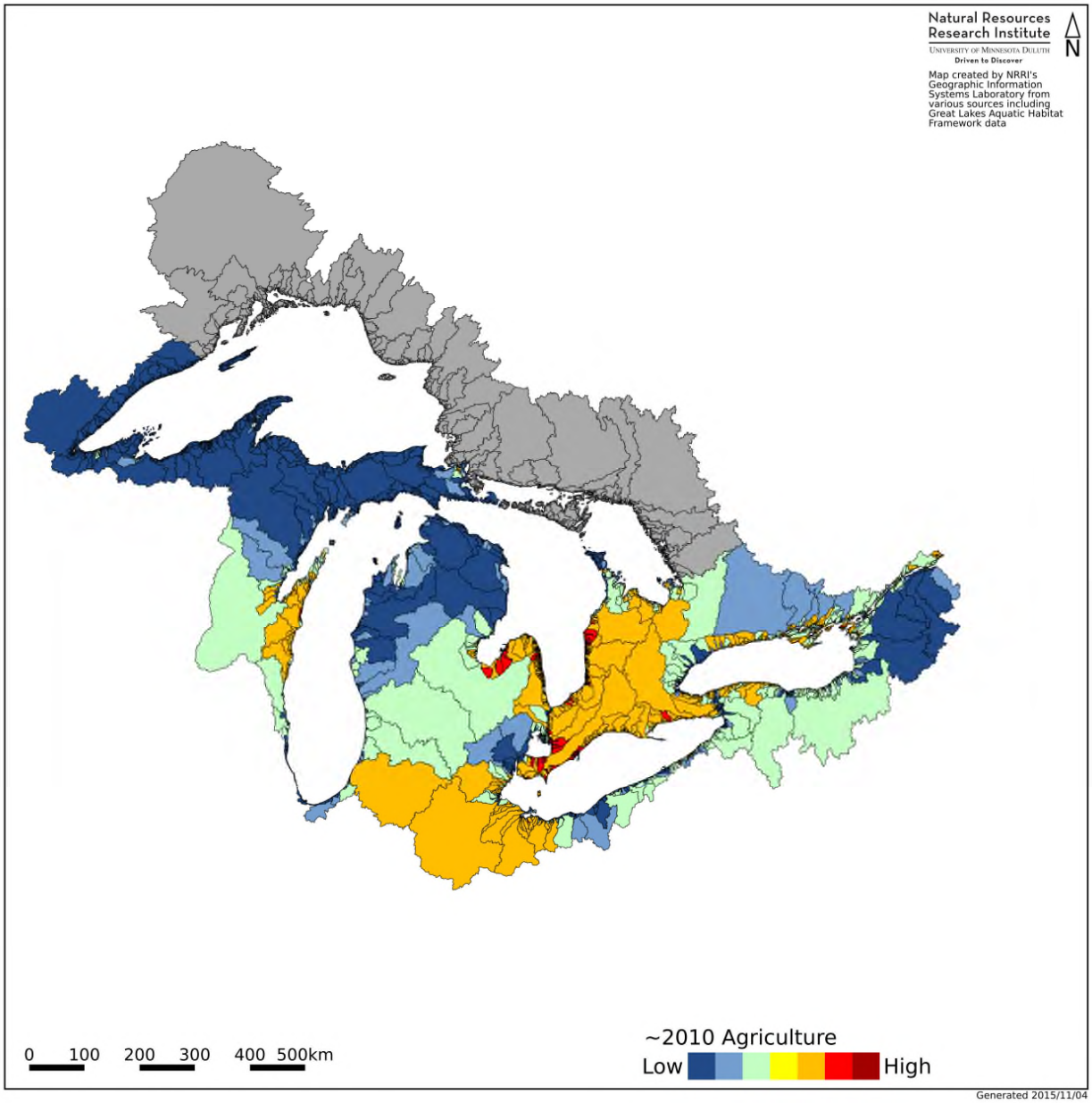


Figure 3. Percent agricultural land for the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins. Note grey area represents a data gap in the Canadian land use data set for this time period.
Source: T. Brown, NRRI. Unpublished.

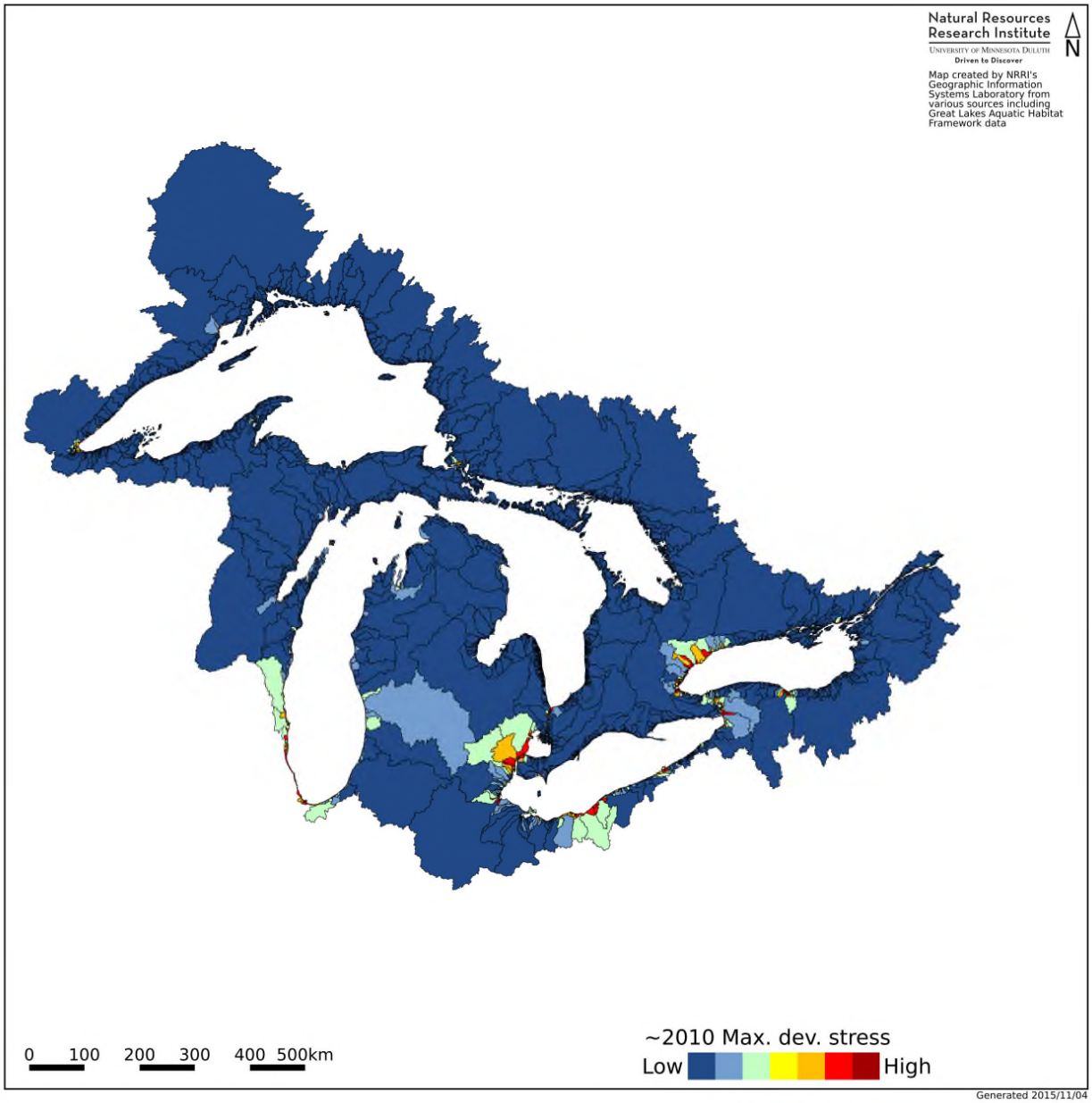


Figure 4. MaxRel Development for the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins. Note grey area represents a data gap in the Canadian land use data set for this time period.
Source: T. Brown, NRRI.

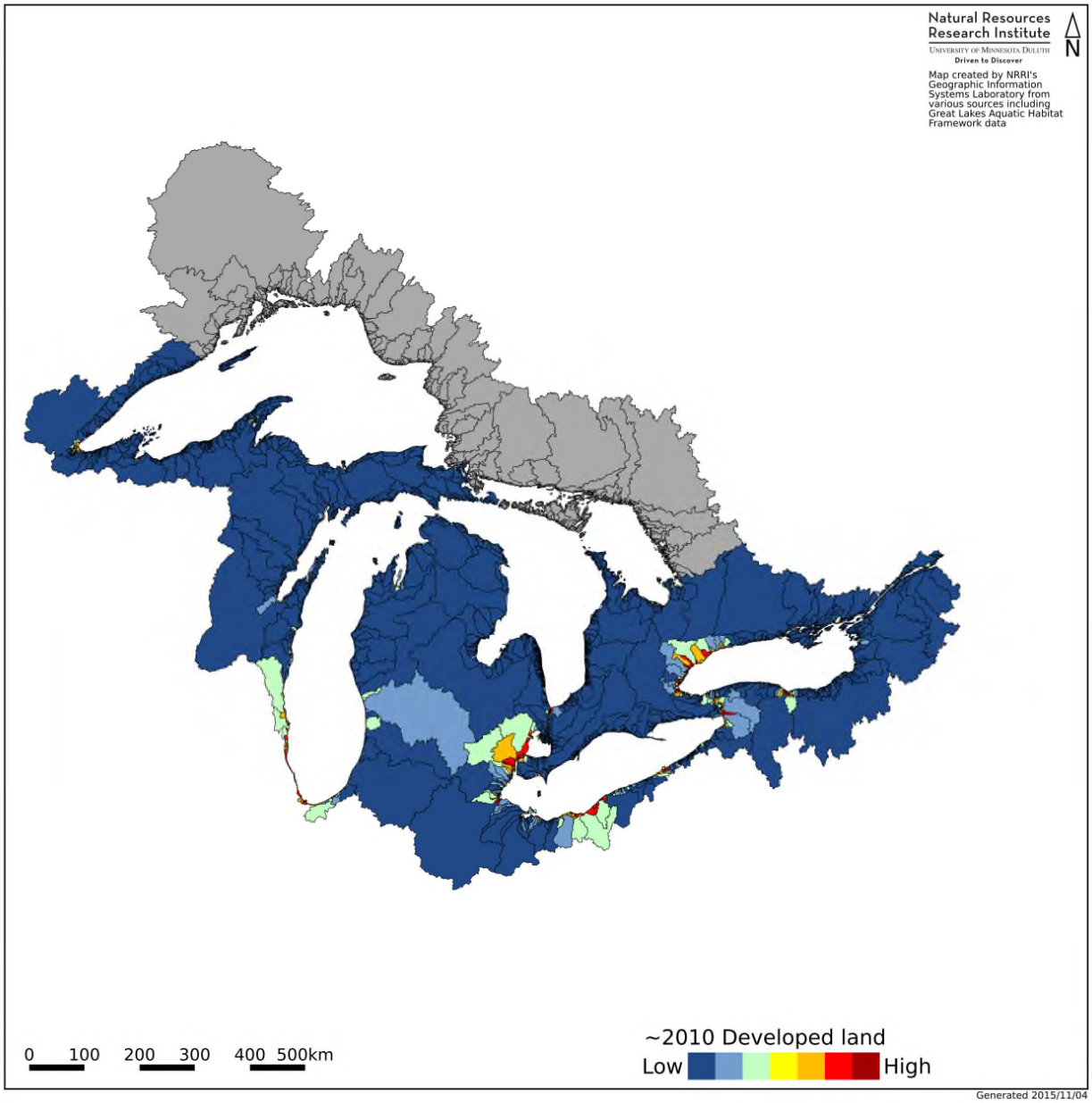


Figure 5. Percent developed land for the Great Lakes basin, 2010. Note the grey area represents a data gap in the Canadian land use data set for this time period. Color classes based on even distribution across 7 bins.
Source: T. Brown, NRRI

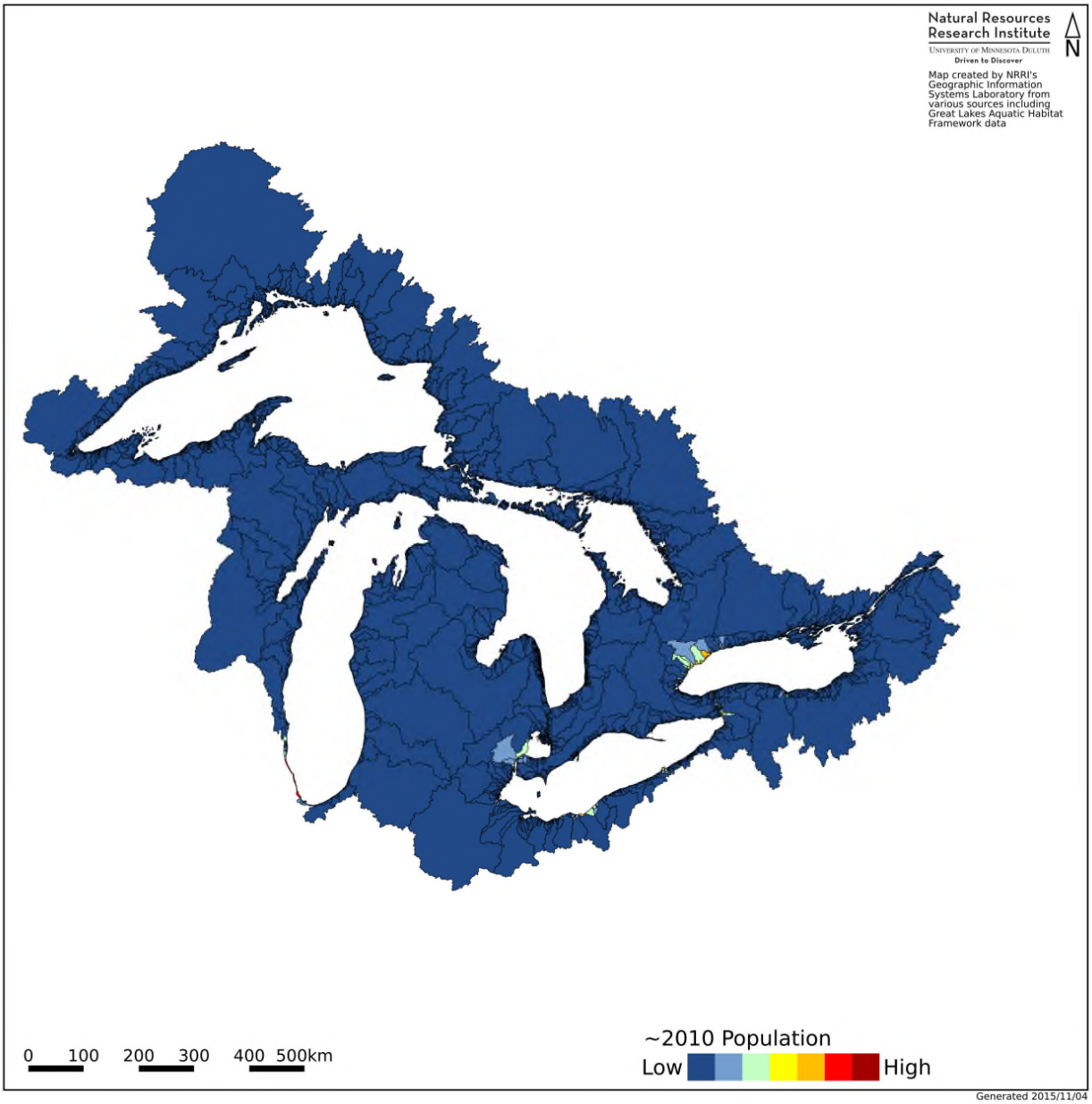


Figure 6. Population density across the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins.
Source: T. Brown, NRRI

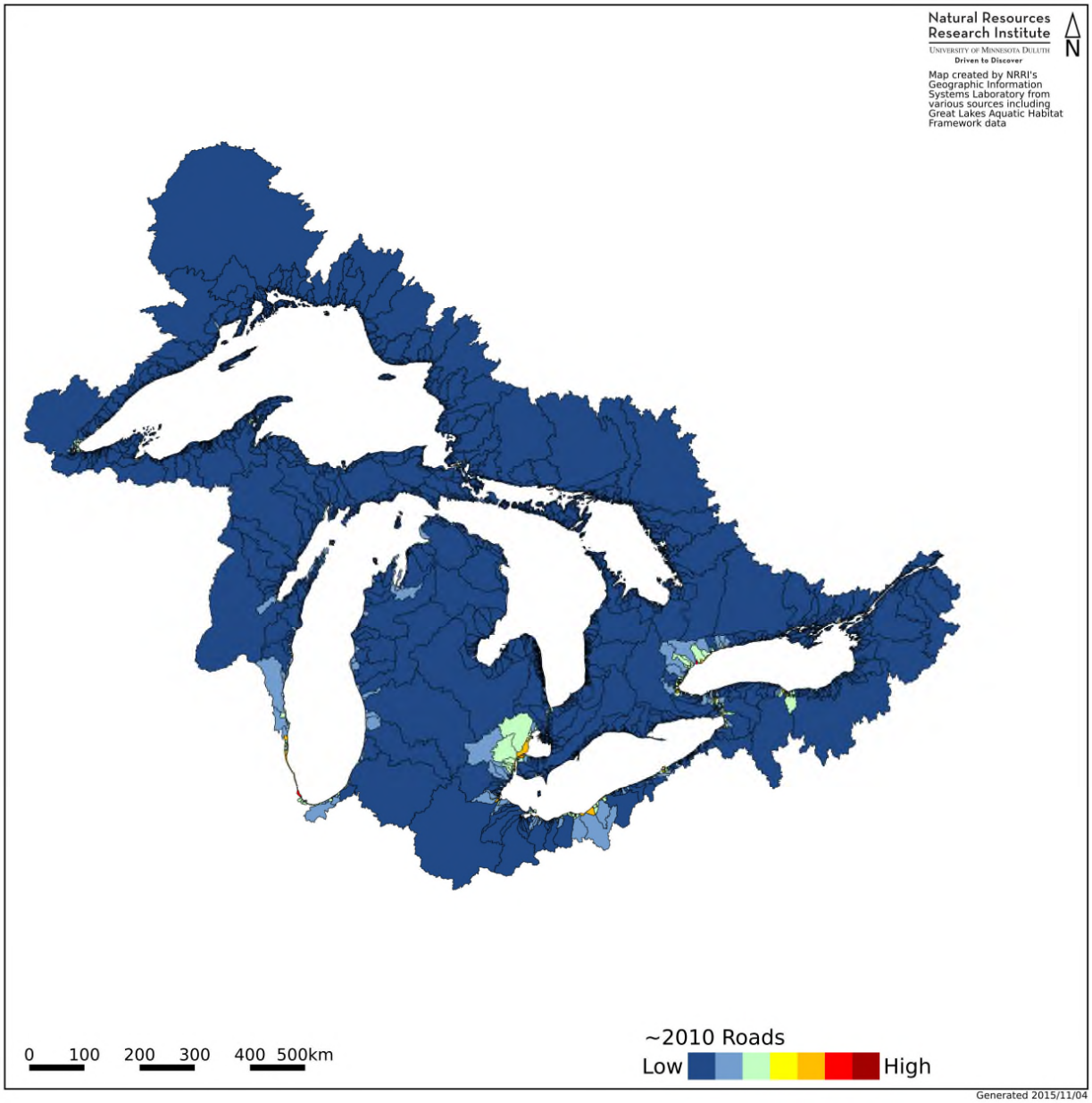


Figure 7. Road density across the Great Lakes basin, 2010. Color classes based on even distribution across 7 bins. Source: T. Brown, NRRI

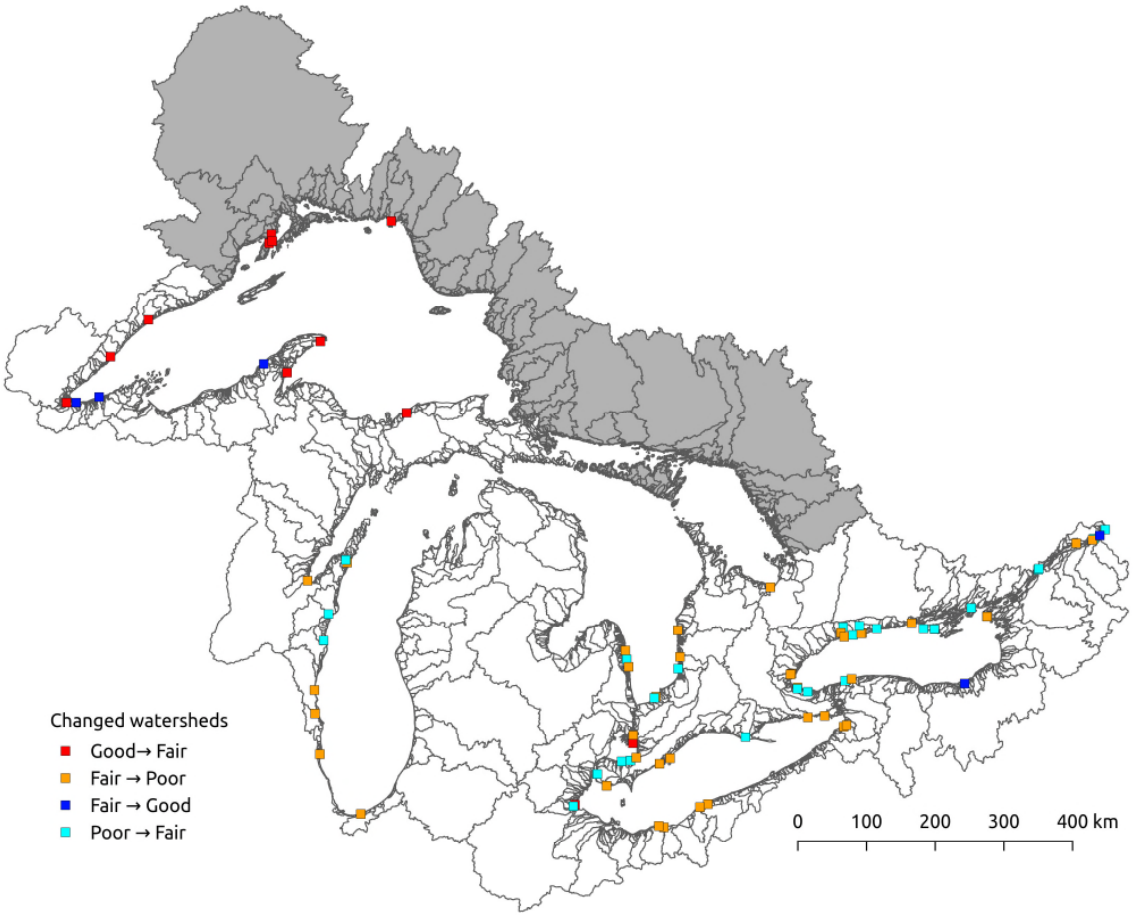


Figure 8. Change in condition from circa ~ 2000 to ~ 2010. Note that there was a data gap in the Ontario land cover data set for the 2010 time period. Change in condition was based on the assumption of 'no change' in agriculture and developed lands for those watersheds (gray), changes shown are driven by population or road density changes. In addition, changes to 34 watersheds on Isle Royale are not shown, as they represent non-existent roads present in the 2010 TIGER dataset, but absent in the 2000 version.
Source: T. Brown, NRRI

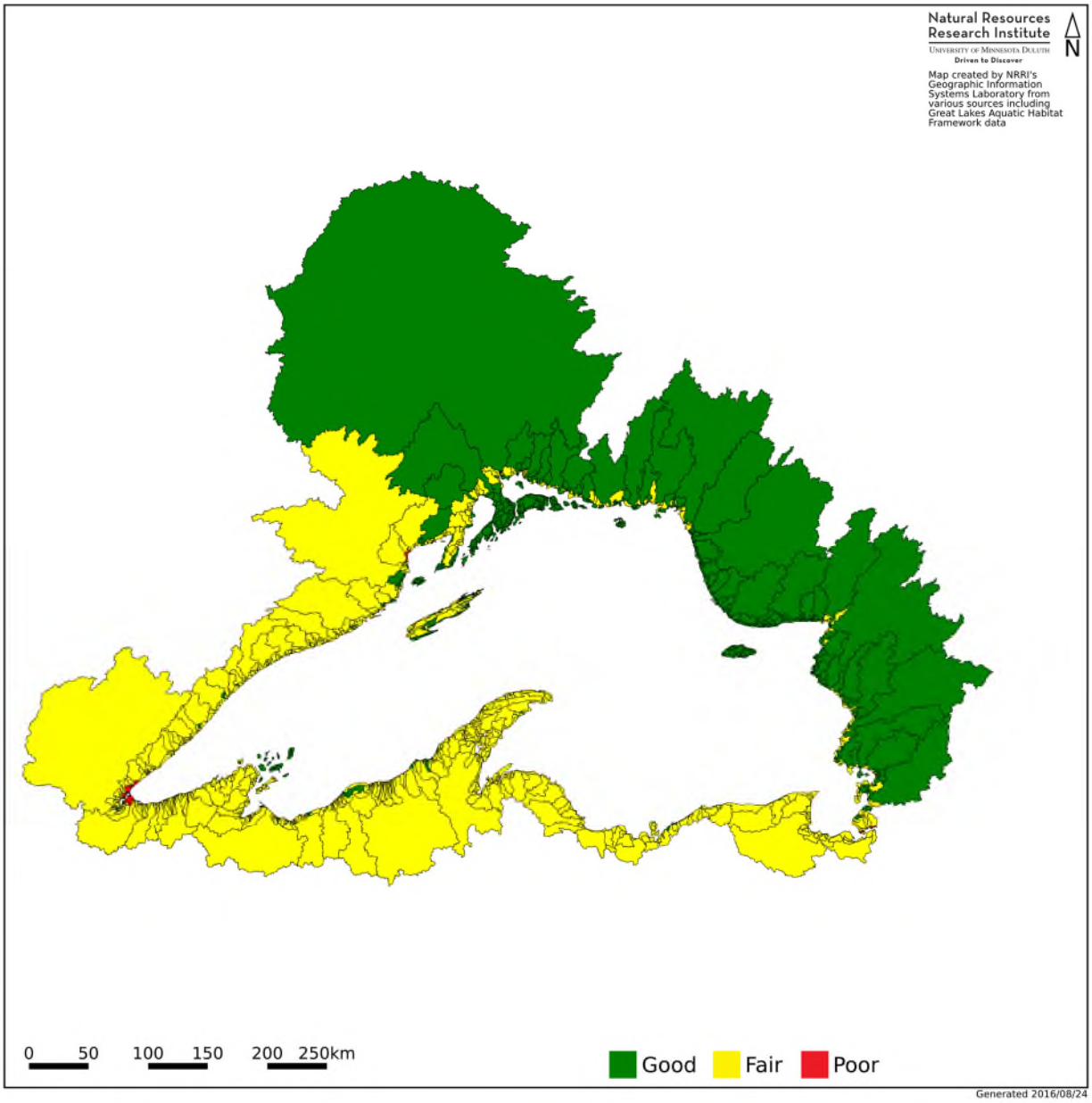


Figure 9. Condition rankings for Lake Superior. circa ~2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: T. Brown, NRRI

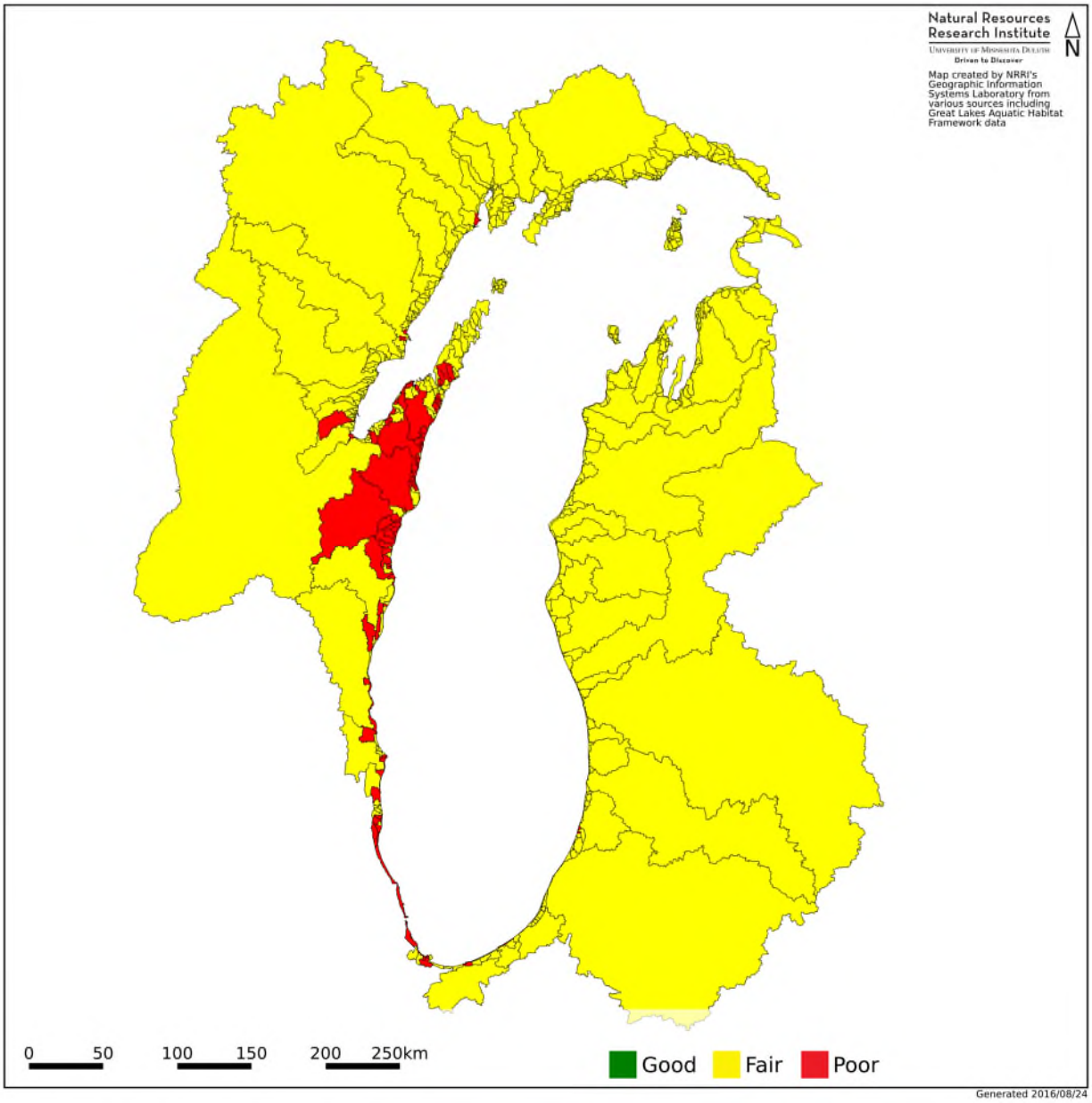


Figure 10. Condition rankings for Lake Michigan watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: T. Brown, NRRI

