**UNITED STATES ENVIRONMENTAL PROTECTION AGENCY**
GREAT LAKES NATIONAL PROGRAM OFFICE
77 WEST JACKSON BOULEVARD
CHICAGO, IL 60604-3590

Sarah Strommen, Commissioner
Minnesota Department of Natural Resources
500 Lafayette Road N
St. Paul, Minnesota 55155

Katrina Kessler, Commissioner
Minnesota Pollution Control Agency
520 Lafayette Road N
St. Paul, Minnesota 55151-4194

Dear Commissioners Strommen and Kessler:

Thank you for your November 28, 2022 request to remove the *Degradation of Fish and Wildlife Populations* Beneficial Use Impairment (BUI) from the St. Louis River Area of Concern (AOC). As you know, we share Minnesota's desire to restore all the Great Lakes AOCs and to formally delist them.

Based upon a review of your submittal and the supporting data, the U.S. Environmental Protection Agency (EPA) approves Minnesota's request to remove this BUI from the St. Louis River AOC. EPA will notify the International Joint Commission (IJC) of this significant positive environmental change at this AOC.

We congratulate you and your staff as well as the many other federal, state and local partners who have been instrumental in achieving this environmental improvement. Removal of this BUI will benefit not only the people who live and work in the St. Louis River AOC, but all residents of Minnesota and Wisconsin and the Great Lakes basin as well.

We look forward to the continuation of this important and productive relationship with the Minnesota Department of Natural Resources, the Minnesota Pollution Control Agency, the Wisconsin Department of Natural Resources, and the St. Louis River Alliance as we work together to delist this AOC in the years to come. If you have any further questions, please contact me at (312) 353-8320 or your staff may contact Leah Medley at (312) 886-1307.

Sincerely,

Chris Korleski, Director
Great Lakes National Program Office

cc: Barbara Huberty, MPCA
Melissa Sjolund, MNDNR
Rick Gitar, Fond du Lac Band of Lake Superior Chippewa
Kendra Axness, WDNR
Matt Steiger, WNDR
Raj Bejankiwar, IJC

**UNITED STATES ENVIRONMENTAL PROTECTION AGENCY**
GREAT LAKES NATIONAL PROGRAM OFFICE
77 WEST JACKSON BOULEVARD
CHICAGO, IL 60604-3590

Rebecca Fedak, Acting Director
Office of Great Waters – Great Lakes & Mississippi River
Wisconsin Department of Natural Resources
P.O. Box 7921
Madison, WI 53707-7921

Dear Ms. Fedak,

Thank you for your November 28, 2022 request to remove the *Degradation of Fish and Wildlife Populations* Beneficial Use Impairment (BUI) from the St. Louis River Area of Concern (AOC). As you know, we share Wisconsin's desire to restore all the Great Lakes AOCs and to formally delist them.

Based upon a review of your submittal and the supporting data, the U.S. Environmental Protection Agency (EPA) approves Wisconsin's request to remove this BUI from the St. Louis River AOC. EPA will notify the International Joint Commission (IJC) of this significant positive environmental change at this AOC.

We congratulate you and your staff as well as the many other federal, state and local partners who have been instrumental in achieving this environmental improvement. Removal of this BUI will benefit not only the people who live and work in the St. Louis River AOC, but all residents of Wisconsin and Minnesota and the Great Lakes basin as well.

We look forward to the continuation of this important and productive relationship with your agency, the Minnesota Pollution Control Agency, the Minnesota Department of Natural Resources, and the St. Louis River Alliance as we work together to delist this AOC in the years to come. If you have any further questions, please contact me at (312) 353-8320 or your staff may contact Leah Medley at (312) 886-1307.

Sincerely,

Chris Korleski, Director
Great Lakes National Program Office

cc: Kendra Axness, WDNR
Cherie Hagan, WDNR
Barbara Huberty, MPCA
Matt Steiger, WNDR
Melissa Sjolund, MNDNR
Rick Gitar, Fond du Lac Band of Lake Superior Chippewa
Raj Bejankiwar, IJC

November 7, 2022

Chris Korleski
Director, Great Lakes National Program Office
United States Environmental Protection Agency
77 West Jackson Boulevard
Chicago, IL 60604-3507

Dear Director Korleski,

The Minnesota Department of Natural Resources (MNDNR), Minnesota Pollution Control Agency (MPCA), and Wisconsin Department of Natural Resources (WDNR) hereby request the approval of the United States Environmental Protection Agency's Great Lakes National Program Office (GLNPO) staff to remove the Degraded Fish and Wildlife Populations Beneficial Use Impairment (BUI) in the St. Louis River Area of Concern (SLRAOC). The SLRAOC team has assessed the status of the management actions and removal target for the Degraded Fish and Wildlife Populations BUI as outlined in the 2013 SLRAOC Remedial Action plan and its subsequent annual updates. Management actions included the following:

- Bird inventory and assessment
- Fish population monitoring and assessment
- Eurasian Ruffe assessment
- Semi-aquatic mammal survey
- Wisconsin Point habitat restoration for Piping Plover
- Interstate Island habitat restoration for Common Tern and Piping Plover

All management actions associated with this impairment have been completed and the removal target has been met. A public review of the removal package and recommendation to remove this BUI has been conducted. Documentation about the public review process, copies of all comment letters and responses from SLRAOC Coordinators are included with the removal package as Appendix I.

Of the letters that were critical of the removal, thirteen were form letters submitted by University of Minnesota Duluth students. Critical comments centered around the following themes:

- Commenter believes that removal is premature and should not be pursued until all remedial or restoration actions associated with the Restrictions on Dredging and Loss of Fish and Wildlife Habitat BUIs are completed;

- Commenter requests additional study and management of the Common Tern colony on Interstate Island and believes that regional recovery goals for the species must be achieved before removing the BUI; and
- Commenter is concerned that fish and wildlife in the SLRAOC may still face limitations caused by factors that are being addressed by other BUIs or factors outside of the AOC program's scope and believes those factors must be addressed prior to BUI removal (e.g., mercury contamination originating from a variety of sources).

While comment letters resulted in some minor edits to the removal package, the SLRAOC Coordinators team has reviewed all comments at length and reached consensus that the objectives and target required to remove this BUI have been addressed. SLRAOC leaders from each of these agencies also support its removal at this time. Therefore, enclosed please find the document that supports the removal of the Degraded Fish and Wildlife Populations BUI prepared by MNDNR and WDNR staff.

We value our continuing partnership with the GLNPO staff, and the funding support provided to the SLRAOC through the Great Lakes Restoration Initiative. It is your significant involvement and our strong partnerships with all federal, state, and local partners that will keep us progressing towards delisting the SLRAOC.

If you need further information about the Minnesota aspects of this removal request please contact Melissa Sjolund, MNDNR AOC Coordinator at 218-302-3245 or melissa.sjolund@state.mn.us.

Sincerely,

Sarah Strommen
Commissioner, MNDNR

Katrina Kessler
Commissioner, MPCA

Enclosure: St. Louis River Area of Concern Beneficial Use Impairment Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment

cc: Leah Medley, SLRAOC Task Force Lead
Amy Roe, USFWS Technical Resource Lead
Kendra Axness, WDNR LAMP and AOC Coordinator
Matt Steiger, WDNR AOC Coordinator
Barb Huberty, MPCA AOC Coordinator
Melissa Sjolund, MNDNR AOC Coordinator
Rick Gitar, Fond du Lac AOC Coordinator

**State of Wisconsin**
**DEPARTMENT OF NATURAL RESOURCES**
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**Madison WI 53707-7921**

**Tony Evers, Governor**
**Preston D. Cole, Secretary**
Telephone 608-266-2621
Toll Free 1-888-936-7463
TTY Access via relay - 711

November 28, 2022

Mr. Chris Korleski
Director, Great Lakes National Program Office
United States Environmental Protection Agency
77 West Jackson Boulevard
Chicago, IL 60604-3507

Dear Director Korleski,

The Wisconsin Department of Natural Resources (WDNR) along with managing agencies Minnesota Department of Natural Resources (MNDNR) and Minnesota Pollution Control Agency (MPCA) requests the concurrence of the U.S. Environmental Protection Agency (USEPA) Great Lakes National Program Office (GLNPO) with the removal of the Degraded Fish and Wildlife Populations Beneficial Use Impairment (BUI) in the St. Louis River Area of Concern (SLRAOC). The SLRAOC team has assessed the status of the management actions and removal target for the Degraded Fish and Wildlife Populations BUI as outlined in the 2013 SLRAOC Remedial Action plan and its subsequent annual updates. The six management actions that were identified for this BUI by the managing agencies in collaboration with local stakeholders are as follows:

- Bird inventory and assessment;
- Fish population monitoring and assessment;
- Eurasian Ruffe assessment;
- Semi-aquatic mammal survey;
- Wisconsin Point habitat restoration for Piping Plover; and,
- Interstate Island habitat restoration for Common Tern and Piping Plover.

All management actions associated with this impairment have been completed and the removal target has been met. The enclosed removal recommendation document provides the supporting information. We held a public comment period for the removal recommendation document from March 28 through April 26, 2022. Information about the public review process, including copies of all comment letters and responses from SLRAOC Coordinators, is included with the removal package as Appendix I. We received comments opposing the removal which we have addressed in Appendix I. After consideration of evidence in support of removal as well as the opposing perspectives, the St. Louis River Alliance (the community advisory committee for the SLR AOC) has voted to provide a letter expressing support for removal (attached in Appendix I).

Studies showing recovering fish and wildlife populations, together with completion of important habitat restoration projects at Wisconsin Point and Interstate Island, demonstrate significant progress since the time of AOC designation. In recognition of this progress and CAC support, and with consensus among the SLRAOC Coordinators team and managing agency leaders that AOC specific BUI removal objectives and targets have been met, we are recommending removal.

While we celebrate the accomplishments made through the AOC program, we recognize that more work is needed beyond the scale of the AOC to realize ecosystem restoration goals. Efforts will continue through the Lake Superior Lakewide Action and Management Plan (LAMP), state fish and wildlife resource management plans (e.g., the Wisconsin Wildlife Action Plan), and other state, federal and local initiatives. These important collective efforts will be in place long term with support from many partners.

We value our continuing partnership with GLNPO and the funding support provided to the SLRAOC through the Great Lakes Restoration Initiative. It is the strong partnerships among all federal, state, and local partners that will keep us progressing towards delisting the SLRAOC.

If you need further information about the Wisconsin aspects of this removal request please contact Matt Steiger, Wisconsin DNR AOC Coordinator at 715-395-6904 or matthew.steiger@wisconsin.gov, Cherie Hagen, Lake Superior Team Supervisor at 715-635-4034 or Cherie.Hagen@wisconsin.gov, or you may contact me.

Sincerely,

Stephen Galarneau, Director
Office of Great Waters – Great Lakes and Mississippi River
Wisconsin Department of Natural Resources
608-266-1956
Stephen.Galarneau@Wisconsin.gov

Enclosure: St. Louis River Area of Concern Beneficial Use Impairment Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment

CC: Todd Nettesheim, GLNPO
Marc Tuchman, GLNPO
Amy Pelka, GLNPO
Leah Medley, GLNPO SLRAOC Task Force Lead
Amy Roe, USFWS Technical Resource Lead
Matt Steiger, WDNR AOC Coordinator
Barb Huberty, MPCA AOC Coordinator
Melissa Sjolund, MNDNR AOC Coordinator
Rick Gitar, Fond du Lac AOC Coordinator
Kendra Axness, WDNR LAMP and AOC Coordinator
Cherie Hagen, WDNR Lake Superior Team Supervisor

**St. Louis River Area of Concern**

## Beneficial Use Impairment Removal Recommendation for

# *Degraded Fish and Wildlife Populations*

**November 28, 2022**

**Submitted to:**

**U.S. EPA-Region 5**

**77 W. Jackson Boulevard**

**Chicago, IL 60604**

Prepared by these implementing agencies:

With major funding support from the Great Lakes Restoration Initiative.

EGAD Number: 3900-2023-03

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**Appendix G**: Piping Plover Habitat Restoration Project Summary

**Appendix H**: Interstate Island Habitat Restoration Project Summary

**Appendix I**: Public Involvement Process

# List of Acronyms

- AOC – Area of Concern
- BUI – beneficial use impairment
- CPUE – Catch per unit effort
- CWA – Clean Water Act
- DLC – Dioxin-like chemical
- FdL – Fond du Lac Band of Lake Superior Chippewa
- MA – management action
- MLT – Minnesota Land Trust
- MNDNR – Minnesota Department of Natural Resources
- MPCA – Minnesota Pollution Control Agency
- NRRI – Natural Resources Research Institute
- PIT – Passive Integrated Technology
- PSD – Proportional stock density
- AOC – Remediation to Restoration
- RAP – Remedial Action Plan
- RST – Restoration Site Team
- SEH – Short, Elliot, and Hendrickson
- SLRA – St. Louis River Alliance
- SLRAOC – St. Louis River Area of Concern
- SLRE – St. Louis River Estuary
- SR – Species Richness
- USACE – United States Army Corps of Engineers
- USFWS – United States Fish and Wildlife Service
- WDNR – Wisconsin Department of Natural Resources
- WLSSD – Western Lake Superior Sanitary District

# Executive Summary

## Background

The United States and Canada designated 43 Areas of Concern (AOC) across the Great Lakes in 1987, including the St. Louis River Area of Concern (SLRAOC). The AOCs were designated because significant environmental damage at those locations caused specific types of Beneficial Use Impairments (BUIs). At the time of AOC designation, the International Joint Commission identified 14 BUIs in the Great Lakes Water Quality Agreement that were to be assessed at each AOC to determine their applicability. Only nine BUIs applied to the SLRAOC. Once the BUIs were identified, removal targets for each were established and management actions (MAs) to achieve the targets for each BUI were identified.

Once the MAs for a BUI are completed and removal targets are met, a removal package is prepared for public review and, ultimately, concurrence by the U.S. Environmental Protection Agency.

This document provides the justifications supporting a removal recommendation for the Degraded Fish and Wildlife Populations BUI (BUI 2) for the SLRAOC. All six MAs that apply to BUI 2 are complete and the BUI 2 removal target has been met. The removal criteria and brief conclusions pertaining to the studies applicable to each are included in this executive summary. Detailed summaries of the studies and findings for each MA are included in the main body of this document, while the study reports prepared for each management action are included in the appendices.

The non-regulatory AOC program was established to address “legacy” issues. These were environmental problems that caused ecosystem impairments at the time of the AOC designation and largely occurred before modern environmental regulations were in place. Legacy issues significantly impacted geographically-defined sites rather than regional-scale stressors.

For the SLRAOC, examples of legacy issues are: unregulated discharge of industrial and municipal waste, dredging and filling in the estuary, wood waste deposited in the river, and extensive logging which exacerbated erosion and sedimentation problems. Since the time of these legacy impacts, the Clean Water Act (CWA) and other environmental regulations were enacted to protect the environment and human health from these types of large-scale problems.

The scope of the AOC program does not include “modern” issues that are the responsibility of variety of state, tribal, and federal agencies under existing natural resources, environmental, and public health program authorities. Some examples of modern issues are: contaminants of emerging concern, water-related climate change impacts, non-compliance with point source permits, and impairments identified and regulated under the CWA.

## The Removal Target Has Been Met

The removal target will have been met when:

> *In consultation with their federal, tribal, local, and nonprofit partners, state resource management agencies concur that diverse native fish and wildlife populations are not limited by physical habitat, food sources, water quality, or contaminated sediments* (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2008).

Removal of BUI 2 will be justified when resource management agencies (Minnesota Department of Natural Resources, Minnesota Pollution Control Agency, and Wisconsin Department of Natural Resources) concur that key native species populations of fish and wildlife are present and not limited by the legacy impairments referenced by the removal target. This is demonstrated by addressing the specific removal objectives for the fish and wildlife species listed below. Removal objectives were established to help identify key species, establish specific management actions, and guide AOC managers toward achieving the removal target. All removal objectives have been met except the Lake Sturgeon objective. Species monitoring has shown the Lake Sturgeon-specific removal objective metric has not yet been met, however resource managers have completed additional studies to justify BUI removal as described in the summary of Management Action 2.02 (p.25) and detailed in **Appendix D**.

Successfully completing management actions triggers a review to verify the removal objectives and target have been achieved. Removal is justified because the objectives and management actions identified as priorities for BUI 2 were successfully addressed. Removal is recommended while acknowledging that St. Louis River fish and wildlife populations may continue to face limitations caused by physical habitat, food sources, water quality, or contaminated sediments that are outside the scope of this BUI . Limitations are at levels that cannot be addressed by additional management actions under BUI 2. Continued efforts to manage habitat, remediate contaminated sediments and restore habitat under other BUIs as well as actions outside of the AOC program, will further benefit native fish and wildlife populations in the estuary.

### Fish

The following BUI 2 objectives were developed for target native and invasive fish species:

- Walleye gillnet catch per unit effort (CPUE) is maintained at or above 5.0 per lift with a proportional stock density between 30 and 60 in at least 50% of years surveyed since 2000.
- Muskellunge trap net CPUE is maintained at or above 1.0 per lift in at least 50% of years surveyed since 1997.
- Document an increasing trend of two to five year old Lake Sturgeon captured in summer index nets, with at least two index values greater than 2.0 per gillnet lift.
- An analysis of historical data that shows the Ruffe, an invasive species, is not inhibiting the native fish population.

CONCLUSION: Populations of Walleye and Muskellunge in the St. Louis River are meeting or exceeding the established objectives and invasive Ruffe are no longer inhibiting native fish populations. While Lake Sturgeon populations are not trending towards recovery goals, research indicates they are not accumulating legacy contaminants at levels that impact reproduction and are likely limited by factors outside of the AOC program's Degraded Populations focus.

### Wildlife

The following BUI 2 removal objectives were established for target wildlife species:

- Piping Plover nesting habitat is created within the SLRAOC.
- Common Tern nesting habitat at Interstate Island is restored and state agencies continue to support habitat management and population monitoring there.
- Great Blue Heron and Bald Eagle presence is recorded during one or more nesting seasons since 1997.
- Wetland bird species are surveyed and compared with 1979 survey results.
- A survey of semi-aquatic mammals in the estuary verifies that the status of small mammal species in the St Louis River Area of Concern is sufficient to remove the beneficial use impairment.

CONCLUSION: Removal objectives for wildlife were achieved. Species including Great Blue Heron and Bald Eagle met numeric targets, while wetland bird species at Remediation to Restoration sites were surveyed and found to have greater abundance and similar species richness when compared to reference sites and similar species richness when compared to historical surveys. Four species of semi aquatic mammals were surveyed and found to be similar to reference populations. Habitat restoration projects targeting Piping Plover and Common Tern were implemented to address legacy habitat loss, which includes long-term monitoring and maintenance plans. Based on post-restoration monitoring at Interstate Island, additional measures may be required outside of the AOC program to conserve the estuary's Common tern breeding colony.

**Developing the Removal Package**

Multiple BUI Technical Teams of subject matter experts were established to evaluate the removal strategy and review the findings from each study and offer recommendations to address any deficiencies until the target and criteria were met.

A public information process was conducted to obtain input from interested parties on the information provided in the removal package.

The BUI 2 Removal Target requires concurrence from resource managers. This is accomplished by reviewing the final recommendation with the AOC Leadership Team, comprised of lead supervisors from the Fond du Lac Band of Lake Superior Chippewa and three state agencies (Minnesota Department of Natural Resources, Minnesota Pollution Control Agency, and Wisconsin Department of Natural Resources.) Upon gaining concurrence from the AOC Leadership Team, the final recommendation is shared with the Interagency Manager's Team. Managers from the three state agencies comprise this team and provide final concurrence and authorization to submit the final BUI removal package to EPA.

Multiple lines of evidence support a removal recommendation for this BUI. The results of the BUI 2 studies, successful implementation of required restoration projects, along with support from the BUI 2 Technical Teams, SLRAOC partners, and stakeholders have resulted in this recommendation by the SLRAOC Coordinators, leaders, and executive managers to remove the Degraded Fish and Wildlife Populations BUI from the SLRAOC.

# Purpose

The purpose of this document is to provide the information needed to support a recommendation to remove the Degraded Fish and Wildlife Populations Beneficial Use Impairment (BUI) in the St. Louis River Area of Concern (SLRAOC).

# St. Louis River Area of Concern Background

The 1987 US-Canada Great Lakes Water Quality Agreement designated the SLRAOC as one of 43 areas with significant environmental degradation. The SLRAOC is spatially large and geographically complex, spanning the Minnesota and Wisconsin state line and including tribal interests (**Error! Reference source not found.**).

The SLRAOC program is jointly managed by four implementing agencies: the Fond du Lac Band of Lake Superior Chippewa (FdL), the Minnesota Department of Natural Resources (MNDNR), the Minnesota Pollution Control Agency (MPCA), and the Wisconsin Department of Natural Resources (WDNR). MPCA and WDNR are the delegated authorities that manage official transactions with the U.S. Environmental Protection Agency - Great Lakes National Program Office. Dozens of stakeholder organizations are also involved in activities related to the SLRAOC.

Efforts to remove the BUIs are located primarily within the 12,000-acre St. Louis River Estuary (SLRE), where water from the St. Louis River and Lake Superior mix. The twin port cities of Duluth, MN and Superior, WI are located on either side of the estuary.

A Stage I Remedial Action Plan (RAP) identified these nine BUIs (MPCA and WDNR, 1992):

1. Restrictions on Fish and Wildlife Consumption
2. Degradation of Fish and Wildlife Populations
3. Fish Tumors or Other Deformities; removed in 2017
4. Degradation of Benthos
5. Restrictions on Dredging Activities
6. Eutrophication or Undesirable Algae (SLRAOC name: Excessive Loading of Sediment and Nutrients); removed in 2020
7. Beach Closings (SLRAOC name: Beach Closing and Body Contact Restrictions)
8. Degradation of Aesthetics; removed in 2014
9. Loss of Fish and Wildlife Habitat

A Stage II RAP was completed in 1995 and it was later superseded by the 2013 St. Louis River Area of Concern Implementation Framework: Roadmap to Delisting (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1995). The 2013 RAP (Roadmap) was a comprehensive listing of the BUIs, their removal targets, and the management actions (MAs) needed to achieve those targets (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013). The 2013 RAP has been updated annually thereafter to document progress and changes to the RAP implementation plan and schedule.

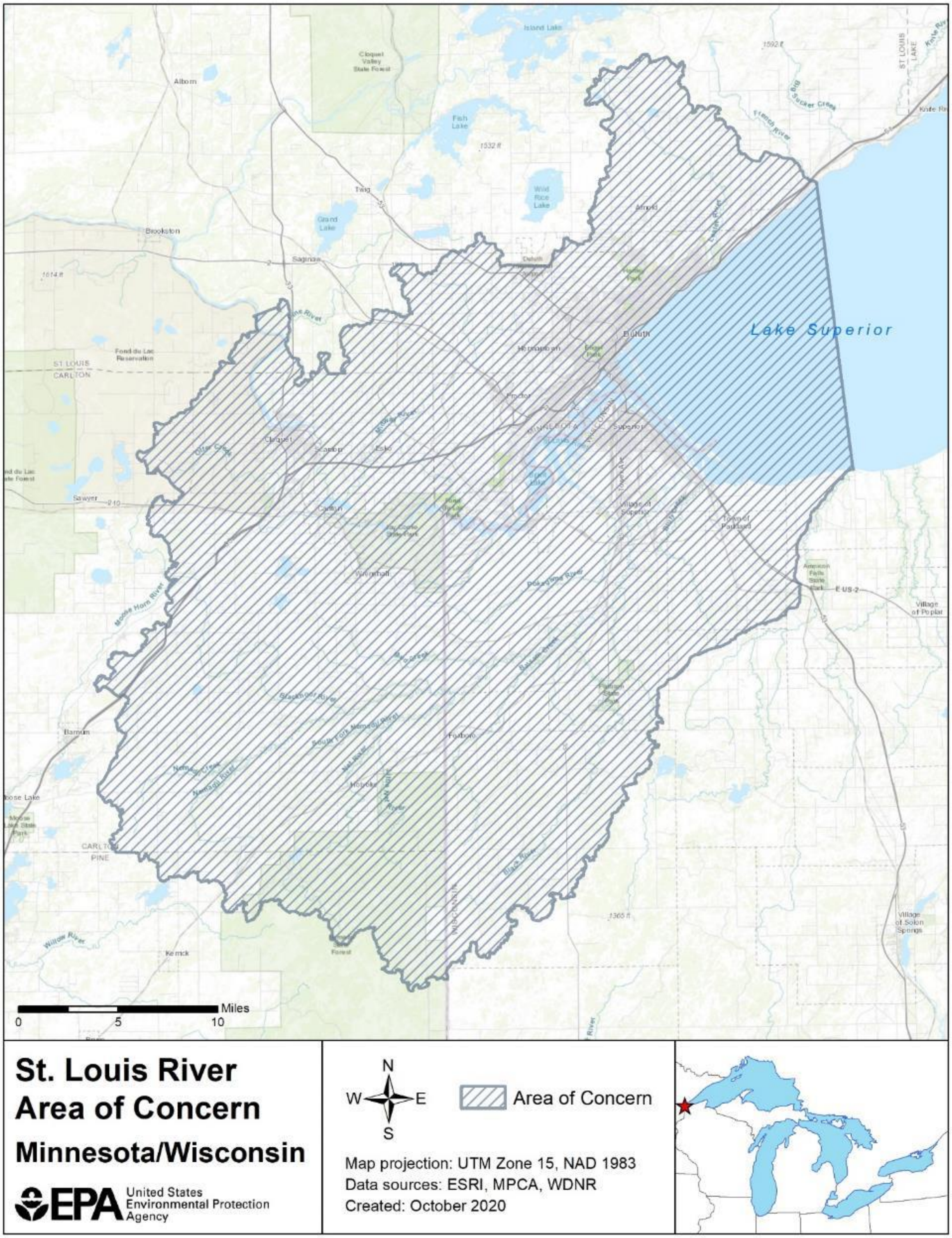


FIGURE 1. EXTENT OF THE ST. LOUIS RIVER AREA OF CONCERN

It is important to understand that the non-regulatory AOC program was created to address "legacy" issues or environmental problems that caused ecosystem impairments at the time of the AOC designation and largely occurred before modern environmental regulations were in place. Legacy issues addressed by the AOC program include stressors which significantly impacted specific, geographically-defined sites as opposed to regional-scale stressors.

For the SLRAOC, examples of legacy issues are unregulated discharge of industrial and municipal waste, dredging and filling in the estuary, wood waste deposited in the river, and extensive logging which exacerbated erosion and sedimentation problems. Since the time of these legacy impacts, the Clean Water Act (CWA) and other environmental regulations were enacted to protect the environment and human health from these types of large-scale problems.

The scope of the AOC program does not include "modern" issues that are the responsibility of a variety of state, tribal, and federal agencies under existing natural resources, environmental, and public health program authorities. Some examples of modern issues are: contaminants of emerging concern, climate change impacts, non-compliance with point source permits, and impairments identified and regulated under the CWA. **Error! Reference source not found.** depicts the differences between the AOC and existing agency programs.

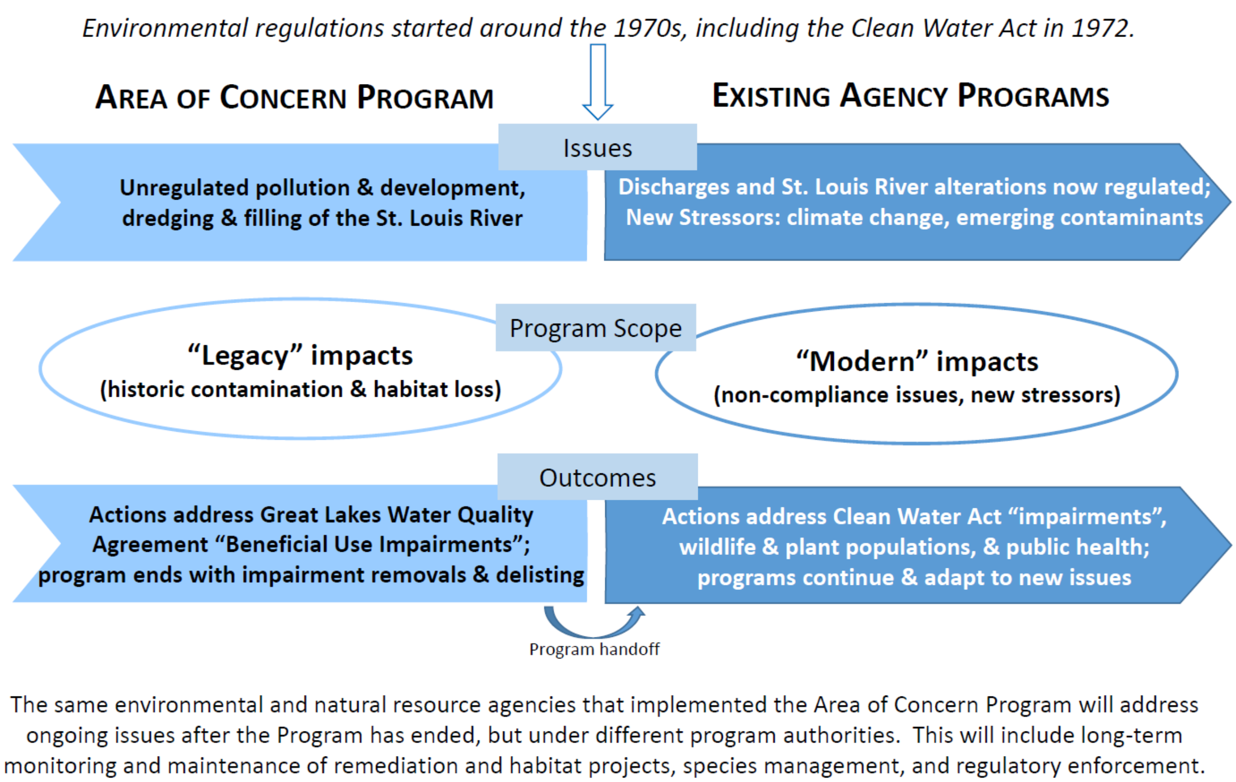


FIGURE 2. "LEGACY VS MODERN"-THE PROGRAM SCOPE OF THE ST. LOUIS RIVER AREA OF CONCERN

Sustaining healthy populations of fish and wildlife in the St. Louis River requires actions to address both legacy and modern impacts. As it relates to the removal of the Degraded Fish and Wildlife Populations BUI discussed in this report, consider the successful nesting of Piping Plover, a federally listed bird species. Plover nest exclusively on unvegetated shoreline, a habitat type lost in the SLRE due to legacy development for industry and navigation. The SLRAOC program can be used to address this legacy impact by creating, restoring, and maintaining Piping Plover habitat. Other factors that influence successful Piping Plover nesting include disturbance by humans and dogs, predation, and regional population trends. These are modern impacts that must be addressed by public land managers and through community outreach and monitoring programs but is not the responsibility of the SLRAOC program. The Future Actions section of this document lists a variety of future needs to be addressed by other agency programs.

# BUI Information and Background

## Rationale for Listing (1992)

An impairment will be listed when fish and wildlife management programs have identified degraded fish or wildlife populations due to a cause within the watershed. In addition, this use will be considered impaired when relevant, field-validated, fish or wildlife bioassays with appropriate quality assurance/quality controls confirm significant toxicity from water column or sediment contaminants. SLRAOC partners worked together to develop the following rationale for the 1992 Stage 1 RAP (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1992):

> *During the period of severe organic pollution before 1979, fish populations were degraded and fish kills were common. Fish populations have been recovering from that era because of improvements in wastewater treatment. However, fish populations are now adversely affected by the proliferation of the Ruffe, an exotic species first found in the AOC in 1987. Other exotics threaten fish populations. The potential effects of toxic substances on fish population health in the AOC is largely unknown. Continuing loss of physical habitat also threatens populations. The loss of wetland habitat and the infestation of the exotic plant, purple loosestrife, have the potential to cause declining fish and wildlife populations. Little population data are available for wildlife with the exception of colonial nesting birds in the AOC. Populations of the Common Tern and the Piping Plover (threatened and endangered species) have declined, probably due to a combination of local and regional factors.*

Since 1979, fish populations have been recovering because of improved water quality that resulted from more complete wastewater treatment after formation of the Western Lake Superior Sanitary District (WLSSD), construction of the WLSSD wastewater treatment plant, and improvement of City of Superior wastewater treatment. However, at the time of listing, fish populations remained adversely affected by alterations and loss of habitat, proliferation of exotic species, and possibly by exposure to toxic substances.

At the time of listing, little population data were available for wildlife except for colonial nesting birds, herons, and gulls. Populations of the Common Tern and the Piping Plover (threatened and endangered species) and Great Blue Heron had declined. Gulls and Mallards had experienced die-offs in the recent past. These problems were attributed to alteration or loss of habitat and possibly toxic contamination.

## Early BUI Recommendations (1995)

Following the 1992 Stage I RAP, discipline-specific working groups were formed to systematically evaluate BUIs and develop recommendations for activities to address the impairments. The work groups followed a protocol, generating a list of 43 approved recommendations, which were presented in a 1995 RAP Progress Report

(Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1995). Specifically, the Habitat Work Group generated the following recommendations related to fish, wildlife, and invasive species.

## Fish

The 1995 RAP Progress Report identified three recommendations related to fish (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1995). One ("Ruffe") is addressed in the BUI Removal objectives for fish in the 2013 RAP (see Section 4.3.1). Two others "Fish Stranding" and "Dam Relicensing" were addressed prior to 2013 through changed requirements to operation of the Fond du Lac Dam as part of the license issued in 1995 (United States Federal Energy Regulatory Commission 1995) by the United States Federal Energy Regulatory Commission (FERC) and by the updated five-year operating plans submitted by Allete Inc., the operator of the St. Louis River Hydroelectric Project (ALLETE, Inc. d.b.a. Minnesota Power 2018).

Walleye, including a major portion of western Lake Superior population, spawn in the St. Louis River below the Fond du Lac Dam (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1992). Restricting the flow of water through the Fond du Lac Dam affected this spawning area and caused incidents of stranding of eggs and spawning adults. Through the 1995 license renewal process and the development of regular, updated operating plans, dam operations, equipment, and licenses have been modified to ensure more flow volume and more gradual rates of change (i.e., ramping rates) to flows through bypassed reaches of the St. Louis River. Article 407 of the current FERC license for the St. Louis River Project requires release of specified minimum flow rates, measured in cubic feet per second, from the Fond du Lac Dam for the protection and enhancement of fish and wildlife resources and riparian vegetation in the Fond du Lac bypassed reach (United States Federal Energy Regulatory Commission 1995). Ramping rates for walleye are required under normal operating conditions and applicable to both increasing and decreasing flows. Appropriate ramping rates were incorporated into required five-year operating plans (ALLETE, Inc. d.b.a. Minnesota Power 2018).

## Wildlife

The 1995 RAP Progress Report identified five recommendations related to wildlife (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1995). Four of these recommendations (Herons, Raptors, Piping Plovers, and Common Terns) are addressed by the removal targets established in the 2013 RAP. One, "Water Birds," was addressed prior to 2013 through an investigation conducted by MPCA on contaminant uptake by mallards at Stryker Bay and the Confined Disposal Facility (CDF) at Erie Pier (Ensor, Pitt and Helwig 1993). The results indicated that waterfowl exposed to dredged sediments at the CDF accumulated polychlorinated biphenyls at levels higher than control, but below Food and Drug Administration's tolerances. Since the 1993 report, contaminated sediments in Stryker Bay have been remediated and contaminated sediments in the Ponds behind Erie Pier will be cleaned up as a management action associated with the Restrictions on Dredging BUI. The Erie Pier CDF was overhauled as a processing and sorting facility in 2010 and now operates under a Management Plan that requires materials testing and regulation (Duluth Seaway Port Authority and Duluth-Superior Metropolitan Interstate Council 2021).

## Invasive Species

Three recommendations from the 1995 RAP Progress Report addressed Aquatic Invasive Species (AIS): "Ballast Water," "Exotics Transport," and "Exotic Mussels (Zebra) Importation." Significant actions were taken outside the AOC program prior to the 2013 RAP update and in the intervening years. These include development of ballast water management protocols, international conventions on ballast water management, USCG discharge standards as well as state regulations to prevent introduction and spread of AIS (Great Lakes Commission 2016). Additionally, the 2012 renegotiated Great Lakes Water Quality Agreement, between the U.S. and Canada included ballast water management commitments to prevent the introduction and spread of AIS (Canada and the United

States of America 2012). The States of Minnesota and Wisconsin have, since 1995, established statewide AIS programs to manage both aquatic and terrestrial invasive species of concern. Together these State, Federal and international efforts have addressed the recommendations from the 1995 RAP Progress report and developed responses to emerging AIS threats.

## Removal Target and Objectives (2008-13)

In 2008, the states of Minnesota and Wisconsin submitted a list of SLRAOC BUI removal targets to the USEPA (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2008). These targets were developed by the two states with input from local stakeholders. They established specific goals, objectives, and indicators to track progress and determine when BUIs can be removed. The removal target for BUI 2 is as follows:

> *"In consultation with their federal, tribal, local, and nonprofit partners, state resource management agencies concur that diverse native fish and wildlife populations are not limited by physical habitat, food sources, water quality, or contaminated sediments."*

The removal target describes a process that leads to delisting through concurrence between State resource management agencies (MNDNR, MPCA, and WDNR) in consultation with other AOC partners. Removal of BUI 2 will be justified when it is shown that key native species populations of fish and wildlife are present and not limited by the legacy impairments referenced by the removal target.

As written, the 2008 removal target does not explicitly state that limitations addressed by the AOC program must be from legacy sources (Figure 2). The AOC Program does not address modern limitations to fish and wildlife populations or limitations of any origin that exist outside of the SLRAOC boundary. The primary legacy sources identified in the SLRAOC are contaminated sediments and degraded habitats. There is significant overlap with other SLRAOC BUIs that are addressing these legacy impairments. With the scope of the AOC program in mind and an understanding of the significant role played by other BUIs, a team was assembled to translate this general target into "removal objectives" and corresponding "management actions" that provide a pathway to removing BUI 2.

Removal objectives are detailed activities or outcomes that are needed to meet the goals established by the removal target. During a multi-year effort, culminating in the completion of the 2013 Roadmap to Delisting, a team of resource managers developed a "Blueprint" for BUI 2. Using information contained in the Lower St. Louis River Habitat Plan as a starting point, the Blueprint Team created a document that evaluated the BUI 2 removal target and guided the selection of removal objectives (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013).

The Blueprint Team acknowledged that there was little specific information in the 1992 Stage 1 RAP that definitively linked fish and wildlife population information to particular causes of decline or recovery. To address this information gap, the team developed a source-stressor model for SLRAOC fish and wildlife populations. The Blueprint Team used existing knowledge and expertise to define key sources and stressors affecting SLRAOC fish and wildlife populations.

The Blueprint Team stated that "impacts to the conservation targets by sources (threats) and resulting stressors have been reduced since the passing of the Clean Water Act and the establishment of the Western Lake Superior Sanitary District in 1979." There was a consensus that significant gains had resulted from improvements to wastewater treatment, agency-supported species rehabilitation programs, and migration from unimpaired reaches of the St. Louis River. This "resulted in the re-establishment of most species that were considered native to the Estuary" (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013).

The Blueprint Team identified the following sources: legacy toxics (sediment/water), habitat alterations, industrial and municipal discharges, exotic species invasions, and hydrologic alteration. These sources are largely addressed by modern day regulations and other BUIs (BUI 5: Restrictions on Dredging, and BUI 9: Loss of Fish and Wildlife Habitat).

After identifying sources and stressors and acknowledging overlap with other programs or BUIs, the Blueprint Team developed measurable status indicators that could be used to assess BUI condition. "It was determined by the team that, although most native fish and wildlife populations have been reestablished, there is a strong need for collection of baseline population information for some species" (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013). Status indicators were evaluated to identify where information gaps existed relative to data collection, synthesis, or index development. The Blueprint provided a pathway by which AOC resource professionals and partners would identify key native, sensitive, and invasive species populations for further assessment and evaluation relative to the BUI. Through this process, specific removal objectives for fish, birds, and small aquatic mammals were developed and incorporated into the BUI removal process as described below.

## Fish

The BUI removal objectives for fish are based on goals established in the MNDNR St. Louis River Estuary Lake Management Plan for three native indicator fish species: Walleye, Muskellunge, and Lake Sturgeon, and one invasive species, Ruffe (Minnesota Department of Natural Resources 2012). The objectives, which must be demonstrated with fish survey data, are listed below. *Italicized* text is quoted directly from the most recent RAP.

**Walleye**

*"Gillnet catch per unit effort (CPUE) is maintained at or above 5.0 per lift with a proportional stock density (PSD) between 30 and 60 in at least 50% of years surveyed since 2000."*

This objective identified an indicator of overall walleye abundance in the St. Louis River estuary and has evolved since first appearing in the 2013 Roadmap. The 2013 objective required a PSD greater than 50 in at least 50% of years surveyed since 2000. This objective was modified to its current form at the recommendation of the Fish Technical Team. The PSD objective identified an indicator of a size vs frequency distribution that suggests a sustainable, balanced population. While a PSD greater than 50 may very well reflect a balanced population in any given year, a PSD which remains greater than 50 most of the time may not reflect a balanced population (R. O. Anderson 1978). As recommended by the Technical Team, the PSD objective was replaced with a range between 30 and 60. This revised range was included in the 2017 RAP.

**Muskellunge**

*"Trap net CPUE is maintained at or above 1.0 per lift in at least 50% of years surveyed since 1997."*

**Lake Sturgeon**

*"Document an increasing trend of two to five year old fish captured in summer index nets, with at least two index values greater than 2.0 per gillnet lift."*

**Invasive Species**

*"An analysis of historical data that shows the Ruffe is not inhibiting the native fish population is required to remove this BUI."*

## Wildlife

AOC resource managers selected the wildlife species and species groups represented in the BUI removal objectives below based on their importance for developing consensus among resource managers that wildlife species are no longer limited by physical habitat, food sources, water quality, or contaminated sediments.

The 2013 RAP established removal objectives for target wildlife species (Piping Plover, Common Tern, Great Blue Heron, Bald Eagle, wetland bird species, and semi-aquatic mammals.) The removal objectives below are quoted directly from the most recent RAP and are *italicized.*

**Piping Plover**

*"Piping Plover populations have been limited by historical habitat losses and may be restricted by factors operating outside of the estuary; however, to support the U.S. Fish and Wildlife Service (USFWS) recovery goal of 150 breeding pairs for the Great Lakes Piping Plover population, efforts are being made to create suitable nesting habitat within the St. Louis River AOC. In order to remove this BUI, implementation of the Piping Plover habitat project (management action 2.05) in the RAP is necessary."*

The location of management action 2.05 is shown on **Error! Reference source not found.**. Though not stated in the RAP, a more detailed version of the USFWS Piping Plover recovery goals states that of the 150 breeding pairs in the Great Lakes, at least 50 pairs must reside outside of Michigan. The St. Louis River is at the far western edge of the Great Lakes Piping Plover's range, with the core population in Michigan.

**Common Tern**

*"Common Tern populations have been limited by historical habitat loss and may be restricted by factors within the estuary such as ice cover, flood events, gull predation, competition for nesting and young rearing habitat by gulls, and by other regional factors outside the estuary. Wisconsin's Common Tern Recovery Plan (WI Recovery Plan) establishes a goal of a 10-year average of 200 nesting pairs with sufficient production of 0.8-1.1 young per breeding pair to maintain population stability in the St. Louis River Estuary* (Matteson 1988)*. To support this goal, efforts are being made to maintain and enhance suitable nesting habitat within the SLRAOC. To remove this BUI, implementation of the Interstate Island restoration project (management action 2.06) in the RAP is necessary. In addition, the state agencies will continue to support habitat management and population monitoring at Interstate Island."*

This objective reflects significant changes adopted in the 2017 and 2019 RAPs upon recommendations from the Technical Team. The 2013 RAP objective required a target Common Tern colony size of 100 nesting pairs. According to data provided by the Technical Team, this objective was neither reflective of historic population size nor supportive of the WI Recovery Plan. The WI Recovery Plan established a goal of 200 nesting pairs in the SLRE as sufficient to maintain population stability; this goal was recommended for adoption by the Technical Team. The previous removal objective also did not include a target production rate, a metric suggested by the Technical Team as an important measure of population stability. Therefore, AOC Coordinators modified the objective in the 2017 RAP to include both the 200 nesting pair and 0.8-1.1 young per breeding pair goals from the WI Recovery Plan.

It is noteworthy that this objective requires support from state resource management agencies towards meeting Common Tern recovery goals, though meeting and/or sustaining the numeric goals is not required for BUI removal. The Technical Team indicated that the most critical support required to recover the SLRE Common Tern population was restoration of habitat at Interstate Island to mitigate for legacy loss of tern habitat in the estuary. AOC Coordinators and agency leaders concurred with this recommendation, including a new BUI 2 management action (2.06) requiring habitat restoration at Interstate Island in the 2019 RAP. See **Error! Reference source not found.** for the location of management action 2.06.

Though not stated as a potential population restriction within the estuary, emerging research conducted by Technical Team members independent of the AOC program indicates that mercury pollution may be an issue currently impacting the Interstate Island Common Tern population. This work is explained in the summary of Management Action 2.01 and recommendations to better understand this have been added to the Future

Actions section of this document. Separate BUIs; Fish Consumption Advisories and Restrictions on Dredging, are addressing legacy contamination and specifically legacy mercury in contaminated sediments in the St. Louis River AOC.

**Great Blue Heron**

*"Removal of this BUI is not dependent on the establishment of a Great Blue Heron rookery, but the recorded presence of the species in the estuary during nesting season since 1997 will provide additional evidence for BUI removal."*

**Bald Eagle**

*"Recovery of the Bald Eagle and the recorded presence of the species in the estuary during nesting season since 1997 is an indicator for BUI removal."*

**Wetland Bird Species**

*"Removal of this BUI is not dependent on populations of wetland-associated wildlife species. An AOC-wide bird follow-up survey to compare to work done in 1979 is necessary evidence for BUI removal."*

**Semi-Aquatic Mammals**

*"Removal of this BUI is not dependent on specific semi-aquatic mammal population numbers. However, to support development of concurrence among state resource management agencies, a semi-aquatic mammal survey will be conducted in the estuary to verify that populations are not limited by physical habitat, food sources, water quality, or contaminated sediments."*

## Removal Strategy (2013-19)

The strategy for BUI 2 removal includes the six management actions listed in **Error! Reference source not found.**. These management actions were designed to address the specific removal objectives described above.

Management Actions 2.01 through 2.05 first appeared in the 2013 Roadmap to Delisting (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013) and were developed through the combined efforts of numerous AOC partners/stakeholders in addition to the AOC coordinators and leaders who represent the agencies responsible for BUI removal and AOC delisting.

Management action 2.02 was expanded in 2018 to include a Lake Sturgeon study. Annual monitoring data showed that sturgeon recruitment was not trending towards BUI objectives. Fish Technical Team members made a formal recommendation to investigate factors limiting Lake Sturgeon recovery and determine whether limiting factors were influenced by legacy contamination. AOC Coordinators and agency leadership concurred with the recommendation, and the addition of the Lake Sturgeon study to management action 2.02 was formalized in the 2018 RAP (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2018).

Management action 2.06 was added in 2019 following a review of local Common Tern population data by the Avian Technical Team. With tern populations trending away from Wisconsin's Common Tern Recovery Plan goals and the increasing instability of Interstate Island due to rising lake levels, it was determined that restoration at this location was the most critical and cost-effective option to improve the tern population and mitigate legacy losses of shorebird habitat in the SLRE. In 2017 and 2018, the Technical Team recommended that restoration of Common Tern nesting habitat at Interstate Island be added as a required management action. AOC Coordinators and agency leadership concurred with the recommendation, and the addition of management action 2.06 was formalized in the 2019 RAP (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2019).

Completing management actions and meeting removal objectives triggers a review of BUI 2 for removal. A removal objective that is not met can be evaluated to determine whether limitations are outside of the scope of the AOC program or will be addressed by other BUIs. In these cases, a BUI 2 removal recommendation can proceed. While we acknowledge that on-going BUIs that remediate legacy pollution and/or restore degraded habitat will further aid in recovering historically degraded fish and wildlife populations, removing other BUIs is not required to remove BUI 2.

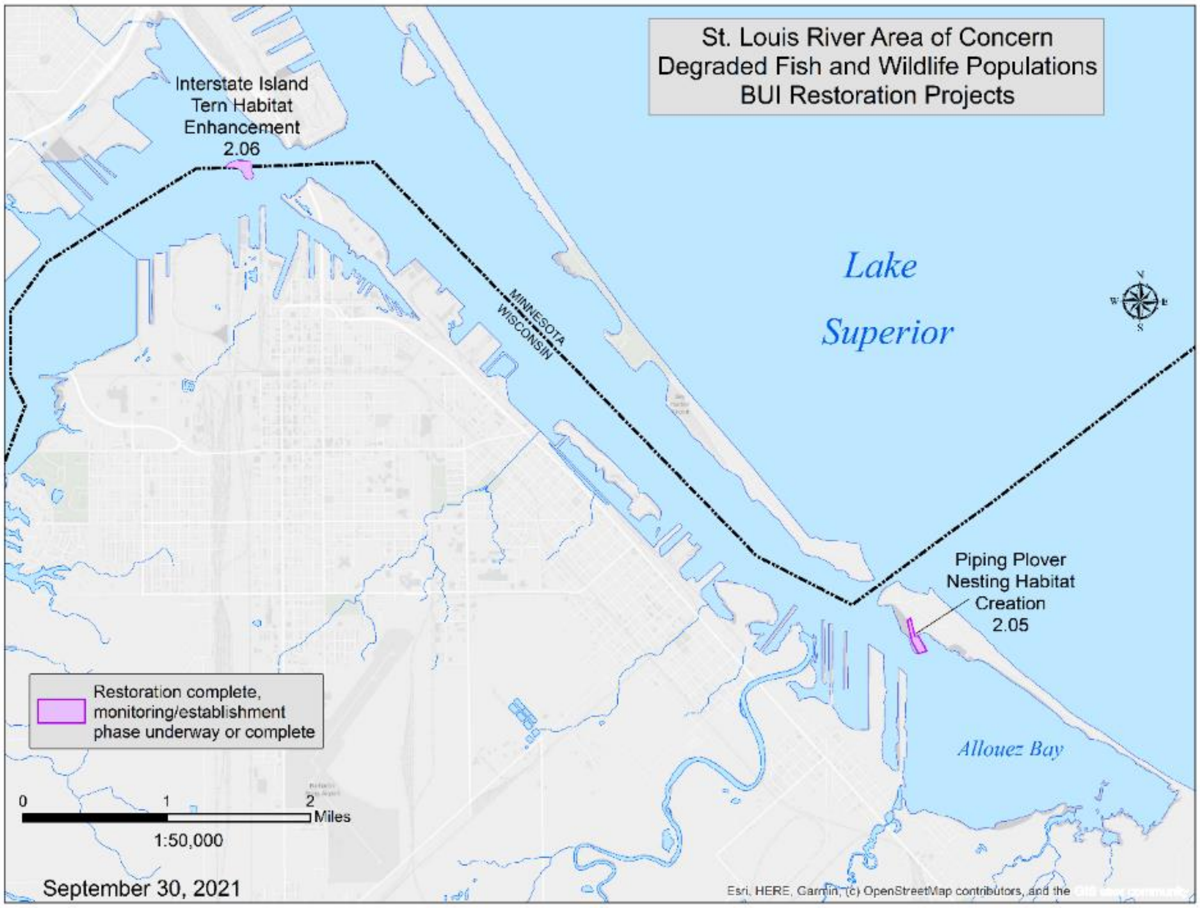


**FIGURE 3. LOCATION OF DEGRADED FISH AND WILDLIFE POPULATIONS BUI HABITAT RESTORATION PROJECTS 2.05 AND 2.06**

TABLE 1. MANAGEMENT ACTIONS NEEDED TO ACHIEVE BUI 2 REMOVAL

| Mgmt Action | Project Name | Project Description | Year Completed |
|---|---|---|---|
| **2.01** | Bird Inventory and Assessment | Conduct an estuary-wide bird inventory for target species to be combined with existing inventory data available.<br>Complete an AOC-wide assessment of bird population status using the combined dataset. | 2016 |
| **2.02** | Fish Population Monitoring and Assessment | Conduct regular MNDNR and WDNR fish population monitoring and evaluate to track status of target fish species against the BUI removal objectives.<br>Conduct study of Lake Sturgeon tissue to assess adverse effects related to legacy contaminants on early life stage and adult fish. | 2021 |
| **2.03** | Ruffe Assessment | Document Ruffe populations in relation to native fish populations within the estuary. | 2017 |
| **2.04** | Semi-Aquatic Mammal Survey | Conduct an estuary-wide semi-aquatic mammal survey. | 2016 |
| **2.05** | Piping Plover Habitat / Beach Nourishment | Increase available nesting habitat within area designated critical habitat. | 2020 |
| **2.06** | Interstate Island Avian Habitat Restoration | Restore and protect critical nesting habitat for Common Tern and stopover habitat for Piping Plover. | 2021 |

## BUI 2 Technical Teams

BUI Technical Teams provide expertise and recommendations to AOC staff and leaders on BUI goals, removal strategies, and the scientific interpretation of BUI status. BUI Technical Teams were originally formed during the 2013 Roadmap development. Since then, these teams changed depending on staff turnover, technical expertise, and interest. BUI Leaders managed three Technical Teams for BUI 2 including Avian, Fish, and Mammal (Table 2). Technical Teams met as needed for their review and recommendations on particular BUI 2 actions.

TABLE 2. CURRENT BUI 2 TECHNICAL TEAMS (DOES NOT INCLUDE PAST TECHNICAL TEAM MEMBERS; PAST TECHNICAL TEAM MEMBERS ARE DOCUMENTED IN THE ANNUAL REMEDIAL ACTION PLAN UPDATES)

**Avian Team**

| Name | Affiliation |
|---|---|
| Reena Bowman | US Fish and Wildlife Service |
| Annie Bracey | UMN Natural Resources Research Institute |
| Gaea Crozier | Minnesota Department of Natural Resources |
| Kris Eilers | St. Louis River Alliance |
| Rick Gitar | Fond du Lac Resource Management Division |
| Cherie Hagen (lead) | Wisconsin Department of Natural Resources |
| Sumner Matteson | Wisconsin Department of Natural Resources |
| Martha Minchak | Minnesota Department of Natural Resources |
| Alexis Grinde | UMN Natural Resources Research Institute |

| Name | Affiliation |
|---|---|
| Fred Strand | Avian expert |

**Fish Team**

| Name | Affiliation |
|---|---|
| Brian Borkholder | Fond du Lac Resource Management Division |
| Deserae Hendrickson | Minnesota Department of Natural Resources |
| Joel Hoffman | US Environmental Protection Agency |
| Paul Piszczek | Wisconsin Department of Natural Resources |
| Henry Quinlan | US Fish and Wildlife Service |
| Amy Roe (BUI 2 Technical Review Lead) | US Fish and Wildlife Service |
| Melissa Sjolund (lead) | Minnesota Department of Natural Resources |
| Darren Vogt | 1854 Treaty Authority |

**Mammal Team**

| Name | Affiliation |
|---|---|
| Shawn Crimmins | University of Alaska – Fairbanks (formerly Wisconsin Department of Natural Resources) |
| John Erb | Minnesota Department of Natural Resources |
| Rick Gitar (lead) | Fond du Lac Resource Management Division |
| Greg Kessler | Wisconsin Department of Natural Resources |
| Martha Minchak | Minnesota Department of Natural Resources |
| Tim Van Deelen | University Wisconsin - Madison |

# Management Actions Review

This section documents the successful completion of the six management actions required to achieve the BUI 2 target. Methods, findings, and conclusions are detailed for each management action.

## Management Action 2.01: Bird Inventory and Assessment

The primary monitoring for this action item was completed in 2016 by the Natural Resources Research Institute (NRRI), University of Minnesota – Duluth under contract managed by the Minnesota Pollution Control Agency (Bracey, Chatterton and Niemi 2016). The goal of the 2016 NRRI study was to provide a contemporary assessment of bird use in the SLRE that can be used to evaluate the status and trends of bird populations in the estuary over time. The study also compared populations against a 1979 survey to identify positive or negative trends attributable to legacy impacts and/or contemporary restoration work. The 1979 survey serves as the primary historical survey of bird use in the SLRE (Niemi, Davis and Hofslund 1979). The complete 2016 NRRI report is included as **Appendix A**. Study objectives included:

**Objective 1:** Summarize and compare contemporary baseline data gathered on bird use at sites planned for restoration and reference sites, and

**Objective 2:** Synthesize and compare contemporary bird use data with similar data gathered for SLRE sites in the late 1970s.

In addition to the 2016 NRRI study, annual monitoring of the Common Tern colony at Interstate Island was completed to meet Objective 3, below. The colony has been monitored using consistent methodology since 1989. This monitoring has been completed by various agencies over the years and is currently being led by MNDNR and WDNR with assistance from NRRI.

**Objective 3.** Measure Interstate Island Common Tern colony size and reproduction rates; compare these metrics against regional population recovery targets.

## Methods

**Objective 1.** NRRI selected ten contemporary AOC sites to compare against five reference locations (Table 3). NRRI sampled the sites during breeding and migration time periods between 2010 and 2015. During the breeding season, NRRI counted birds from a fixed point beginning at 0.5 hour before sunrise to 4.5 hour after sunrise. During each 15-minute count, all individuals seen or heard were recorded. Playback recordings helped elicit responses from hard-to-detect species. NRRI sampled each site two times during the breeding season.

TABLE 3. AOC AND REFERENCE SITES IN THE SLRAOC WHERE BIRD USE WAS DOCUMENTED

| AOC Site | Corresponding Reference Site |
|---|---|
| Minnesota Slip | Minnesota Point |
| Slip C | |
| 21st Avenue West | Little Pokegama Bay |
| 40th Avenue West | |
| Grassy Point | |
| Spirit Lake West | Spirit Lake East |
| Radio Tower (Cedar Yard) Bay | North Bay |
| Kingsbury Bay | |
| Mud Lake | Rask Bay |
| Perch Lake | |

Migration season counts lasted 10 minutes and occurred between sunrise and early afternoon from a fixed shoreline or boat location. All individuals seen or heard from the point were recorded. NRRI recorded the location of each observation, including flyovers, on aerial photo sheets which were digitized into Geographic Information System (GIS) format. Observations were classified into 16 unique species groups, with species of special concern (e.g. Common Tern and Piping Plover) also considered separately. NRRI sampled each site four times during each migration season (spring and fall).

**Objective 2:** NRRI also selected ten sites sampled from 1976-1979 to compare against the ten recent AOC sites. Historic sites were chosen based on their use of similar sampling techniques, objectives, and biological timing. NRRI digitized data from the 1970s data and aerial photo sheets. NRRI employed various statistical techniques to determine overall differences in community composition, species richness, and species of special interest or conservation concern between recent and historic surveys.

**Objective 3:** Since 1989, the Interstate Island Common Tern colony has been monitored by MN and WI wildlife biologists using a consistent methodology to estimate peak nest count and reproduction rates. Biologists visit the

Common Tern nesting colony annually at least twice a week, beginning in mid-May. Active nests are counted and marked through peak nest count (mid- to late-June) to determine the number of nesting pairs. The fate of all marked nests is recorded during each visit to determine nest success (hatched or failed). Chicks are captured as soon as possible after hatching and banded. To calculate fledgling rates, chicks are recaptured during each visit through mid-August, when most chicks have fledged. Fence maintenance and vegetation management are performed as needed.

## Findings

**Objective 1.** A total of 117,235 individual bird observations of 177 species were recorded during the migratory and breeding seasons at all sites sampled from 2010-2015. Researchers performed a total of 11 surveys at each site. Observations were grouped into three categories of birds: waterfowl, shorebirds, and waterbirds.

For waterfowl, there were approximately 880 individuals per AOC site, compared to 288 individuals per reference site. Of the waterfowl species observed at AOC sites, 80% were Canada Geese and Mallards. For shorebird species, there were six individuals per AOC site and two individuals per reference site. Most shorebird species were observed at 40$^{th}$ Avenue West complex. There were more waterbirds observed at reference sites (386 individuals per site) than AOC sites (67 individuals per site). Most of the waterbird observations were American Coot. Rails and Wrens were observed in low numbers at both AOC and reference sites, and there were no differences between sites. Great Blue Heron (a BUI 2 indicator species) were observed at higher frequencies at AOC sites (mean 0.33, Range 0-7) as compared to Reference locations (mean 0.04, range 0-1).

Species observed in AOC sites only, included six species of shorebirds: Killdeer, Greater Yellowlegs, Dunlin, Least Sandpiper, White-rumped Sandpiper, and Semipalmated Sandpiper, most of which were observations of single individuals, five species of waterfowl: Gadwall, Northern Shoveler, Greater Scaup, Red-breasted Merganser and Ruddy Duck, and one Sedge Wren. There were two species of waterfowl observed in reference sites only (Trumpeter Swan and Northern Pintail) and one shorebird (Pectoral Sandpiper).

There were no statistically significant differences in cumulative species richness (SR) between AOC and reference sites when all sites were pooled. This included richness calculated for all species observations as well as for water-obligate species only (i.e., rails, waterbirds, waterfowl, and shorebirds.)

Site-specific SR results were mixed. When comparing AOC sites against reference, four sites showed significant differences in cumulative SR: Minnesota Slip, Slip C, Radio Tower (Cedar Yard) Bay, and 40$^{th}$ Avenue West. At sites where differences were significant, cumulative SR was higher in the reference site with the exception of 40th Avenue West. In general, the AOC and reference sites had substantial overlap in their respective bird communities, primarily because of the high variability in bird species.

Since the completion of the avian survey in 2015, remediation and restoration are either complete or in development at all of the AOC sites listed in Table 3. These projects are associated with the Restrictions on Dredging and Loss of Fish and Wildlife Habitat BUIs. With the exception of remediated slips, these sites are anticipated to provide improved habitat to estuary bird populations after a period of recovery.

**Objective 2.** A total 196 species were observed in the historical and recent surveys at the 10 sites sampled during both survey periods in the SLR. There were 16,911 individual bird observations of 133 species (historical) and 11,042 individual bird observations of 132 species (recent) included in the analysis.

Of the 196 total species, 29 were observed in recent surveys only and 31 were observed in historical surveys only. However, many of the species unique to either historical or recent surveys were observed in small numbers (less than 5 individuals). All of the water-obligate species observed in historical surveys only were uncommon, rare, or very rare in the SLR and, therefore, the lack of observation of many of these species was partly due to their rarity and not likely due to local conditions.

Similar with the AOC and reference comparisons, NRRI found no significant differences in species richness between historical and recent surveys when all sites were pooled. This included richness measured for all species observations as well as for water-obligate species only. However, comparisons of site-specific species richness indicated significant differences in cumulative SR for three sites: 20th Avenue West, 27th Avenue West, and Spirit Lake West. For comparisons of water-obligate species, four sites had significant differences: 20th Avenue West, 27th Avenue West, 40th Avenue West, and Spirit Lake West. At each of these sites, with the exception of 40th Avenue West, cumulative SR was greater in historical surveys.

Findings specific to the BUI 2 indicator species are as follows:

- ***Great Blue Heron*** have declined in the SLRE relative to historical counts. Recent counts of Great Blue Heron at AOC sites produced a mean of 0.32 and range of 0-6 individuals. This shows a decline of approximately 1.73%/year when compared to a historical mean of 0.73 and range of 0-4. Great Blue Heron reductions may be associated with changes in the location of their colony site. During the historical surveys, Great Blue Herons nested near Kimball's Bay. This nesting site no longer exists, likely due to residential development and/or Bald Eagle predation. While researchers are unaware of the colony's current location, it is likely further from the SLRE, influencing their presence on the river. The Great Blue Heron specific removal objective has been met based on the report generated by management action 2.01, and therefore recorded presence, of Great Blue Herons within the AOC, which provide additional evidence for BUI removal.
- ***Bald Eagle*** populations have increased substantially in the SLRE compared with the historical period (1970s) when the species did not nest at all. Today, there are up to five nesting Bald Eagle pairs in the SLR, but no information on their overall nesting success. Researchers noted that recovery of the Bald Eagle is supportive of BUI removal, but its recovery has had little to do with changes in the SLRE. The species has recovered because of the banning of DDT, the focused management efforts to protect nest sites, the improvement in reduced contaminant loads in food supplies, and its increased tolerance to human disturbance. The Bald Eagle specific removal objective has been met based on the report generated by management action 2.01, and therefore recorded presence, of Bald Eagles within the AOC, which provide additional evidence for BUI removal.
- ***Wetland Bird Species*** at AOC sites were surveyed and found to have greater abundance and similar species richness when compared to reference sites and similar species richness when compared to historical surveys. When comparing specific AOC-reference site pairs, results were mixed. When comparing historic and current species data, lower use by Wood Duck, Northern Pintail, and Common Loon were inconsistent with regional trends. The Wetland Bird Species specific removal objective has been met based on the report generated by management action 2.01, and therefore comparison to 1979 Survey results, of Wetland Bird Species, within the AOC, which provide additional evidence for BUI removal.

**Objective 3.** Peak Common Tern nest counts at Interstate Island have been estimated annually, dating back to 1977. However, current monitoring methods have stayed consistent since 1989, making the 1989-2020 data set the most accurate for assessing population size and reproductive rates (**Appendix B**). 2014 was a significant year for Interstate Island, in that it marked the beginning of a sudden and sustained increase in Lake Superior water levels (Figure 4). This rise in water level flooded portions of Interstate Island, increasing pressure from co-nesting Ring-billed Gulls and threatening to inundate the tern nesting area.

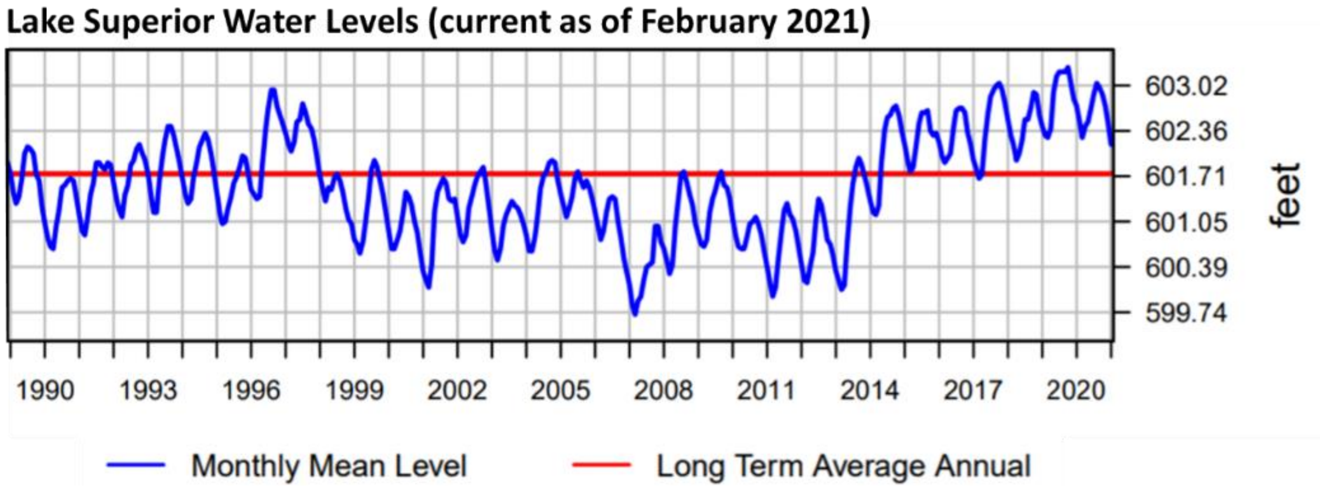


FIGURE 4. MONTHLY MEAN AND LONG-TERM AVERAGE ANNUAL LAKE SUPERIOR WATER LEVELS (1989 - 2021). USACE DATA AVAILABLE AT: HTTPS://LRE-WM.USACE.ARMY.MIL/FORECASTDATA/GLBASINCONDITIONS/LTA-GLWL-GRAPH.PDF

<u>Common Tern Population Size</u>

A summary of Common Tern population size metrics is presented in Table 4 and Figure 5. The Wisconsin recovery goal as referenced in the RAP is 200 nesting pairs (measured as a rolling 10-year average). Between 1989 and 2021, the average annual population size was 178 nesting pairs.   During this time frame, the annual population count met or exceeded the 200-pair target in 11 years (33%). The 10-year average (calculated for years 1998-2021) met or exceeded the 200-pair target in nine years (38%).

Due to sudden and sustained high water levels, 2014 marked the beginning of a decline in the size of the tern nesting population.  In 2015, the colony was only 101 nesting pairs, the lowest number since 1989 (Figure 5). The annual population count prior to 2014 averaged 194; after 2014, the average fell to 131. The 200 pair goal was met 11 times prior to 2014; since 2014, the 200 pair goal has not been met.  Based on a rolling 10-year average (calculated for years 1998-2021), the 200 pair goal was met eight times prior to 2014, and only once since 2014 (Table 4 and Figure 5). Lower populations observed during recent counts were likely caused by habitat loss due to the high-water levels.  Resource managers implemented emergency repairs in 2015 and eventually added a more comprehensive project, Interstate Island Avian Habitat Restoration, to the AOC program as Management Action 2.06.

<u>Common Tern Reproductive Rate</u>

Annual fledgling counts (number of young fledged per nest) were recorded annually at Interstate Island starting in 1989 (**Appendix B**).  A summary of Common Tern reproductive rate metrics is presented in Table 5 and **Error! Reference source not found.**.  The RAP references a WI Common Tern recovery goal to have a minimum fledgling rate between 0.8 and 1.1 young fledged per nest.  These two goals encompass a range in recommendations from various researchers studying Great Lakes Common Tern colonies (Matteson 1988).  The 0.80 fledgling rate can be considered a “bare minimum,” for recovery, while the 1.1 rate is more desirable to ensure long-term population stability.

TABLE 4. SUMMARY OF COMMON TERN ANNUAL NESTING PAIR COUNTS AT INTERSTATE ISLAND (1989-2021).

| Metric | Annual Nesting Pair Count | 10-year Nesting Pair Average |
|---|---|---|
| 1989-2021 | | |
| **n** | 33 | 24 |
| **Average** | 178 | 194 |
| **Years ≥ 200** | 11 | 9 |
| **Percent ≥ 200** | 33 | 38 |
| 1989-2013 | | |
| **n** | 25 | 16 |
| **Average** | 194 | 200 |
| **Years ≥ 200** | 11 | 8 |
| **Percent ≥ 200** | 44 | 50 |
| 2014-2021 | | |
| **n** | 8 | 8 |
| **Average** | 127 | 183 |
| **Years ≥ 200** | 0 | 1 |
| **Percent ≥ 200** | 0 | 13 |

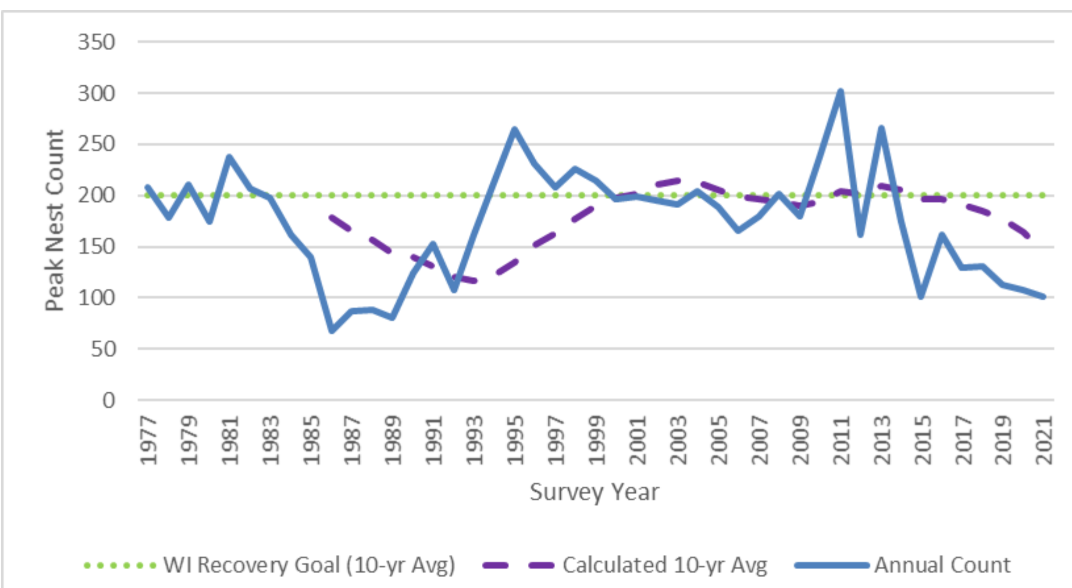


FIGURE 5. ESTIMATED NUMBER OF PAIRS OF COMMON TERNS NESTING IN THE ST. LOUIS RIVER ESTUARY (1989-2021). THE WISCONSIN RECOVERY GOAL IS 200 NESTING PAIRS (10-YR AVERAGE).

Between 1989 and 2021, the average annual fledgling rate was 0.87. During this time frame, the annual fledgling rate met or exceeded the 0.80 goal 18 times (55%) and met or exceeded the 1.1 goal 11 times (33%). The 10-year rolling average (calculated for years 1998-2021) met or exceeded the 0.80 goal 13 times (54%) but did not ever meet the 1.1 goal. The rolling 10-year average shows oscillations around a slight, steady downward trend (Figure 6).

High water levels beginning in 2014 have had a less dramatic impact on fledgling rates at Interstate Island. Factors such as weather and mammalian/avian predation on chicks are important factors influencing fledgling success. The annual fledgling rate prior to 2014 averaged 0.88; after 2014, the average fell slightly to 0.81. Interestingly, 2013 was the last year that the rolling 10-year average was at or above the 0.80 goal, while the 1.1 goal has not ever been met based on a 10-year average (Figure 6).

TABLE 5. SUMMARY OF ANNUAL COMMON TERN FLEDGLING RATES AT INTERSTATE ISLAND (1989-2021).

| Metric | Annual Fledgling Rate (No. fledged per nest) | 10-year Average Fledgling Rate (No. fledged per nest) |
|---|---|---|
| 1989-2021 | | |
| **n** | 33 | 24 |
| **Average** | 0.87 | 0.81 |
| **Years ≥ 0.8** | 18 | 13 |
| **Percent ≥ 0.8** | 55 | 54 |
| **Years ≥ 1.1** | 11 | 0 |
| **Percent ≥ 1.1** | 33 | 0 |
| 1989-2013 | | |
| **n** | 25 | 16 |
| **Average** | 0.88 | 0.86 |
| **Years ≥ 0.8** | 13 | 13 |
| **Percent ≥ 0.8** | 52 | 81 |
| **Years ≥ 1.1** | 8 | 0 |
| **Percent ≥ 1.1** | 32 | 0 |
| 2014-2021 | | |
| **n** | 8 | 8 |
| **Average** | 0.81 | 0.70 |
| **Years ≥ 0.8** | 5 | 0 |
| **Percent ≥ 0.8** | 63 | 0 |
| **Years ≥ 1.1** | 3 | 0 |
| **Percent ≥ 1.1** | 38 | 0 |

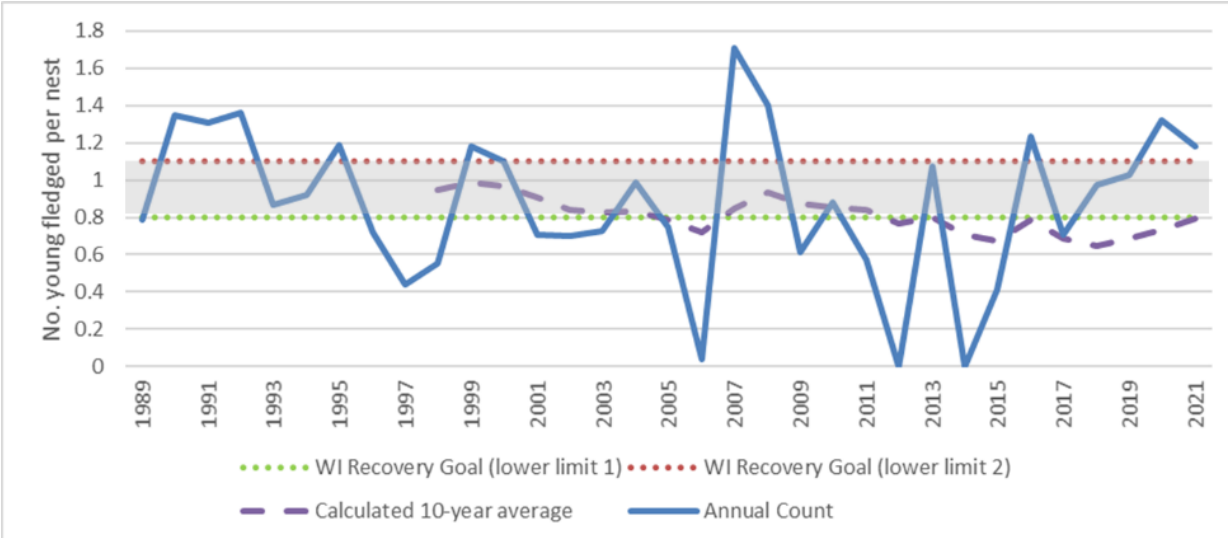


**Figure 6. Number of Common Tern young fledged per nest (1989-2021). The Wisconsin recovery goal establishes a minimum rate between 0.8 and 1.1 (shaded area).**

The extensive monitoring effort on Interstate Island has provided important information relative to the estuary's Common Tern population. The statewide and regional significance of the colony reinforces the value of maintaining this long-term effort. Until recently, about two-thirds of all the Common Terns breeding in the Lake Superior Basin nested on Interstate Island. With the success of the constructed nesting colony at Ashland, WI and continued loss of Interstate Island habitat, that proportion has fallen to less than half in recent years, though the long-term nesting pair average remains higher at Interstate Island (Strand 2020). With the Interstate Island tern colony trending away from regional recovery goals as described above, island restoration was determined necessary and incorporated into the BUI 2 removal strategy as management action 2.06 in (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2019).

In addition to the monitoring activities described above, area researchers have further studied the SLRAOC Common Tern population. Bracey et al. (2018) studied potential impacts to Common Tern populations during the nonbreeding season using geolocators. Researchers found the majority (70%) of the inland Common Tern colonies (including the Great Lakes) spending the nonbreeding season on the coast of Peru. Such aggregation could make the terns vulnerable to local negative effects, impacting population size and breeding success in the SLRAOC. Though population impacts occurring during the nonbreeding season are outside of the SLRAOC program influence, researchers emphasized the importance of continued colony management at sites such as Interstate Island.

Bracey et al. (2021) focused additional Common Tern research on patterns of dietary mercury (Hg) exposure. Measurements of Hg in blood and feathers indicated higher Hg accumulation in the summer (breeding season). Hg concentrations were also correlated with increased industrial influence and riverine-based diet. Mercury levels in chicks hatched on Interstate Island were often above published toxicological risk thresholds, whereas chicks hatched at the Ashland, WI colony were below the threshold, suggesting greater mercury exposure in the SLRE relative to Chequamegon Bay. Because chicks obtain most of their diet from the river, the study highlights the importance of identification and remediation of estuary contamination.

A 2021 study demonstrated that mercury is still actively cycling in the estuary food web and is present in sediments outside the boundaries of identified SLRAOC remediation and restoration projects at concentrations below remediation action levels (Janssen, et al. 2021). At each sampling location, inorganic mercury originated from two or three of the following sources: industrial/legacy, watershed, and precipitation. The AOC program targets legacy sources of mercury, but not those from modern watershed and precipitation sources. A strategy is being implemented under other BUIs to identify and remediate hot spots (those sites where contamination exceeds state action levels), with the understanding that it is not feasible to remediate low levels of contamination in sediments elsewhere in the estuary. AOC restoration and remediation project sites are used as forage locations for Common Terns and an outcome of completed projects will be a reduction of mercury entering the food web. A lag between completion of AOC projects and observable food web impacts is likely and may be two or more years.

While Common Terns in the SLRE may continue to face contaminant exposures unique to the estuary, as well as beyond the estuary during the non-breeding season, the primary factor limiting colony stability is habitat suitability. Accordingly, habitat restoration and long-term management remained the primary BUI 2 management action targeting terns. While the findings of Bracey et al. (2021) did not prompt an additional BUI 2 management action, continued study of mercury's impacts on Common Tern chicks is included as a recommended future action. Separate BUIs, Fish Consumption Advisories and Restrictions on Dredging, are addressing the contribution of legacy mercury sediment contamination to the food web in the St. Louis River AOC.

## Conclusions

Based on the results of the 2016 bird study and planned long-term monitoring and maintenance of the restored Interstate Island habitat, objectives for the chosen avian indicator species have been met and Management Action 2.01 is complete (Table 6). As described above, numeric objectives were set for some species, based on measured indicators of populations. These numeric objectives were met.

For other indicator species, action-based objectives such as studies or on-the-ground projects supportive of regional population goals were established. For both the Piping Plover and Common Tern, habitat was chosen as the most significant population limitation for the AOC program to address under BUI 2. Therefore, the objectives and associated management actions for these species are limited to restoring critical habitat and supporting long-term monitoring and maintenance. While achieving regional recovery goals for Piping Plover and Common Tern is desired by all involved it is not required for BUI 2 removal.

Successful completion of Management Action 2.01 supports a BUI 2 removal target focused on legacy impacts to key indicator species. Through implementation and review of these actions, resource managers, species experts, and researchers developed recommendations for future actions. These actions will provide continued support for avian populations in the SLRE and would take place outside of the AOC program and framework. See Section 6.0 for a summary of recommended future actions.

TABLE 6. SUMMARY AND STATUS OF BUI 2 AVIAN SPECIES OBJECTIVES

| Objective | Status | AOC Objective Met? |
|---|---|---|
| **Piping Plover:**<br>To remove this BUI, implementation of the Piping Plover habitat project (Project 2.05) is necessary. | Piping plover populations may be restricted by factors operating outside the estuary. The required restoration project was completed in 2020 (See action 2.05 for additional information.) | Yes |
| **Common Tern:**<br>State agencies will continue to support habitat management and population monitoring at Interstate | Population metrics for the Common Tern are not currently meeting goals established by | Yes |

| Objective | Status | AOC Objective Met? |
|---|---|---|
| Island to support the following population goals established by Wisconsin's Common Tern Recovery Plan:<br>A. 10-year average of 200 nesting pairs<br>B. Production of 0.8-1.1 young per breeding pair<br>*Achieving the numeric population goals is not required to remove BUI 2.* | Wisconsin's Common Tern Recovery Plan (Figure 5, Figure 6) though achievement of these goals is not necessary to meet the BUI removal objective for this species.<br>State agencies restored habitat at Interstate Island (see management action 2.06). A long-term monitoring, maintenance, and management plan for the habitat is in place and supported by state resource management agencies. | |
| **Great Blue Heron:**<br>Removal of this BUI is not dependent on the establishment of a rookery, but the recorded presence of the species in the estuary during nesting season since 1997 will provide additional evidence for BUI removal. | Between 0-6 Great Blue Herons were observed at AOC sites in the SLR during the study period, though no colonies have been located for many years. | Yes |
| **Bald Eagle:**<br>Recovery of the Bald Eagle and the recorded presence of the species in the estuary since 1997 is an indicator for BUI removal. | Today there are up to five nesting Bald Eagle pairs in the SLR, but no information on overall nesting success. Recovery is supportive of BUI removal but has little to do with changes in the SLR. | Yes |
| **Wetland Bird Species:**<br>Removal of this BUI is not dependent on populations of wetland-associated wildlife species. BUI removal requires an AOC-wide bird follow-up survey to compare to work done in 1979. | Site-specific comparisons produced mixed results. Overall, AOC sites had greater abundance and similar species richness when compared to reference sites and similar species richness when compared to historical surveys.<br>Wetland habitat is targeted for restoration by many projects under the Loss of Fish and Wildlife Habitat BUI (BUI 9) and will benefit wetland bird species. | Yes |

## Management Action 2.02: Fish Population Monitoring and Assessment

Management Action 2.02 contains two components: annual fish population monitoring targeting key species, and a study of Lake Sturgeon to assess potential limitations to population recovery.

### Fish Population Monitoring

Walleye historically was an important fish species in the St. Louis River for subsistence fishing by native people, early commercial harvest, and for recreational fishing (Kaups 1984). Declines in water quality from industrial and municipal discharges, hydrologic alteration from construction and operation of dams, and loss of aquatic habitat due to dredging and filling associated with port development all are cited as contributing to declines in Walleye populations in the estuary (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1992).

Walleye and Muskellunge populations in the St. Louis River are monitored as part of routine fisheries assessments conducted by both Minnesota and Wisconsin Departments of Natural Resources. They were selected as a target species for BUI removal due to their historic subsistence harvest importance and their importance as recreational gamefish.

The Lake Sturgeon population in the SLR was extirpated in the early 1900s due to overfishing and habitat changes caused by dam construction and pollution. Since then, management actions such as stocking and spawning habitat restoration have helped reintroduce Lake Sturgeon to the St. Louis River. Fry, fingerlings, and/or yearlings were periodically stocked between 1984 and 2000 from two sources (Wolf River, WI and Sturgeon River in MI). Schram et al. (1999) reported that spawning habitat in the upper St. Louis River estuary “remained relatively unchanged and adequate.” In 2009, rock riffles were constructed below the Fond du Lac dam, creating spawning cells and in 2015, additional spawning habitat was constructed at Chambers Grove. In 2011, the first documented naturally reproduced larval Lake Sturgeon were captured during a drift netting survey below the Fond du Lac dam on the St. Louis River, demonstrating that the stocking and habitat improvement efforts have been somewhat successful (Anselmo, Bogyo and Borkholder 2020). The successes realized in 2011 were translated by fisheries resource managers into population recovery metrics that appeared feasible at the time they were incorporated into the 2013 Roadmap. In 2019, the MNDNR completed a Lake Sturgeon Management Plan to document the history of Lake Sturgeon Management in the St. Louis River and guide future management.

All Lake Sturgeon captured in the SLR since 2007 by MNDNR or WDNR have been implanted with a Passive Integrated Technology (PIT) tags. Approximately 1,800 individuals have been PIT tagged with less than one percent of those being verified as females. Most of the Lake Sturgeon captured in the SLRE are captured during the spring near the known spawning habitat. The adult Lake Sturgeon population is monitored annually utilizing a variety of methods, including boom electrofishing, backpack electrofishing and hand dipping. River flow rates vary greatly in the spring on the St. Louis River and have a large impact on the number of Lake Sturgeon captured each year. Also, Lake Sturgeon do not spawn every year adding more variability to the number of Lake Sturgeon captured yearly. These factors contribute to making adult Lake Sturgeon captured during spring spawning surveys an unreliable metric for measuring the recovery of the Lake Sturgeon population. For this reason, metrics for adult Lake Sturgeon were not developed for BUI 2.

Despite past efforts to recover SLRAOC Lake Sturgeon populations via fingerling stocking, natural recruitment has not been observed at anticipated levels. Lake Sturgeon objective metrics were established during the period of fingerling stocking and considered the number of years since stocking was initiated and current age of the population. Since then, the population has not even approached the BUI objectives, though there is evidence that some natural reproduction is occurring. Spring surveys conducted by the 1854 Treaty Authority and Fond du Lac Resource Management Division between 2011-2020 have captured between two and 1,028 larval Lake Sturgeon in drift nets. The number of larvae captured have been highly variable, likely due to many influencing factors including survey timing/methods, water levels, and number of spawning females. However, data suggest that Lake Sturgeon have continued to naturally reproduce in the St. Louis River to some degree. Larval survey methods were standardized in 2019, making future analysis of reproductive success easier (Anselmo, Bogyo and Borkholder 2020).

## Methods

The Minnesota and Wisconsin Department of Natural Resources, together with the 1854 Treaty Authority and the Fond du Lac Band of Lake Superior Chippewa, monitor fish populations annually. Populations of Walleye, Muskellunge, and Lake Sturgeon were selected as objectives for assessing health of the fish population. Fisheries managers share monitoring efforts and reports; these data were compiled for this report by MNDNR Duluth Area Fisheries staff.

An annual summer gillnet assessment is completed by the MNDNR in late June/early July and used to monitor Walleye and Lake Sturgeon populations. These data were used to evaluate BUI 2 removal objectives for Walleye and Sturgeon. Experimental gillnets are set at the same locations and in the same orientation each year to make catch rates comparable. Data collected as part of this assessment are used to calculate CPUE, PSD and other metrics.

The Muskellunge population is monitored in spring using trapnet assessments that take place approximately every two to five years. Historically, when BUI 2 was written, assessments were completed only during the peak of Muskellunge spawning and at a limited number of locations with high quality spawning habitat. More recent assessments attempt to capture the entire spawning season by setting nets when water temperatures are near 44 degrees Fahrenheit and rising and continuing until Muskellunge are being captured in very low numbers post-spawn. Nets are also being set in more locations to look for additional potential spawning areas. These assessment changes would be reflected in the 2017 and 2018 sampling discussed in this document and in future assessments.

### Findings

Based on monitoring results, BUI 2-specific removal objective metrics have been met for Walleye and Muskellunge, but not for Lake Sturgeon, triggering additional study (

Table 7). Complete data sets for these indicator species are attached as **Appendix C**.

The specific removal objective metrics for Walleye included a gillnet CPUE greater than 5.0 and PSD between 30 and 60 in at least 50% of years sampled since 2001. Based on survey results, both metrics were met, with CPUE and PSD meeting the objectives in 81% and 59% of years sampled, respectively (Figure 7, Figure 9). For Muskellunge, the specific removal objective metric required trapnet CPUE greater than or equal to 1.0 in at least 50% of years surveyed since 1997. Based on survey results, this metric was met with CPUE meeting the objective in 75% of years (Figure 8).

The specific removal objective metrics for Lake Sturgeon included documenting an increasing trend in two to five year old fish captured in summer index nets (Figure 10), and measuring at least two index values greater than 2.0 fish per lift (Figure 11). Since 2002, the number of sturgeon age five or less has been close to zero with no increasing trend. Similarly, index values have also been below the 2.0 goal since 2002. Failure to meet established targets triggered an additional study detailed below to justify BUI removal.

TABLE 7. COMPARISON OF FISH POPULATION MONITORING AGAINST FISH SPECIES OBJECTIVES

| Objective | Result | Objective Met? |
|---|---|---|
| **Walleye:**<br>A. Gillnet catch per unit effort (CPUE) ≥ 5.0 in at least 50% of years surveyed since 2000<br>B. Proportional stock density (PSD) between 30 and 60 in at least 50% of years surveyed since 2000. | A. The objective is met in 13 of 16 years (81%) surveyed since 2000 (Figure 7).<br>B. The objective is met in 10 of 17 years (59%) surveyed since 2000 (Figure 9). | Yes |
| **Muskellunge:**<br>Trapnet CPUE ≥ 1.0 in at least 50% of years surveyed since 1997. | Muskellunge CPUE was 1.0 or greater in 4 of 6 years (75%) surveyed since 1997 (Figure 8). | Yes |
| **Lake Sturgeon:**<br>A. Document an increasing trend of two to five year old fish captured in summer index nets.<br>B. At least 2 index values greater than 2.0 per lift. | A. Populations of Lake Sturgeon under 5 years of age are not observed to be increasing (Figure 10).<br>B. Lake Sturgeon index values have not been measured above 2.0 per lift since 2000 (Figure 11). | No – for this reason, a supplemental study was initiated in 2018. |

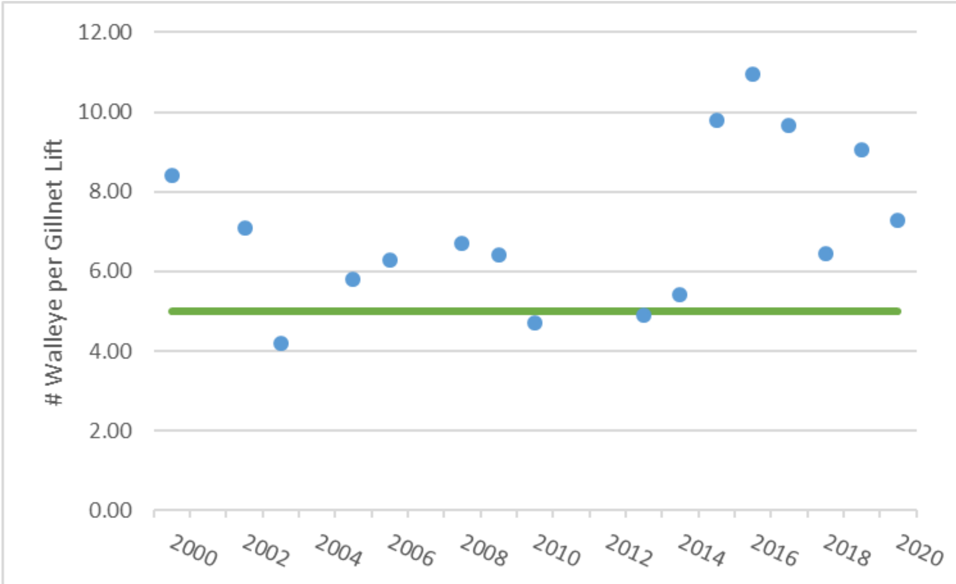


**FIGURE 7. CPUE MONITORING DATA FOR WALLEYE. THE GREEN LINE INDICATES THE BUI 2 METRIC, THE OBJECTIVE BEING TO ACHIEVE THIS IN AT LEAST 50% OF YEARS.**

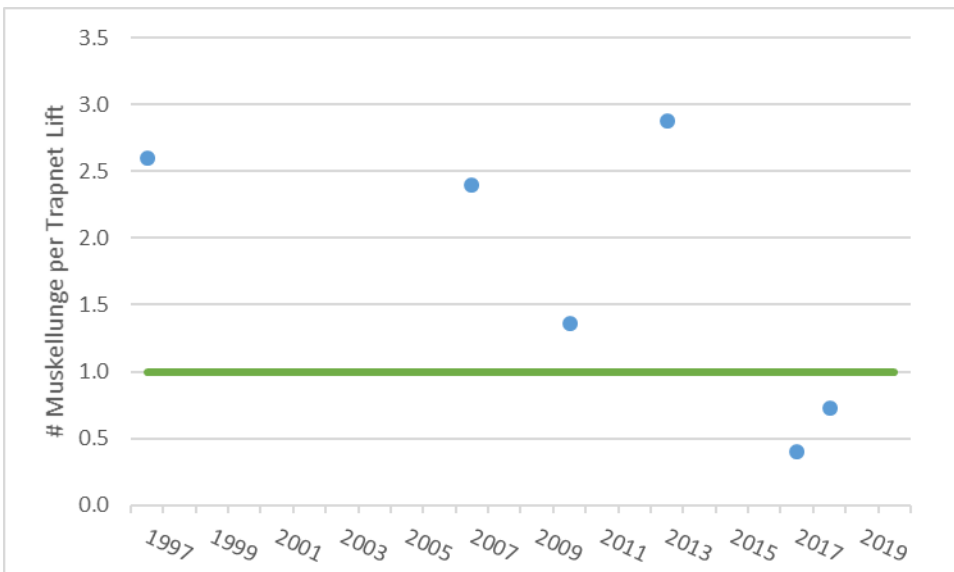


**FIGURE 8. CPUE MONITORING DATA FOR MUSKELLUNGE. THE GREEN LINE INDICATES THE BUI 2 OBJECTIVE.**

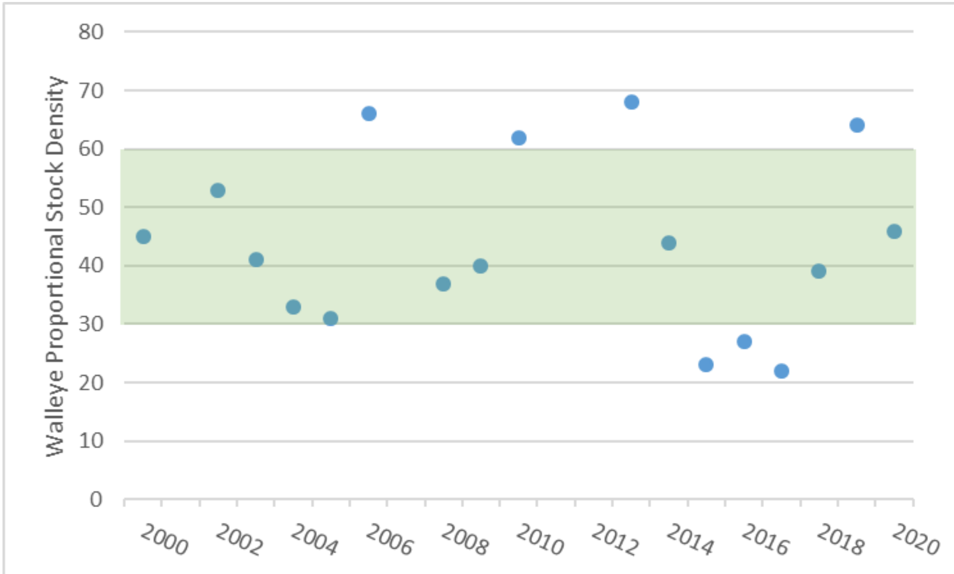


**Figure 9. PSD monitoring data for Walleye. The corresponding BUI 2 objective range is shaded in green**

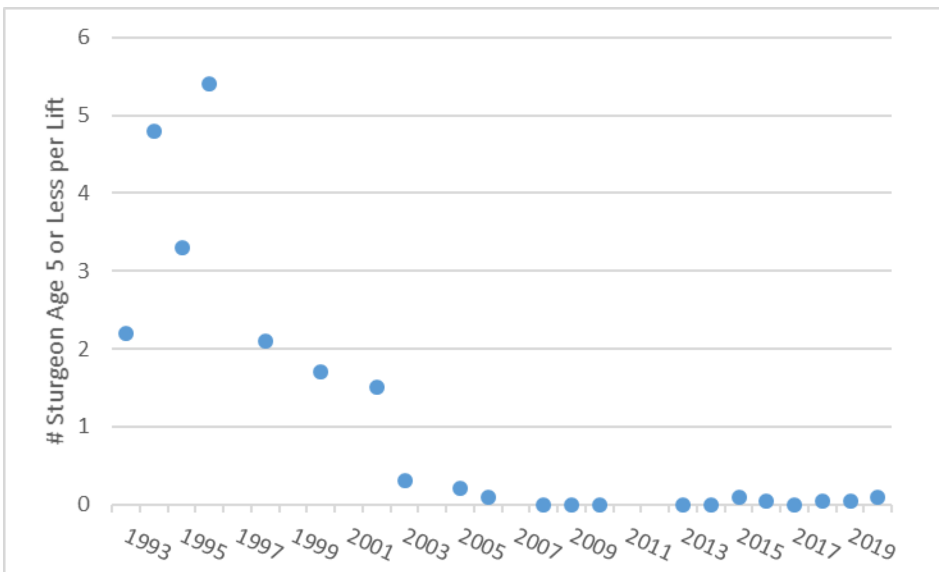


**Figure 10. Number of Lake Sturgeon age five or less captured per lift in summer index nets. The corresponding BUI 2 objective is to document an increasing trend.**

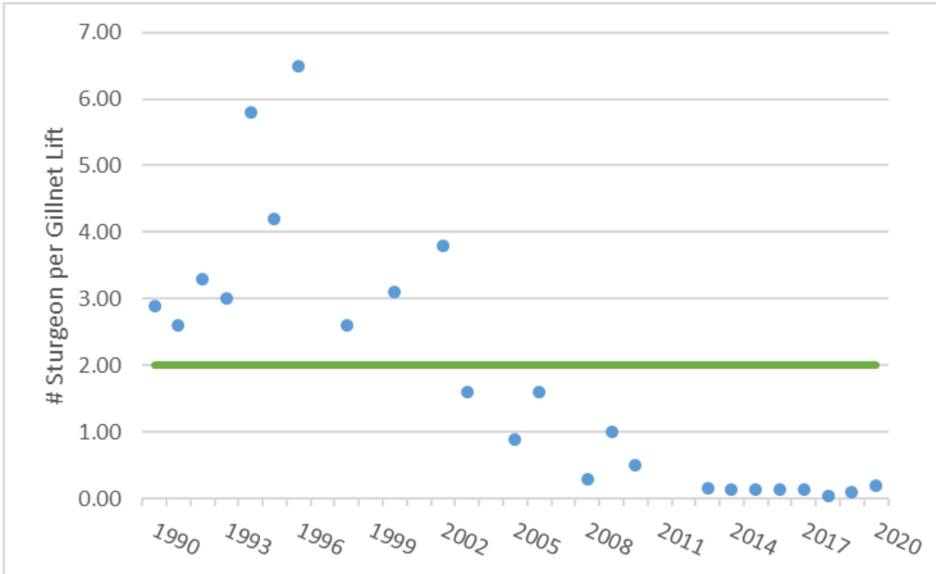


FIGURE 11. TOTAL NUMBER OF LAKE STURGEON CAPTURED PER LIFT. THE GREEN LINE INDICATES THE CORRESPONDING BUI 2 OBJECTIVE.

## Conclusions

All removal objectives have been met except the Lake Sturgeon objective. In 2017, Fish Technical Team members concurred that population objectives for Walleye and Muskellunge had been met. Fisheries data collected since 2017 continues to support these objective metrics (

Table 7).

Species monitoring has shown the Lake Sturgeon specific removal objective metric has not yet been met, however in 2017, Fish Technical Team members made a formal recommendation to assess potential factors limiting Lake Sturgeon recovery and determine whether limiting factors were influenced by legacy contamination. This additional study to justify BUI removal was completed in 2021 is summarized below. A detailed report is included as **Appendix D**.

## Lake Sturgeon Study

Despite past efforts to recover Lake Sturgeon on the SLRAOC via fingerling stocking, restoring habitat, and improving water quality, annual population measurements indicate that recruitment was not being observed at anticipated levels or trending towards BUI 2 removal targets. Resource managers and fisheries experts working in the SLRAOC identified a need to assess potential factors limiting Lake Sturgeon recovery and determine whether limiting factors are influenced by legacy contamination. Lake Sturgeon are particularly susceptible to bioaccumulation of legacy contaminants because it is an extremely long-lived, late-spawning, bottom-dwelling species that consumes organisms from the bottom sediments, where many contaminants occur in the St. Louis River. The Technical Team hypothesized that various legacy contaminants may accumulate in adult sturgeon and maternally transfer to their developing embryos, potentially leading to decreased survival of early life-stage fish. This prompted a study in 2018, led by Dr. Jon Doering (former EPA GL-TED researcher), to determine the effects of legacy contaminants on early life stage and adult Lake Sturgeon. The study suggested low likelihood of adverse impacts (see final report in **Appendix D**).

### Methods

During the springs of 2017, 2018, and 2019, resource managers captured spawning Lake Sturgeon in the SLRE between the Fond du Lac dam and Highway 23. Captured sturgeon were measured, weighed, and their sex determined. Samples collected included blood, eggs (if female), and pelvic fin clips (for genetic stock identification purposes). A total of seven egg samples were collected from captured females between the three sampling efforts.

The egg and plasma samples were analyzed for concentrations of contaminants known as dioxin-like chemicals (DLCs). DLCs are known to have great toxicity to wildlife, producing a range of issues from wasting disease to deformities and early life mortality. To express the total toxicity of a mixture of compounds as a single number, the suite of DLCs measured in eggs and plasma were converted to toxic equivalents (TEQs). The summed potency for each egg sample, in terms of TEQs, were compared to a mortality curve which predicts the percent mortality of Lake Sturgeon as a function of TEQs present in eggs. TEQs measured in St. Louis River sturgeon were also compared to a healthy reference population.

Due to field sampling yielding only seven egg samples, plasma samples from a larger population of females, males, and immature individuals were analyzed to get a broader scope of DLC exposure and bioaccumulation.

### Findings

DLCs were detected in egg samples collected from all seven female Lake Sturgeon caught in the SLRE. DLC concentrations in St. Louis River sturgeon eggs were generally at levels higher than the reference population. However, the resulting TEQs were all below the level predicted to cause early life stage mortality. Further, all TEQs were below levels predicted to cause developmental anomalies.

Plasma samples were collected in 2018 and 2019 from a total of six female, two male, and ten immature SLRAOC Lake Sturgeon. DLCs were detected in plasma collected from all but two of the females. Of the 19 DLCs analyzed, individual plasma samples had between one and seven DLCs measured at detectable levels.

TEQs in SLRAOC Lake Sturgeon plasma were greater than EQs in eggs. However, TEQs in plasma remained well below the calculated threshold corresponding to sublethal or lethal effects in adults. This suggests that neither lethal nor sublethal adverse effects would be expected to occur in female, male, or immature adult individuals because of bioaccumulated DLCs.

Long-term exposures to DLCs in Rainbow Trout have shown that effects on adults, such as decreased survival, altered behavior, and impaired reproduction, begin to occur at body burdens that result in maternal transfer of doses that cause toxicities in early life stages. Because adult Lake Sturgeon in the SLRAOC are not accumulating DLCs at levels known to cause lethal or sublethal effects, body burdens are such that females likely are not transferring DLCs to eggs at concentrations known to cause mortality.

### Conclusions

The study demonstrates that DLCs are present in SLRAOC Lake Sturgeon and are bioaccumulating to levels greater than the reference population. However, DLC concentrations being maternally transferred from adult fish to embryos are well below concentrations known to cause lethal or sublethal effects. In addition, concentrations measured in adult fish suggest that neither lethal nor sublethal effects are expected. Therefore, bioaccumulated or maternally transferred DLCs are unlikely to be a contributing factor to the failed recruitment of Lake Sturgeon in the SLRAOC (**Error! Reference source not found.** and **Error! Reference source not found.**). See **Appendix D** for the complete study report.

Females Lake Sturgeon may live from 80 to 150 years. They do not reach sexual maturity until 18 to 27 years of age, and even then will spawn intermittently. Data from the Bad River Lake Sturgeon population suggests that Lake Sturgeon inhabiting the relatively colder waters of Lake Superior mature later than most other populations and spawn less frequently (Schloesser and Quinlan 2019). It is likely that too few female Lake Sturgeon in the St. Louis River have reached sexual maturity and spawned for fishery assessment to detect a change in abundance of age two-to-five-year-old fish. It is even less likely that females have spawned multiple times and first-time spawners are less fecund or successful. Larval sturgeon were first captured in 2011 and numbers have been steadily increasing. It is evident from increasing captures during the spring spawning run that spawning stock biomass is continuing to increase which should result in an increase in larval and age 2-5 fish over the next decade.

While the BUI removal objective targets for Lake Sturgeon have not been reached, findings from the Lake Sturgeon study indicate that legacy contaminants associated with the SLRAOC are not contributing to slow sturgeon recruitment. Historic habitat loss associated with the SLRAOC has been addressed via restoration of Lake Sturgeon spawning habitat at multiple locations in the estuary and restoration of fish habitat at multiple sites associated with the Loss of Fish and Wildlife BUI (BUI 9). While it is likely that patience is still required for Lake Sturgeon recovery, the SLRAOC program has completed the key actions needed to address legacy impacts to the species and BUI removal is justified despite failure to meet removal targets.

## Management Action 2.03: Ruffe Assessment

The Eurasian Ruffe was first identified by Wisconsin DNR in the St. Louis River Estuary in 1987, likely introduced via ship ballast water discharged by a vessel arriving from a Eurasian port. By 1993, it was considered the most abundant of the 60 species found in Duluth Harbor.

Ruffe exhibit rapid growth and high reproductive output and adapt to a wide range of habitat types. There was concern that ruffe may have a detrimental effect on native species in the St. Louis River, such as Yellow Perch and Walleye, by feeding on the young of these species, or by competing for food.

Subsequent analysis demonstrated “that it is not possible to implicate ruffe for decreasing densities of native fishes in the St. Louis River during the study period” (Bronte, et al. 1998). The researchers reported that the variability in abundance and distribution of Ruffe in the SLRE did not negatively impact recruitment of other species in the estuary.

Despite an intense focus on Ruffe during the 1990s, and its subsequent spread to other Great Lakes, resource managers knew little of how its abundance has since changed in the St. Louis River over the last two decades (Gutsch 2017). To better understand the dynamic between Ruffe and native Lake Superior fish, management action 2.03 was created.

## Methods

University of Minnesota PhD candidate Michelle Gutsch addressed this action item by completing an analysis of historical fish population data to confirm that Ruffe are not currently inhibiting native fish populations (Gutsch 2017). A complete copy of Ms. Gutsch's dissertation is included as **Appendix E,** noting that the research specific to this action item is captured in *Chapter 2: Population change of an invasive fish, Ruffe, thirty years post-introduction: boom or bust?*

The study's purpose was to determine if Ruffe populations in the St. Louis River conform to typical invasive species "boom-bust" patterns and to identify interactions between Ruffe and native prey and predatory fishes.

Annual St. Louis River fisheries data were available between 1993 and 2015. Bottom trawl surveys were used to sample the fish community. Ruffe invasions in other systems were found to directly compete with species such as Round Goby, Trout Perch, Yellow Perch, Spottail Shiner, Emerald Shiner, and Johnny Darter. These other species were classified as "competitor species" for analysis. Gill net and creel surveys were used to sample the predator fish community. Predator fish are those large enough to consume Ruffe and included Walleye, Northern Pike, Smallmouth Bass, and Muskellunge.

**Determining CPUE**

CPUE was calculated separately for trawl, gill net, and creel survey data. Trawl CPUE data were standardized to account for variations in trawl width, tow duration, and vessel speed. Trawling CPUE for Ruffe and competitors was calculated as the number of fish caught per hectare. For gill net surveys, CPUE was calculated as the number of fish caught per net using only those individual predatory fish determined to be large enough to consume Ruffe. Average CPUE for Ruffe, competitors, and predators were calculated across sample dates for each year.

**Modeling Population Dynamics**

Researchers applied an exponential growth model to look for a "boom" pattern in Ruffe CPUE. The model incorporated comparable data sets from 1985-1992 to document the beginning of the Ruffe invasion in the SLR. Researchers developed additional statistical models to determine whether competitor and predatory fish species' populations were correlated with each other and/or with Ruffe and how strong those correlations were. Using this method, the species with the strongest effect on Ruffe could be identified.

## Findings

Ruffe populations in the SLR increased in the ten years following its first detection (1985-1995), followed by a decline from 1996-2015. In SLR, the Ruffe population conforms to the typical invasion theory "boom-bust" model and is currently in the "bust" phase. The rate of increase (1985-1995) contrasted with the more moderate rate of decline (1996-2015).

During the study period, researchers found the following correlations between Ruffe CPUE and other species' CPUE:

- Ruffe and Trout Perch (competitor) = strong, negative correlation
- Ruffe and Yellow Perch (competitor) = strong, positive correlation
- Ruffe and Emerald Shiner (competitor) = moderate, negative correlation
- Ruffe and Northern Pike (predator) = moderate, positive correlation

Negative correlations between Ruffe and competitor CPUE in SLR suggests that competition for food, space, or other resources may be contributing to the Ruffe decline. As Ruffe populations decrease, Trout Perch and Emerald Shiner may be outcompeting Ruffe and Yellow Perch. Researchers hypothesized that competition for spawning habitat and food were likely factors contributing to these correlations.

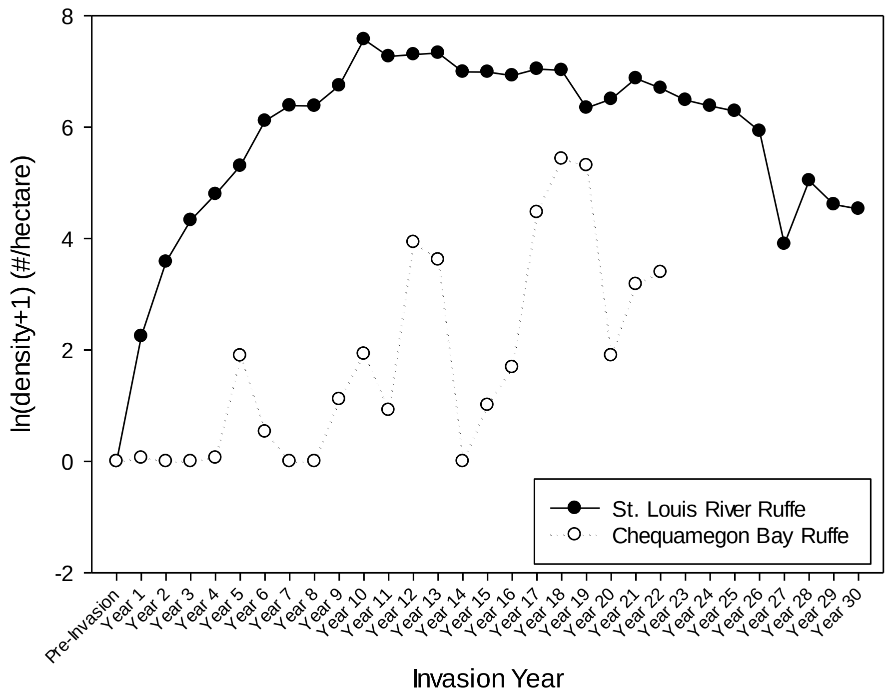


FIGURE 12. RUFFE POPULATIONS IN THE ST. LOUIS RIVER BY INVASION YEAR. ADAPTED FROM GUTSCH (2017); REFER TO APPENDIX E FOR A DISCUSSION OF THE CHEQUAMEGON BAY STUDY.

## Conclusions

Research conducted by Gutsch (2017) supports earlier conclusions made by Bronte et al. (1998) that Ruffe was not causing a decline in native SLR fishes. Gutsch (2017) also concluded that the SLR Ruffe population has been declining for two decades and was in the "bust" phase of the invasion at the time of the study. In 1995, the Ruffe CPUE reached a maximum, possibly indicating the population had reached or exceeded its carrying capacity, and then slowly declined.

The Fish Technical Team reviewed the Ruffe assessment results and confirmed that the BUI 2 specific removal objective for the invasive species has been met. Armed with a better understanding of the boom-bust cycle of Ruffe invasion in the SLR, resource managers can better formulate management decisions related to its control. Though the action item is complete, SLR Ruffe populations continue to be measured and monitored through ongoing fisheries data collection by resource management partners.

# Management Action 2.04: Semi-Aquatic Mammal Survey

"*Removal of this BUI is not dependent on specific small aquatic mammal population numbers. However, to support development of concurrence among state resource management agencies, a small mammal survey will be conducted in the estuary to verify that populations are not limited by physical habitat, food sources, water quality,* *or contaminated sediments*" (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013).

The primary objective of the semi-aquatic mammal study undertaken in 2015 is to determine if the SLRAOC supports similar populations of mammal species versus a non-AOC area with comparable habitat. The justification for BUI removal must rely on this contemporary comparison of wildlife populations because historic (pre-degradation) wildlife data are not available for the SLRAOC. Only limited anecdotal accounts are available; for example, WDNR staff report comments from long-time trappers that muskrat numbers today are a fraction of historic levels (personal communication with Greg Kessler, WDNR)

## Methods

The assessment for this action item was completed by the University of Wisconsin – Madison under contract administered by Wisconsin Department of Natural Resources. Project partners included the MPCA, and Fond du Lac Band of Lake Superior Chippewa. The GLRI funded the study. The data were also the focus of a master's thesis by University of Wisconsin – Madison Department of Forest and Wildlife Ecology master's candidate Bryn Evans. A complete report on the study is included as **Appendix F**.

**Target Species**

Ms. Evans completed an estuary-wide survey of select semi-aquatic mammal species with the objective to determine if the SLRAOC currently supports populations of native mammal species in similar abundances as areas with less extensive impairment (Evans 2016). Four native species dependent on aquatic resources were selected to study: river otter, mink, beaver, and muskrat. These species span several trophic levels and have distinct roles within aquatic ecosystems, which presents both unique risks for species decline linked to industrialization as well benefits linked to species recovery. All four species are regulated as furbearers, and experienced extensive reductions in population size following European settlement of the area.

**Study Areas**

The research focused on the following areas within the SLRAOC: the St. Louis River from the Fond du Lac dam to the Bong Bridge, and the Nemadji River from six miles above Crawford Creek to its outlet in Allouez Bay. This area encompasses the diversity of flow regimes and habitat types in the SLRAOC. Due to a lack of data on semi-aquatic mammals in the SLRE prior to degradation, the study used reference areas to determine if target species' populations meet AOC recovery requirements based on statistical equivalency test demonstrating the equality of a degraded system and a control system. Although the estuary's size and characteristics are unmatched in the region, The Boulder Lake Reservoir and St. Croix River were selected as examples of relatively unimpaired lake and riverine reference systems, respectively.

**Field Methods**

To collect data on beaver, otter, muskrat, and mink, motion-triggered trail cameras were deployed within the AOC and two reference sites. Three deployments using 28, 29, and 65 cameras occurred during multiple seasons spanning 2014-2016. In addition to camera traps, aerial surveys by fixed wing aircraft were used to collect fall and winter data on beaver, otter, and muskrat in 2015-2016. Data collected on the flights included a GPS track log of the flight path; starting and stopping locations and times for each segment of the surveys; and waypoints for any sign recorded. Observable sign for beaver consisted of lodges, food caches, and wood chips and downed trees because of chewing activity. Muskrat sign consisted of "push-ups" in the fall, which were generally not observable after snow had fallen. Otter sign consisted of tracks in snow, which were only observable during periods of ice cover and were discernable from other animal tracks by the distinctive sliding pattern in the snow. Detections of target species by aerial surveys and trail cameras were modeled to determine occupancy rates.

## Findings

Detections of beaver, mink, muskrat and otter within the SLRAOC were found to be statistically similar to those in reference sites during all three seasons of the study. The study concluded that the removal objective for small semi-aquatic mammals is being met as there is no evidence that a current lack of suitable habitat, resources or pollution was impeding their ability to naturally repopulate the area. The data cannot ascertain if aspects of habitat, food availability, or water quality are sub-optimal, but there is support that the ecosystem is healthy to the degree required for these species to meet their life requirements at levels similar to areas without the same history of degradation. Tech Team members have reviewed the study and affirmed that the metric for removal of the beneficial use impairment related to small aquatic mammals has been met.

## Conclusions

A Small Mammal Technical Team was established and reviewed the results of this study. In 2017, the subcommittee accepted the report and concluded that the status of small mammal species in the St Louis River Area of Concern is sufficient to remove the beneficial use impairment.

# Management Action 2.05: Piping Plover Habitat / Beach Nourishment

Piping Plovers are on the threatened and endangered species lists for Minnesota and Wisconsin, as well as being federally listed.  Of the three Piping Plover breeding populations, the Great Lakes population is the only one listed as endangered. In the Great Lakes region, Piping Plovers use sparsely vegetated beaches, cobble pans, and sand spits to breed and raise their young for a period of approximately three to four months, annually.  Wintering grounds range from North Carolina to Florida and along the Florida Gulf Coast to Texas, Mexico, and the Caribbean Islands. Threats to Piping Plovers include the following: habitat destruction and degradation, human disturbance, and contaminants.  Plovers are also impacted by the genetic and geographic consequences of their small population size (U.S. Fish and Wildlife Service 2003).

Management action 2.05 was officially included in the SLRAOC “Roadmap to Delisting” in 2013 (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013) and was determined necessary to restore historically lost nesting habitat for the endangered Piping Plover in the SLRE and support the 2003 USFWS Great Lakes Recovery Plan for the Great Lakes Piping Plover (U.S. Fish and Wildlife Service 2003).  The 2003 Recovery Plan’s ultimate objective is to remove the Great Lakes population from the list of Threatened and Endangered Species, requiring that specific recovery criteria for population size, reproduction, habitat, and long-term protection are met.

## Methods

Management action 2.05 was led by the WDNR with the following partners: City of Superior, Fond du Lac Band of Lake Superior Chippewa, St. Louis River Alliance (SLRA), USACE, USEPA, USFWS, and the University of Wisconsin Sea Grant.  This project was funded with approximately $4 million from the Great Lakes Restoration Initiative.

WDNR coordinated the project with support from a Restoration Site Team (RST) of local and regional species experts.  The RST evaluated SLRE sites for their potential to attract and retain a Piping Plover population, ultimately choosing the Wisconsin Point Bird Sanctuary (Bird Sanctuary).  The Bird Sanctuary site is a fenced-in area owned by the WDNR.  WDNR, USWFS, Douglas County, and the St. Louis River Alliance (SLRA) have all supported efforts to actively manage the Bird Sanctuary site since the 1980s for both Piping Plovers and Common Terns.   Past management activities included grading, vegetation control, signage, fencing, removal of large woody debris, monitoring, and public outreach. The area is closed seasonally from April 1 through August 1 to reduce human impacts during breeding season.

In 2017, WDNR entered into an agreement with the USACE to design the project.  Primary objectives of the design included the following:

- Provide adequate nesting acreage at locations that have shown plover activity in the past
- Create nesting areas that can be maintained over time using anticipated funding
- Sustain available habitat for 25-30 years

The design process recognized that constructing the habitat may provide ancillary benefits to the estuary such as creating a more sheltered place in Allouez Bay for manoomin (wild rice), fish spawning habitat, and dual use habitat for other shorebirds (USACE - Detroit District 2017).