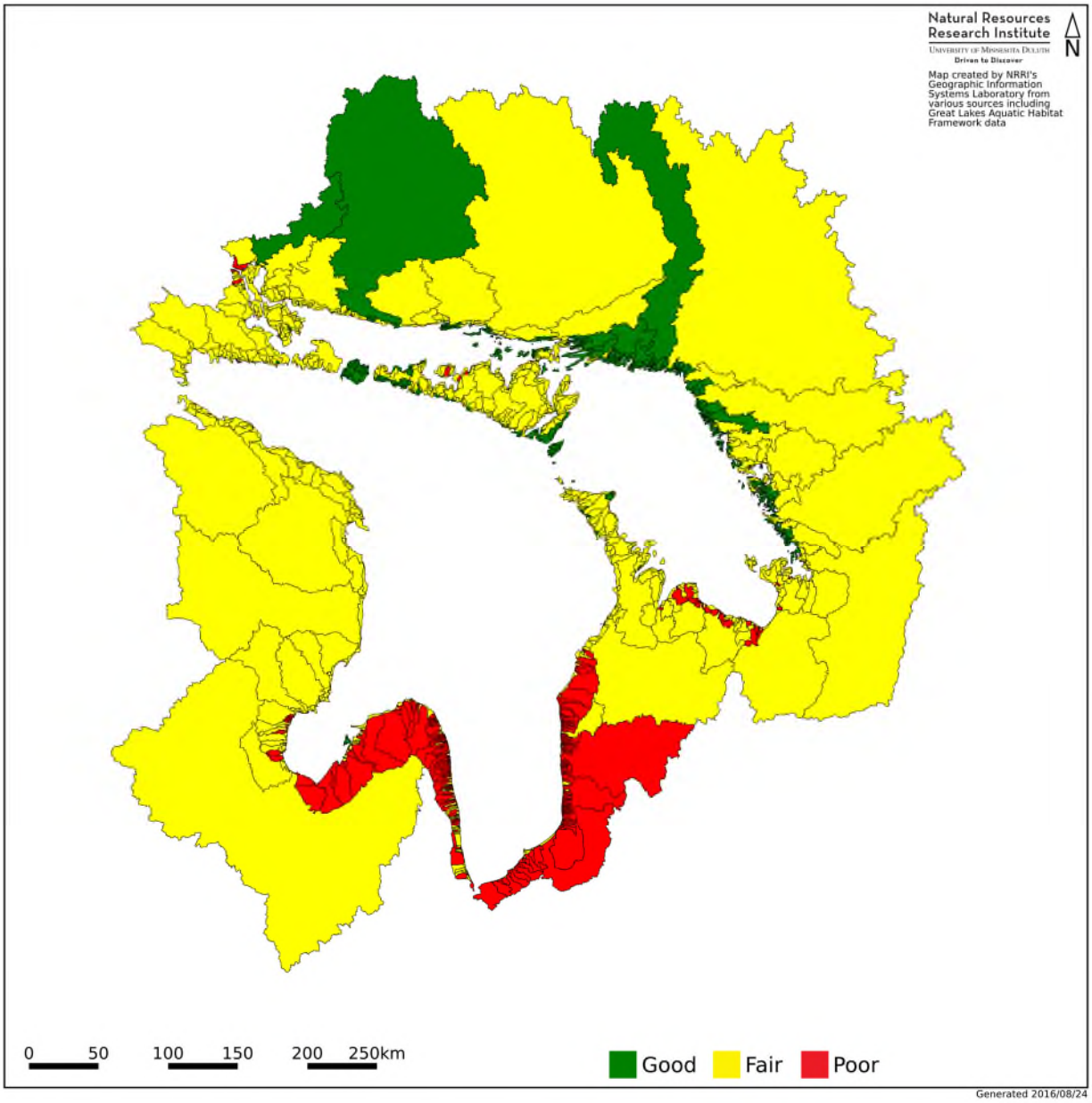


Figure 11. Condition rankings for Lake Huron watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cut-off values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: T. Brown, NRRI

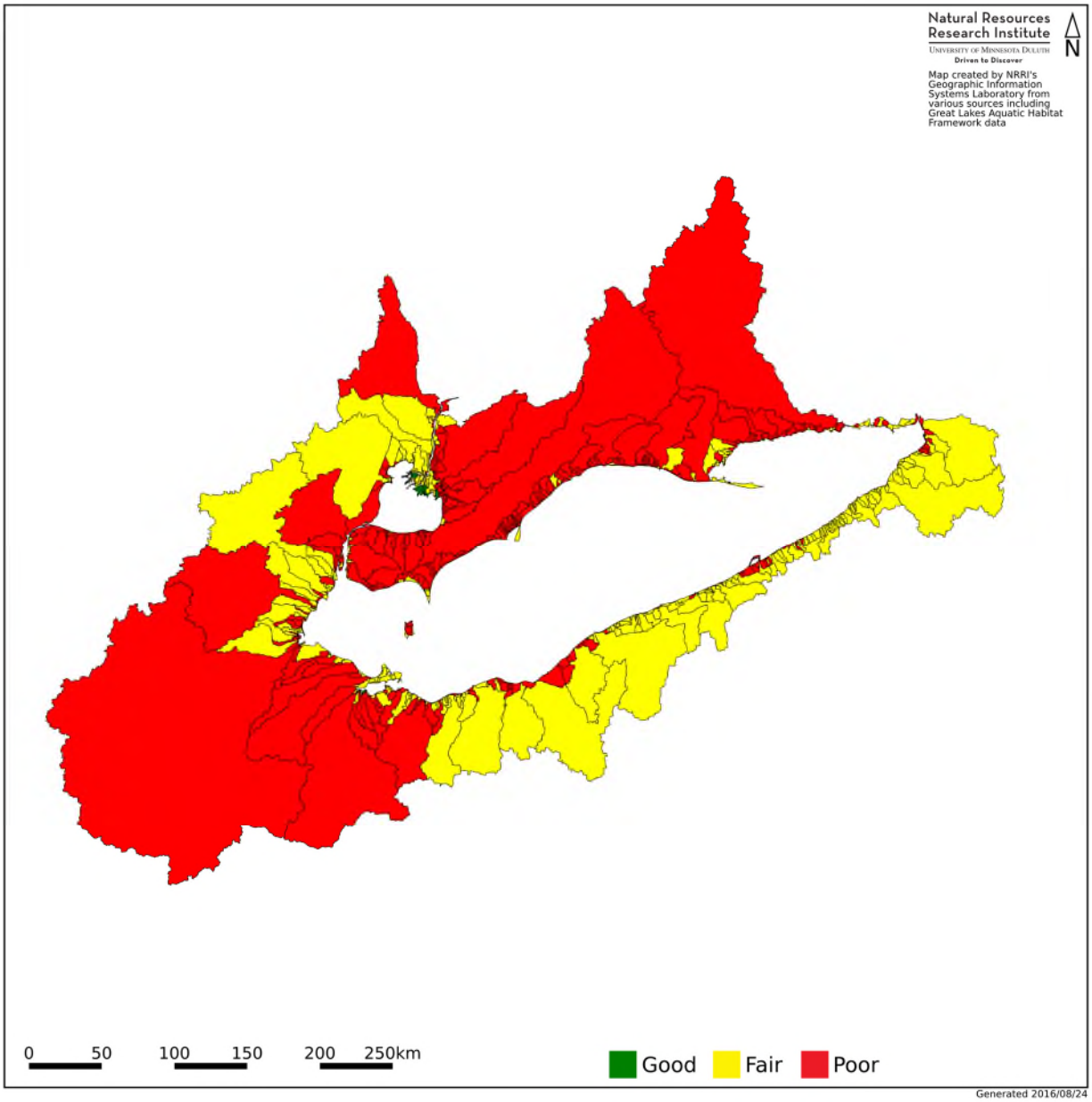


Figure 12. Condition rankings for Lake Erie watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Cut-off values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: T. Brown, NRRI

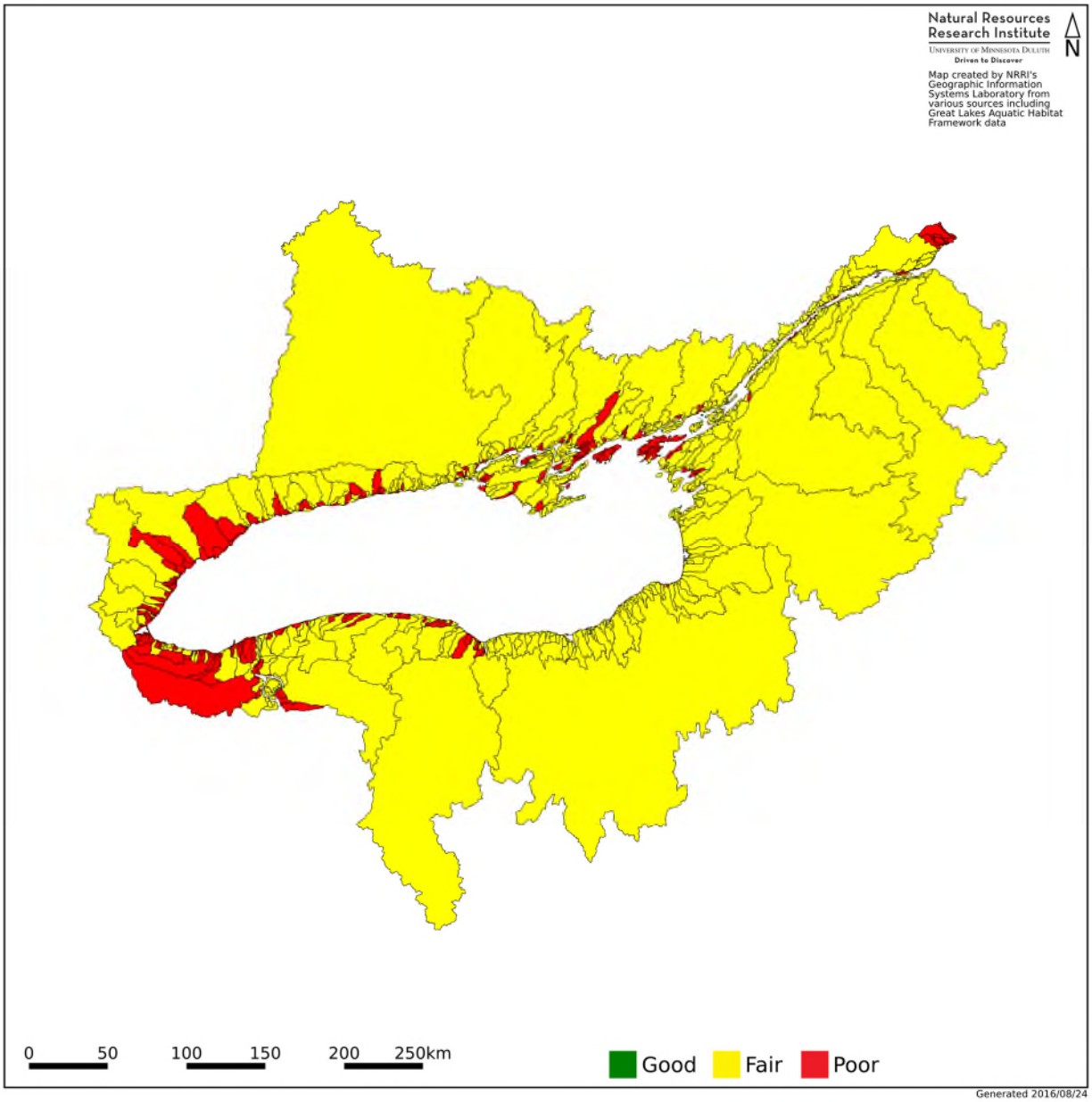


Figure 13. Condition rankings for Lake Ontario watersheds, circa ~ 2010. Classes are based on lower and upper quintiles of the Ag-Dev distribution for the entire Great Lakes basin and then applied to Lake Superior watersheds. Note that watersheds in the St. Lawrence River system were not included in the calculations of the AgDev score, but condition classes are shown for those watersheds. Cutoff values of AgDev score for the boundary of good / fair = 0.01228; fair / poor = 0.673.
Source: T. Brown, NRRI

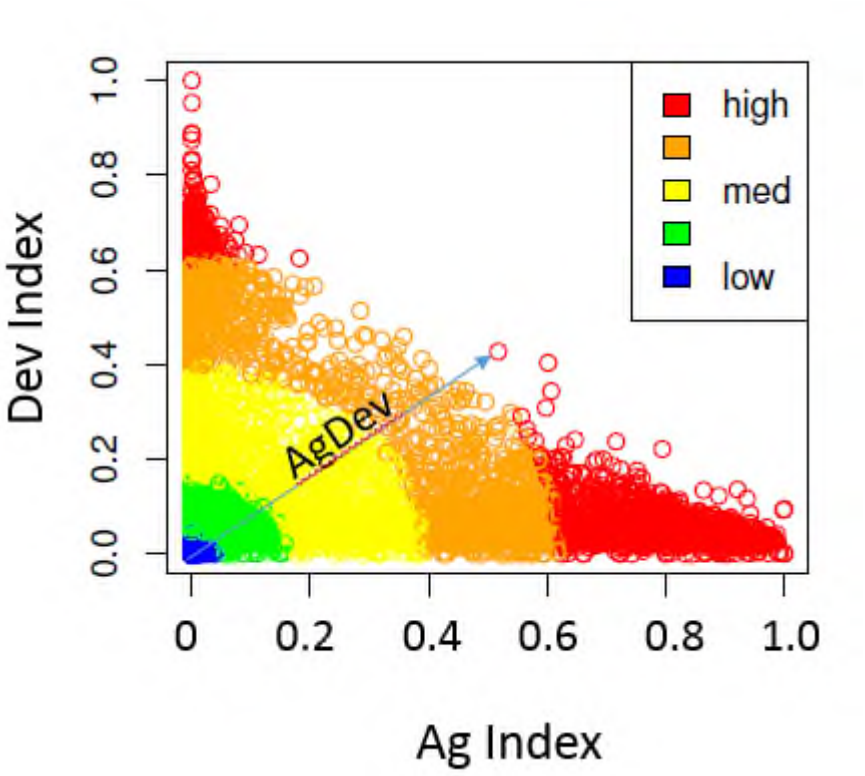


Figure 14. Distribution of Great Lakes watersheds along agricultural and development stressor gradients. Colors are quintiles from low to high stress. The AgDev index is the Euclidean distance from the origin to any given point. Source: NRRI

Sub-Indicator: Base Flow Due to Groundwater

Overall Assessment
Trends:
10-Year Trend: Undetermined
Long-term Trend: Undetermined
Rationale: For stream gages in the U.S. portion of the Great Lakes basin, the Base Flow Index is near the long-term average. Although base flow appears to be increasing in the western part of the basin and decreasing in the central and eastern part of the Great Lakes basin, the changes in Base Flow Index (ratio of base flow to overall streamflow) are small, indicating that climatic variation during the time periods analyzed may be driving the overall base flow values. Specific subwatershed trends have not been determined for each stream gage in the Great Lakes basin, and examination of local trends may indicate impacts on base flow not apparent at the lake-basin scale. The stream gage data used in this analysis were only from the U.S. portion of the basin and the time period from 2005-2013 was compared to records prior to 2005. Other analysis of trends such as annual evaluations may reveal more subtle patterns.

Lake-by-Lake Assessment
Individual lake basin assessments were not prepared for this report.

Trend Assessment Definitions
Increasing: Base Flow Index is increasing
Unchanging: Base Flow Index has not changed over time
Decreasing: Base Flow Index is decreasing
Undetermined: Data are not available to report on a trend

Targets or Endpoint
Anthropogenic factors are not responsible for deviations in the base flow characteristics of sub-watersheds. No quantifiable endpoint or reference value is available at this time.

Sub-Indicator Purpose
- This sub-indicator estimates the contribution of base flow to total stream flow by sub-watershed (lake-scale)
- To detect changes in the source of water (base flow versus direct runoff) to total streamflow. Large changes in base flow or base flow-index would indicate potential changes in the groundwater contribution to streamflow that could affect the ecological function of the stream or riparian users.

Ecosystem Objective
The capacity of base flow to maintain in-stream flow conditions and aquatic habitat at, or near potential is not compromised by anthropogenic factors.

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
Aquatic ecosystems in the streams of the Great Lakes basin have developed in response to natural variations in flow including low-flow conditions. Because base flow maintains streamflow volume and helps regulate stream temperature, it is considered important in maintenance of aquatic ecosystems. Long term average base flow relative to streamflow is referred to as a Base Flow Index. The Base Flow Index is a dimensionless value between 0 and 1 where increasing values of the index indicate increasing groundwater discharge and base flow; e.g. a Base Flow Index value of 0.28 implies that 28% of stream flow is estimated to be from groundwater sources or base flow. Significant extents of sand and gravel within a watershed often result in relatively large values of base flow index while significant extents of clay often result in relatively small values. Human impacts on base flow can potentially be detected using trend analysis of base flow, however, climatic factors can also result in Base Flow Index changes

over time.

Ecological Condition

Background

Base flow provides a relatively stable supply of usually high quality water with stable temperatures. The Base Flow Index indicates how much of a stream's flow comes from base flow compared to total streamflow. A high Base Flow Index indicates that direct runoff contributes little to overall streamflow. Changes in the Base Flow Index through time indicate variability in the relative contribution of base flow and runoff to streamflow, and these changes may have implications for the health of stream ecosystems. Changes in the Base Flow Index through time may result from climate variability and (or) anthropogenic influences.

A significant portion of precipitation over the inland areas of the Great Lakes basin returns to the atmosphere by evapotranspiration. Water that does not return to the atmosphere either flows across the ground surface or infiltrates into the subsurface and recharges groundwater. Water that flows across the ground surface discharges into surface water features (rivers, lakes, and wetlands) and then flows toward and eventually into the Great Lakes. Water that infiltrates into the subsurface and recharges groundwater also results in flow toward the Great Lakes. Groundwater contributes more than 50 percent of the flow in streams that discharge into the Great Lakes (Grannemann et al. 2000). Accordingly, groundwater is a critical factor in the establishment and maintenance of aquatic habitat (Mortsch et al. 2003) because groundwater discharge often provides thermal refuges for fish in summer when temperatures are high. Most recharged groundwater flows at relatively shallow depths at local scales and discharges into adjacent surface water features as base flow. However, groundwater also flows at greater depths at regional scales and discharges either directly into the Great Lakes or into distant surface water features. The quantities of groundwater flowing at these greater depths can be significant locally but are generally believed to be modest relative to the quantities flowing at shallower depths.

The component of stream flow due to runoff from the ground surface is rapidly varying and transient, and results in the peak discharges of a stream. Groundwater discharge to surface water features in response to precipitation is greatly delayed relative to surface runoff. The stream flow resulting from groundwater discharge is, therefore, more uniform. In the Great Lakes region, groundwater discharge is often the dominant component of base flow. Base flow is the less variable and more persistent component of total streamflow (example hydrograph in figure 1).

For this report, base flow represents the consistent part of streamflow that is not surface-runoff. In the Great Lakes region, groundwater discharge is often the dominant component of base flow. However, anthropogenic activities such as flow regulation, the storage and delayed release of water using dams and reservoirs, creates a steady streamflow signature that is similar to that of groundwater discharge. This analysis does not distinguish between base flow from groundwater discharge and base flow-type streamflow patterns resulting from anthropogenic activities.

Status of Base Flow

Base flow is frequently determined using hydrograph separation. This process uses streamflow monitoring information as input and partitions the observed flow into rapidly and slowly varying components, i.e., surface runoff and base flow, respectively. Figure 1 illustrates the daily streamflow monitoring information and the results of hydrograph separation for the Nith River at New Hamburg, Ontario, for January 1 to December 31, 1993. The rapidly varying response of streamflow to precipitation and snow melt are in contrast to the more slowly varying base flow.

Application of hydrograph separation to daily streamflow monitoring information results in lengthy time series of output. Various measures are used to summarize this output. For example, the Base Flow Index is a simplified summary of the relative contribution of base flow to streamflow that is appropriate for use in regional scale studies (Neff et al., 2005). Base Flow Index is defined as the average rate of base flow relative to the average rate of total streamflow, has no units, and varies from zero to one where increasing values indicate an increasing contribution of base flow to streamflow. The value of Base Flow Index for the data shown in Figure 1 is 0.28, which implies that 28% of the observed flow is estimated to be base flow

For this sub-indicator, five base flow separation techniques were applied to daily mean streamflow data from 227 gages operated by the U.S. Geological Survey (USGS) in the U.S. portion of the Great Lakes basin. The five techniques were: BFI (Base Flow Index method or the UKIH method, Wahl and Wahl, 2003), HYSEP local minimum, HYSEP fixed interval, and HYSEP sliding interval (Sloto and Crouse, 1996), and PART (Rutledge, 1998). The 227 gages were selected according to the following criteria; the gages: (1) had data available in the

USGS NWIS database, (2) were included in the Neff et al. (2005) report, and (3) had at least 3 continuous years of complete daily streamflow data for each time period (pre-2005 and 2005-2013). The Base Flow Index in each time period for each basin is the mean of five base flow separation techniques. The gages are a subset of the 959 gages that Neff et al. (2005) used to develop a regression model to predict base flow in ungagged basins (Figure 2). The pre-2005 values presented in this report are not taken directly from Neff et al. (2005), but were re-calculated using the same 5 base flow separation methods as applied to the 2005-2013 data. The base flow indices in this report may therefore be slightly different from the Neff et al (2005) report, but are well-suited for a direct comparison between pre-2005 and 2005-2013 (Figure 3 and Table 1).

When the base flow indices for all 227 gages are aggregated together, there is very little change in the overall mean Base Flow Index, suggesting there was not a consistent shift in Base Flow Index for the entire U.S. portion of the Great Lakes basin (Table 1). The change in the mean and maximum Base Flow Index values using any of the analysis methods are generally less than one percent. The estimated minimum Base Flow Index value is sensitive to small changes and has a higher percent change. Overall, this simple comparison does not indicate significant trend in the sub-indicator. However, many individual gages show differences in the Base Flow Index between the two time periods (Table 2). Of the 227 gages, 142 had increases or decreases in all 5 base flow separate techniques. This change was considered a “strong” signal and these basins are shown as red or blue points in Figure 3 and are summarized in Table 2. The largest within-basin negative change was -0.15 and the largest positive change was 0.15. This value represents a large change in Base Flow Index (recalling that the scale goes from 0 to 1). The positive and negative responses show some spatial clustering (Figure 3) suggesting that regional climatic patterns and (or) regional land use changes are driving at least part of the change in Base Flow Index. However, this entire analysis is very coarse and a more rigorous analysis of individual streamflow records at a more refined time interval may be necessary to identify trends.

Additional information is also required to determine the extent to which human activities have impaired groundwater discharge. Change in base flow over time may be subtle and difficult to quantify (e.g., variations in the relation of base flow to climate) and may be continuous (e.g., a uniform increase in base flow due to aging water supply infrastructure and increasing conveyance losses) or discrete (e.g., an abrupt reduction in base flow due to a new consumptive water use). Change may also be the result of cumulative impacts due to a range of historical and ongoing human activities, and may be more pronounced and readily detected at local scales than at the scales that are typical of continuous stream flow monitoring.

The discharge of groundwater to surface water features is the end-point of the process of groundwater recharge, flow, and discharge. Human activities impact groundwater discharge by modifying the components of this process where the time, scale, and to some extent the severity, of these impacts is a function of hydrogeological factors and the proximity of surface water features. Increasing the extent of impervious surfaces during residential and commercial development and installation of drainage to increase agricultural productivity are examples of activities that may reduce groundwater recharge and ultimately groundwater discharge.

Withdrawals of groundwater as a water supply and during dewatering (pumping groundwater to lower the water table during construction, mining, etc.) remove groundwater from the flow regime and may also reduce groundwater discharge. Groundwater discharge may be impacted by activities such as the channelization of water courses that restrict the motion of water across the groundwater and surface water interface. Human activities also have the capacity to intentionally, or unintentionally, increase groundwater discharge. Induced storm water infiltration, conveyance losses within municipal water and wastewater systems, and closure of local water supplies derived from groundwater are examples of factors that may increase groundwater discharge. Climate variability and change may compound the implications of human activities relative to groundwater recharge, flow, and discharge.

Linkages

Base flow due to the discharge of groundwater to the rivers, inland lakes and wetlands of the Great Lakes basin is a significant and often major component of stream flow, particularly during low-flow periods. Base flow frequently satisfies flow, level, quality and temperature requirements for aquatic species and habitat. Water supplies and the capacity of surface water to assimilate wastewater discharge are also dependent on base flow. Base flow due to groundwater discharge is therefore critical to the maintenance of water quantity, quality, and integrity of aquatic species and habitat. Pressures such as urban development and water use, in combination with the potential for climate change impacts, may alter base flow. Reductions in base flow may compromise the assimilative capacity of surface water for wastewater discharge during periods of otherwise low flow and result in reduced water quality.

The higher the Base Flow Index value, the less sensitive the watershed is to climate change because these areas have a higher buffer capacity. Areas with low base flow index values are more sensitive to climate change based on the Sensitivity Mapping and Local Watershed Assessments for Climate Change Detection and Adaptation Monitoring report.

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 Limitations

- Data are largely available. The limitation is the significant processing and interpretation that is needed to assess base flow.
- Streamflow monitoring of the full land mass is neither technologically nor economically feasible.
- Methods of determining base flow from total streamflow data are not standardized and the use of differing methods may produce inconsistent results.
- Differing summary statistics of base flow may also yield inconsistent results.
- Changes in base flow are a delayed indication of changes in net groundwater recharge (recharge from precipitation minus water withdrawal by pumping) and, in some settings, changes in base flow due to anthropogenic factors may not be evident for extended periods of time.
- Sub-basin (lake) scale averages of base flow may not be particularly useful targets because geological conditions within the sub-basins can be quite variable.
- Water management practices such as flow regulation replicate base flow characteristics and disable the calculations of natural base flow in sub-watersheds where these practices are significant.
- Wastewater discharge similarly disables the calculation of natural base flow in sub-watersheds where this discharge is significant.
- Statistical methods are used to detect changes in indices of base flow with respect to time.
- These methods are also used to differentiate natural factors from anthropogenic factors as the cause of these changes.
- Natural factors such as climate variability modify both average rates of base flow and the annual distribution of flow.
- A consistent, standard method needs to be developed and implemented to assist in the understanding and documentation of the results.
- Necessary to devise a better form for summary for reporting at lake-scale, as much of the spatial variation of base flow index and human impacts are lost at this scale when summarized as an average.

Additional Information

Base flow is a proxy for groundwater discharge in areas where the extents of other water storage and release mechanisms that generate base flow are modest.

Base Flow Index is frequently (but not always) easier to use than base flow because it directly communicates the contribution of groundwater to streamflow and is independent of factors such as size of the watersheds and total precipitation.

Base flow is the more slowly varying component of total streamflow and is often attributed to groundwater discharge to wetlands, lakes, and rivers. Base flow is determined from total streamflow data using mathematical algorithms. Unlike point measurements of groundwater levels, base flow is an integrated measure of groundwater conditions and impacts upstream of the streamflow gage. Various anthropogenic factors can impact the base flow characteristics of a sub-watershed. For example, increasing extents of paved and other impermeable surfaces due to urban development can reduce recharge and therefore decrease base flow. In contrast, conveyance losses (the quantity of water that is lost in the subsurface in municipal water and wastewater systems) can increase base flow. Anthropogenic factors in rural settings such as tile drainage and changes in vegetation coverage can impact base flow. The withdrawal of groundwater by pumping or through the drainage of quarries and other excavations can also impact base flow.

Analysis of base flow can potentially indicate consumptive water use within the gaged watersheds with high groundwater withdrawals.

Groundwater has important societal and ecological functions across the Great Lakes basin. Groundwater is typically a high quality water supply that is used by a significant portion of the population, particularly in rural areas where it is often the only available source of water. Groundwater discharge to rivers, lakes, and wetlands is also critical to aquatic species and habitat and to in-stream water quantity and quality. These functions are concurrent and occasionally conflicting.

Pressures such as urban development and water use, in combination with the potential for climate impacts and further contamination of the resource, may increase the frequency and severity of these conflicts. In the absence of systematic accounting of groundwater supplies, use, and dependencies, it is the ecological function of groundwater that is most likely to be compromised.

Managing the water quality of the Great Lakes requires an understanding of water quantity and quality within the inland portion of the basin, and this understanding requires recognition of the relative contributions of surface runoff and groundwater discharge to stream flow. The results described in this report indicate the significant contribution of groundwater discharge to flow within the tributaries of the Great Lakes.

The dynamics of groundwater flow and transport are different than those of surface-water flow. Groundwater discharge responds more slowly to climate and maintains stream flow during periods of reduced water availability, but this capacity is known to be both variable and finite. Contaminants that are transported by groundwater may be in contact with geologic materials for years, decades, and perhaps even centuries or millennia. As a result, there may be considerable opportunity for attenuation of contamination prior to discharge. However, the lengthy residence times of groundwater flow also limit opportunities for the removal of contaminants, in general, and non-point source contaminants, in particular.

Further research and analysis is required to determine if the changes in base flow noted are due to human causes and to determine conditions on a lake-by-lake basis.

Recent investigations on trends in streamflow characteristics (Hodgkins et al. 2007) could be expanded to the Canadian part of the basin. Similarly, analyses of trends in groundwater recharge (Rivard et al. 2009) could be completed in greater detail across both the Canadian and U.S. portions of the basin.

The Base Flow Index and the Tributary Flashiness Index may provide the opposite type of information that, in combination, may provide a more complete assessment of the streamflow conditions to support General Objective 9. In theory, these two sub-indicators inversely correlate, i.e. if tributary flashiness is high than base flow is low.

Acknowledgments

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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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List of Tables

Table 1. Comparison of Base Flow Index for 227 gages with at least 3 continuous years of daily streamflow data in two time periods: pre-2005 and 2005 to 2013. The gages are all in the U.S. portion of the Great Lakes basin and were included in the Neff et al. (2005) analysis of base flow.
Source: unpublished data from J. Trost, U.S. Geological Survey

Table 2. Number of gages (grouped by the magnitude of the change) in which all 5 base flow separation methods indicated either an increase or decrease in Base Flow Index from pre-2005 to 2005-2013. These correspond to the red and blue points in figure 3.
Source: unpublished data from J. Trost, U.S. Geological Survey

List of Figures

Figure 1. Hydrograph of observed total stream flow (black) and calculated base flow (red) for the Nith River at New Hamburg during 1993.
Source: Environment Canada and the U.S. Geological Survey.

Figure 2. Location of gages with streamflow records used in Neff et al. (2005) to predict base flow in ungagged basins.
Source: Environment Canada and the U.S. Geological Survey (Neff et al., 2005).

Figure 3. Difference in Base Flow Index for selected stream gages on the U.S. part of the Great Lakes Basin for the period 2005-2013 and data prior to 2005.
Source: unpublished analysis from J. Trost, U.S. Geological Survey

Last Updated

State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

Method	Pre-2005			2005-2013		
	minimum	mean	maximum	minimum	mean	maximum
BFI	0.071	0.575	0.955	0.120	0.578	0.958
HYSEP fixed interval	0.257	0.708	0.957	0.241	0.702	0.960
HYSEP local minimum	0.226	0.642	0.955	0.196	0.633	0.954
HYSEP sliding	0.259	0.708	0.957	0.231	0.702	0.959
PART	0.064	0.711	0.965	0.058	0.710	0.966

Table 1. Comparison of Base Flow Index for 227 gages with at least 3 continuous years of daily streamflow data in two time periods: pre-2005 and 2005-2013. The gages are all in the U.S. portion of the Great Lakes Basin and were included in the Neff et al. (2005) analysis of base flow.
Source: Neff et al. (2005) and U.S. Geological Survey

Magnitude of change between time periods	Increase	Decrease
	Number of gages	
0 to 0.05 change	42	74
0.05 to 0.10 change	7	14
more than 0.10 change	4	1
Total	53	89

Table 2. Number of gages (grouped by the magnitude of the change) in which all 5 base flow separation methods indicated either an increase or decrease in Base Flow Index from pre-2005 to 2005-2013. These correspond to the red and blue points in figure 3.
Source: U.S. Geological Survey

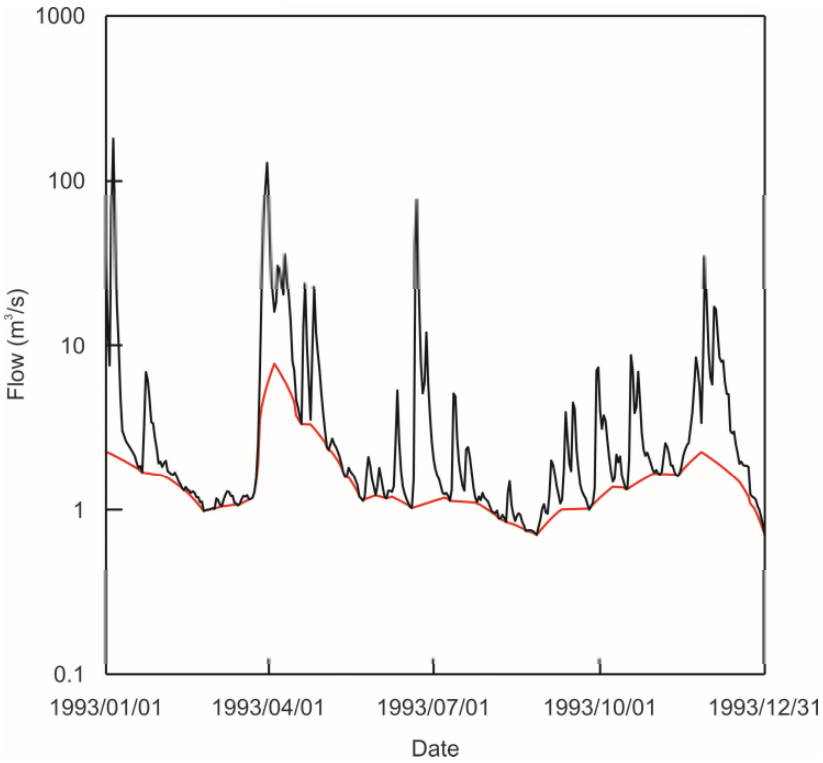


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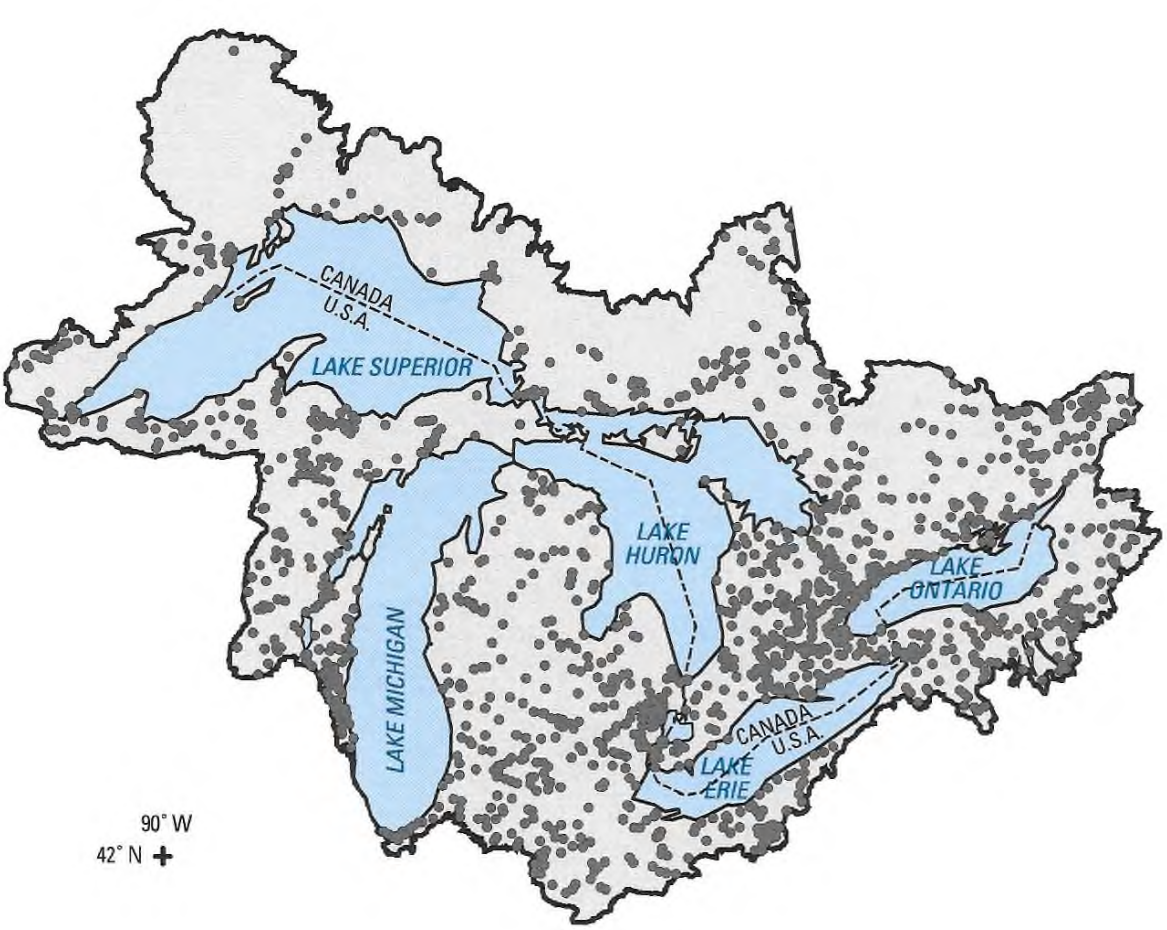


Figure 2. Location of gages with streamflow records used in Neff et al. (2005) to predict base flow in ungagged basins.

Source: Environment Canada and the U.S. Geological Survey (Neff et al., 2005)

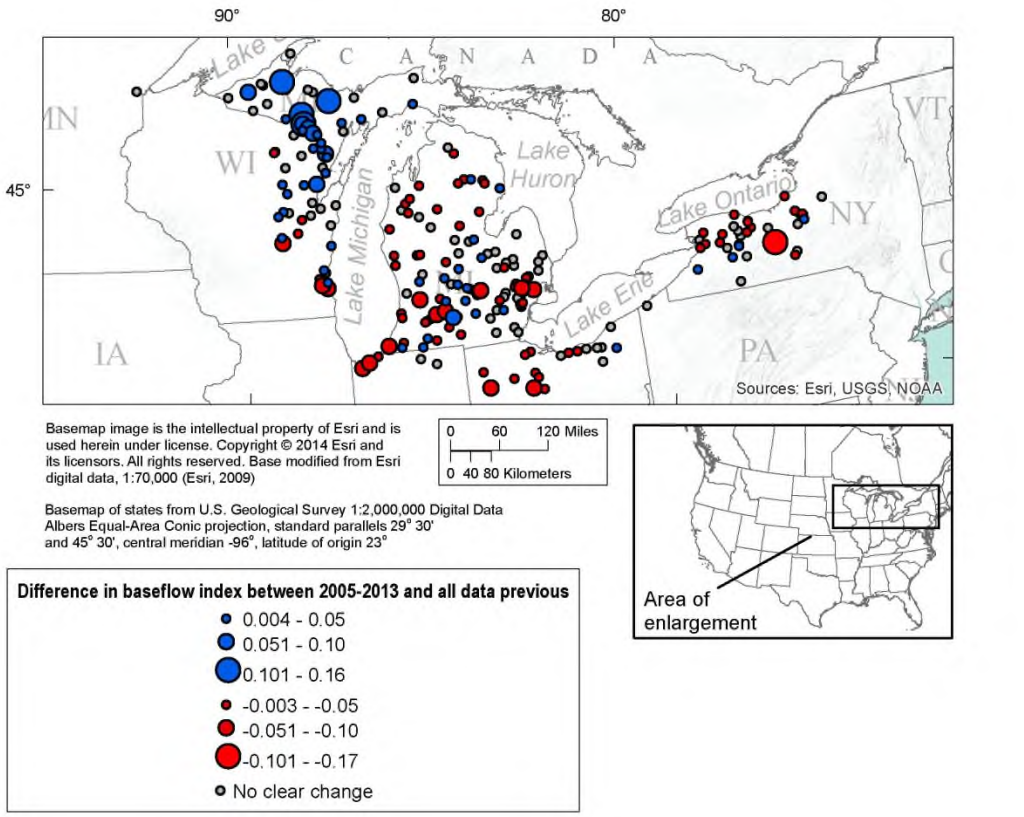


Figure 3. Difference in Base Flow Index for selected stream gages on the U.S. portion of the Great Lakes basin for the period 2005-2013 and data prior to 2005.
Blue = higher in 2005-2013 than previous
Red = lower BFI in 2005-2013 compared to previous.
Source: unpublished analysis from J. Trost, U.S. Geological Survey

Sub-Indicator: Tributary Flashiness

Overall Assessment
Trends:
10-Year Trend: Unchanging
Long-term Trend (1958-2017; some rivers have shorter data period): Unchanging
Rationale: The R-B Flashiness Index (RBI) was either unchanging or decreasing in most rivers in the long-term (18/27), and most rivers had improved or remained unchanged for the 10-year trend (13/27). Attention should focus on 3 rivers that had deteriorating trends (10-year trends are increasing and long-term trends unchanging or decreasing), which include the Muskegon (Michigan), Portage (Erie), and Cattaraugus (Erie) Rivers. Also, efforts should focus on 6 rivers that have had continued increasing trends in the short- and long-term, which include the Manistee (Michigan), Pere Marquette (Michigan), White (Michigan), Thames (Erie), Seneca (Ontario), and Don (Ontario).

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

Lake-by-Lake Assessment
Individual lake basin assessments were not prepared for this report because rivers appropriate for the calculation of tributary flashiness (long-term data, few dams, not too small) cover less than half of any of the lake basins.

River-by-River Assessment – Lake Superior
Pic River (CAN)
10-Year Trend: Unchanging
Long-term Trend (1971-2016): Unchanging
Rationale: The RBI exhibits no significant trend over either the long-term (p=0.69) and the average RBI decreased only slightly from 1998-2007 (0.106) to 2008-2016 (0.099).

Pigeon River (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017): Unchanging
Rationale: There is no significant trend in RBI (p=0.21) over the long-term nor has the average RBI changed in the past two decades (both are 0.112).

River-by-River Assessment – Lake Michigan
Fox River (US)
10-Year Trend: Decreasing
Long-term Trend (1989-2017): Unchanging
Rationale: The long-term trends were not significant (p=0.10); however, there is a substantial decrease in RBI from 1998-2007 (0.152) compared to 2008-2017 (0.134). In the last report, the Fox River was unchanging in the 10-year trend so this indicated a slight improvement.

Muskegon River (US)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Decreasing
Rationale: Although the long-term trend is significantly downward (r=-0.63; p<0.001), there is a significant increase since 1998 (r=0.80; p<0.001) and the average RBI increased substantially from 1998-2007 (0.062) to 2008-2017 (0.080).

Manistee River (US)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Increasing

Rationale: The long-term trend is upward (r=0.52; p<0.001), and the largest increases in average RBI have occurred in the past two decades (0.039 in 1998-2007 and 0.043 in 2008-2017). Though the past 2 years have had slightly lower RBIs (0.035 and 0.033, respectively) indicating this trend may be shifting.

Pere Marquette River (US)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Increasing
Rationale: There is a significant upward long-term trend in RBI (r=0.59; p<0.001), and the most recent decade (2008-2017) is higher on average (0.056) compared to the past decade (1998-2007; 0.051). However, since 2014 when RBI was 0.062, each year has been successively lower (0.056, 0.052, 0.047) indicating this trend may be shifting as in the Manistee.

White River (US)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Increasing
Rationale: The RBI has a significant upward trend (r=0.48; p=0.009), and the average RBI has increased primarily in the past decade from 0.067 in 1998-2007 to 0.078 in 2008-2017.

Escanaba River (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017): Decreasing
Rationale: The long-term trend is significantly downward (r=-0.71; p<0.001), though the decreases were largely in the late 1960s and the average RBI is similar between 1998-2007 (0.105) and 2008-2017 (0.109).

Grand River (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017): Decreasing
Rationale: There is a significant downward long-term trend in RBI (r=-0.30; p=0.014), and although there was a slight increase in the mid-1990s, the average RBI was similar between 1998-2007 (0.076) and 2008-2017 (0.074). The downward 10-year trend in the last report has dissipated and is now unchanging.

River-by-River Assessment – Lake Huron

French River (CAN)
10-Year Trend: Unchanging
Long-term Trend (1962-2017): Unchanging
Rationale: Neither the long-term trend (p=0.21) nor the average RBI have changed substantially from 1998-2007 (0.024) to 2008-2017 (0.026).

Au Sable River (US)
10-Year Trend: Unchanging
Long-term Trend (1997-2017): Increasing
Rationale: There is a significant long-term increase in RBI starting in 1997 (r=0.44; p=0.047), yet that increase largely occurred from 2000-2006 and over the past decade RBI has remained fairly stable with an average of 0.051.
Since the Last Report: The long-term increasing trend was not significant in the past report.

Magnetawan River (CAN)
10-Year Trend: Unchanging
Long-term Trend (1973-2017): Unchanging
Rationale: There is not a significant long-term trend since 1973 (p=0.94) and the average RBI has been similar across all 4 decades (0.049-0.052).

Maitland River (CAN)
10-Year Trend: Unchanging
Long-term Trend (1958-2017; missing 1997-2001): Unchanging
Rationale: There is no significant long-term trend in RBI (p=0.66), and the average RBI from 1985-1996 was identical to 2003-2014 (0.291). Data were missing for 1997-2002.

Thunder Bay River (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017; missing 1981-2001): Unchanging
Rationale: The RBI has no significant long-term trend (p=0.21) even though there were substantial decreases in the earlier record that were not maintained. Since measurements restarted in 2002, the average RBI from 2002-2007 (0.081) was very similar to past 10 years (0.087). This trend is very similar to the past report and the recent years have led to a non-significant decrease in the early record.

Wanapitei River (CAN)
10-Year Trend: Decreasing
Long-term Trend (1958-2017): Unchanging
Rationale: The long-term data exhibit no significant trend (p=0.64). The recent average RBI (2008-2017) was much lower (0.068) than the past decade (0.080 from 1998-2007). However, the RBI has increased in the past 5 years (average of 0.074) indicating this trend is shifting.

Saginaw River (US)
10-Year Trend: Unchanging
Long-term Trend (1997-2017): Unchanging
Rationale: There is no significant long-term trend since 1997 (p=0.22) and while the average RBI from 2008-2017 (0.178) is lower than the previous 2 decades (0.198, 0.203), RBIs for the past 3 years (0.246, 0.217, 0.205) are higher than prior RBI indicating this trend may be starting to change. The 10-year trend was decreasing in the last report, but high RBIs in the recent 3 years suggest that trend is changing, resulting in an unchanging designation for 2019.

Nottawasaga River (CAN)
10-Year Trend: Decreasing
Long-term Trend (1993-2017): Decreasing
Rationale: The long-term trend since 1993 has decreased significantly (r=-0.462, p=0.02), and the average RBI has decreased each decade from 0.075 in 1988-1997, to 0.070 in 1998-2007, and to 0.065 in 2008-2017. In the past report, the long-term decrease was not significant, but it is significant in this report.

River-by-River Assessment – Lake Erie

River Raisin (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017): Unchanging
Rationale: There is no significant long-term trend in RBI (p=0.79), and the average RBI was similar between 1998-2007 (0.164) and 2008-2017 (0.160).

Grand River (OH-US)
10-Year Trend: Unchanging
Long-term Trend (1975-2017): Unchanging
Rationale: There is no significant long-term trend (p=0.22), and the average RBI is similar between 1998-2007 (0.361) and 2008-2017 (0.357).

Maumee River (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017): Increasing
Rationale: Over the long-term record, there has been a significant increase in RBI (r=0.55; p<0.001), yet the past ten years have a similar average RBI (0.280) to 1988-2007 (0.278) and 1998-2007 (0.289). The long-term trend is the same as the past report. The 10-year trend was decreasing previously but recent RBI indicate a slight increase counteracting that trend.

Sandusky River (US)
10-Year Trend: Unchanging
Long-term Trend (1958-2017): Increasing
Rationale: Although the long-term trend is significantly upward (r=0.35; p=0.004), the average RBI was very similar between 1998-2007 (0.381) and 2008-2018 (0.380). As with the Maumee, the long-term trend is the same, but the

previous decreasing 10-year trend is now statistically insignificant.

Thames River (CAN)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Increasing
Rationale: There is a significant long-term increase in RBI since 1958 (r=0.25; p=0.047) and the average RBI increased over the past two decades from 0.202 in 1998-2008 to 0.221 in 2008-2017. Recent increases in RBI have led to a significant long-term increase, which was unchanging previously.

Cattaraugus River (US)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Unchanging
Rationale: Although the long-term trend is not significant (p=0.20), the average RBI increased from 0.354 in 1998-2007 to 0.406 in 2008-2018.

Portage River (US)
10-Year Trend: Increasing
Long-term Trend (1958-2017): Unchanging
Rationale: There was no significant long-term trend in RBI (p=0.07), yet the average RBI has increased consistently from 1998-2007 (0.520) to the recent decade (0.528). However, recent years have been trending downward, so this trend appears to be shifting.

River-by-River Assessment – Lake Ontario
Humber River (CAN)
10-Year Trend: Unchanging
Long-term Trend (1963-2017; missing 2004-2007): Unchanging
Rationale: The RBI exhibits no significant trend (p=0.83) and the average RBI from 1998-2007 (0.233) is similar to 2008-2017 (0.237).

Don River (CAN)
10-Year Trend: Increasing
Long-term Trend (1963-2017): Increasing
Rationale: The long-term trend is upward (r=0.35; p=0.009), though much of that increase was from 1955 to 1975, there has also been a slight increase in the average RBI from 0.509 in 1998-2007 and 0.539 in 2008-2017.

Seneca River (US)
10-Year Trend: Increasing
Long-term Trend (1997-2017): Increasing
Rationale: The long-term trend in RBI since 1997 is significantly upward (r=0.53, p=0.014) and the average RBI has increased from 0.077 in 1998-2007 to 0.097 in 2008-2017.

Status Assessment Definitions

For the purposes of this sub-indicator report, only an analysis of trends in tributary flashiness will be carried out. There is no universal scale for the R-B Index (RBI) and absolute values of the RBI are therefore not indicative of either “Good” or “Poor” conditions, especially if comparisons are made between watersheds of different types or sizes. However, when considering watersheds with similar flow regimes (i.e. event, variable, stable, or super stable) it may be possible characterize a statistically appropriate range of RBI values by watershed type or ecoregion (Baker et al., 2004). Additional analyses are needed to develop appropriate criteria and measures to assess differences between absolute values of the RBI within similar watershed types.

Trend Assessment Definitions

Trend**s** will be reported on a watershed-by-watershed basis across the Great Lakes basin every three years. In almost all cases, reductions in the RBI (becomes less flashy) would be considered desirable. Conversely, increases in the RBI (becomes more flashy) would be considered undesirable.

Increasing: The trend in flashiness over time is increasing (positive relationship)

Decreasing: The trend in flashiness over time is decreasing (negative relationship)
Unchanging: No trend in flashiness (no significant relationship with time).
Undetermined: Metrics do not indicate a clear overall trend for tributary flashiness, or data are not available to report on a trend.

Endpoints and/or Targets
Desirable outcomes are lack of trend in flashiness, or in most cases of altered ecosystems, reductions in flashiness. Urbanizing watersheds typically show increases in flashiness over time that parallel increases in imperviousness in the watershed.

To assess endpoints, two approaches were used. For long-term trends, we used a Spearman's rank correlation with statistical significance determined at the α=0.05 level. This non-parametric test does not assume normality or equal variance in the data, nor does it assume a linear trend. It does however, test for monotonic trends. For short-term trends, the 10 year running average in the data was examined over the past two decades (see Figures 1-5) and compared the average RBI from 1998-2007 with 2008-2017, where possible.

Sub-Indicator Purpose

- The purpose of this sub-indicator is to quantify the nebulous concept of flashiness, which is an important aspect of the hydrologic regime as it reflects the frequency and rapidity of short term changes in river flow to which aquatic ecosystems are adapted.

Ecosystem Objective
The ecosystem objective is to avoid hydrologic alteration. Periodic changes in flow rate are characteristic of streams and rivers, and the organisms that live in them have adapted to those changes. Spring floods may be important in opening up spawning areas or nurseries. Higher energies associated with storm runoff flush finer sediment from gravel beds, improving them as habitats for invertebrates and as spawning sites for salmonids. However, changes in the hydrologic regime, either by reduced flashiness such as occurs when a dam is constructed, or by increased flashiness such as occurs with urbanization, require adaptation by the resident organisms; if the changes are great enough, they can lead to the displacement of the native community and its replacement by another, often less desirable community.

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 the flashiness of hydrological response of a stream or river to precipitation (rainfall) and runoff (snowmelt) events. The Richards-Baker Flashiness Index (R-B Index/RBI for short) is calculated from mean daily flows from the U.S. Geological Survey or Environment Canada, usually on an annual basis, by dividing the sum of the absolute values of day-to-day changes in mean daily flow by the total discharge over that time interval (Baker et al. 2004).

$$\frac{\sum_{n=1}^{365} |q_n - q_{n-1}|}{\sum_{n=1}^{365} q_n}$$

Candidate streams for this sub-indicator are listed in Table 1. Most of these streams have a long flow records and are part of a proposed national monitoring network (http://acwi.gov/monitoring/network/design/). They cover a range of flashiness and land use as well as have few to no dams within the watershed. Rivers with dams present were reviewed to ensure that sampling stations are a good distance away (3+ miles) and the hydrograph was examined for any obvious influence.

Ecological Condition
Tributary flashiness is a measure that reflects the frequency of short-term changes in streamflow; the flow of a flashy stream increases and decreases dramatically in hours or a few days in response to rainfall.

Status
Overall, the long-term and 10-year (short-term) trends in RBI varied by river. Over the short-term (past decade), 9 of the 27 rivers had increasing RBI, 3 had decreasing trends, and 15 had unchanging trends (Table 2, Figures 1-5). Together, this suggests that 18 of the rivers, or 67%, were at or approaching desirable outcomes for flashiness. Compared to the previous Great Lakes Indicator report, over the short-term 26 rivers have the exact same or similar status, 1 river had trends that have improved (changed from increasing to unchanging or decreasing), and no rivers have trends that have worsened (changed from unchanging or decreasing to increasing). This data implies that flashiness has been relatively stable since the last report.

However, it is also important to compare the short-term and long-term trends to see if there have been declines in the RBIs (Table 2). Most of the rivers exhibited similar trends in the long- (entire period of record) and short- (since 2007) term– 19 of the 27 rivers either didn't change or went from decreasing to unchanging. Five rivers (Fox, Au Sable, Wanapetei, Maumee, Sandusky) improved in the recent decade over the long-term trends (i.e., changed from unchanging to decreasing, or increasing to decreasing). Yet, 9 rivers had increasing trends with 3 of the 9 in particular, showing a deteriorating trend, i.e. RBI changed from decreasing or unchanging to increasing. These included the Muskegon (Michigan), Portage (Erie), and Cattaraugus (Erie) Rivers. In the previous report, we identified the Thames (Erie) as a deteriorating trend (RBI in the long-term was unchanging but increasing in the short-term). Unfortunately, with the additional 3 years, both the long and short-term trends in the Thames are now increasing. It is important to note that 5 other rivers have increasing RBI over both the long and short-term time periods, which include the Manistee (Michigan), Pere Marquette (Michigan), White (Michigan), Seneca (Ontario), and Don (Ontario). In summary, most rivers (13) are showing stable long-term trends, 5 rivers have improved, only 3 have worsened, but 6 have continued worsening trends.

Collated by lake, Lake Superior and Lake Huron were the only lakes with no rivers with increasing flashiness. Lakes Michigan, Erie, and Ontario, in contrast, had multiple rivers with increasing flashiness. In both Lakes Erie and Michigan, a substantial number of rivers had increasing trends (3 and 4, respectively), yet rivers with decreasing or unchanging flashiness made up a greater proportion of the watershed (>27%). Yet, any of the lakes with a substantial number of smaller rivers exhibiting increasing trends (i.e., Michigan and Erie) should be closely monitored in case these smaller rivers are showing changes that would take longer to detect in larger watersheds. In Lake Ontario two of the three rivers assessed exhibited increasing flashiness, which may suggest this lake should be monitored more closely for affects such as increased erosion and fine sediment export, decreased habitat, and displacement of native biological communities.

Some of the rivers assessed (i.e., the Maumee and Sandusky Rivers) are known to have increasing discharge. It is important to note that the RBI may not reflect the influence of increasing discharge if storm events have changed in character to become more dispersed and cover more days. The influence of higher discharges may be quite similar to higher flashiness, hence some rivers may have negative impacts, such as high fine sediment export and erosion, even with declining flashiness trends.

Most rivers had the same trends as in the last report. The Fox River (Michigan) improved since the last report and previously had unchanging trends in the short-term that have now switched to decreasing. The Portage River (Erie) also improved, with increasing long-term trends in the past report that are unchanging. Two rivers (Au Sable- Huron, Thames- Erie) exhibited increasing long-term trends in this report, but were previously unchanging.

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

- Precipitation Amounts in the Great Lakes Basin – an increase in the amount or intensity of precipitation can impact a stream or river's flashiness by saturating or overwhelming the soil's capacity to retain water resulting in water entering the river much faster than under normal precipitation conditions. Changes in the intensity and frequency of precipitation could have large impacts on this sub-indicator, most likely leading to increases in the RBI. The exact response will depend on the magnitude of the changes in precipitation, the potential for concurrent drought, and the interactions with practices on the land. Climate change is expected to further exacerbate these effects by increasing the frequency and magnitude of heavy rain events, especially in the spring when soil conditions tend to be saturated.

- Land Cover – a variety of land use and land management changes may lead to increased or decreased flashiness, which can be detrimental to aquatic life and water quality. For example, increased impervious surfaces from urbanization tend increase flashiness.
- Water Quality in Tributaries – flashier watersheds can lead to elevated nutrient and sediment loadings that tributaries deliver to lakes, which can ultimately degrade water quality of the lake. Poor tributary water quality reflects the increase runoff potential, lower ability to attenuate nutrients and sediments during high flows, and higher erosive potential exacerbating streambank erosion.
- Base flow due to Groundwater – increased flashiness may reduce groundwater recharge at local and regional scales, which could impact base flow in rivers in streams; water supply for certain wetlands; surface water temperatures which can impact habitat and fish spawning areas; and habitat and conditions for benthic aquatic life.
- Coastal Wetlands – Increased flashiness can negatively impact downstream ecosystems such as wetlands or riparian buffers, as coastal wetlands could get flooded and water residence time decrease due to increased amounts of water, reducing their abilities to filter contaminants from water and provide habitat for wetland species. Furthermore, increased stream flashiness and peakflow could enhance scouring of the stream bottom and subsequent reductions in reproductive success of aquatic species if nesting areas are destroyed during high flow events (Johnson, background paper)

This sub- indicator also links directly to the other sub-indicators in the Watershed Impacts category, particularly Land Cover.

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 Limitations

- For small streams or streams with steep gradients, the hydrologic response may be too rapid to be resolved by daily flow data.
    - Small streams tend to be flashier than large rivers, and this is reflected in generally higher R-B Index values for small streams.
    - Streams with steep gradients and/or impervious watersheds will have high index values, even if they are not impacted by human activities (e.g. rock basin mountain streams).
    - For such systems, a version of the RBI based on hourly flow data can be used. However, Index comparisons between daily and hourly flow data are generally not meaningful. It is therefore critical that consideration be given to the type of flow data (daily or hourly) be used to calculate RBI values when making year-to-year comparisons or watershed-to-watershed comparisons.
    - Since the best use of the RBI Index is to track the hydrologic response of a stream through time, the index based on daily data is still useful for small streams, even if it under-represents the true flashiness.

Additional Information

The RBI is easy to calculate from widely available data. Possible range of values is from 0 to 2. Typical values are from 0.05 (very stable) to about 1.2 (very flashy). The Index integrates all flow data rather than specific percentile data. The Index is relatively stable from year-to-year (i.e. insensitive to weather effects) and consequently is a reliable indicator of longer-term trends (Baker et al., 2004).

The RBI is believed to be the only flashiness index or index of hydrologic alteration which incorporates the temporal sequence of flows, a very important part of the concept of flashiness. Overall, the RBI is positively correlated with increasing frequency and magnitude of storm events, and negatively correlated with base flow and watershed area.

For each gauged river, a graph can be prepared showing the RBI value on the y-axis and years on the x-axis. Changes in the RBI can also be displayed graphically to illustrate trend or mapped geospatially to show geographic changes in the distribution of flashiness. Streams with long flow records (20 years plus) flowing across a diverse range of landscapes and watersheds would be desirable. The State of Michigan is calculating RBI values for all gauged streams in the state.

This index offers an integrated perspective on changing hydrology in selected, and hopefully representative, major Great Lakes tributaries. It can be used to track the effects of, and guide decisions about, land use changes as they affect hydrology and its impact on riverine ecosystems. It utilizes basic flow data from the U.S. Geological Survey and Environment and Climate Change Canada, however, a number of rivers have patchy or incomplete data sets.