The USACE evaluated historic water levels, resulting in a design incorporating target elevations that minimize impacts of water variability and shoreline erosion in the establishment of plover habitat.  Beach widths, slopes, and open areas for breeding, nesting, and foraging were designed based on recommendations from RST species experts.

The project has created approximately 14 acres of open sand and cobble beach suitable for Piping Plover nesting and foraging habitat.  The sand required to construct the beach was obtained through annual Operations and Maintenance dredging of the federal navigational channel by USACE.  WDNR developed physical and chemical criteria for these construction materials to ensure their suitability for Piping Plover. In 2019, approximately 87,485 CY of approved dredged materials were placed to create the habitat.  The existing spit feature was widened to encourage long-term connectivity, and a new beach was created.  Following sand placement, the habitat was enhanced with cobbles, native dune grass restoration and a fence upgrade to deter predators.

See Figure 13 for photos showing the completed restoration.


FIGURE 13. BEFORE AND AFTER IMAGES SHOWING THE COMPLETED HABITAT RESTORATION AT WISCONSIN POINT. IMAGE SOURCE: GOOGLE EARTH

Partners will continue teaming up for Piping Plover, focusing efforts on long-term habitat establishment and management, outreach and education, and monitoring. WDNR, USFWS and the St. Louis River Alliance, with GLRI funding, are planning to:

- Assess habitat twice yearly to identify management actions necessary to maintain suitable habitat
- Conduct management actions to maintain suitable habitat (i.e., remove unwanted vegetation/wood, maintain slopes)
- Develop education and outreach materials to protect Piping Plover habitat from human activity and predators

- Monitor site for Piping Plovers, document nesting pairs, fledgling survival and success at the site

Outlined below are components of the AOC project's Establishment Phase. This work is supported with AOC program resources and will ensure the high-quality habitat is established and plans are in place for long term management.

1. Native dune grass will be planted on approximately 1 acre of the site outside of the nesting area.
2. Habitat assessments will be conducted twice per year by species experts to identify any actions needed to ensure the habitat is meeting the species criteria.
3. Habitat management actions identified will be conducted by the St. Louis River Alliance and WDNR. Actions to maintain the habitat criteria may include: woody debris and vegetation management, shoreline grading to maintain slope and manage cliffing, and cobble pan maintenance.
4. Property management actions including gate installation, fencing, and signage will be conducted by WDNR staff.
5. Monitoring of Plover Tiger Beetle (food source) re-colonization of the site.

## Findings

The habitat has attracted Piping Plover in the few seasons that it has been available. During site monitoring in 2020 and 2021, 1 plover was observed using the habitat in each year. Annual monitoring begins in late April and continues daily until chicks are fledged or no nesting occurs (June-July).

Please see **Appendix G** for a complete summary (including photos) on the WI Point project

## Conclusions

The project was successfully constructed as designed and therefore the Piping Plover specific removal objective has been met. Early monitoring results are encouraging and plans for long-term maintenance, management, and nest protection are in place to ensure the high-quality habitat is sustained into the future. A complete project summary report, including photos, is included as **Appendix G.**

The 2019 project is not the only work in the estuary to benefit the Piping Plover and long-term plans are in place to continue assisting this species recovery. AOC partners and agencies have been involved in managing the species and habitats for many years. Partners have conducted significant work related to piping plover habitat and outreach including plover monitoring, creating a educational curriculum, engaging the media and developing nesting attractants and improvements at Shafer beach.

# Management Action 2.06: Interstate Island Avian Habitat Restoration

Interstate Island is a small island within the Duluth-Superior Harbor, constructed by the United States Army Corps of Engineers (USACE) in the late 1930s using dredge materials. In the 1980s, the site became of interest as habitat for Common Terns as human disturbance and site development in other nesting locations in the estuary made those places no longer viable for the species (Matteson 1988). A 1989 restoration project cleared all vegetation completely to expose sand substrate to attract Common Terns, which are listed as threatened in Minnesota and endangered in Wisconsin. The entire breeding population of the SLRE was subsequently attracted to the site in 1989 and 1990 (Penning 1993). The island has since been managed by MNDNR and WDNR as a Wildlife Management Area and Wildlife Refuge, respectively (Minchak and Staffon 2007).

A legacy of habitat loss in the SLRAOC has confined Common Terns to one, increasingly unstable site. A colony of Ring-billed Gulls also nests on the island. Competition for tern nesting habitat by Ring-billed Gulls has increased in recent years as gull nesting habitat has decreased due to flooding and erosion of the island, increasing the vulnerability of the tern colony. This vulnerability has been expressed in terms of decreased colony sizes and even complete colony failure in some years due to flooding, predation, or other unknown issues.

Interstate Island is the only federally designated critical habitat for Piping Plovers in the state of Minnesota. While Piping Plover have not nested at Interstate Island and are not likely to nest there while the Ring-billed Gull colony is present, the island may be used as stopover habitat for Piping Plover, as well as other migrating shorebirds (United States Fish and Wildlife Service 2001). NRRI researchers monitoring shorebirds at Interstate Island observed 15 different species in 2019-20, including multiple plover and sandpiper species (Kolbe 2021)

In 2014, rising lake levels and increased storm surges resulted in significant and sustained flooding at Interstate Island. In 2015, the colony was only 101 nesting pairs, the lowest number since 1989 (**Error! Reference source not found.**). This prompted local resource managers to increase the elevation of the tern nesting area and build a protective berm around it in 2015 to protect it from destruction by flooding. Though the 2015 construction successfully protected the nesting area for the time being, NRRI researchers and resource managers at MNDNR and WDNR watched water levels continually rise, while nesting and reproduction rates remained well below recovery goals (**Error! Reference source not found.**).

The BUI 2 Avian Tech Team (Table 2) recommended that SLRAOC Coordinators review and revise the necessary management actions related to Common Tern habitat as the population is in decline due to legacy habitat loss. This recommendation resulted in the Interstate Island avian habitat restoration project being included as a BUI 2 management action in 2019 (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2019).

## Methods

The Minnesota Land Trust (MLT) managed the Interstate Island avian habitat restoration project for the MNDNR and its partner the WDNR. Federal partners included the USEPA, USFWS, and the USACE. The project cost approximately $2,800,000 to manage, design, construct, and monitor. Funding support came to MNDNR and MLT from the GLRI via USEPA, Minnesota Outdoor Heritage Fund, NOAA Coastal Program, USFWS Great Lakes Fish and Wildlife Restoration Act, and USFWS Midwest Region Coastal Program. Through a USACE partnership, Harbor Maintenance Trust Funds were used to contract the excavation and placement of sand acquired through annual navigational channel operations and maintenance dredging.

A project Restoration Site Team (RST) comprised of project coordinators and local avian species experts began meeting in 2019 to develop the following restoration purpose and objectives and generate a concept design. The Project's primary purpose is to maintain viability of the largest Common Tern nesting colony in the Lake Superior watershed. Objectives supporting this purpose include the following:

1. Restore at least 5.5 acres of stable upland (above 605.5 feet IGLD85) habitat for nesting and rearing use by Common Terns and nesting and stopover use by other migratory shorebirds, including Piping Plover;
2. Increase island elevation to protect against flooding;
3. Stabilize the island perimeter to prevent scour and further habitat loss;
4. Enhance substrate composition and control woody vegetation;
5. Quantify target populations' status, nest success, and habitat usage; and
6. Develop and implement a proactive program to sustain habitat quality.

In 2019, the MLT awarded a design contract to SEH, Inc. (SEH). In consultation with the RST, SEH developed project plans and specifications that reflect habitat requirements listed in scientific literature and promote long-term habitat resiliency. The design incorporated specific habitat requirements of Common Terns and Piping Plover while providing sandy beach habitat for use by a variety of shorebirds. Specific requirements included shoreline slopes of approximately 8% and a minimum of 5.5 acres of habitat above an elevation of 605.5' IGLD85. This is equal to the historic Lake Superior high elevation of 604.5' IGLD85, with an additional foot incorporated to ensure island protection and resiliency to increasingly unpredictable Lake Superior water levels and storm surges

The design accommodated the use of both commercially-sourced sand and sandy material obtained through annual USACE operations and maintenance dredging in the Duluth-Superior Harbor. The project team worked with the USACE to identify suitable materials, resulting in the project's construction being divided into two separate contracts managed by the MLT and USACE.

The MLT awarded a contract to JF Brennan Company (JF Brennan) in March 2020 to begin restoring habitat in the island's existing footprint. In April 2020, JF Brennan removed all woody vegetation to eliminate predatory bird perches. JF Brennan then placed commercially-sourced sand to elevate flooded areas of the island. The work was completed prior to May 1 to vacate the island prior to the Common Tern nesting season.

Following the nesting season and beginning in late August 2020, JF Brennan rebuilt the Common Tern nesting area. The new 30,000 sqft nesting area featured an embankment at a higher elevation to eliminate future flood risk. They also installed a permanent fencing system to assist in managing the nesting area and constructed rock vanes to protect the island against wind erosion.

Beginning in late September 2020 through early November, the USACE led the expansion of the island on the Wisconsin side to approximately double its previous size. The USACE awarded a contract to Roen Salvage Company (Roen Salvage) for the work, which beneficially used dredge materials from yearly navigation channel operation and maintenance dredging. Roen Salvage hydraulically placed and graded dredged sand in two separate lifts to expand the island footprint and tie in to the existing island. Two lifts were required to keep the highest quality sand at the surface. JF Brennan revisited the newly expanded island in April 2021 to install additional rock vanes in the island expansion area and perform any needed touch-up grading or fill placement. Though not directly related to Common Tern habitat restoration or required for BUI 2 removal, native dune vegetation will be planted on Interstate Island beginning in summer 2021 to provide additional stability (contractor and planting list to be determined).

See Figure 14 for aerial photos depicting the progression of construction at Interstate Island.

During the 2020 nesting season (mid-May through mid-August), NRRI researchers visited the island at least twice weekly to count Common Tern nests, monitor and band chicks, and monitor and band adults. Starting in mid-May, NRRI also researchers documented other migratory shorebirds through weekly surveys (through September) and perimeter camera traps, which take one photo every 10 minutes during daylight hours (through October). Annual tern and shorebird monitoring is contracted using project grant funds through 2023; tern monitoring will likely continue using state natural resource funding.

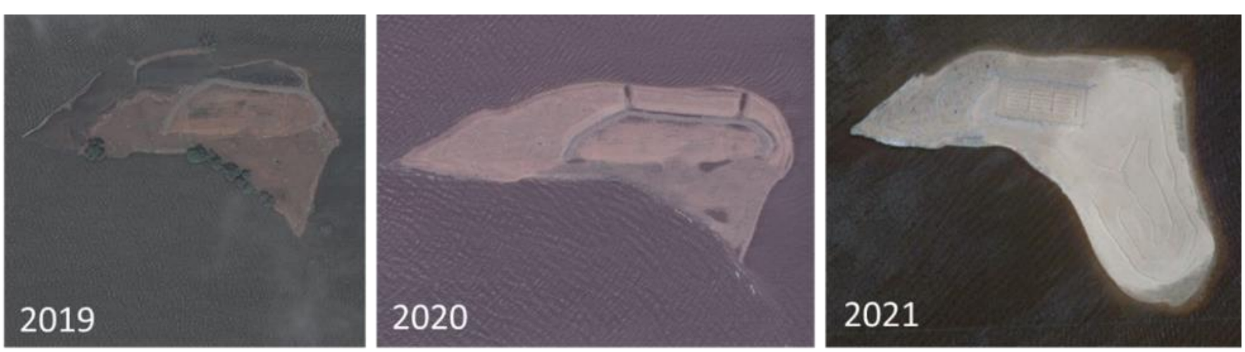


FIGURE 14. PROGRESSION OF RESTORATION CONSTRUCTION AT INTERSTATE ISLAND. IMAGE SOURCE: GOOGLE EARTH

**Long-term Monitoring & Maintenance**

In 2020, SEH developed a comprehensive Long-term Monitoring and Maintenance Plan (Plan) that identifies the goals, methods, schedule, and triggers for ongoing monitoring and maintenance of the restored Interstate Island habitat (Short Elliott Hendrickson Inc. 2020). MN and WI wildlife managers have a history of jointly managing Interstate Island and will continue to do so, using the Plan as guidance.

The Plan incorporates measurable objectives related to the amount of stable upland habitat, nesting area and other island substrates, woody and invasive vegetation, and target avian populations' nesting and reproductive success. The Plan establishes performance standards, monitoring protocols, and reporting requirements. Any maintenance triggered by the plan would occur outside of the Common Tern nesting season (March 1 to August 30). The Plan is adaptive, with the MN and WI resource managers performing an annual review, updating the plan with any required changes or updates.

## Findings

See **Appendix H** for a complete summary (including photos) on the Interstate Island project. Successful implementation of the project plans and specifications by JF Brennan and Roen Salvage were documented through field observation by project engineers, review of periodic field reports, and post-construction surveys. JF Brennan and Roen Salvage submitted final "as built" surveys which demonstrated the project produced 6.7 acres, slightly exceeding the project objectives for long-term, stable, upland habitat.

The 2020 nesting season was successful, with 108 nesting pairs and 1.32 young fledged per nest (**Error! Reference source not found.** and **Error! Reference source not found.**). The fledgling rate was the highest observed since 2008. NRRI researchers continue to observe a diverse assemblage of migratory shorebirds using Interstate Island (Kolbe 2021). A total of four years of Common Tern and migratory shorebird monitoring will be completed under the project's funding, and it is assumed that long-term Common Tern monitoring by MN and WI wildlife managers will continue into the foreseeable future. These data will continue to demonstrate the impacts of this restoration work on target species.

The 2020 long-term monitoring and maintenance plan for Interstate Island in 2020 was developed by the project's design engineers with a high level of input and review from MN and WI resource managers and researchers. The plan is explicitly linked to measurable project objectives, allows for adaptive management, and has support from the state wildlife managers who jointly manage the island resource.

## Conclusions

Due to the high level of input and involvement from local species experts, the design and specifications for the Interstate Island restoration project were highly tailored to the target species, appropriately protective/conservative, and are therefore have a high probability for long-term success. The project's construction contractors effectively implemented the project plans and specifications to meet objectives, as reflected in the review and acceptance of final as-built surveys. While several on-the-ground adjustments to the design were required to accommodate actual conditions, these adjustments did not compromise the achievement of the project's purpose and objectives.

The Common Tern removal objective has been met based on the implementation of the Interstate Island restoration project. The state agencies will continue to support habitat management and population monitoring at Interstate Island.

Select members of the Avian Technical Team who research and monitor the Interstate Island Common Tern colony remain concerned with the population's recovery and provided the following statement:

> *"The restoration of Interstate Island that was completed in 2021 as part of the BUI effort was a critical first step in protecting the breeding colony of Common Terns. However, avian experts continue to be concerned about the current population trends and overall productivity of breeding terns on Interstate Island. The intent of the restoration was to increase and protect tern breeding habitat and reduce Ring-billed Gull competition for nesting habitat and predation on tern eggs and chicks. At this point of the monitoring (2 year post-restoration), the data indicate that population metrics have not shown a positive response to the restoration efforts as expected. Further, on-going research indicates that breeding adults are foraging in contaminated areas of the St.Louis River Estuary) and as a consequence, chicks hatched on Interstate Island have higher mercury levels compared to breeding populations that are not located in impaired areas, despite the fact that they share the same wintering areas. At the time the BUI delisting package is being created, additional research is being conducted to better understand the source of the mercury, links across the food chain, and to identify high risk foraging areas. The results of the on-going mercury study will provide important information regarding risk of contaminants and impacts to the Common Tern population. The status of the Common Tern breeding population on Interstate Island continues to be precarious. Continued research to determine the role of contaminants on population declines along with consistent monitoring is critical to prevent continued declines of the population and conserve the breeding colony in the St. Louis River Estuary."*

Degradation of the Interstate Island habitat was previously identified by Technical Team members as the greatest limitation to Common Terns in the estuary, leading to restoration and maintenance of the island habitat being chosen as the required action for this target species under BUI 2. Current and future study of limitations facing the estuary tern colony are not required for BUI removal and are referenced in the Future Actions section of this document. These actions may be pursued outside of the AOC program.

# Future Actions

Extensive investments into the recovery of SLRE fish and wildlife populations are being made through the AOC Program. While the scope and duration of the SLRAOC is limited, the need for continued focus on SLRE fish and wildlife populations into the future is necessary to protect AOC investments, research and/or address continued limitations, and generally support the estuary resource. Therefore, Technical Team members provided the following recommendations regarding future planning, management, and study of fish and wildlife in the estuary. These recommendations go beyond the scope of the AOC program and may be pursued under other existing or future programs and are not required for BUI removal. Recommendations listed are in different stages of development, funding and implementation and inclusion in this document does not guarantee a recommendation will be implemented.

## Recommendations for Fish

- Provide annual monitoring updates so the data sets for indicator species can be maintained as current.
- Continue the acoustic tagging study to investigate Lake Sturgeon movement patterns.
- Conduct molecular sexing to evaluate sex ratios of spawning and non-spawning Lake Sturgeon (MNDNR Fisheries Genetics Lab).
- Implement all activities recommended by the 2019 MNDNR Lake Sturgeon Management Plan for the SLRE with emphasis on:
  - Utilize stationary PIT tag receiver downstream of the spawning habitat below the Fond du Lac Dam to passively monitor PIT tagged Lake Sturgeon to collect additional data on the timing, duration and frequency of spawning movements in the SLR.
  - Investigating low female capture rate.
  - Expanding current drift netting efforts to better determine larval drift rates and compare to regional data.
  - Developing outreach and education.

## Recommendations for Birds

- Restore, enhance, and protect marsh bird habitat. Example given: hemi marsh habitat creation and upland bird habitat restoration at Grassy Point (and beyond) and city of Duluth forested areas.
- Shallow marsh habitat is still limited; conduct future survey/research to determine if wetland bird populations are similar to local reference sites
- The Lake Superior Common Tern population should continue to be managed and monitored annually.
- By 2025, reevaluate, assess, and restate (if necessary) Wisconsin's Common Tern Recovery Plan (Matteson 1988) target objectives of 200 breeding pairs for the Duluth Superior Harbor.
- Additional research on mercury exposure in the food web for breeding birds on Interstate Island.
- Continue Common Tern population modeling to refine recovery targets
- Additional research is needed to determine limiting factors for population growth for the Interstate Island Common Tern population and to anticipate long-term impacts of climate change.

- Research and implement plans on Interstate Island to prevent gulls from establishing nests and taking over the tern nesting area this winter or spring
- As recommended by Bracey et al. (2016)
  - The overall low use of the SLRE by shorebirds deserves further study
  - Attracting and reestablishing breeding Great Blue Herons in the SLR will most likely require keeping multiple large undisturbed areas of the appropriate habitat available or, if that is not feasible, possibly installing nest platforms. Many individual Great Blue Herons were observed in the SLR during the avian study period, but no colonies have been located for many years. Several local bird watchers in the area have suggested that a colony site exists in the Superior Municipal Forest. The authors suggest that an effort be made to search for the colony or colonies and provide adequate protection of these sites if possible.
- Continue to maintain piping plover habitat and monitor and protect the species in the SLR AOC.
- Investigate the impacts of contaminants (e.g., mercury) on the survival and productivity of breeding avian species.
- Perform Piping Plover habitat restoration on Minnesota Point
- Conduct toxicology monitoring for waterfowl species to explore impacts to consumption and breeding success
- Conduct post-restoration and long-term monitoring of bird populations in the SLRE

## Recommendations for Mammals

- Conduct mink and muskrat toxicology monitoring to explore potential site-specific impacts to breeding success and fitness.

# Assessment of the BUI Removal Target

Prior to the development of a BUI 2 removal target (2008), objectives (2013), and management actions (2013-19), the 1995 RAP Progress Report provided a list of recommended actions. As discussed in Section 4.2, these recommendations have since been implemented prior to 2013, or translated into six management actions via the 2013 Roadmap to Delisting and subsequent Remedial Action Plan updates. Completion of the six management actions has led the AOC Managing Agencies to consult with the BUI Technical Team to review and evaluate the BUI Removal Target and form a removal recommendation.

The BUI 2 Removal Target requires concurrence from resource managers. This is accomplished by reviewing the final recommendation with the AOC Leadership Team, comprised of lead supervisors from the Fond du Lac Band of Lake Superior Chippewa and three state agencies (Minnesota Department of Natural Resources, Minnesota Pollution Control Agency, and Wisconsin Department of Natural Resources.) Upon gaining concurrence from the AOC Leadership Team, the final recommendation is shared with the Interagency Manager's Team. Managers from the three state agencies comprise this team and provide final concurrence and authorization to submit the final BUI removal package to EPA.

## The Removal Target Has Been Met

The removal target will have been met when:

> *In consultation with their federal, tribal, local, and nonprofit partners, state resource management agencies concur that diverse native fish and wildlife populations are not limited by physical habitat, food sources, water quality, or contaminated sediments.*

The final delisting target for BUI 2 considers the following four key limitations to diverse, native fish and wildlife populations in the SLRAOC:

1. Water Quality
2. Physical Habitat
3. Contaminated Sediment
4. Food Sources

States establish and implement their own water quality standards, which provide benchmarks for assessing the Estuary's water quality limitations. To address the remaining three limitations, resource managers have selected common measures to demonstrate whether conditions support fish and wildlife populations, though formal measurable targets have not been established by the states. Each of these limitations are described further below.

## Water Quality

Historically, degradation of water quality by industrial and urban discharges limited the ability of the aquatic habitat to support macrophytes and other healthy ecological functions. Contaminated sediments, suspended sediments and organic sediments historically discharged into the SLRE resulted in the impairment of fish and wildlife habitat and populations with an overall reduction of biological productivity of the system. Prior to the improvements in wastewater treatment in the late 1970s, water quality and biological investigations characterized the estuary as low in dissolved oxygen and high in total phosphorus and total suspended solids (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1992). By the time the Stage I RAP was developed in 1992, many of these point source discharges were being treated as required by the Clean Water Act, and the primary concerns for the AOC were legacy contamination and historical habitat degradation, as well as excess sediment and nutrient inputs prior to 1972.

Water quality protection is an important responsibility of federal and state government and the 1972 US Clean Water Act established water quality standards to monitor the condition of public waters and assure that waters support their designated uses. SLRAOC partner states adopt their own water quality standards into statute and use them to assess Clean Water Act impairments. The AOC program is tasked with addressing Beneficial Use Impairments and poor water quality resulting from pre-Clean Water Act discharges.

## AOC Management of Water Quality

Water quality in the SLRAOC is addressed by multiple BUIs, as listed below. BUI statuses are current as of the 2020 SLRAOC RAP Update (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2020).

*BUI 1: Fish Consumption Advisories*. The removal target for BUI 1 requires that tissue concentrations of contaminants of concern are not significantly elevated from background. The strategy for BUI removal is closely linked to water quality, as sources of bioaccumulative contaminants of concern are closely linked to legacy pollution in sediments, which influences water quality. All four management actions developed to remove BUI 1 are in progress and are scheduled to be completed between 2022 and 2024.

*BUI 5: Restrictions on Dredging*. The removal target for BUI 5 requires remediation of contaminated sediment project sites in the SLRAOC. Contaminated sediments may contain a variety of toxic and/or bioaccumulative contaminants that are detrimental to water quality. The remediation sites are identified in the management action list for BUI 5; they are scheduled for completion in 2025.

*BUI 6: Excessive Loading of Sediment and Nutrients*. The removal target for BUI 6 requires that nutrient and sediment levels are not impairing water quality and habitat, based on specific criteria for: discharge permit compliance, total phosphorus concentrations in Lake Superior and the SLR, and wastewater overflows to the SLR contributing to organic matter and algal growth. The five management actions developed to remove BUI 6 were completed and the BUI was removed in 2020.

## Continued Water Quality Limitations

As described above, completion of management actions associated with BUIs 1, 5, and 6 will improve SLRE water quality by addressing legacy-based fish consumption advisories, remediating legacy contaminant hot spots, and demonstrating acceptable sediment and nutrient loading. However, the strategy for removing BUI 2 does not require completion of management actions associated with other BUIs. BUI 2 is recommended for removal upon the successful completion of six required management actions with the understanding that water quality limitations may persist within the SLRE and ongoing water quality management is required.

Extensive sampling completed during RAP development and through ongoing research indicate the presence of contaminated sediments outside of targeted remediation projects, originating from a combination of legacy, precipitation, and watershed sources (Janssen, et al. 2021) (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013). While the SLRAOC strategy involves identifying and remediating key legacy hot spots to acceptable levels, it is likely that: 1) there will be a lag between completing remediation projects and observing improvements to water quality and the food web, and 2) it is not feasible for AOC work to address the totality of pollution in the estuary including pollution below action levels, natural sources, and modern sources that are the ongoing responsibility of natural resource management agencies.

Regardless of AOC status, resource management agencies continue to monitor, assess, and regulate SLRE water quality through the Clean Water Act and associated programs to ensure that conditions support the most sensitive species, as intended by applicable water quality standards. For example, a forthcoming Total Maximum Daily Load (TMDL) will determine the mercury reductions needed for lakes and rivers in the St. Louis River watershed to meet state water quality standards and support healthy consumption of fish. Also, county and tribal partners are currently developing “One Watershed, One Plan,” a comprehensive watershed management plan focused on protecting and restoring the St. Louis River watershed. This stakeholder-driven process will guide decisions on how and where to complete projects that restore and protect natural resources. Once the plan is adopted, the watershed will be eligible for funding to complete the prioritized work.

## Physical Habitat

Physical habitat is the ecological setting that supports aquatic life in the estuary. Legacy impairments to physical habitat identified in the SLRAOC include loss through dredging and filling activities and decline in the quality of wetlands from invasion of non-native vegetation. Since 1861, approximately 3,400 acres of wetlands have been lost in the estuary through a combination of dredging and filling; this includes approximately 1,700 acres of shallow, open-water aquatic habitat in St. Louis Bay and Superior Bay that was converted to deep shipping channels (Hollenhorst, et al. 2013).

The 2002 Lower St. Louis River Habitat Plan (Habitat Plan) was developed to establish conservation targets for aquatic and terrestrial habitat in the Estuary using a source/stressor model (St. Louis River Citizens Action Committee 2002). Specific habitat restoration projects were then prioritized to achieve approximately 1,700 acres of restored aquatic habitat (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013).

The Habitat Plan documents the ecological values of shallow sheltered bays, which are the objective of many SLRAOC fish and wildlife habitat improvements. Sheltered bays are critical spawning habitat for many species, including forage and non-game species. The shallow sheltered bay has the highest diversity of fish species and the highest abundance of fish; the bays also provide critical habitat for obligate wetland species.

Physical habitat for birds covers the important life behaviors of nesting, feeding, and migratory stopovers. Terns, Gulls and Plover nest on gravelly substrate and forage in shallow areas. Sandy beaches are stopovers for migrating Plovers. Waterfowl and marsh birds use vegetated wetlands for nesting and feeding.

### AOC Management of Physical Habitat

Physical Habitat in the SLRAOC is addressed by multiple BUIs, as listed below. BUI statuses are current as of the 2020 SLRAOC RAP Update (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2020).

*BUI 5: Restrictions on Dredging.* The removal target for BUI 5 requires remediation of contaminated sediment project sites in the SLRAOC, which may also facilitate improvements to physical habitat where practicable. A “remediation to restoration” approach has been developed and adopted by SLRAOC partners to simultaneously address contaminated sediments and degraded habitat in remediation sites other than the shipping slips. Remediation sites are identified in the management action list for BUI 5; they are scheduled for completion by 2025.

*BUI 9: Loss of Fish and Wildlife Habitat.* The removal target for BUI 9 requires completion of identified aquatic habitat restoration projects, in addition to protection and rehabilitation of habitat in AOC watersheds. Management actions for BUI 9 were heavily influenced by the 2002 Habitat Plan and include 17 restoration sites, plus 275 acres of wild rice restoration at multiple sites. A variety of habitat types will be restored including coastal wetlands, shallow sheltered bays, fish passages, stream banks, cold-water streams, sand dunes, and wild rice beds. As of the 2020 RAP, nine of 21 management actions are complete. The anticipated timeline for BUI 9 removal is 2025.

The BUI 9 removal target also requires an assessment of AOC habitat restoration and protection projects completed in Wisconsin prior to the 2013 Roadmap. As summarized in a 2015 WNDR report, these efforts included over 17,600 acres of habitat protection, over 345 acres of restoration (including 50 acres aquatic), 60,000 tons of contaminated sediment removed from Newton Creek and Hog Island, and invasives control throughout the estuary (Wick 2015).

Though not a BUI 9 removal requirement, MNDNR prepared a 2020 companion report documenting AOC habitat restoration and protection efforts completed in Minnesota prior to the 2013 Roadmap. The accomplishments summarized in this report include ten protection projects and 22 restoration projects totaling approximately 27,170 acres (Collins 2020).

### Continued Physical Habitat Limitations

As described above, completion of management actions associated with BUI 5 and BUI 9 will improve physical habitat at key locations in the SLRE by remediating contamination and restoring habitat. Where needed to improve physical habitat for key species (i.e. Piping Plover and Common Tern), additional habitat restoration projects were identified as BUI 2 management actions. While these projects were strategically chosen to maximize SLRAOC outcomes, it is understood that: 1) there will likely be a lag between completing habitat restoration projects and documenting associated benefits to fish and wildlife populations, and 2) it is not feasible for the AOC program to address the totality of impacts to physical habitat in the SLRE and continued management and improvement is the ongoing responsibility of natural resource agencies and partners outside the AOC program. An example of continued management is "One Watershed, One Plan," a comprehensive watershed management plan focused on protecting and restoring the St. Louis River watershed. Lead by county and tribal partners, this stakeholder-driven process will guide decisions on how and where to complete projects that restore and protect natural resources. Once the plan is adopted, the watershed will be eligible for funding to complete the prioritized work.

Recognizing that management actions associated with BUIs 5 and 9 will complete the AOC-wide objective for addressing physical habitat, the strategy for removing BUI 2 does not require completion of management actions associated with other BUIs. BUI 2 is recommended for removal upon the successful completion of six required management actions with the understanding that some physical habitat limitations may persist within the SLRE and ongoing management and improvement of fish and wildlife habitat will continue. The Future Actions section provides recommendations to better understand and address continued limitations outside the AOC program.

## Contaminated Sediments and Food Sources

These two conditions are assessed together since the legacy challenge for food sources, as identified in the 1992 RAP, is the potential contamination of the food chain in locations with elevated contaminants

(Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 1992). There is no clear documentation on how the various constituent units of the Duluth-Superior area handled their solid and liquid wastes prior to the 1970s, but it has also been established that a number of industries discharged directly and indirectly into the river or bay. Consequently, a number of sites within the AOC contain legacy pollutants from historical contamination with chemicals or toxic waste products.

At the time of AOC listing, restrictions on dredging was a beneficial use that was clearly identified as impaired in the SLRAOC. Sediments in many parts of the AOC exceeded guidelines developed by regulatory agencies to characterize in-place sediments and contained a variety of toxic or bioaccumulative contaminants, which have been shown to cause adverse effects to aquatic and terrestrial organisms. In addition, serious economic and social consequences were thought to be imposed upon resource users through special dredging requirements and obligations for long-term sediment containment.

## AOC Management of Contaminated Sediments and Food Sources

Sediment contamination in the AOC contributes directly or indirectly to eight of the nine BUIs (BUI 6: Excess Loading of Sediment and Nutrients is the exception); remediation of contaminated sediments is an obvious focus of AOC restoration efforts, not only from an ecological standpoint but also from the standpoint of stakeholder concern.

Legacy-contaminated sediments and food sources in the SLRAOC are addressed by multiple BUIs, with the two primary BUIs listed below. BUI statuses are current as of the 2020 SLRAOC RAP Update (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2020).

*BUI 1: Fish Consumption Advisories.* The removal target for BUI 1 requires that tissue concentrations of contaminants of concern are not significantly elevated from background. The strategy for BUI removal is closely linked to sediment, as sources of bioaccumulative contaminants of concern are closely linked to legacy pollution in sediments. All four management actions developed to remove BUI 1 are in progress and are scheduled to be completed between 2022 and 2024.

*BUI 5: Restrictions on Dredging*. The removal target for BUI 5 involves assessment and remediation when state action levels are exceeded of twenty-three contaminated sediment project sites in the SLRAOC. Sediment contaminant levels were investigated and quantified through several research efforts. The SLRAOC was divided into Sediment Assessment Areas (SAAs) to establish a common framework for assessing and displaying sediment contaminant data. Staff from MPCA and WDNR determined the need for remedial action at specific locations across the AOC. Remediation of contaminated sediments above action levels, as well as other necessary restorative actions, must be evaluated, designed, and implemented in support of any identified ecological endpoint objectives.

## Continued Contaminated Sediment & Food Source Limitations

As described above, completion of management actions associated with BUIs 1, and 5 will improve the SLRE food web by addressing legacy-based fish consumption advisories and remediating legacy contaminant hot spots. However, the strategy for removing BUI 2 does not require completion of management actions associated with other BUIs. BUI 2 is recommended for removal upon the successful completion of six required management actions with the understanding that some limitations to the food web may persist within the SLRE and ongoing management is required beyond the AOC program.

Extensive sampling completed during RAP development and through ongoing research indicate the presence of contaminated sediments below action levels outside of targeted remediation projects, originating from a combination of legacy, precipitation, and watershed sources (Janssen, et al. 2021)

(Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources 2013). While the SLRAOC strategy involves identifying and remediating key legacy hot spots to reduce exposure risk, it is likely that: 1) there will be a lag between completing remediation projects and observing improvements to the food web, and 2) it is not feasible for AOC work to address the totality of pollution in the estuary including legacy pollution below action levels, natural sources, and modern sources that are the ongoing responsibility of natural resource management agencies.

Regardless of AOC status, resource management agencies continue to monitor, assess, and regulate potential food web contaminant sources. The Future Actions section provides recommendations to better understand and address continued limitations.

# The Removal Objectives Have Been Addressed

Removal of BUI 2 will be justified when it is shown that key native species populations of fish and wildlife are present and not limited by the legacy impairments referenced by the removal target. This is demonstrated by addressing the specific removal objectives for the fish and wildlife species listed below. Removal objectives were established to help identify key species, establish specific management actions, and guide AOC mangers toward achieving the removal target. All removal objectives have been met except the Lake Sturgeon objective. Species monitoring has shown the Lake Sturgeon specific removal objective metric has not yet been met, however resource managers have completed additional studies that justify BUI removal.

## Fish

The following BUI 2 objectives were developed for target native and invasive fish species:

- Walleye gillnet catch per unit effort is maintained at or above 5.0 per lift with a proportional stock density between 30 and 60 in at least 50% of years surveyed since 2000.
- Muskellunge trap net CPUE is maintained at or above 1.0 per lift in at least 50% of years surveyed since 1997.
- Document an increasing trend of two to five year old Lake Sturgeon captured in summer index nets, with at least two index values greater than 2.0 per gillnet lift.
- An analysis of historical data that shows the Ruffe, and invasive species, is not inhibiting the native fish population.

CONCLUSION: Populations of Walleye and Muskellunge in the St. Louis River are meeting or exceeding the established objectives and invasive Ruffe are no longer inhibiting native fish populations. Lake Sturgeon populations are not currently meeting the BUI 2 objective, however justification to remove the BUI has been shown through additional study of legacy contaminants. Lake Sturgeon are not accumulating legacy contaminants at levels that impact reproduction and are likely limited by factors outside of the AOC program's focus. Lake Sturgeon recovery is part of agency program goals that extend beyond the AOC program's scope and timeline.

Based on evaluation of completed management actions, resource managers have reached consensus that SLRAOC fish populations are not limited by legacy impacts to water quality, physical habitat, contaminated sediment, or food sources at levels that require additional management actions under BUI 2. Continued efforts to remediate contaminated sediments and restore habitat under other BUIs will further benefit native fish populations in the estuary.

## Wildlife

The following BUI 2 removal objectives were established for target wildlife species:

- Piping Plover nesting habitat is created within the SLRAOC.
- Common Tern nesting habitat at Interstate Island is restored and state agencies continue to support habitat management and population monitoring there.
- Great Blue Heron and Bald Eagle presence is recorded during one or more nesting seasons since 1997.
- Wetland bird species are surveyed and compared with 1979 survey results.
- A survey of semi-aquatic mammals in the estuary verifies that the status of small mammal species in the St Louis River Area of Concern is sufficient to remove the beneficial use impairment.

CONCLUSION: Removal objectives for wildlife were achieved. Species including Great Blue Heron and Bald Eagle met targets, while wetland bird species at Remediation to Restoration sites were surveyed and found to have greater abundance and similar species richness when compared to reference sites and similar species richness when compared to historical surveys. Four species of semi aquatic mammals were surveyed and found to be similar to reference populations. Habitat restoration projects targeting Piping Plover and Common Tern were implemented to address legacy habitat loss, which includes long-term monitoring and maintenance.

Based on evaluation of completed management actions, resource managers have reached consensus that SLRAOC wildlife populations are not limited by legacy impacts to water quality, physical habitat, contaminated sediment, or food sources at levels that require additional management actions under BUI 2. Continued efforts to manage critical nesting habitat, remediate contaminated sediments and restore habitat under other BUIs will further benefit native wildlife populations in the estuary.

## Public Involvement Process

Many types of public involvement activities are conducted as part of the SLRAOC program. Some are specific to projects and BUIs and others are related to the SLRAOC program more broadly and they are too numerous to be mentioned here. Three specific activities fall in the public involvement realm for this BUI:

1. The activities associated with the BUI 2 technical teams (see Table 2 for the current members and their affiliations). The technical teams' members assisted the SLRAOC Coordinators with activities associated with reaching the BUI 2 removal target, including: making recommendations on restoration project details, data collection and analyses, reviewing the findings, and providing input on the removal package.
2. The process to obtain public input on the BUI removal package. A thirty-day public comment period about the BUI 2 removal recommendation was held from March 28, 2022, through April 26, 2022. The draft removal document was placed on MNDNR's St. Louis River web site. A public meeting was held on April 14, 2022. Appendix I contains public input received and responses to comments.
3. Additional outreach. A presentation about the BUI 2 removal recommendation was made at the St. Louis River Summit on March 1, 2021. About 285 people attended the Summit.
4. SLRA Letter of Support. The St. Louis River Alliance is the designated citizens' action committee for the SLRAOC. Information about the BUI 2 removal recommendation was presented to the members of the Alliance's Board on March 16 and September 12, 2022. As a result of their review, a letter of support for the removal of BUI 2 was submitted on behalf of the St. Louis River Alliance (see **Appendix I**).

## Removal Recommendation

Throughout the process of implementing management actions for this BUI, the SLRAOC staff consulted with the BUI technical teams, stakeholders, and federal staff assigned to this BUI. They analyzed the collected data in the context of the SLRAOC RAP BUI targets, actions, and removal objectives for the Degraded Fish and Wildlife Populations BUI.

The results of the scientific assessments, the input from the BUI Technical Teams, and the support of the St. Louis River Alliance and other stakeholders form the basis for this removal recommendation. Accordingly, the MPCA and the WDNR, with the concurrence of the MNDNR and the Fond du Lac Band of Lake Superior Chippewa, recommend that the USEPA concur with this recommendation to remove the Degraded Fish and Wildlife Populations BUI from the SLRAOC.

While BUI 2 removal is based on the successful completion of its listed management actions, continued benefits to fish and wildlife populations will also be realized through activities associated with other SLRAOC BUIs. These management actions for other BUIs, which include investigations, remediation, and restoration throughout the estuary, will positively impact the following legacy impairments to fish and wildlife populations: physical habitat, food sources, water quality, and contaminated sediments.

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# Appendix A

# 2016 St. Louis River AOC Bird Assessment Report

# St. Louis River AOC R2R Support Projects: Ecological Monitoring and Assessment (CR#6403) Final Report

Objectives 1 & 2, Task D:
Sample Birds at R2R and Reference Locations
Compare bird use in historical (1976-1979) and recent (2010-2015) survey periods

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## Executive Summary

The goal of this project was to provide a contemporary assessment of bird use of the St. Louis River Freshwater Estuary (SLR), a designated Great Lakes Area of Concern (AOC), located in the extreme western end of Lake Superior, Minnesota-Wisconsin, USA. These data and analyses will be used to assess the current status of the beneficial use impairment (BUI) on 'Degraded Fish and Wildlife Populations' that exists in the SLR. Removal of such a BUI is contingent upon evidence that native populations of fish and wildlife are not limited by physical habitat, food sources, water quality, or sediment contamination. To provide a perspective on the BUI, the project consisted of two broad objectives: 1) summarize and compare contemporary baseline data gathered on bird use at sites planned for restoration and reference sites with reduced disturbances within the SLR, and 2) synthesize and compare these contemporary bird use data with similar data gathered for sites in the late 1970s within the SLR.

Sites selected for contemporary sampling in objective 1 were those identified as the Remediation-to-Restoration (R2R) sites by the Minnesota Pollution Control Agency (MPCA), while reference sites were selected based on locations that were most relevant for comparison with R2R sites. A total of 10 R2R sites were selected and up to 10 potential reference sites were originally identified. Based on a review of these potential reference sites, five were selected as most appropriate for comparison with the R2R sites, but data for the remaining five sites were also included in the dataset as "additional sites." Sampling of these sites were grouped into breeding and migration (Spring and Fall) time periods. Bird counts during the breeding season were completed from fixed point count locations and gathered from 0.5 hr before sunrise to 4.5 hr after sunrise. Each count was 15 min in duration, which included use of playback recordings of hard to detect species, such as rails, and all individuals seen or heard from the point were recorded. Counts during the migration seasons were also recorded from a fixed point location from sunrise to early afternoon. Each count was 10 min in duration and all individuals seen or heard from the point were recorded. Depending on accessibility, sites were sampled either from the shoreline or by boat. All locations of observations, including flyovers, were estimated on aerial photo field sheets and digitized in ArcGIS®. For ease of interpretation, all observations were classified into 16 unique species groups based on taxonomy, life history, and physiological similarities; however, species of special concern (e.g., Common Tern and Piping Plover) were also considered separately.

Grassy Point, a Remediation-to Restoration site in the St. Louis River Estuary. Photo credit: A. Bracey

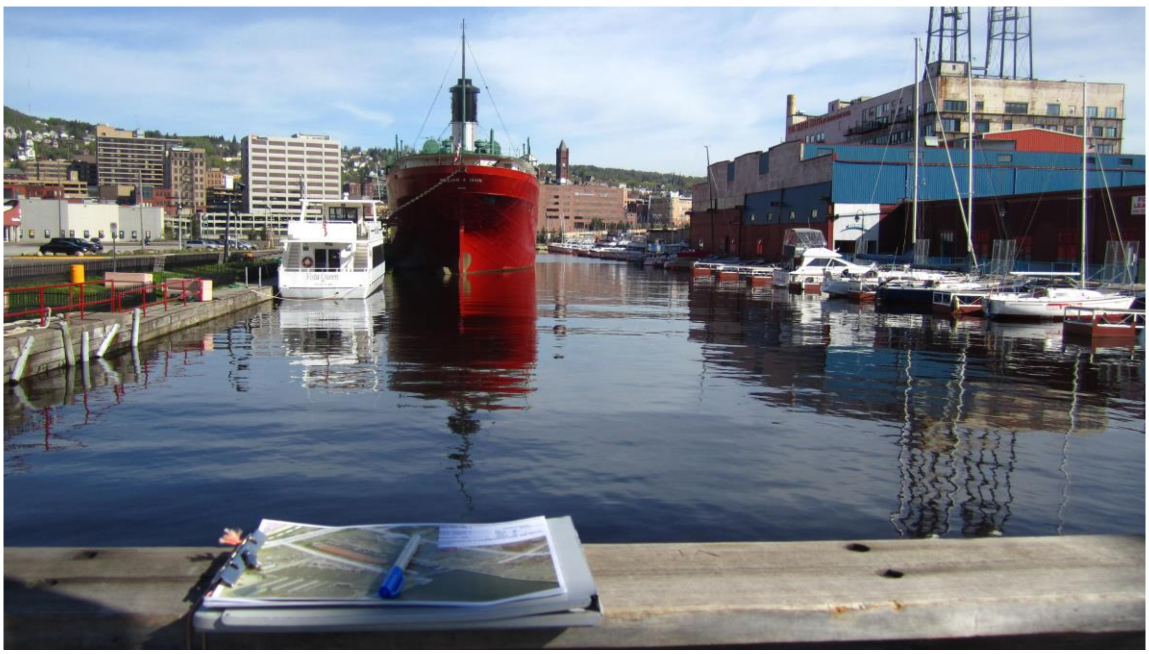
For objective 2, we identified 10 sites that matched the areas sampled in objective 1 with those sampled using similar techniques from 1976 to 1979 in the SLR. Major considerations included similar areas sampled, similar sampling techniques, close phenological time periods of sampling, and a representative distribution across the SLR. Although time of sampling varied in the 1970 counts, the major objective of both sampling regimes was to do a complete count of all bird species and individuals detectable within the sample area, therefore, we believe the methods are comparable. In addition, one of the co-authors, GJN, was involved in gathering the 1970s data and he verified their comparability at the ten sites selected. Data gathered in the 1970s were digitized from the original field sheets which were also gathered on aerial photos.

The focus of the analyses were to compare R2R with reference sites (Objective 1) and compare historical (1970s) with contemporary surveys (2010s); a difference of over 30 years (Objective 2). Various statistical techniques were used to determine overall differences in community composition, species richness, and to document differences for species of special interest or conservation concern, including the Least Bittern, Great Blue Heron, Great Egret, Piping Plover, Black Tern, Common Tern, and Forster's Tern. Groups of species of specific interest to the BUI included waterfowl, waterbirds, rails, raptors (e.g., Peregrine Falcon and Bald Eagle), or songbirds of concern such as Sedge and Marsh Wren. Special attention was also made with reference to potentially problematic or nuisance species such as Canada Goose and Ring-billed Gull.

Interstate Island Wildlife Management Area. Photo credit: D. Hamilton

**Objective 1**. A total of 117,235 individual bird observations of 177 species were recorded during the migratory and breeding seasons at all sites sampled from 2010-2015. Each site was sampled at least 10 times, including four during spring migration, four during fall migration, and two during the breeding season. Sites varied in size from 10 to 480 acres. Overall, cumulative species richness (SR) and water-obligate species richness did not differ between R2R and reference sites when all sites were pooled. However, site-specific comparisons provided mixed results. The 40$^{th}$ Avenue West R2R site had higher cumulative and water-obligate only SR, compared with its reference site (Little Pokegema Bay). The 21$^{st}$ Avenue West complex also had higher water-obligate SR compared with its reference site (Little Pokegema Bay). In contrast, the R2R sites Minnesota Slip and Slip C had low overall use by birds and lower SR for all species and water-obligate species only compared with their reference site (Minnesota Point). The R2R sites Cedar Yard Bay had lower SR when all species were included compared with its reference site (North Bay) and the R2R site Perch Lake had lower SR for water-obligate species compared with its reference site (Rask Bay). In general, the R2R and reference sites have substantial overlap in their respective bird communities, primarily because of the high variability in bird species found at R2R sites. This was especially due to the high use by many bird species at the 40th Avenue West and 21st Avenue West R2R sites. These data provide a solid baseline to assess future changes in these communities over time. Any future changes can be assessed within each respective area as a natural experiment in progress.

The SLR is a complex system which has been influenced by substantial human activity. The cumulative impacts of current activities, which vary from recreation to heavy industrial activity, and legacies of past activities (logging, shipping, development) influence both migratory and breeding bird communities in the estuary. Extensive and constant human activity at the Minnesota Slip and Slip C sites renders use of these areas by birds as extremely low. In contrast, the 40th Avenue West site is a very active industrial zone, but it is also very heavily used by birds. Despite the industrial activity, human activity levels in this area are very low because of its remoteness, plus it has a diverse habitat base with wetlands, open water, and shrubby, forested riparian areas. Cedar Yard Bay also has high potential for use by birds as it shares the low human activity and isolation with 40th Avenue West, despite major differences in industrial activity.



Minnesota Slip located in Canal Park. Photo credit: A. Bracey

**Objective 2**. A total of 16,911 individual bird observations of 133 species (historical) and 11,042 individual bird observations of 132 species (recent) were included in the analysis of historical and recent bird use in the estuary. Of the water-obligate species only, 13 were unique to the historical surveys and seven were unique to recent surveys. The number of surveys included ranged from 4-17 and was dependent on the number of replicate samples available between the time periods. Sites varied in size from 35 to 664 acres but were matched to be the same size in paired comparisons. Similar with the R2R and reference comparisons, we found no significant differences in SR between historical and recent surveys when all sites were pooled. However, comparisons of site-specific SR indicated significant differences (when all species were included and water-obligate species only) with higher cumulative SR for three historical sites compared with recent sites at 20th Avenue West, 27th Avenue West, and Spirit Lake West. However, recent counts at 40th Avenue West were higher than historical counts (water-obligate species only). In contrast with comparisons between R2R and reference sites, recent and historical sites did not overlap as extensively. Based on community composition, the most influential water-obligate species contributing to differences between the historical and recent surveys were the extremely high Canada Goose populations and lower Blue-winged Teal, American Coot, and Lesser Scaup populations observed during recent surveys compared with historic counts.

Interpretation of the differences between historical and recent surveys requires consideration of how populations of bird species have changed over the past 30 years independently of the changes that have occurred in the SLR. Many waterfowl are still common and widespread in the region and across North America and generally waterfowl populations have increased over the past five decades (NABCI 2016), while some have changed substantially – both increasing and decreasing. In contrast to many areas of North America that have continued to see reductions in water quality and expansion of agriculture and human populations, the SLR has improved in water quality with the addition of WLSSD and agriculture is a negligible issue in the region. In addition, DDT was banned in the early 1970s and overall contaminant levels have declined in exposure for aquatic-associated species. All of these factors have an effect on population levels for each bird species and interpretation of these interacting effects is beyond the scope of this report.

In general, comparison of recent and historical waterfowl populations indicate that Canada Geese have increased substantially in the SLR, but Wood Duck, Blue-winged Teal, and Northern Pintail were observed less frequently. Population changes in Canada Geese and Blue-winged Teal are consistent with regional changes in their populations over the past 40+ years, but Wood Duck and Northern Pintail have declined despite regional population increases. Other water-dependent bird species also indicate mixed results in these comparisons. Double-crested Cormorants have increased in the SLR compared with historical counts, while Common Loon, American Bittern, Great Blue Heron, American Coot, and Black Tern have declined. All of these changes, except for Common Loon, are consistent with regional population trends in these species over the past 40+ years. Fewer observations of Common Loon in the SLR compared with historical counts are inconsistent with their increases over the past 40+ years. However, the total number of observations of this species in the SLR was relatively small in both historical and recent periods.

Overall shorebird use was relatively low in the recent sampling periods. The lower number of observations during the recent sampling period for Killdeer, Spotted Sandpiper, and Wilson's Snipe are consistent with declining regional population trends for these species. However, the fewer observations for Black-bellied Plover, Pectoral Sandpiper, and Semipalmated Sandpiper compared with historical counts do not have regional population trends available for comparison. All three of these species are migrants that nest in the Arctic tundra. The overall lack of use by shorebirds in the SLR is a concern and deserves further study. It is unclear whether availability of suitable, breeding or stopover habitat is an issue in the SLR for shorebirds compared with the past.

Based primarily on observations, raptorial species, especially Bald Eagle and Peregrine Falcon, populations have increased substantially in the SLR compared with the historical period (1970s) when neither species nested. Several pairs of both species have nested or attempted to nest in the SLR over the past 5-10 years. Increases in their populations have largely been attributed to the banning of DDT and focused management such as reintroduction programs for Peregrine Falcons and nesting habitat protection for the Bald Eagle. The population recovery of these species represents a massive success story in wildlife species conservation. Analyses of two rail species, Sora and Virginia, plus two wren species, Sedge and Marsh, provided no significant differences in historical or current population levels in the SLR.

## Background

The SLR was designated an AOC under the 1987 Great Lakes Water Quality Agreement, and efforts towards delisting this area are in progress. The MPCA is currently developing a comprehensive, long-term plan to delist the SLR AOC under a grant from the U.S. Environmental Protection Agency (EPA) and other project partners (MPCA 2013). The potential removal of beneficial use impairment (BUI) #2: 'Degraded Fish and Wildlife Populations' is contingent upon evidence that native populations of fish and wildlife are not limited by physical habitat, food sources, water quality, or sediment contamination (MPCA 2013). Documenting avian use throughout the AOC is fundamental to prioritizing project areas, establishing objectives, and successfully implementing R2R project activities. By documenting avian diversity and abundance, in conjunction with sediment, benthic, fish, vegetation and water quality sampling, it will be possible to better define biotic and abiotic relationships that collectively indicate ecological condition.

Our primary objective (Objective 1) of this report was to summarize the baseline data collected at priority sites selected for potential future restoration (R2R) and their corresponding reference sites in the SLR AOC (2010-2015) with a focus on the richness and abundance of species that use water as their primary habitat (i.e., waterfowl, waterbirds, shorebirds, rails). The secondary objective (Objective 2) was to compare historical (1978-1979) and recent (2010-2015) data on bird use at sites that were surveyed during both sampling periods. Both objectives involve comparisons, objective 1 contrasts R2R and reference sites using contemporary data, while objective 2 compares contemporary data with those sampled in the 1970s; albeit the latter with slightly different methodologies.

We will discuss how these data can be used to address BUI targets and provide a summary of species of particular interest (e.g., Piping Plover, Common Tern) identified by the Minnesota and Wisconsin Departments of Natural Resources (MDNR and WDNR, respectively). This information will be summarized in the context of abundance in the SLR as well as trends across each species range.

## Methods

### *Objective 1. Documenting bird use in R2R and Reference sites in the SLR AOC*

#### *Sample Locations*

To document bird use in the SLR AOC, we sampled 10 R2R sites and 5 reference sites. Reference sites were chosen based on location within the estuary and size of site (acres). Reference sites were also considered less impacted by human disturbance (e.g., farther from industrial activity, non-hardened shoreline). In addition to the five reference sites, during the first sampling period in 2013, we sampled 5 additional locations, considered potential reference sites, to determine locations that would be the most appropriate reference sites (i.e., met the criteria above and were accessible; Table 1: '*Additional Sites*'). Minnesota Point was chosen as the reference site for R2R sites: Minnesota Slip and Slip C. Little Pokegema Bay was selected as the reference site for R2R sites: 21$^{st}$ Avenue West, 40$^{th}$ Avenue West, and Grassy Point. Spirit Lake East was chosen as the reference site for R2R site: Spirit Lake West. North Bay was chosen as the reference site for R2R sites Cedar Yard Bay and Kingsbury Bay, and Rask Bay was chosen as the reference site for R2R sites: Mud Lake and Perch Lake.

In 2014-2015, we sampled a subset of all 10 R2R sites and 5 reference sites (Table 1, Fig. 1). R2R sites selected for sampling in 2014-2015 were selected primarily for logistical reasons: 1) to ensure that reference sites would be sampled in the same year as their corresponding R2R sites when possible, and 2) for consistent accessibility (i.e. boat vs. land surveys). Three of the R2R sites (21st Avenue West, 40th Avenue West, and Grassy Point) had been previously sampled by researchers at the Natural Resources Research Institute (NRRI) in 2010-2012 (Host et al. 2012, 2013). Because of the extensive data available for these sites, we limited our sampling to the fall of 2013. Cedar Yard Bay was also only sampled in fall 2013 and May-June 2014 because restoration activities were initiated in 2014. In lieu of sampling at Cedar Yard Bay, MPCA requested that we survey Kingsbury Bay in 2015 (Table 1, '*R2R Sites*').

Polygons for R2R sites were provided from the MPCA project officer to ensure all sampling occurred within appropriate site boundaries (Table 1, Fig 1). For sites where polygons were not provided, we created polygons using ArcGIS® software by Esri, version 10.2.2. Sampling density within each site was dependent on size and accessibility of each site (Table 1). Data were collected from each site either by boat or from land. A total of 12 surveys were conducted at most survey point from 2013 to 2015. Surveys were a minimum of seven days apart. Of the 12 surveys, five occurred during the spring migration (March-May), five during fall migration (August-November), and two during the breeding season (June). A few dates were logistically unfeasible for sampling because of unsafe conditions for water travel such as high winds or river was iced over. However, we attempted to conduct those remaining surveys in another year. Detailed methodology can be found in the MPCA Bird Sampling QAPP 'CR#6403: Migration and Breeding Bird Distribution and Abundance' as well as Host et al. (2012 & 2013).

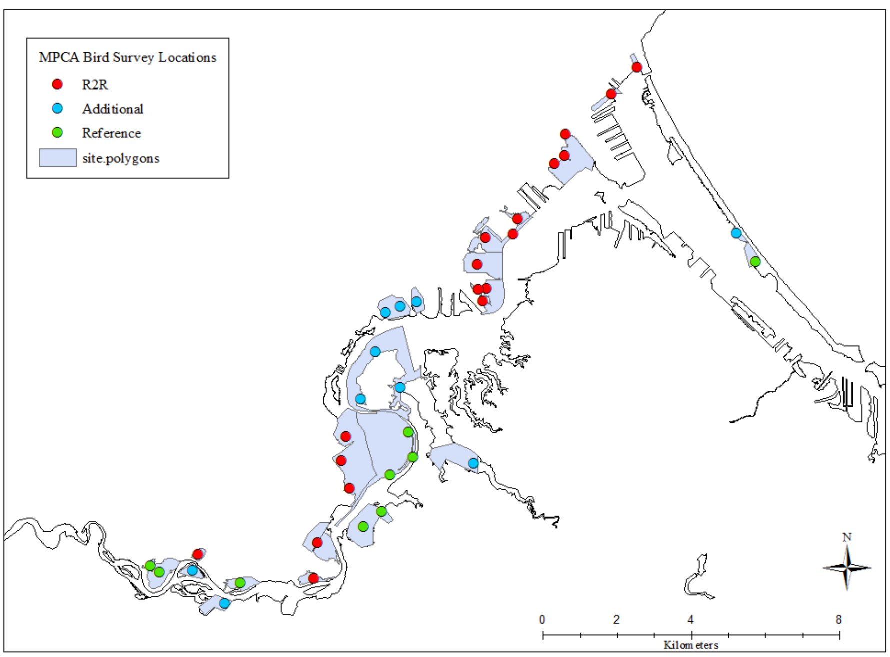



**Figure 1.** Location of sites surveyed in the St. Louis River Area of Concern (SLR AOC). A total of 10 R2R sites, 5 Reference sites, and 6 additional sites were surveyed (2010-2015).

**Table 1.** Location of sites surveyed in the St. Louis River Area of Concern (SLR AOC). A total of 10 R2R sites, 5 reference sites, and 6 additional sites were surveyed (2010-2015). For each site, the location of the centroid of each site polygon is provided in NAD_1983_UTM_Zone_15N, area of site surveyed (acres), number of survey points, and year(s) surveyed as well as number of replicates per season/year (Reps).

| **Site Name** | **Location (UTM)** | | **Area (acres)** | **# of Points** | **Survey Year(s) and (Reps)** | | |
|---|---|---|---|---|---|---|---|
| | **x coordinate** | **y coordinate** | | | **Fall** | **Spring** | **Summer** |
| ***R2R Sites*** | | | | | | | |
| Minnesota Slip | 568932.0987 | 5181429.415 | 10 | 1 | 2013(4), 2015(5) | 2014(4), 2015(5) | 2014(2), 2015(2) |
| Slip C | 568146.893 | 5180336.653 | 28 | 1 | 2013(4), 2014(4), 2015(1) | 2014(4) | 2014(2) |
| 21st Avenue West Complex* | 567234.7284 | 5178629.812 | 201 | 3 | 2012(10), 2013(5) | 2012(7) | 2012(2) |
| 40th Avenue West Complex* | 565095.9511 | 5176399.507 | 321 | 4 | 2010(11), 2013(5) | 2011(8) | 2011(5) |
| Grassy Point Complex* | 565084.7939 | 5174889.351 | 115 | 3 | 2010(11), 2013(4), 2014(3) | 2011(8), 2014(4) | 2011(5), 2014(2) |
| Spirit Lake West | 561344.4729 | 5170475.449 | 250 | 3 | 2013(5), 2014(4) | 2014(5) | 2014(2) |
| *Kingsbury Bay**** | 562495.9847 | 5174547.985 | 36 | 2 | 2015(6) | 2015(5) | 2015(2) |
| Mud Lake East & West | 560373.3906 | 5168013.218 | 123 | 2 | 2013(5), 2015(6) | 2014(4), 2015(5) | 2014(2), 2015(2) |
| Cedar Yard Bay** | 560196.2854 | 5167027.719 | 38 | 1 | 2013(5) | 2014(4) | 2014(2) |
| Perch Lake | 557172.1823 | 5167700.839 | 21 | 1 | 2013(5), 2014(3), 2015(3) | 2014(4) | 2014(2) |
| ***Reference Sites*** | | | | | | | |
| Minnesota Point | 572077.3183 | 5176012.388 | 37 | 1 | 2014(2), 2015(6) | 2014(4), 2015(5) | 2014(2), 2015(2) |
| Little Pokegema Bay | 561697.4797 | 5168511.646 | 189 | 2 | 2013(3), 2014(4) | 2014(4) | 2014(2) |
| Spirit Lake East | 562218.5785 | 5170744.569 | 480 | 3 | 2013(5), 2014(3) | 2014(4) | 2014(2) |
| North Bay | 558324.5876 | 5166891.417 | 60 | 1 | 2013(1), 2015(6) | 2014(4), 2015(5) | 2014(2), 2015(2) |
| Rask Bay | 556137.9853 | 5167213.874 | 98 | 2 | 2013(5), 2014(4), 2015(2) | 2014(4) | 2014(2) |
| ***Additional Sites*** | | | | | | | |
| Southworth Marsh | 571731.2127 | 5176570.682 | 18 | 1 | – | – | 2014(2) |
| Clough Island | 562075.786 | 5172539.183 | 82 | 3 | 2013(6) | 2014(1) | 2014(1) |
| Pokegema Bay | 564586.7451 | 5170216.221 | 70 | 2 | 2013(3) | – | – |
| *Stryker Bay**** | 563066.2283 | 5174737.414 | 42 | 1 | 2015(6) | – | – |
| Weasel Bay | 557604.1257 | 5166309.617 | 59 | 1 | 2013(4) | – | – |
| Horseshoe Bay | 556984.2951 | 5167216.649 | 36 | 1 | 2013(5) | 2014(1) | – |

*Site was sampled for entire year in previous study and therefore only surveyed in Fall 2013

**Site was undergoing remediation/restoration activities during sampling period and therefore only surveyed in Fall 2013 and once in May and June 2014

***Site included as an additional sample, not because it was considered a potential reference site but because of a request by MPCA

### ***Data Collection***

Due to differences in the seasonal distribution of species, sampling protocols varied between breeding (June) and migration (spring/fall) surveys. Surveys were designed to obtain a complete count of bird use in each survey location (site), during each visit. This technique was used in the late 1970s by Niemi et al. (1979; *see methods of Objective 2*). For all surveys, we used unlimited distance counts at designated point locations within each site and counted all species identified by both visual and auditory observations. All bird observations were identified to specific locations on aerial photo field sheets and digitized in ArcGIS® (e.g., Fig. 2). Accuracy was approximately 30 m in open water and 20 m near or on shore. Observation type was based on behavior and included 1) singing, 2) calling, 3) drumming (woodpeckers), 4) visual observation, or 5) flyover (i.e. species not actively using study area). Flyover observations are included in raw data and total species list, but excluded from site summaries and analyses. Species were classified into 16 unique groups based on taxonomy and physiological similarities as well as individual species groups of interest. These groups are as follows: gulls, waterfowl, waterbird, raptor, shorebird, blackbird, songbird, corvid, pigeon, woodpecker, dove, rail, hummingbird, pheasant, grouse, and invasive. Grouping individuals based on taxonomy and physiological similarities is useful to simplify mapping and to identify specific groups of species of interest (e.g., water associated species).

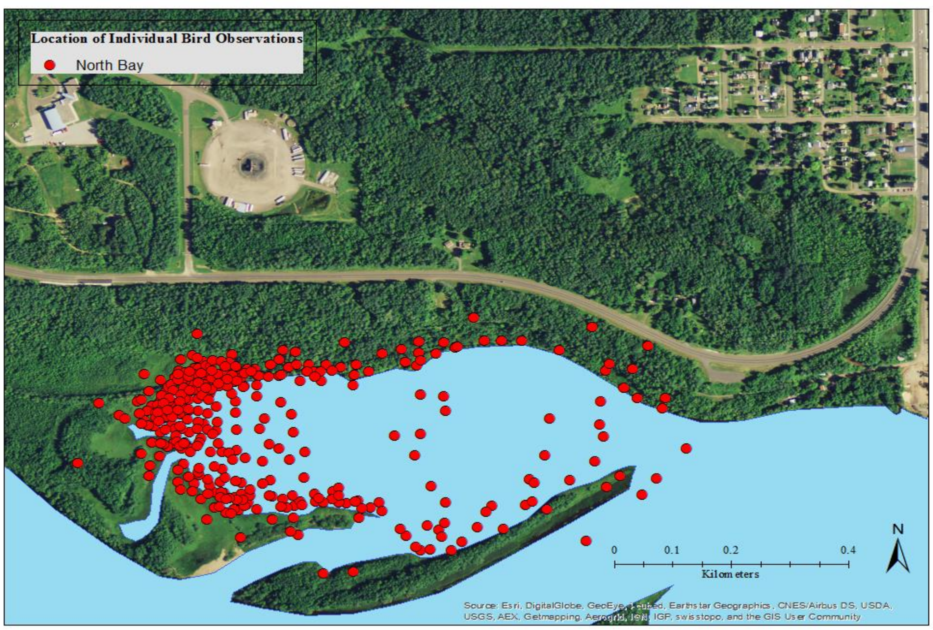



**Figure 2.** An example of the digitized spatial locations of individual bird observations in North Bay (reference site).

Spotting Scope used to identify birds on the water. Photo credit: A. Bracey

Surveys conducted during migration were completed from a fixed point location within each site for 10 min or, in rare situations when no birds were present; the count was abated early. During breeding season surveys, we extended our point counts to 15 minutes, which incorporated the use of playbacks, a series of recordings of secretive marsh bird calls, to target this group of hard to detect species. The broadcast calls consisted of 30 seconds of vocalization followed by 30 seconds of silence in the following order for each of six focal species: Least Bittern, Sora, Virginia Rail, a mixture of American Coot and Common Gallinule, and Pied-billed Grebe. Surveys were conducted from 0.5 hr before sunrise to 4.5 hr after sunrise in the breeding season and from sunrise until early afternoon during spring and fall migration; all completed during suitable weather conditions (e.g., minimal wind or precipitation). Detailed sampling methodology can be found in the MPCA Bird Survey Standard Operating Procedures document (Appendix A). For objective 1, we restricted analysis to include four spring surveys, two breeding surveys, and five fall surveys from each site. For sites with more than 11 samples, we randomly removed surveys by year and month, thereby making sample size equal and comparable between sites.

### *Objective 2: Comparison of bird use in the SLRAOC between historical (1976-1979) and recent (2010-2015) time periods*

#### *Sample Locations*

St. Louis River historical bird survey data were obtained using original data sheets from three projects conducted in the 1970s: Phase I and Phase II of the Assessment of Habitat Types and Bird Populations in the Duluth-Superior Area (Niemi et al. 1977, Davis et al. 1978) and Distribution and Relationships of Habitats and Birds in the St. Louis River Estuary (Niemi et al 1979a). The original field data sheets were used, rather than the summarized data found in the appendices, to ensure that dates and locations of bird observations matched with those of the recent MPCA surveys (Table 2). Only historical data sheets with dates closely corresponding to dates of the recent MPCA surveys (e.g., within the same month) were used. Since the survey areas involved in this analysis were of varying shapes and sizes, ten sites that could be closely matched between the historical and recent survey data were selected. Site polygons were created in ArcGIS® to represent the locations where historical and recent data were collected (Fig. 3).

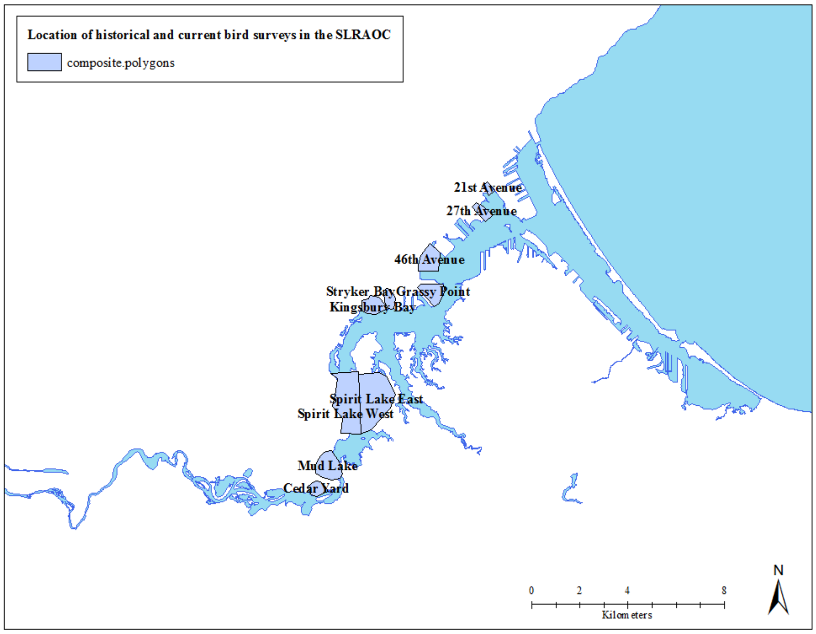



**Figure 3.** Location of historical (1976-1979) and recent (2010-2015) bird surveys in the St. Louis River Area of Concern (SLR AOC). Each of these 10 sites has both historical and recent bird data associated with them and are included in the temporal comparison.

The number of surveys included ranged from 4-17 and was dependent on the number of replicate samples available between the time periods, sites also varied in size from 35 to 664 acres (Table 2). Survey time and total effort were included whenever possible for historical surveys, but it was necessary to include historical surveys with missing effort information due to a large number of surveys that did not include this information. Co-author GJN was involved in these surveys and he confirmed that the overall objective of the surveys in the 1970s and the contemporary surveys was to obtain a complete count of the individual birds within a specific area. The historical data sheets included specific locations and area on aerial photographs that could be matched with the contemporary survey areas. Hence, we felt these comparisons were reasonable.

**Table 2.** Location of ten sites where historical (1977-1979) and recent (2010-2015) surveys were conducted. The location at each site is represented by the centroid of the polygon [Location (UTM)]. The total area in acres and the number of surveys included in the analyses are also provided for each location. At each site, the number of surveys was equal for both historical and recent surveys.

| Site Name | Location (UTM) | | Area (acres) | # of Surveys |
|---|---|---|---|---|
| | x coordinate | y coordinate | | |
| 20th Avenue West | 567191.2395 | 5179207.368 | 35 | 10 |
| 27th Avenue West | 566922.8469 | 5178263.966 | 76 | 12 |
| 40th Avenue West | 564737.3271 | 5176323.484 | 190 | 14 |
| Cedar Yard Bay | 564865.6061 | 5175003.808 | 85 | 9 |
| Grassy Point | 562385.5466 | 5174472.7 | 157 | 8 |
| Kingsbury Bay | 560526.3187 | 5167964.834 | 142 | 10 |
| Mud Lake | 562418.634 | 5170684.202 | 231 | 17 |
| Spirit Lake East | 561342.3492 | 5170581.749 | 664 | 8 |
| Spirit Lake West | 560045.4171 | 5167080.039 | 579 | 11 |
| Stryker Bay | 563055.8593 | 5174736.049 | 74 | 4 |

***Data Collection***

Birds were surveyed during three sampling periods: fall migration - September 1 – December 31; spring migration - January 1 – May 31; and breeding season - June 1 – August 31. We used the following field methods: Waterbirds - by spot checks, transects, spotting with scopes from the bay or lakeshore, and by boat; Shorebirds – by boat and by spot checks at regular intervals (along shoreline when waves were hazardous); Colonial birds – estimated at each visit and, if accessible, site was visited once during breeding season on foot to accurately count all nests, eggs, and young; and all other bird species – by transect counts and modified transects or spot checks where terrain was difficult. For all observations the estimated location of the bird was recorded on maps of the area being surveyed. Surveys covered the area from the Arrowhead Bridge to Lake Superior, including Minnesota and Wisconsin Points, but excluded many industrial, residential, and recreational areas. Surveys at these sites included all open water to 0.25 mi inland from the land-water interface. As with the recent MPCA data, all bird observations were assigned to groups based on species associations.

Assessment of changes in bird species use of the SLR in the late 1970s with the contemporary counts were grouped into several categories based on comparisons of the paired study areas: 1) species observed less in contemporary versus historical counts, 2) species observed less in historical counts versus contemporary counts, and 3) species that are too rare to make confident comparisons between the two periods. In addition, we also consider the changes that have occurred in species populations from 1966 to 2013 using the North American Breeding Bird Survey, commonly known as the BBS (Sauer et al. 2014). We primarily used the changes that have occurred in the regional population defined as Bird Conservation Region 12 (BCR 12) which encompasses northern Minnesota, northern Wisconsin, northern Michigan, and southern/western Ontario – basically the area surrounding the western Great Lakes Region. In some cases where a species population is not assessed sufficiently within this region, we included a broader area of the BBS and used the survey-wide results.

**Data Analyses**

For both objective 1 and objective 2, we were interested in determining changes in community composition between R2R and reference sites (objective 1) and between historical and recent surveys (objective 2). The sample size of R2R and reference sites was unequal (10 and five, respectively), therefore we used the non-parametric Wilcoxon rank sum test to compare the median of the difference between R2R and reference sites for water-obligate species. The sample size of historical and recent surveys was equal and the same locations were surveyed in both time periods, therefore we used a paired t-test to compare means of water-obligate species.

To assess differences in community composition we compared species richness (SR) between communities using the package 'rich' in R, version 3.2.3 (Rossi 2011, R Core Team 2015). Using the function *c2cv*, we were able to compare cumulative richness between locations. This function calculates difference between the values ($d=S_1-S_2$) and compares to $n$ similar differences $d_{rand}$ obtained after randomizing samples between communities. This technique allows us to determine if differences in richness are significant or due to sampling fluctuations (Manly 1997). This function tests observed values of $d$ as compared to the quantiles of the randomized values of a user-fixed probability level (Rossi 2011). For this analysis we used $n$=999 randomizations and a probability level for quantile computations of 0.025 – 0.975.

We calculated dissimilarity among samples (replicate and temporal) to determine if differences in community composition were larger than sampling variation alone. We calculated dissimilarity distances using nonmetric multidimensional scaling (NMDS) in R, using package 'vegan' (Oksanen et al. 2016). We created two-dimensional plots to visualize the dissimilarity distances. To measure the magnitude of change we used the Bray-Curtis distance (Bray and Curtis 1957), which is calculated from differences in species abundance. Because this distance measure uses abundance it can be influenced by large differences in species counts. Therefore, we first standardized the species data by converting species abundance to relative proportions of species across sites. We then transformed the proportions using an arcsine square root transformation. We used hierarchical clustering via Ward's Method on the set of calculated Bray-Curtis dissimilarities. We then used the function *adonis* in package 'vegan' (Oksanen et al. 2016) which calculates analysis of variance using distance matrices. Significance tests use F-tests based on sum of squares from permutations of the raw data.

We also calculated indices of beta diversity with package 'vegan' (Oksanen et al. 2016), using the most commonly used index of beta diversity: $\beta\_w = S/\alpha - 1$, where **S** is the total number of species and **α** is the average number of species per site (Whittaker 1960, 1972) and measured variance in beta diversity between groups using multivariate analysis of variance, with distance matrices describing how variation is attributed to different groups. Calculating the average distance of group members to the group centroid, we determined if variances were different between groups. PCoA axes represented distances between groups, with negative axes being a consequence of using a dissimilarity index other than Euclidean.

To determine which species were driving differences between groups we used the function *simper* (Clarke 1993) in package 'vegan' (Oksanen et al. 2016). *Simper* calculates the contribution of individual species to overall dissimilarity between two groups using Bray-Curtis dissimilarities. This function performs pairwise comparisons of groups and determines the average contribution of each species to the average overall Bray-Curtis dissimilarity, displaying the most important species for each pair of groups (Oksanen et al. 2016).

## Results

A total of 196 bird species were observed in the SLRAOC (1977-2015; Appendix B). Not all of the species included in this list were included in the analysis. Excluded species were those only observed as flyovers or that fell outside of the survey boundaries delineated for the comparison of historical and recent surveys.

### *Objective 1. Documenting bird use in R2R and reference sites in the SLR AOC*

A total of 117,235 individual bird observations and 177 species were recorded during migration and the breeding season in all sites surveyed (2010-2015; Fig. 4). Counts of individuals observed in each group are listed by site (Appendix C).

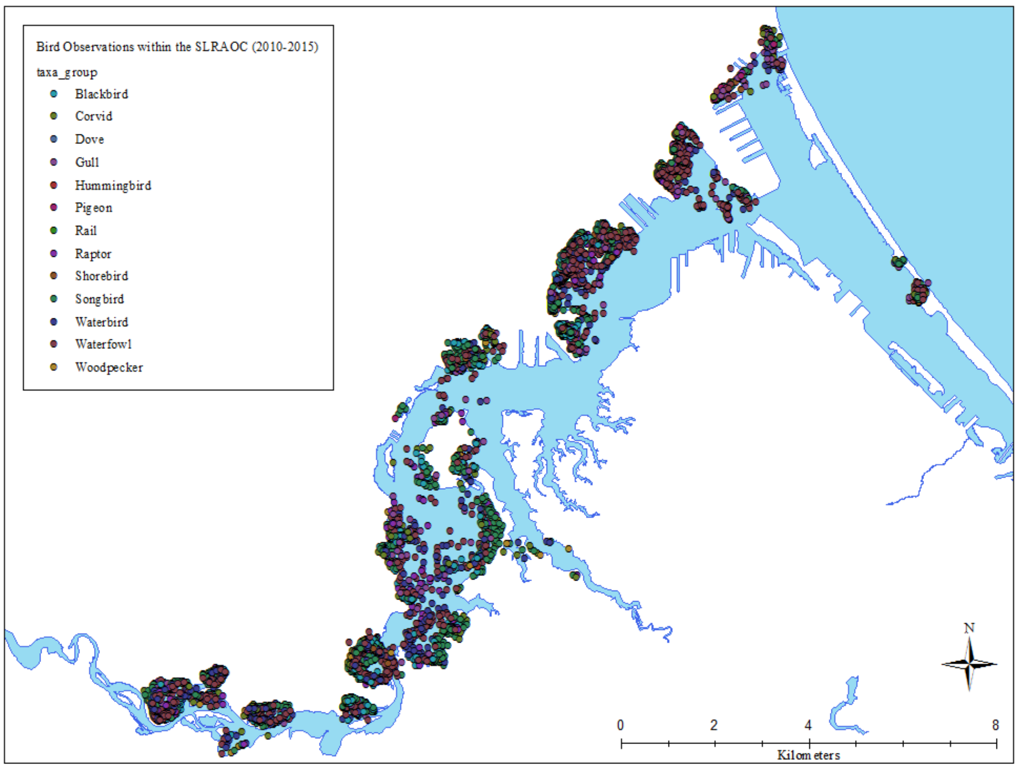



**Figure 4.** All bird observations digitized from aerial field sheets. Observations are based on bird groups.

*Wilcoxon Rank Sum Test*

Based on 11 surveys at each site, waterfowl and shorebirds were observed in R2R sites in greater abundance than reference sites. For waterfowl, there were approximately 880 individuals per R2R site (8,801 individuals/10 sites) and 288 individuals per reference site (1,438 individuals/5 sites; Table 3). Of the waterfowl species observed in R2R sites, 80% were Canada Geese and Mallards. For shorebird species, there were roughly six individuals per R2R site (61 individuals/10 sites) and two individuals per reference site (9 individuals/5 sites). The majority of the shorebird species were observed at the 40$^{th}$ Avenue West complex (72%). There were more waterbirds observed in reference sites than R2R sites, with roughly 67 individuals per R2R site (671 individuals/10 sites) and 386 individuals per reference site

(1,928 individuals/5 sites), the majority of which were observations of American Coot (88%). Rails and wrens were observed in low numbers and there were no differences between R2R and reference sites. There was an average of 1.6 rails per R2R site (16 individuals/10 sites) and 2.8 rails per reference site (14 individuals/5 sites) and an average of 1.5 wrens per R2R site (15 individuals/10 sites) and 0.8 wrens per reference site (4 individuals/5 sites).

Species observed in R2R sites only, included six species of shorebirds: Killdeer, Greater Yellowlegs, Dunlin, Least Sandpiper, White-rumped Sandpiper, and Semipalmated Sandpiper, most of which were observations of single individuals, five species of waterfowl: Gadwall, Northern Shoveler, Greater Scaup, Red-breasted Merganser and Ruddy Duck, and one Sedge Wren (Table 3). There were two species of waterfowl observed in reference sites only (Trumpeter Swan and Northern Pintail) and one shorebird (Pectoral Sandpiper).

**Table 3.** Number of water-obligate species observed in R2R and reference sites. Counts include four spring surveys, two breeding surveys, and five fall surveys from each site. Species absent from R2R or reference sites are highlighted in gray. Species with significantly different population medians between R2R and Reference sites, based on Wilcoxon rank sum test (95% CI), are highlighted in blue.

| | R2R | | | Reference | | |
|---|---|---|---|---|---|---|
| **Species** | **Mean** | **Range** | **Median** | **Mean** | **Range** | **Median** |
| ***Waterfowl*** | | | | | | |
| Canada Goose | 42.45 | 0-493 | 8 | 10.93 | 0-113 | 1 |
| Trumpeter Swan | 0 | 0-0 | 0 | 0.69 | 0-35 | 0 |
| Tundra Swan | 0.31 | 0-32 | 0 | 0.24 | 0-13 | 0 |
| Wood Duck | 0.19 | 0-6 | 0 | 0.09 | 0-3 | 0 |
| Gadwall | 0.05 | 0-2 | 0 | 0 | 0-0 | 0 |
| American Wigeon | 0.33 | 0-20 | 0 | 0.07 | 0-3 | 0 |
| American Black Duck | 0.17 | 0-7 | 0 | 0.02 | 0-1 | 0 |
| Mallard | 20.76 | 0-629 | 0.5 | 3.29 | 0-75 | 0 |
| Blue-winged Teal | 0.18 | 0-8 | 0 | 0.02 | 0-1 | 0 |
| Northern Shoveler | 0.31 | 0-20 | 0 | 0 | 0-0 | 0 |
| Northern Pintail | 0 | 0-0 | 0 | 0.18 | 0-10 | 0 |
| Green-winged Teal | 0.34 | 0-12 | 0 | 0.73 | 0-37 | 0 |
| Canvasback | 1.01 | 0-96 | 0 | 0.04 | 0-1 | 0 |
| Redhead | 3.14 | 0-220 | 0 | 0.45 | 0-12 | 0 |
| ***Waterfowl, cont.*** | | | | | | |
| Ring-necked Duck | 3.02 | 0-179 | 0 | 1.85 | 0-21 | 0 |
| Greater Scaup | 0.07 | 0-8 | 0 | 0 | 0-0 | 0 |
| Lesser Scaup | 1.23 | 0-33 | 0 | 5.09 | 0-56 | 0 |
| Bufflehead | 1.05 | 0-35 | 0 | 1.53 | 0-20 | 0 |
| Common Goldeneye | 0.65 | 0-25 | 0 | 0.4 | 0-15 | 0 |
| Hooded Merganser | 0.42 | 0-14 | 0 | 0.18 | 0-8 | 0 |
| Common Merganser | 0.42 | 0-16 | 0 | 0.09 | 0-2 | 0 |

*continued on next page*

| | R2R | | | Reference | | |
|---|---|---|---|---|---|---|
| **Species** | **Mean** | **Range** | **Median** | **Mean** | **Range** | **Median** |
| ***Waterfowl, cont.*** | | | | | | |
| Red-breasted Merganser | 0.18 | 0-7 | 0 | 0 | 0-0 | 0 |
| Ruddy Duck | 0.03 | 0-2 | 0 | 0 | 0-0 | 0 |
| Horned Grebe | 0.15 | 0-7 | 0 | 0.16 | 0-2 | 0 |
| ***Waterbirds*** | | | | | | |
| Common Loon | 0.05 | 0-2 | 0 | 0.04 | 0-1 | 0 |
| Pied-billed Grebe | 0.52 | 0-10 | 0 | 0.36 | 0-7 | 0 |
| Ring-necked Grebe | 0.07 | 0-4 | 0 | 0.05 | 0-1 | 0 |
| Double-crested Cormorant | 1.99 | 0-50 | 0 | 2.22 | 0-69 | 0 |
| American White Pelican | 0.02 | 0-2 | 0 | 0.07 | 0-4 | 0 |
| American Bittern | 0.02 | 0-1 | 0 | 0.02 | 0-1 | 0 |
| Great Blue Heron | 0.33 | 0-5 | 0 | 0.04 | 0-1 | 0 |
| Green Heron | 0.03 | 0-1 | 0 | 0.02 | 0-1 | 0 |
| American Coot | 0.95 | 0-29 | 0 | 30.38 | 0-496 | 0 |
| Common Tern | 1.63 | 0-75 | 0 | 1.24 | 0-16 | 0 |
| Belted Kingfisher | 0.17 | 0-2 | 0 | 0.04 | 0-1 | 0 |
| ***Shorebirds*** | | | | | | |
| Killdeer | 0.1 | 0-4 | 0 | 0 | 0-0 | 0 |
| Spotted Sandpiper | 0.32 | 0-10 | 0 | 0.04 | 0-2 | 0 |
| Greater Yellowlegs | 0.01 | 0-1 | 0 | 0 | 0-0 | 0 |
| Dunlin | 0.01 | 0-1 | 0 | 0 | 0-0 | 0 |
| Least Sandpiper | 0.02 | 0-2 | 0 | 0 | 0-0 | 0 |
| White-rumped Sandpiper | 0.01 | 0-1 | 0 | 0 | 0-0 | 0 |
| Pectoral Sandpiper | 0 | 0-0 | 0 | 0.11 | 0-6 | 0 |
| Semipalmated Sandpiper | 0.07 | 0-3 | 0 | 0 | 0-0 | 0 |
| Wilson's Snipe | 0.02 | 0-2 | 0 | 0.02 | 0-1 | 0 |
| ***Rails*** | | | | | | |
| Virginia Rail | 0.06 | 0-2 | 0 | 0.05 | 0-1 | 0 |
| Sora | 0.08 | 0-2 | 0 | 0.15 | 0-2 | 0 |
| ***Wrens*** | | | | | | |
| Sedge Wren | 0.11 | 0-3 | 0 | 0 | 0-0 | 0 |
| Marsh Wren | 0.03 | 0-2 | 0 | 0.07 | 0-2 | 0 |

*Species Richness*

Based on cumulative species richness (SR), there were no significant differences between R2R and reference sites when all sites were pooled. This included richness calculated for all species observations as well as for water-obligate species only (i.e., rails, waterbirds, waterfowl, and shorebirds; Fig. 5). However, comparisons of site-specific SR indicated significant differences in cumulative SR for four R2R sites: Minnesota Slip, Slip C, 40th Avenue West and Cedar Yard Bay (Fig. 6). For comparisons of water-obligate species only, five R2R sites had significant differences: Minnesota Slip, Slip C, 21th Avenue West, 40th Avenue West, and Perch Lake (Fig. 7). At sites where differences were significant, cumulative SR was higher in the reference site with the exception of 40th Avenue West, which had higher SR when all species were included as well as when only water-obligate species were included. 21st Avenue West also had significantly higher SR of water-obligate species compared to its reference site ($p \leq 0.05$).

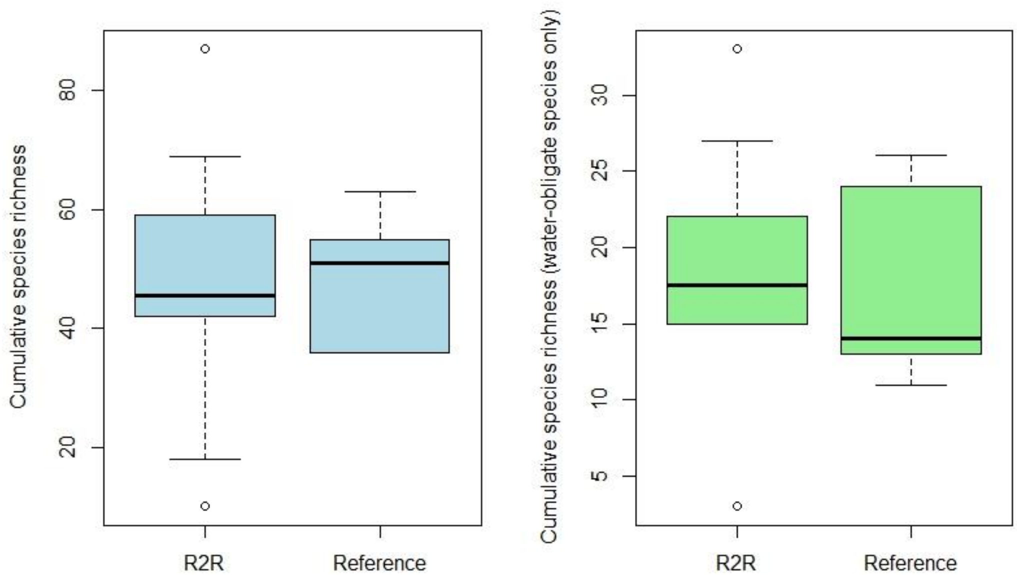



**Figure 5.** Cumulative species richness (SR) calculated using all species observations for R2R and reference sites (left) and for water-obligate species only (right).

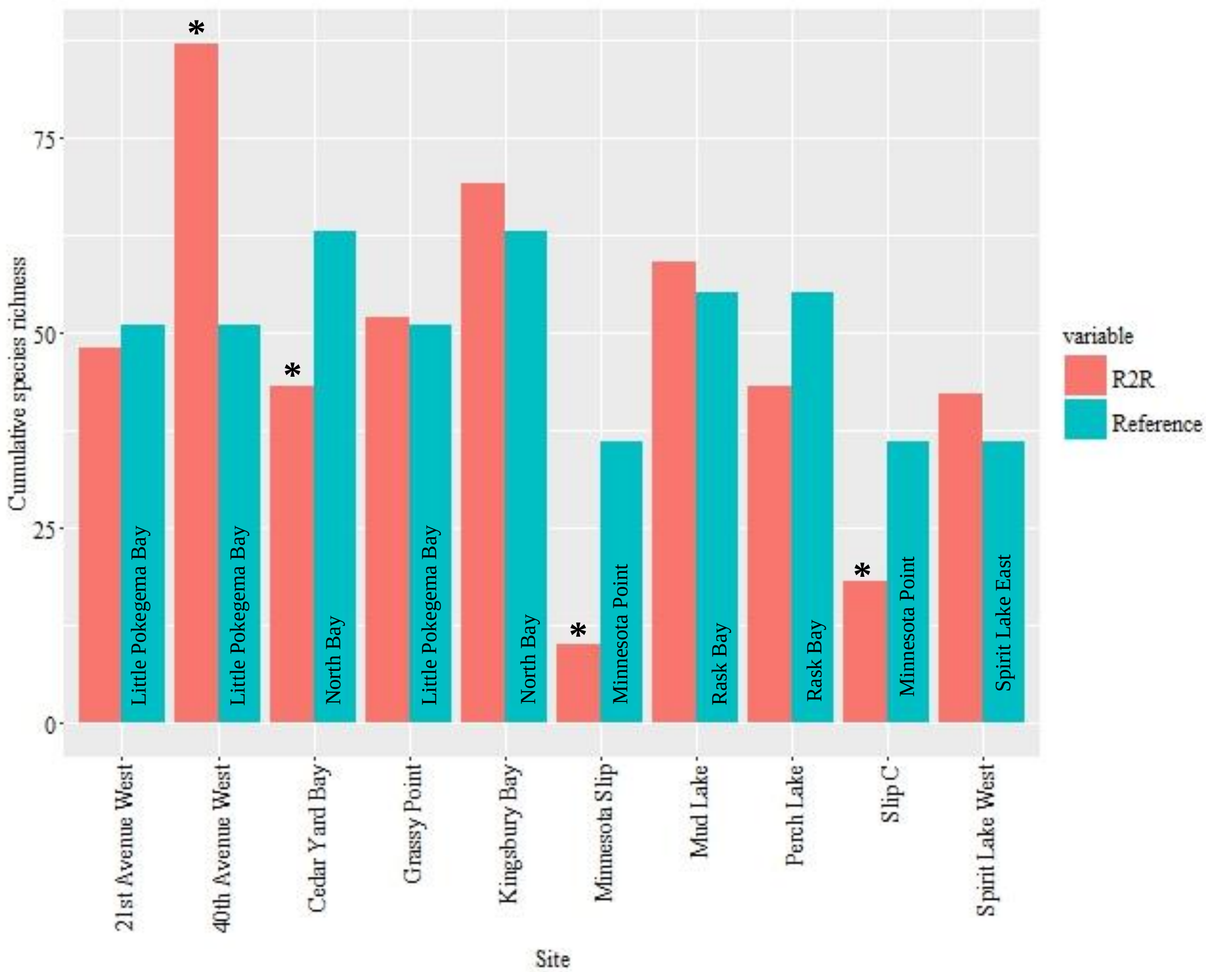



**Figure 6.** For each R2R site, the cumulative species richness (SR) of all species observed relative to their corresponding reference site. Asterisks represent sites where differences in SR were significant at $p \leq 0.05$.

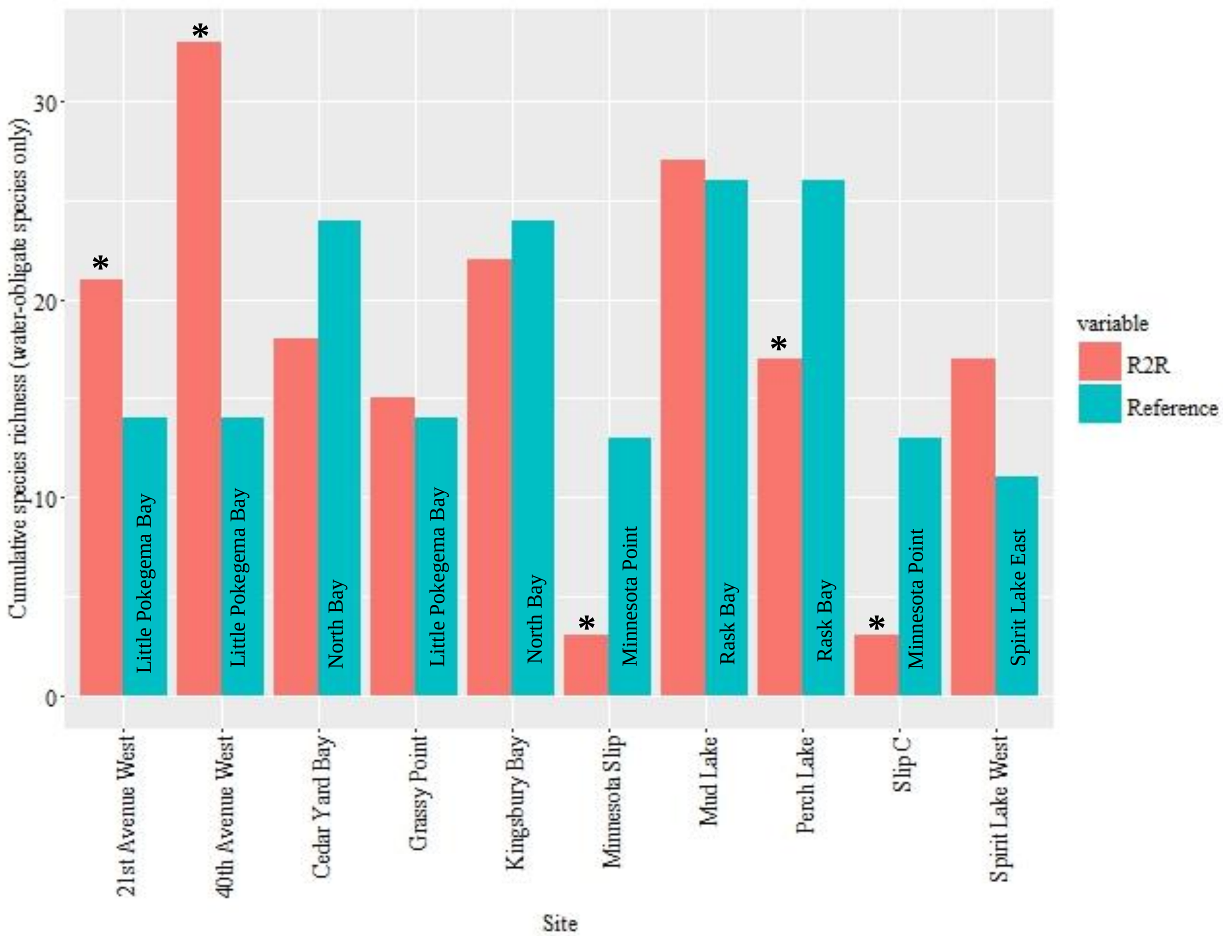



**Figure 7.** For each R2R site, the cumulative species richness (SR) of water-obligate species only relative to their corresponding reference site. Asterisks represent sites where differences in SR were significant at $p \leq 0.05$.

*Dissimilarity*

To visualize differences in water-obligate communities we first calculated dissimilarity indices using NMDS and then used hierarchal clustering based on those dissimilarity indices. These analyzes suggest that sites tend to cluster based on site type (i.e., reference or R2R). For instance, reference sites were more similar to other reference sites and R2R sites were more similar to other R2R sites (Fig. 8). There were no significant differences between R2R and reference sites based on beta diversity (Fig. 9) and the variability of species within the R2R sites completely encompassed that of the reference sites.

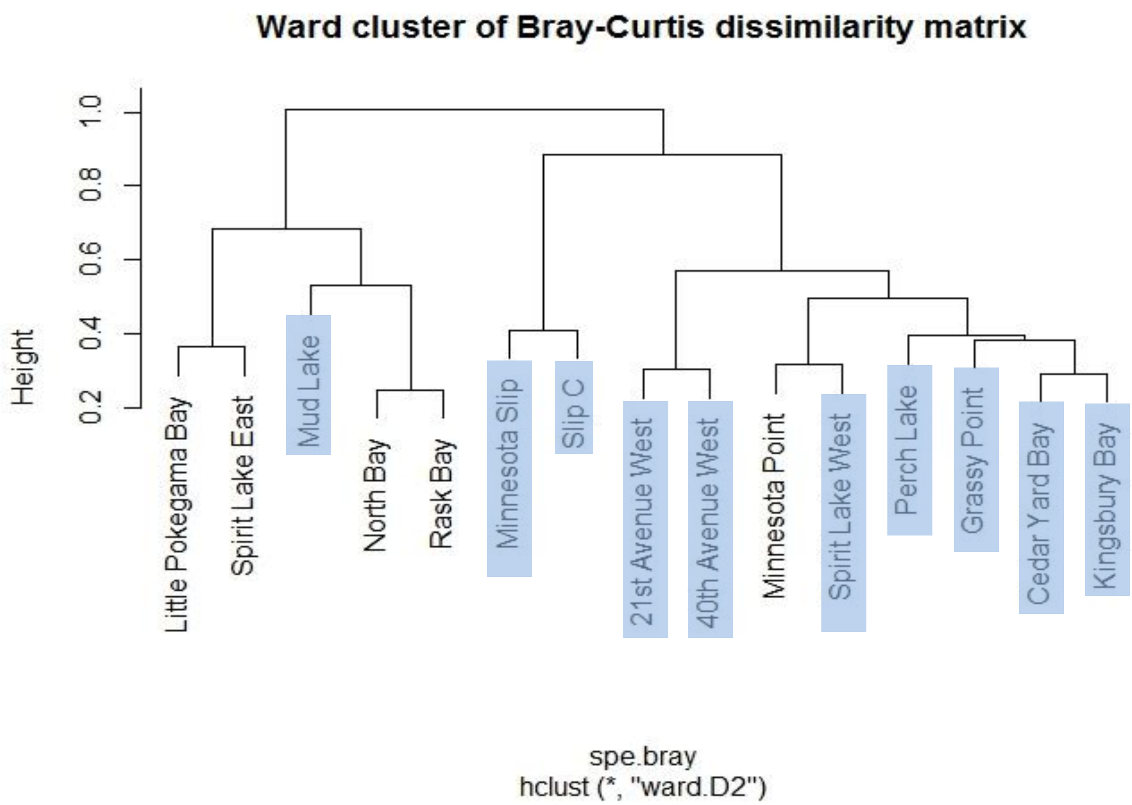



**Figure 8.** Cluster dendogram showing the relationship between R2R (blue) and reference sites in the St. Louis River based on a Bray-Curtis dissimilarity matrix using water-obligate species only.

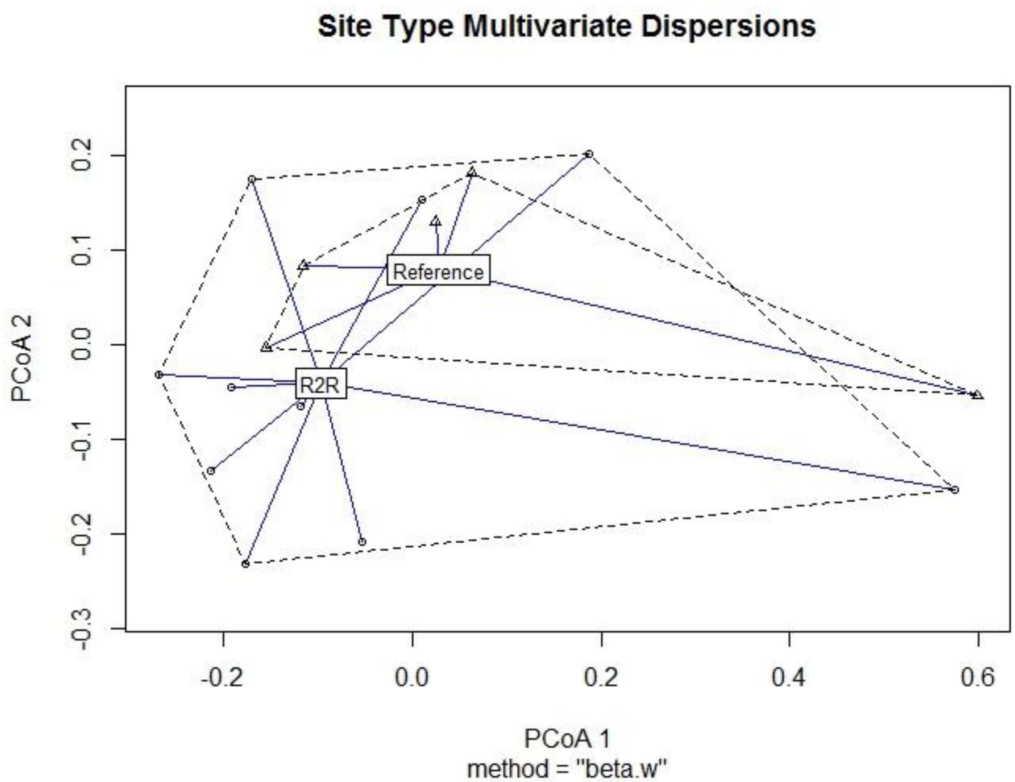



**Figure 9.** Measure of multivariate dispersion between site types (R2R and Reference), for water-obligate species only, for 11 surveys (4-spring, 2-breeding, and 5-fall) at 10 R2R sites and 5 Reference sites. Dispersions between groups (site type) were not significantly different.

### *Objective 2: Bird use in the SLR AOC a comparison of historical (1976-1979) and recent (2010-2015) data*

A total 196 species were observed in the historical and recent surveys at the 10 sites sampled during both survey periods in the SLR (Appendix B). There were 16,911 individual bird observations of 133 species (historical) and 11,042 individual bird observations of 132 species (recent) included in the analysis. Of these species, 29 were observed in recent surveys only and 31 were observed in historical surveys only (Appendix D). However, many of the species unique to either historical or recent surveys were observed in small numbers (<5 individuals).

*Paired t-test*

For water-obligate species only, 13 were unique to the historical surveys and seven were unique to recent surveys (Table 4). For many of these species we include Breeding Bird Survey (BBS) trend estimates (1966-2013; Sauer et al. 2014) for Bird Conservation Region 12 (BCR-12) 'Boreal Hardwood Transition' (http://www.nabci-us.org/bcr12.html). If trends were not available for BCR-12, we list survey-wide trend estimates and denoted them with an asterisk (Table 4). Trend estimates were not available for species with breeding ranges that fall outside of the BBS survey area and were denoted with *NA*. We list BBS trend estimates for species if they are significantly increasing (+) or decreasing (-). There were no trend estimates for many of the shorebird species and some waterfowl and waterbirds due to the extent of their breeding ranges. Caveats associated with these trend estimates are provided in detail in Sauer et al. (2014) and should be considered when interpreting trends for any particular species.

Water-obligate species observed in recent counts but not in historical counts include the following: Trumpeter Swan, Canvasback, Greater Scaup, Red-breasted Merganser (-), Red-necked Grebe, American White Pelican (+), Great Egret (+*). In contrast, species found in historical surveys but not in recent surveys included American Bittern, Least Bittern, Black-crowned Night Heron (-), Black-bellied Plover, Semipalmated Plover, Killdeer (-), Solitary Sandpiper, Sanderling, Dunlin, White-rumped Sandpiper, Wilson's Phalarope, Black Tern (-) and Forster's Tern. All of these species are uncommon, rare, or very rare in the SLR and, therefore, the lack of observation of many of these species is partly due to their rarity. Observations of all species ($n \geq 10$ individuals) that were present in historical surveys but absent in recent surveys include: Black Tern (-), Purple Martin (-), and Yellow-headed Blackbird (-). In contrast, species observed in recent counts but not in historical counts included: Canvasback, Red-necked Grebe, Peregrine Falcon, Common Raven (+), and Black-and-white Warbler. Many of the species unique to the historical or recent surveys were either present in other areas of the estuary, were not included in analysis, or were observed in very low numbers.

**Table 4.** Species summaries for historical and recent surveys. The mean, range, and median are provided for each species within each group (*Waterfowl, Waterbirds, Shorebirds, Rails, and Wrens*). Species with significantly different population means between historical and reference sites, based on paired t-tests (*df*= 102, 95%CI), are highlighted in blue. When available, North American Breeding Bird Survey trends were provided from Sauer et al. (2014). Trends represent %change/year for the Bird Conservation Region 12 (BCR-12), the northern Great Lakes region of North America. When trend estimates were not available for BCR-12 we used survey-wide estimates (represented by an asterisk). When trend estimates were not available for a particular species it is denoted *NA*. Trend estimates judged significant based on 95% credible intervals are indicated in (red = significant decreases) and (green = significant increases).

| | Historical | | | Recent | | | |
|---|---|---|---|---|---|---|---|
| **Species** | **Mean** | **Range** | **Median** | **Mean** | **Range** | **Median** | **BBS Trend (%/yr)** |
| ***Waterfowl*** | | | | | | | |
| Canada Goose | 1.51 | 0-56 | 0 | 30 | 0-232 | 10.5 | 17.7 |
| Trumpeter Swan | 0 | 0-0 | 0 | 0.06 | 0-6 | 0 | *NA* |
| Tundra Swan | 0.15 | 0-8 | 0 | 0.31 | 0-32 | 0 | *NA* |
| Wood Duck | 0.64 | 0-13 | 0 | 0.03 | 0-1 | 0 | 3.1 |
| Gadwall | 0.07 | 0-5 | 0 | 0.05 | 0-2 | 0 | 2.65* |
| American Wigeon | 1.21 | 0-18 | 0 | 0.53 | 0-20 | 0 | -2.64* |
| American Black Duck | 0.23 | 0-10 | 0 | 0.23 | 0-4 | 0 | -3.93 |
| Mallard | 14.89 | 0-110 | 5.5 | 15.47 | 0-135 | 4 | 0.99 |
| Blue-winged Teal | 12.41 | 0-254 | 3.5 | 0.35 | 0-13 | 0 | -3.8 |
| Northern Shoveler | 0.07 | 0-4 | 0 | 0.5 | 0-20 | 0 | 9.1 |
| Northern Pintail | 0.35 | 0-12 | 0 | 0.06 | 0-2 | 0 | 4.41 |
| Green-winged Teal | 1.75 | 0-80 | 0 | 0.4 | 0-9 | 0 | -1.6 |
| Canvasback | 0 | 0-0 | 0 | 1.11 | 0-96 | 0 | 0.99* |
| Redhead | 0.08 | 0-3 | 0 | 3.45 | 0-220 | 0 | 0.86* |
| Ring-necked Duck | 5.86 | 0-212 | 0 | 2.9 | 0-179 | 0 | 2.69 |
| Greater Scaup | 0 | 0-0 | 0 | 0.08 | 0-8 | 0 | *NA* |
| Lesser Scaup | 7.78 | 0-210 | 0 | 1.96 | 0-58 | 0 | -4.46 |

*continued on next page*

| Species | Historical Mean | Historical Range | Historical Median | Recent Mean | Recent Range | Recent Median | BBS Trend (%/yr) |
|---|---|---|---|---|---|---|---|
| ***Waterfowl, cont.*** | | | | | | | |
| Bufflehead | 0.51 | 0-22 | 0 | 0.92 | 0-35 | 0 | 2.78* |
| Common Goldeneye | 3.77 | 0-180 | 0 | 0.68 | 0-20 | 0 | 0.86 |
| Hooded Merganser | 0.28 | 0-6 | 0 | 0.49 | 0-19 | 0 | 4.29 |
| Common Merganser | 0.65 | 0-27 | 0 | 0.25 | 0-9 | 0 | 1.75 |
| Red-breasted Merganser | 0 | 0-0 | 0 | 0.24 | 0-7 | 0 | -7.71 |
| Ruddy Duck | 0.06 | 0-3 | 0 | 0.02 | 0-2 | 0 | 0.81* |
| ***Waterbirds*** | | | | | | | |
| Common Loon | 0.1 | 0-3 | 0 | 0.01 | 0-1 | 0 | 1.38 |
| Pied-billed Grebe | 0.29 | 0-4 | 0 | 0.44 | 0-10 | 0 | 0.42 |
| Horned Grebe | 0.15 | 0-12 | 0 | 0.14 | 0-7 | 0 | -1.61* |
| Red-necked Grebe | 0 | 0-0 | 0 | 0.05 | 0-4 | 0 | 3.36 |
| Double-crested Cormorant | 0.04 | 0-3 | 0 | 0.55 | 0-7 | 0 | 11.03 |
| American White Pelican | 0 | 0-0 | 0 | 0.19 | 0-17 | 0 | 9.08 |
| American Bittern | 0.04 | 0-1 | 0 | 0 | 0-0 | 0 | -0.47 |
| Least Bittern | 0.03 | 0-2 | 0 | 0 | 0-0 | 0 | 7.57 |
| Great Blue Heron | 0.73 | 0-6 | 0 | 0.32 | 0-4 | 0 | -1.73 |
| Great Egret | 0 | 0-0 | 0 | 0.03 | 0-1 | 0 | 2.11* |
| Green Heron | 0.1 | 0-5 | 0 | 0.03 | 0-2 | 0 | 0.31 |
| Black-crowned Night-Heron | 0.02 | 0-1 | 0 | 0 | 0-0 | 0 | -13.52 |
| American Coot | 26.65 | 0-318 | 0 | 1.05 | 0-29 | 0 | -4.88 |
| Black Tern | 0.51 | 0-18 | 0 | 0 | 0-0 | 0 | -3.58 |
| Common Tern | 0.66 | 0-17 | 0 | 0.36 | 0-11 | 0 | 0.09 |
| Forster's Tern | 0.02 | 0-2 | 0 | 0 | 0-0 | 0 | -6.1 |
| Belted Kingfisher | 0.32 | 0-8 | 0 | 0.23 | 0-2 | 0 | -1.57 |
| ***Shorebirds*** | | | | | | | |
| Black-bellied Plover | 0.2 | 0-8 | 0 | 0 | 0-0 | 0 | *NA* |
| American Golden-Plover | 0.19 | 0-9 | 0 | 0.01 | 0-1 | 0 | *NA* |
| Semipalmated Plover | 0.15 | 0-6 | 0 | 0 | 0-0 | 0 | *NA* |
| Killdeer | 2.39 | 0-28 | 0 | 0 | 0-0 | 0 | -4.05 |
| Spotted Sandpiper | 0.73 | 0-14 | 0 | 0.25 | 0-5 | 0 | -4.99 |
| Solitary Sandpiper | 0.02 | 0-2 | 0 | 0 | 0-0 | 0 | -11.55 |
| Lesser Yellowlegs | 0.26 | 0-9 | 0 | 0.06 | 0-3 | 0 | -4.76* |
| Stilt Sandpiper | 0.02 | 0-2 | 0 | 0.01 | 0-1 | 0 | *NA* |
| Sanderling | 0.02 | 0-1 | 0 | 0 | 0-0 | 0 | *NA* |
| Dunlin | 0.35 | 0-20 | 0 | 0 | 0-0 | 0 | *NA* |
| Least Sandpiper | 0.09 | 0-4 | 0 | 0.03 | 0-1 | 0 | *NA* |
| White-rumped Sandpiper | 0.02 | 0-2 | 0 | 0 | 0-0 | 0 | *NA* |
| Pectoral Sandpiper | 0.21 | 0-3 | 0 | 0.02 | 0-1 | 0 | *NA* |
| Semipalmated Sandpiper | 2.77 | 0-120 | 0 | 0.03 | 0-3 | 0 | *NA* |
| Wilson's snipe | 0.34 | 0-13 | 0 | 0.02 | 0-2 | 0 | -1.43 |
| Wilson's Phalarope | 0.15 | 0-15 | 0 | 0 | 0-0 | 0 | *NA* |
| ***Rails*** | | | | | | | |
| Virginia Rail | 0.06 | 0-3 | 0 | 0.03 | 0-1 | 0 | 0.13 |
| Sora | 0.11 | 0-2 | 0 | 0.06 | 0-2 | 0 | -2.94 |
| ***Wrens*** | | | | | | | |
| Sedge Wren | 0.04 | 0-3 | 0 | 0.04 | 0-3 | 0 | 0.61 |
| Marsh Wren | 0.49 | 0-41 | 0 | 0.01 | 0-1 | 0 | -3.35 |

*Species Richness*

Based on cumulative SR, there were no significant differences between historical and recent surveys when all sites were pooled. This included richness measured for all species observations as well as for water-obligate species only (Fig. 10). However, comparisons of site specific SR indicated significant differences in cumulative SR for three sites: 20$^{th}$ Avenue West, 27$^{th}$ Avenue West, and Spirit Lake West (Fig. 11). For comparisons of water-obligate species, four sites had significant differences: 20$^{th}$ Avenue West, 27$^{th}$ Avenue West, 40$^{th}$ Avenue West, and Spirit Lake West (Fig. 12). At each of these sites, with the exception of 40$^{th}$ Avenue West, cumulative SR was greater in historical surveys ($p \leq 0.05$).

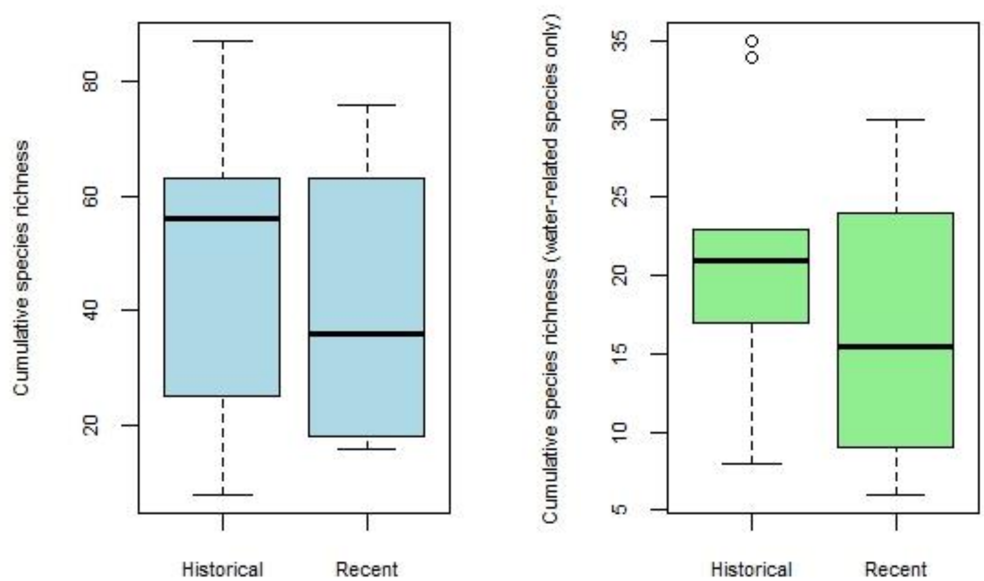



**Figure 10.** Cumulative species richness (SR) calculated using all species observations for historical and recent surveys (left) and for water-obligate species only (right).

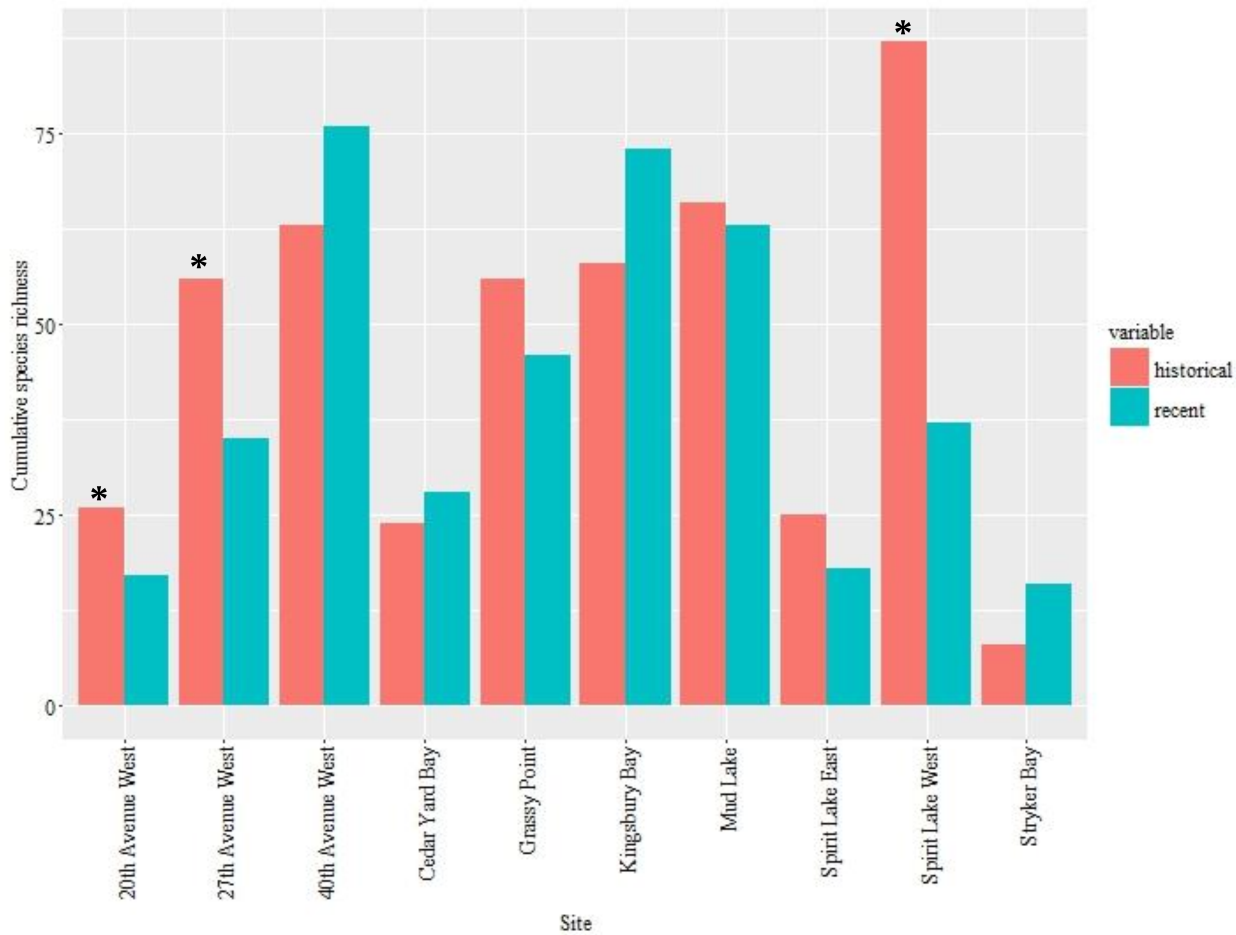



**Figure 11.** Cumulative species richness (SR) of all species by site. Asterisks represent sites where differences in historical versus recent SR was significant at $p \leq 0.05$.