For small streams, the hydrologic response is too rapid to be adequately resolved by daily flow data. For such systems, a version of the RBI based on hourly flow data can be used. However, index values derived from hourly data cannot be directly compared with those derived from daily data. Since the best use of the RBI is to track the hydrologic response of a stream through time, the index based on daily data is still useful for small streams, even if it under-represents the true flashiness. Most of the watersheds selected for this sub-indicator are large, and flows change relatively slowly, so daily data are adequate for calculating the RBI. Most of these streams cover a range of flashiness and land use, have long flow records, have a large watershed area, and stations are of a great distance (over 3 miles) from a dam and sufficiently upstream of the Great Lake it feeds. More information about the RBI, and some applications in the Midwestern United States, can be found in the paper cited below.

Given the observed increases in discharge in some of the rivers, this metric may be improved by further examining potential increases in long-term and short-term trends in discharge along with RBI across all rivers. This would further flush out potential reasons for trends in RBI as well as serve as a linkage to the effects of other sub-indicators such as precipitation and the watershed stressor index.

Most of the candidate watersheds currently selected for this sub-indicator are large and likely to change flashiness slowly, so if the intent of this sub-indicator is to be some form of an “early warning system”, it might be preferable to choose smaller watersheds. However, smaller watersheds cover less of the Great Lakes basin and may not be a good representation of wide-spread changes. Therefore, using the daily flow data for the RBI on large watersheds into the Great Lakes appears to best match the objectives of this sub-indicator.
The Base Flow Index and the Tributary Flashiness Index may provide the opposite type of information that, in combination, may provide a more complete assessment of the streamflow conditions to support General Objective 9. In theory, these two sub-indicators inversely correlate, i.e. if tributary flashiness is high than base flow is low.

Acknowledgments
Authors: Laura Johnson, National Center for Water Quality Research, Heidelberg University, Tiffin, OH, USA.

Information Sources
Literature Citation:
Baker, D.B., R.P. Richards, T.T. Loftus, and J.K. Kramer. 2004. A New Flashiness Index: Characteristics and Applications to Midwestern Rivers and Streams. Journal of the American Water Resources Association 40(2): 503-522.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/ See Table 1 for station identification codes.

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Grannemann G, Van Stempvoort D. (Eds.) 2016. Groundwater science relevant to the Great Lakes Water Quality Agreement: A status report. Prepared by the Annex 8 Subcommittee for the Great Lakes Executive Committee, Final version, May, 2016. Published (online) by Environment and Climate Change Canada and U.S. Environmental Protection Agency.

Johnson, Lucinda, Katherine Hayhoe, George Kling, John Magnuson, Brian Shuter. *Confronting Climate Change in the Great Lakes Region – Wetland Ecosystems*, Technical Appendix, found at https://www.ucsusa.org/sites/default/files/legacy/assets/documents/global_warming/greatlakes_final.pdf https://www.ucsusa.org/resources/great-lakes-report-technical-appendix-climate-change-projections-impacts-aquatic. Last accessed July 25, 2019.

List of Tables

Table 1. Rivers used for the Tributary Flashiness sub-indicator. When a stream includes several HUC8s but does not comprise a complete HUC6, the HUC8 is listed that includes the gaging station from which the flow data are derived.
Table 2. Short-term (10 year) and long-term (period of record) trends in the RBI. Trends are denoted as unchanging in green (-), increasing in red (↑), or decreasing in blue (↓).

List of Figures

Figure 1. The RBI for tributaries to Lake Superior. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/
Figure 2. The RBI for tributaries to Lake Michigan. Note differences in y-axis scales. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt
Figure 3. The RBI for tributaries to Lake Huron. Note differences in y-axis scales. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/
Figure 4. The RBI for tributaries to Lake Erie. Note differences in y-axis scales. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/
Figure 5. The RBI for tributaries to Lake Ontario. Note differences in y-axis scales. Solid lines indicate the 10 year running average. Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/

Last Updated

State of the Great Lakes 2019 Technical Report

Lake	River	Country	Latitude	Longitude	USGS or EC ID	HUC	Drainage Area (mi^2)	Drainage Area (km^2)	Any dams present?
Superior									
	Pic	Canada	48°46'26"	86°17'47"	02BB003		1649	4271	No
	Pigeon	Canada	48°00'44"	89°36'58"	4010500 02AA001	04010101	235	609	Yes
Michigan									
	Fox	US	44°31'43"	88°00'36"	40851385	04030204	6330	16395	No
	Muskegon	US	43°26'05"	85°39'55"	4122000/ 4121970	04060102	2313	5991	Yes
	Grand	US	42°57'52"	85°40'35"	4119000	04050006	4900	12691	Yes
	Escanaba	US	45°54'31"	87°12'49"	4059000	04030110	870	2253	Yes
	Manistee	US	44°26'11"	85°41'55"	4124000	04060103	857	2220	Yes
	Pere Marquette	US	43°56'42"	86°16'43"	4122500	04060101	681	1764	No
	White	US	43°27'51"	86°13'57"	4122200	04060101	406	1052	No
Huron									
	Saginaw	US	43°25'19"	83°57'07"	4157005	04080206	6060	15695	No
	French	Canada	46°04'6"	80°36'41"	02DD010		5367	13900	Yes
	Au Sable	US	44°36'46"	83°50'16"	4136900	04070007	1513	3919	Yes
	Wanapitei	Canada	46°20'44"	80°50'22"	02DB005		1218	3154	Yes
	Magnetawan	Canada	45°46'23"	80°28'56"	02EA011		1096	2839	Yes
	Nottawasaga	Canada	44°29'6"	79°57'57"	02ED027		1037	2686	Yes
	Maitland	Canada	43°53'12"	81°19'35"	02FE002		635	1644	Yes
	Thunder Bay	US	45°07'27"	83°38'08"	4133501	04070006	586	1518	Yes
Erie									
	Maumee	US	41°30'00"	83°42'46"	4100009	04100009	6330	16395	Yes
	Thames	Canada	42°32'41"	81°58'2"	02GE003		1660	4299	Yes
	Sandusky	US	41°18'28"	83°09'32"	4198000	04100011	1251	3240	Yes
	Raisin	US	41°57'38"	83°31'52"	4176500	04100002	1042	2699	Yes
	Grand (OH)	US	41°43'08"	81°13'41"	4212100	04110004	685	1774	No
	Cattaraugus	US	42°27'48"	78°56'03"	4213500	04120102	436	1129	Yes
	Portage	US	41°26'58"	83°21'41"	4195500	04100010	428	1109	Yes
Ontario									
	Humber	Canada	43°48'40"	79°37'39"	02HC025		114	296	Yes
	Seneca	US	43°04'43"	76°38'44"	4235600	04140201	2815	7291	Yes
	Don	Canada	43°41'9"	79°21'41"	02HC024		123	319	Yes

Table 1. Rivers used for the Tributary Flashiness sub-indicator

*HUC stands for hydrologic unit code. The United States is divided and sub-divided into successively smaller hydrologic units which are classified into four levels (regions, sub-regions, accounting units, and cataloging units). The hydrologic units are arranged or nested within each other, from the largest geographic area to the smallest. Each hydrologic unit is identified by a unique HUC consisting of two to eight digits based on the four levels of classification in the hydrologic unit system. HUCs for units in the Great Lakes basin all begin with '04' to classify them as belonging to the Great Lakes region (USGS Water Boundary Dataset, 2014)

Lake	River	Country	Drainage Area (mi^2)	Drainage Area (km^2)	Any dams present?	10-year trend	Long term trend	
Superior								
	Pic	Canada	1649	4271	No	–	–	
	Pigeon	Canada	235	609	Yes	–	–	
Michigan								
	Fox	US	6330	16395	No	↓	–	
	Muskegon	US	2313	5991	Yes	↑	↓	*
	Grand	US	4900	12691	Yes	–	↓	
	Escanaba	US	870	2253	Yes	–	↓	
	Manistee	US	857	2220	Yes	↑	↑	*
	Pere Marquette	US	681	1764	No	↑	↑	*
	White	US	406	1052	No	↑	↑	*
Huron								
	Saginaw	US	6060	15695	No	–	–	
	French	Canada	5367	13900	Yes	–	–	
	Au Sable	US	1513	3919	Yes	–	↑	
	Wanapitei	Canada	1218	3154	Yes	↓	–	
	Magnetawan	Canada	1096	2839	Yes	–	–	
	Nottawasaga	Canada	1037	2686	Yes	↓	↓	
	Maitland	Canada	635	1644	Yes	–	–	
	Thunder Bay	US	586	1518	Yes	–	–	
Erie								
	Maumee	US	6330	16395	Yes	–	↑	
	Thames	Canada	1660	4299	Yes	↑	↑	*
	Sandusky	US	1251	3240	Yes	–	↑	
	Raisin	US	1042	2699	Yes	–	–	
	Grand (OH)	US	685	1774	No	–	–	
	Cattaraugus	US	436	1129	Yes	↑	–	*
	Portage	US	428	1109	Yes	↑	–	*
Ontario								
	Humber	Canada	114	296	Yes	–	–	
	Seneca	US	2815	7291	Yes	↑	↑	*
	Don	Canada	123	319	Yes	↑	↑	*

Table 2. Short-term (10-year) and long-term (period of record) trends in the RBI. Trends are denoted as unchanging in green (-), increasing in red (↑), or decreasing in blue (↓). Rivers with trends that have worsened in the recent 10 years relative to the long-term or have had continued increased trends are denoted with an asterisk (*).

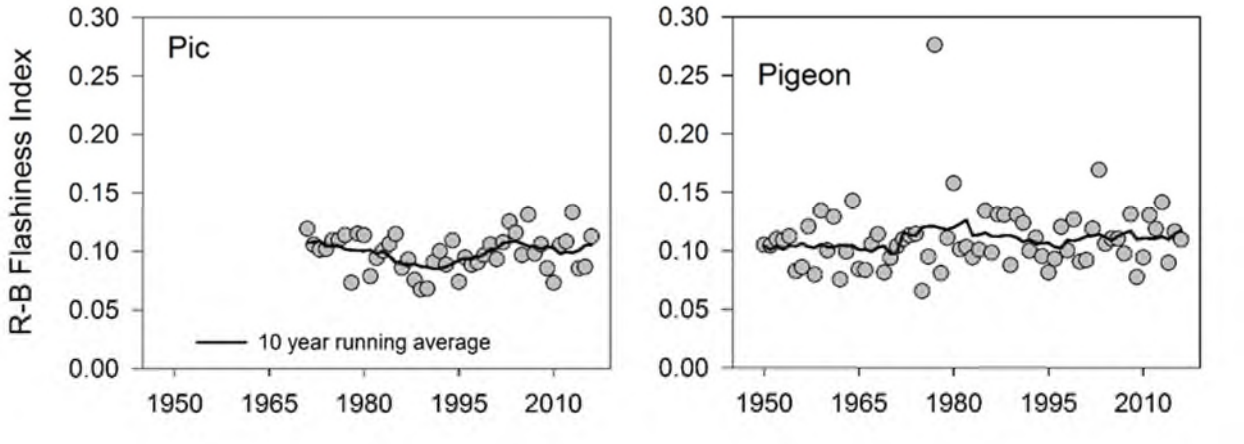


Figure 1. The RBI for tributaries to Lake Superior. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/

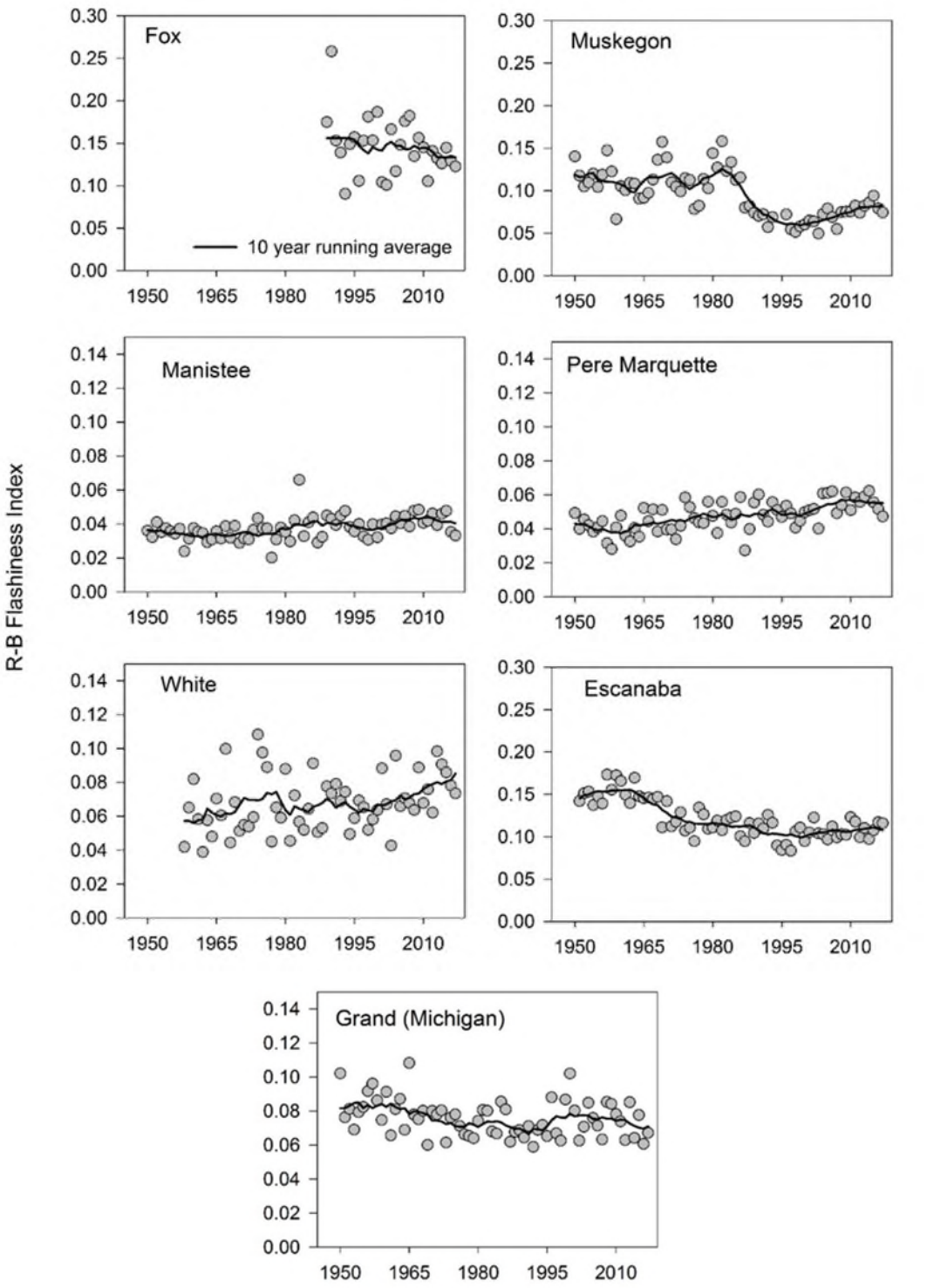


Figure 2. The RBI for tributaries to Lake Michigan. Note differences in y-axis scales. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt

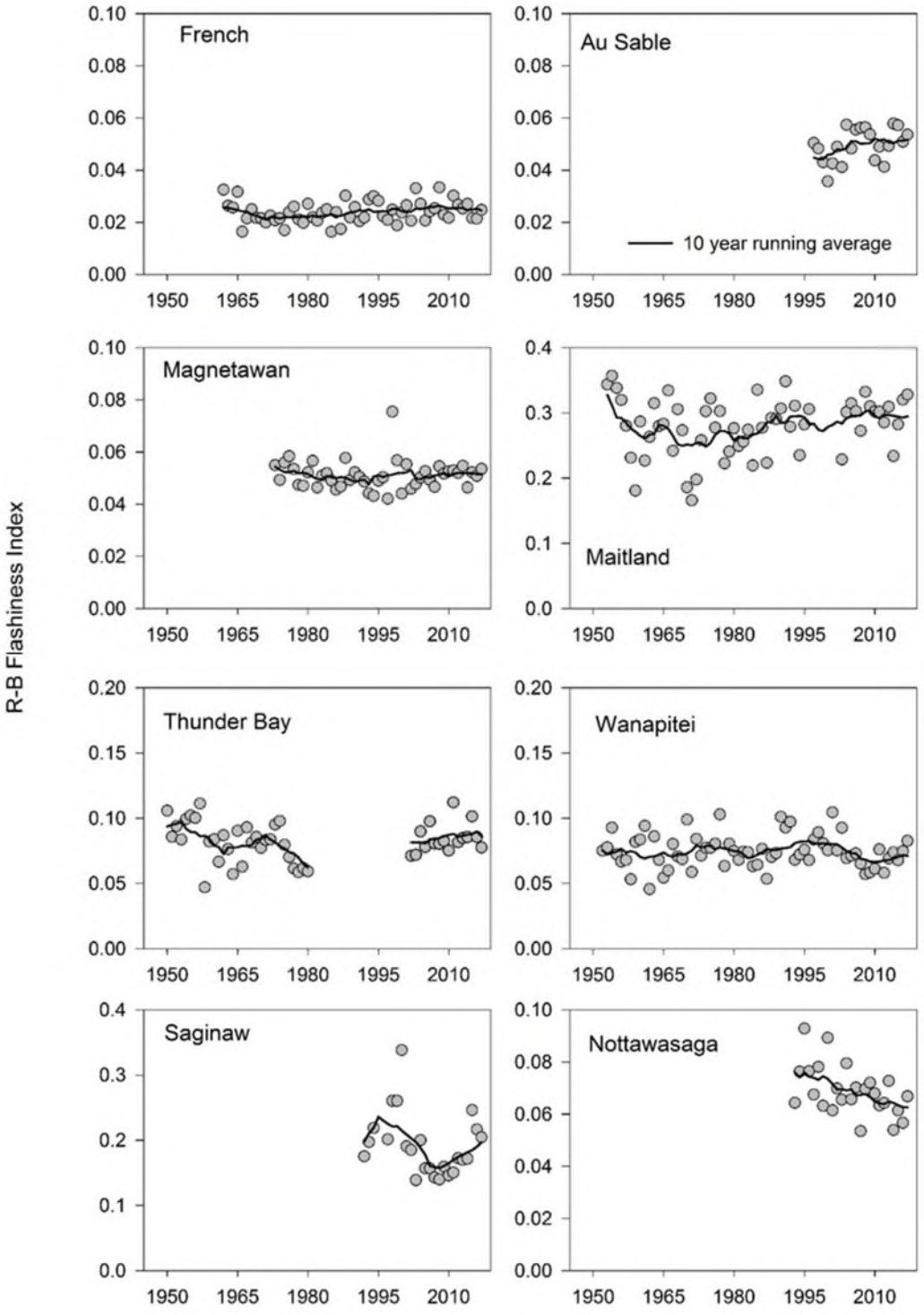


Figure 3. The RBI for tributaries to Lake Huron. Note differences in y-axis scales. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/

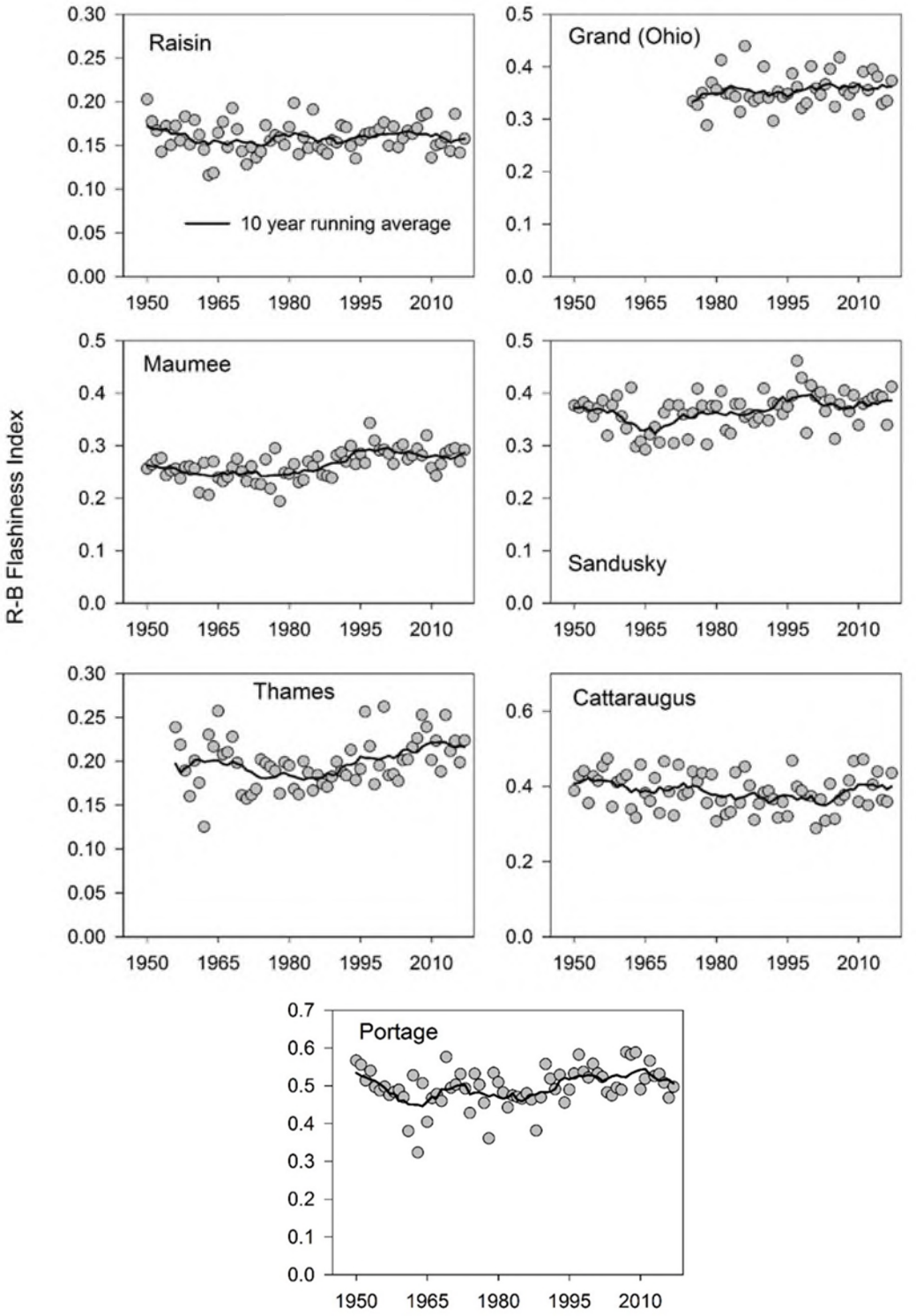


Figure 4. The RBI for tributaries to Lake Erie. Note differences in y-axis scales. Solid lines indicate the 10 year running average.
Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/

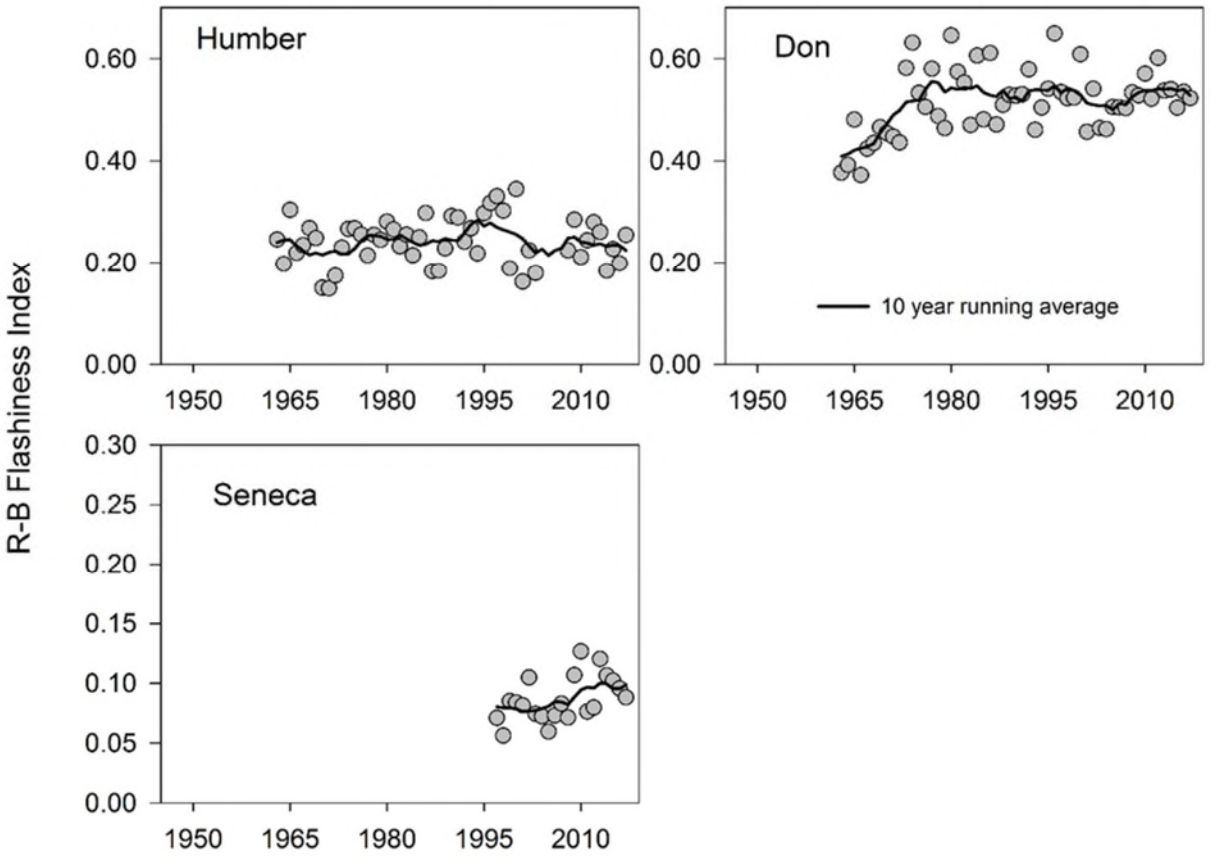


Figure 5. The RBI for tributaries to Lake Ontario. Note differences in y-axis scales. Solid lines indicate the 10 year running average. Data sources: http://waterdata.usgs.gov/nwis/rt and http://wateroffice.ec.gc.ca/

Sub-Indicator: Human Population

Overall Assessment
Trends:
10-Year Trend (2000-2010/2001-2011): Increasing
Long-term Trend (1970-2011): Increasing
Rationale: The long-term trend of the total population in the Great Lakes is increasing. The region has experienced 19.3% growth in population from 1971 to 2011. The 10-year trend reflects an increase of 3.4% for the Great Lakes basin. However, growth has slowed in the more recent shorter-term (2006 to 2011) to a 1.5% increase. No new data has been added to the 2017 State of the Great Lakes report so the long-term trends remain the same. The 2019 sub-indicator report offers a 10-year trend in population for the overall and lake-by-lake assessments.

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

Lake-by-Lake Assessment

Lake Superior
10-Year Trend (2000-2010/2001-2011): Unchanging
Long-term Trend (1970-2011): Decreasing
Rationale: Over the long term, the Lake Superior basin has experienced a 3.7% decrease in population. The 10-year trend indicates a 0.8% decrease in population which is noted as “Unchanging” for this period of time.

Lake Michigan
10-Year Trend: (2000-2010): Increasing
Long-term Trend (1970-2010): Increasing
Rationale: Population in the Lake Michigan basin has increased by 15.9% over the long term. The 10-year trend indicates a 1.7% increase in population in the years 2000-2010.

Lake Huron (including St. Marys River)
10-Year Trend: (2000-2010/2001-2011): Increasing
Long-term Trend (1970-2011): Increasing
Rationale: From 1971 to 2011, the population has increased by 34.1% in the Lake Huron basin with the Canadian side of the basin having more than a 60% increase in population over the same time period. The 10-year trend indicates a 3.7% increase in population.

Lake Erie (including St. Clair-Detroit River Ecosystem)
10-Year Trend: (2000-2010/2001-2011): Unchanging
Long-term Trend (1970-2011): Increasing
Rationale: The long-term trend indicates a 2.7% increase in population from 1971 to 2011. Over the 10-year period, there has been a 0.6% decrease in population. From 2000-2010, the U.S. Lake Erie basin experienced a 2.4% decrease in population; whereas from 2001-2011, the Canadian Lake Erie basin had a significant 8.5% increase.

Lake Ontario (including Niagara River)
10-Year Trend: (2000-2010/2001-2011): Increasing
Long-term Trend (1970-2011): Increasing
Rationale: From 1971 to 2011, the population has increased by 51.7% in the Lake Ontario basin. Over the 10-year period, the population increased by 11.5%. Population is only included for the Great Lakes basin watershed, so population from the St. Lawrence River watershed is not included in this assessment.

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 assess these conditions as “Good” or “Poor”.

Increasing: Net increase of the human population in the Great Lakes basin >1%
Unchanging: No 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

Endpoints and/or Targets
Human population growth in the Great Lakes basin is occurring sustainably, with emphasis on reducing urban sprawl and restoring and maintaining the chemical, physical, and biological integrity of the Waters of the Great Lakes.

Sub-Indicator Purpose
To assess the current human population trends in the Great Lakes region.

Ecosystem Objective
Humans are a key driving force in the overall impact on the environment in the Great Lakes basin, and emphasis should be placed on ensuring humans are working, playing, and living sustainably.

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." However, human population has an impact on achieving all General Objectives of the Great Lakes Water Quality Agreement.

Measure
This sub-indicator will measure total human population using Canadian census data, and U.S. census and intercensal data by watershed (Table 3). The Canadian census data and U.S. census and intercensal data each draw their data from different time periods, so the data do not align. However, the intent of this sub-indicator is to identify population fluctuations across the years, and not to compare Canadian with U.S. population growth. Intercensal data was used to provide a reasonable estimate of population and a greater amount of insight into the states and lake basins that are experiencing fluctuations in population. Population data by watershed is available through Statistics Canada and will be used in this measure.

There are numerous approaches to determining population distribution in the Great Lakes basin. In the United States, population figures are not available by watershed as they are in Canada. The U.S. County Adjusted Ratio Approach was developed, which uses two levels of adjustment to U.S. county census data to determine the population in the Great Lakes basin. A geographic information system was used to calculate the proportion of a county within the Great Lakes basin. The first level of adjustment multiplies the county population by the proportion of the county within the Great Lakes basin. This step assumes that population distribution across counties is uniform, which is not necessarily the case.