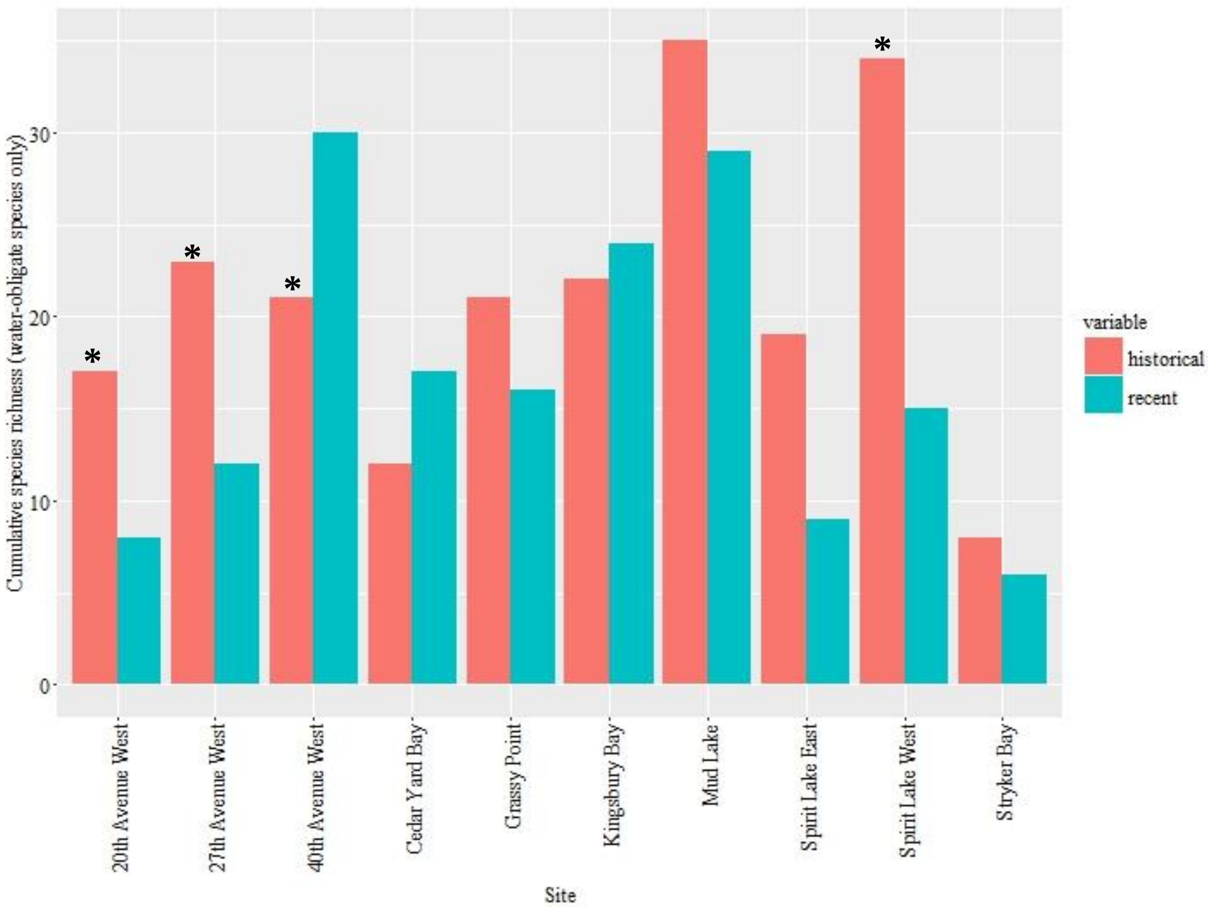



**Figure 12.** Cumulative species richness (SR) of water-obligate species by site. Asterisks represent sites where differences in SR were significant at $p \leq 0.05$.

*Dissimilarity*

As with comparisons of R2R and reference sites, we also calculated dissimilarity indices using NMDS and then used hierarchal clustering based on those dissimilarity indices. These results showed sites clustering primarily based on time period (historical vs. recent), which resulted in historical sites being more similar to other historical sites and recent sites being more similar to other recent sites (Fig. 13). In contrast with comparisons in R2R and reference sites, recent and historical sites did not overlap as extensively, primarily because of differences in several bird communities, with significant temporal difference in group heterogeneity based on beta diversity (F=5.1153, p=0.001; Fig. 13).

The cumulative impact of the five most influential water-obligate species, contributing to differences between the historical and recent surveys were the following, in order of highest to lowest contribution: Canada Goose (CANG; 0.29), American Coot (AMCO; 0.42), Mallard (MALL; 0.55), Blue-winged Teal (BWTE; 0.65), and Lesser Scaup (LESC; 0.72). These species accounted for ~72% of the explained dissimilarity (Figure 15). When comparing the site-specific influence for species, the top three influential species varied by location (Figure 16). At 20$^{th}$ Avenue West, Canada Goose, American Coot, and Mallard accounted for ~60% the dissimilarity, whereas for 27$^{th}$ Avenue and 40$^{th}$ Avenue West it was

~45%, Cedar Yard Bay - ~31%, Grassy Point -~38%, Kingsbury Bay - ~34%, Mud Lake -~34%, Spirit Lake East -~54%, Spirit Lake West-~63%, and Stryker Bay -~27%. Based on BBS trends (1966-2013) in BCR-12, Canada Geese have increased significantly and Blue-winged Teal have declined significantly (Sauer et al. 2014; Table 4). BBS trends for other species such as Mallard, American Coot, and Lesser Scaup were not significant (Sauer et al. 2014; Table 4).

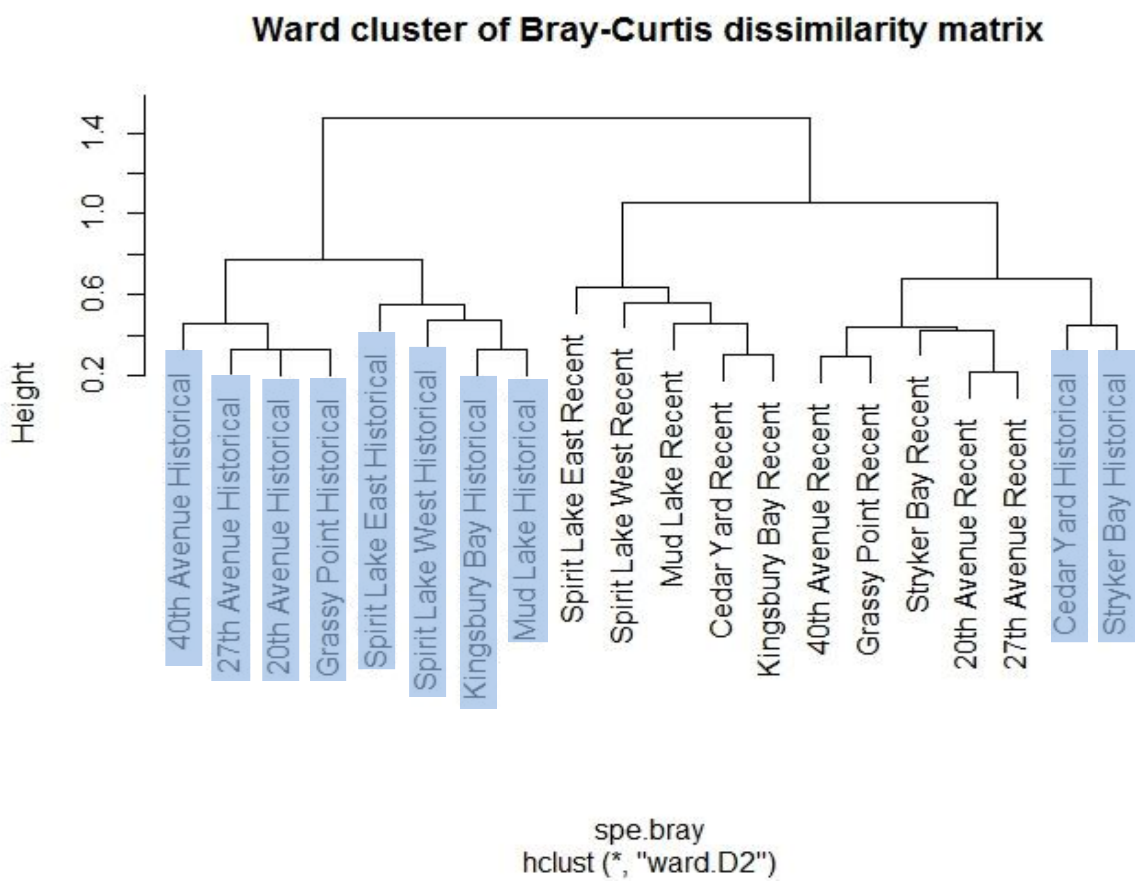



**Figure 13.** Cluster dendogram showing the relationship between historical (blue) and recent surveys at 10 sites located in the St. Louis River based on a Bray-Curtis dissimilarity matrix using water-obligate species only.

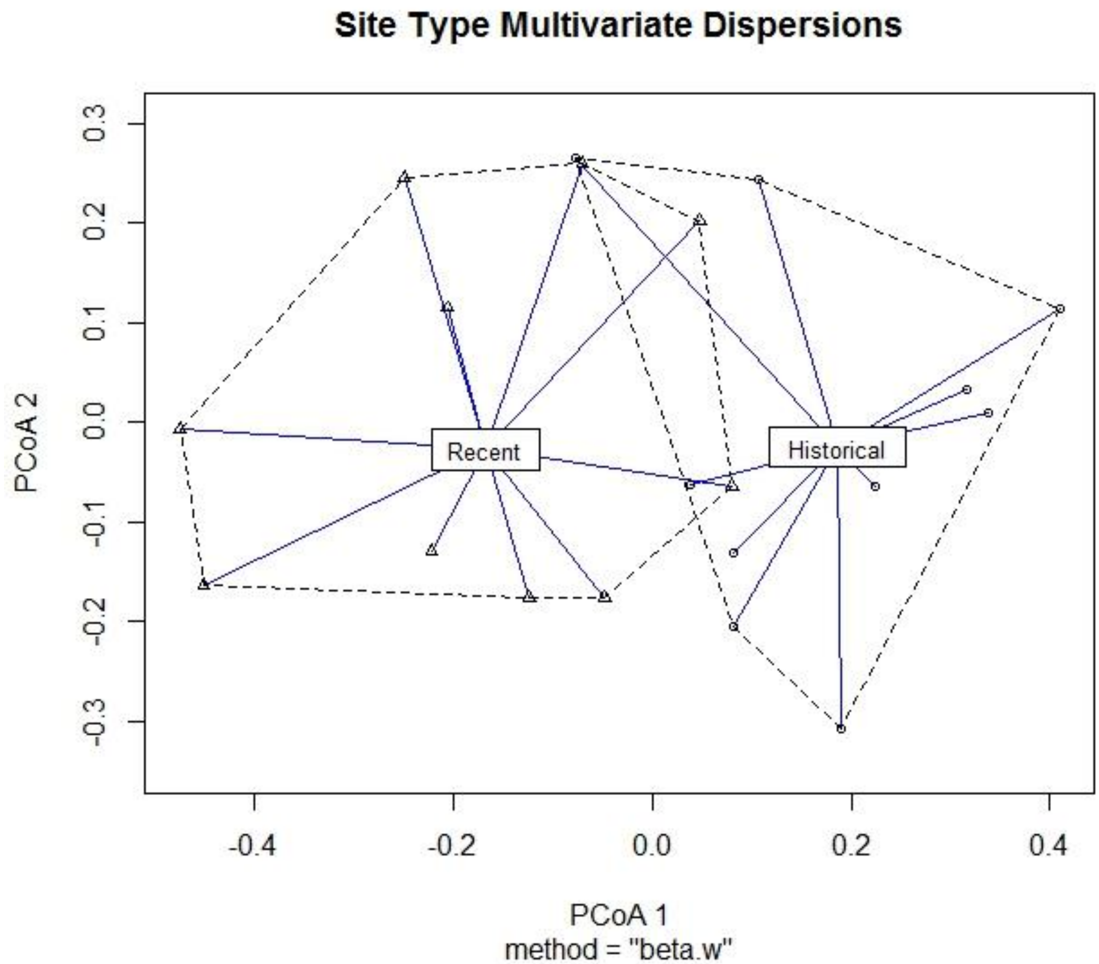



**Figure 14.** Measure of multivariate dispersion between site types (historical and recent) for water-obligate species only for 10 Recent and Historical sites. Dispersions between groups (site type) were significantly different at p = 0.001.

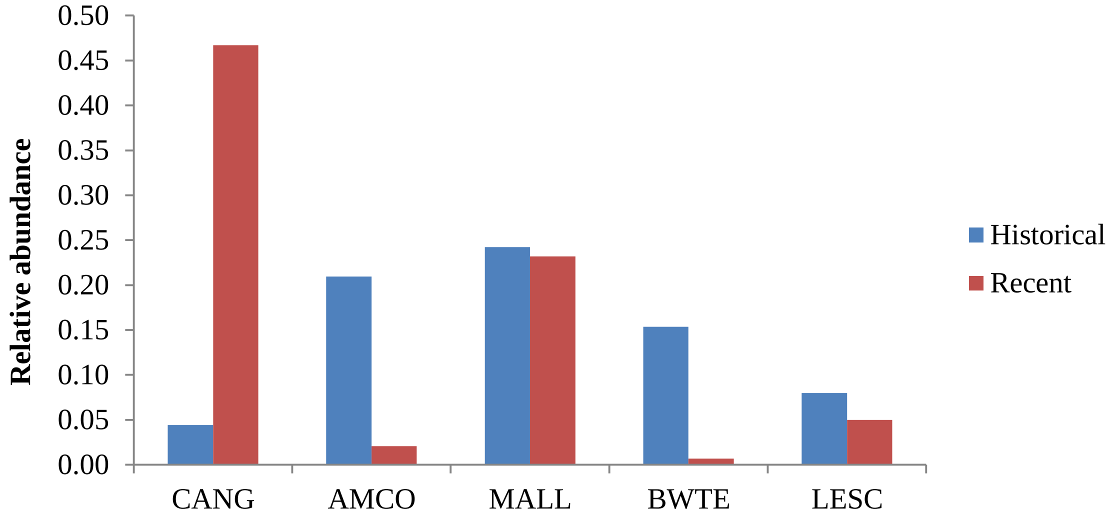



**Figure 15.** Relative abundance of the five most influential water-obligate species: Canada Goose (CANG), American Coot (AMCO), Mallard (Mall), Blue-winged Teal (BWTE) and Lesser Scaup (LESC). These species collectively account for ~72% of the explained dissimilarity between historical and recent surveys.

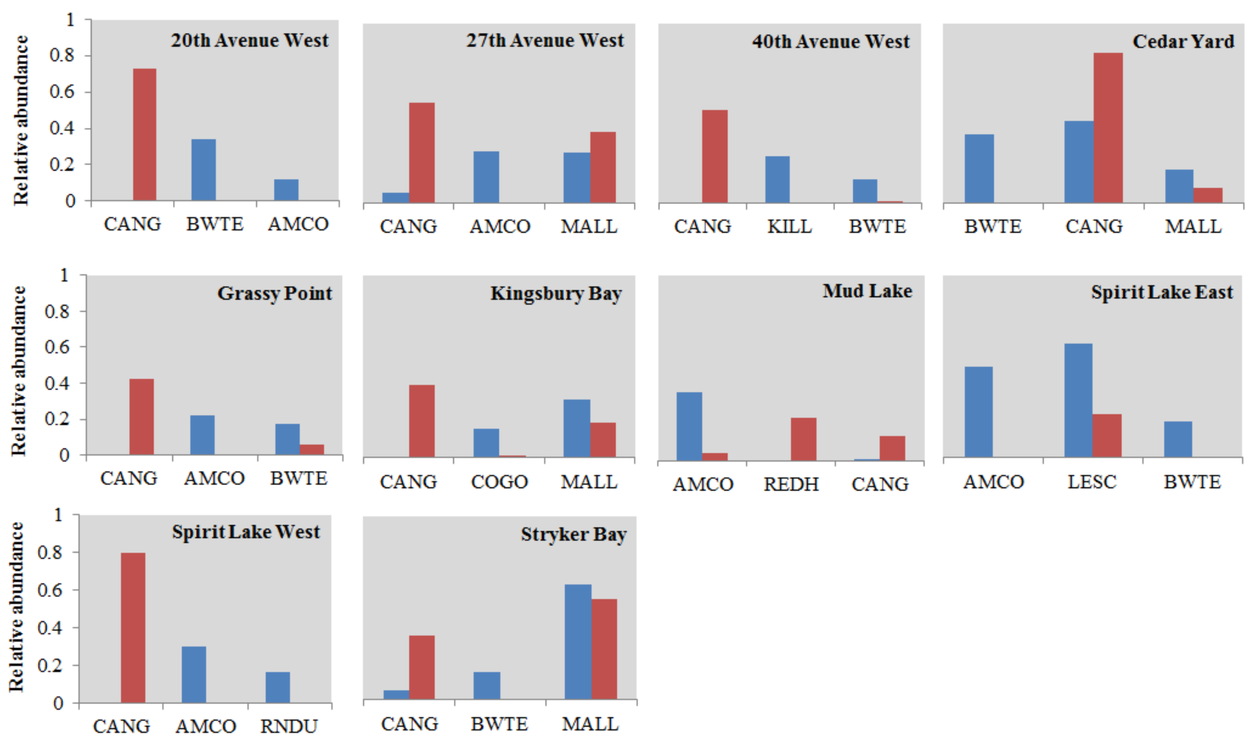



**Figure 16.** Relative abundance of the three most influential water-obligate species at each of the 10 sites based on Bray-Curtis dissimilarity. Species include Canada Goose (CANG), Blue-winged Teal (BWTE), American Coot (AMCO), Mallard (MALL), Killdeer (KILL), Common Goldeneye (COGO), Redhead (REDH), and Lesser Scaup (LESC). Blue bars represent historical surveys and red bars represent recent surveys.

## Discussion

There are many reasons a species may be present or absent from a given location and although changes or differences in species composition can be quantified, they are not always easy to interpret (Philippi et al. 1998). The presence of a species at a given site or set of sites implies these locations provide a similar set of conditions which allows a species to exist and potentially persist (Borcard et al. 2011). However, if a species is absent, it is difficult or impossible to discern why it is not present. There are many reasons why a species may be absent or undetected including 1) poor site condition, 2) lack of detection, in which the species was present but not observed, and 3) factors outside the sampled area such as an overall declining population and a retraction of the species range.

*Objective 1*

Although there were not significant differences in overall species richness or species composition between R2R and reference sites, there were some notable, and significant, differences in the species found in individual R2R sites within the estuary relative to reference sites. For example, there were nearly 13 times as many Canada Geese and Mallards observed in R2R sites compared to reference sites. However, there were overall larger numbers of waterfowl and shorebirds observed in R2R sites, many of which were located in areas where shorelines and water depths have been manipulated by human activity For instance, at 40th Avenue West these changes have resulted in varied shoreline types and differences in bathymetry related to dredging of shipping canals and removal of vegetation and trees from the shoreline. These human disturbances have created different types of available habitat. For example, habitat suitability for diving ducks is species dependent, with some preferring shallower areas such as Redheads and others preferring greater depths such as Lesser Scaup. Many of the reference sites in the estuary were chosen, in part, because they were considered less disturbed by humans and therefore, tended to be in shallower, more protected areas, often with heavily vegetated shorelines. Therefore, shorebirds that prefer unvegetated shorelines and waterfowl that use deep water were less abundant in reference sites.

Based on cumulative SR of reference and R2R sites, Minnesota Slip and Slip C had lower SR relative to Minnesota Point likely because neither of these sites have a natural shoreline, plus human activity is very intense. Shorelines consist of cement shipping channels and most species were those associated with built environments such as Ring-billed Gull, Rock Pigeon, and House Sparrow. Cedar Yard Bay also had lower overall species richness relative to North Bay but still had relatively high SR relative to other R2R sites and was actively undergoing restoration activities during the survey period. Perch Lake had lower water-obligate species richness than Rask Bay, this may have been due to the more extensively vegetated shoreline and sheltered inlet at North Bay. The 21st Avenue West and 40th Avenue West sites both had higher SR than Little Pokegema Bay. These two sites are located in heavily industrialized areas of the SLR but despite being close to the city, are relatively isolated. Portions of their shorelines are sandy while others are vegetated. These factors may be associated with the high species diversity observed at these sites particularly during migration. Comparing R2R and reference sites based on dissimilarity indices suggested that although sites tended to cluster based on site type and in space, overall there were no significant differences in beta diversity among sites.

*Objective 2*

Interpretation of the historical surveys and recent surveys of the same area requires consideration of how populations of bird species have changed over the past 30 years independent of the changes that have occurred in the SLR. Many waterfowl and shorebird species were still common and widespread in the region and across North America. Overall waterfowl populations have increased over the past five decades (NABCI 2016), while some have changed substantially – both increasing and decreasing. In contrast to many areas of North America which have continued to see reductions in water quality and expansion of agriculture and human populations, the SLR has improved in water quality with the addition of WLSSD and agriculture is a negligible issue in the region. In addition, DDT was banned in the early 1970s and overall contaminant levels have declined.

**Waterfowl.** A total of 4 of 23 waterfowl species that were compared with paired t-tests of historical and recent surveys indicated one species, Canada Goose, was more abundant in recent surveys. Three species, Wood Duck, Blue-winged Teal, and Northern Pintail were more abundant in historical surveys than presently. Both Wood Duck, and Northern Pintail, were less abundant despite both species having increased regional populations from 1966 to 2013. These increased trends were only significant for the Wood Duck. Reduced populations of Blue-winged Teal in recent surveys are consistent with reductions in regional populations for this species. Overall, there is little basis to state that waterfowl populations have changed considerably in the SLR, except there clearly has been a massive increase in the Canada Goose population over the past 40 + years. The effect of this population increase on other species of waterfowl is unclear.

**Waterbirds.** Six of 17 species of waterbirds had significant differences in paired t-tests between historical and recent surveys. Five of the six species had consistent differences between the two periods that were also consistent with their regional population trends. Double-crested Cormorants have increased significantly over the past 40+ years, while the Great Blue Heron, American Bittern, American Coot, and Black Tern have all declined; though only significantly for the Great Blue Heron and Black Tern. The anomaly includes the fewer observations of the Common Loon in recent surveys compared with historical counts, despite significant increases in regional populations of the Common Loon. However, the number of observations of the Common Loon in the SLR is very small and provides limited emphasis on the overall interpretation of changes in the SLR. The rarity of the American Bittern in the SLR also must be considered cautiously. Reductions in the Great Blue Heron may be associated with changes in the location of their colony site. During the historical surveys, this species nested near Kimball's Bay, but its colony site no longer exists in close proximity to the SLR. We are unaware of the current location of the colony site. Presumably, its greater travel distance from its colony site has had some influence on its presence in the SLR. As with the waterfowl, there is no strong basis for major changes in the waterbird community in the SLR, except for the substantial increase in Double-crested Cormorants.

**Shorebirds.** Six of 16 species of shorebirds compared between historical and recent surveys were different and all indicated significantly fewer observations of shorebirds in the recent period. Three of the six species with fewer observations, Killdeer, Spotted Sandpiper, and Wilson's Snipe, were consistent with significant regional population declines. The fewer observations of Black-bellied Plover, Pectoral Sandpiper, and Semipalmated Sandpiper in recent surveys have no support from regional populations because none of these species nest in the continental U.S. They only occur in the SLR as migrants and all nest in the northern tundra. The overall lack of use by shorebirds in the SLR is a concern and is deserving of further study. It is unclear whether suitable breeding or stopover habitat is an issue in the SLR for shorebirds compared with the past.

**Rails and Wrens.** There were no significant differences in historical or recent surveys for the two rail species that were most common in the SLR or for the two wren species that have been identified at the state level or nationally as species of concern.

**Species Richness and Composition.** Based on cumulative SR of historical versus recent surveys, 20th Avenue West, 40th Avenue West, and Spirit Lake West had significantly higher SR in historical surveys than in recent surveys. This was also true for 27th Avenue West, when only water-obligate species were included. These areas have undergone considerable changes in industrial activity over the past 30+ years. WLSSD in the 20th to 27th Avenue West area was being developed during the late 1970s and has considerably expanded its operation since the 1970s. Similarly, considerable changes have occurred in the 40th Avenue West region at Erie Pier and the addition of the Bong Bridge. The piling of dredge material at Erie Pier had not begun in the late 1970s and construction of the Bong Bridge was initiated in 1982 and finished in 1985. All of these changes may have had considerable influence on the bird use of these areas.

It is not unusual for dissimilarity in species composition to increase with temporal separation (Philippi et al. 1998), which was the case with the historical and recent comparisons of species composition. In contrast to comparisons of R2R and reference sites, recent and historical surveys did not overlap as extensively, primarily because of differences in several bird species. The primary species associated with these differences included Canada Goose, whose presence throughout the estuary has increased immensely since the 1970s. Note that the Canada Goose population has increased by almost 18 %/yr in BCR-12 from 1966 to 2013. In contrast, American Coot was present in several locations within the estuary in historical surveys but absent from these areas in recent surveys. However, the species was still present in large numbers throughout the estuary in sites that were not included in these analyses (e.g., Rask Bay). Mallards remain an abundant species in the SLR but showed considerable variability among sites during both time periods. Overall differences between the two time periods appear to be minimal.

## Individual Species Accounts

### Piping Plover

Commercial hunting for feathers in the 19th century decimated the North American Piping Plover population. With the signing of the Migratory Bird Treaty Act in 1918 their population began to recover and the Great Lakes population was estimated to be as high as 802 breeding pairs in the 1930s (Russell 1983, Hyde 1999, Haig and Elliott-Smith 2004). This regional population began to decline again in the late 1970s and reached a low of only 17-19 nesting pairs in 1982 (Russell 1983). The Great Lakes population of Piping Plover was listed as federally endangered in 1985 (USFWS 2003).

The Piping Plover was first documented in the St. Louis River Estuary in 1936 when a few birds were found on Minnesota Point (Russell 1983, Price and Cuthbert 2002). Until the 1980s, the Duluth-Superior area had annually seen small numbers of nesting plovers. An average of five nesting pairs was common throughout the 1970s (Russell 1983) with a high of six pairs in 1977 (Davis et al. 1978, Niemi and Davis 1979b; Fig.17). This population steadily dwindled to 3 pairs in 1985, none of which successfully hatched young (Guertin and Pfannmuller 1985). The first year that Piping Plover were not observed nesting in the estuary was 1986 (Davis 1986).

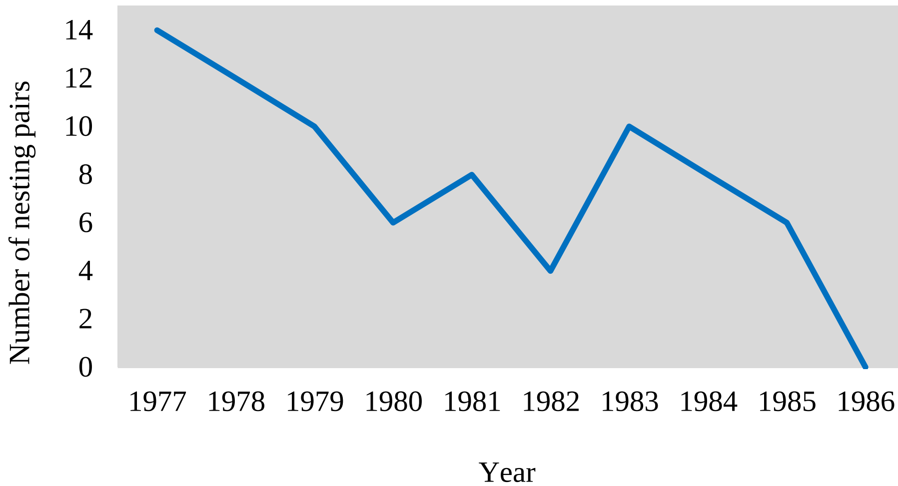



**Figure 17.** Summary of recent Piping Plover populations in the St. Louis River Estuary from 1977-1986. No nesting has been observed since 1986; though individuals have been regularly observed during migration since 1986 (Data from Guertin and Pfannmuller 1985).

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Since 1986 there have been periodic observations of Piping Plovers in the estuary. For instance, according to eBird (http://ebird.org/content/ebird/), an on-line system for recording bird observations, the species has been observed every year for the last 10 years, except in 2011. Most observations are of a single bird, but occasionally two have been sighted. Locations of the observations were generally along the beach on Minnesota Point, Wisconsin Point, Hearding Island, or at Erie Pier. These data suggest that Piping Plovers are still returning to areas where nesting has been documented in the past.

Some of the factors that have contributed to the decline of the Piping Plover in the Great Lakes Region are habitat loss to development, disturbances from recreational activities, predation, and high lake water levels (Russell 1983). The Great Lakes shoreline experienced intense commercial and residential development during the post-World War II period. With new businesses, marinas, and homes being built wetlands were filled and erosion control methods were employed in areas that were prone to shifting shorelines. Additionally, recreational activities that accompany shoreline development such as frequent foot traffic, off-road vehicles, and fireworks may startle birds from their nests, endangering eggs and chicks at crucial stages of their development (USFWS 2003). There were also increased disturbance and predation by dogs, cats, and other predators such as skunks, raccoons, fox, and crows (USFWS 2003). High water levels also reduce suitable nesting habitat and increases nest vulnerability to wave action. The dramatic increase in the Ring-billed Gull population nesting throughout the Great Lakes has also contributed to further loss of habitat as well as increased risk of predation (Hyde 1999, Haig and Elliott-Smith 2004, Haig et al. 2005).

Successful recovery efforts in Michigan have led to the only sizable population in the Great Lakes region with 90 individuals reported in the most recent 2011 International Piping Plover Census (Elliot-Smith et al. 2011). This population has seen some fluctuations but has remained relatively stable through conservation efforts by the U.S. Fish and Wildlife Service (USFWS) in coordination with researchers who had developed strategies to improve fledging survival. The USFWS implemented nest patrolling in 1994 with volunteers monitoring known nests over holiday weekends. This program expanded to include other departments in Michigan and eventually expanded to the Apostle Islands area of Wisconsin. Currently all known nests are surrounded with an exclosure made of wire fencing and monofilament line to reduce disturbance and predation. In areas with frequent foot traffic, a 30 m buffer is placed around nest sites with signs to deter people from entering the area. A program has also been implemented to salvage eggs from abandoned nests to hatch and raise chicks in captivity. This program has been successful with a fledge rate of 90% compared to 25-76% in wild raised chicks (Hyde 1999, USFWS 2015).

Similar conservation efforts are being carried out by the St. Louis River Alliance's Piping Plover Monitoring Project over the past five years. This program trains volunteers to search area beaches for Piping Plover and inform beachgoers of the hazards these birds face from recreational activities and dogs running loose on the beach. The Alliance has also obtained permission from Douglas County to close Lakeshore Road leading to Shafer Beach on Wisconsin Point in an effort to minimize beach traffic. In May 2015 two birds were observed at the Park Point Recreation Area beach. Actions were immediately taken to close that area of the beach in an attempt to encourage the birds to nest but the birds did not stay in the area. If Piping Plovers do begin nesting in the area, the USFWS has plans to construct exclosures, close the beach, and potentially provide 24-hour surveillance to protect the nest site from intrusion or predation.

A Piping Plover habitat and recovery assessment for the St. Louis River Estuary was conducted in 2002 by Price and Cuthbert (2002). For the Great Lakes Piping Plover population to recover birds need to recolonize or colonize historic or new habitat. Eight sites in the Duluth-Superior Harbor were originally identified as having good potential (Wemmer et al. 2001, Price and Cuthbert 2002). Of these eight sites, Minnesota Point was considered the most suitable based on biophysical beach characteristics. The other seven sites were deemed unsuitable in their current condition due to human disturbance, development, heavy vegetation, narrow beaches, or large numbers of nesting gulls (Price and Cuthbert 2002). However, recommendations for restoration activities at each of these sites are provided in the document and should be used as a reference for any potential Piping Plover restoration projects in the SLR.

Attracting Piping Plovers to the SLR will remain a challenging task, but attraction of birds and protection of nest sites is essential. We believe that the main concerns for this species in the SLR are the availability of suitable, undisturbed sandy-cobble beach habitat, plus the low population levels of this species in the western Great Lakes region which restricts the availability of suitable colonizers. The two closest nesting areas for this species in the region include a small population in Ashland, Wisconsin and a small population on islands in Lake of the Woods, northern Minnesota. The latter population was recently confirmed as the only known population in Minnesota during the recent Minnesota breeding bird atlas project (2009-2013). In summary, the availability of suitable physical habitat is still a factor in restricting the re-establishment of this population in the SLR.

## Common Tern

The number of Common Terns nesting in Minnesota was estimated at 2,000 pairs in the 1930s which at that time was still recovering from being hunted for the millinery trade in the late 1800s. By the 1970s the number of nesting pairs was again in decline and by 1984 only 880 pairs remained in the state (Pfannmuller 2014b). In Minnesota, Common Terns currently nest on four major sites including Mille Lacs Lake, Leech Lake, Lake of the Woods, and the St. Louis River Estuary (Pfannmuller 2014b).

Common Terns were first documented in the SLR when a breeding pair was discovered at the Sky Harbor Airport in 1937 (Engstrom 1940, Davis and Niemi 1980, McKearnan 1986). For about 50 years the tern population in this area continued to increase but then experienced a rapid decline in the 1980s (Penning 1993, Fig. 18). During the intensive study period by Niemi et al. (1979a), Common Terns were found nesting at four sites in the Duluth-Superior Harbor (number of breeding adults 1977-1979): Sky Harbor Airport (14-18), Port Terminal (296-370), Hibbard Power Plant (6-10), and Grassy Point Islands (22-40) (Davis and Niemi 1980). In the early 1980s Interstate Island was cleared of trees and the Port Terminal was being developed for increased shipping activity. Common Terns began to establish a colony on Interstate Island, an 8-acre dredge spoil island situated in the Duluth-Superior Harbor, in 1985 when 50 pairs were documented as nesting on the island. At that time Ring-billed Gulls were nesting in three main locations in the estuary: the Minnesota Power and Light Hibbard Plant, the Duluth Port Terminal, and very nearby at the Peavey Globe Elevator. By 1990, Ring-billed Gulls had begun nesting on Interstate Island when 572 nesting pairs were recorded (Penning 1993).


Common Terns nesting on Interstate Island. Photo credit: K. Rewinkel

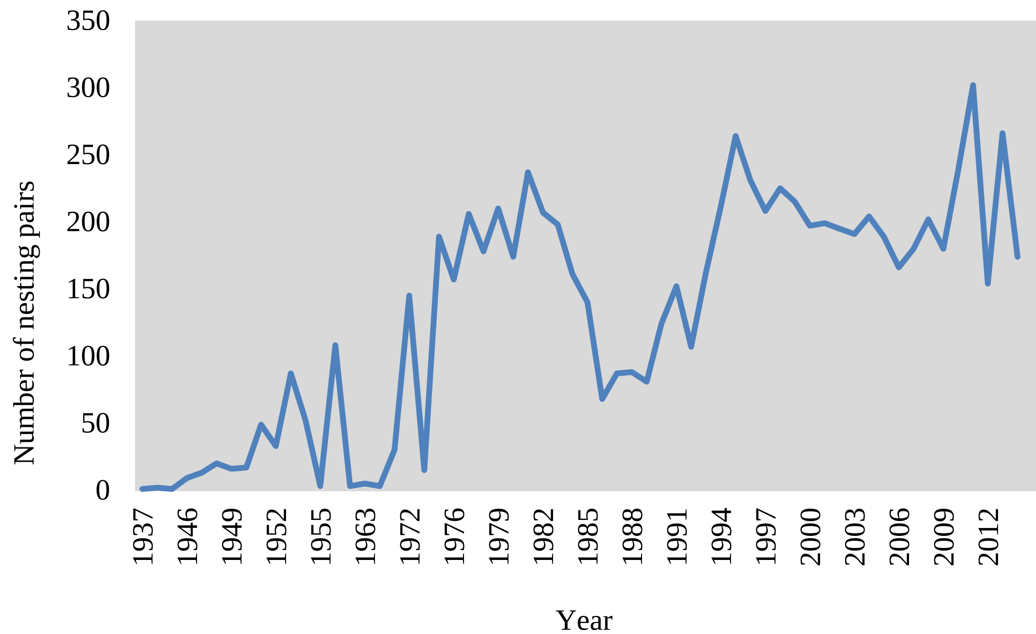


**Figure 18.** Estimated number of pairs of Common Terns nesting in the St. Louis River Estuary (1937-2015). Estimates from 1937-1984 from Penning (1993). Estimates from 1985-2015 for Interstate Island (data provided by F. Strand (WDNR)).

A number of factors, such as predation, human disturbance, and competition with Ring-billed Gulls for nesting habitat have contributed to their decline (Cuthbert et al. 2003, Pfannmuller 2014b). The rapidly increasing population of Ring-billed Gulls drastically reduced available breeding habitat for Common Terns (Fig. 19). Ring-billed Gulls arrive and begin nesting 2-4 weeks earlier in the spring than Common Terns and have effectively eliminated terns from many of their established colonies. (Courtney and Blokpoel 1980, Pfannmuller 2014b).

Great Horned Owls are also a major threat and have been known to cause frequent disturbances to Common Tern colonies. When adult terns temporarily abandon their nests in response to the threat of owl predation, unsheltered eggs and chicks become vulnerable to cooler nighttime temperatures and other predators such as raccoons, fox, rats, and other birds (Erwin et al. 2001,Wires and Cuthbert 2001). Total nest failure at Interstate Island was caused by a Great Horned Owl in 1985 and a history of owl predation has been documented at other sites in the St. Louis River Estuary (Penning 1993). Fluctuating water levels can also cause problems for colonies of nesting terns. Rising water levels reduce suitable nesting area along shorelines by erosion and can destroy nests in low lying areas during storms, whereas falling water levels can create land bridges to island colonies, which allow for increased access by mammalian predators and encroachment of vegetation (Wire and Cuthbert 2001). Terns on Interstate Island may be disturbed by human activity in the estuary as there is frequent boating and shipping traffic in the area due to the island's proximity to the shipping channel. This type of disturbance could cause the birds to abandon their chick and eggs at crucial times in their development leaving them prone to exposure and predation (Courtney and Blokpoel 1983).

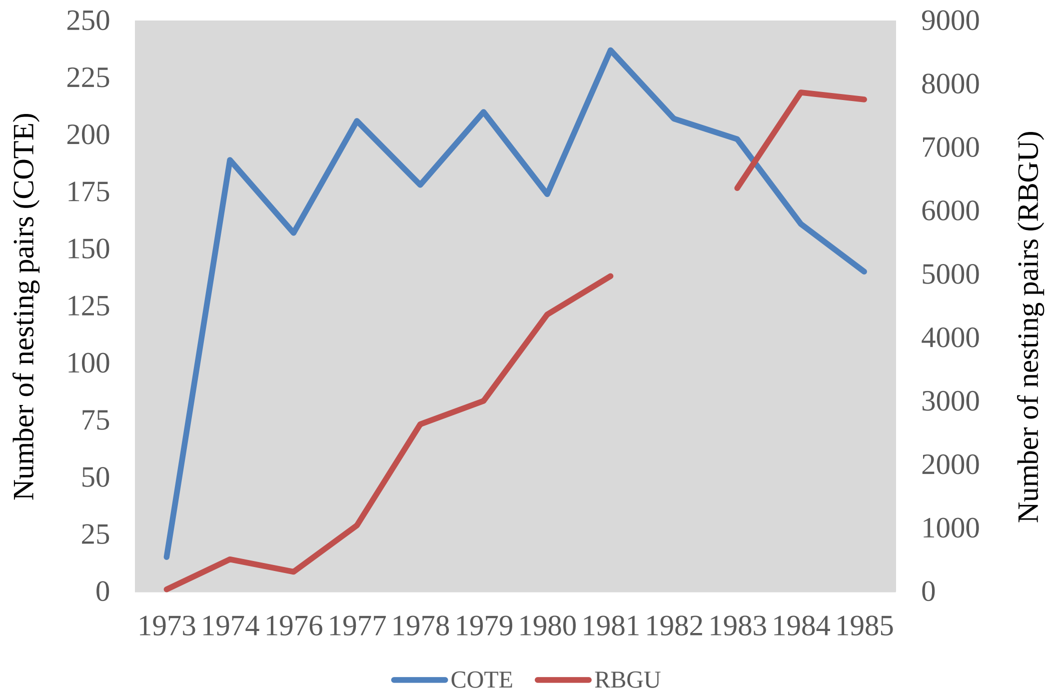


**Figure 19.** Total number of nesting pairs of Common Terns (COTE) and Ring-billed Gulls (RBGU) in the St. Louis River Estuary (1973-1985). Data from Penning (1993).

Most large Common Tern breeding colonies in the Great Lakes region require continuous management to sustain colony numbers (Cuthbert et al. 2003, Wires et al. 2010, Morris et al. 2012). Management techniques have included habitat restoration and protection, predator control, use of exclosures, and construction of artificial nesting structures (Jones and Kress 2011). Because the Common Tern is listed as threatened in Minnesota and endangered in Wisconsin, the legal status of the Common Tern requires that future development does not adversely impact this species.

Since 1985, the estimated number of nesting pairs on Interstate Island has ranged from 68 to 302 (Fig. 18). However, because birds are highly mobile, declines in colony size due to low productivity and survival may be masked by recruitment from a larger area (Weegman et al. 2016). Although counting breeding pairs of birds is a reliable indicator of colony stability, this method cannot differentiate between birth, immigration, death, and emigration events and is therefore unable to identify underlying factors driving population changes. For Common Terns breeding in Lake Superior nest success has varied significantly from year to year and from colony to colony and so it is important to identify site-specific reasons for nest failure and to estimate productivity over time.

A better understanding of the ecology and status of this species breeding in the Duluth-Superior harbor is essential to providing an accurate assessment of the status and condition of this important breeding colony in the SLR. Research and monitoring efforts are necessary to identify best management practices that minimize risk of local extirpation and enhance colony productivity. To achieve these goals it is imperative that these birds have suitable nesting habitat where predation risk is low. Pfannmuller (2014b) outlines and synthesizes state and federal conservation plans for the Common Tern. These goals and recommendations are useful guidelines for continual and effective conservation planning for Common Terns in the SLR and include: 1) protecting and maintaining three island nesting colonies in

Minnesota and *work to restore or enhance one nesting colony site*, 2) Minnesota colonies must produce at least 1.1 young per breeding pair for the state to maintain its current population. Minnesota's Common Tern nesting success rate has ranged from 0-1.35 fledglings/pair, with most falling below 1.0, with Interstate Island's annual reproductive success rate averaging 0.91 (1989-2010; Pfannmuller 2014b). Although below the target of 1.1 fledglings/pair, it is higher than the other Minnesota colonies that document fledgling rates. The Common Tern colony in the SLR needs continual management to sustain colony numbers and to insure successful reproduction. Compared with the late 1970s the Common Tern now only nests at one highly, vulnerable site in the SLR. Because Interstate Island is one of the most important nesting colonies in Minnesota and Lake Superior, we suggest protection and maintenance, including restoration and enhancement of the island be a priority to MPCA. Any delisting of the Fish and Wildlife BUI in the SLR should strongly consider its effect on the future of the Common Tern. We recommend that MPCA support current and future restoration efforts to maintain suitable breeding habitats and support continued intensive monitoring and management efforts.

## Black Tern

The Black Tern is listed as endangered in Wisconsin and had been listed as a Species of Greatest Conservation Need in Minnesota; though it no longer has that designation (Shuford 1999, Pfannmuller 2014a). Breeding populations have significantly declined from 1966-2013 in Minnesota (-6.1%/year), Wisconsin (-5.6%/year), and throughout the United States (-2.9%/year) (Sauer et al. 2014). The decline in Minnesota translates into a 95% decline in the population over the last 50 years. The population in the Great Lakes Region has also been declining even more drastically. The Great Lakes Marsh Monitoring Program has reported that this species has declined at a faster rate than any other bird monitored in the program with a population decrease of 10.5% annually between 1995 and 2012 (Pfannmuller 2014a).

Black Terns are semi-colonial nesting birds that prefer breeding in shallow open wetlands larger than 20 ha with sparse emergent vegetation such as cattails, bulrush, or bur-reed. This type of habitat has been rapidly disappearing from the landscape with increased industrial, residential, and agricultural development and general degradation of wetlands (Dunn and Argo 1995, Pfannmuller 2014a).

There is little historical population information available for breeding Black Terns in the Duluth-Superior area. Reports from the Duluth Bird Club in June 1953 of nesting activity at what is now Southworth Marsh included eggs and hatched young (Bronoel 1953). He also mentions breeding activity observed in this location much earlier by Olga Lakela in 1937 and she noted that Black Terns were absent for a number of years after 1937 but had begun to return (Bronoel 1953). Niemi et al. (1979a) stated the Black Tern "nested in six marsh communities in the estuary" in the SLR from 1977-1979. They included the following areas with the number of breeding adults in parentheses (based on the number of nests found): Allouez Bay, WI (20 in 1977 and 32 in 1979), Pokegema River, WI (10 in 1978 and 8 in 1979), Indian Point, MN (2 each in 1978 and 1979), Morgan Park mudflats (50 in 1978 and 8 in 1979), Mud Lake, MN (4 in 1978), and South Spirit Lake, WI (20 in 1978 and 20 in 1979).

Allouez Bay and adjacent Wisconsin Point comprise one of the largest wetland complexes in the SLR (Niemi et al. 1977, Davis et al. 1978). The earlier observations of Black Terns by Lakela and Bronoel as well as the observations of nesting in the 1970s indicate that the species has been a frequent, if not permanent, breeding resident of the SLR in the past. The Wisconsin Breeding Bird Atlas documented nesting by the Black Tern in or near Allouez Bay during the period from 1995-2000 (Cutright et al. 2006). However, there is no other recent documentation of nesting by this species in other parts of the SLR. An extensive breeding bird inventory of the SLR in 1999 revealed no Black Tern observations (Niemi et al. 2000).

Since the late 1970s the SLR has had many changes and Black Tern populations have declined over a large area of their range. The species has seen extensive retraction of its range, especially in the northeastern portion of Minnesota. There are many potential reasons for its decline including increased eutrophication of water bodies, sedimentation, and pollutants such as mercury, dioxins, pcbs, and other xenobiotic chemicals. Wetlands in the SLR have also been affected by invasive aquatic plant species (Kitson and Jensen 2015). Studies of the Black Terns in South Dakota indicated that vegetation structure was more important than vegetation composition. Showing they required either short dense or tall sparse vegetation to provide them with enough cover for chicks to avoid predation and ease of flight for adults to defend nests (Naugle et al. 2000). Since these types of wetlands cycle through stages with differing levels of vegetation it would be necessary to preserve and manage several wetland areas in regenerative and degenerative states to attract and maintain a breeding population of Black Terns (Matteson et al. 2012).

Black Terns have been affected by anthropogenic habitat and landscape changes to wetlands, but they are also vulnerable to the effects of climate change. Freshwater wetlands will be affected by temperature increases and lower precipitation levels predicted to accompany climate change in this region. Hence, freshwater wetland habitats are predicted to become dryer with increased vegetation, which may further reduce their already limited habitat.

Although Black Terns have been known to nest in cattail marshes, the invasion of narrowleaf cattail (*Typha angustifolia*), and its hybridization with native broadleaf cattail (*Typha glauca*), may also be contributing to changes in habitat (Kudell-Ekstrum and Rinaldi 2004). The increased biomass and density of cattails reduce the available number of quality breeding sites through the loss of open water and sparsely vegetated areas. Quality food sources are also diminished by the lowered diversity of invertebrates found in these monotypic stands (Boers et al. 2006, Linz and Blixt 1997).

The creation of nest platforms can be beneficial for breeding Black Terns in areas where quality nesting habitat is limited. Although it can be difficult to attract the terns to use nest platforms, when they are used numbers of nesting birds and hatching success have both increased. (Shealer 2005). Wyman and Cuthbert (2016) found that key predictors of Black Tern colony persistence in the U.S. Great Lakes were wetland area, wetland type (emergent vegetation, open water, or combination), and area of wetlands available for foraging within 2 km of the colony.

In summary, the North American Black Tern population has declined substantially over the past 50 years. It is currently extirpated from the SLR as well as many areas in the northeastern distribution of the species range in Minnesota. It is unclear the extent that changes in the SLR have contributed to its extirpation or lack of re-colonization in the area. On the surface there appears to be suitable habitat still available in the SLR, especially in the Allouez Bay area. However, given the species steep population decline, recovery of its North American population may be a prerequisite for its return to the SLR. Substantial effort should be included every year to determine whether individuals are still being observed in mid to late May in the SLR and, as with the Piping Plover, efforts made to protect potential nesting activity. The substantial changes in water levels that have occurred over the past 50 years in the Great Lakes should also be examined with respect to Black Tern nesting. Higher water levels were purported to be a problem from 1978 to 1979 when the Black Tern colony at the Morgan Park mudflats dropped from 50 to 8 breeding pairs (Niemi et al. 1979a).

## Caspian Tern

This species is listed as endangered in Wisconsin. It has primarily been recorded as a spring and fall migrant in the SLR. A variety of nesting records have been recorded in Wisconsin dating back to the late 1800s, primarily from Lake Michigan. The most recent nesting records are also from Lake Michigan, but "possible" nesting was identified in the 1995-2000 WI Breeding Bird Atlas project from Allouez Bay and the Ashland area of WI (Cutright et al. 2006). Historical accounts from Roberts (1932) indicate no nesting records in the state; though statewide coverage was sparse, especially in the northern portion up to the 1930s. The only known records of nesting in Minnesota are both from Leech Lake with two nests in 1969 (*Loon* 41: 83-84) and confirmed nesting during the MN Breeding Bird Atlas project (2009-2013) (www.mnbba.org). No nests were found during the 1970s in the SLR, but up to 100 individuals were observed during peak migration counts in late May (Niemi et al. 1979a). In addition, approximately 12 individuals were regularly sighted at Allouez Bay during the summer of 1979, but no nests were located.

Cuthbert and Wires (1999) report that the Caspian Tern has been increasing in the Great Lakes; likely due to protection and management of nesting sites as well as reduction in the use of organochlorines. The lack of history of nesting by the Caspian Tern in the SLR suggests that it does not enter into consideration regarding the Fish and Wildlife BUI. However, physical habitat protection of sandy-cobble nesting areas, similar to the Common Tern and Piping Plover, would potentially allow the species to colonize and use the SLR in the future. Caspian Terns nest within Common Tern colonies at Leech Lake, MN and have frequently nested in association with Ring-billed Gulls in Wisconsin (Cutright et al. 2006). Breeding activity of this species at Allouez Bay and other suitable nesting areas of the SLR should be monitored annually.

### Forster's Tern

There are no documented nesting records for this species in the SLR. Current nesting in Minnesota primarily occurs in the western and southern areas of the state. Roberts (1932) also emphasized its western breeding distribution and its primary occurrence as a migrant in the eastern portion. This is also true of the species in the SLR (Niemi et al. 1979a, Green and Niemi 2011). In Wisconsin, this species appears to be largely confined to the central and southern regions, in locations that were historically identified as important nesting sites, such as Winnebago Pool, Lake Puckaway, Rush Lake, Big Muskego Lake, Horicon Marsh, and Green Bay (Cutright 2006). Despite the species extensive use of large riverine wetland ecosystems, like the SLR in other parts of its range, so far it has not been found nesting here. However, there appears to be potential given the extensive wetlands that exist in the SLR. Currently there is no basis for consideration of this species in removal or maintaining the Fish and Wildlife BUI.

### Great Blue Heron

The Great Lakes population of Great Blue Herons was flourishing between 1977 and 1991 when the Great Lakes Colonial Waterbird Censuses found an increase of 43% in the number of nests located in this region. After 1991 the population began to decline, with a decrease of 26% through 1999 and a continued decrease of 18% from 1999 to 2008. This decline can be attributed to changes in land use, water quality or food availability, frequent human or natural disturbance, interspecific competition, or predation (Rush et al. 2015). These birds are adaptable in their breeding habitat but appear to prefer locations that are inaccessible to predatory mammals and have low rates of human disturbance. They feed primarily on fish in slow moving or calm water and rarely nest more than 20 km from their foraging habitat. Colonies located in areas of high disturbance are prone to frequent relocation and may resettle in smaller groups (Vennesland and Butler 2011). Since trees used by nesting colonies of Great Blue Herons eventually die, due to old age or the acidity of the heron droppings, it is important that areas with appropriate nesting habitat be preserved for future colony locations (Danz et al. 2007).

The St Louis River Estuary was documented as hosting a colony of Great Blue Herons at the northernmost point of Kimball's Bay, with 110 to 186 breeding adults recorded from 1977 to 1979, respectively (Niemi et al. 1979a). This colony has since disappeared and a private residence has been developed in the former nesting area. It is unknown what prompted the species to desert their colony but encroaching development could have been a factor. Nest and colony abandonment have been known to increase in areas with high human activity. Most colonies require a buffer of at least 300 m where humans are excluded, especially during the breeding and nesting seasons, to prevent desertion (Watts and Bradshaw 1994, Vennesland and Butler 2011). Increased accessibility of this colony to predators which are more common in residential areas, such as raccoons, could also have contributed to the loss of this colony. Raccoons, once they locate a colony, will often prey on eggs and nestlings until none are left causing the birds to permanently abandon their colony (Rodgers 1987). A recent example of this would be the large colony at Peltier Lake in east central Minnesota. This colony at its peak in the 1990s contained more than 1,000 nests but for unknown reasons the population started to decline about ten years later and by 2005 only 25 nests remained. In 2004 cameras that were installed to monitor the colony showed raccoons preying on eggs and nestlings. Predation was so extreme that no young survived in 2004. Remediation efforts have been successful in preventing some of the raccoon predation and the following year four young survived (Von Duyke 2009).

Predation by Bald Eagles could also have contributed to this abandonment. Bald Eagles are one of the few predators of adult Great Blue Heron and frequently prey on nestlings and eggs (Forbes 1987, Norman et al. 1989). The enormous population recovery of the Bald Eagle has been shown to have been a factor in the desertion of colonies in the Pacific Northwest. As eagles became more prevalent in the post-DDT era they have been documented preying on nestlings at Great Blue Heron colonies which have resulted in colony abandonment (Kelsall and Simpson 1980, Norman et al. 1989, Jones et al. 2013). The Great Blue Heron colony located in Kimball's Bay could have faced similar pressures as eagle populations in Minnesota and the Great Lakes region have substantially increased (Bowerman et al. 1995).

Great Blue Heron colonies have been known to relocate if disturbance becomes too great and may attempt to resettle nearby often in smaller splinter colonies (Vennesland and Butler 2011). Attracting and reestablishing breeding Great Blue Herons in the SLR will most likely require keeping multiple large undisturbed areas of the appropriate habitat available or, if that is not feasible, possibly installing nest platforms. Many individual Great Blue Herons have been observed in the SLR during the current study period, but no colonies have been located for many years. Several local bird watchers in the area have suggested that a colony site exists in the Superior Municipal Forest. We suggest that an effort be made to search for the colony or colonies and provide adequate protection of these sites if possible.

## American White Pelican

This species has experienced an exponential increase in its population since the turn of the century in 2000. Breeding Bird Survey trends in Minnesota and Wisconsin from 1966 to 2013 were 14%/yr ($n$ = 37 routes) and 80%/yr ($n$ = 6 routes), respectively (Sauer et al. 2014). Changes during the past 11 years (2003-2013) have been similar. Hence, even though a few American White Pelicans were observed in the 1970s, the much larger number observed during the recent period is a reflection of this large increase in the population. During the 1995-2000 WI Breeding Bird Atlas only two colonial nest sites were identified: Green Bay and Horicon Marsh. The MN Breeding Bird Atlas (2009-2013) revealed 13 colony sites ranging from Lake of the Woods, Leech Lake, and the remainder in southern and southwestern Minnesota.

The increase in American White Pelican has little to do with activities in the SLR where the species does not nest nor are there any historical records of nesting in the SLR. Banning of DDT and related organochlorine chemicals, protection and management of nest sites, and reduction in illegal shooting are among the reasons for the increase in the population of this species. There is a possibility that this species could nest in the SLR in the future. Like the Forster's Tern, this species should have little influence on determination of the Fish and Wildlife BUI because of its limited distribution in this region; however, improvements in fish populations, water quality, and availability of isolated islands or protected open gravel or sandy areas for nesting will greatly benefit this species.

## Great Egret

The current breeding distribution of this species in Minnesota is primarily in southern, west-central, and southeastern portions of the state (Green and Janssen 1975, Janssen 1987). It was labelled a straggler from the south by Roberts (1932) and the first breeding records in the state were in the late 1930s (Green and Janssen 1975). The MN Breeding Bird Atlas project found confirmed nesting in 24 locations as far north as Becker County which is approximately the same latitude as Duluth. In Wisconsin where it is a threatened species, it is primarily found nesting along the Mississippi and Horicon Marsh/Lake Winnebago area (Cutright et al. 2006). It is a colonial nesting species that nests in large trees, most often in lowland forests adjacent to large rivers or lakes. The species has been significantly increasing from 1966-2013 based on the MN Breeding Bird Survey (4.8%/yr) and throughout the United States (2.1%/yr; Sauer et al. 2014).

The species was observed in the 1970s and during recent counts, but it is still rare and usually observed as single individual in the SLR. Because of its increasing population in Minnesota and to some extent in Wisconsin, we could expect more frequent observations of this species in the SLR as well as a potential nesting species in the future. As with the American White Pelican, we would not expect this species to be considered in the decisions regarding the Fish and Wildlife BUI because it likely was never part of the "recent" native avifauna of the SLR. From 1870-1910 over 95% of this species population was reduced by killing for their plumes. This was primarily an issue in the southern US states, but its recovery is still in process as evidenced by its continued population expansion. The species would greatly benefit from healthy fish, reptile and amphibian, and invertebrate populations; good water quality, and the availability of suitable, large trees relatively close to the SLR for potential nesting.

## Black-crowned Night Heron

Like the Great Egret, the Black-crowned Night Heron is a rare species and represented by a few individual observations in the SLR; both in the 1970s and in recent surveys. In Wisconsin, its primary breeding range is in the central and southeastern parts of the state (Cutright et al. 2006). The MN Breeding Bird Atlas identified 10 nesting locations; all in southcentral Minnesota. The species has had a relatively stable population over the past 50 years but has not shown the same type of expansion in its population like the Great Egret. It is our opinion that the species should not be considered in the decisions regarding the Fish and Wildlife BUI in the SLR because of its rarity and low probability of future colonization in the future; though as with many species it would benefit from healthy, fish, reptile and amphibian, and invertebrate populations as well as the availability of large trees that it uses for its colonial nesting sites.

### Least Bittern

This species is rare and one of the "least" known members of the heron family. It is a secretive species found in dense wetland vegetation, usually in emergent marshes, where it constructs its nest in cattails, bulrushes, or sedges (Poole et al. 2009). The species was formally recognized as a Species of Conservation Concern in the upper Midwest Region (USFWS 2008). Collectively, its status in Minnesota and Wisconsin, as documented by Roberts (1932), Green and Janssen (1975), Janssen (1987), Robbins (1991), and Cutright et al. (2006), suggests little evidence of nesting in northwestern Wisconsin or northeastern Minnesota. There is some suggestion that this species population has declined, likely with the reduction of available wetland habitat over the past 150 years (Poole et al. 2009). However, this species is very difficult to monitor because of its secretive habits, indistinct vocalizations, and the remoteness of its breeding habitat in wetlands. Niemi et al. (1979a) documented that this species "occurred regularly" in the Allouez Bay, Mud Lake, and Spirit Lake Marshes. It was not documented in counts of 39 wetland areas in the SLR in 1999 (Niemi et al. 2000). Recently Bracey (pers. comm.) has detected at least one individual in the wetland area around Clough Island in 2012 and in Little Pokegema River in 2014. These sites were sampled as part of the Great Lakes Coastal Wetland Monitoring Program (Cooper et al. 2014). Because there is little historical data on the presence of this species in the SLR, we cannot make an argument for its consideration in retaining or elimination of the BUI for fish and wildlife in the SLR at the current time.

### Bald Eagle

This species has made a remarkable recovery since the banning of DDT in 1972. The last formal counts of nests in Minnesota in 2005 indicated 872 active nests and an estimated 1,312 nests in the state. The number has clearly grown since that time where the Minnesota Breeding Bird Atlas project recorded Bald Eagles nesting in nearly every county in the state, except Lincoln and Pipestone Counties in the extreme southwest. Wisconsin has shown similar results of expansion, documented by Cutright et al. (2006) in the Wisconsin Breeding Bird Atlas. Counts in the 1970s by Niemi et al. (1979a) did not document any nesting of this species in the SLR; however, several large concentrations (e.g., 44 individuals on April 7, 1978 in the Spirit Lake and Oliver Bridge areas) of migrating Bald Eagles were observed, especially prior to ice-out of interior

Bald Eagle at Boat Club Point near Spirit Lake West. Photo credit: E. Zlonis.

lakes. Today, there are up to five nesting Bald Eagle pairs in the SLR, but no information on their overall nesting success. It is highly likely that populations existed in the SLR during historical, pre-European times. Hence, the recovery of this species in the SLR is supportive of BUI removal, but its recovery has had little to do with changes in the SLR. The species has recovered because of the banning of DDT, the focused management efforts to protect nest sites, the improvement in reduced contaminant loads in food supplies, and its increased tolerance to human disturbance.