The second level adjustment, involves a review of the population identified for each U.S. county. Every county with a population close to 100,000 people (or 40,000 people for the Lake Superior basin) was examined to ensure the population calculated in the ratio approach accurately reflected the distribution of the county's population. 40,000 and 100,000 are not hard limits because inconsistencies will occur when populations are adjusted one year, but not the next. For example, if a county had a population of 38,000 in 1971 and was 30% contained in the basin, 30% of the 38,000 would be included in the population count. However, if the same county's population was 45,000 in 1976, it should be reviewed according to the adjusted ratio method to make sure that the population count is truly reflective of the spatial distribution of the country's population. The unadjusted 1971 population would be 11,400 (0.3*38,000) and if the ratio for 1976 was adjusted to something like 50% (to be more reflective of the spatial distribution of the county's population), that would be 22,500 (0.5*45,000). That would nearly be a doubling in population. It is reasonable to adjust the ratio for the 1971 population to better reflect the true population, though the county has a population less than the 40,000 threshold. For counties that are close to the 40,000 or 100,000 population mark, large population centres that lie outside the basin should be visually inspected regardless of the proportion of the county that lies within the basin. For example, a county that lies just 2% in the basin, but has a population of 105,000 would be examined to ensure that there are no large population centres nearby. If the ratio for a county's population is being adjusted, ensure that the county resides in only one basin. The county's total area within all 5

basins must still be equal or less than 100% or population will be over-estimated. If a county of 105,000 is located 50% within the Erie basin and 40% in the Ontario basin, the ratio for either basin cannot be adjusted more than 10% as it would exceed 100%. Adjustments were also necessary in the Chicago Metropolitan Area (Cook, DuPage, Lake and Will counties) to reflect the 6.4 million that draw drinking water from Lake Michigan. Only 4.2% of Cook County is located within the Lake Michigan basin, which would result in a first level adjusted population of approximately 220,000 people. However, the second level adjusted population of 4.9 million people in Cook County more accurately reflects the county's impact on the basin. In total, 21 counties of 653 had ratios adjusted to more accurately reflect their population calculations and those population numbers are included in the total population calculated for the Great Lakes basin (Table 2).

A few adjustments have been made to the methodology since the 2011 reporting cycle and these improvements were implemented in the 2017 and 2019 assessments. In previous reports, U.S. intercensal data was used as it would align with Canadian census years. In this report, U.S. census data has been used to improve accuracy of population figures. However, this results in an imperfect alignment of census data since U.S. census data is collected a year before Canadian census data. The lengths of the census cycles are also different; U.S. census data is collected every 10 years, while Canadian census data is collected every 5 years. This report follows the shorter Canadian census cycle; data from 1971 represents 1970 U.S. census data and 1971 Canadian census data (Table 3). Also, a larger number of counties were reviewed in preparation for this report to determine if a second level of adjustment was warranted to ensure greater accuracy of population values. Previously, only counties that reached a population threshold of 100, 000 people or greater in the Lake Erie, Huron, Michigan and Ontario basins, or 40,000 or greater in the Lake Superior basin in 1971 would be reviewed for a second level of adjustment. In this report, any county that reached a population of 100,000 people or greater from 1971 to 2011, or 40,000 or greater in the Lake Superior basin, was reviewed for a second level of adjustment.

The calculation of percentage change in population is necessary to determine human population trends in the Great Lakes basin.

To calculate the percentage change in population:

$$\text{Percentage change in population} = \frac{(\text{Population of later year} - \text{Population of earlier year}) \times 100}{\text{Population of earlier year}}$$

Example: 1971 Population in Lake Superior 603,507
1976 Population in Lake Superior 621,886

$$\text{Percentage change in population} = \frac{(621{,}886 - 603{,}507) \times 100}{603{,}507}$$
$$= 0.03045 \times 100$$
$$= 3.045\% \text{ increase in population in the Lake Superior basin from 1971 to 1976}$$

Other possible measurements:

- The ratio of total increase in population which is occurring in the Canadian Census Metropolitan Areas (CMAs) and U.S. Metropolitan Statistical Areas (MSAs) of the basin as compared to overall total increase.
- Qualitative assessment of the pressure points of human population impacts.
- Comparison of Canadian and U.S. census data (instead of Canadian census data and U.S. intercensal data). However, census years are not the same as the Canadian census cycle occurs every 5 years, while the U.S. census occurs every 10 years. However, data quality could be improved since U.S. population figures would no longer be estimates.

Ecological Condition
In this report, the Great Lakes basin is defined as the watershed of the Great Lakes.

Total Populations in the Great Lakes Region (Ontario and Eight Great Lakes States)
In 2011, the Great Lakes region was home to about 39,385,438 people (Table 4); 28.5% reside in Canada while 71.5% reside in the United States. The Lake Ontario basin continued to experience the largest growth. Population in

the Lake Michigan and Lake Huron basins stayed relatively static in the short- term (2006-2011) while population in Lake Superior and Lake Erie experienced a slight decline. Of the Great Lakes region, the Province of Ontario continued to exhibit the largest amounts of growth with a 6.0% increase in population from 2006 to 2011. From 1971 to 2011, the population has increased by 51.7% in the Lake Ontario basin. The population of the Province of Ontario has also grown the most over the long-term; the population increased 70.1% from 1971 to 2011. Furthermore, 87% of Ontario's population resides in the Great Lakes basin (Figure 1).

Ten-Year (10-year) Trend Analysis
Over the 10-year period, population trends have remained stable in Lake Superior in both U.S. and Canada (Figure 9). The percentage of population in the U.S. Lake Huron basin increased slightly by 1%, but increased significantly in Canada by 9.6%. Furthermore, Lake Ontario remained stable in the U.S. with only a very small decrease of 0.8%, but rose substantially in Canada by 16%. Population in the Lake Erie basin had a decrease in the U.S. by 2.4%, but the population increased in the Canadian Lake Erie basin by 8.5% (Figures 7 and 8).

This differing trend between the two nations may be due to people and immigrants settling within metropolitan areas, in particular in southern Ontario, because of increased job opportunities, academic institutions, cultural diversity and access to health care.

Lake Basins
The total population in the Great Lakes basin has increased since 1971 (Figure 2), though population growth rates have varied across the years (Figure 3).

The population in the Lake Superior basin is significantly smaller than in the other basins; 1.5% of the total population in the Great Lakes Region live in the Lake Superior basin. Furthermore, it is the only basin in the Great Lakes where both the short- and long-term trends indicate a decline in population (Figure 3).

The Lake Michigan basin is home to 33.8% of the total Great Lakes population, the most populated basin of all the Great Lakes. The population in this basin reflects population adjustments as well as the 6.4 million people in Cook, DuPage, Lake, and Will counties that draw drinking water from Lake Michigan. This basin is completely contained within the United States (Figures 1 and 4); more specifically, the states of Illinois, Indiana, Michigan and Wisconsin are located in the Lake Michigan basin.

Lake Huron's population has remained relatively static as population growth from 2006 was 0.2%. On average, the population in the Lake Huron basin has made up roughly 7.7% of the total population in the Great Lakes Basin (Figure 6). Over time, the proportion of the population living on the Canadian side of the basin has increased. A 40-year span yielded an 8.1% increase in the proportion of Canadians living in the Lake Huron basin.

In the Lake Erie basin, population has remained fairly stable and is the second most populated of the Great Lakes basins. The greatest population change occurred from 1991 to 1996 with a population increase of 3.0%. Approximately 31% of the Great Lakes population resides in the basin (Figure 6) – 25.3% of Lake Erie's population lived in the U.S. basin and 5.6% lived in the Canadian basin.

The Lake Ontario basin has consistently experienced the greatest amount of growth in the Great Lakes region. The population change in the Lake Ontario basin since 1971 has exhibited an average growth of 5.3% (per 5-year period). This lake basin has been home to the greatest proportion of Canadians living in the Great Lakes region from 1971-2011; 18.8% of the total population in the Great Lakes region are Canadians that live in the Lake Ontario basin (Figure3). The population of the province of Ontario has also grown the most over the long term. The population increased by 70% over the 40-year period (1971-2011).

Linkages
Humans are a key driving force in the overall impact on the environment. Emphasis should be placed on ensuring humans are working, playing and living sustainably. Further analysis in population trends, urban sprawl and consumption rates can help understand and calculate the different impacts humans can have on the environment. As the population continues to grow, cities could expand outwards, resulting in the potential loss of agricultural and natural lands. Associated impacts of urban growth include an increase in air pollution through gridlock, fragmentation of

habitats and greater strains on water systems for drinking water. Growth in human population and resource consumption are drivers for more development. Other things being constant, high conversion rates in land cover are associated with rapid rates of development. The spatial pattern of land use conversion also affects wildlife habitat and associated wildlife populations and communities (Land Cover 2019 draft sub-indicator).

As coastal communities continue to grow, there will be increasing stress on natural resources, leading to environmental degradation. For example, Ribic et al. (2010) identified several mechanisms through which human population can drive the proliferation of marine and coastal debris and population size is of particular significance. Increased marine and coastal debris is linked to population increases and proximity to the water and coasts.

This sub-indicator is essential in identifying areas within the Great Lakes basin that may be facing increased environmental pressures as a result of large amounts of population growth. In the Greater Toronto Area, located on the northwestern shores of Lake Ontario, urban areas have grown roughly 20% from 1985 to 1995 and roughly 15% from 1995 to 2005 (Furberg & Ban, 2012). Areas of high population can be identified on the northwestern shores of Lake Ontario, the southwestern tip of Lake Michigan and along the northern and western shores of Lake Erie/St. Clair-Detroit River Ecosystem (Figure 6). This is contrasted by the Lake Huron and Lake Superior basins that have very few large population centres.

Ontario's population is projected to grow by 30.2 per cent, or almost 4.3 million, over the next 24 years, from an estimated 14.2 million in July 2017 to almost 18.5 million by July 2041. The Greater Toronto Area (GTA) is projected to be the fastest growing region of the province, with its population increasing by 2.8 million, or 40.8 %, to reach almost 9.7 million by 2041. The GTA's share of provincial population is projected to rise from 48.3 % in 2017 to 52.3 % in 2041 (Ontario Population Projections). Population estimates are also available from Statistics Canada. Preliminary estimates for Ontario in 2017 reflect 14,193,384 people living in the province which is an increase of over 1.3 million people since 2011 when the population was 12,851,821, Statistics Canada 2018. Rapid population growth can result in a great amount of stress exerted on the environment and water quality. These concerns can include inadequate transportation systems resulting in more vehicles on the roads and emissions, conversion of natural lands to developed lands, demands for infrastructure including waste water treatment and drinking water treatment (Mikovits, Rauch & Kleidorfer, 2014; Addie, 2013).

Human activities are a source of excess nutrients in the Great Lakes basin. Humans contribute to nutrient pollution through activities related to agriculture, storm water runoff, wastewater, and the use of fossil fuels and household products (USEPA, 2017). Increases in human population around the basin could mean greater amounts of nutrient pollution in the Great Lakes. Nutrients like nitrogen and phosphorus can feed algae, and contribute to the growth of harmful algal blooms (NOAA, 2018). Harmful algal blooms can also prevent human activities in the water such as swimming, and can affect the quality of drinking water. HABs can be harmful to human health in other ways, causing rashes, respiratory illnesses, neurological problems, and illness in the stomach or liver (USEPA, 2017).

Increasing populations will also have the potential to increase contaminants in water (Dovers and Butler, 2015). Human beings use and dispose of products containing chemicals such as fluorinated surfactants, flame retardants, chlorinated paraffins, phthalates, and other compounds in personal care products (Ecology Center, 2015). These chemicals can have negative effects on water quality which can have direct or indirect impacts on fish, birds and other species. Toxic chemicals produced from other activities, including industrial processes, can also impact source and drinking water quality (Denchak, 2018). Large populations tend to consume a greater amount of land, food, fossil fuels, and mineral resources which can contaminate lake water through runoff, emissions and other routes of entry. Humans are a driver of various sources of pollution, which can be harmful to the environment (Green Facts, 2006) and water quality.

Climate trends in Precipitation Amounts, Water Levels, Ice Cover and Surface Water Temperature can have indirect impact on the human population. More extremes weather events can cause flooding, property damage, and impact water quality for recreational use and drinking.

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 Limitations

- This measure does not assess population density and therefore does not infer the impacts of the human population; but it is an easier measurement to provide accurately. Human population density can be used as a surrogate indicator of the extent of human pressures on the surrounding landscapes, whereas change in total population does not provide these insights.
- This sub-indicator would be best serving if it were able to assess urban population density in the Great Lakes basin, by looking at growth in the Canadian Census Metropolitan Areas and U.S. Metropolitan Statistical Areas in the Great Lakes basin. While density is a readily understandable measure, it proves challenging to quantify because of the difficulty in estimating true urban extent in a consistent and unbiased way. The political geographic extents of MSAs and CMAs give approximate indications of relative city size, but also tend to contain substantial areas of rural land use. There is insufficient data for all urban centres in the basin to determine a trend.
- Information for this sub-indicator of population density would need to be collected perhaps every 5 or 10 years as changes in density take place relatively slowly.
- This indicator is useful in comparing municipalities to each other, but would need to be aggregated into an index in order to be represented as a basin-wide measure.
- U.S. population figures derived from intercensal data and will not be as accurate as census data.

Additional Information

The Great Lakes basin cities of Toronto and Chicago are the 4th and 5th largest cities in North America, respectively. However, of the 100 fastest growing cities in the U.S. in 2018, none were in the Great Lakes basin (Census Bureau, 2018).

Sustainable development requires that cities expand and grow in a way so that they will be environmentally sensitive, and not compromise the environment for future generations. Increasing human population densities in urban areas while maintaining low densities in rural and fringe areas, reduces the amount of land consumed by urban sprawl.

Urban density is a relative measure of efficiency. In general, and other things being equal, higher density land use is less energy and resource consuming and thus is more efficient from an ecosystem perspective. For example, transportation in higher density areas may be less resource demanding since distances are shorter and public transportation is often more available and inexpensive. Consequently, air pollution should be lower in more densely populated areas. In addition, since inefficient land use for urban development implies loss of land use for natural and other purposes there are significant biodiversity dimensions to inefficient land use. In general, the less land used for development, the greater the opportunities that exist for natural biodiversity goals to be met. Urban densities have been declining over time as urban development has become much more sprawling with the vast majority of new development occurring on former agricultural or natural lands. This has resulted in greater reliance for urban residents on the automobile as virtually the only method of public transit for these widespread and low density new communities has become impractical.

Acknowledgments
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List of Tables

Table 1. Population Estimate Approaches (2011).
Source: Data taken from Statistics Canada and U.S. Census Bureau
Table 2. Counties requiring second level ratio adjustment to better reflect population numbers in basin.
Source: Data taken from Government of Ontario and U.S. Census Bureau
Table 3. Years of Intercensal and Census Data.
Source: Data taken from Statistics Canada and U.S. Census Bureau
Table 4. Total Population in each Great Lakes basin.
Source: Data taken from Statistics Canada and U.S. Census Bureau

List of Figures

Figure 1. Population in Ontario by Census Subdivision (2011).
Source: Government of Ontario, Statistics Canada
Figure 2. Total population in the Great Lakes Region from 1971-2011.
Source: Data taken from Statistics Canada and U.S. Census Bureau
Figure 3. Population change (%) in each Great Lakes Basin from 1971-2011.
Source: Data taken from Statistics Canada and U.S. Census Bureau
Figure 4. Proportion of Total Great Lakes Population by Canadian and U.S. Lake basin.
Source: Data taken from Statistics Canada and U.S. Census Bureau
Figure 5. Proportion of Canadian and U.S. Population located in each Great Lakes basin.
Source: Data taken from Statistics Canada and U.S. Census Bureau
Figure 6. Population by County or Census Subdivision in the Great Lakes basin (2010/2011).

Source: Data taken from Government of Ontario, Statistics Canada and U.S. Census Bureau
Figure 7. U.S. population located in each Great Lakes basin from 2000-2010.
Source: Data taken from U.S. Census Bureau
Figure 8. Canadian population located in each Great Lakes basin from 2001-2011.
Source: Data taken from Statistics Canada
Figure 9. Total population in each Great Lakes basin from 2000-2011.
Source: Data taken from Statistics Canada and U.S. Census Bureau

Last Updated
State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

Approach	**Population Estimates (2011)**			
	Ontario	**Quebec**	**Eight Great Lakes States**	**Total**
Great Lakes and St. Lawrence River Region – All of Ontario, Quebec and Eight Great Lakes States	12,851,821	7,903,001	83,805,970	104,560,792
Great Lakes Region – All of Ontario and All Eight Great Lakes States	12,851,821	-	83,805,970	96,657,791
Great Lakes Basin – Canadian population in Great Lakes Basin and population in U.S. Counties partially or wholly contained within basin	11,234,177	-	37,681,537	48,915,714
Great Lakes Basin – Canadian population in Great Lakes Basin and population in U.S. using County Adjusted Ratio Approach (without Chicago Metropolitan Area)	11,234,177	-	21,785,962	33,020,139
Great Lakes Basin – Canadian population in Great Lakes Basin and population in U.S. using County Adjusted Ratio Approach (with Chicago Metropolitan Area), used in this report	**11,234,177**	**-**	**28,151,261**	**39,385,438**

Table 1. Population Estimate Approaches (2011)**.**
Source: Data taken from Statistics Canada and U.S. Census Bureau.

County	Basin – Original Ratio	Basin – Adjusted Ratio	Rationale
Cook, IL	Lake Michigan – 4.24%	Lake Michigan – 95%	Population extracts drinking water from Lake Michigan
DuPage, IL	Lake Michigan – 0%	Lake Michigan – 50%	
Lake, IL	Lake Michigan - 11.51%	Lake Michigan – 90%	
Will, IL	Lake Michigan – 0%	Lake Michigan – 50%	
LaPorte, IN	Lake Michigan 24.09%	Lake Michigan – 35%	Accounting for Michigan City
St. Joseph, IN	Lake Michigan – 39.68%	Lake Michigan – 70%	Accounting for South Bend and Mishawaka
Jackson, MI	Lake Erie – 16.24% Lake Michigan – 83.76%	Lake Erie – 7% Lake Michigan – 93%	Larger population centres in Michigan basin than Erie
Marquette, MI	Lake Michigan – 55.97% Lake Superior – 44.03%	Lake Michigan – 45% Lake Superior – 55%	Larger population centres in Superior basin than Michigan
Oakland, MI	Lake Erie – 79.65% Lake Huron – 20.35%	Lake Erie – 90% Lake Huron – 10%	Larger population centres in Erie basin than Huron
Saginaw, MI	Lake Huron – 97.81% Lake Michigan – 2.19%	Lake Huron – 99% Lake Michigan – 1%	Larger population centres in Huron basin than Michigan
St. Louis, MN	Lake Superior – 49.27%	Lake Superior – 90%	Accounting for Duluth
Chemung, NY	Lake Ontario – 9.10%	Lake Ontario – 3%	Few population centres in basin
Jefferson, NY	Lake Ontario – 57.8%	Lake Ontario – 75%	Numerous population centres in basin
Onondaga, NY	Lake Ontario – 93.07%	Lake Ontario – 98%	Numerous population centres in basin
Steuben, NY	Lake Ontario – 12.75%	Lake Ontario – 5%	Few population centres in basin
Tompkins, NY	Lake Ontario – 81.70%	Lake Ontario – 92%	Numerous population centres in basin
Erie, PA	Lake Erie – 52.89%	Lake Erie – 80%	Accounting for Erie
Douglas, WI	Lake Superior – 57%	Lake Superior – 80%	Accounting for Superior
Kenosha, WI	Lake Michigan – 22.8%	Lake Michigan – 75%	Accounting for Kenosha
Racine, WI	Lake Michigan – 47.77%	Lake Michigan – 75%	Accounting for Racine
Waukesha, WI	Lake Michigan – 7.72%	Lake Michigan – 15%	Accounting for Menomonee Falls

Table 2. Counties requiring second level ratio adjustment to better reflect population numbers in basin.
Source: Data taken from the Government of Ontario and U.S. Census Bureau

Population Data (1970/1971 – 2010/2011)

Canada	1971	1976	1981	1986	1991	1996	2001	2006	2011
U.S.	1970	1976*	1980	1986*	1990	1996*	2000	2006*	2010

*Intercensal population estimates

Table 3. Years of intercensal and census data.
Source: Data taken from Statistics Canada and U.S. Census Bureau

Lake Basin	1971	1976	1981	1986	1991	1996	2001	2006	2011
Superior	603,507	621,886	618,958	585,591	586,389	584,622	585,630	581,722	581,093
Michigan	11,497,315	11,718,373	11,796,578	11,857,716	12,042,678	12,561,047	13,101,190	13,211,346	13,325,057
Huron	2,332,911	2,519,332	2,612,775	2,617,271	2,776,130	2,917,142	3,018,271	3,123,897	3,129,153
Erie	11,863,069	11,829,173	11,784,483	11,559,464	11,741,188	12,098,730	12,258,143	12,266,000	12,180,736
Ontario	6,703,266	7,036,916	7,241,352	7,570,122	8,213,685	8,653,646	9,121,919	9,621,761	10,169,399
Total	**33,000,068**	**33,725,680**	**34,054,145**	**34,190,163**	**35,360,070**	**36,815,187**	**38,085,152**	**38,804,726**	**39,385,438**

Table 4. Total population in each Great Lakes basin.
Source: Data taken from Statistics Canada and U.S. Census Bureau.

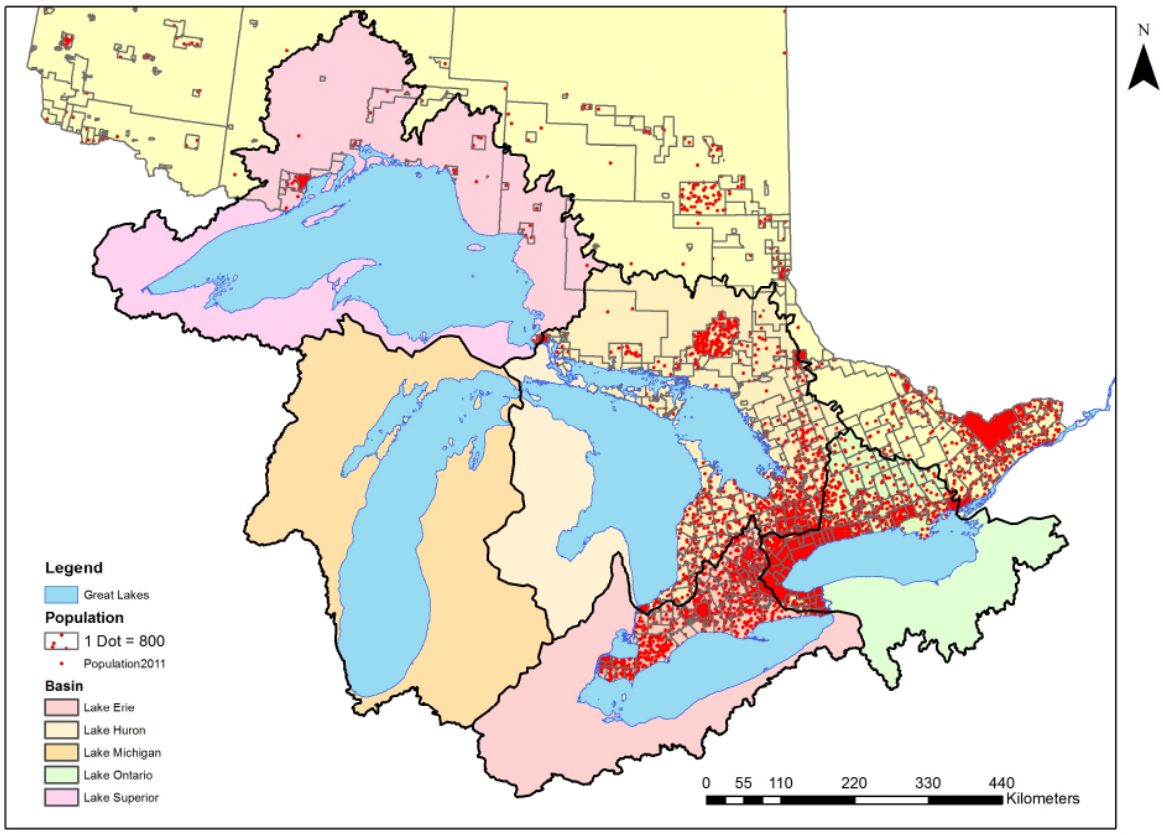


Figure 1. Population in Ontario by census subdivision (2011).
Source: Data taken from Government of Ontario, Statistics Canada

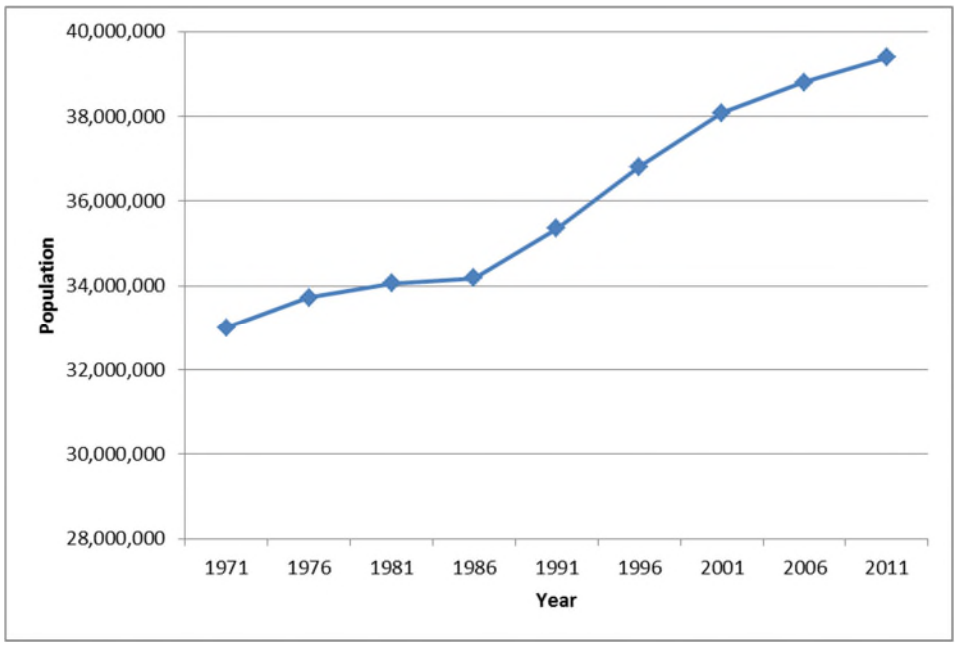


Figure 2. Total population in the Great Lakes region (1971-2011).
Source: Data taken from Statistics Canada and U.S. Census Bureau

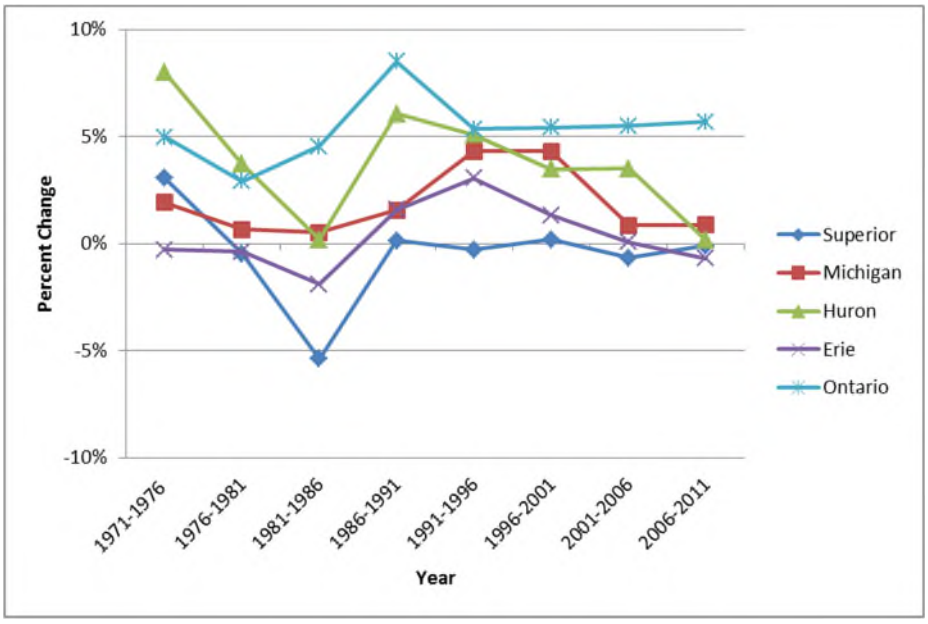


Figure 3. Population change (%) in each Great Lakes basin from 1971-2011.
Source: Data taken from Statistics Canada and U.S. Census Bureau.

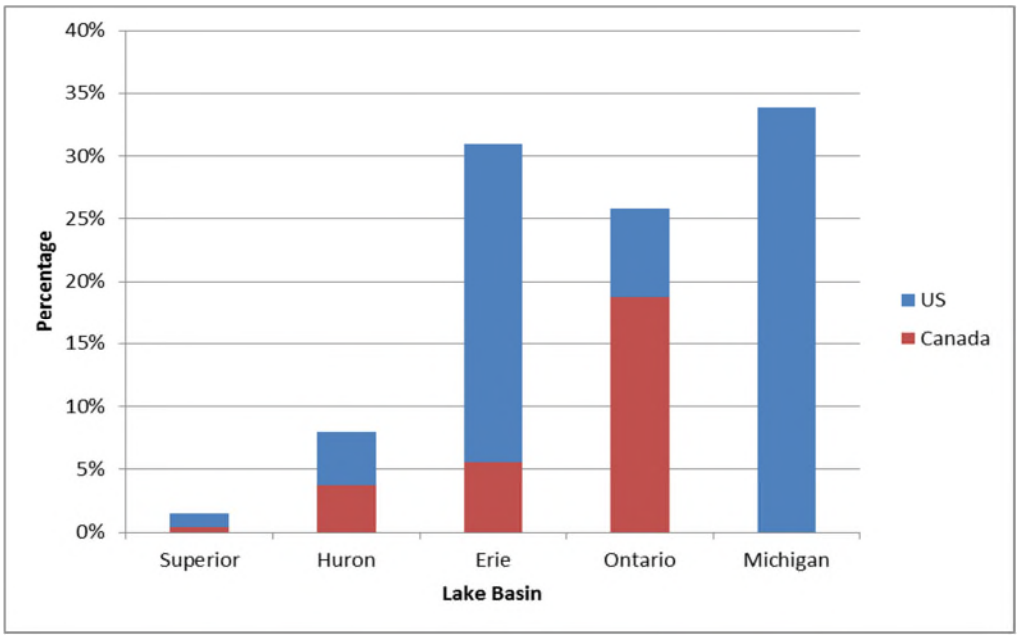


Figure 4. Proportion of total Great Lakes population by Canadian and U.S. Lake basin (2011).
Source: Data taken from Statistics Canada and U.S. Census Bureau.