### Peregrine Falcon

Peregrine Falcon populations were extremely low in the 1960s and no nesting was reported in Minnesota from 1965 to 1969 (Janssen 1987). A reintroduction program was initiated in 1982 at the University of Minnesota, Twin Cities (www.midwestperegrine.org). Since that time the population has increased substantially in Minnesota. The Minnesota Breeding Bird Atlas identified 79 nesting areas in Minnesota from 2009-2013. This species has nested successful at the Greysolon Plaza Hotel and most recently at the Torrey Building in downtown Duluth, plus a new site at the SP Duluth Ore Docks in 2015 (Fallon 2015). The species has also periodically nested or attempted to nest on the Blatnik and Bong Bridges as well as the Hibbard Steam Plant, but nesting success has been highly variable. This species has clearly increased in its population within the SLR since the 1970s, but nesting success continues to be highly variable and has been greatly aided by the successful, reintroduction program in the upper Midwestern U.S.

## Problematic or nuisance species

### Canada Goose

The rapid expansion of settlers throughout North America in the 19th century drastically reduced the population of Canada Geese across the continent. These people, who were often near starving after harsh winters with little available food, turned to hunting large numbers of geese as the birds returned from their wintering grounds. They also gathered goose eggs in spring as a supplemental food source. Many wetlands were also drained and developed for farmland during this time, reducing suitable habitat for these birds. The Canada Goose population began a slow recovery after the passing of the Migratory Bird Treaty Act of 1918 when unregulated hunting was abated and many wildlife refuges were created to preserve wetland habitat for use by breeding, migratory, and over-wintering waterfowl (Cooper 1978, Smith 1999).

In 1927, Kellogg Bird Sanctuary in Michigan established the first successful reintroduction of a breeding population of Canada Geese in North America. Since then many programs have restored populations of geese to areas where they had formerly occurred as well as to areas that were outside of their historical range (Cooper 1978). Since the diet of these birds includes a high proportion of grasses, they were naturally attracted to the manicured residential lawns, golf courses, and other large expanses of open grassy areas that many urban areas provide (Smith et al. 1999). Many groups of Canada Geese have recently been found to stay in their urban and suburban breeding areas year round. These permanent resident populations of Canada Geese experience increased survival and reproductive rates over wild populations because they are protected from hunting due to firearms laws within city limits, few predators, and are often fed by humans (Smith et al. 1999). Since the 1980s many populations of Canada Geese could be found wintering much farther north in agricultural areas where they consume carbohydrate-rich waste grain rather than the native wetland plants they had consumed historically. To reduce the number of nuisance geese, these birds have frequently been relocated to areas that are farther south than their historical range (Mowbray 2002). All of these situations have contributed to a huge increase in populations of Canada Geese.

Urban flocks of Canada Geese can number anywhere from 10s, to hundreds, to thousands, and even tens of thousands in some areas. Data from a country-wide survey of USDA State Directors found that geese were a problem for more than 100 urban areas in 37 states. These problem flocks numbered anywhere from 10 to 27,500 birds and surveys indicated that a total population of nuisance urban geese was estimated at 299,720 individuals (Forbes 1993). Droppings from flocks this large can create a number of public health issues such as closure of swimming areas and reduction in water quality. The high concentration of nitrogen can cause the eutrophication of urban ponds and lakes resulting in excessive growth of algae. The congregation of large numbers of geese on open grassy areas can also result in trampled grass and packed down soil leading to a ground surface devoid of vegetation which also results in erosion and destruction of habitat.

### Ring-billed Gull

Ring-billed Gulls experienced a population explosion and expansion westward through the Great Lakes in the mid to late 1960s. By 1967 there were an estimated 300,000 individuals in Lakes Huron and Michigan (Ludwig 1974). The first nesting record of the species in the Duluth-Superior Harbor was at Barker's Island in 1957 within a Common Tern colony (Cohen 1958). He stated that the Ring-billed Gull "is the first found of that species in this area in recent years" which implies that the species was nesting in this area in previous years. In 1974, 500 pairs were documented at the Minnesota Power and Light Hibbard Plant and this population increased rapidly until it reached a high count of 8,361 breeding pairs in 1986 (Penning 1993). From 2000-current, an annual nest count of Ring-billed Gulls breeding in the SLR is conducted. From 2000-2004 nests were counted at Minnesota Power Hibbard Plant where nest numbers declined annually from 643 nests to 24. From 2000-2005 nests were counted at South Hibbard Islet, with nest numbers declining from 299 to 0. On Interstate Island total nest counts were conducted from 2000-2016 with nest numbers fluctuating from 8,734 – 14,383. The majority of nesting Ring-billed Gulls in the SLR currently nest on Interstate Island.

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**Appendix A.** St. Louis River AOC R2R Support Project: Ecological Monitoring and Assessment (CR#6403): Migration and Breeding Bird Distribution and Abundance Standard Operating Procedure.

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## Survey Protocol Summary

*Spring/Fall Migration:*

- Each point at each site needs to be surveyed for 10 minutes. If it is not possible to count all birds within 10 minutes, stay until all birds have been counted and write survey duration on accompanying field sheet
- All birds seen or heard should be placed on the maps in the location in which it was observed. Observation type (e.g. singing, observed, flyover) should also be recorded.
- A field sheet will be provided with each map and should be filled out completely during each visit. This will contain site level information (e.g. date, survey duration, location, observer, temperature, etc.).

*Breeding Season:*

- Breeding season surveys will be extended to 15 minute surveys and include use of playbacks

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1. Samples: Bird surveys will be conducted 16 times at each point annually.
    a. Surveys will be conducted:
        i. 6 times during spring migration (March- May)
        ii. 4 times during the breeding season (May-July)
        iii. 6 times during fall migration (August-November)

    b. Sites will be revisited with a minimum of:
        i. 5 days between surveys during the breeding season
        ii. 7 days between surveys during migration periods

2. Survey weather
    a. Because the majority of observations will be visual, wind strength is less likely to affect the quality of the survey. However, it is optimal to conduct surveys when the wind strength is less than 4 on the Beaufort wind scale (i.e. wind < 15 mph or < 20 kmh) for identifying birds aurally.
    b. Surveys should only be conducted when there is little or no precipitation.
        i. If the precipitation is heavier than a drizzle, you should discontinue the survey. Moderate to heavy rain will decrease bird vocalization and other activity levels.
    c. Wind and precipitation during breeding season surveys could affect your ability to detect territorial vocalizing males and therefore it is more important that survey conditions are optimal.
    d. The decision to discontinue a survey due to weather conditions is made at the discretion of the field crew leader.
    e. If survey is conducted during questionable weather conditions, be sure to provide comments on the data sheet, such as why the survey was continued.

3. Sample periods
    a. Be sure to get accurate sunrise and sunset times for your location
    b. All breeding season surveys are morning surveys: sampling can begin from 0.5hr before sunrise to 4.5hrs after sunrise.
    c. Surveys during migration can begin at sunrise and continue into the afternoon.
    d. Surveyors will survey each point within a given location until all birds present have been counted (approximately 10 minutes at each point within a site).

4. Sites and sample points
    a. Each site can contain from 1-4 bird sample points
    b. Sample points
        i. Points will be located near the most convenient access point
        ii. The location of each point will be marked using a GPS unit prior to the first sampling period (June 2013). These locations will not change during the project, unless a safety or accessibility issue arises during the project.
        iii. Points will be saved in the GPS unit as a waypoint as well as in an excel database.
        iv. Once point locations have been established, proceed to the provided point location to conduct surveys.
        v. All points must be marked on the field maps, and notes such as how to access each point must be recorded.

5. Record site data
    a. Before beginning the survey, fill out the following:
        i. Date: Format of MM/DD/YY (e.g. 06/04/13)
        ii. Point ID: Each point has an associated ID (e.g. Site 1 pt.1)
        iii. Observer: Observer first initial and last name (J. Doe)
        iv. Time (start): Record in 24-hour format (e.g. 4:30am is 0430)
        v. Temperature: Record in $^{o}$Celsius
        vi. Wind (code): Beaufort wind scale codes (see chart below)
        vii. Sky (code): Assign and record the appropriate sky cover code (see chart below)
        viii. Noise (code): Assign and record the appropriate background noise code (see chart below)
        ix. Weather: Circle the appropriate description: dry, damp/haze/fog, drizzle, or rain
        x. Site description/notes: Any additional information that you think will be important to record about the survey location. Observations that could affect counts (e.g. ice covering the bay, boat activity in the area) or any other information that may be of interest (e.g. other animals using the area, e.g. beaver or otter)

**BEAUFORT WIND SCALE**

| | |
|---|---|
| 0 | Calm; smoke rises vertically |
| 1 | Light air movement; smoke drifts; leaves barely move |
| 2 | Slight breeze; wind felt on face; small twigs move |
| 3 | Gentle breeze; leaves & small twigs in constant motion |
| 4 | Moderate breeze; small branches moving, raises dust & loose paper |
| 5 | Large branches & small trees sway |

**NOISE CODES**

| | |
|---|---|
| 0 | No appreciable effect (owl calling) |
| 1 | Slightly affecting sampling (distant traffic, dog barking, car passing) |
| 2 | Moderately affecting sampling (distant traffic, 2-5 cars passing) |
| 3 | Seriously affecting sampling (continuous traffic nearby, 6-10 cars passing) |
| 4 | Profoundly affecting sampling (continuous traffic passing, construction noise) |

**SKY CODES**

| | |
|---|---|
| 0 | clear (<10%) |
| 1 | scattered (10-50%) |
| 2 | broken (60-90%) |
| 3 | overcast (>90%) |
| 4 | fog |
| 5 | light mist |
| 6 | water dripping off vegetation |
| 7 | rain during last 5 minutes of census |
| 8 | rain during last 7 minutes of census |
| 9 | rain during entire census |

6. Conduct the survey
    a. Each survey point will be visited for approximately 10 minutes, or until all observations have been recorded.
        i. Using a spotting scope and binoculars, make a preliminary scan of the survey location to identify all individuals present. This is important, as some species may leave the area due to your presence.

    b. We will use unlimited-distance counts, to complete a thorough inventory of bird use, counting all species identified by both visual and aural surveys

c. All bird observations will be identified to specific locations on aerial photo field sheets; accuracy will be approximately 25 m in open water and 10 m near or on shore.
   i. Record the 4-letter alpha code for each species observed at the corresponding spatial location on the aerial map provided for each point.
   ii. Each individual bird observed must be recorded, whether you were able to identify it or not. Individuals which cannot be positively identified should be recorded as unidentified (e.g. unidentified sparrow (USPA), unidentified passerine (UPBD). See < http://www.birdpop.org/alphacodes.htm > for alpha codes). The inability to identify every individual bird is expected. However, not recording individuals because you are unable to identify them is not acceptable, as this can greatly affect survey results.

d. Flyover observations will be excluded because these birds are not using the study area.

e. Record the behavior of the individual. Notation is listed below and on each data sheet. For instance, if it was singing, circle the alpha code; if it was calling, underline it. "Observed" means you saw the bird and it wasn't doing anything else such as calling, singing, or drumming. NOTE: record the "highest" level of observation. For instance, if a bird is first observed calling and later sings, record that observation as singing. This is most important to record during the breeding season when territorial males are singing.

   i. 1. The order of observations is as follows (highest to lowest):
      1. a. 2 males simultaneous singing
         b. Singing/woodpecker drumming
         c. Calling
         d. Observed (sight only)

| | | | | |
|---|---|---|---|---|
| NAWA | NAWA | NAWA | NAWA \| ~~\|-NA~~WA | DOWO$_D$ |
| observed | calling | singing | 2 males simultaneous singing | woodpecker drumming |

f. For surveys conducted during the breeding season (June-July), record the breeding evidence code by using a subscript after the alpha code. Evidence codes can be found, along with descriptions, see <http://www.mnbba.org/pdf/BreedingEvidenceCodes_Tips.pdf >. Record the "highest" level of breeding evidence. For instance, if a bird is first observed doing a distraction display and later you see it occupying a nest, record it as occupied nest. This is a definite breeding observation, whereas a distraction display is a probable breeding observation.
   i. **Examples:**

| | | |
|---|---|---|
| TRES$_{ON}$ | MOWA$_{NB}$ | RWBL$_{FY}$ |
| Observed an occupied nest cavity of a Tree Swallow (adult seen entering/exiting) | Observed a Mourning Warbler building a nest | Observed a Red-winged Blackbird carrying food for young |

g. If a bird moves to a different location during the survey, only record the location where the bird was originally detected within the site. If a bird is initially not using the site but moves in during the survey, it should be recorded.

h. If a bird is detected at multiple points, record it on the data sheet for each of the points where it is observed. The location where the bird was first detected is where the observation should be recorded. At all other locations where the bird was observed record the bird and use a superscript asterisk. In the site description/notes section, write that this bird is a duplicate seen at point X. When entering the data, do not enter birds that have an asterisk denoting a duplicate observation.

i. Observations of large groups of birds (single species) should be recorded with the number of individuals in parentheses in front of the species code. For example, a group of 80 Double-crested Cormorants observed on the water would be recorded as:

(80) DCCO

j. Aerial foragers that <u>are foraging</u> should be recorded. A bird that is aerial foraging is using the airspace above the territory for foraging, catching insects in the air, using the airspace for fishing (terns), etc. It is different from a flyover in that a bird flying over the territory is traveling, not foraging.

7. Breeding Season Surveys
    a. During the 4 breeding season surveys, surveys will last 15 minutes and will be broken down in the following way;
        i. 0-5 minutes: passive listening (0:00 to 5:00)
        ii. 5-10 minutes: broadcast (5:00 to 10:00)
        iii. 10-15 minutes: passive listening (10:00 to 15:00)
    b. Equipment must be capable of broadcasting at an 80 dB level with minimal distortion. A decibel meter should be used at the beginning of the first survey each day to determine that speakers are projecting at 80dB at 1m distance from the speaker.
    c. Hold speaker above the level of vegetation and broadcast in the direction of the site you are surveying.
    d. Broadcast order:
        i. 30 seconds LEAST BITTERN (LEBI)
        ii. 30 seconds silence
        iii. 30 seconds SORA (SORA)
        iv. 30 seconds silence
        v. 30 seconds VIRGINIA RAIL (VIRA)
        vi. 30 seconds silence
        vii. 30 seconds COMMON MOORHEN(COMO)
        viii. 30 seconds silence
        ix. 30 seconds PIED-BILLED GREBE (PBGR)
        x. 30 seconds silence

8. Data Management
    a. Crews will check over data sheets after each survey, checking that all fields have been filled in, filled in properly and for readability.

    b. Data sheets will be maintained at the Natural Resources Research Institute in Duluth, Minnesota. Results from the field surveys will be stored in an excel database. They will eventually be deposited in a location to be designated by the MPCA project officer.

c. Recommended prep for entering data:
    i. Using a red ultra-fine sharpie marker, number each species code/observation in sequential order on the data sheet. This method allows you to easily follow along the numbering system during actual entry into the database and helps to eliminate mistakes.

9. Safety, Materials & Equipment
    a. a. Because bird surveys are being conducted during daylight hours, observers may survey alone but are required to check in with their field crew leader on a daily basis. Field crew leaders will work out a feasible daily check-in system with their crew to ensure safety in the field.
    b. This survey can be a single or multiple observer protocol.
    c. Surveyors will be equipped with the following:
        i. Data sheets
        ii. Standard Operating Procedures
        iii. Clipboard
        iv. Waterproof, permanent pens/markers (Rite in the Rain pen, ultra-fine tip Sharpie marker)
        v. Thermometer, in metal or plastic case
        vi. Site/point map(s)
        vii. GPS unit, with points loaded
        viii. Extra batteries
        ix. Each crew will carry spare equipment and materials

**Appendix B.** List of all 197 species observed in the St. Louis River Estuary (1977-2015). Not all of the species included in this list were included in the analysis. Excluded species were those only observed as flyovers or that fell outside of the survey boundaries (for historical vs. recent surveys). The 4-letter (English Name) Alpha Code and Scientific Name are listed for each bird species in accordance with the 56th AOU Supplement (2015). Species detected in a survey = Yes and species not detected = No.

| English Name | Scientific Name | Taxa Code | Historical | Recent |
|---|---|---|---|---|
| Snow Goose | *Chen caerulescens* | SNGO | Yes | Yes |
| Canada Goose | *Branta canadensis* | CANG | Yes | Yes |
| Trumpeter Swan | *Cygnus buccinator* | TRUS | No | Yes |
| Tundra Swan | *Cygnus columbianus* | TUSW | Yes | Yes |
| Wood Duck | *Aix sponsa* | WODU | Yes | Yes |
| Gadwall | *Anas strepera* | GADW | Yes | Yes |
| American Wigeon | *Anas americana* | AMWI | Yes | Yes |
| American Black Duck | *Anas rubripes* | ABDU | Yes | Yes |
| American Black Duck X Mallard Hybrid | *Anas rubripes x platy.* | ABDH | No | Yes |
| Mallard | *Anas platyrhynchos* | MALL | Yes | Yes |
| Blue-winged Teal | *Anas discors* | BWTE | Yes | Yes |
| Northern Shoveler | *Anas clypeata* | NSHO | Yes | Yes |
| Northern Pintail | *Anas acuta* | NOPI | Yes | Yes |
| Green-winged Teal | *Anas crecca* | GWTE | Yes | Yes |
| Canvasback | *Aythya valisineria* | CANV | No | Yes |
| Redhead | *Aythya americana* | REDH | Yes | Yes |
| Ring-necked Duck | *Aythya collaris* | RNDU | Yes | Yes |
| Greater Scaup | *Aythya marila* | GRSC | No | Yes |
| Lesser Scaup | *Aythya affinis* | LESC | Yes | Yes |
| White-winged Scoter | *Melanitta fusca* | WWSC | No | Yes |
| Bufflehead | *Bucephala albeola* | BUFF | Yes | Yes |
| Common Goldeneye | *Bucephala clangula* | COGO | Yes | Yes |
| Hooded Merganser | *Lophodytes cucullatus* | HOME | Yes | Yes |
| Common Merganser | *Mergus merganser* | COME | Yes | Yes |
| Red-breasted Merganser | *Mergus serrator* | RBME | Yes | Yes |
| Ruddy Duck | *Oxyura jamaicensis* | RUDU | Yes | Yes |
| Ring-necked Pheasant | *Phasianus colchicus* | RNEP | Yes | No |
| Ruffed Grouse | *Bonasa umbellus* | RUGR | Yes | No |
| Common Loon | *Gavia immer* | COLO | Yes | Yes |
| Pied-billed Grebe | *Podilymbus podiceps* | PBGR | Yes | Yes |
| Horned Grebe | *Podiceps auritus* | HOGR | Yes | Yes |
| Red-necked Grebe | *Podiceps grisegena* | RNGR | No | Yes |
| Double-crested Cormorant | *Phalacrocorax auritus* | DCCO | Yes | Yes |
| American White Pelican | *Pelecanus erythrorhynchos* | AWPE | No | Yes |
| American Bittern | *Botaurus lentiginosus* | AMBI | Yes | Yes |
| Least Bittern | *lxobrychus exilis* | LEBI | Yes | No |

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| English Name | Scientific Name | Taxa Code | Historical | Recent |
|---|---|---|---|---|
| Great Blue Heron | *Ardea herodias* | GBHE | Yes | Yes |
| Great Egret | *Ardea alba* | GREG | No | Yes |
| Green Heron | *Butorides virescens* | GRHE | Yes | Yes |
| Black-crowned Night-Heron | *Nycticorax nycticorax* | BCNH | Yes | Yes |
| Turkey Vulture | *Cathartes aura* | TUVU | Yes | Yes |
| Osprey | *Pandion haliaetus* | OSPR | No | Yes |
| Bald Eagle | *Haliaeetus leucocephalus* | BAEA | Yes | Yes |
| Northern Harrier | *Circus cyaneus* | NOHA | Yes | Yes |
| Sharp-shinned Hawk | *Accipiter striatus* | SSHA | Yes | Yes |
| Cooper's Hawk | *Accipiter cooperii* | COHA | No | Yes |
| Broad-winged Hawk | *Buteo platypterus* | BWHA | Yes | Yes |
| Red-tailed Hawk | *Buteo jamaicensis* | RTHA | Yes | Yes |
| Rough-legged Hawk | *Buteo lagopus* | RLHA | Yes | Yes |
| Virginia Rail | *Rallus limicola* | VIRA | Yes | Yes |
| Sora | *Porzana carolina* | SORA | Yes | Yes |
| American Coot | *Fulica americana* | AMCO | Yes | Yes |
| Sandhill Crane | *Grus canadensis* | SACR | No | Yes |
| Black-bellied Plover | *Pluvialis squatarola* | BBPL | Yes | Yes |
| American Golden-Plover | *Pluvialis dominica* | AMGP | Yes | Yes |
| Semipalmated Plover | *Charadrius semipalmatus* | SEPL | Yes | No |
| Killdeer | *Charadrius vociferus* | KILL | Yes | Yes |
| Spotted Sandpiper | *Actitis macularius* | SPSA | Yes | Yes |
| Solitary Sandpiper | *Tringa solitaria* | SOSA | Yes | No |
| Greater Yellowlegs | *Tringa melanoleuca* | GRYE | Yes | Yes |
| Willet | *Tringa semipalmata* | WILL | No | Yes |
| Lesser Yellowlegs | *Tringa flavipes* | LEYE | Yes | Yes |
| Stilt Sandpiper | *Calidris himantopus* | STSA | Yes | Yes |
| Sanderling | *Calidris alba* | SAND | Yes | No |
| Dunlin | *Calidris alpina* | DUNL | Yes | Yes |
| Baird's Sandpiper | *Calidris bairdii* | BASA | No | Yes |
| Least Sandpiper | *Calidris minutilla* | LESA | Yes | Yes |
| White-rumped Sandpiper | *Calidris fuscicollis* | WRSA | Yes | Yes |
| Buff-breasted Sandpiper | *Calidris subruficollis* | BBSA | No | Yes |
| Pectoral Sandpiper | *Calidris melanotos* | PESA | Yes | Yes |
| Semipalmated Sandpiper | *Calidris pusilla* | SESA | Yes | Yes |
| Wilson's Snipe | *Gallinago delicata* | WISN | Yes | Yes |
| Wilson's Phalarope | *Phalaropus tricolor* | WIPH | Yes | No |
| Bonaparte's Gull | *Chroicocephalus philadelphia* | BOGU | Yes | Yes |
| Ring-billed Gull | *Larus delawarensis* | RBGU | Yes | Yes |
| Herring Gull | *Larus argentatus* | HERG | Yes | Yes |
| Lesser Black-backed Gull | *Larus fuscus* | LBBG | No | Yes |

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| English Name | Scientific Name | Taxa Code | Historical | Recent |
|---|---|---|---|---|
| Glaucous Gull | *Larus hyperboreus* | GLGU | No | Yes |
| Great Black-backed Gull | *Larus marinus* | GBBG | No | Yes |
| Caspian Tern | *Hydroprogne caspia* | CATE | Yes | No |
| Black Tern | *Chlidonias niger* | BLTE | Yes | No |
| Common Tern | *Sterna hirundo* | COTE | Yes | Yes |
| Forster's Tern | *Sterna forsteri* | FOTE | Yes | No |
| Rock Pigeon | *Columba livia* | ROPI | Yes | Yes |
| Mourning Dove | *Zenaida macroura* | MODO | Yes | Yes |
| Black-billed Cuckoo | *Coccyzus erythropthalmus* | BBCU | No | Yes |
| Common Nighthawk | *Chordeiles minor* | CONI | Yes | Yes |
| Chimney Swift | *Chaetura pelagica* | CHSW | Yes | Yes |
| Ruby-throated Hummingbird | *Archilochus colubris* | RTHU | Yes | Yes |
| Belted Kingfisher | *Megaceryle alcyon* | BEKI | Yes | Yes |
| Red-headed Woodpecker | *Melanerpes erythrocephalus* | RHWO | Yes | No |
| Red-bellied Woodpecker | *Melanerpes carolinus* | RBWO | No | Yes |
| Yellow-bellied Sapsucker | *Sphyrapicus varius* | YBSA | No | Yes |
| Downy Woodpecker | *Picoides pubescens* | DOWO | Yes | Yes |
| Hairy Woodpecker | *Picoides villosus* | HAWO | Yes | Yes |
| Northern Flicker | *Colaptes auratus* | NOFL | Yes | Yes |
| Pileated Woodpecker | *Dryocopus pileatus* | PIWO | No | Yes |
| American Kestrel | *Falco sparverius* | AMKE | Yes | Yes |
| Merlin | *Falco columbarius* | MERL | Yes | Yes |
| Peregrine Falcon | *Falco peregrinus* | PEFA | No | Yes |
| Alder Flycatcher | *Empidonax alnorum* | ALFL | Yes | Yes |
| Least Flycatcher | *Empidonax minimus* | LEFL | Yes | Yes |
| Eastern Phoebe | *Sayornis phoebe* | EAPH | Yes | Yes |
| Great Crested Flycatcher | *Myiarchus crinitus* | GCFL | Yes | Yes |
| Eastern Kingbird | *Tyrannus tyrannus* | EAKI | Yes | Yes |
| Northern Shrike | *Lanius excubitor* | NSHR | Yes | Yes |
| Yellow-throated Vireo | *Vireo flavifrons* | YTVI | No | Yes |
| Blue-headed Vireo | *Vireo solitarius* | BHVI | No | Yes |
| Warbling Vireo | *Vireo gilvus* | WAVI | Yes | Yes |
| Red-eyed Vireo | *Vireo olivaceus* | REVI | Yes | Yes |
| Gray Jay | *Perisoreus canadensis* | GRAJ | Yes | No |
| Blue Jay | *Cyanocitta cristata* | BLJA | Yes | Yes |
| American Crow | *Corvus brachyrhynchos* | AMCR | Yes | Yes |
| Common Raven | *Corvus corax* | CORA | No | Yes |
| Horned Lark | *Eremophila alpestris* | HOLA | No | Yes |
| Purple Martin | *Progne subis* | PUMA | Yes | No |
| Tree Swallow | *Tachycineta bicolor* | TRES | Yes | Yes |
| Northern Rough-winged Swallow | *Stelgidopteryx serripennis* | NRWS | Yes | Yes |

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| English Name | Scientific Name | Taxa Code | Historical | Recent |
|---|---|---|---|---|
| Bank Swallow | *Riparia riparia* | BANS | Yes | Yes |
| Cliff Swallow | *Petrochelidon pyrrhonota* | CLSW | Yes | Yes |
| Barn Swallow | *Hirundo rustica* | BARS | Yes | Yes |
| Black-capped Chickadee | *Poecile atricapillus* | BCCH | Yes | Yes |
| Red-breasted Nuthatch | *Sitta canadensis* | RBNU | No | Yes |
| White-breasted Nuthatch | *Sitta carolinensis* | WBNU | No | Yes |
| Brown Creeper | *Certhia americana* | BRCR | No | Yes |
| House Wren | *Troglodytes aedon* | HOWR | Yes | Yes |
| Winter Wren | *Troglodytes hiemalis* | WIWR | Yes | Yes |
| Sedge Wren | *Cistothorus platensis* | SEWR | Yes | Yes |
| Marsh Wren | *Cistothorus palustris* | MAWR | Yes | Yes |
| Golden-crowned Kinglet | *Regulus satrapa* | GCKI | No | Yes |
| Ruby-crowned Kinglet | *Regulus calendula* | RCKI | Yes | Yes |
| Veery | *Catharus fuscescens* | VEER | Yes | Yes |
| Gray-cheeked Thrush | *Catharus minimus* | GCTH | Yes | No |
| Swainson's Thrush | *Catharus ustulatus* | SWTH | No | Yes |
| Hermit Thrush | *Catharus guttatus* | HETH | No | Yes |
| American Robin | *Turdus migratorius* | AMRO | Yes | Yes |
| Gray Catbird | *Dumetella carolinensis* | GRCA | Yes | Yes |
| Brown Thrasher | *Toxostoma rufum* | BRTH | Yes | Yes |
| European Starling | *Sturnus vulgaris* | EUST | Yes | Yes |
| American Pipit | *Anthus rubescens* | AMPI | Yes | Yes |
| Cedar Waxwing | *Bombycilla cedrorum* | CEDW | Yes | Yes |
| Lapland Longspur | *Calcarius lapponicus* | LALO | Yes | Yes |
| Snow Bunting | *Plectrophenax nivalis* | SNBU | Yes | Yes |
| Ovenbird | *Seiurus aurocapilla* | OVEN | Yes | Yes |
| Northern Waterthrush | *Parkesia noveboracensis* | NOWA | Yes | Yes |
| Black-and-white Warbler | *Mniotilta varia* | BAWW | No | Yes |
| Tennessee Warbler | *Oreothlypis peregrina* | TEWA | Yes | No |
| Orange-crowned Warbler | *Oreothlypis celata* | OCWA | Yes | Yes |
| Nashville Warbler | *Oreothlypis ruficapilla* | NAWA | Yes | Yes |
| Mourning Warbler | *Geothlypis philadelphia* | MOWA | Yes | Yes |
| Common Yellowthroat | *Geothlypis trichas* | COYE | Yes | Yes |
| American Redstart | *Setophaga ruticilla* | AMRE | Yes | Yes |
| Cape May Warbler | *Setophaga tigrina* | CMWA | No | Yes |
| Northern Parula | *Setophaga americana* | NOPA | No | Yes |
| Magnolia Warbler | *Setophaga magnolia* | MAWA | Yes | Yes |
| Blackburnian Warbler | *Setophaga fusca* | BLBW | No | Yes |
| Yellow Warbler | *Setophaga petechia* | YEWA | Yes | Yes |
| Chestnut-sided Warbler | *Setophaga pensylvanica* | CSWA | Yes | Yes |
| Blackpoll Warbler | *Setophaga striata* | BLPW | Yes | Yes |

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| English Name | Scientific Name | Taxa Code | Historical | Recent |
|---|---|---|---|---|
| Palm Warbler | *Setophaga palmarum* | PAWA | Yes | Yes |
| Yellow-rumped Warbler | *Setophaga coronata* | YRWA | Yes | Yes |
| Black-throated Green Warbler | *Setophaga virens* | BTNW | No | Yes |
| Canada Warbler | *Cardellina canadensis* | CAWA | No | Yes |
| Wilson's Warbler | *Cardellina pusilla* | WIWA | Yes | Yes |
| American Tree Sparrow | *Spizelloides arborea* | ATSP | Yes | Yes |
| Chipping Sparrow | *Spizella passerina* | CHSP | Yes | Yes |
| Clay-colored Sparrow | *Spizella pallida* | CCSP | Yes | Yes |
| Vesper Sparrow | *Pooecetes gramineus* | VESP | Yes | Yes |
| Savannah Sparrow | *Passerculus sandwichensis* | SAVS | Yes | Yes |
| Fox Sparrow | *Passerella iliaca* | FOSP | No | Yes |
| Song Sparrow | *Melospiza melodia* | SOSP | Yes | Yes |
| Lincoln's Sparrow | *Melospiza lincolnii* | LISP | Yes | Yes |
| Swamp Sparrow | *Melospiza georgiana* | SWSP | Yes | Yes |
| White-throated Sparrow | *Zonotrichia albicollis* | WTSP | Yes | Yes |
| Harris's Sparrow | *Zonotrichia querula* | HASP | Yes | No |
| White-crowned Sparrow | *Zonotrichia leucophrys* | WCSP | Yes | Yes |
| Dark-eyed Junco | *Junco hyemalis* | DEJU | Yes | Yes |
| Scarlet Tanager | *Piranga olivacea* | SCTA | No | Yes |
| Northern Cardinal | *Cardinalis cardinalis* | NOCA | No | Yes |
| Rose-breasted Grosbeak | *Pheucticus ludovicianus* | RBGR | Yes | Yes |
| Indigo Bunting | *Passerina cyanea* | INBU | Yes | No |
| Red-winged Blackbird | *Agelaius phoeniceus* | RWBL | Yes | Yes |
| Yellow-headed Blackbird | *Xanthocephalus xanthocephalus* | YHBL | Yes | No |
| Rusty Blackbird | *Euphagus carolinus* | RUBL | Yes | Yes |
| Brewer's Blackbird | *Euphagus cyanocephalus* | BRBL | Yes | No |
| Common Grackle | *Quiscalus quiscula* | COGR | No | Yes |
| Brown-headed Cowbird | *Molothrus ater* | BHCO | Yes | Yes |
| Baltimore Oriole | *Icterus galbula* | BAOR | Yes | Yes |
| Pine Grosbeak | *Pinicola enucleator* | PIGR | Yes | No |
| House Finch | *Haemorhous mexicanus* | HOFI | No | Yes |
| Purple Finch | *Haemorhous purpureus* | PUFI | Yes | Yes |
| Common Redpoll | *Acanthis flammea* | CORE | Yes | Yes |
| Pine Siskin | *Spinus pinus* | PISI | Yes | Yes |
| American Goldfinch | *Spinus tristis* | AMGO | Yes | Yes |
| Evening Gosbeak | *Coccothraustes vespertinus* | EVGR | Yes | No |
| House Sparrow | *Passer domesticus* | HOSP | Yes | Yes |

**Appendix C.** Total number of birds observed per bird group per site for current surveys of R2R, reference, and additional sites, excluding flyover observations.

| Site | Blackbird | Corvid | Dove | Gull | Hummingbird | Invasive | Pigeon | Rail | Raptor | Shorebird | Songbird | Waterbird | Waterfowl | Woodpecker |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 21st Avenue West | 227 | 2256 | 0 | 64839 | 0 | 3284 | 170 | 0 | 29 | 70 | 915 | 543 | 13406 | 5 |
| 40th Avenue West | 557 | 156 | 2 | 954 | 1 | 34 | 19 | 5 | 39 | 196 | 1368 | 412 | 9366 | 38 |
| Cedar Yard Bay | 105 | 5 | 0 | 43 | 0 | 0 | 0 | 2 | 0 | 0 | 74 | 47 | 138 | 0 |
| Clough Island | 24 | 14 | 0 | 37 | 0 | 0 | 0 | 1 | 6 | 0 | 71 | 16 | 68 | 10 |
| Grassy Point | 363 | 7 | 5 | 8 | 1 | 1 | 0 | 1 | 13 | 17 | 365 | 73 | 894 | 9 |
| Horseshoe Bay | 1 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 7 | 64 | 80 | 1 |
| Kingsbury Bay | 294 | 15 | 1 | 7 | 1 | 4 | 0 | 8 | 3 | 5 | 325 | 56 | 750 | 4 |
| Little Pokegema Bay | 57 | 10 | 0 | 2 | 1 | 0 | 0 | 2 | 8 | 0 | 90 | 216 | 241 | 3 |
| Minnesota Point | 91 | 33 | 0 | 164 | 0 | 4 | 0 | 1 | 1 | 23 | 184 | 22 | 281 | 13 |
| Minnesota Slip | 0 | 12 | 0 | 201 | 0 | 20 | 28 | 0 | 0 | 1 | 37 | 114 | 78 | 0 |
| Mud Lake | 255 | 22 | 1 | 7 | 0 | 0 | 0 | 7 | 5 | 5 | 287 | 139 | 1623 | 6 |
| North Bay | 220 | 24 | 0 | 7 | 1 | 0 | 0 | 11 | 2 | 2 | 183 | 430 | 376 | 6 |
| Perch Lake | 39 | 9 | 0 | 0 | 0 | 0 | 0 | 2 | 3 | 0 | 76 | 54 | 561 | 3 |
| Pokegema Bay | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 4 | 0 | 3 |
| Rask Bay | 33 | 9 | 0 | 13 | 0 | 0 | 0 | 5 | 7 | 0 | 85 | 1615 | 1058 | 4 |
| Slip C | 2 | 2 | 1 | 2 | 0 | 2 | 26 | 0 | 1 | 0 | 149 | 4 | 54 | 0 |
| Southworth Marsh | 9 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 28 | 2 | 1 | 0 |
| Spirit Lake East | 41 | 8 | 0 | 1 | 0 | 0 | 0 | 0 | 6 | 0 | 87 | 41 | 311 | 4 |
| Spirit Lake West | 115 | 14 | 0 | 40 | 0 | 0 | 0 | 0 | 16 | 2 | 141 | 50 | 788 | 5 |
| Stryker Bay | 18 | 12 | 0 | 2 | 0 | 0 | 8 | 0 | 0 | 0 | 24 | 6 | 106 | 2 |
| Weasel Bay | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 8 | 93 | 13 | 2 |

**Appendix D.** Species not observed in historical or recent surveys. This list includes all species observations.

| Species | Historical | Recent | Species | Historical | Recent |
|---|---|---|---|---|---|
| Trumpeter Swan | 0 | 6 | Eastern Phoebe | 0 | 2 |
| Northern Shoveler | 0 | 20 | Northern Shrike | 0 | 1 |
| Canvasback | 0 | 113 | Common Raven | 0 | 11 |
| Greater Scaup | 0 | 8 | Purple Martin | 3 | 0 |
| Red-breasted Merganser | 0 | 24 | Bank Swallow | 37 | 0 |
| Ring-necked Pheasant | 2 | 0 | White-breasted Nuthatch | 0 | 2 |
| Red-necked Grebe | 0 | 5 | Winter Wren | 0 | 1 |
| American White Pelican | 0 | 19 | Golden-crowned Kinglet | 0 | 1 |
| American Bittern | 4 | 0 | Gray-cheeked Thrush | 1 | 0 |
| Least Bittern | 3 | 0 | Swainson's Thrush | 0 | 1 |
| Great Egret | 0 | 3 | Brown Thrasher | 4 | 0 |
| Black-crowned Night-Heron | 2 | 0 | Snow Bunting | 27 | 0 |
| Turkey Vulture | 0 | 8 | Black-and-white Warbler | 0 | 1 |
| Cooper's Hawk | 0 | 2 | Tennessee Warbler | 5 | 0 |
| Rough-legged Hawk | 11 | 0 | Mourning Warbler | 0 | 1 |
| Black-bellied Plover | 22 | 0 | Cape May Warbler | 0 | 2 |
| Semipalmated Plover | 15 | 0 | Magnolia Warbler | 1 | 0 |
| Killdeer | 244 | 0 | Blackburnian Warbler | 0 | 1 |
| Solitary Sandpiper | 2 | 0 | Blackpoll Warbler | 0 | 1 |
| Sanderling | 2 | 0 | Canada Warbler | 0 | 1 |
| Dunlin | 36 | 0 | Wilson's Warbler | 0 | 4 |
| White-rumped Sandpiper | 2 | 0 | Clay-colored Sparrow | 1 | 0 |
| Wilson's Phalarope | 15 | 0 | Vesper Sparrow | 3 | 0 |
| Black Tern | 52 | 0 | Savannah Sparrow | 20 | 0 |
| Forster's Tern | 2 | 0 | Indigo Bunting | 1 | 0 |
| Black-billed Cuckoo | 0 | 1 | Yellow-headed Blackbird | 43 | 0 |
| Common Nighthawk | 1 | 0 | Brewer's Blackbird | 3 | 0 |
| Red-bellied Woodpecker | 0 | 1 | Evening Gosbeak | 31 | 0 |
| Yellow-bellied Sapsucker | 0 | 1 | House Sparrow | 7 | 0 |
| Pileated Woodpecker | 0 | 6 | | | |
| Peregrine Falcon | 0 | 4 | | | |

# Appendix B

# Interstate Island Common Tern Monitoring Data

# (1977-2021)

## St. Louis River Estuary History Annual Summary

| Year | Peak Nest Count | Number Young Fledged (4A+) | Number Young Fledged/Peak Nest Count | Notes, Important Events, Observations |
|---|---|---|---|---|
| 1977 | 208 | | | Tom Davis reports: No nest counts given, "breeding populations were determined by counting all nests." |
| 1978 | 178 | | | Tom Davis reports: No nest counts given, "breeding populations were determined by counting all nests." |
| 1979 | 210 | | | Tom Davis reports: No nest counts given, "breeding populations were determined by counting all nests." |
| 1980 | 174 | | | Tom Davis reports: No nest counts given, "breeding populations were determined by counting all nests." |
| 1981 | 237 | | | Tom Davis reports: No nest counts given, "breeding populations were determined by counting all nests." |
| 1982 | 207 | | | Tom Davis reports: No nest counts given, "breeding populations were determined by counting all nests." |
| 1983 | 198 | | | 1983 to 1989 from Penning, 1989 |
| 1984 | 161 | | | |
| 1985 | 140 | 0 | | Fox predation, non breeders present. |
| 1986 | 68 | 0 | | Mink, GH Owl, fox predation?, non-breeders present. |
| 1987 | 87 | 0 | | Mammal predation & human disturbance, non-breeders present. |
| 1988 | 88 | 0 | | WI PT: mink predation and storm destroyed nests, non-breeders present. |
| 1989 | 81 | 64 | 0.79 | Habitat work at Interstate completed, terns colonize. WI PT: mink predation & storm destroyed nests. |
| 1990 | 124 | 168 | 1.35 | All at Interstate Island. |
| 1991 | 152 | 199 | 1.31 | Possible GH Owl or P Falcon predation. Adults present which were banded as chicks in 1988 at Ashland. |
| 1992 | 107 | 146 | 1.36 | Adults present which were banded as chicks in 1989 at II. |
| 1993 | 162 | 141 | 0.87 | |
| 1994 | 212 | 197 | 0.92 | |
| 1995 | 264 | 313 | 1.19 | Herring gull predation. |
| 1996 | 231 | 166 | 0.72 | Herring gull predation. |
| 1997 | 208 | 92 | 0.44 | Herring gull predation. |
| 1998 | 226 | 124 | 0.55 | |
| 1999 | 215 | 254 | 1.18 | GH Owl predation at end of the season |
| 2000 | 197 | 217 | 1.10 | |
| 2001 | 199 | 141 | 0.71 | Franklin GS egg predation early, excessive vegetation later |
| 2002 | 195 | 136 | 0.70 | Unidentified predation later part of fledging period |
| 2003 | 191 | 139 | 0.73 | Fledging probably higher than data shows. |
| 2004 | 204 | 201 | 0.99 | |
| 2005 | 189 | 130 | 0.75 | ST Weasel egg and at least 1 adult predation early. P Falcon predation on young later. Fledging probably better than data shows. |
| 2006 | 166 | 6 | 0.04 | Chronic egg and chick predation by unknown predator/s |
| 2007 | 180 | 307 | 1.71 | egg predation early by unknown predator/s |
| 2008 | 202 | 283 | 1.40 | some chick predation late in season by unknown predator |
| 2009 | 180 | 109 | 0.61 | egg predation early by unknown predator/s |
| 2010 | 238 | 210 | 0.88 | Less nest losses increased nesting synchrony and hence the peak nest count |
| 2011 | 302 | 173 | 0.57 | Prolonged cold & wet weather at the peak of hatching in mid June caused significant mortality. Substantial renesting was partially successful. Population increase is do to past reproduction: 36% of adults were hatched in 2007 & 08. |
| 2012 | 161 | 0 | 0.00 | Repeated egg predation by unidentified predator/s. Trapping was unsuccessful. Abandoned by early July |
| 2013 | 266 | 285 | 1.07 | Nesting area reduced to 1/3 of former size. Geolocators placed on 15 birds. Some apparent avian predation late in the season. |
| 2014 | 174 | 0 | 0.00 | Chronic egg predation by gulls; and chick predation probably by gulls |
| 2015 | 101 | 42 | 0.42 | Chronic egg predation by gulls; and chick loss probably by gull predation. A substantial renesting effort was partially successful. A major habitat restoration work was completed in the fall. |
| 2016 | 162 | 200 | 1.23 | Unusual large number of dead adults, cause undetermined; AI sampling; GPS foraging study; GPS migration study |
| 2017 | 129 | 91 | 0.71 | GPS migration study initiated |
| 2018 | 131 | 128 | 0.98 | GPS migration study concluded |
| 2019 | 113 | 116 | 1.03 | Motus migration study initiated; Chronic egg predation by gulls |
| 2020 | 108 | 143 | 1.32 | Spring: Island restoration/stabilization project; Motus study concluded; Chronic egg predation by gulls; Fall: large island expansion and tern area restoration project, chain link fencing installed |
| 2021 | 101 | 119 | 1.18 | Chronic egg predation by gulls |
| **1989-'20 Averages** | 180 | 154 | 0.85 | |
| **1989-'21 Totals** | 5691 | 4886 | 0.86 | II History 9.9.21 |

# Appendix C

# St. Louis River AOC BUI 2 Fisheries Data

# St. Louis River Estuary Fisheries Statistics for BUI 2 Indicator Species

## Standard Gillnetting CPUE (Early July Sampling)

| Year | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Effort | 44 | 15 | | 14 | 15 | | 14 | 14 | | | 22 | 22 | 21 | 21 | 13 | 21 | 21 | | 21 | |
| Walleye | 0.80 | 4.40 | | 8.20 | 8.00 | | 5.40 | 4.60 | | | 4.80 | 3.00 | 4.50 | 2.30 | 4.30 | 3.90 | 4.50 | | 4.80 | |
| Lake Sturgeon | 0.00 | 0.00 | | 0.10 | 0.20 | | 1.40 | 0.10 | | | 2.90 | 2.60 | 3.30 | 3.00 | 5.80 | 4.20 | 6.50 | | 2.60 | |
| Lake Sturgeon # age 5 or less per lift | | | | | | | | | | | | | | 2.2 | 4.8 | 3.3 | 5.4 | | 2.1 | |

## Muskellunge Trap Net CPUE (April/May Sampling)

| Year | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Effort | | | | | | | | | | | | | | | | | | 28 | | |
| Muskellunge | | | | | | | | | | | | | | | | | | 2.6 | | |

## Gillnetting PSD - Summer

| Year | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Walleye | 44 | 44 | | 41 | 33 | 27 | 38 | 40 | | | 55 | 63 | 67 | 35 | 66 | 66 | 51 | | 38 | |

## Standard Gillnetting CPUE (Early July Sampling)

| Year | 2000 | 2001 | 2002 | 2003 | 2004* | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Effort | 20 | | 20 | 20 | | 21 | 20 | | 19 | 20 | 21 | | | 19 | 21 | 21 | 21 | 21 | 21 | 21 | 21 |
| Walleye | 8.40 | | 7.10 | 4.20 | | 5.80 | 6.30 | | 6.70 | 6.40 | 4.70 | | | 4.89 | 5.43 | 9.80 | 10.95 | 9.67 | 6.43 | 9.05 | 7.29 |
| Lake Sturgeon | 3.10 | | 3.80 | 1.60 | | 0.90 | 1.60 | | 0.30 | 1.00 | 0.50 | | | 0.16 | 0.14 | 0.14 | 0.14 | 0.14 | 0.05 | 0.10 | 0.19 |
| Lake Sturgeon # age 5 or less per lift | 1.7 | | 1.5 | 0.3 | | 0.2 | 0.1 | | 0 | 0 | 0 | | | 0 | 0 | 0.1 | 0.05 | 0.00 | 0.05 | 0.05 | 0.10 |

## Muskellunge Trap Net CPUE (April/May Sampling)

| Year | 2000 | 2001 | 2002 | 2003 | 2004* | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Effort | | | | | | | | 56 | | | 58 | | | 65 | | | | 629 | 312 | | |
| Muskellunge | | | | | | | | 2.4 | | | 1.4 | | | 2.9 | | | | 0.4 | 0.7 | | |
| Gillnetting PSD - Summer | | | | | | | | | | | | | | | | | | | | | |
| Year | 00 | 01 | 02 | 03 | 04 | 05 | 06 | 07 | 08 | 09 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 |
| Walleye | 45 | | 53 | 41 | 33 | 31 | 66 | | 37 | 40 | 62 | | | 68 | 44 | 23 | 27 | 22 | 39 | 64 | 46 |

***2004 data removed to different methodology and sampling season**

CPUE = Catch Per Unit Effort

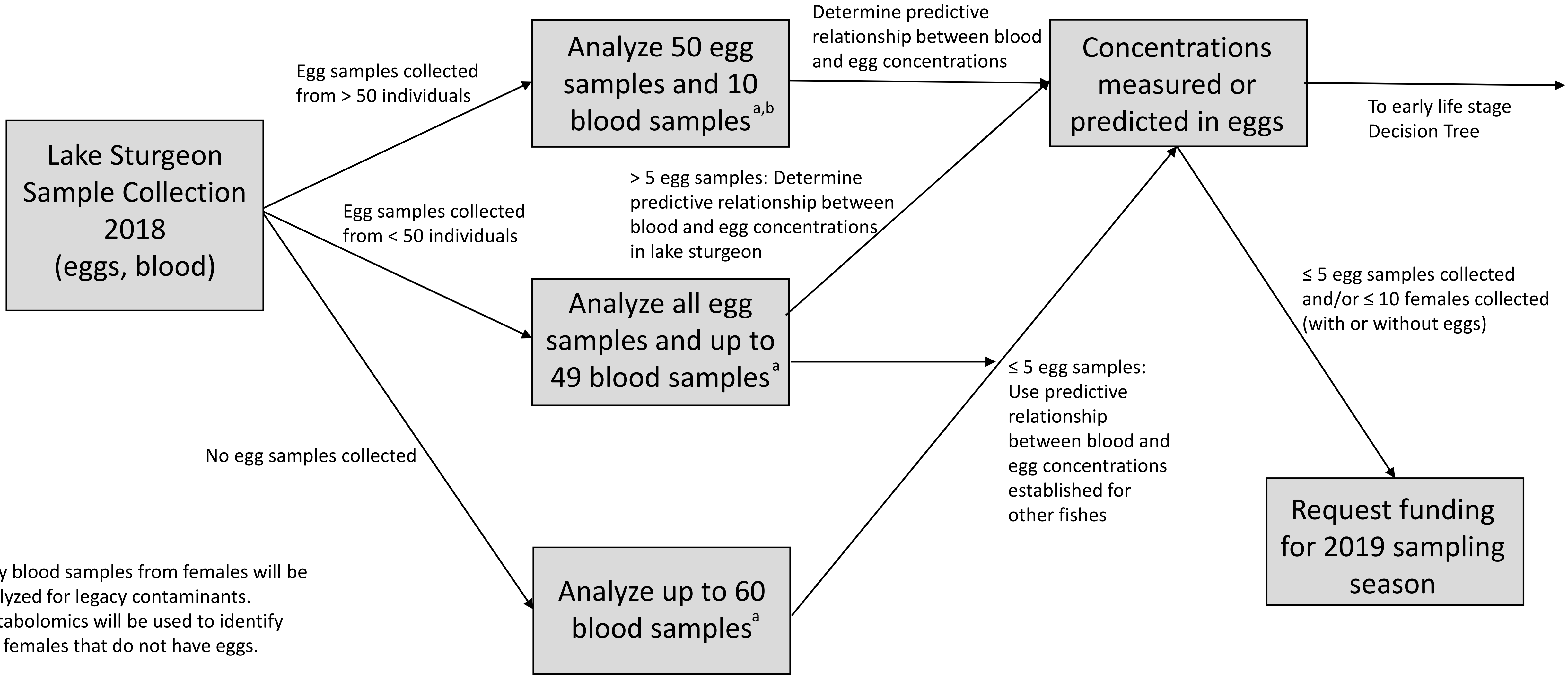
PSD = Proportional Stock Density

# Appendix D

# Lake Sturgeon Study Decision Tree

# and Final Report

# Decision Tree for Sample Selection for Chemical Analysis




[a] Only blood samples from females will be analyzed for legacy contaminants. Metabolomics will be used to identify any females that do not have eggs.

[b] Blood samples will be selected from individuals with a range of egg concentrations (low, medium, high), if a range in egg concentrations is present.

# Decision Tree for Assessment of Early Life Stages

**Total Toxicity of Legacy Contaminants**[a]

- Mortality predicted by model for ≥ 1 individual (Figure 1) → **Proceed: Develop Sturgeon-specific Total Toxicity**[b]
  - Mortality predicted by model for ≥ 1 individual (Figure 1) → **Proceed: Resource management action to be considered**
  - No mortality predicted by model (Figure 1) → **Stop: Exposure unlikely to cause elevated mortality**[c]
- No mortality predicted by model (Figure 1) → **Stop: Exposure unlikely to cause elevated mortality**[c]

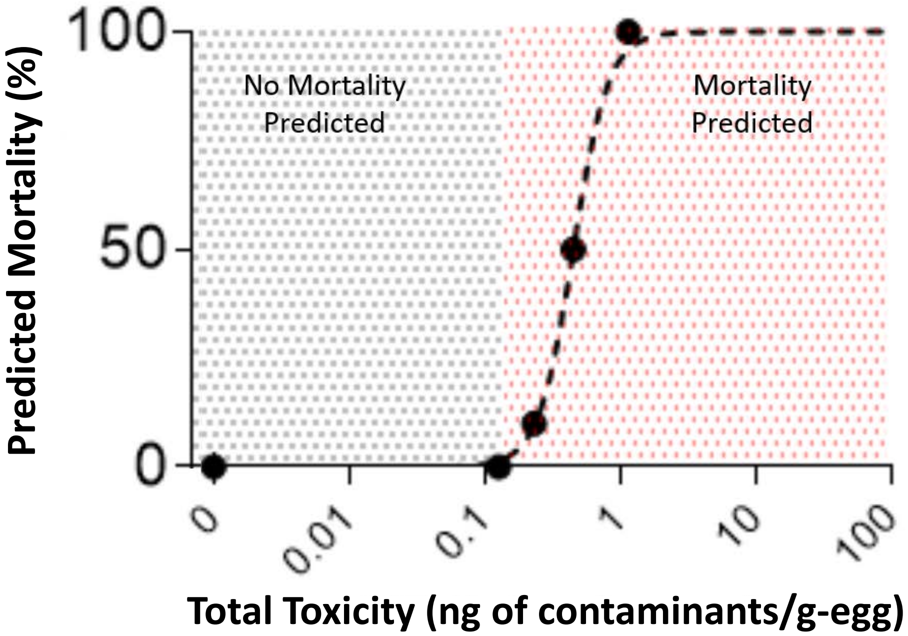
**Proceed: Resource management action to be considered** ↔ **Request funding for 2019 sampling season**: > 10-fold range in total toxicity of legacy contaminants among individuals

[a] Total toxicity of the mixture of legacy contaminants is calculated using established potencies based largely on rainbow trout and which are intended for initial screening purposes only.

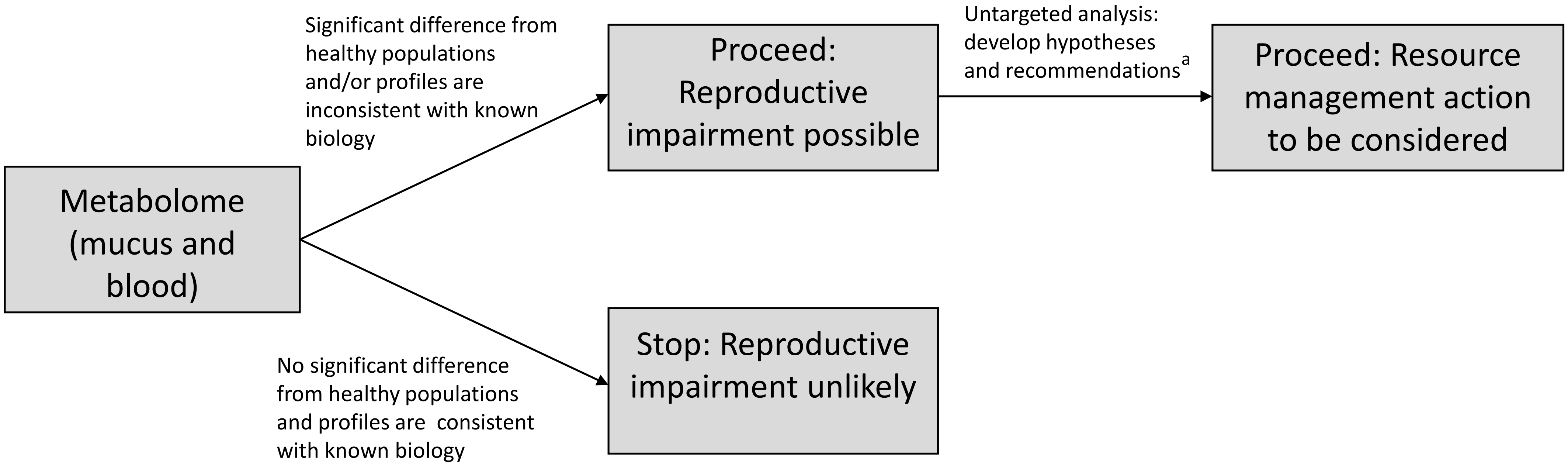
[b] Specific chemicals chosen for development of sturgeon-specific potency values will be based on expert opinion and take into consideration the concentrations measured, contribution to total toxicity, known differences in potency of different chemicals among species, and frequency of detection among individuals.

[c] Assessment only predicts mortality caused by exposure to chlorinated aromatic hydrocarbons (dioxins/furans and PCBs) and does not consider other contaminants (ex. metals, DDT, etc).

**Figure 1**: Model Output for Toxicity to Lake Sturgeon



# Decision Tree for Assessment of Adults




[a] Since metabolomics is an untargeted analysis that can measure thousands of metabolites it is not possible to develop concrete hypotheses at this time. Expert opinion will be used to interpret the data in order to make hypotheses and provide regulatory recommendations. Possible information derived from metabolomics include phenotypic sexing (even with no external sex characteristics), evidence of reproductive health or reproductive dysfunction, evidence of delayed maturation, and evidence of exposure to other chemicals of potential concern, such as polycyclic aromatic hydrocarbons, wastewater effluents, and others.

# Possible Scientifically Defensible Recommendations Resulting from this Research:

1) **No egg mortality predicted, no indications of reproductive impairment:** Recruitment failure is likely a result of factors unrelated to the Area of Concern.

2) **Elevated egg mortality predicted, no indications of reproductive impairment:** Recruitment failure is likely a result of bioaccumulated legacy contaminants being maternally transferred to embryos.

3) **No egg mortality predicted, indications of reproductive impairment:** Recruitment failure is likely a result of reduced reproductive potential of adults. Depending upon results of metabolomics, this reproductive impairment could be predicted to result from exposure to legacy contaminants, present day contaminants, or some other biotic or abiotic stressor. It is possible that results will suggest a delayed maturation and that healthy reproduction should occur in the future.

4) **Elevated egg mortality predicted, indications of reproductive impairment:** Recruitment failure is likely a combined result of maternally transferred legacy contaminants and reduced reproductive potential of adults. Suggestive of severe, contaminant related pressures on recruitment in lake sturgeon.

# Assessing whether maternal transfer of legacy dioxin-like chemicals is limiting lake sturgeon natural recruitment in the St. Louis River Area of Concern

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**ABSTRACT:** The purpose of this research is to answer questions about whether impairment of natural recruitment of lake sturgeon (*Acipenser fulvescens*) in the St. Louis River (SLR) Area of Concern (AOC) could be related to exposure to and bioaccumulation of legacy contaminants, such as polychlorinated dibenzo-*p*-dioxins/furans (PCDD/Fs) and polychlorinated biphenyls (PCBs), which are collectively known as dioxin-like chemicals (DLCs). Lake sturgeon are identified as an indicator species with specific population targets linked to the removal of SLR-AOC Beneficial Use Impairment (BUI) 2: Degraded Fish and Wildlife Populations. Despite past efforts to recover SLR-AOC lake sturgeon populations via fingerling stocking, recruitment is not being observed at anticipated levels and is not trending towards BUI targets. Therefore, resource managers and fisheries experts working in the SLR-AOC have identified a need to assess potential factors limiting lake sturgeon recovery and determine whether limiting factors are influenced by legacy contamination. In particular, DLCs can accumulate in sturgeons and be maternal transferred to embryos leading to decreased survival of early life-stages. Therefore, samples of eggs were collected from SLR-AOC lake sturgeon using a combination of egg mats and releases from ripe females collected during the MN DNR 2017 to 2019 sturgeon spawning assessments. Additionally, plasma was collected from adult female, male, and immature individuals. Concentrations of the 15 dioxin-like PCDD/Fs and 4 dioxin-like PCBs were quantified in each sample. Toxic equivalents (TEQs) in eggs and plasma were calculated using World Health Organization (WHO) toxic equivalency factors (TEFs) for fish and predicted lake sturgeon-specific TEFs developed from a quantitative adverse outcome pathway (qAOP) model. The calculated TEQs for eggs were compared to lake sturgeon mortality curves and TEQs for plasma were compared to a predicted effect threshold for adults. These comparisons determined that bioaccumulation and maternal transfer of DLCs is below levels that would be expected to cause any known toxicities in lake sturgeon. This evidence suggests that legacy contamination of DLCs is unlikely to be a factor in recruitment failure and provides support for other hypothesized drivers, possibly unrelated to the SLR-AOC.