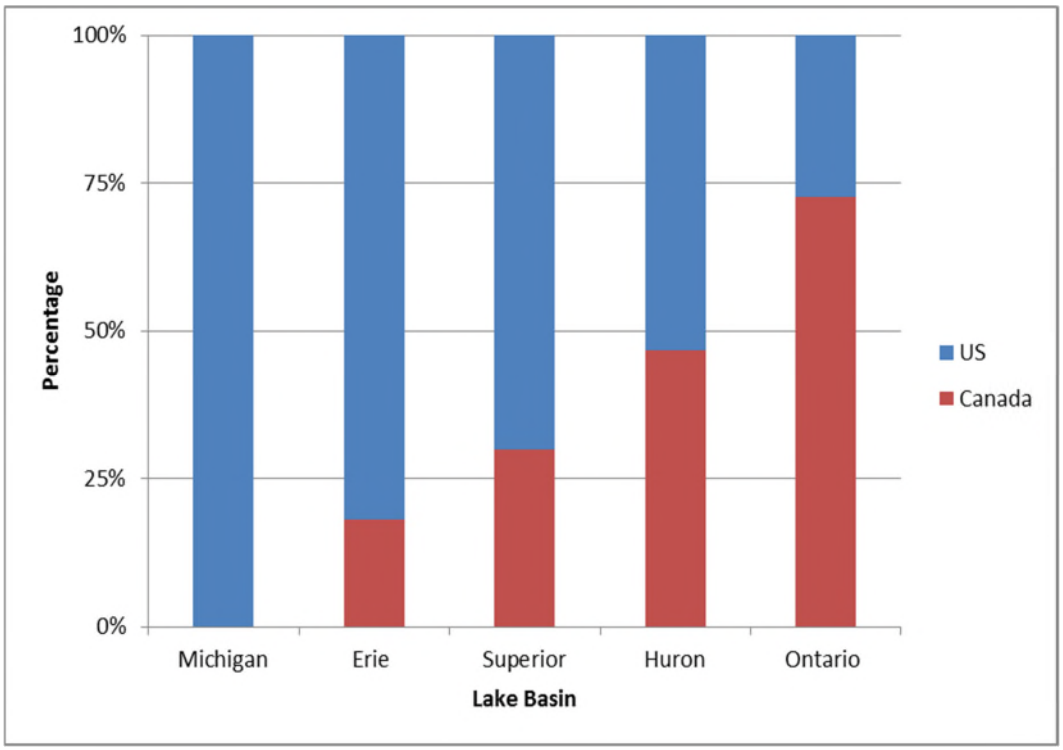


Figure 5. Proportion of Canadian and U.S. population located in each Great Lakes basin (2011).
Source: Data taken from Statistics Canada and U.S. Census Bureau.

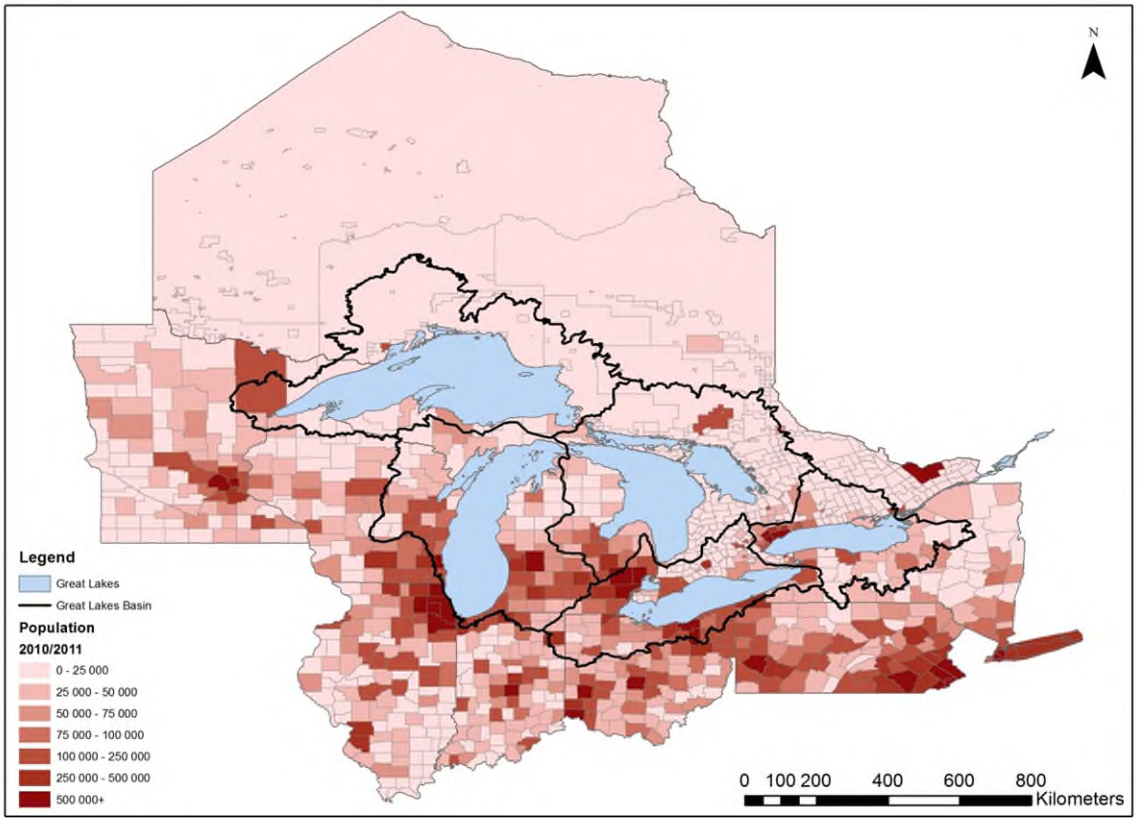


Figure 6. Population by county or census subdivision in the Great Lakes basin (2010/2011).
Source: Data taken from Government of Ontario, Statistics Canada and U.S. Census Bureau

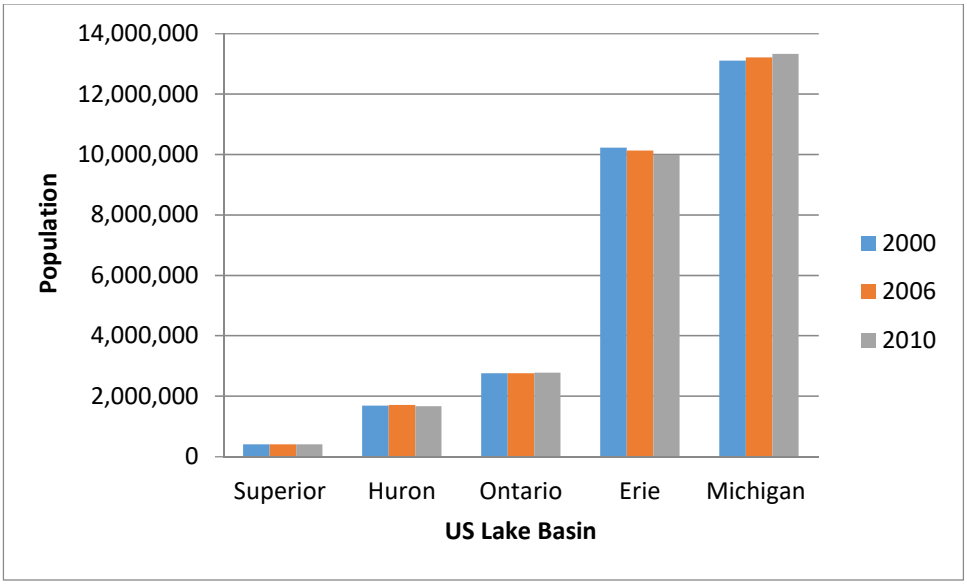


Figure 7. U.S. population located in each Great Lakes basin from 2000-2010.
Source: Data taken from U.S. Census Bureau.

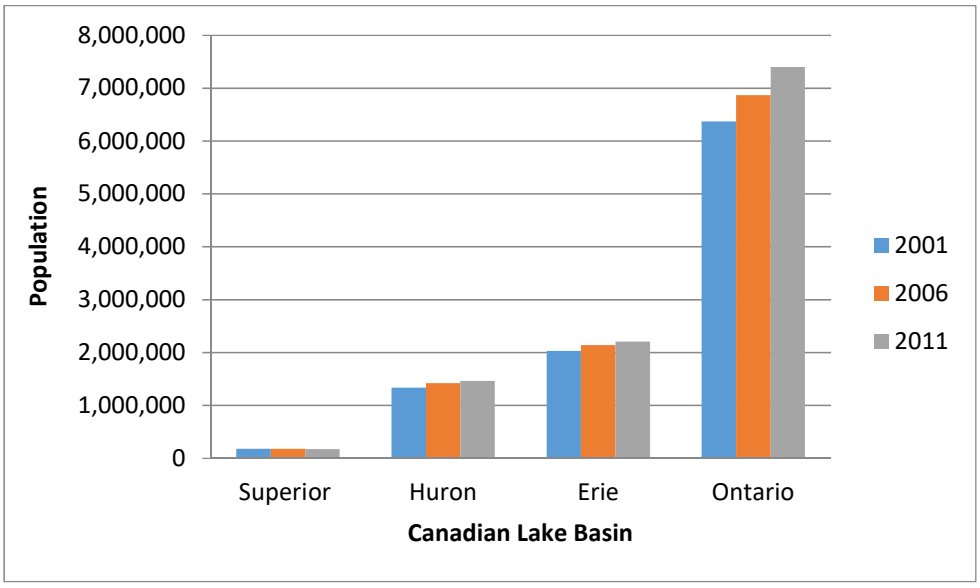


Figure 8. Canadian population located in each Great Lakes basin from 2001-2011.
Source: Data taken from Statistics Canada.

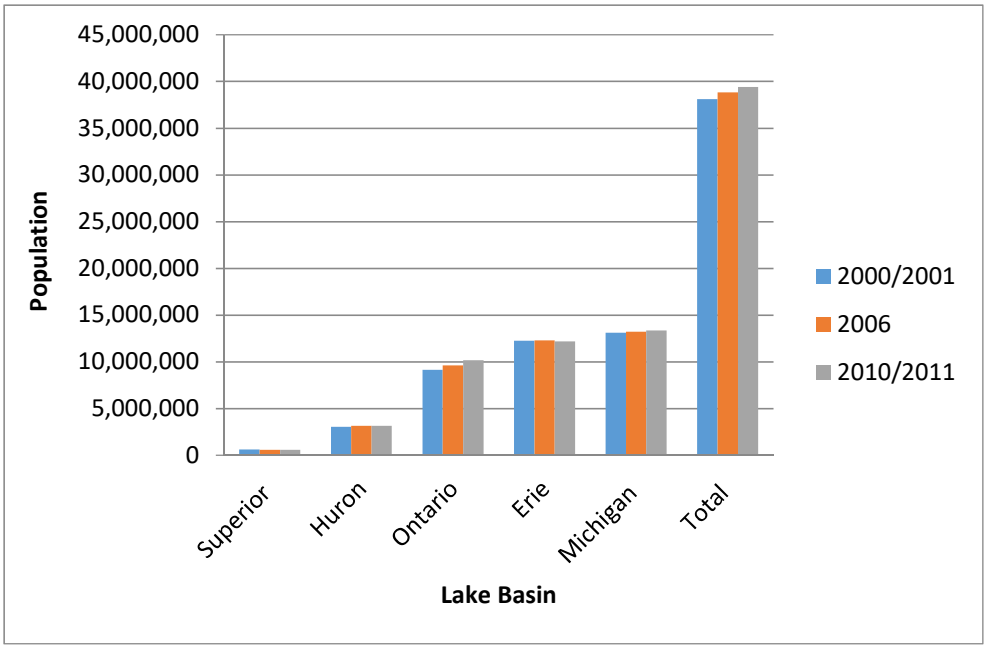


Figure 9. Total population in each Great Lakes basin from 2000 (US)/2001(CAN) and 2010 (US)/2011 (CAN).
Source: Data taken from Statistics Canada and U.S. Census Bureau.

Sub-Indicator: Precipitation Amounts in the Great Lakes Basin

Overall Assessment – Canadian Only
Trends:
10-Year Trend: Undetermined
30-Year Trend (1986-2015): Unchanging
Long-term Trend (1948- 2015): Increasing
Rationale: The annual precipitation anomaly (the departure from the 1961–1990 base-period average) for the long-term period of study (1948–2015) for Canadian stations within the Great Lakes basin displays a statistically significant increasing trend of 10.9% over the study period. Similarly, the trend for the 30-year period of study (1986-2015) was determined to be a positive trend of 6.2%, but this trend was not statistically significant and is therefore noted as Unchanging.

Lake-by-Lake Assessment
Individual lake basin assessments were not prepared for this report.

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: The calculated historical trend in annual or seasonal precipitation amount is positive and large enough to be distinguished from the variability of the data (is statistically significant)
Unchanging: The calculated historical trend in annual or seasonal precipitation amount is too small to be distinguished from the variability of the data (is not statistically significant)
Decreasing: The calculated historical trend in annual or seasonal precipitation amount is negative and large enough to be distinguished from the variability of the data (is statistically significant)
Undetermined: Data are not available to report on a trend or the time period is too short for viable precipitation trend analysis

Endpoints and/or Targets
No endpoint is needed for the climate-based sub-indicator reports. The frequency, seasonal distribution, and magnitude of precipitation will vary naturally from year to year.

Sub-Indicator Purpose
The purpose of this 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, 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 to the frequency, seasonal distribution, or magnitude of precipitation will impact the hydrological system of the entire Great Lakes basin, having effects such as altering water levels or changing the rates and patterns of storm runoff – directly 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 provides the annual and seasonal precipitation anomalies for the period of study (1948–2015) in the Great Lakes basin. For this sub-indicator, an annual or seasonal anomaly is the difference between the precipitation amount and reference value, which is further divided by the reference value and multiplied by 100. This was completed at each climatological station reporting precipitation measurements. The reference values used in this sub-indicator are the annual and seasonal precipitation averages for the base period of 1961–1990. Seasons are defined as winter (December, January, February); spring (March, April, May); summer (June, July, August); and autumn (September, October, November).

The 1961–1990 base period was chosen because the World Meteorological Organization (WMO 2007) recognizes it as a standard reference period for long-term climate change assessments. Moreover, this period consists of mostly manual and rigorously quality controlled observations before the onset, in the 1990s, of automation of observations.

Since stations reporting precipitation amounts are irregularly distributed over the Great Lake Basin, precipitation anomalies were interpolated to evenly spaced point locations (50-km spaced grid points) for a better spatial representation of the climate variations (Vincent et al. 2015). Annual and seasonal grid point values were then averaged together in order to produce annual and seasonal time series representing the Great Lake Basin.

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 the 2019 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 for the four climate related sub-indicator 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 four reports, but include a minimum of 15 years of data that are reliable and comprehensive to calculate the base values. 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. 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. 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, vary greatly from year-to-year (e.g. there is no inherent link in precipitation amounts from one year to the next). The moving average smooths the fluctuations so that any value of the running average is a better representation of the state of the variable than a single annual value alone.

Analysis was performed using the 147 grid-points of the Canadian Gridded Temperature and Precipitation Anomalies Dataset (CANGRD) that fell within the definition of the Great Lakes basin. The CANGRD is derived from the Adjusted and Homogenized Canadian Climate Data (AHCCD) dataset, a collection of rigorously treated land-based station records. CANGRD data is only calculated for grid-points occurring over land. More information about the precipitation data used in CANGRD can be found in Mekis and Vincent (2011). Also refer to the Canadian Environmental Sustainability Indicators (CESI) – Precipitation Change in Canada indicator.

Trend values were estimated using the method of Sen (1968). 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 using Kendall’s test (Kendall 1955) at the 0.05 level. While trends deemed statistically insignificant may still be true, there is also a tangible chance that they instead represent cyclic variations in the data.

Ecological Condition

The annual precipitation anomaly for 1948–2015 is shown in Figure 1. It displays a statistically significant (at the 0.05 level) increasing trend of 10.9% over the period. An increasing trend of 6.2% was found for the 30-year period from 1986–2015; however, this trend was found to be statistically insignificant which means the trend cannot be confidently discerned from the variability in the data signal over this time period.

The precipitation anomaly 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 (1948–2015) are shown in Figure 2. Autumn was found to have a statistically significant increasing trend of 15.2% over the study period while winter, spring, and summer each displayed statistically insignificant but increasing trends of 7.8%, 11.7%, and 8.9%, respectively.

While 2015 showed only a slight positive anomaly, the number and magnitude of positive anomalies is generally larger towards the end of the period (1948–2015). Five of the 10 years showing the most total annual precipitation during the entire study period, shown in Table 1, have occurred since 2000 with all but one (1977) of the 10 occurring in the later-half of the period. In contrast, the largest negative precipitation anomalies (shown in Table 2) tend to occur earlier in the study period, with 8 of the 10 falling in the first half. The largest positive precipitation anomaly (21.5%) was recorded in 2008 and the largest negative (-18.3%) in 1963.

Changes in precipitation indices show considerable spatial and temporal variability. The number of days with precipitation and with heavy precipitation (precipitation amount > 90th percentile) has increased over the past 7 decades at a number of stations in the Great Lake basin (Vincent et al. 2018).

Linkages

An increase in global temperature will enhance the ability of the atmosphere to store and transport water vapour, which will affect storm evolution and geographical distribution. In any one region, this may result in both the amount and type of precipitation varying at rates greater than those predicted by local climate change alone. This could result in significant changes in both precipitation event frequency and magnitude in the Great Lakes basin, which will affect 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 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 links directly to almost all other sub-indicators in the suite as precipitation events are a driving force in hydrology, nutrient and toxin distribution, and shoreline and wetland health. More intense precipitation events increase the risk of flooding and increase soil erosion 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 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			
Clarifying Notes: No U.S. data were included in this report.				

Data Limitations

Precipitation data suffers from discontinuities due to factors such as changes in observation method, equipment wear and replacement, station relocation and automation. To mitigate the effect of discontinuities, an analysis was completed using homogenized and adjusted station data (Mekis and Vincent 2011).

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 shorter, as only very large trends will be deemed statistically significant by tests.

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 (Logfren 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).

The inclusion of US-based data in future reports would be beneficial to help distinguish basin-wide changes in precipitation amounts from shifts in geographical precipitation distribution. The inclusion of stations south of the border would also improve the geographical representativeness of the data.

Future reporting cycles could benefit from an analysis of the number of extreme storms—defined as any storm 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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The Canadian Gridded Temperature and Precipitation Anomalies Dataset (CANGRD): http://open.canada.ca/data/en/dataset/3d4b68a5-13bc-48bb-ad10-801128aa6604

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List of Tables

Table 1. The 10 years showing the most annual precipitation (in % anomaly above the 1961–1990 mean) in the Great Lakes basin. Bolded years refer to five of the 10 years showing the most total amount of precipitation during the entire study period which have occurred since 2000.
Source: Environment and Climate Change Canada
Table 2. The 10 years showing the least annual precipitation (in % anomaly above the 1961-1990 mean) in the Great Lakes basin.
Source: Environment and Climate Change Canada

List of Figures

Figure 1. Annual precipitation anomaly (from the 1961–1990 mean) for the Great Lakes basin over the period 1948–2015. 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 1952 and the last is 2011.
Source: Environment and Climate Change Canada
Figure 2. Nine-year running means of seasonal precipitation anomalies (from the 1961–1990 seasonal means) for the Great Lakes basin over the period of record (1948–2015). 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 1952 and the last is 2011.
Source: Environment and Climate Change Canada

Last Updated
State of the Great Lakes 2019 Technical Report
(no new data for SOGL 2019)

Rank	Year	% Anomaly
1	**2008**	21.5
2	1985	18.1
3	**2013**	17.9
4	**2014**	16
5	1996	15.3
6	1977	11.2
7	**2004**	10.2
8	1995	10
9	1988	9.9
10	**2001**	9.3

Table 1. The 10 years showing the most annual precipitation (in % anomaly above the 1961–1990 mean) in the Great Lakes basin. Bolded years refer to five of the 10 years showing the most total amount of precipitation during the entire study period which have occurred since 2000.
Source: Environment and Climate Change Canada

Rank	Year	% Anomaly
1	1963	-18.3
2	1958	-17.1
3	1987	-13.1
4	1981	-10.9
5	1962	-10.6
6	1952	-10.5
7	1998	-10.4
8	1978	-9
9	1961	-8.4
10	1956	-8.1

Table 2. The 10 years showing the least annual precipitation (in % anomaly above the 1961-1990 mean) in the Great Lakes basin.
Source: Environment and Climate Change Canada

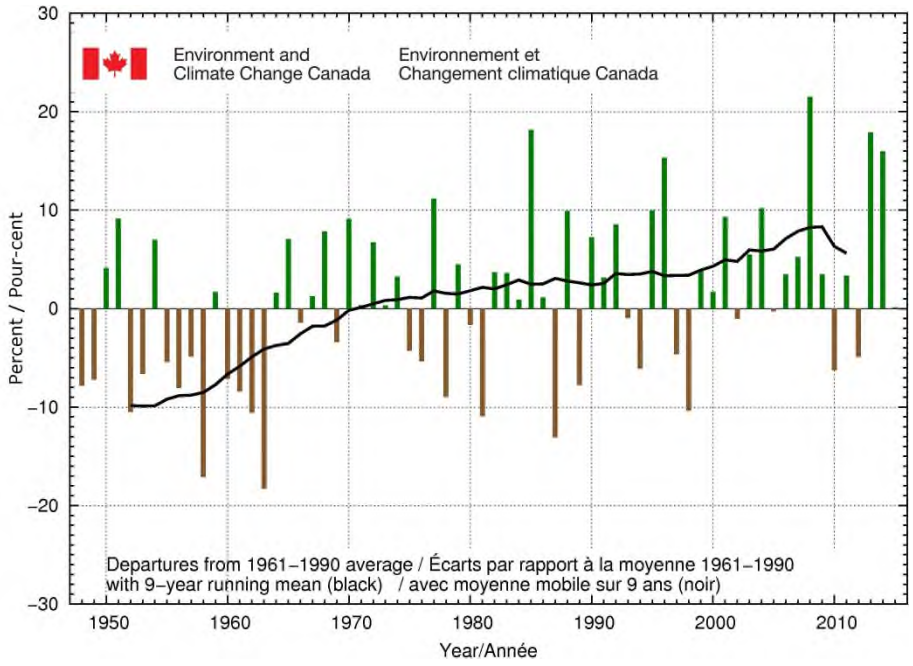


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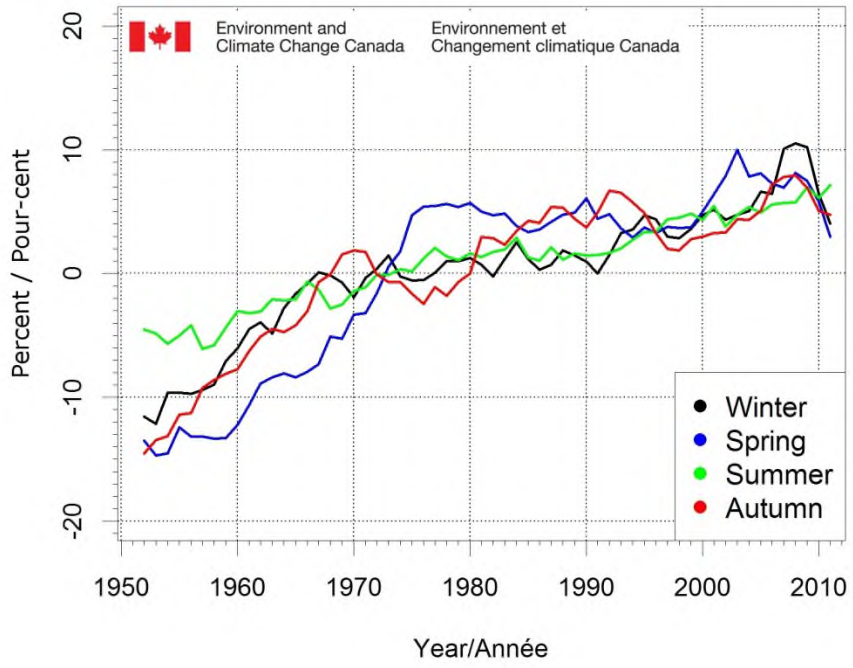


Figure 2. Nine-year running means of seasonal precipitation anomalies (from the 1961–1990 seasonal means) for the Great Lakes basin over the period of record (1948–2015). Note that the mean for a particular 9-year interval is centred on the middle year, meaning the first year for which the running mean can be defined is 1952 and the last is 2011.
Source: Environment and Climate Change Canada

Sub-Indicator: Water Levels

Overall Assessment
Trends:
10-Year Trend (2008-2017): Unchanging to Increasing, depending on lake
30-Year Trend (1988-2017): Unchanging
Long-term Trend (1918-2017): Unchanging to Increasing, depending on lake
Rationale: Notable basin-scale water level dynamics for the 1918-2017 period of record include a significant decline in the late 1990s, persistent and record-low monthly mean levels on Superior (August and September 2007) and Michigan-Huron (December 2012 and January 2013), a record setting water level rise on both Superior and Michigan-Huron between January 2013 and December 2014, record high water levels on Lake Ontario in May to July 2017, and generally above average water levels from 2014 to 2017 on Lakes Superior, Michigan-Huron and Erie. 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 2017) 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 in order to be consistent with other State of the Great Lakes sub-indicators. However in the future, 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 the lakes except Lake Ontario. 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 100-year period from 1918 to 2017, only Lakes Erie and Ontario showed a statistically significant upward trend in water level.

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

Lake-by-Lake Assessment
Lake Superior (including St. Marys River)
10-Year Trend (2008-2017): Increasing
30-Year Trend (1988-2017): Unchanging
Long-term Trend (1918-2017): 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 centimetres below record high values by the end of 2017. The rise during 2013 and 2014 was the largest rise observed over a 2-year period in the historical record (1918-2017). Although water levels on Lake Superior are higher than they were at the time of the previous State of the Great Lakes (SOGL) reports, it is difficult to determine whether this is due to interannual variability in climatic conditions or whether it is an indication of longer term trends. Neither the 30-year nor the long-term trends showed a significant trend relative to the base period.

The 5-year increasing trend from the 2017 report was similar to the shorter term 10-year trend due to the influence of the below average levels between 1999 and mid-2014 and the rise to above average conditions from mid-2014 to present. Both the 30-year and long-term trend changed from decreasing in the 2017 analysis to unchanging, which is likely due to the influence of the recent higher water levels. Note that this change in trend from one analysis to another is not surprising due to the interannual variability and even multi-year variability of water levels that includes periods

of high and periods of low water levels. 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 regards to water depths, velocities and wetted perimeter would need more detailed study.

Lake Michigan-Huron
10-Year Trend (2008-2017): Increasing
30-Year Trend (1988-2017): Unchanging
Long-term Trend (1918-2017): 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 record-setting 2-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 2017. Although water levels on Lake Michigan-Huron are higher than they were at the time of the previous SOGL report, the inter-annual variability in climate conditions and the regional hydrologic cycles makes it difficult to determine whether the 10-year trend will persist. 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.

No significant trend was reported for the 5-year analysis period in the 2017 report. However, the 10-year trend in this report is increasing due to the influence of the below average levels during the 2000s, which were followed by a rapid rise in water levels from 2013 to 2017. The 30-year trend in the 2017 analysis indicated a decreasing trend in water levels, which was also found in the 2019 report however, this trend was not statistically significant (P=0.09).

Lake Erie
10-Year Trend (2008-2017): Increasing
30-Year Trend (1988-2017): Unchanging
Long-term Trend (1918-2017): 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 2017 shows an upward trend relative to the base period. However, water level conditions on Lake Erie (unlike conditions on Lakes Superior and Michigan-Huron in 2012 and 2013) did not approach record annual and monthly lows during this period. It is informative to note, however, that water levels on Lake Erie rose dramatically in 2011 (in response to widespread increased annual precipitation) and then declined (starting in December 2011) for 10 straight months. Compared with the 5-year analysis in 2017, which was found to have no significant change, the 10-year analysis of this report found a significant increasing trend. Comparing the 30-year analysis from each report, the 2017 report analysis indicated a decreasing trend while the 2019 calculated trend indicated no significant change. The wetter conditions in the last few years could help explain these differences. In both the 2017 and 2019 SOGL reports, the 100-year trends show a significant increasing trend for the lake level. However, there is no indicator that would suggest that this wetter condition will continue.

Lake Ontario (including the St. Lawrence River)
10-Year Trend (2008-2017): Unchanging
30-Year Trend (1988-2017): Unchanging
Long-term Trend (1918-2017): Increasing
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, and is expected 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 impacts of implementation of Plan 2014 in this analysis is considered negligible as it was only in implementation for the part of 2017, and due to record high precipitation during its time of implementation, lake levels were driven by precipitation and not changes in outflow regulation due to the regulation plan changes. Despite record high water levels in 2017, the short, medium, and long-term trends did not change from the 2017 report to this report.

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
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 the linear model fit. It is noted that this method is relatively simple compared to other possibilities and consideration could be made for use of more sophisticated methods in the future. 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 (1918-2017), which is not surprising due to interannual and multi-year variability of Great Lake water levels. The 1918 to 2017 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, and are determined from a network of gauges throughout each lake. Data are available from 1860 to 1917 but were not used in the analysis because they were computed from only a single gage on each lake. The use of only one gage to represent a lakewide average may not account for local variations in water levels. However an analysis of data over the 1860 to 2017 period was performed and changes in long-term trends were found (i.e. Lake Michigan-Huron was found to have a significant decreasing trend, and both Lakes Erie and Ontario are unchanging). This indicates sensitivity of the analysis to period of record chosen for the analysis. In addition to computing trends, Figure 1 also includes a line showing the 10-year rolling average anomaly.

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

Endpoints and/or Targets
It is generally thought that naturally fluctuating water levels are supportive of a healthy Great Lakes ecosystem. For this reason, no endpoint or targets will be developed for this sub-indicator. No endpoint is needed for the climate-based sub-indicator reports. The ecosystem adapts to periods of high and low water levels – it's having a prolonged period of either that could pose problems.

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 1918-2017).

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-average water level measurements. Lake-wide average 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 measures used are based on monthly average water level records from gage data collected between 1918 and 2017.

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 2019 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. However, it was found that the analysis was not particularly sensitive to the base period chosen.

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 test comparison using the past methods and the anomaly-based approach was conducted with the same data sets and it was found that the methods gave similar results indicating that any differences found in trends compared to previous reports is due to the additional data of 2016 and 2017. The moving average of 10 years of data was provided for consistency with other sub-indicators and general comparison.

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. These are influenced by natural and anthropogenic factors, and long-term climate trends (Baedke and Thompson 2000; Booth and Jackson 2003). Changes in Great Lakes water levels not only impact the flooded 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 de-

grees 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.

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, while an important contribution to the water cycle, 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).

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, and from the early 1970s to mid-1990s. Pronounced low lake level periods occurred in the mid-1920s, the mid-1930s and the mid-1960s. Though less-well documented, low levels also occurred in the late 1890s, following a long period of high lake levels.

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 be 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.

Other Spatial Scales

Water level fluctuations at finer spatial scales (including the fluctuations on Lake St. Clair) are not considered in this report.

Linkages to Other Sub-indicators

Great Lakes water levels are linked to a number of other sub-indicators. The natural hydrological cycle is the primary determinant of all lake levels. Key components of this are precipitation and runoff on the watershed, and evaporation from the lake surface. In turn, evaporation off the lakes is significantly influenced by the difference in air and surface water temperature and lake ice cover, with the highest evaporation rates from the lakes occurring in the fall and ice free winter. The relationships to other sub-indicators is complex and efforts are currently being undertaken to better quantify the interrelationship. Changes in Great Lakes water levels are also interrelated with the lakes coastal processes (e.g. interrelationship of lake level, wave action, shoreline substrate on sediment supply and transport) and shoreline management practices (e.g. shoreline protection, dredging, dyking, beach nourishment)

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*, Preyfish, 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.

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 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. One of the more potentially significant impacts on interpretation of water level indicators are short-term perturbations in the regional climate system including, for example, the 1997-1998 strong winter El Niño coincident with a period-of-record (1918-2017) extreme water level decline, and the extreme cold winter of 2013-2014 and basin wide high precipitation events of 2017, which were coincident with period-of-record extreme rates of water level rise.

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. 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.

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.

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. Interactions 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 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.

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-2017)
2. Magnitude of seasonal rise and decline: an assessment of trends over time in the magnitude of spring rise, and the magnitude of fall decline. (data from late-1990s-2017)
3. 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 1918-2017)
4. Changes in coastal processes on the Great Lakes with changes in water levels and associated impacts on habitat structures.
5. The conclusions of this report are based on the best available data set at the time of writing this report (i.e. 1918 to 2017). It is important to note that Lakes Erie and Ontario reached period-of-record-high values and lakes Superior and Michigan-Huron were at or above seasonal-period-of-record-high values during the late spring and summer of 2019, which was not available at the time of preparing these results. Trends should be reevaluated as data becomes available.

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Keith Kompoltowicz, United States Army Corps of Engineers
The authors would like to acknowledge use of material from the 2017 SOGL Technical Report: Author Drew Gronewold, University of Michigan

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List of Tables

Table 1. Trend line P-values. A value $P<0.05$ was considered significant.
Data from USACE

List of Figures

Figure 1. Water level anomalies relative to average of the baseline period 1918-1990 for each Great Lake over the period 1914-2017. Note that the mean of each 9-year interval is centered on the middle year so the first year of the running mean is 1918.
Data from USACE

Figure 2. Monthly and yearly mean annual water levels for the Great Lakes.
Source: Fisheries and Oceans Canada – Canadian Hydrographic Services.

Last Updated

State of the Great Lakes 2019 Technical Report

Trend	Great Lake			
	Superior	Michigan-Huron	Erie	Ontario
10-year	0.00	0.01	0.03	0.37
30-year	0.77	0.09	0.37	0.81
Long-term	0.25	0.57	0.00	0.00

Table 1. Trend line P-values for anomalies for 1918-2017 computed using 1918-1990 baseline period. A value P<0.05 was considered significant.
Data from USACE

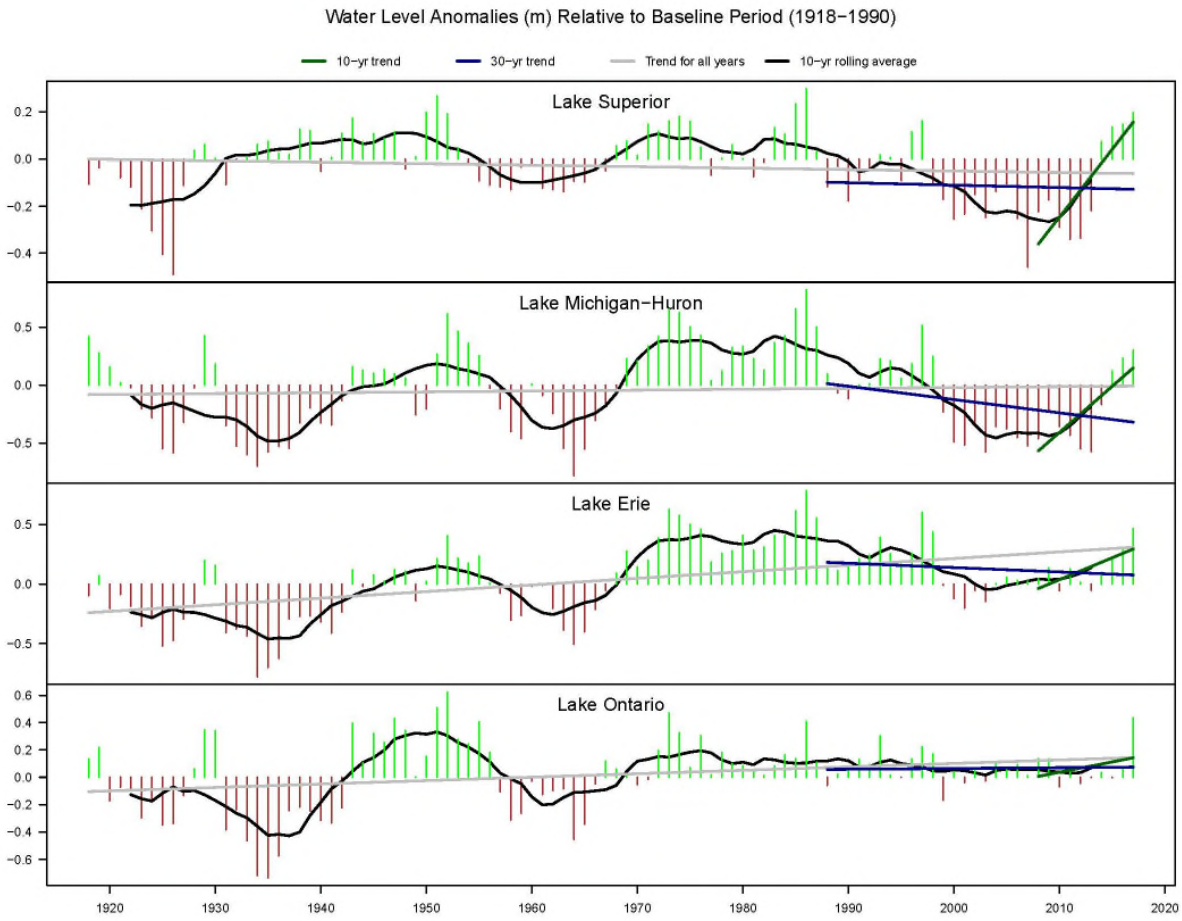


Figure 1. Water level anomalies relative to average of the baseline period 1918-1990 for each Great Lake over the period 1914-2017. Note that the mean of each 9-year interval is centered on the middle year so the first year of the running mean is 1918.
Data from USACE

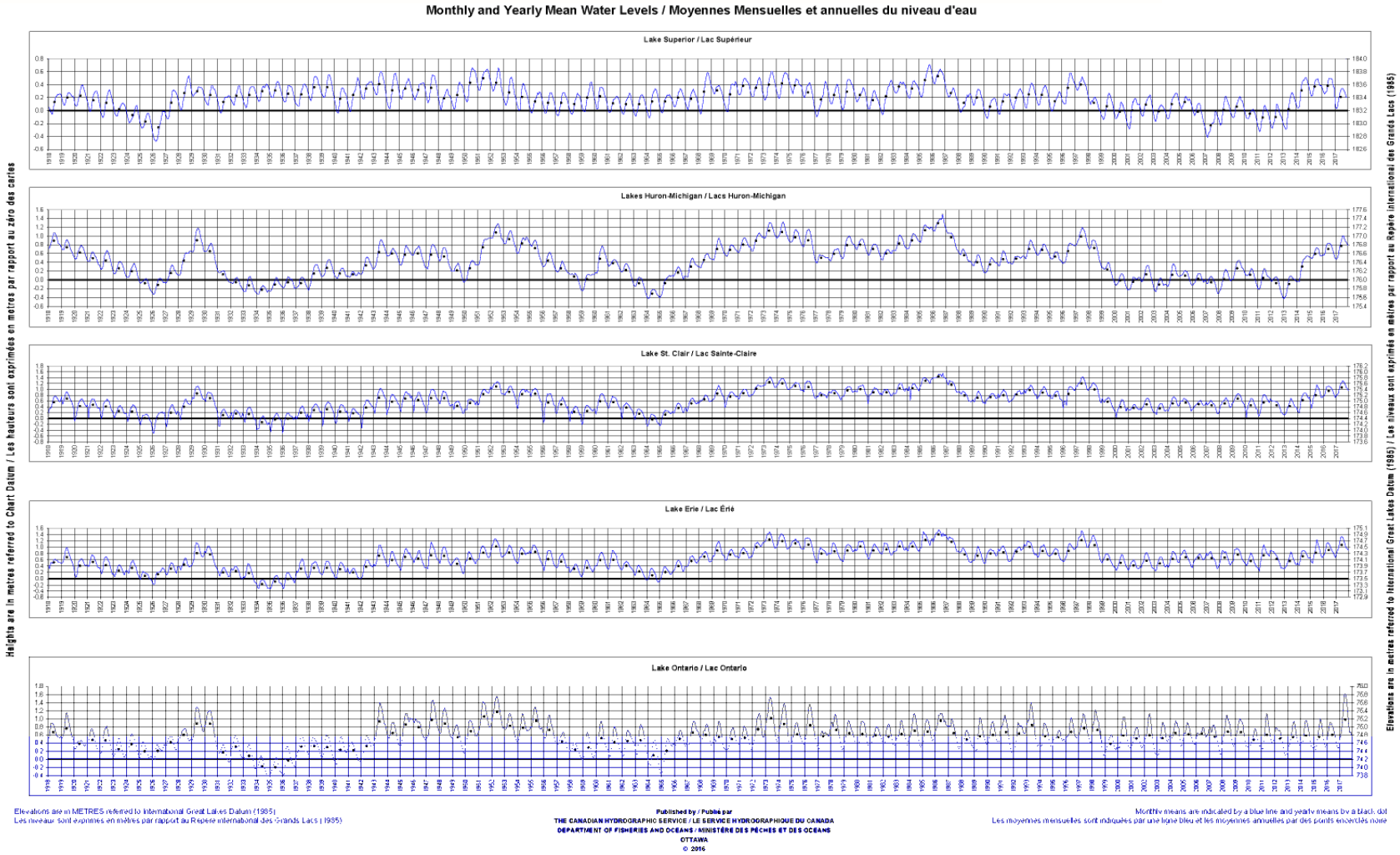


Figure 2. Monthly and yearly mean annual water levels for the Great Lakes.
Source: Fisheries and Oceans Canada – Canadian Hydrographic Service

Sub-Indicator: Surface Water Temperature

Overall Assessment
Trends:
10-Year Trend: Undetermined
30-Year Trend: Increasing
Long-term Trend (1980-2017): Increasing
Rationale: An overall assessment is difficult to assign given that the different lakes appear to be responding differently. However, in general, based on the buoy observations (Figure 2), it appears that the upper Great Lakes (Superior, Michigan, Huron) show statistically significant trends towards higher temperatures over the last 30 years, and will be classified as "Increasing". However, the lower Great Lakes (Erie and Ontario) do not appear to show significant trends, and will be classified as "Unchanging" or "Undetermined" (see Lake-by-Lake analysis below). Model output (Figure 3 and 4) is generally consistent with the buoy output (Figure 5) 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 are not statistically meaningful and will not be assessed.

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

Lake-by-Lake assessment
Lake Superior
10-Year Trend: Undetermined
30-Year Trend: Increasing
Long-term Trend (1980-2017): Increasing
Rationale: The average temperature over the last 10 years, estimated by the numerical model, is approximately 3C higher than the baseline period average. The average date of the onset of stratification over the last 10 years is 25 days earlier. Data is collected by three surface buoys in Lake Superior during the period of interest. All three buoys display a significant trend towards warmer surface waters, on the order of 0.07 °C/yr, a little less than 1°C per decade. This is exacerbated by a significant step-change in 1998, after which temperatures are typically (but not entirely) warmer than previous years. The year 2014 stands out as a particularly cold summer, although 2012 is essentially tied with 1998 for the warmest summer in the instrumental record. Model output from LLTM confirms this, with temperatures after 1997 significantly higher than the "baseline" temperature.

Lake Michigan
10-Year Trend: Undetermined
30-Year Trend: Increasing
Long-term Trend (1980-2017): Increasing
Rationale: The average temperature over the last 10 years, estimated by the numerical model, is approximately 4C higher than the baseline period average. The average date of the onset of stratification over the last 10 years is 39 days earlier. Data is collected by two surface buoys in Lake Michigan during the period of interest. Both display statistically significant increasing surface water temperature trends, on the order of 0.05 °C /yr, about 0.5 °C per decade. This is exacerbated by a significant step-change in 1998, after which temperatures are typically (but not entirely) warmer than previous years. Model output from LLTM confirms this, with temperatures after 1997 significantly more than the "baseline" temperature.

Lake Huron
10-Year Trend: Undetermined
30-Year Trend: Increasing
Long-term Trend (1980-2017): Increasing
Rationale: The average temperature over the last 10 years estimated by the numerical model is approximately 3C higher than the baseline period average. The average date of the onset of stratification over the last 10 years is 18 days earlier. Data is collected by two surface buoys in Lake Huron during the period of interest. Both display

statistically significant increasing surface water temperature trends, on the order of 0.07 °C/yr, a little less than 1 °C per decade. This is exacerbated by a significant step-change in 1998, after which temperatures are typically (but not entirely) warmer than previous years. 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
10-Year Trend: Undetermined
30-Year Trend: Unchanging
Long-term Trend (1980-2017): Unchanging
Rationale: The average temperature over the last 10 years estimated by the numerical model is approximately 1C higher than the baseline period average. The average date of the onset of stratification over the last 10 years is 20 days earlier. Data from a single buoy on Lake Erie had a reported increasing linear trend in surface water temperature of 0.016 °C/yr, just barely above the uncertainty in that rate (0.012 °C/yr). The LLTM model output is consistent with this result, showing no appreciable trend.

Lake Ontario
10-Year Trend: Undetermined
30-Year Trend: Undetermined
Long-term Trend (1989-2017): Undetermined
Rationale: The average temperature over the last 10 years, estimated by the numerical model, is approximately 1C higher than the baseline period average. The average date of the onset of stratification over the last 10 years is 13 days earlier. As data from the NDBC buoy in Ontario dates back only to 2002, insufficient data exists with which to make an assessment of a long term trend. The linear trend (0.02 °C/yr) is significantly less than the uncertainty in that trend (0.06 °C/yr). The LLTM model output also shows no discernible trend.

Status and Trend 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: A site was classified as "Increasing" if the linear trend (determined by standard methods of linear regression analysis) was positive and significantly in excess of the uncertainty associated with that estimate.
Unchanging: A site was classified as "Unchanging" if the linear trend (determined by standard methods of linear regression analysis) was of the same order or smaller than the uncertainty associated with that estimate.
Decreasing: A site was classified as "Decreasing" if the linear trend (determined by standard methods of linear regression analysis) was negative and significantly in excess of the uncertainty associated with that estimate.
Undetermined: A site was classified as "Undetermined" if insufficient data was available to make an estimate with any statistical significance.

Endpoints and/or Targets
No endpoint is needed for the climate-based sub-indicator reports.

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 duration and spatial extent of 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
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 2019 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 include a minimum of 15 years of data that are reliable and comprehensive to calculate the base values. 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. 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. 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.

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) (Figure 1), which goes back as far as 1979, data from Environment and Climate Change Canada (ECCC), and model output from the Large Lake Thermodynamic Model (LLTM) (Croley 1989), which goes back as early as 1950. For the numerical model output, 1960-1990 will be used as a "baseline" period in order to display results in terms of anomaly. However, by doing so, it is not intended to suggest that this baseline period was "normal" in any sense of the word. *In-situ* buoy data will be presented in absolute terms since it is difficult to determine a baseline value. As there are offsets between buoy data and model output, it is not appropriate to use the model baseline values as a baseline for the buoy data; however, the buoy data ground-truths the model trends. To assess trends, average July-August-September surface water temperature data will be used as an annual metric of thermal status. As an additional metric, date of the onset of stratification will be used, defined as the last day in the spring when the surface water temperature is below 4 °C. 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.

Ecological Condition
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". While the date of the onset of winter conditions would also be a useful metric, it is a more difficult date to determine, and most buoys are not left in late enough into the season to make consistent observations of it.

There is a great deal of natural inter-annual variability that sits on top of the warming trend. 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.

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 (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.

The previous report in 2017 included data through 2014. This report includes data through 2017. Therefore, not enough additional data is present to rationalize a statistically significant change in trend, either overall or on a lake-by-lake basis.

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 (unpublished) work has shown a strong but unsurprising link between average winter air temperatures and the amount of ice cover, suggesting a series of statistically significant 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) suggest 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 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 5) 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 due to the fact that the model is attempting to provide a "lakewide" average temperature, whereas the buoy observations are made at distinct points within these large, spatially heterogeneous systems.

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 fresh water, 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.

Increased atmospheric temperatures will result in increased water temperatures in the Great Lakes region. Water temperature is an integral component of the physical, chemical, and biological processes of the basin, including the timing and duration of stratification periods.

Due to the correlation between trends in water temperature and global mean temperatures, the Great Lakes may serve as a metric for observing regional temperature changes.

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.

This sub-indicator measures the thermal properties of the Great Lakes that affects the ecosystems' function and influences water evaporation from the lakes that affects lake's water level (if higher surface water temperatures persists, this will may lead to reduced winter ice cover and increased water evaporation from the lakes resulting in lower water levels). Recent work suggests that ice cover trends may ultimately not affect lake levels.

The observed trends towards earlier stratification and higher summer water temperatures are driven by long-term changes in atmospheric conditions, primarily air temperature, which is widely acknowledged to be a consequence of changing atmospheric chemistry, specifically the addition of carbon dioxide (and to a lesser extent methane) from the burning of fossil fuels. It is important, however, that the Great Lakes community continue to develop a better understanding of the impacts of warming on lake ecosystems, as well as to continue to carefully document these trends.

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. 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.

Since the fall overturn is important to reporting, the need to have mooring buoys deployments extend to late October of each year may be required and recommended in order to capture more robust data with respect to fall turnover.

Consider the addition of satellite imagery (GLERL) and the GLSEA (Great Lakes Surface Environmental Analysis) in future reports for judging spatial variability within the lakes. Additional collection and analysis of data for lakes Ontario and Erie would be beneficial.

Acknowledgments
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Ram Yerubandi, Environment and Climate Change Canada
Tim Hunter, National Oceanic and Atmospheric Administration

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List of Figures

Figure 1. Location of surface buoys used in this report.
Source: National Data Buoy Center (NDBC) and Environment and Climate Change Canada (ECCC)
Figure 2. Mean summer surface (JAS) water temperature, 1979 to 2017.
Source: National Data Buoy Center (NDBC) and Environment and Climate Change Canada (ECCC)

Figure 3. Mean summer (JAS) surface water temperature anomalies from LLTM output, 1950 to 2017. Shaded region represents baseline period.
Source: Large Lake Thermodynamic Model
Figure 4. Date of onset of stratification (>4C) from LLTM model output, 1950 to 2017. Shaded region represents baseline period.
Source: Large Lake Thermodynamic Model
Figure 5. Model output- buoy data comparison. WT = Water Temperature.
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Last Updated
State of the Great Lakes 2019 Technical Report



Figure 1. Location of surface buoys used in this report.
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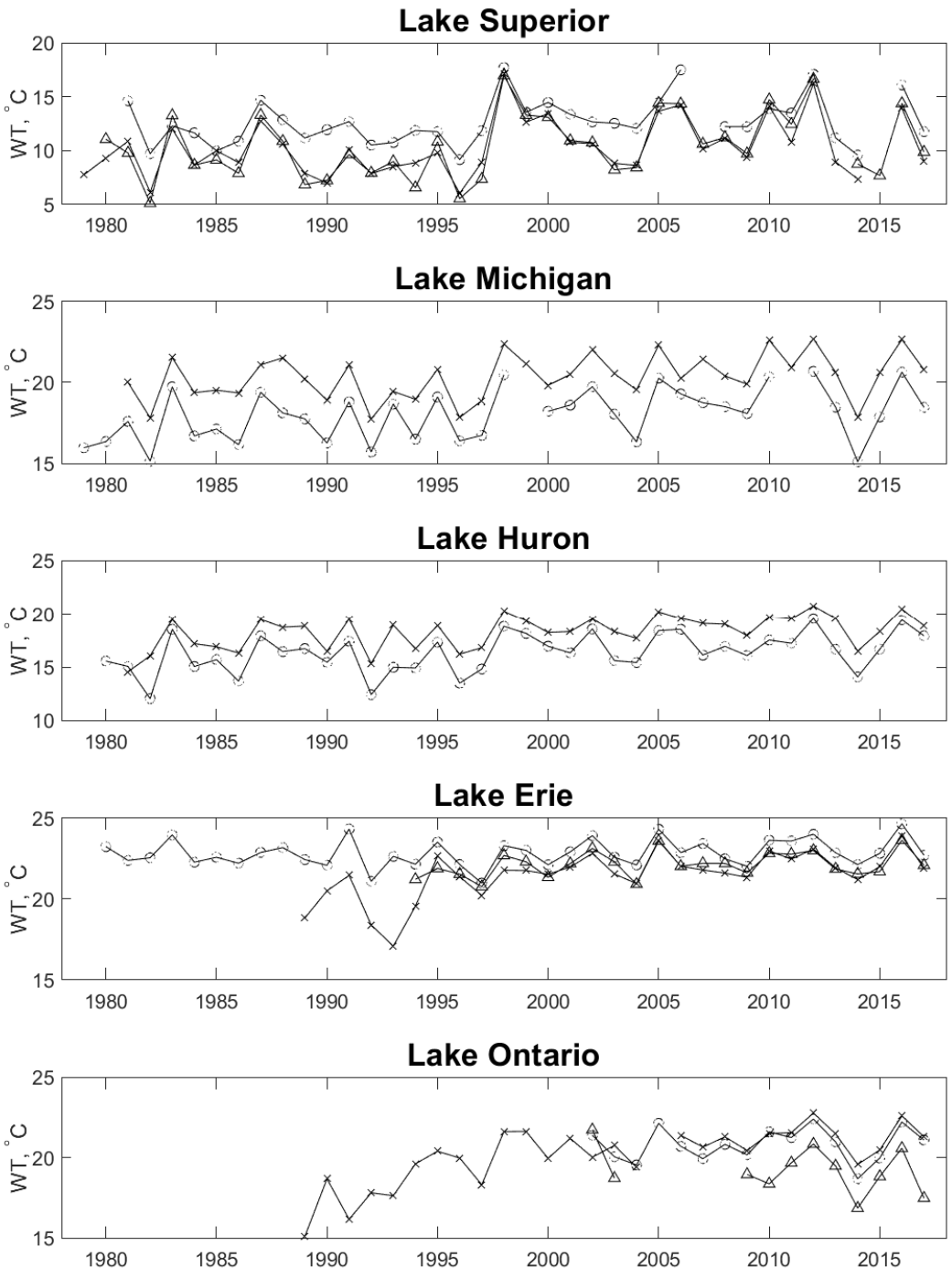


Figure 2. Mean summer surface (JAS) water temperature, 1979 to 2017.
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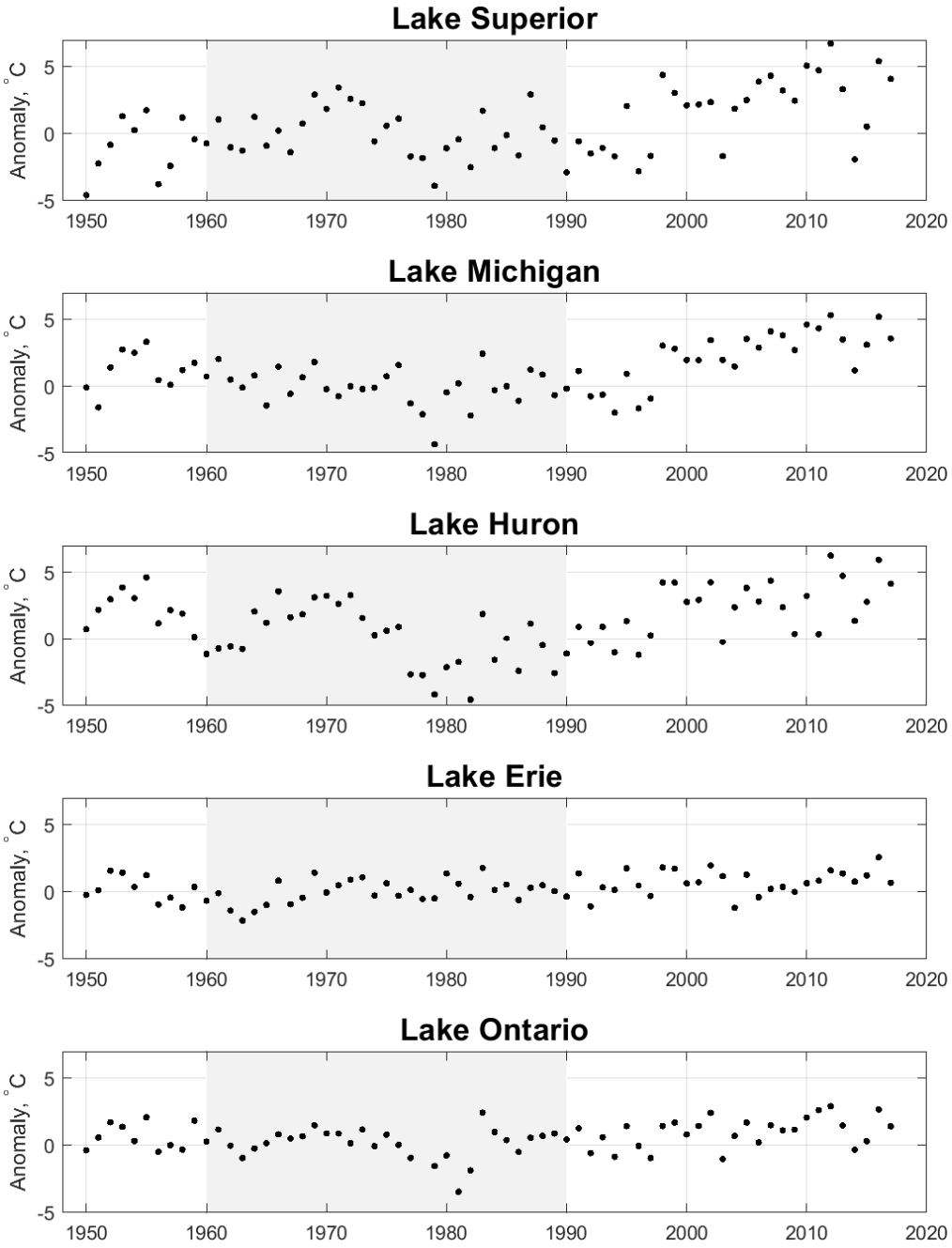


Figure 3. Mean summer (JAS) surface water temperature anomalies from LLTM output, 1950 to 2017. Shaded region represents baseline period.
Source: Large Lake Thermodynamic Model

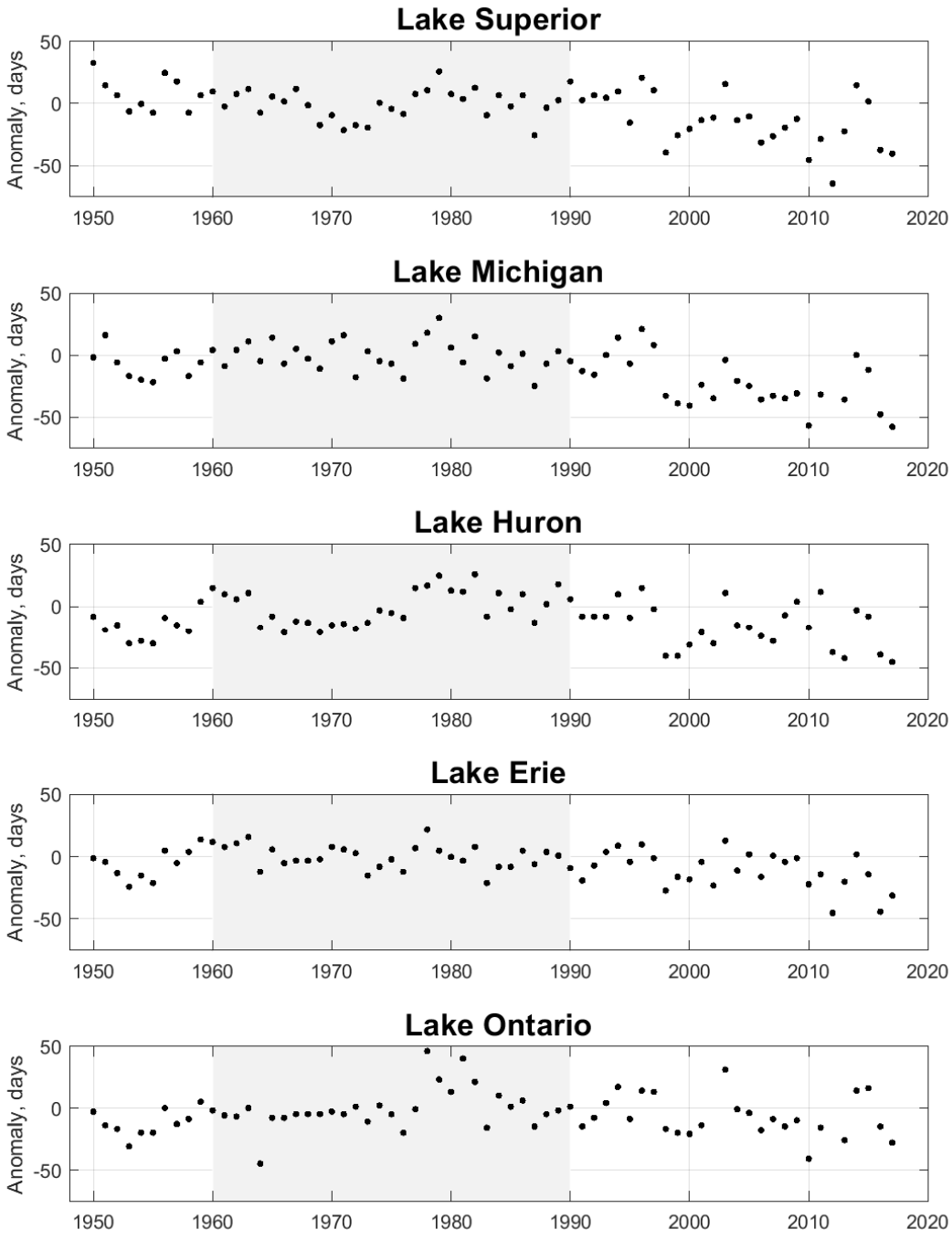


Figure 4. Date of onset of stratification (>4C) from LLTM model output, 1950 to 2017. Negative anomaly values represent earlier onset of stratification. Shaded region represents baseline period.
Source: Large Lake Thermodynamic model