**INTRODUCTION**

Historical municipal and industrial waste disposal and improper land-use practices before the onset of modern environmental protection laws have created a complex set of issues in the St. Louis River (SLR), a tributary of Lake Superior situated in Minnesota and Wisconsin. In 1987 the SLR was identified as a Great Lakes Area of Concern (AOC), leading to development of a comprehensive remedial action plan (RAP) to restore all beneficial use impairments (BUIs), including BUI-2: degraded fish and wildlife populations (EPA, 2020). As part of removal of BUI-2, lake sturgeon (*Acipenser fulvescens*) were identified as an indicator species with specific population targets (EPA, 2020). The population of lake sturgeon in the SLR were considered extirpated in the early 1900's due to a combination of habitat degradation, impaired water quality, and overharvest (Auer, 1996; Schram et al., 1999). Beginning in 1983, a restoration stocking program was initiated by the Minnesota Department of Natural Resources (MNDNR) and Wisconsin Department of Natural Resources (WDNR) to support recovery of the SLR lake sturgeon population in conjunction with restoration of spawning habitat, improved water quality through treating domestic and industrial effluent, and reduced exploitation through conservative fishing regulations (Schram et al., 1999). Fry and fingerling lake sturgeon from the Wolf River of the Lake Michigan watershed were stocked into the SLR from 1983 to 1994 and from the Sturgeon River of the Lake Superior watershed from 1998 to 2000 (Estep et al., 2020; Schram et al., 1999). In total, 781,000 fry and 142,180 fingerling lake sturgeon were stocked into the SLR (Estep et al., 2020). However, more than 30 years since the first fish were stocked, recruitment is not trending towards BUI-2 targets (SLR-AOC Fish Tech Team, personal communication). As a result, resource managers and fisheries experts working in the SLR-AOC have identified a need to assess potential factors that could be limiting lake sturgeon recovery.

Legacy contaminants could be one factor limited lake sturgeon natural recruitment in the SLR-AOC. Elevated levels of sediment-associated legacy contaminants have been documented in the SLR-AOC, including complex mixtures of polychlorinated biphenyl (PCB), polychlorinated dibenzo-*p*-dioxin (PCDD), and polychlorinated dibenzofuran (PCDF) congeners (Crane & MacDonald, 2003). A total of 12 PCBs, 7 PCDDs, and 10 PCDFs are collectively known as dioxin-like chemicals (DLCs) and have the greatest toxicity to wildlife due to their planar structure which allows them to bind with relatively great affinity to the aryl hydrocarbon receptor (AHR) (Denison & Nagy, 2003). The AHR is a ligand-activated transcription factor that regulates a range of physiological processes in vertebrates and whose dysregulation can cause adverse biological effects, including wasting syndrome, hepatotoxicity, immune suppression, impaired endocrinology, carcinogenesis, developmental deformities, and early life mortality (Elonen et al., 1998; Giesy et al., 2002; King-Heiden et al., 2012; Spitsbergen et al., 1986, Walter et al., 2000). DLCs can bioaccumulate in wildlife, undergo biomagnification through trophic levels, and be maternally transferred to embryos (Lohmann & Jones, 1998; Opperhuizen & Sijim, 1990). Lake sturgeon are uniquely susceptible to bioaccumulation of DLCs because they live in close association with sediments where these contaminants are most persistent and feed primarily on benthic organisms (Hochleithner & Gessner, 2001). Additionally, lake sturgeon are extremely long-lived (>100 years), do not spawn until late in life (12 to 27 years), and then only spawn intermittently (every 2 to 6 years). These attributes of lake sturgeon also mean that embryos have increased likelihood of being exposed to elevated concentrations of bioaccumulated DLCs via maternal transfer which might cause increased early life mortality.

The objective of the present study was to assess whether maternal transfer of legacy DLCs could be limiting lake sturgeon natural recruitment in the SLR-AOC. Assessments of ecological risk posed by complex environmental mixtures of PCB, PCDD, and PCDF congeners are performed using the TCDD-equivalency factor (TEF) approach (Van den Berg et al., 1998). TEFs are order of magnitude consensus values for the potency of a DLC congener relative to the prototypical reference congener, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD), which can be used in combination with tissue residue measurements to calculate TCDD-equivalents (TEQs) of a mixture of DLCs in terms of potency equivalents to 2,3,7,8-TCDD (Van den Berg et al. 1998). Fish TEFs have been published by the World Health Organization (WHO) based entirely on results of early life toxicity assays with salmonids (Van den Berg et al. 1998). However, the potency of DLCs relative to 2,3,7,8-TCDD can differ by more than an order of magnitude among species which introduces significant uncertainty when interpreting species-specific risk (Eisner et al., 2016). Development of TEFs specific for lake sturgeon using traditional early life toxicity assays is largely impractical because performing these assays for a suite of DLCs would be expensive, time consuming, and require large numbers of this difficult to acquire species; in addition to challenges related to unique life history characteristics which make sturgeon difficult to use in early life toxicity assays (Tompsett et al., 2014). However, 21$^{st}$ century advances in predictive ecotoxicology have resulted in the development of mechanism-based biological models, known as quantitative adverse outcome pathways (qAOPs), capable of accurately predicting species-specific relative potencies for DLCs using only cell-based assays that do not require lethal samples (Doering et al., 2018). Therefore, the specific objectives of the present study were to calculate TEQs in eggs and plasma collected from SLR-AOC lake sturgeon using WHO TEFs for fish and predicted lake sturgeon-specific relative potencies. The calculated TEQs could be compared to lake sturgeon mortality curves to determine whether maternal transfer would be expected to cause elevated early life mortality and potentially explain the observed recruitment failure.

## MATERIALS AND METHODS

### *Sample collection*

Collection of lake sturgeon for sampling was performed as described previously (Estep et al., 2020). Briefly, lake sturgeon were captured in the SLR estuary between the Fond du Lac dam and Highway 23 in the springs of 2017, 2018, and 2019 during the sturgeon spawning season when water temperatures reach 8 to 10 °C. Lake sturgeon were captured via a combination of boat electrofishing, dip-netting, backpack electrofishing, and angling. All captured fish were measured for total length, girth, and weight. A small pelvic fin clip was taken from each captured lake sturgeon for genetic stock identification as reported previously (Estep et al., 2020). The sex of each individual was determined visually using manual extrusion of gametes. From each female, a sample of the extruded oocytes were collected and transported on dry ice before being stored at -80 °C until analyzed. Approximately 6 mL of blood was collected from each individual through the caudal vein using a 21 G, 1" needle, attached to a 6 mL heparinized vacutainer. Blood was transported on wet ice before being centrifuged at 1750 g for 10 min at 10 °C to separate plasma. Plasma was stored at -80 °C until analyzed.

### *Exposure assessment*

Concentrations of the selected dioxin-like PCDDs, PCDFs, and PCBs were measured in eggs and plasma collected from SLR-AOC lake sturgeon by use of high-resolution isotopedilution mass spectrometry (HRMS) according to Methods 1668A and 1613B. Total lipid content (as % wet weight) was measured by use of the microgravimetric of Radin method. Lipid content was used for lipid normalization in order to directly compare concentrations of selected dioxin-like PCDDs, PCDFs, and PCBs between eggs and plasma, which differ significantly in lipid content. Lipid normalization was performed by dividing the chemical concentration of the sample by the total lipid content of the sample.

*Hazard assessment and risk characterization*

Measured concentrations of the selected dioxin-like PCDDs, PCDFs, and PCBs in eggs and plasma of SLR-AOC lake sturgeon were multiplied by WHO TEFs or lake sturgeon-specific TEFs predicted using the qAOP for each chemical in each sample (Table 1). The qAOP was developed and previously validated for application to lake sturgeon and is known to produce accurate predictions for this species (Doering et al., 2018). Predictions were based on lake sturgeon AHR function reported previously (Doering et al., 2015). In cases where the selected dioxin-like PCDD, PCDF, or PCB were not detected in the sample, each TEF was multiplied by the detection limit (as reported) to produce the most conservative estimate of possible exposure. Previously published concentrations of the selected dioxin-like PCDDs, PCDFs, and PCBs in eggs from lake sturgeon from a known healthy population (Tillitt et al., 2017) were also multiplied by WHO TEFs (Van den Berg et al., 1998) or lake sturgeon-specific TEFs predicted using the qAOP to act as a reference. The toxicity of each chemical in each sample were summed to produce TEQs in terms of potency of the mixture relative to 2,3,7,8-TCDD for each sample using both sets of TEFs. The TEQs in eggs were compared to the predicted toxicity curve of 2,3,7,8-TCDD for lake sturgeon predicted using the qAOP (Doering et al., 2018) and effect concentrations for 2,3,7,8-TCDD in limited toxicity assays performed with lake sturgeon (Tillitt et al., 2017). The qAOP uses not lipid normalized egg TEQs and therefore lipid normalized values were not used for these comparisons.

**RESULTS AND DISCUSSION**

The objective of the present study was to assess whether maternal transfer of legacy DLCs could be limiting lake sturgeon natural recruitment in the SLR-AOC. To assess maternal transfer of bioaccumulated DLCs to embryos, eggs were collected for chemical analysis from 1 female lake sturgeon from the SLR-AOC in 2017, 3 females in 2018, and 3 females in 2019. DLCs were detected in eggs collected from all 7 female lake sturgeon caught in the SLR-AOC (Table 2). The detected DLCs were predominantly PCDFs and PCBs, with 2,3,7,8-TCDD and 1,2,3,4,6,7,8-heptachlorodibenzo-*p*-dioxin (1,2,3,4,6,7,8-HCDD) being the only 2 detected PCDDs, and only in 1 of the egg samples (Table 2). The most frequently detected DLCs in SLR-AOC lake sturgeon eggs were 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TCDF), 1,2,3,6,7,8-hexachlorodibenzofuran (1,2,3,6,7,8-HCDF), 3,3',4,4'-tetrachlorobiphenyl (PCB 77), and 3,3',4,4',5-pentachlorobiphenyl (PCB 126) (Table 2). The DLC with the overall greatest measured concentration in SLR-AOC lake sturgeon eggs was PCB 126, however PCB 77 was the DLC with the greatest measured concentration in 4 of the 7 egg samples (Table 2).

To assess possible toxicities of exposure to maternally transferred DLCs, TEQs were calculated using both WHO TEFs for fish and lake sturgeon-specific TEFs predicted using the qAOP. TEQs in SLR-AOC lake sturgeon eggs ranged from 2.9 to 9.0 pg of 2,3,7,8-TCDD/g of egg using WHO TEFs and from 10.3 to 76.8 pg of 2,3,7,8-TCDD/g of egg using lake sturgeon-specific TEFs (Table 3). Most of the TEQs calculated for eggs from SLR-AOC lake sturgeon using either set of TEFs were greater than TEQs calculated for eggs from lake sturgeon from a healthy reference population (Figure 1). For most of the egg samples from SLR-AOC lake sturgeon, the greatest contributor to the TEQs using WHO TEFs was 1,2,3,6,7,8-HCDF, while using lake sturgeon-specific TEFs the greatest contributors to the TEQs were 1,2,3,6,7,8-HCDF, PCB 126, and 2,3,7,8-TCDF (Table 1; Table 2). The TEQs calculated using lake sturgeon-specific TEFs were on average 5-fold greater than TEQs calculated using WHO TEFs (Table 3). The greater TEQs using lake sturgeon-specific TEFs is driven heavily by increased potency of PCB 126 and 2,3,7,8-TCDF. These differences illustrate how using WHO TEFs as is standard in risks assessments for DLCs could lead to underestimation of risk in some cases and for certain species, as has been suggested previously (Doering et al., 2014; Eisner et al., 2016).

Early life toxicities of exposure to DLCs in fishes is well established and great differences in sensitivity are known to exist (Doering et al., 2013). The sensitivity of lake sturgeon has previously been assessed using predictive methods and in embryo toxicity assays (Doering et al., 2015; 2018; Tillitt et al., 2017). These studies have demonstrated lake sturgeon to be a moderately sensitive species with a 50% lethal dose (LD50) of 2,3,7,8-TCDD of 610 pg/g-egg and the no observed effect level (LOEL) is 360 pg/g-egg (Tillitt et al., 2017). Based on these studies, the lake sturgeon is approximately 10-fold less sensitive than the most sensitive known species of fish, the lake trout (*Salvelinus namaycush*) (Doering et al., 2013). In eggs from SLR-AOC lake sturgeon, TEQs calculated using either set of TEFs were all below levels predicted to cause any toxicities in early life stages based on predictive methods or embryo toxicity assays (Figure 1). Further, all TEQs were below known effect concentrations for sublethal endpoints in lake sturgeon based on results of embryo toxicity assays, including effects on long-term growth, developmental anomalies, swim performance, or behavior (Figure 1). Therefore, the TEQs measured in eggs from SLR-AOC lake sturgeon in context with results of existing studies and predictions from the qAOP suggest that neither lethal nor sublethal adverse effects would be expected to occur in early life stages as a result of maternally transferred dioxin-like PCDDs, PCDFs, or PCBs.

Eggs from only 7 female SLR-AOC lake sturgeon were able to be collected for chemical analysis between 2017 and 2019. Therefore, to get a broader scope of possible exposure to and bioaccumulation of dioxin-like PCDDs, PCDFs, and PCBs among SLR-AOC lake sturgeon, plasma samples from females, males, and immature individuals were also collected for analysis. Plasma was not collected from any fish in 2017. In 2018, plasma was collected and analyzed from 1 female, 2 male, and 3 immature SLR-AOC lake sturgeon (Table 4-6). In 2019, plasma was collected and analyzed from 5 female and 7 immature SLR-AOC lake sturgeon (Table 4-6). As was the case with the eggs, the detected DLCs were predominantly PCDFs and PCBs, with PCDDs being detected in only 1 female caught in 2019 (Table 4) and 1 immature individual caught in 2019 (Table 6). The most frequently detected DLCs in plasma from female SLR-AOC lake sturgeon were 2,3,7,8-TCDF and 1,2,3,4,6,7,8-HCDF, but the DLC with the greatest measured concentration was 1,2,3,6,7,8-HCDF (Table 4). Three DLCs were detected in plasma from male SLR-AOC lake sturgeon, specifically 2,3,7,8-TCDF, 1,2,3,6,7,8-HCDF, and PCB 126; with PCB 126 having the greatest measured concentration in both males (Table 5). In common with female and male SLR-AOC lake sturgeon, the most frequently detected DLCs in plasma from immature individuals were 2,3,7,8-TCDF, 1,2,3,4,6,7,8-HCDF, and PCB 126; with PCB 126 having the greatest measured concentration (Table 6).

In common with eggs, TEQs for plasma were calculated using both WHO TEFs for fish and lake sturgeon-specific TEFs predicted using the qAOP. However, to directly compare TEQs between eggs and plasma, TEQs were calculated following lipid normalization for plasma (Table 7). Lipid normalization was subsequently performed on concentrations measured in eggs and lipid normalized TEQs calculated (Table 7). Lipid normalized TEQs in SLR-AOC lake sturgeon eggs ranging from 0.4 to 1.4 pg of 2,3,7,8-TCDD/g of egg using WHO TEFs and from 1.3 to 8.4 pg of 2,3,7,8-TCDD/g of egg using lake sturgeon-specific TEFs (Table 7). However, lipid normalized TEQs in SLR-AOC lake sturgeon plasma were greater than in eggs, with a range from 2.6 to 10.5 pg of 2,3,7,8-TCDD/g of plasma using WHO TEFs and from 10.7 to 29.3 pg of 2,3,7,8-TCDD/g of plasma using lake sturgeon-specific TEFs (Table 7). Again, the TEQs calculated using lake sturgeon-specific TEFs were greater and driven heavily by the increased potency of PCB 126 and 2,3,7,8-TCDF.

Less is known about toxicities of exposure to DLCs in adult fish and nothing is known about toxicities in adult lake sturgeon. However, long-term exposure studies to environmentally relevant concentrations of DLCs in rainbow trout (*Oncorhynchus mykiss*) have shown the potential for toxicities in adult fishes, including decreased survival, altered behavior, and impaired reproduction (Giesy et al., 2002). These toxicities in adults were shown to begin to occur at body burdens that result in maternal transfer of doses that cause toxicities in early life stages (Giesy et al., 2002). Using this knowledge, a plasma TEQ (lipid normalized) that is predicted to represent a threshold for sublethal or lethal effects in adult lake sturgeon was predicted by converting the embryo TEQ for threshold for effects to the plasma TEQ. This was done by dividing the embryo effect threshold of 360 pg TCDD/g-egg (Tillet et al., 2017) by the average egg lipid content (8.3%) (Table 2) allowing extrapolation across tissues. This results in a predicted lipid normalized TEQ effect threshold for sublethal or lethal effects on adults of 43 pg TCDD/g of tissue. The TEQs calculated in plasma from SLR-AOC lake sturgeon ranged from 2.6 to 10.5 pg of 2,3,7,8-TCDD/g of plasma using WHO TEFs and from 10.7 to 28.5 pg of 2,3,7,8-TCDD/g of plasma using lake sturgeon-specific TEFs (Table 7). All these TEQs measured in plasma from adult SLR-AOC lake sturgeon are below the predicted effect threshold of 43 pg TCDD/g of plasma which suggests that neither lethal nor sublethal adverse effects would be expected to occur in female, male, or immature adult individuals as a result of bioaccumulated dioxin-like PCDDs, PCDFs, or PCBs.

In conclusion, the present study demonstrates that SLR-AOC lake sturgeon are bioaccumulating dioxin-like PCDDs, PCDFs, and PCBs to levels greater than at a reference site, but concentrations being maternally transferred to embryos are well below concentrations known to cause any lethal or sublethal toxicities in this species. Additionally, TEQs measured in plasma from adult SLR-AOC lake sturgeon also suggest that neither lethal nor sublethal adverse effects would be expected to occur in adult female, male, or immature individuals. Therefore, these results support that bioaccumulated or maternally transferred DLCs are unlikely to be a contributing factor to the recruitment failure of lake sturgeon in the SLR-AOC. As a result, other limiting factors that could cause the observed recruitment failure should be assessed, such as potential toxicity from other contaminants of concern, competition or predation from invasive species, habitat suitability, or the extremely long generation time of this species.

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**Table 1.** Comparison of WHO TEFs for fish and TEFs specific for lake sturgeon.

| DLC Congener | WHO Fish TEFs[a] | Lake Sturgeon TEFs | Fold-difference |
|---|---|---|---|
| PCDDs | | | |
| 2,3,7,8-TCDD | 1.0 | 1.0 | 1 |
| 1,2,3,7,8-PCDD | 1.0 | 1.0 | 1 |
| 1,2,3,4,7,8-HCDD | 0.5 | 0.5 | 1 |
| 1,2,3,6,7,8-HCDD | 0.01 | 0.01 | 1 |
| 1,2,3,7,8,9-HCDD | 0.01 | 0.01 | 1 |
| 1,2,3,4,6,7,8-HCDD | 0.01 | 0.01 | 1 |
| PCDFs | | | |
| 2,3,7,8-TCDF | 0.05 | 1 | 20 |
| 1,2,3,7,8-PCDF | 0.05 | 0.5 | 10 |
| 2,3,4,7,8-PCDF | 0.5 | 2.5 | 5 |
| 1,2,3,4,7,8-HCDF | 0.1 | 1 | 10 |
| 1,2,3,6,7,8-HCDF | 0.1 | 1 | 10 |
| 1,2,3,7,8,9-HCDF | 0.1 | 1 | 10 |
| 2,3,4,6,7,8-HCDF | 0.1 | 1 | 10 |
| 1,2,3,4,6,7,8-HCDF | 0.01 | 0.1 | 10 |
| 1,2,3,4,7,8,9-HCDF | 0.01 | 0.1 | 10 |
| Non-*ortho* PCBs | | | |
| PCB 77 | 0.0001 | 0.05 | 500 |
| PCB 81 | 0.0005 | 0.1 | 200 |
| PCB 126 | 0.005 | 0.1 | 20 |
| PCB 169 | 0.00005 | 0.01 | 200 |

[a] TEFs as shown previously (Van den Berg et al., 1998).

**Table 2.** DLCs measured in eggs (pg DLC/g egg) collected from SLR-AOC lake sturgeon in 2017, 2018, and 2019.[a]

| Analyte | 2017 | 2018 | | | 2019 | | |
|---|---|---|---|---|---|---|---|
| | Female 1 | Female 1 | Female 2 | Female 3 | Female 1 | Female 2 | Female 3 |
| Lipid (%) | 4 | 9 | 11 | 10 | 8 | 8 | 8 |
| PCDDs | | | | | | | |
| 2,3,7,8-TCDD | | | | 0.4 | | | |
| 1,2,3,7,8-PCDD | | | | | | | |
| 1,2,3,4,7,8-HCDD | | | | | | | |
| 1,2,3,6,7,8-HCDD | | | | | | | |
| 1,2,3,7,8,9-HCDD | | | | | | | |
| 1,2,3,4,6,7,8-HCDD | | | | 0.6 | | | |
| PCDFs | | | | | | | |
| 2,3,7,8-TCDF | | 2.7 | | 8.1 | | | 3.9 |
| 1,2,3,7,8-PCDF | | 2.2 | | | 0.7 | | |
| 2,3,4,7,8-PCDF | | | | | | | |
| 1,2,3,4,7,8-HCDF | | 2.5 | | | | | |
| 1,2,3,6,7,8-HCDF | 18.0 | 21.0 | | 58.0 | | | |
| 1,2,3,7,8,9-HCDF | | | | | | | |
| 2,3,4,6,7,8-HCDF | | | | | | | |
| 1,2,3,4,6,7,8-HCDF | 1.7 | | | | | | |
| 1,2,3,4,7,8,9-HCDF | | | | | | | |
| Non-*ortho* PCBs | | | | | | | |
| PCB 77 | 58.0 | 41.0 | | | 54.8 | 10.1 | 10.2 |
| PCB 81 | | | | | | | |
| PCB 126 | 24.0 | 91.0 | 120.0 | | | | |
| PCB 169 | | | | | | 6.2 | 8.8 |

[a] DLCs that were not detected in eggs are left blank. Detection limit was 1 pg/g-egg for PCDD/PCDFs and 5 pg/g-egg for PCBs.

**Table 3.** TEQs of SLR-AOC lake sturgeon eggs calculated using WHO TEFs or lake sturgeon-specific TEFs show as pg of 2,3,7,8-TCDD/g of egg.

| Year | Individual | WHO TEQ | Lake Sturgeon-specific TEQ |
|---|---|---|---|
| 2017 | Female 1 | 5.4 | 33.7 |
| 2018 | Female 1 | 6.3 | 46.2 |
| | Female 2 | 4.2 | 23.5 |
| | Female 3 | 9.0 | 76.8 |
| 2019 | Female 1 | 2.9 | 10.6 |
| | Female 2 | 3.2 | 10.3 |
| | Female 3 | 3.9 | 12.8 |

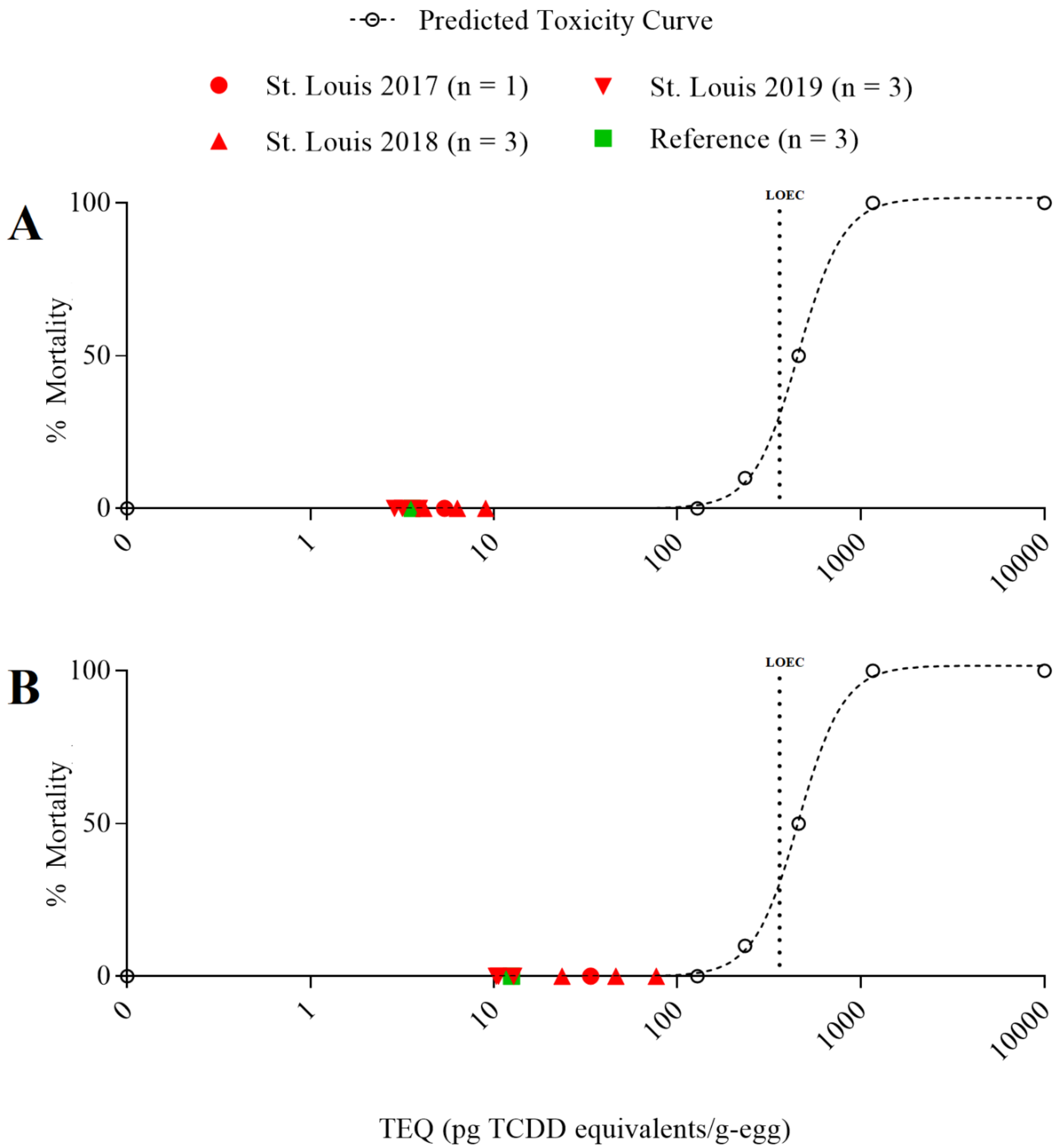


**Figure 1.** TEQs of SLR-AOC lake sturgeon eggs calculated using WHO TEFs (A) or lake sturgeon-specific TEFs (B) relative to predicted mortality response curve (open circles connected by curved dotted line). Lowest observed effect concentration (LOEC) in lake sturgeon following laboratory toxicity testing is shown (vertical dotted line) (Tillitt et al., 2017). TEQs calculated for lake sturgeon eggs from a healthy population is shown as a reference (Tillitt et al., 2017).

**Table 4.** DLCs measured in plasma (pg DLC/g plasma) collected from female SLR-AOC lake sturgeon in 2018 and 2019.[a]

| Analyte | 2018 | 2019 | | | | |
|---|---|---|---|---|---|---|
| | Female 1 | Female 1 | Female 2 | Female 3 | Female 4[b] | Female 5[b] |
| Lipid (%) | 1.2 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 |
| PCDDs | | | | | | |
| 2,3,7,8-TCDD | | | | | | 2.6 |
| 1,2,3,7,8-PCDD | | | | | | 4.6 |
| 1,2,3,4,7,8-HCDD | | | | | | |
| 1,2,3,6,7,8-HCDD | | | | | | |
| 1,2,3,7,8,9-HCDD | | | | | | |
| 1,2,3,4,6,7,8-HCDD | | | | | | |
| PCDFs | | | | | | |
| 2,3,7,8-TCDF | 1.3 | 3.3 | | | | |
| 1,2,3,7,8-PCDF | | | | | | |
| 2,3,4,7,8-PCDF | | | | | | 2.5 |
| 1,2,3,4,7,8-HCDF | | | | | | 2.5 |
| 1,2,3,6,7,8-HCDF | 9.0 | | | | | 2.2 |
| 1,2,3,7,8,9-HCDF | | | | | | |
| 2,3,4,6,7,8-HCDF | | | | | | 2.8 |
| 1,2,3,4,6,7,8-HCDF | 0.6 | | 3.4 | | | 4.1 |
| 1,2,3,4,7,8,9-HCDF | | | | | | |
| Non-*ortho* PCBs | | | | | | |
| PCB 77 | | | | | | |
| PCB 81 | | | | | | |
| PCB 126 | | | | | | |
| PCB 169 | | | | | | |

[a] DLCs that were not detected in plasma are left blank. Detection limit was 1 pg/g-egg for PCDD/PCDFs and 5 pg/g-egg for PCBs.

[b] Plasma sample, but no egg sample, collected from these individuals.

**Table 5.** DLCs measured in plasma (pg DLC/g plasma) collected from male SLR-AOC lake sturgeon in 2018.[a]

| Analyte | 2018 | |
|---|---|---|
| | Male 1 | Male 2 |
| Lipid (%) | 1.5 | 1.1 |
| PCDDs | | |
| 2,3,7,8-TCDD | | |
| 1,2,3,7,8-PCDD | | |
| 1,2,3,4,7,8-HCDD | | |
| 1,2,3,6,7,8-HCDD | | |
| 1,2,3,7,8,9-HCDD | | |
| 1,2,3,4,6,7,8-HCDD | | |
| PCDFs | | |
| 2,3,7,8-TCDF | 0.5 | 0.6 |
| 1,2,3,7,8-PCDF | | |
| 2,3,4,7,8-PCDF | | |
| 1,2,3,4,7,8-HCDF | | |
| 1,2,3,6,7,8-HCDF | 4.3 | 5.3 |
| 1,2,3,7,8,9-HCDF | | |
| 2,3,4,6,7,8-HCDF | | |
| 1,2,3,4,6,7,8-HCDF | | |
| 1,2,3,4,7,8,9-HCDF | | |
| Non-*ortho* PCBs | | |
| PCB 77 | | |
| PCB 81 | | |
| PCB 126 | 18 | 14 |
| PCB 169 | | |

[a] DLCs that were not detected in plasma are left blank. Detection limit was 1 pg/g-egg for PCDD/PCDFs and 5 pg/g-egg for PCBs.

**Table 6.** DLCs measured in plasma (pg DLC/g plasma) collected from SLR-AOC lake sturgeon in 2018 and 2019 that could not be sexed.[a]

| Analyte | 2018 | | | 2019 | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Fish 1 | Fish 2 | Fish 3 | Fish 1 | Fish 2 | Fish 3 | Fish 4 | Fish 5 | Fish 6 | Fish 7 |
| Lipid (%) | 0.7 | 0.5 | 0.6 | 0.6 | 0.6 | 0.6 | 0.6 | 0.6 | 0.6 | 0.6 |
| PCDDs | | | | | | | | | | |
| 2,3,7,8-TCDD | | | | | | | | | | |
| 1,2,3,7,8-PCDD | | | | | | | | | | |
| 1,2,3,4,7,8-HCDD | | | | | | | | | | |
| 1,2,3,6,7,8-HCDD | | | | | | | | | | |
| 1,2,3,7,8,9-HCDD | | | | | | | | | | |
| 1,2,3,4,6,7,8-HCDD | | | | 1.6 | | | | | | |
| PCDFs | | | | | | | | | | |
| 2,3,7,8-TCDF | 0.7 | | | 2.3 | 3 | | 3.7 | 3.1 | 3.2 | 2.9 |
| 1,2,3,7,8-PCDF | | | | | | | | | | |
| 2,3,4,7,8-PCDF | | | | | | | | | | |
| 1,2,3,4,7,8-HCDF | | | | | | | | | | |
| 1,2,3,6,7,8-HCDF | 3.0 | 2.3 | 2.8 | | 0.7 | 0.9 | | | | 0.9 |
| 1,2,3,7,8,9-HCDF | | | | | | | | | | |
| 2,3,4,6,7,8-HCDF | | | | | | | | | | |
| 1,2,3,4,6,7,8-HCDF | | | | | | | | | | |
| 1,2,3,4,7,8,9-HCDF | | | | | | | | | | |
| Non-*ortho* PCBs | | | | | | | | | | |
| PCB 77 | 6.8 | | | | | | | | | |
| PCB 81 | | | | | | | | | | |
| PCB 126 | 11 | 24 | 14 | | | | | | | |
| PCB 169 | | | | | | | | | | |

[a] DLCs that were not detected in plasma are left blank. Detection limit was 1 pg/g-egg for PCDD/PCDFs and 5 pg/g-egg for PCBs.

**Table 7.** TEQs of SLR-AOC lake sturgeon plasma calculated using WHO TEFs or lake sturgeon-specific TEFs show as lipid normalized pg of 2,3,7,8-TCDD/g of plasma or egg.

| Year | Individual | WHO TEQ | | Lake Sturgeon-specific TEQ | |
|---|---|---|---|---|---|
| | | Egg | Plasma | Egg | Plasma |
| 2017 | Female 1 | 1.4 | | 8.4 | |
| 2018 | Female 1 | 0.7 | 3.7 | 5.1 | 16.9 |
| | Female 2 | 0.4 | 0.9 | 2.1 | 7.7 |
| | Female 3 | 0.9 | | 7.7 | |
| | Male 1 | | 2.6 | | 10.7 |
| | Male 2 | | 3.7 | | 15.3 |
| | Fish 1 | | 5.5 | | 20.9 |
| | Fish 2 | | 7.3 | | 29.3 |
| | Fish 3 | | 5.9 | | 22.7 |
| 2019 | Female 1 | 0.4 | 4.7 | 1.3 | 13.9 |
| | Female 2 | 0.4 | 1.3 | 10.5 | 28.5 |
| | Female 3 | 0.5 | 4.6 | 1.6 | 15.0 |
| | Female 4 | | 3.7 | | 11.5 |
| | Female 5 | | 6.8 | | 22.8 |
| | Fish 1 | | 5.9 | | 20.5 |
| | Fish 2 | | 5.3 | | 21.4 |
| | Fish 3 | | 6.8 | | 22.5 |
| | Fish 4 | | 7.3 | | 26.3 |
| | Fish 5 | | 7.4 | | 26.7 |
| | Fish 6 | | 7.9 | | 24.2 |
| | Fish 7 | | 6.7 | | 24.9 |

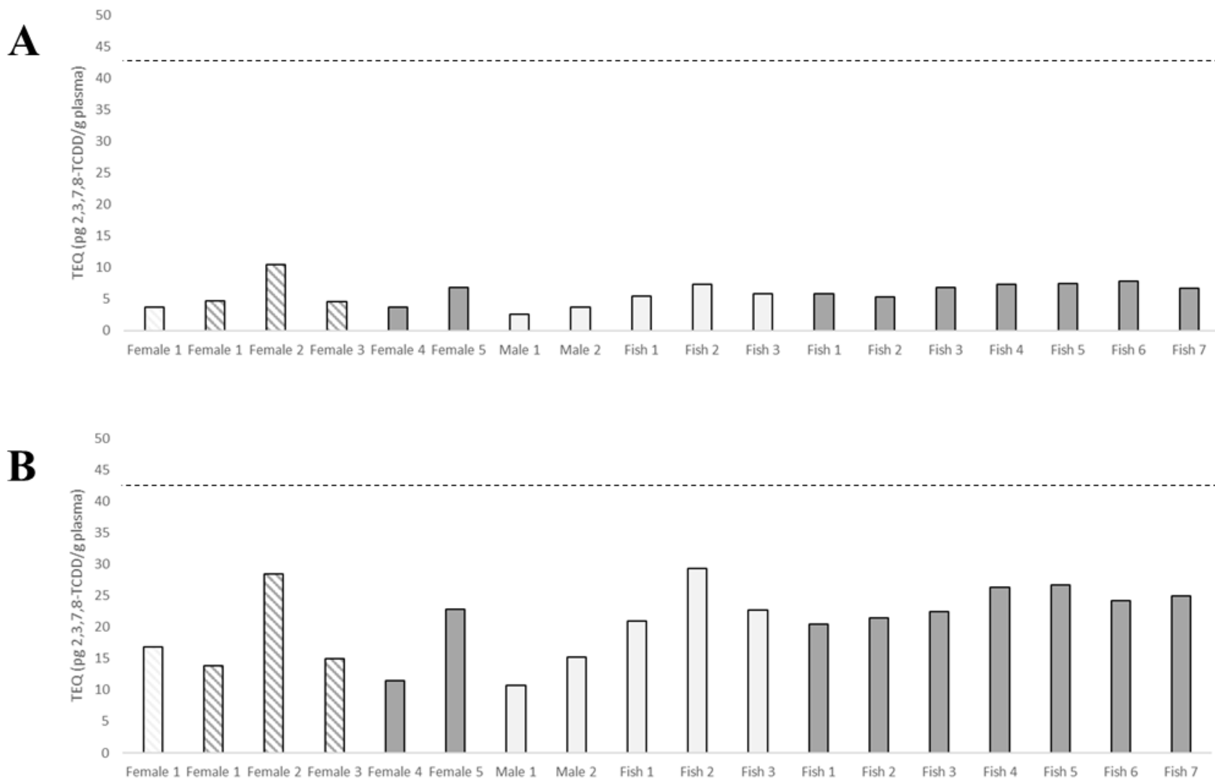


**Figure 2.** Lipid normalized TEQs of SLR-AOC lake sturgeon plasma calculated using WHO TEFs (A) or lake sturgeon-specific TEFs (B) from females, males, and individuals that could not be sexed that were caught in 2018 (light bars) and 2019 (dark bars). Females with both plasma and egg TEQs are highlighted (hashed bars). Labels for each individual fish as presented previously (Table 7). The predicted effect threshold of 43 pg TCDD/g of plasma is indicated (dotted horizontal line).

# Appendix E

# Gutsch (2017) Ruffe Study Dissertation

# The rise and fall of the Ruffe (*Gymnocephalus cernua*) empire in Lake Superior

A DISSERTATION

SUBMITTED TO THE FACULTY OF

UNIVERSITY OF MINNESOTA

BY


Michelle Kathleen Gutsch


IN PARTIAL FULFILLMENT OF THE REQUIREMENTS

OF THE DEGREE OF

DOCTOR OF PHILOSOPHY

Dr. Joel C. Hoffman, Advisor

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# Dedication

I dedicate this dissertation to my husband and Baby Maggie. Thank you for your love and support throughout this endeavor.

## Abstract

Invasive species are a global problem, impacting property, habitats, ecosystem function, and native species. Our ability to predict future habitat and spread of aquatic invasive species is limited because it is challenging to collect and integrate information regarding life history, movement, and habitat, especially across continents. Invasive Ruffe (*Gymnocephalus cernua*) has caused substantial ecological damage in North America, parts of Western Europe, Scandinavian countries, and the United Kingdom. Given the potential for ecological impacts, such as native fish declines, ongoing concern regarding the spread of Ruffe is warranted. But there are significant research gaps regarding life history, movement, and Ruffe distribution in the native and non-native range. Therefore, the overall goals of my dissertation were to acquire life stage-specific data for Ruffe, including dispersal, seasonal, and spawning movements, and characterize their life cycle, and to develop a lake-scale species distribution model for Ruffe at a 30-m resolution. First, I found that Ruffe exhibits plasticity with regard to chemical, physical, biological, and habitat requirements (Chapter One). Adult Ruffe has characteristics that allow it to adapt to a range of environments, including rapid maturation, relatively long life and large size, batch spawning, genotypic and phenotypic plasticity, tolerance to a wide range of environmental conditions, broad diet, and multiple dispersal periods. Notably, there is variability among these characteristics between the native, non-native North American, and European non-native populations. Second, I found that Ruffe populations in both the St. Louis River and Chequamegon Bay are at different invasion stages (Chapter Two). In the St. Louis River, the population

increased from the initial invasion in 1986 up to 1995 and has been in decline for the past two decades (1996-2015). In Chequamegon Bay, the overall population is increasing, but is doing so by oscillating every 5-7 years. I concluded that Ruffe populations in both systems partially conform to the typical “boom-bust” patterns seen with other invasive fish species. Third, carbon and nitrogen stable isotope ratios ($\delta^{13}C$, $\delta^{15}N$) revealed size-specific movements between coastal wetland and Lake Superior. I found significant differences in $\delta^{13}C$ and $\delta^{15}N$ values between Ruffe captured in Lake Superior and those captured in the St. Louis River, but not among locations within the river (Chapter Three). I found size-based differences as well; medium-sized fish, 60-80 mm standard length (SL), had a $\delta^{13}C_{\text{lipid corrected}}$ of about -25‰ to -45‰, lower than either small (<60 mm SL) or large (80-148 mm SL) Ruffe (-38.2‰ to -14.2‰). Importantly, extremely $^{13}C$-depleted fish (<-36‰ $\delta^{13}C$) indicate that some Ruffe captured within coastal wetlands were feeding in a methane-based trophic pathway. Finally, a variety of species distribution models constructed to predict Ruffe suitable habitat in Lake Superior based on environmental data resolved to a variety of scales all performed similarly but varied substantially in the area of habitat predicted (Chapter Four). Among the six distribution models (250-m, 500-m, 1000-m, 2000-m, and 2000-m selected model) constructed using catch and environmental data from various spatial resolutions, the best performing model used 500 m data and the worst performing model used 2000 m data. The important geographic discrepancies in potential habitat occurred around the Apostle Islands, WI, Isle Royale, MN, Grand Marais, MI, Whitefish Point, MI, and

Red Rock and Nipigon in Canada. Multiple models performed similarly according to the area under the curve (AUC) scores, but had different results with respect to the area and distribution of suitable habitat predicted. I further examined whether there were differences among species distribution models developed from cumulative time-series (cumulative decades) or discrete time stanzas (decades treated separately). The separate time-series models all performed similarly well, but the performance of the cumulative models declined as data were added to subsequent models. Despite relatively strong performance, the species distribution models indicated offshore habitat and exposed, rocky nearshore habitat were suitable habitat, which is not corroborated by my research on the habitat preference and movement ecology of Ruffe (Chapter 1, 2, 3). I conclude that, to interpret the outputs of the Ruffe species distribution models, both model performance and the ecology of Ruffe must be considered to better characterize its fundamental niche. Broadly, I demonstrate the importance of synthesizing the life stage-specific biology and distribution of an invasive species with species distribution models to advance our ability to predict the future habitat of an invasive species.

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# Dissertation Introduction

Invasive species are a global problem, causing destruction of property, habitats, and threatening native species. Biological invasions can impact agriculture, forestry and health, all of which affect human economic wealth (Pimentel *et al.* 2001); invasions can further alter ecosystem function (Brooks *et al.* 2004) and threaten native biodiversity (Mack *et al.* 2000). In recent decades, the spread of species from their native ranges has increased dramatically, both in frequency and extent, due to the increase in global and international trade, as well as an increase in human movements (McNeely 2001; Thuiller *et al.* 2005). Once an introduced species has become established in a novel environment, it is nearly impossible to eradicate (Sindel and Michael 1992; Hastings 1996; Perrings *et al.* 2002; Peterson 2003). Preventing the introduction of potential invaders is the best, most cost-effective management strategy; however, when prevention is not possible, early detection tools can be used to help monitor new introductions and spread (Hoffman *et al.* 2016). One such tool is an ecological niche model (Peterson and Vieglais 2001).

The Laurentian Great Lakes have been severely impacted by aquatic invasive species (AIS) in the past two centuries (USEPA 2011). Owing to the severity of the invasion, a Great Lakes-wide aquatic invasive species (AIS) early detection and rapid response network is required under the Great Lakes Water Quality Agreement (GLWQA 2012). The goal of an early detection and rapid response network is to detect an invasive species at an early stage in its introduction when it is rare and geographically isolated (Hulme 2006). The success of eradication efforts, quarantines, and public education is increased during this early invasion stage before the invasive species becomes established, and these actions become much more costly (Gherardi and Angiolini 2004). To establish an effective network, locations of high risk for introduction of AIS need to be identified (Vander Zanden *et al.* 2010), and high-efficiency methods, including detection techniques that are more sensitive than traditional population monitoring need to be put in place (Trebitz *et al.* 2009; Vander Zanden *et al.* 2010; Hoffman *et al.* 2011).

Identifying locations of high risk for invasive species requires some understanding of vectors for spread, relative propagule pressure, and the suitability of the chemical, physical, and biological conditions (Colautti and MacIsaac 2004). Niche modeling is one way that has been shown to predict whether or not introduced species will be able to establish and spread throughout the landscape (Peterson 2003). Niche models are cost effective because they often use already existing data to model species' potential distributions, so there is no need for costly field efforts (Fielding and Bell 1997). However, these models have limitations based on how they are constructed. Typically, ecological niche models use global climate data as their ecological component and data from the native range of the organism. Often the prediction maps are at such a large scale that managers only have a vague idea (e.g., all of the Great Lakes) of where an invasive species might be able to establish a population. A model using data from the non-native range and environmental data that is at a resolution closer to the scale at which the animal lives may provide model outputs with finer geographic resolution to predict suitable habitat.

The overall goals of my dissertation were to acquire life stage-specific data for Ruffe, including dispersal, diet, seasonal, and spawning movements and characterize their life cycle and to develop a lake-scale species distribution model for Ruffe at a 30-m resolution. Ruffe is an invasive species that has caused ecological and economic damage in places it has invaded around the world (Maitland and East 1989; Adams and Tippett 1991; Selgeby and Edwards 1993; Adams 1994; Kalas 1995; Ogle *et al.* 1996; Selgeby 1998; Lorenzoni *et al.* 2009). By learning about its complete life history in the Laurentian Great Lakes and creating a lake-scale model of its suitable habitat, I have provided better information for targeted monitoring of Ruffe; further, these methods and this model can be used for other invasive species in Lake Superior.

I had three goals for Chapter One. First, I identified Ruffe's native and non-native range; second, I examined the chemical, physical, biological, and habitat requirements of Ruffe; and third, I characterized Ruffe's life cycle. For Chapter Two, my goal was to determine whether Ruffe populations in the St. Louis River and Chequamegon Bay conform to typical invasive species boom-bust patterns; moreover, as an exploratory analysis, I compared Ruffe abundance to potential predator and competitor abundance through time to identify species that might have strong interactions with Ruffe in the St. Louis River and Chequamegon Bay. For Chapter Three, I used carbon and nitrogen stable isotope ratios to identify trophic pathways supporting Ruffe in the St. Louis River, Chequamegon Bay, and Lake Superior. I measured carbon and nitrogen stable isotope ratios of Ruffe, used a stable isotope mixing model to estimate diet contributions from both Lake Superior (benthic periphyton) and wetland sources (including methane-oxidizing bacteria), and then characterized size-based movement between the wetland and Lake Superior based on the output of the mixing model. Finally, for Chapter Four, my goal was to apply lake-scale catch data and environmental variables to develop a Ruffe species distribution model (Maxent model) for Lake Superior. I evaluated the effects of resolving the data at a variety of spatial and temporal scales on the model output (i.e., the area within Lake Superior that is classified as suitable habitat). For the spatial analysis, I compared the model output among six different occurrence point distance buffers, including all points, 250-m, 500-m, 1000-m, 2000-m, and a 2000-m selected point removal procedure. In addition, I ran a cumulative and a separate time-series analysis on data from 1986, 1996, 2006, and 2014. To compare the model outputs, I determined the percent of suitable habitat for the lake for all models, as well as three zones—offshore, nearshore, and in-shore.

Ruffe can adapt to almost any aquatic environment (lakes, rivers, ponds, bays, brackish waters, tidal estuaries, non-tidal estuaries, and reservoirs (Hölker and Thiel 1998)). That adaptability is what makes it an effective invasive species (Adams and Tippett 1991; Ruffe Task Force 1992; Ogle *et al.* 1995, 1996; Mayo *et al.* 1998). Even though it is not a highly migratory fish, Ruffe has spread and established populations across continents (Matthey 1966; Maitland and East 1989; Adams 1991; Winfield 1992; Kalas 1995; Stepien *et al.* 1998; Eckmann 2004; Winfield *et al.* 2010, 2011, 2004; Lorenzoni *et al.* 2009; Volta *et al.* 2013). Also, Ruffe is highly competitive in low-light conditions and has the potential to alter population dynamics of prey (benthic invertebrates and zooplankton prey), competitors (forage fish), and fish predators (including through egg-consumption; (Mikkola *et al.* 1979; Sterligova and Pavlovskiy 1984; Pavlovskiy and Sterligova 1986; Adams and Tippett 1991; Kangur and Kangur 1996; Selgeby 1998; Kangur *et al.* 2000)). Notably, management actions to prevent the spread of Ruffe are critical because Ruffe matures rapidly and has high fecundity, and thus can quickly establish a population (Fedorova and Vetkasov 1974; Collette *et al.* 1977; Kolomin 1977; Lind 1977; Craig 1987; Neja 1988; Jamet and Lair 1991; Kovac 1998; Lappalainen and Kjellman 1998; Lorenzoni et al. 2009). In this dissertation, I present a detailed description of Ruffe life history and native and non-native range; a current and past description of its population dynamics and how that fits into invasion theory; detailed descriptions about its movements and trophic pathways based on stable isotope ratios; and a series of prediction maps showing suitable habitat of Ruffe for Lake Superior using 30-m-scale environmental variables.

# Chapter 1: A review of Ruffe (*Gymnocephalus cernua*) life history in its native versus non-native range

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## Abstract

Invasive Ruffe (*Gymnocephalus cernua*) has caused substantial ecological damage in North America, parts of Western Europe, Scandinavian countries, and the United Kingdom. The objectives of this review are to define Ruffe's native and non-native range, examine life history requirements, explore the life cycle, and differentiate between life stages. I compare data from its native and non-native ranges to determine if there are any differences in habitat, size, age, genotype, or seasonal migration. Literature from both the native and non-native ranges of Ruffe, with some rare, translated literature, is used. In each life stage, Ruffe exhibit plasticity with regard to chemical, physical, biological, and habitat requirements. Adult Ruffe has characteristics that allow them to adapt to a range of environments, including rapid maturation, relatively long life and large size (allowing them to reproduce many times in large batches), batch spawning, genotype and phenotype (having plasticity in their genetic expression), tolerance to a wide range of water quality, broad diet, and multiple dispersal periods. There is, however, variability among these characteristics between the native, non-native North American, and European non-native populations, which presents a challenge to managing populations based on life history characteristics. Monitoring and preventative strategies are important because, based on Ruffe's variable life history strategies and its recent range expansion, all of the Laurentian Great Lakes and many other water bodies in the U.K., Europe, and Norway are vulnerable to Ruffe establishment.

## Introduction

Although Ruffe (*Gymnocephalus cernua*), a small freshwater fish, is an invasive species in Europe and North America, less than thirty years ago there was a commercial fishery for it along the coastal regions of the Baltic Sea. The Ruffe fishery dated back to 1886 in the Elbe River estuary, Germany. Historically, Ruffe fisheries were found in Denmark, Scandinavian countries, Holland, and the former USSR, including Estonia (Johnsen 1965; Hölker and Thiel 1998), harvesting up to 1759 tons per year (Johnsen 1965). Although once popular as a food fish, Ruffe is no longer commercially harvested. Rather, it has since been widely introduced outside of their native range, to water bodies in North America, the United Kingdom, Western Europe (defined for the purposes of this paper as Italy, Germany, France, Belgium, the Netherlands, Austria, Spain, Portugal, and Denmark), and Norway.

Once established, invasive Ruffe disrupts interactions among native organisms. It competes with native fishes for food resources due to niche overlap (Maitland and East 1989; Ruffe Task Force 1992; McLean 1993; Ogle *et al.* 1995). It also consumes fish eggs, especially those of *Coregonus* spp. (Mikkola *et al.* 1979; Sterligova and Pavlovskiy 1984; Pavlovskiy and Sterligova 1986; Adams and Tippett 1991; Kangur and Kangur 1996; Selgeby 1998; Kangur *et al.* 2000), and preys on young-of-the-year fish or small fishes (Kozlova and Panasenko 1977; Holker and Hammer 1994; Kangur and Kangur 1996). In the water bodies it has successfully invaded (i.e., established a reproducing population), Ruffe has outcompeted native fishes and evaded native piscivores (Ogle *et al.* 1995, 1996; Mayo *et al.* 1998).

In the 1980s, Ruffe was accidentally introduced to the Laurentian Great Lakes in North America via ballast water, and parts of Western Europe, Scandinavian countries, and the United Kingdom via canals, shipping, and bait bucket transfers. There are concerns about the adaptability of this fish to introduced water bodies due to their rapid and steady range expansion. To better understand its potential for further range expansion, it is important to characterize the chemical, physical, biological, and habitat requirements of Ruffe, as well as its interactions with other organisms. Substantial knowledge gaps remain regarding its habitat use and ecology. First, there is a lack of a complete description of Ruffe's native range, particularly in Asia, which is necessary to determine the extent of their native habitat. Second, seasonal movements and dispersal need to be characterized to fully describe the ecological niche of Ruffe.

The goals of this review are to (1) define Ruffe's native and non-native range; (2) examine the chemical, physical, biological, and habitat requirements of Ruffe; and (3) characterize Ruffe's life cycle. For this literature review, I conducted an exhaustive search of published literature and available reports from both the native and non-native range of Ruffe. Throughout, I examine differences with respect to habitat, size, age, genotype, or seasonal migration between populations from the native and non-native ranges.

## Methods

To conduct the review, I searched for published literature using Google Scholar with key phrases, including "Ruffe habitat," "Ruffe life cycle," "Ruffe diets," and "Ruffe ecology." Historical literature, including unpublished reports, was identified using sources cited in primary literature and review articles. Most literature from the non-native range was published between 1980 and 2000, a period of rapid spread. Literature from the native range was from Russia, Denmark, Western Europe, Norway, and the former USSR, including Estonia, and was published between 1940 and 2000.

To describe the life cycle, I used four discrete stages—egg (embryonic), larva, juvenile, and adult. Ontogeny specific to Ruffe was based on Kovac (1994).

To describe the native range, location data came from any paper that mentioned Ruffe was present, even if Ruffe was not the topic of the paper (i.e., papers about parasites in Ruffe were common, as were papers examining the mechanisms of sensory organs in fish). The range map for their native distribution was based on literature descriptions; I associated Ruffe with the water bodies (i.e., rivers, lakes, and seas) surrounding the 229 native occurrence points, and below an elevation of 964 m above sea level (the highest elevation Ruffe are known to occur). For the range map, native and non-native occurrences were differentiated based on literature descriptions. England included both native and non-native occurrences; however, I was unable to find any occurrence coordinates for southern England although Ruffe is native to this region (Collette and Banarescu 1977; Kalas 1995; Winfield et al. 1998). Ruffe occurrences for southern England were interpreted from a UK map from the National Biodiversity Network (NBN Gateway 2013). For the marine coastal habitat, I applied a 15 km buffer from the shoreline because this is the furthest distance away from shore that Ruffe has been documented (Selgeby 1998).

Non-native populations of Ruffe usually were emphasized in specific articles, allowing us to identify native and non-native populations. For the non-native North American occurrence map, data (N=5,898 sampling events over a 29-year period) in the Laurentian Great Lakes was mostly provided by USGS, USFWS, and USEPA, including published and unpublished data.

## FINDINGS

I discuss review findings, including the native and non-native ranges, life history requirements, Ruffe life cycle, and details of adult Ruffe.

### *NATIVE RANGE*

Ruffe is native to a large part of Europe and Asia, ranging from the northeast of France (Berg 1965; Rösch *et al.* 1996) and southern England (Collette and Banarescu 1977; Kalas 1995; Winfield et al. 1998) to parts of Siberia and Russia (Berg 1949; McLean 1993; Mills *et al.* 1994; Gunderson *et al.* 1998; Mayo *et al.* 1998; Ogle 1998, 2009; Selgeby 1998; Ogle *et al.* 2004; Dawson *et al.* 2006) (Figure 1). Its range extends almost to the coast of the Arctic seas in eastern Scandinavia, including rivers entering the Baltic and White Seas at the northernmost part of its range (Holcik and Hensel 1974; Collette and Banarescu 1977; Kalas 1995; Popova *et al.* 1998; Brown *et al.* 1998; Lorenzoni *et al.* 2009). Ruffe exist throughout all of Siberia; it is present in the Kolyma River, but not in the Amur River (Holcik and Hensel 1974; Collette and Banarescu 1977; Kalas 1995; Brown *et al.* 1998; Lorenzoni *et al.* 2009). The Ob' and Nadym River in Russia comprise Ruffe's eastern border (Petlina 1967; Kolomin 1977; Matkovskiy 1987; Popova *et al.* 1998; Stepien *et al.* 1998). In

Slovakia, Ruffe is found throughout the Danube River, including the Little Danube and its side channels and tributaries in the lower parts of the river and on the Large Danube Island (Hensel 1979; Kovac 1998). The Danube River and Black and Caspian Seas form the southern border of Ruffe's native range (Popova *et al.* 1998).

### *NON-NATIVE RANGE*

Ruffe has established populations in Lake Piediluco (Lorenzoni *et al.* 2009), Lake Ghirla, and Lake Mergozzo, Italy (Volta *et al.* 2013); Bassenthwaite Lake (Stepien *et al.* 1998; Winfield *et al.* 2004), Derwent Water, and Windermere, England (Winfield *et al.* 2010, 2011); Loch Lomond, Scotland (Maitland and East 1989; Adams 1991); Llyn Tegid (Bala Lake), Wales (Winfield 1992; Winfield et al. 1998; Winfield et al. 2011); Lake Constance, Germany, Austria, and Switzerland (Winfield et al. 1998; Eckmann 2004); Lake Geneva, Switzerland and France (Matthey 1966; Winfield et al. 1998); and Lake Mildevatn, Norway (Kalas 1995) (Figure 1).

In North America, Ruffe was introduced to the Laurentian Great Lakes in the 1980s via ballast water releases, establishing populations in both US and Canadian waters of Lake Superior, Lake Michigan, MI, and Lake Huron, MI. Propagule pressure (i.e., the abundance and frequency of Ruffe introduced) on the Great Lakes has been low (Kolar and Lodge 2001); genetic evidence suggests there was a single founding population from the Elbe River drainage region, Germany (Stepien *et al.* 2005). Among the Great Lakes, Ruffe is most abundant in Lake Superior (Figure 2); the highest densities have been observed in the St. Louis River, MN-WI (Figure 2A), and Chequamegon Bay, WI (Figure 2B).

### ***LIFE HISTORY REQUIREMENTS: CHEMICAL***

Ruffe tolerate a wide range of salinity (0-12 ppt) (Lind 1977) and pH (as eggs 6.5-10.5) (Kiyashko and Volodin 1978) (Table 1). It lives in waters ranging from oligotrophic to eutrophic but prefer eutrophic waters (Fedorova and Vetkasov 1974; Disler and Smimov 1977; Leach *et al.* 1977; Hansson 1985; Johansson and Persson 1986; Bergman 1988a, 1990, 1991; Bergman and Greenberg 1994; Rösch *et al.* 1996; Popova *et al.* 1998; Lehtonen *et al.* 1998; Brown *et al.* 1998). Ruffe may thrive in eutrophic waters for several reasons: it has a sophisticated lateral line system and sensory organs that aid mechanoreception in turbid waters (Disler and Smimov 1977; Johansson and Persson 1986; Bergman 1988a, 1990, 1991; Popova *et al.* 1998); Ruffe prefers to consume benthic invertebrates, and there may be an abundance of benthic organisms in eutrophic waters (Leach *et al.* 1977); and there may be less predation pressure and competition than in oligotrophic waters because its adaptations to low-light conditions aid avoidance of native piscivores and provide a foraging advantage compared to native demersal fishes (Bergman 1991; Lehtonen et al. 1998).

### ***LIFE HISTORY REQUIREMENTS: PHYSICAL***

Although Ruffe is considered a ‘temperature generalist,’ it is adapted for cold water rather than warm water (Bergman 1987; Hölker and Thiel 1998).

Adult Ruffe can feed at temperatures as low as 0.2°C (Lake Vortsjarv, Estonia) (Kangur *et al.* 1999) (Table 1) and is active and feeding at 4-6°C in other locations (Bergman 1987; Eckmann 2004; Tarvainen *et al.* 2008). In the Danube River, when the temperature is 16.2-23.0°C, Ruffe embryos hatch in 8 days and larvae transition to juveniles in 20 days (Kovac 1998) (Table 1). Hokanson (1977) stated that the optimal growth temperature for larval Ruffe is 25-30°C (Table 1). For juveniles, after an acclimation temperature of 20°C for 11 days, the upper incipient lethal temperature (i.e., the temperature at which 50% of individuals will die if exceeded) is 30.4°C (Alabaster and Downing 1966; Hokanson 1977); whereas, with an acclimation in the field with temperatures ranging from 24.1-25.7°C, the juveniles' critical thermal maximum (i.e., the temperature at which locomotory activity becomes disorganized) is 34.5°C (Horoszewicz 1973; Hokanson 1977) (Table 1). Based on a bioenergetics model, maximum consumption in laboratory conditions for adults occurs at 18-22°C (Tarvainen *et al.* 2008).