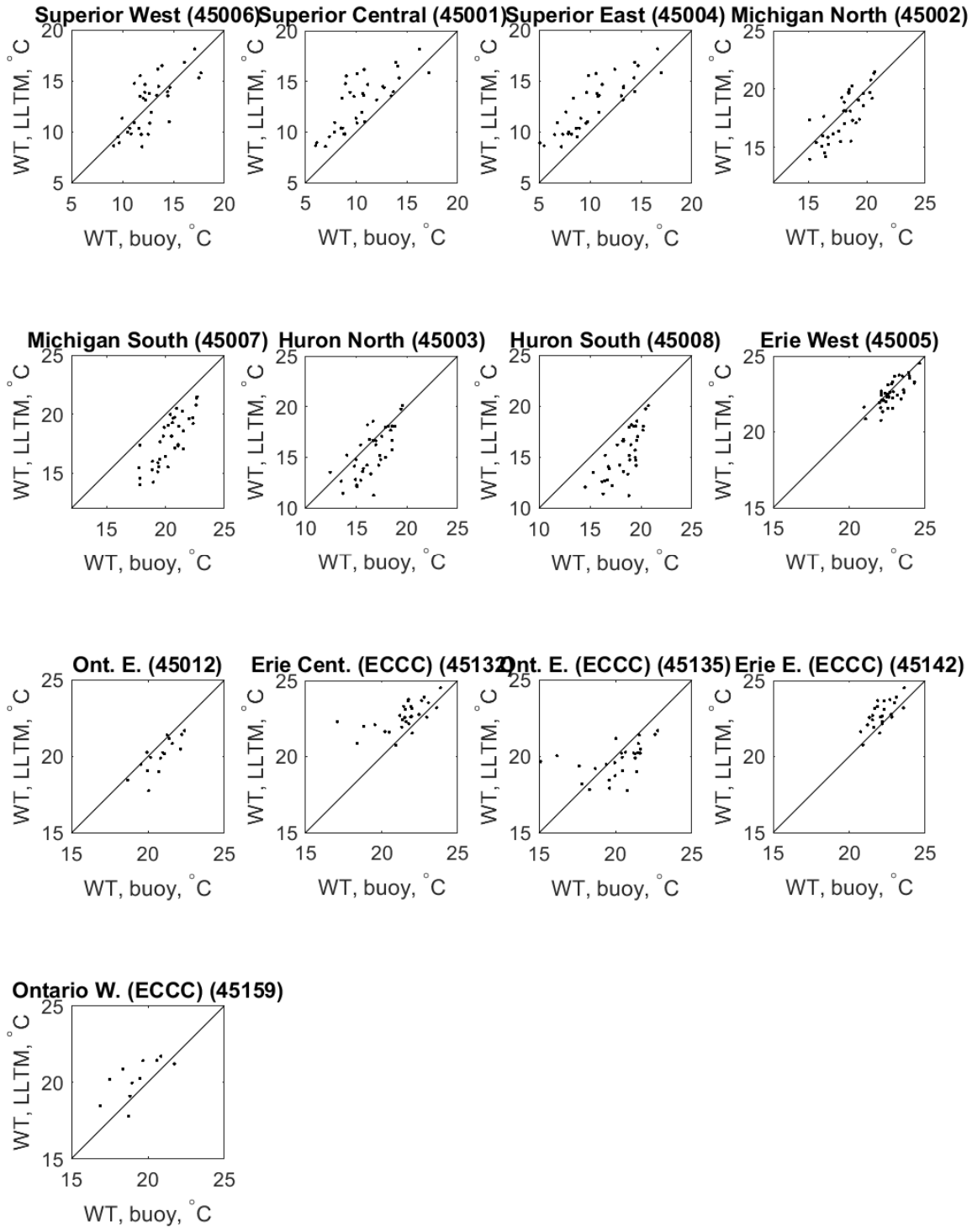


Figure 5. Model output- buoy data comparison. WT = Water Temperature.
Source: Source: National Data Buoy Center (NDBC) and Environment and Climate Change Canada (ECCC)

Sub-Indicator: Ice Cover

Overall Assessment
Trends:
10-Year Trend (2009-2018): Slightly increasing
30-Year Trend (1989-2018): Decreasing
Long-term Trend (1973-2018): Decreasing
Rationale: The trends are based on maximum ice cover data. The annual maximum ice cover (AMIC) anomaly (derived from the 1973 to 1990 base period average) for the period of 1973 to 2018 demonstrates a statistically significant trend of -0.53% annually for the Great Lakes. This implies a basin-wide ice loss of 24.6% over the period of 1973 to 2018. Note that a separate determination of 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. The 10-year and 30-year trends in maximum ice cover anomalies are 0.80% and -0.22%, respectively but are not statistically significant.

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

Lake-by-Lake Assessment
Lake Superior
10-Year Trend (2009-2018): Slightly increasing
30-Year Trend (1989-2018): Decreasing
Long-term Trend (1973-2018): Decreasing
Rationale: The long-term trend for the ice cover anomaly is decreasing at an annual rate of -0.75%, translating into a lake-wide decrease in maximum ice cover of 34.61% from 1973 to 2018. Lake Superior has experienced the highest diminishment of maximum lake ice coverage over the historical period of all the Great Lakes, because the deepest lake has the largest positive ice/lake albedo feedback that results in the largest warming in water temperature, leading to more ice loss than the other lakes. The trends for the 10-year and 30-year periods are unchanging as they were deemed statistically insignificant, and were 0.27% and -0.96% respectively.

Lake Michigan
10-Year Trend (2009-2018): Increasing
30-Year Trend (1989-2018): Slightly increasing
Long-term Trend (1973-2018): Decreasing
Rationale: Lake Michigan maximum ice cover anomalies show a decreasing (-0.28% per year) but statistically insignificant trend for the long-term period. Statistically insignificant 10-year and 30-year trends in maximum ice cover anomaly tendencies show an increasing (1.02%) trend for the 10-year period and (0.25%) in the 30-year timeframe. These trends may be the result of recent heavy ice seasons in 2013/14 and 2014/15 that have opposed the trend of decreasing maximum ice cover anomalies and these outliers have reduced the statistical significance of the trend in the dataset.

Lake Huron (including St. Marys River)
10-Year Trend (2009-2018): Increasing
30-Year Trend (1989-2018): Slightly decreasing
Long-term Trend (1973-2018): Decreasing
Rationale: A statistically insignificant annual decrease in the maximum ice cover anomaly of -0.42% was noted for the long-term timeframe at the 0.05 level. Shorter period indicators for Lake Huron were also deemed to be statistically insignificant and thus unchanging, however the trend demonstrated for these statistically insignificant relationships was -0.20% over the last 30 years and 0.88% for the last 10 years. These trends may be the result of recent heavy ice seasons in 2013/14 and 2014/15 that have opposed the trend of decreasing maximum ice cover anomalies and these outliers have reduced the statistical significance of the trend in the dataset.

Lake Erie (including St. Clair-Detroit River Ecosystem)
10-Year Trend (2009-2018): Decreasing
30-Year Trend (1989-2018): Decreasing
Long-term Trend (1973-2018): Decreasing
Rationale: Lake Erie exhibits an unchanging pattern in all maximum lake ice coverage anomalies calculated for all the specified time periods due to a lack of statistical significance. Maximum ice cover anomalies all trend negatively on an annual basis across the long-term (-0.43%), 30-year (-0.07%) and 10-year (-1.34%) periods of examination. The long-term trend translates into a lake-wide decrease in maximum ice cover of 19.63% from 1973 to 2018. Lake Erie is the shallowest and most diverse lake so this loss of lake ice can have significant impacts in the ecosystem. These trends may be the result of recent heavy ice seasons in 2013/14 and 2014/15 that have opposed the trend of decreasing maximum ice cover anomalies and these outliers have reduced the statistical significance of the trend in the dataset.

Lake Ontario (including the St. Lawrence River)
10-Year Trend (2009-2018): Increasing
30-Year Trend (1989-2018): Slightly increasing
Long-term Trend (1973-2018): Decreasing
Rationale: A similar pattern in the anomaly analysis emerges for Lake Ontario as the long-term trend in maximum ice cover is decreasing at a rate of -0.35% per year for the 1973 to 2018 period. This trend however was not found to be statistically significant at the 0.05 level. Statistically insignificant trends of 1.54% for the 10-year period and 0.18% for the 30-year timeframe of maximum ice cover anomalies were noted. These trends may be the result of recent heavy ice seasons in 2013/14 and 2014/15 that have opposed the trend of decreasing maximum ice cover anomalies and these outliers have reduced the statistical significance of the trend in the dataset.

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 – it's having a prolonged period of either that 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 and accompanying physical changes to each lake over time by measuring the thermal properties of the Great Lakes. Thermal properties affect the ecosystems' function and influence water evaporation 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. 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, from erosion by waves and storms.

Ecosystem Objective
Change in lake ice cover during the winter due to climate change will affect water temperature 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 2018 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 outliners (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 2018, 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 was 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 multidecadal variability in lake ice caused by multidecadal 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 for 2019 State of the Great Lakes reports.

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 Condition

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. More importantly, ice cover directly controls lake water temperature change, duration of stratification, and fish behaviours. 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 AMIC trend. The highest trend is located along heavy ice-covered coasts in Lake Superior, Georgian Bay, northern Lake Huron, and northern Lake Michigan. Offshore ice cover has smaller trend, since 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 distribution 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 Nino, 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 one of the few factors that control the formation of ice on that surface. Colder winter temperatures increase the rate of heat released 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 to give their eggs protection against predators 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 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.

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 control the heat and moisture exchange between the lake and the atmosphere
- Precipitation Amounts – ice cover can control the lake-effect snow.
- Coastal Wetlands: Extent and Composition – ice cover can protect coastal erosion.
- Hardened Shorelines – less ice cover exposes the shoreline to waves generated by winter storms that accelerates erosion.
- Dreissenid Mussels – ice cover can influence the water stability and temperature structure and warming duration,
- 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, as ice can protect the eggs from wind mixing.
- Lake white fish, lake herring, and yellow perch species are dependent on ice cover.

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			

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, therefore, this is a good indicator of climate effects. This data is 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 to ecosystem in the Great Lakes (Bai et al. 2015; Wang et al. 2012b, 2018).

Acknowledgments

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All data analyzed and charts created by the authors.

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List of Tables

Table 1. Regression linear trend of AMIC anomalies (using a 1973-1990 baseline), $x=a+bt$ in percentage (t in years starting with 0 in 1973, a is constant, and b is the trend/slope).
Source: Wang et al. (2017a)

Table 2. Statistic parameters (mean and standard deviation) of AMIC anomalies (using a 1973-1990 baseline) for Great Lakes using data from 1973-2018.
Source: Wang et al. (2017a)

List of Figures

Figure 1. Time series of five Great Lake AMIC for the period 1973-2018, 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 1973-2018, 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 1973-2018, 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 1973-2018, 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 1973-2018, 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 1973-2018, 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 1973-2018, 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.

Last Updated

State of the Great Lakes 2019 Technical Report

	a	b: 10-year trend (annual loss or gain %)	b: 30-year trend (annual loss or gain %)	b: Long-term trend (annual loss %)	Total loss up to 2018 (%)
Superior	6.61	0.27	-0.96	**-0.75***	-34.61
Michigan	1.13	1.02	0.25	-0.28	-12.85
Huron	1.82	0.88	-0.19	-0.41	-18.94
Erie	2.36	-1.34	-0.07	-0.43	-19.63
Ontario	2.02	1.54	0.18	-0.35	-15.97
Basin	2.77	0.80	-0.22	**-0.53***	-24.56

Table 1. Regression linear trend of AMIC anomalies (using a 1973-1990 baseline), $x=a+bt$ in percentage (t in years starting with 0 in 1973, a is constant, and b is the trend/slope). *Note that the long-term trends for Lake Superior and the Great Lakes basin are statistically significant shown in bold.
Source: Wang et al. (2017a)

From 1973 to 2018	Superior	Michigan	Huron	Erie	Ontario	Basin
Mean	-11.07	-5.44	-7.86	-7.66	-6.14	-9.79
standard deviation	30.12	21.54	22.87	25.88	19.71	23.25

Table 2. Statistic parameters (mean and standard deviation) of AMIC anomalies (using a 1973-1990 baseline) for Great Lakes using data from 1973-2018.
Source: Wang et al. (2017a)

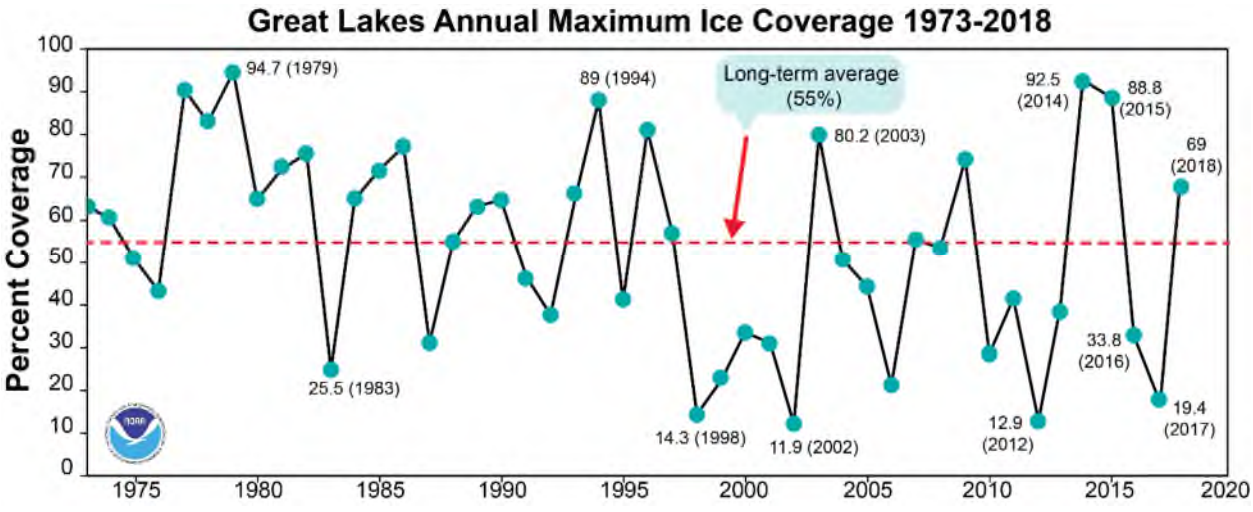


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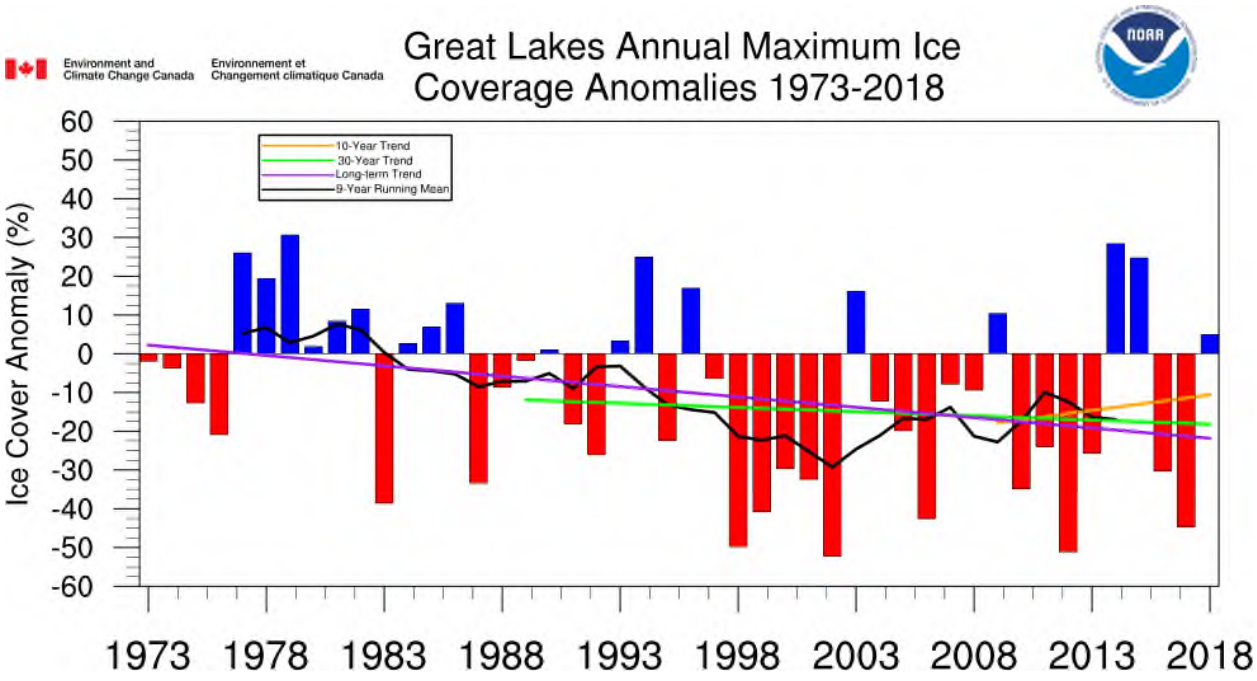
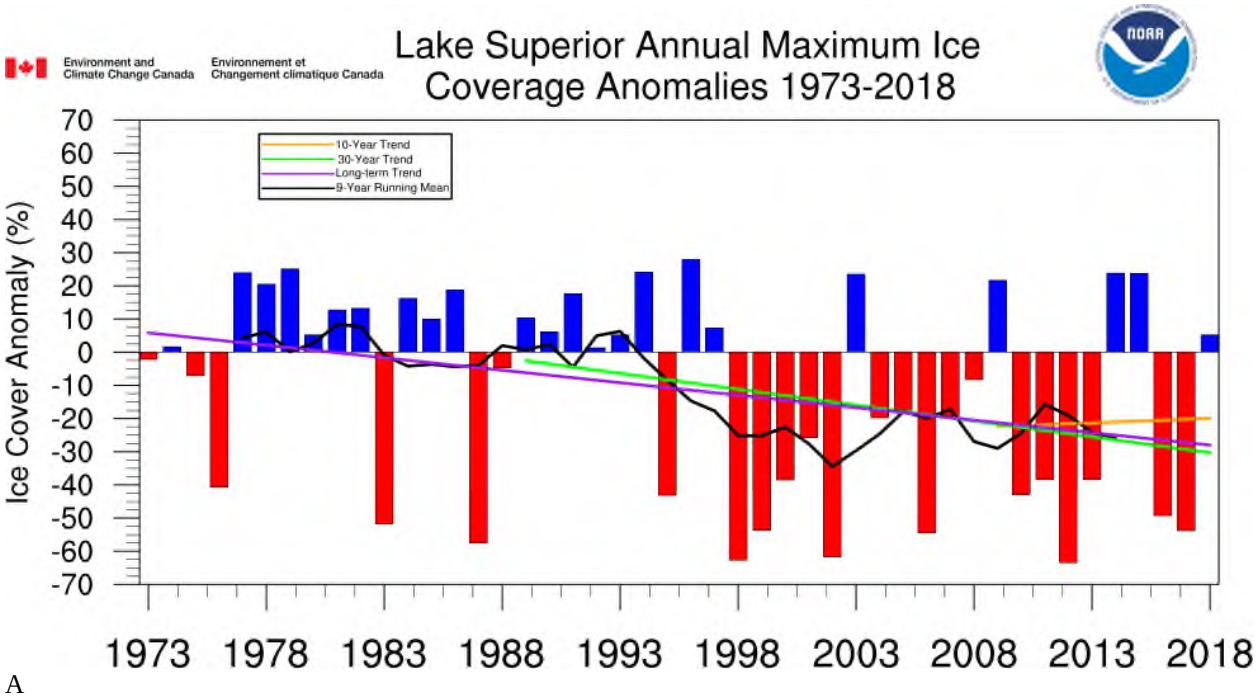


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Source: ECCC and NOAA



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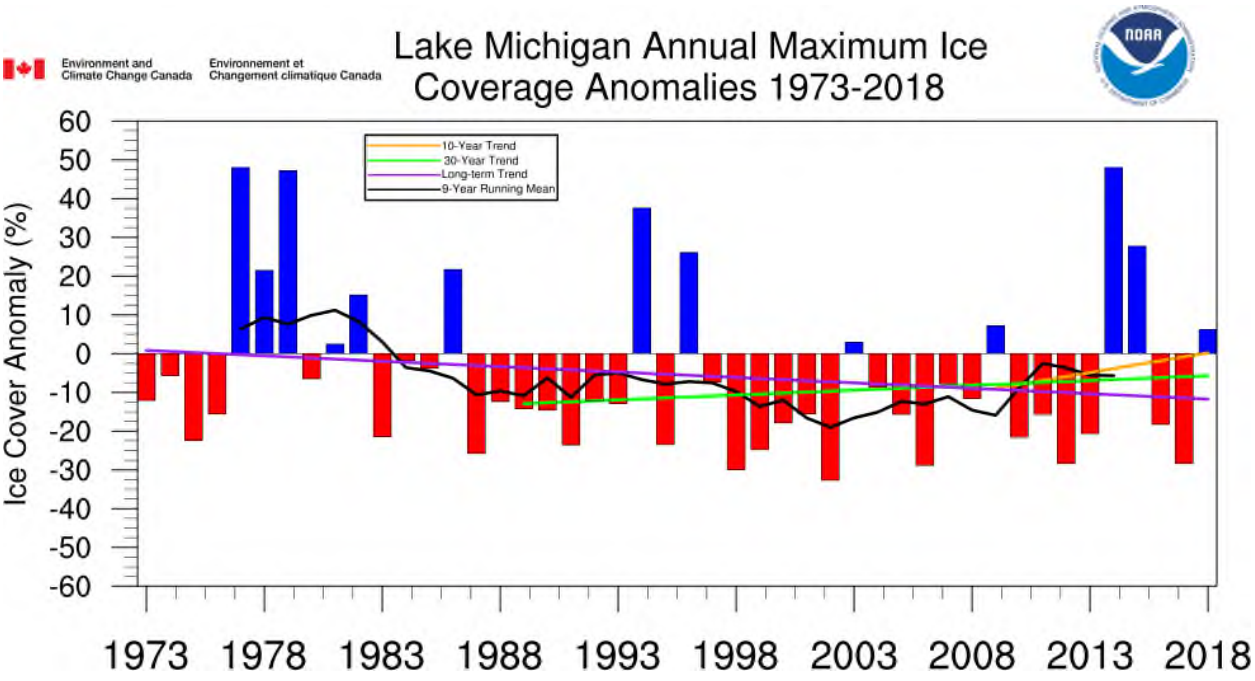


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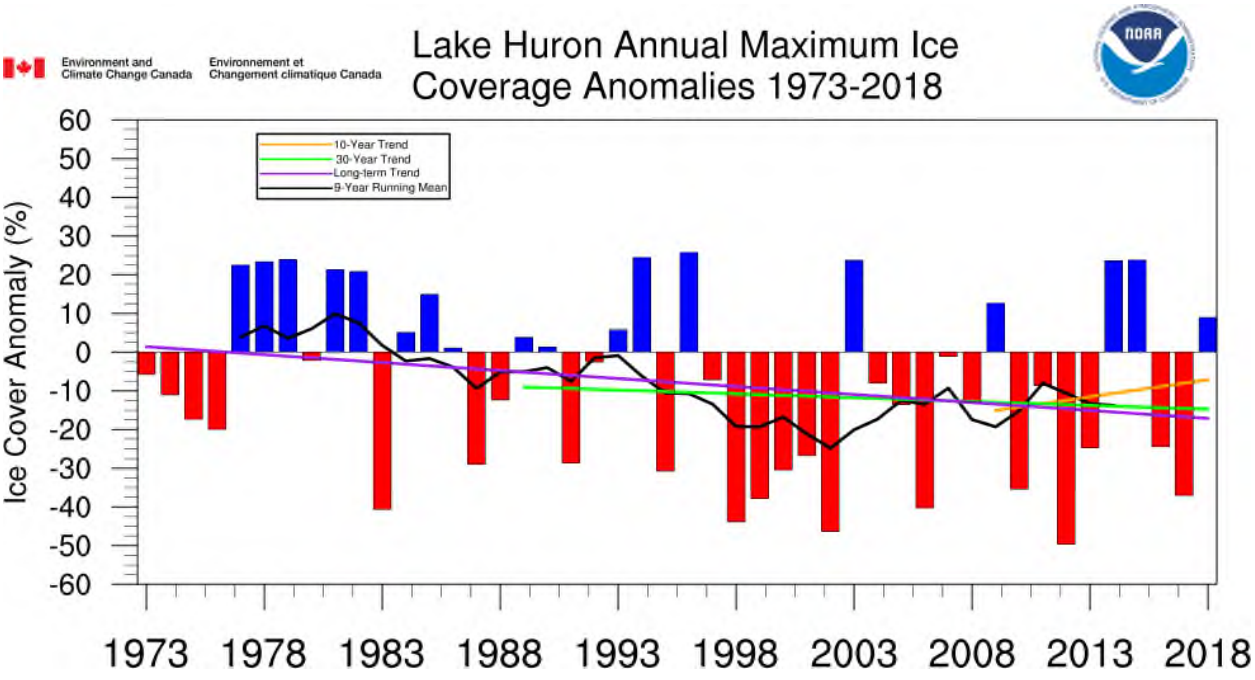


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Source: ECCC and NOAA

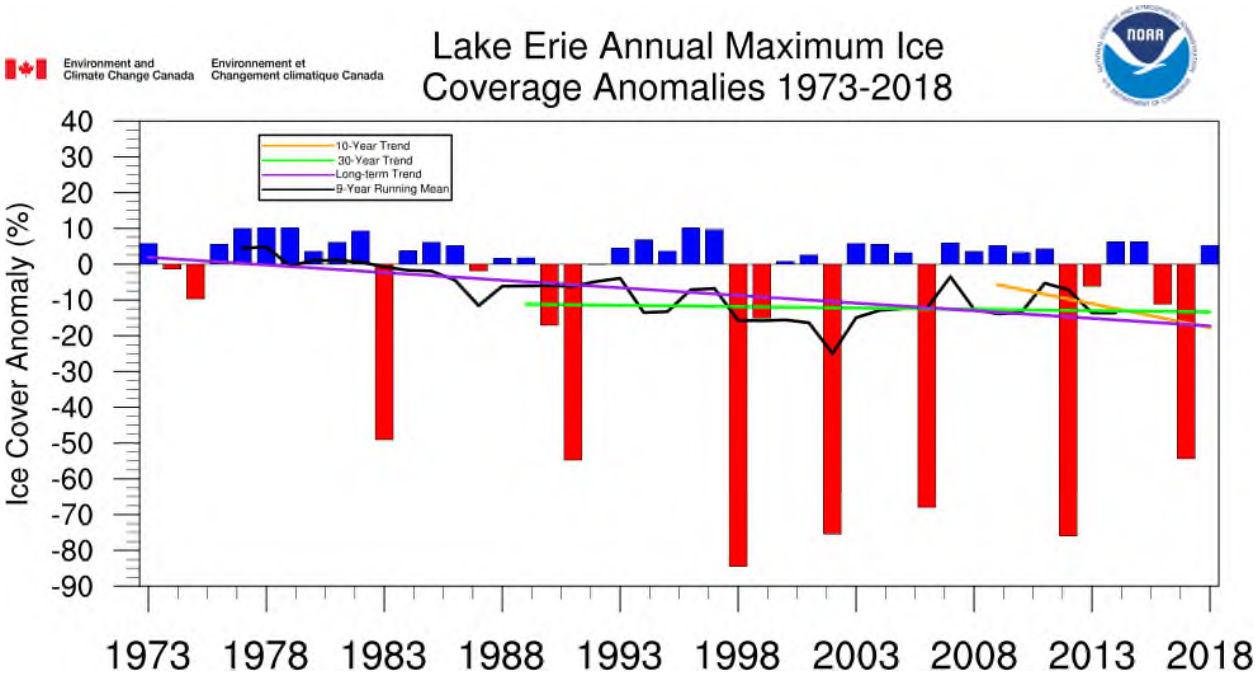


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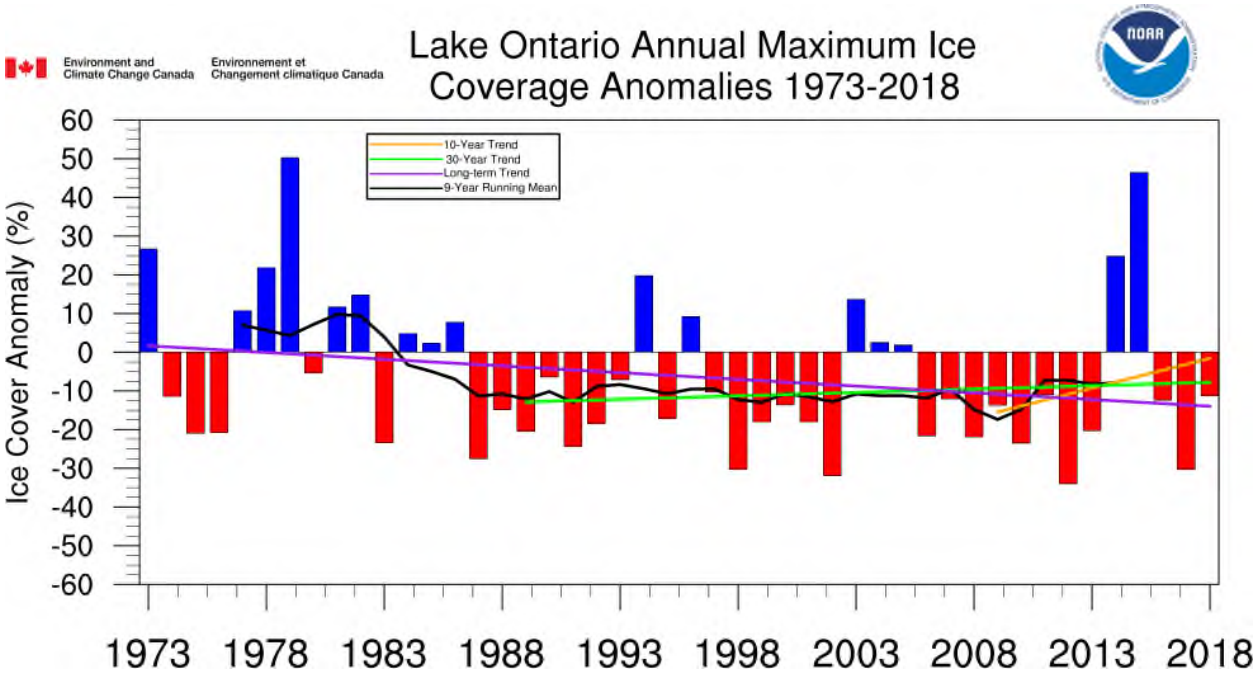


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