Ruffe spawns between 5-18°C in the non-native North American range (Brown *et al.* 1998). Notably, the minimum spawning temperature reported in the native range was 11.6°C, whereas the maximum reported was 18°C (Hokanson 1977).

Ruffe has been captured at depths of 0.25-85 m (Nilsson 1979; Van Densen and Hadderingh 1982; Sandlund *et al.* 1985) in its native range (Table 2). However, in Lake Superior, USA, Ruffe has been captured from 0.2-205 m (USGS, personal comm., 2014) (Table 2). In the eastern portion of their non-native range, Ruffe was caught as shallow as 4.9 m in Mildavetn, Norway (Kalas 1995) and as deep as 70 m in Lake Constance, Germany (Eckmann 2004) (Table 2).

### *LIFE HISTORY REQUIREMENTS: BIOLOGICAL- FEEDING HABITS AND BEHAVIORS*

Adult Ruffe often lives in shoals (Kontsevaya and Frantova 1980; Popova *et al.* 1998). In North America, it competes for food resources with native fishes, such as Emerald Shiner (*Notropis atherinoides*), Yellow Perch (*Perca flavescens*), Trout-perch (*Percopsis omiscomaycus*), and other benthic planktivores (Ogle *et al.* 1995; Fullerton *et al.* 1998; MN Sea Grant 2013). Ruffe possesses a tapeta lucidum and sensitive lateral line systems, allowing it to forage in low-light conditions (Hölker and Thiel 1998). On each side of the head are three large lateral line canals (Jakubowski 1963; Wubbels 1991), inside of which are neuromasts that contain approximately 1000 hair cells and are innervated by about 100 afferent fibers (Wubbels *et al.* 1990). These canals provide directional sensitivity (especially to sound frequencies lower than 20 Hz (Gray and Best 1989)), allowing Ruffe to detect prey in low-light conditions when vision cannot be used (Wubbels 1991). In addition, it is speculated that Ruffe is fine-tuned to detect sound frequencies of their primary food item, chironomid larvae, which live in the surface of the mud on the bottom of a water body (Gray and Best 1989). This well-adapted foraging technique gives Ruffe a significant advantage over many fishes for feeding in deep, dark water, especially at night and during ice-cover (Eckmann 2004). Native fishes select against Ruffe; Mayo et al. (1998) found that native predators in Lake Superior, USA, including Northern Pike (*Esox lucius*), Smallmouth Bass (*Micropterus dolomieu*), Brown Bullhead (*Ameiurus nebulosus*), Walleye (*Sander vitreus*), and Yellow Perch, preferentially selected native fish species to eat even when Ruffe composed 71-88% of the available prey biomass in the environment.

### *LIFE HISTORY REQUIREMENTS: HABITAT*

Adult Ruffe generally is demersal (Holcik and Mihalik 1968; Sandlund *et al.* 1985; Bergman 1988a) and prefer sandy, silty, well-aerated, slow-moving water with little or no vegetation (Kontsevaya and Frantova 1980; Popova *et al.* 1998; Ogle 1998) (Table 1). Ruffe inhabit lakes, rivers, ponds, bays, brackish waters, tidal estuaries, non-tidal estuaries, and reservoirs in its native range (Hölker and Thiel 1998). In non-native regions in North America, Ruffe is found in rivers, lakes, and coastal wetlands (Pratt 1988; Fairchild and McCormick 1996; Sierszen *et al.* 1996; Brown *et al.* 1998; Selgeby 1998; Stepien *et al.* 1998; Ogle *et al.* 2004; Ogle 2009; Peterson *et al.* 2011; USGS 2014); whereas, in other non-native regions, Ruffe is restricted to lakes and reservoirs (Wootten 1974; Maitland and East 1989; Duncan 1990; Kalas 1995; Eckmann 2004; Winfield *et al.* 2004; Lorenzoni *et al.* 2009; Volta *et al.* 2013) (Table 2).

Ruffe readily alters its behavior when introduced to a new water body. For example, Kalas (1995) demonstrated that Ruffe underwent a change in habitat use and prey consumption after introduction to Mildevatn, Norway, a lake that differs with respect to its fish and prey community structure from lakes in Ruffe's native range. Ruffe in Mildevatn fed primarily on zooplankton during June-

September. Further, it was mainly active during the day; 84% were caught during the day, significantly more compared to night capture (Kalas 1995) (Table 2). This finding is unusual, as Ruffe is typically nocturnal (Jamet and Lair 1991) or crepuscular (Westin and Aneer 1987).

### *EGGS*

Ruffe can spawn multiple times per season (Fedorova and Vetkasov 1974; Kolomin 1977; Ogle 1998); spawning is intermittent and asynchronous (Hokanson 1977). Multiple studies report that Ruffe in its native range batch spawn (i.e., release multiple clutches of eggs throughout the spawning season) (Koshelev 1963; Fedorova and Vetkasov 1974; Hokanson 1977; Kolomin 1977) (Table 2). In Lake Glubokoe in the Moscow region of Russia, Ruffe spawned up to three batches in a two-month period (Koshelev 1963). Ruffe has the capacity to release up to three clutches of eggs (Lake Glubokoe, Russia (Koshelev 1963)); however, only two clutches typically are released in their native habitat (Fedorova and Vetkasov 1974; Hokanson 1977; Kolomin 1977) (Table 2). In the North American population, Brown et al. (1998) noted a prolonged spawning period, but they were unable to provide evidence for Ruffe laying multiple clutches of eggs (Table 2).

The first batch of eggs matures over winter (165 days (Hokanson 1977)) and is laid in the spring or early summer. The second batch, if there is one, matures during the summer (30 days (Hokanson 1977)) and is laid during the late summer (Koshelev 1963; Ogle 1998). During maturation, oocyte resorption of unspawned ova from a previous batch can occur without interfering with the growth of the current batch (Hokanson 1977).

Ruffe eggs are adhesive and laid on a variety of substrates (Balon *et al.* 1977; Collette *et al.* 1977) (Table 1, Figure 3A). A study conducted in the St. Louis River, USA, found the spawning period to last about 8 weeks, spanning April to June (depending on the year), during which temperatures ranged from 5-18°C (Brown *et al.* 1998) (Table 1). Hokanson (1977) stated that because of the fast rate of oocyte maturation, Ruffe requires relatively high temperatures (>11.6°C) (Bastl 1969) for spawning in their native range when compared with other percids, including Walleye, Eurasian Perch (*Perca fluviatilis*), Yellow Perch, and Pikeperch (*Sander lucioperca*), which all have lower spawning temperature limits (2-5°C). Ruffe embryos may require high dissolved oxygen concentrations because they lack a subintestinal-vitelline system and segmental vessels (Kovalev 1973; Kovac 1993); therefore, spawning grounds may need to be well-oxygenated (Table 1).

Fecundity is size-dependent and varies among water bodies (Kovac 1998). Neja (1988) found that absolute fecundity (total number of eggs per female) is less correlated to body length (r=0.752) than to body weight (r=0.801). In a study conducted in the side-arm of the Danube River in Baka, Slovakia (native range), the mean absolute fecundity for the first batch of a spawning female with a mean length of 96.3 mm was 23,731 eggs; the mean relative fecundity was 1,284 eggs/ gram of body weight (Bastl 1988; Kovac 1998). Fecundity estimates in the non-native range are limited. In Lake Piediluco, Italy (non-native) fecundity estimates were much smaller than those observed in most regions in the native range, although there was no information on batch spawning (Lorenzoni *et al.* 2009) (Table 2): the mean absolute fecundity was highly correlated with size—absolute fecundity ranged from 550 to 52,000 and the mean relative fecundity was 240 eggs/ g (Lorenzoni *et al.* 2009).

Absolute fecundity estimates for the first spawning batch range from 1,000 (Kovac 1998) to 200,000 eggs (Fedorova and Vetkasov 1974; Collette *et al.* 1977; Kolomin 1977; Neja 1988). Relative fecundities range from 585 to 1,540 eggs/ g (Neja 1988; Kovac 1998) in the native range but from 72 to 513 eggs/ g in the non-native range (Lorenzoni *et al.* 2009). The second batch was documented as being substantially smaller than the first batch in the native range: 352 – 6,012 eggs (Kolomin 1977). Kolomin (1977) determined that the first batch can be almost six times larger than the second batch.

Ruffe ovaries contain three types of eggs, only two of which are used during the spawning season (Neja 1988; Ogle 1998). The type that is not used is small, colorless, and glassy in appearance. The two that are used for spawning are in two different groups: 1) larger, opaque, whitish or light yellow to yellow or orange and 2) large, partly glassy, yellow or orange (Neja 1988; Ogle 1998). In the Danube River, Slovakia, Ruffe eggs were spherical and yellow (Kovac 1993, 1998).

Various ranges of egg diameter have been reported: 0.97-1.07 mm (Kovac 1998), 0.5-1 mm (Collette *et al.* 1977), 0.90-1.21 (Kolomin 1977), 0.71-1.59 mm (Lorenzoni *et al.* 2009), and 0.64-0.98 mm (Neja 1988) (Table 1). Ruffe in the Danube River and central and eastern Europe is thought to undergo saltatory ontogeny, described as seven embryonic stages and three larval stages prior to juvenile transition (Balon 1990). The embryonic period lasts approximately eight days when the water temperature is 16.2-23°C (Kovac 1998). The time to hatch is temperature-dependent. At 10-15°C, Ruffe eggs hatch 5-12 days post-fertilization (Maitland 1977; Craig 1987); whereas eggs hatch 4-6 days after fertilization when temperatures range 16.2-23°C (Balon 1990; Kovac 1998) (Table 1).

***LARVAE***

Ruffe is 3.35-4.40 mm long at hatch (Fedorova and Vetkasov 1974; Kovac 1998) (Figure 3B, Table 1). It is stationary on the bottom of the water body for 3-7 days until they grow to 4.5-5.0 mm (Disler and Smimov 1977). Temperature for optimum growth in its native range is 25-30°C (Hokanson 1977) (Table 1). Approximately one week after hatch, larvae transition to exogenous feeding (French III and Edsall 1992) and remain demersal (Disler and Smimov 1977) (Table 1). At this stage, it is about 6-8 mm long and feeds primarily on zooplankton and small benthic invertebrates (Popova *et al.* 1998).

Although Ruffe generally is demersal after yolk sac absorption, it may temporarily occupy pelagic habitats to feed on large zooplankton prey (Popova et al. (1998) (native), Kalas (1995) (non-native)). By the end of the larval stage (16-18 mm), its prey includes large zooplankton (e.g., cladocerans, large copepods), ostracods, and small chironomids (Johnsen 1965; Ogle *et al.* 1995; Kangur and

Kangur 1996; Werner *et al.* 1996; Popova *et al.* 1998). The larval stage is about 20 days when temperatures range from 16.2-23°C (Kovac 1998) (Table 1).

Larvae can undertake both horizontal (i.e., between inshore and offshore) and vertical movements. Because it is sensitive to hypoxia, larval Ruffe may leave shallow spawning sites (less than 5 m) for deeper, cooler, well-oxygenated areas (Popova *et al.* 1998) (Table 1). In the Al. Stamboliiski Reservoir, Bulgaria (south), and the Votkinskoe Reservoir, eastern Russia (temperate), diel vertical migration (DVM) was observed in which larvae were concentrated at the surface (0-1 m) at night and concentrated at the bottom (5-6 m) during the day (Popova *et al.* 1998). Despite this isolated example, Ruffe larvae typically do not typically undergo DVM (Johnsen 1965; Fedorova and Vetkasov 1974; Disler and Smimov 1977; Ogle 1998).

### *JUVENILES*

After the embryonic (8 days) and larval stage (20 days), the juvenile stage begins about 28 days after hatching (Kovac 1998) (Figure 3C, Table 1). Juveniles forage during the day, dawn, and night, although, more so at dawn and night (Disler and Smimov 1977; French III and Edsall 1992). Unlike larvae, they typically live in shoals (Disler and Smimov 1977; French III and Edsall 1992) and will undergo DVM during the summer, occupying deep water at night and shallow inshore habitat at dawn (Kovac 1998; Peterson *et al.* 2011). Juveniles may migrate from upstream reservoirs to downstream water bodies (Kovac 1998). However, in a survey of 22 lakes and reservoirs in temperate and northern Russia (native range), downstream movement of Ruffe was only observed in

54% of cases, while movements by European Perch and Pikeperch were more frequent, 75% and 100%, respectively (Popova *et al.* 1998). In temperate regions, from about June to July, juvenile Ruffe has been found to move from littoral to profundal areas in lakes in the former USSR (native range) (Mikheev and Pavlov 1993; Popova *et al.* 1998) (Table 1).

Juveniles also make seasonal movements. For example, in Russia (native range), they move to the deepest part of the body of water in which they reside, regardless of whether it is a lake, river, reservoir, or estuary to overwinter (Kovac 1998) (Figure 3E-F, Table 2). In June and July, juvenile Ruffe (40-60 mm) in the St. Louis River, USA (non-native range), was collected to determine habitat use; based on stable isotope ratios, half of the sample demonstrated recent use of Lake Superior habitat, and the other half showed recent use of river habitat (Hoffman *et al.* 2010) (Table 1).

In both the native and non-native range, juvenile Ruffe primarily consumes benthic invertebrates (Popova *et al.* 1998; Hoffman *et al.* 2010) (Table 1). However, if there is high abundance of large zooplankton prey, adult and juvenile Ruffe will ascend to the pelagic zone to feed periodically (Popova *et al.* 1998) (Table 1).

### *ADULTS: AGE AND SIZE AT MATURITY*

Age at maturity for Ruffe varies from 1-4 years (Fedorova and Vetkasov 1974; Craig 1987; Neja 1988; Jamet and Lair 1991) (Figure 3H, Table 2). At the northern range of their climate, Ruffe matures at 2-3 years of age (Lind 1977; Maitland 1977; Ogle 1998). Presumably due to the northern climate, Ruffe in

Finland reached maturity at the age of 2-3 (Lind 1977; Lappalainen and Kjellman 1998) (Table 2). In the Nadym River basin, Russia (northern portion of the native range), Ruffe mature as early as age 2 but usually at age 3 or 4; most spawning Ruffe were reported to be 3-7+ years, between 20-30 grams and 110-120 mm (Kolomin 1977) (Table 2). However, in the Baka system of the Danube River (southern border of the native range), females matured between 57-90 mm and males matured at 80+ mm (Bastl 1988) (Table 2). Early maturity could be caused by a response to high mortality rates at the population level (Lind 1977) or to warmer water at a physiological level (Fedorova and Vetkasov 1974; Craig 1987).

No studies have been conducted on the age and size at maturity of the North American population; however, Ogle (1998) reported estimates of 2-3 years of age and 110-120 mm, based on Lind's (1977) Finland study and Maitland's (1977) fish guide to Britain and Europe. In the non-native population in Lake Piediluco, Italy, the age of maturity for both sexes was age 1; however, size of maturity varied between sexes—females matured at 78.74 $\pm$ 0.83 mm while males matured smaller at 69.42 $\pm$ 1.91 mm (Lorenzoni *et al.* 2009) (Table 2). In Loch Lomond, Scotland (non-native range), female Ruffe matures at 11.67 g and males at 7.5 g (Devine *et al.* 2000) (Table 2).

***ADULTS: MAXIMUM AGE AND SIZE***

Reports from Ruffe's native range in Finland and parts of Europe and non-native range in Britain indicate females live up to 11 years and males up to 7 years of age (Lind 1977; Maitland 1977; Crosier and Molloy 2007) (Table 2).

Whereas, in the Ob' River, Russia (native range), Ruffe was as old as 20 years of age (Popova *et al.* 1998) (Table 2). Popova et al. (1998) noted that there are regional age differences—in temperate water bodies, the maximum age is typically 10 years, but in southern water bodies, the maximum age is closer to 8 years (Table 2).

Maximum age in the North American population (non-native range) was extrapolated from the native range. Given that the majority of Ruffe occurrences are in the Great Lakes fall in the 30°N temperate zone, the maximum age should be about 10 years based on former USSR information from Popova et al. (1998) (Table 2). Similarly, in the non-native ranges in Europe, Britain, and Scandinavia, one can infer the maximum age to be 8-10 years (Popova *et al.* 1998) (Table 2) because the introduced populations span from temperate to the southern regions. In Lake Piediluco, Italy (non-native range), the maximum age is 6 years (Lorenzoni *et al.* 2009) (Table 2).

The most-cited maximum length (290 mm) reported for Ruffe was from the Elbe River estuary (as cited in Holker and Thiel 1998), where adult Ruffe average size is about 250 mm (Holker and Hammer 1994) (Table 2). According to Berg (1949), a 500 mm Ruffe was caught in Siberia; however, this report has never been confirmed (Sanjose 1984) (Table 2). In Finland, it was reported that Ruffe only reach 200 mm (Lind 1977) (Table 2). Ruffe often do not grow to a large size in freshwater habitats. In the non-native North American population, the maximum size recorded was 207 mm (Ogle and Winfield 2009) (Table 2). In European non-native populations, Eckmann (2004) state Ruffe obtains lengths of

124 mm (Lake Constance, Germany), and Lorenzoni et al. (2009) report that the maximum length in Lake Piediluco, Italy is 191 mm and maximum weight is 141 g (Table 2).

### *ADULTS: FEEDING HABITS*

In their native range, adult Ruffe primarily feeds on benthic organisms, generally chironomid larvae or pupae (Johnsen 1965; Polivannaya 1974; Kozlova and Panasenko 1977; Boikova 1986; Nagy 1988; Jamet and Lair 1991; Kangur and Kangur 1996; Werner *et al.* 1996; Kangur *et al.* 2000). Ruffe also consumes *Chaoborus* (Glassworm) larvae, Perlodidae (Stonefly) larvae, Culicidae (Mosquito) pupae, Ceratopogonidae (Biting Midge) larvae, (Jamet and Lair 1991), Tricoptera (Caddisfly) larvae (Polivannaya 1974; Jamet and Lair 1991; Ogle *et al.* 1995; Kangur *et al.* 2000), Odonata (Dragonfly) larvae, and Ephemeroptera (Mayfly) larvae (Ogle et al., 1995; Polivannaya, 1974); crustaceans (Johnsen 1965; Kozlova and Panasenko 1977) – *Asellus* (isopods), Ostracoda (Johnsen 1965; Kangur and Kangur 1996; Kangur *et al.* 2000), mysids, and brown shrimp (Holker and Hammer 1994) – and, when large enough, juvenile fish, such as Yellow Perch (Kozlova and Panasenko 1977) or Rainbow Smelt (*Osmerus mordax*) (Kozlova and Panasenko 1977; Holker and Hammer 1994; Kangur and Kangur 1996). Adult Ruffe periodically feeds on zooplankton (Kozlova and Panasenko 1977; Kangur *et al.* 2000), including copepods (Johnsen 1965; Boikova 1986; Holker and Hammer 1994; Kangur and Kangur 1996; Werner *et al.* 1996), cladocerans (Johnsen 1965, Boikova 1986, Kangur and Kangur 1996, Werner et al. 1996), and adult *Chaoborus* (Boikova 1986; Werner *et al.* 1996).

Ruffe eats Vendace (*Coregonus albula*) and Powan (*Coregonus lavaretus*) eggs in their native and non-native range (Kangur and Kangur 1996; Selgeby 1998; Kangur *et al.* 2000), potentially impacting populations of these fishes in some invaded areas (Adams and Tippett 1991) (Table 2). Lab experiments have been conducted to determine predation effects on Whitefish (*Coregonus* spp.) eggs (Mikkola *et al.* 1979; Sterligova and Pavlovskiy 1984; Pavlovskiy and Sterligova 1986) and demonstrated that Ruffe will eat the eggs, especially if there is no other prey (Sterligova and Pavlovskiy 1984) or if the eggs are fertilized (Mikkola *et al.* 1979). When Ruffe establishes populations in new water bodies, however, its feeding habits can shift to acclimate to the local habitats.

Adult Ruffe primarily feeds in shallow, littoral habitats at night (Leszczynski 1963; Holcik and Mihalik 1968; Jamet and Lair 1991) or twilight (Westin and Aneer 1987) and move to deeper waters during the day (Holcik and Mihalik 1968; Ogle *et al.* 1995) (Figure 3J, Table 1). However, in the St. Louis River, USA (non-native), adult Ruffe fed during the day in deep water (Ogle *et al.* 1995).

### ***ADULTS: MOVEMENTS***

Ruffe populations undergo routine movements throughout their life cycle (Figure 3); these movements vary by season and life stage and influence their distribution among habitats. Some important abiotic factors that affect its distribution include current velocity (in rivers), temperature, oxygen concentration, and salinity (in estuaries); however, food availability is probably the most important factor influencing movements and distributions of Ruffe (Popova *et al.* 1998). On a daily basis, predation risk can also cause Ruffe to change habitats and activity patterns (Popova *et al.* 1998).

### ***ADULTS: SEASONAL MOVEMENTS***

Adult Ruffe moves seasonally from shallow water during summer months to deep water (up to 70 m) in the fall and during spring ice-out to overwinter (Johnsen 1965; Kolomin 1977; Sandlund *et al.* 1985; Kovac 1998; Popova *et al.* 1998; Brown *et al.* 1998; Selgeby 1998; Eckmann 2004) (Figure E-G). Factors influencing the timing and location of seasonal movements include refuge from water currents, dissolved oxygen, salinity, or food availability (Johnsen 1965; Sandlund *et al.* 1985; Kovac 1998; Popova *et al.* 1998).

In the native Nadym River basin, Russia, many of the flood-plain lakes experience extreme hypoxia and freezing conditions, persisting from February to March. As a result, Ruffe moves seasonally, descending into Ob' Bay in late October and early November to overwinter and returning to the Nadym River in the spring (Kolomin 1977).

In the non-native range in North America, Ruffe was observed several kilometers offshore during December in Lake Superior, USA, at a depth of 15-30 m where they fed on Mysis (*Mysis diluviana*) and Cisco (*Coregonus artedi*) eggs (Selgeby 1998). It is likely these Ruffe return to nearby tributaries, such as the St. Louis River, USA, to spawn in the spring (Figure G-J). Ruffe may also remains in tributaries during the winter but moves to deep, channel habitats. In deep channels in the St. Louis River, USA, Ruffe was more abundant at ice out than during the summer months and was observed returning to deep channels when winter returned (Brown *et al.* 1998).

Ruffe was captured at depths of 30-70 m in the winter in Lake Constance, Germany (non-native), which suggests it had moved offshore (Eckmann 2004). In another invaded lake, Loch Lomond, Scotland, gut contents analysis showed no difference between winter and summer diets, possibly indicating these Ruffe were not moving; however, there was no mention of where the fish were captured (Adams and Tippett 1991). In Lake Mildevatn, Norway (non-native), Ruffe stayed in deep water in the winter and moved to shallow water in the summer, possibly due to spawning, change in diet, or reduced oxygen concentration (Kalas 1995).

### *ADULTS: SPAWNING MOVEMENTS*

Ruffe spawning habitat varies with respect to both water quality and substrate. Spawning occurs in shallow water, approximately three meters or less, with pH levels of 6.5-10.5 for normal egg development (Kiyashko and Volodin 1978) (Table 1). Temperatures need to range from 6-18°C (Kovalev 1973; Fedorova and Vetkasov 1974; Kolomin 1977; Willemsen 1977; Kiyashko and Volodin 1978; Neja 1988; Ogle 1998) (Table 1). Spawning substrate varies; Ruffe can deposit their eggs on submerged plants, branches, rocks, or logs (Balon *et al.* 1977) (Table 1). Collette et al. (1977) found that Ruffe lays their eggs on sand, clay, or gravel substrates (Table 1). Field studies have supported both of these findings (Kovalev 1973; Fedorova and Vetkasov 1974; Kolomin

1977), suggesting that Ruffe spawning substrate is either population- or environment-specific.

Prior to spawning, Ruffe moves in shoals from their deep, overwinter habitats toward shallow, nearshore habitats (Figure 3G-J). Ruffe moves along the shoreline and concentrate at the mouths of rivers in its native habitat in the former USSR, including Kursian Bay, Syam Lake, rivers Prut and Dniester basins, lakes of the Bolshezemelskaya Tundra, bays of Ob' and Taz, and Lake Zaisan (Kontsevaya and Frantova 1980; Popova *et al.* 1998). In the waters of Kazakhstan, Ruffe shoals appear under the ice in March prior to spawning (Popova *et al.* 1998). Further, in the lakes of the Bolshezemelskaya Tundra, spawning Ruffe was already in shoals near the shore during the break-up of ice. Females arrived two days after the males to the spawning grounds (Popova *et al.* 1998).

### *ADULTS: GENOTYPE AND MORPHOLOGY*

There are genetic and phenotypic differences among native and non-native populations. Stepien et al. (1998) identified five mitochondrial DNA control region haplotypes: a North American (Laurentian Great Lakes, USA) and Danube River, Slovakia haplotype; a Bassenthwaite Lake, United Kingdom haplotype; a St. Petersburg, Russia, including the Neva River Embankment and Komsomolskoe Lake haplotype; and two haplotypes in the Ob' River at Novosibirsk, Siberia, Russia (Table 2).

Within these haplotypes, there are two distinct groups that are genetically and morphologically different: a North America-Danube-Elbe River group and a

Bassenthwaite Lake-St. Petersburg-Ob' River group. Between the two groups, there is a mean genetic distance of 0.010 $\pm$ 0.0035, which is close to the distance (0.016 $\pm$ 0.005) separating two species of *Gymnocephalus* that Stepien et al. (1998) also examined. Multiple Ruffe experts have stated that Ruffe in the Danube River (same as North American Ruffe) are morphologically different than Ruffe in any other European regions, and the Danube River Ruffe was previously classified as a distinct morphotype (Stepien *et al.* 1998), *G. cernuus* natio *danubica*. There are four significantly different morphological traits among the five haplotypes of Ruffe. These traits include the relative length of the caudal peduncle, the number of pre-opercular spines, the relative length of the anal fin, and the number of soft spines in the dorsal fin (Stepien *et al.* 1998).

Based on mitochondrial DNA, the North American population matches the Danube and Elbe River population (Stepien *et al.* 1998); Stepien et al. (2005) had similar findings based on mtDNA and found that the Elbe River population matched the North American population. However, based on nuclear DNA, Stepien et al. (2005) determined that the Great Lakes population was established by a single founding population from the Elbe River drainage. More recent results using 10 nuclear DNA microsatellite loci confirm that the North American Ruffe population genetically matches that from the Elbe River region (C. Stepien, personal comm.). Moreover, Ruffe in North America has remained genetically similar over 20 years, with no evidence of additional introduction events, indicating that spread throughout the northern Great Lakes stemmed from the original population that was established at Duluth, MN (an international maritime freshwater port) in the St. Louis River, a tributary to Lake Superior (C. Stepien, personal comm.).

## Summary/ Conclusion

Ruffe has a wide tolerance for chemical, physical, and biological conditions. This tolerance reflects their wide geographic distribution and utilization of a broad range of aquatic habitat types, including lakes, rivers, ponds, bays, coastal wetlands, brackish waters, tidal estuaries, non-tidal estuaries, and reservoirs. Ruffe also demonstrates variable movement and feeding strategies that are responsive to local environmental conditions. These characteristics help to explain the ability of Ruffe to successfully invade a wide variety of lakes and reservoirs. Yet, I did find that non-native populations have more restricted habitat use compared to native populations. Further, I found differences among native, non-native North American, and European non-native populations with regard to life stage-specific characteristics (i.e., number of eggs, reproduction, feeding habits, movements, and size and age). Several key knowledge gaps include geographic discrepancies and lack of data with respect to the native range; lack of reproduction information for populations in the non-native range, specifically from North America; and an overall lack of overwintering studies in both the native and non-native ranges. These topics are specifically addressed in the discussion. I further discuss the ecological implications of variability in life history characteristics between the native and non-native range, as well as management implications for Ruffe spread and invasion.

### ***UNCERTAINTIES IN NATIVE AND NON-NATIVE RANGE***

Despite an exhaustive literature search, my proposed range is fragmented in some regions, indicating undocumented introduction or lack of occurrence data (i.e., I could not distinguish between the absence of studies citing Ruffe captures in specific areas and actual Ruffe absences). The largest of these gaps is between the Nadym River, Russia (East) and Volga River, Russia (West). Although water bodies connect these rivers, I found no known Ruffe occurrences in this area.

Further, the biogeographic information is lacking and ambiguous in some regions. For example, many literature sources state that southern England is part of the native Ruffe range, but I could not find specific occurrences by water body in this region. Stepien and Haponski (2015) indicate the range of Ruffe is somewhat more widespread than my range, especially in Russia and Asia, a region for which the range has been poorly described. I found few occurrences within Eurasia, and thus the range within Eurasia should be interpreted with due caution. Stepien and Haponski (2015) also include regions within Ukraine in the range, whereas I found no published occurrences for that region. As with Russian and Asia, the range description would benefit from additional occurrence data here. I also have more discontinuities throughout Norway than Stepien and Haponski (2015). This discrepancy is likely due to my elevation cut-off, which was based on the highest elevation native Ruffe has been found.

Further, Ruffe may have been introduced to more locations than we are presently aware. For example, in the southwest region of the map, there is a native population in Lake Aydat, France, that is not connected to the rest of the range. Nearby, there is a cluster of introduced populations in Italy and Germany. The population in Lake Aydat could be introduced but was not documented as such because Ruffe was already present when the study was conducted. The author of the study simply states, “These fishes are widely distributed in European waters” (Jamet and Lair 1991). Lake Vastra Kyrksundet on the Aland Islands in Finland is another example of a potentially undocumented introduction (Bonsdorff and Storberg 1990). Ocean surrounds the island on all sides (on the eastern side there is a series of islands), but a native Ruffe population exists in a lake in the middle of the island. In 1932, a small artificial canal was built connecting the lake to the Baltic Sea. This tributary was blocked by a dam in 1979 in an attempt to return the lake to its original hydrological and ecological conditions (Bonsdorff and Storberg 1990). It is unknown if Ruffe was in the lake prior to 1932, but Bonsdorff and Storberg (1990) suggest it was. Ruffe is native and present along the coasts of Sweden and Finland in the Baltic Sea, Gulf of Finland, and the Gulf of Bothnia, so it is feasible that Ruffe could have established there naturally.

### *KNOWLEDGE GAPS AND UNCERTAINTIES*

I determined that there were several substantial knowledge gaps in the scientific literature, specifically, reproduction information from the non-native range and overwintering ecology. Based on my review, data on fecundity, age and size at maturity, and spawning movements are all lacking for populations in the non-native range, especially North America. Few studies have been conducted on overwintering ecology and movements, an important stage for temperate fishes because it potentially represents a “bottleneck” for population size due to poor habitat condition (Reimers 1963, Cunjak and Power 1987, Nickelson et al. 1992, Giannico and Hinch 2003). Overwintering is also a period during which Ruffe may disperse. There are a few studies addressing overwintering in native and non-native European ranges and only one in North America. Despite the difficulty of sampling during the winter, there need to be more studies to identify overwintering habitat, including location, environmental character (i.e., depth, temperature, food availability), and differences between adults and juveniles.

***NATIVE VERSUS NON-NATIVE POPULATIONS***

I found substantial differences in certain life history characteristics, including maturity, size and growth, and temperature, between the native and non-native range. I found age at maturity to be based on latitude—generally, Ruffe further north matures later than southern Ruffe populations. Also, maximum length of Ruffe is almost always greater in native ranges than non-native ranges, possibly because in the native range Ruffe inhabits highly productive brackish water that provides high amounts of food, whereas Ruffe solely lives in freshwater in the non-native range. In addition to food abundance, Hölker and Thiel (1998) proposed that Ruffe has higher growth rates in brackish water due to temperature or salinity (or both).

Finally, Ruffe demonstrates adaptability to temperature differences between native and non-native ranges. For example, Hokanson (1977) stated that the optimal growth temperature for larval Ruffe is 25-30$^{o}$C, a temperature range that is rarely reached in its non-native North American range. Similarly, in the non-native North American range, Ruffe begins spawning at temperatures as low as 5$^{o}$C (Brown *et al.* 1998), but Ruffe requires a higher temperature (>11.6$^{o}$C) for spawning in the native range (Hokanson 1977). Ruffe seems to be well-adapted to the cooler temperatures of some of its native and non-native habitats; however, this adaptation is not without consequence. Ruffe in colder climates at more northern latitudes generally is shorter in maximum length (Eckmann 2004; Hölker and Thiel 1998; Lind 1977; Lorenzoni et al. 2009; USFWS, personal comm. 2014), matures later (Lind 1977; Maitland 1977; Ogle 1998), is smaller at maturity (and therefore likely less fecund) (Kolomin 1977; Devine *et al.* 2000; Lorenzoni *et al.* 2009), and requires longer for eggs to hatch (Maitland 1977; Craig 1987), leaving it vulnerable to predators for a longer period of time.

### ***IMPLICATIONS FOR SPREAD AND ESTABLISHMENT***

Even though it is not a highly migratory fish, Ruffe has spread and established populations across continents. Ruffe is particularly able to disperse and spread during the larval stage and the overwintering period. During the larval stage when Ruffe is a few millimeters long, water currents can potentially disperse it long distances. Further, although larvae are generally demersal, they can move into open waters where they are vulnerable to entrainment in ballast water by commercial ships and subsequent inadvertent translocation (as with the introduction to North America). At this small stage, accidental, human-mediated transport by bait bucket is also possible (commonly implicated with introductions in England).

Juvenile and adult Ruffe can move long distances to overwintering grounds. In some cases, these grounds were greater than 15 kilometers away from the summer rearing grounds; however, the fidelity to a specific spawning location is not known. Characterizing movements between spawning grounds and overwintering grounds, as well as straying rates when returning to natal spawning grounds, is likely important to understand spread across large, hydrologically-connected landscapes. Spread may be limited by spawning habitat availability. Each year, mature Ruffe must find warm (5-18˚C) and shallow (<5 m) habitat to spawn. However, Ruffe overwinters at depths greater than 15 m, so individuals must move inshore to spawn. In aquatic landscapes where suitable spawning habitat is widely geographically separated, this could limit dispersal.

Multiple traits combine to facilitate the successful establishment of Ruffe in an introduced water body. Ruffe has a broad tolerance for environmental conditions, including salinity, pH, and trophic level, and thus are able to inhabit a broad array of aquatic habitat types and conditions. Ruffe rapidly matures and can reproduce annually thereafter. It has a high fecundity with the ability to batch spawn for a prolonged spawning period, which is a useful trait for successful reproduction in variable environments (Koshelev 1963; Fedorova and Vetkasov 1974; Hokanson 1977; Kolomin 1977). It has multiple defenses against predators, such as a large dorsal spine, sensitive lateral line, and strong night vision. Ruffe is also an effective competitor, especially in dark environments, due to their tapeta lucidum, and sensitive lateral line (making hunting for food easier). Further, Ruffe can change its diet preference to select for the most abundant prey, which is a useful trait when introduced to a new water body.

Based on Ruffe's life history strategies and occurrence patterns in its native and non-native ranges, all of the Laurentian Great Lakes and many water bodies, particularly lakes, in the U.K., Europe, and Scandinavian countries are vulnerable to a Ruffe invasion. However, my review suggests there may be broad constraints to the spread and ecological impact of Ruffe establishment. To date, the types of water bodies in which it has established have been limited to lakes and reservoirs. Because Ruffe prefers turbid (eutrophic) and cool systems, this habitat preference may further constrain their spread.

Given the potential for ecological impacts, ongoing concern regarding the spread of Ruffe is warranted. Notably, management actions to prevent the spread of Ruffe are critical because Ruffe matures rapidly and has high fecundity, and thus can quickly establish a population. Upon establishment, Ruffe populations can increase rapidly and exceed the local carrying capacity, but then subsequently decline (Ruffe Task Force 1992, Peterson et al. 2011). Thus, the ecological impact of Ruffe establishment may be diminished over time. However, in an introduced water body, native predators may initially be reluctant to prey on Ruffe (Mayo *et al.* 1998). Also, Ruffe is highly competitive in low-light conditions and has the potential to alter population dynamics of prey (benthic invertebrates and zooplankton prey), competitors (forage fish), and fish predators (including through egg-consumption). One particular area of concern are isolated, inland lakes, exemplified by invasions in Western Europe and the UK, including Lake Constance, Germany, Austria, and Switzerland (Matthey 1966; Winfield et al. 1998), Loch Lomond, Scotland (Maitland and East 1989; Adams 1991), Lake Bassinthwaite, England (Winfield *et al.* 2004), and Lake Mildevatn, Norway (Kalas 1995). These lakes possessed environmental conditions suitable for Ruffe, and because they are closed systems with relatively low biodiversity, Ruffe has had a substantial effect on the benthic invertebrate, zooplankton, and prey fish communities.

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# Chapter 2: Population change of an invasive fish, Ruffe, thirty years post-introduction: boom or bust?

## Abstract

Invasive species often show a period of rapid initial increase (boom) followed by a population crash (bust) before settling into a relatively stable equilibrium population size. The purpose of this study was to determine trends in abundance of Ruffe at two locations and how they relate to the typical “boom-bust” population invasion patterns. Further, to identify potential interactions with native fishes, I compared the Ruffe catch per unit effort (CPUE) time-series to the corresponding time-series for common prey and predatory fishes in both the St. Louis River, MN/ WI, USA, and Chequamegon Bay, WI, USA, from 1993-2015. These systems were invaded by Ruffe at different time periods, both have similar fish communities, and CPUE data has been collected in both locations since Ruffe invaded. I found that Ruffe populations in the two systems are at different stages of invasion. In the St. Louis River, overall the population decreased from 1993-2015; the population increased from the initial invasion up to 1995 and has been declining for two decades (1996-2015). In Chequamegon Bay, the overall population is increasing, but is oscillating every 5-7 years. I conclude that Ruffe populations in both systems partially conform to the typical “boom-bust” patterns seen with other invasive fish species.

## Introduction

Invasive species threaten biodiversity globally, alter the ecological function of invaded ecosystems, and cause extensive economic damage (D'Antonio *et al.* 2001; Arim *et al.* 2006). Invasive species can impact native species through both direct interactions, such as competition, predation, mutualism, herbivory, and parasitism, and indirect interactions, such as habitat alteration, cascading trophic interactions, and apparent predation (Sakai *et al.* 2001). However, the severity of a particular invasion depends on the invasive species' competitive ability and how the species interacts with its new environment (Blossey and Kamil 1996).

The Laurentian Great Lakes are among the most invaded ecosystems in North America; they have been subject to biological invasions since at least the early 1800s, following settlement by Europeans (Mills *et al.* 1994). They are vulnerable to invasion because of high shipping traffic, particularly transoceanic cargo ships, ballast water discharge, and a history of pollution and ecological disturbance (Stepien *et al.* 2005). The economic and ecological costs of some invasive species have been immense (Pimentel *et al.* 2005).

The timeline of population growth and spread of an introduced species can be conceptualized as a series of invasion stages (Sakai *et al.* 2001; Colautti and MacIsaac 2004; Simberloff and Gibbons 2004). In stage 0, propagules of the introduced species are in the donor region; in stage 1, the introduced species is transported outside of its current range; in stage 2, individuals are released and introduced into a new region. In stage 3, the species becomes established, distributed in a small area and is numerically rare. In stage 4, the species' population is either spatially widespread but numerically rare, or localized but abundant. Finally, in stage 5, organisms are widespread and dominant (Colautti and MacIsaac 2004). Often with invasive species, there is a lag time between stage 3 and stage 4 or 5, after which there is exponential growth (Sakai *et al.* 2001). Another common feature seen in invasive populations is a “boom-bust” cycle. A population crash (“bust”) is often seen following the exponential growth (“boom”) in some invasive populations (Simberloff and Gibbons 2004; Cooling and Hoffmann 2015). Lags and “boom-bust” cycles are thought to exist because of adaptive evolution as a part of the colonization and establishment process. During colonization there may be genetic constraints on the probability of successful invasion (Sakai *et al.* 2001). Once the population overcomes these genetic constraints, it has the ability to “boom” or grow very rapidly. This boom can result in dense local population or rapid range expansion (Sakai *et al.* 2001). At some point, these populations crash (Simberloff and Gibbons 2004). Studying these post-boom population declines may help us to understand the timeline and pattern of introductions.

The focus of this study is Ruffe (*Gymnocephalus cernua*: Percidae), a small-bodied, demersal fish native to Europe and Asia; specifically, its native range is from parts of Siberia and Russia to northeast France and southern England (Berg 1965; Kalas 1995; Rösch *et al.* 1996; Winfield *et al.* 1998b; Ogle 1998; Dawson *et al.* 2006; Gutsch and Hoffman 2016). Ruffe is invasive in parts of both Europe and North America (Gutsch and Hoffman 2016). In North America, Ruffe was first introduced to the Lake Superior basin, presumably via ballast water from transoceanic commercial vessels (Pratt *et al.* 1992a). Ruffe was first discovered in the Duluth-Superior harbor (the Great Lakes largest commercial shipping port), which is located at the far western end of Lake Superior in the St. Louis River (SLR), in 1986 (Bowen and Keppner 2013). It subsequently spread eastward, likely by dispersal along the southern shore of Lake Superior (MN Sea Grant 2016), and by 1993, Ruffe established in Chequamegon Bay (CB), a large embayment about 110 km east of Duluth-Superior harbor (MN Sea Grant 2016). Ruffe continued to spread along the south shore of Lake Superior, eventually reaching Whitefish Bay in the far east end of the lake in 2006. It was found in Lake Huron in 1995, and Lake Michigan in 2002 (Bowen and Keppner 2013). By 1998, Ruffe inhabited 16 tributaries on the south shore of the western arm of Lake Superior (Mayo *et al.* 1998). Bronte et al. (1998) concluded the increase in Ruffe was due to recruitment of large year classes in 1990, 1994, and 1995.

Ruffe can potentially reduce native fish diversity and abundance (Gutsch and Hoffman 2016). After it was first detected in Lake Superior, there was substantial concern that Ruffe would compete with native species (Ruffe Task Force 1992; Selgeby 1994; Evrard *et al.* 1998; Czypinski *et al.* 2002). During the early 1990s, when the Ruffe population size was rapidly increasing in the St. Louis River, the abundance of many native species were declining, including Yellow Perch (*Perca flavescens*), Emerald Shiner (*Notropis atherinoides*), Spottail Shiner (*Notropis hudsonius*), Trout Perch (*Percopsis omiscomaycus*), and Johnny Darter (*Etheostoma nigrum*) (Selgeby and Edwards 1993; Bronte *et al.* 1998). At that time, Mayo et al. (1998) conducted a diet study of native piscivores including Northern Pike (*Esox lucius*), Walleye (*Sander vitreus*), Smallmouth Bass (*Micropterus dolomieu*), large Brown Bullhead (*Ameiurus nebulosu*), and large Yellow Perch, and found that Northern Pike were the only predator that consumed a substantial biomass of Ruffe, but all predators consumed some Ruffe (Mayo *et al.* 1998). Notably, comparisons between the St. Louis River and Chequamegon Bay were useful to diagnose the effects of Ruffe by examining common trends in fish abundance; based on a set of comparisons between these two systems, Bronte et al. (1998) concluded that Ruffe was not causing declines in native fishes.

Despite an intense, regional focus on Ruffe during this time period, and the subsequent spread of this fish to other US Great Lakes, we know little of how its abundance has since changed in either the St. Louis River (SLR) or Chequamegon Bay (CB) over the past two decades. The objective of this study was to determine whether Ruffe populations in SLR and CB conform to typical invasive species boom-bust patterns. The boom-bust pattern is defined by an exponential increase followed by an exponential decrease to some equilibrium. Further, as an exploratory analysis, I compared Ruffe abundance to potential predator and competitor abundance through time to identify species that might have strong interactions with Ruffe in SLR and CB. For this study, my main hypothesis was that Ruffe populations conform to initial exponential growth and subsequent exponential decline (i.e., a boom-bust pattern). I tested the hypothesis separately for populations in the St. Louis River, WI/ MN, USA, and Chequamegon Bay, WI, USA.

# Methods

### *STUDY AREA*

The St. Louis River (SLR) is located in the western arm of Lake Superior (Figure 4, A). Its lower 30 km is classified as a drowned river mouth coastal wetland, also known as a “freshwater estuary,” which extends from Fond du Lac, MN, to the mouth at Lake Superior, and has a surface area of about 44 square km. The Port of Duluth-Superior is located where the river enters Lake Superior, and is afforded protection by a 16 km long barrier beach. The thalweg has a maximum depth of 16 m in the harbor and 8 m at the upper end of the river (Angradi *et al.* 2015). The river is mesotrophic (Bellinger *et al.* 2016), unlike Lake Superior, which is oligotrophic (Bronte *et al.* 1998). The turbidity is generally high with total suspended solids between the harbor, bay, and the river ranging from 10.2-13.0 mg/L (Bellinger *et al.* 2016). Mean dissolved oxygen in June and July is 7.82 mg/L (2.78-10.30 mg/L) (Bellinger *et al.* 2016). The maximum temperature is about 29$^{o}$C (G. Peterson, personal comm.). As of 2014, there were 52 documented fish species in SLR, most of which were cool or cold-water species (Peterson *et al.* 2011; Hoffman *et al.* 2016).

Chequamegon Bay (CB), WI, is located in southwestern Lake Superior (Figure 4, B). The surface area of CB is about 160 square km. It has a maximum depth of 23 meters and a mean depth of 9 meters. The bay is also mesotrophic (Bronte *et al.* 1998). Typically, total suspended solids range from non-detect to 3 or 4 mg/L (R. Lehr and M. Hudson, Northland College, personal comm.). The maximum temperature is 23$^{o}$C, and the average dissolved oxygen concentration is 10.5 mg/L (8.5-14.3 mg/L) between April and August (R. Lehr and M. Hudson, Northland College, personal comm.). It has 53 known fish species, 41 of them in common with SLR as of 2014 (USGS, personal comm.). Chequamegon Bay is a useful location for comparison to the St. Louis River because Ruffe established in CB shortly after the SLR population began to increase rapidly, and because the two systems have a similar fish assemblage, are part of the same drainage, are at the same latitude, and have been compared in previous studies (Bronte *et al.* 1998).

### *COMPETITOR AND PREDATOR SPECIES*

For the St. Louis River, Ruffe and competitor catch data came from bottom trawl surveys conducted by US Fish and Wildlife Service (USFWS), 1854 Treaty Authority, US Environmental Protection Agency (USEPA), and US Geological Survey (USGS) (Table 3); predator capture data were from the Minnesota Department of Natural Resources (MN DNR) gill net survey. For Chequamegon Bay, Ruffe and competitor catch data came from bottom trawl surveys conducted by USFWS and USGS; predator catch data were from a Wisconsin DNR creel survey. Annual data were available for both systems from 1993-2015. During this time period, USFWS, USEPA, and 1854 Treaty Authority all used the same equipment and methods for bottom trawling; however, the methods used by the USGS varied slightly (Table 3).

I standardized trawl catch data for area swept catch per unit effort (CPUE; number of fish/ hectare) based on trawl width, tow duration, and vessel speed, assuming that the vessel type and speed did not affect trawl performance (Table 3). I calculated CPUE using the following equations:

$$\frac{\#\ of\ fish\ caught}{Tow\ time\ (min)} \times \frac{60\ min}{hour} = \#\ of\ fish/hour$$

Eq. 1

$$\frac{\#\ of\ fish/hour}{\#\ of\ hectares/hour} = \#\ of\ fish/hectare$$

Eq. 2

A notable concern is that different vessels were used for different time stanzas, that bottom trawl CPUE is density-dependent (i.e., the number of fish ahead of the bottom trawl affects catchability) (Godø *et al.* 1999) and influenced by environmental factors such as water clarity (Buijse *et al.* 1992) and substrate, and the St. Louis River in particular has undergone substantial ecological change over the course of the time-series (Bellinger *et al.* 2016). However, species-specific catches within the same system from vessel to vessel are generally consistent with regard to spatio-temporal effects (Benoit and Swain 2003). As such, I present the data throughout with due caution.

For the MN DNR gill net survey (76.2 m length, 1.83 m height, 5- 15.24 m panels with corresponding mesh sizes of 19.05 mm, 25.4 mm, 31.75 mm, 38.1 mm, and 50.8 mm), I calculated CPUE by dividing the mean summed total by the total number of net sets in a given year. I also analyzed gear selectivity to determine if the gill net was catching predator fish that were large enough to consume Ruffe (Figures A-1-4). I determined that the majority of predator fish caught in the gill nets were large enough (>300 mm) to consume adult Ruffe because most fish this size have a gape large enough to consume a fish that is an average of 70-120 mm (Scharf *et al.* 2000).

To calculate creel survey CPUE, I divided the annual sum of fish caught each year reported by anglers by the number of angling hours. This estimate is my least reliable relative measure of fish abundance, but is the only annual measure of game fish for Chequamegon Bay.

### *ANALYSIS*

I estimated average Ruffe CPUE for sampling dates and sites and vessels per year for each system (SLR and CB). I used this same method for all competitors (Round Goby, Trout Perch, Yellow Perch, Spottail Shiner, Emerald Shiner, and Johnny Darter) and predators (Walleye, Northern Pike, Smallmouth Bass, and Muskellunge) of interest, as well. I chose the competitor species because they were the main fish affected by the Ruffe invasion back in the 1980s (Ruffe Task Force 1992); whereas, I chose the predator species because they were found to eat Ruffe (Mayo *et al.* 1998) or are large enough to eat Ruffe. Each species had a column of CPUE data and each row represented an average year of sampling. I standardized the samples to a common level of effort, and the level of effort used was one year. Some species had missing values for several years (Table A-1-4); I used a cubic spline method to impute data for those species (R package CRAN). The cubic spline method achieves a smooth interpolating function by creating a formula in which the first and second derivative are continuous and minimize error (Brumback and Rice 1998; Junninen *et al.* 2004). Columns of fish CPUE with too many (more than three) missing values together were removed from the analysis because the data were insufficient to support imputation. I analyzed SLR and CB data separately. The SLR dataset had 10 species: four predator and six competitor species. The CB dataset had 7 species: two predator species and five competitor species.

To test my hypothesis that Ruffe exhibited exponential growth (“boom”) or decay (“bust”), I used an exponential growth model to estimate r using $N(t)= N(0)e^{rt}$, assuming N(0) and $t_0$ are population size and time at first detection, respectively, where r= relative growth rate, t= time, and N(t)= population after a time t has passed. I fit a linear model to the plot of ln(Ruffe CPUE + 1) vs Year for my dataset from 1993-2015. I conducted this analysis separately for SLR and CB, which allowed for comparisons. In addition, I expanded my analysis to include data from Pratt (1988), Ruffe Task Force (1992), and USGS from 1985-1992 in SLR to determine boom-bust cycles from the beginning of the Ruffe invasion. This addition allowed me to view the entire invasion period of Ruffe in SLR from 1985-2015. This data pre-1993 was not calculated by me, but CPUE was estimated using the same methods as data post-1993, and the data was collected using similar methods, so I considered it comparable. For this second analysis, I fit two linear models to the data: one from 1985-1995 (introduction to the maximum CPUE) and one from 1996-2015 (decline following maximum CPUE).

To test for a monotonic change in competitor or predator CPUE through time, I used the Mann-Kendall (MK) test (Mann 1945; Kendall 1975; Gilbert 1987), using the Kendall package in R (Hirsch *et al.* 1982). To determine which species were correlated with one another and with Ruffe, I analyzed each dataset using a Pearson correlation matrix and used the Pearson r value scale to classify the correlation strength: 0.00-0.19 = “very weak,” 0.20-0.39 = “weak,” 0.40-0.59 = “moderate,” 0.60-0.79 = “strong,” and 0.80-1.00 = “very strong” (Evans 1996).

To determine which species had the strongest statistical effect on Ruffe CPUE, I used univariate generalized linear models (GLMs) with Gaussian distributions for each of the variables in each of the systems (independent variables: competitor or predator species CPUE; dependent variable: Ruffe CPUE). I compared separate univariate models rather than multivariate GLMs due to model assumption violations and variable correlations. I natural log-transformed all catch data (i.e., (ln(CPUE +1)). For SLR, I had 11 models, and for CB, I had 8 models, including each competitor or predator species and intercept only model. I used Akaike Information Criterion (AIC) for model selection, correcting for small sample size ($AIC_c$). All analyses were conducted using R.

I ran an additional preliminary analysis examining fish lengths of Yellow Perch, Trout Perch, and Ruffe between the two systems to try to determine a condition factor between the populations. In SLR, I examined years 1989, 1995, and 2016. In CB, I examined 1998, 2011, and 2015. I chose these years because the first year was just after the Ruffe invasion, the second year was the peak of the Ruffe invasion so far, and the third year was the most recent data I had in that system. SLR and CB Ruffe, Yellow Perch, and Trout Perch lengths were collected from USGS and USFWS catch data. I averaged all recorded lengths for the three species for the specified dates. To compare ratios of total Yellow Perch, Trout Perch, and Ruffe in each system to one another, I multiplied average length by CPUE as a surrogate for biomass.

## Results

Ruffe in SLR exponentially declined significantly from 1993-2015 (ln(Ruffe CPUE+1) = -0.113(Year) + 231.942, adj. $R^2$ = 0.59, p<0.001) (Figure 5). Incorporating the additional data for SLR, I found that the Ruffe population significantly increased from 1985-1995 (ln(Ruffe CPUE+1) = 0.634(Year) – 1256, adj. $R^2$ = 0.88, p<0.001) in the ten years immediately following its first detection, and declined from 1996-2015 (ln(Ruffe CPUE+1) = -0.147(Year) + 301.227, adj. R2 = 0.725, p<0.001) (Pratt 1988; Ruffe Task Force 1992, USGS, personal comm.) (Figure 6 and 7). In contrast, the Ruffe population in CB has undergone a significant exponential increase (ln(Ruffe CPUE+1) = 0.196(Year) -390.398, adj. $R^2$ = 0.50, p<0.001; Figure 8) since its first detection, but with apparent oscillations (Figure 6).

Based on the Mann-Kendall test, Ruffe CPUE in SLR has decreased overall from 1993-2015 (p<0.001, tau=0.66). In SLR, the CPUE of several fishes did change significantly over time. The CPUE of Northern Pike (p=0.0013, tau=0.488) and Yellow Perch (p=0.02, tau=0.352) both decreased, whereas the CPUE of Trout Perch (p<0.001, tau= 0.589), Round Goby (p<0.001, tau=0.544), and Emerald Shiner (p=0.035, tau=0.32) increased over time. Spottail Shiner, Johnny Darter, Muskellunge, Smallmouth Bass, and Walleye CPUE did not change over time (Figures 9 and 10). Ruffe CPUE in CB increased overall since its introduction from 1993-2015 ($p<0.001$, tau=0.561). According to the MK test, no other fish CPUE changed significantly over time in CB (Figure 11 and 12), recognizing that the predator CPUE data are from a creel survey. However, based on the plots in CB, Yellow Perch, Emerald Shiners, Spottail Shiners, and Johnny Darters all showed similar trends. They had a relatively high CPUE between 1993-2000, then decreased from 2001-2008, and increased again from 2009-2015 (Figure 11).

Based on the Pearson correlation matrices, I found that Ruffe and Trout Perch CPUE were strongly, negatively correlated and that Ruffe and Yellow Perch CPUE were strongly, positively correlated in SLR (Table 4). Ruffe CPUE was also moderately, negatively correlated with Emerald Shiner, and moderately, positively correlated with Northern Pike. Further, Northern Pike and Yellow Perch CPUE were strongly, positively correlated and Walleye and Spottail Shiner CPUE were also strongly, positively correlated (Table 4).

In CB, Ruffe CPUE had very weak to weak (Evans 1996) correlations with all other fish CPUE. Among the other fishes, Spottail Shiner and Emerald Shiner CPUE were very strongly, positively correlated (Table 5). Spottail Shiner CPUE was also strongly, positively correlated with Johnny Darter and Yellow Perch CPUE. Yellow Perch and Johnny Darter CPUE, too, were strongly, positively correlated. Emerald Shiner CPUE was moderately, positively correlated with Johnny Darter and Yellow Perch CPUE (Table 5).

Based on the generalized linear models, four univariate models make up 99% of the model weight for SLR (Table 6). As Ruffe CPUE decreased, so did

Yellow Perch and Northern Pike CPUE, while Trout Perch and Emerald Shiner CPUEs increased (Figure 13). The best model was the Yellow Perch model (62% of the $AIC_c$ weight), followed by the Trout Perch model (24%), the Northern Pike model (7%), and the Emerald Shiner model (5%). Three of the four top models were competitors, and the proportion of model weight in the top 99% associated with competitors was about 93% (Table 6). All of the variables in the top four models were significant (their 95% confidence limits did not encompass zero); parameter estimates for Yellow Perch and Northern Pike were both positive, whereas parameter estimates for Trout Perch and Emerald Shiner were both negative (Table 7). Trout Perch and Emerald Shiner have been the most abundant fish in SLR recently, with the highest abundances of all time in 2013.

For CB, the generalized linear model was inconclusive. All of the models were within two $AIC_c$ points of each other (Table 8). The two models with the most weight were the null model (intercept only) and the Northern Pike model, each of which made up 18% of the model weight (Table 8). That is, none of the species were significantly related to the increase of Ruffe in CB. Yellow Perch is the most abundant fish in CB, with the highest recorded abundances in 1998 and 2013.

In SLR, since the Ruffe invasion, average Yellow Perch lengths have decreased by about 20 mm and have not changed in CB (Figure 14). In both systems, Trout Perch lengths have not changed. In SLR, Ruffe lengths have stayed approximately the same and in CB they have increased by about 30 mm (Figure 14). The carrying capacity of the native fishes (Yellow Perch and Trout

Perch) in SLR seems to be about 4 times that of the carrying capacity in CB (Figure 15).

## Discussion

I found that Ruffe CPUE in SLR significantly decreased overall, and Ruffe CPUE significantly increased from 1985-1995 and subsequently decreased from 1996-2015. In SLR, the Ruffe population conforms to the typical invasion theory "boom-bust" model and is currently in the "bust" phase. I also found that Ruffe CPUE in SLR is related to the CPUE of numerous potential competitors. Ruffe CPUE in CB significantly increased from 1993-2015. In CB, the Ruffe population partially conforms to the "boom-bust" model and is in the "boom" phase. CPUE in CB is not related to the CPUE of any potential competitor or predator species examined. Here, I discuss the CPUE patterns of Ruffe in SLR and CB and whether they conform to a boom-bust cycle, the weight of evidence for interactions with respect to both potential competitors and predators, and different factors responsible for fish population dynamics in SLR and CB.

### *CPUE PATTERNS OF RUFFE AND INVASION THEORY*

By my analysis, the Ruffe population in the St. Louis River has been declining for two decades and was in the "bust" phase of the invasion at the time of the study. In 1995, the Ruffe CPUE reached a maximum, possibly indicating the population had reached or exceeded its carrying capacity, and then slowly declined. In the initial analysis, there was a modest rate of decline in CPUE from 1993-2015 (r=-0.113). In the additional analysis, I found a similarly modest rate of decline from 1996-2015 (r=-0.147), which contrasted strongly with a much greater rate of increase from 1985-1995 (r=0.634; Figure 6). Overall, the Ruffe population in SLR is at or past invasion stage 5 (Colautti and MacIsaac 2004) because it has been established for at least 30 years. Being in this stage should mean Ruffe is everywhere and regularly found within the ecosystem, not just captured in the original “hotspot” areas.

I found exponential growth in CB after 1993 but no evidence of recent long-term decline. Overall, there was a significant increase in Ruffe CPUE in CB, but the rate of increase is relatively small (r=0.196), much smaller than the rate of increase in SLR and similar to the rate of decline in SLR. It does not completely match the typical “boom” of most invasive species in a new environment, which usually has a very high rate of change after a lag period (Ruffe Task Force 1992; Simberloff and Gibbons 2004; Branstrator *et al.* 2017). That is, in SLR, in the first ten years of being established, the population boomed to over 1808 fish/ hectare; whereas in CB, in the first ten years of establishment, the captured population size was only about 7 fish/ hectare (Figure 6). The Ruffe population in CB either was slow to establish since it was first discovered in 1993, or agencies had difficulties catching it, because there was nearly 0 CPUE for the first five years of its invasion. This lag time in population growth is similar to the lag time that was described by Sakai et al. (2001) and Branstrator et al. (2017) commonly found in aquatic invasive populations between stage 3 and stage 4 or 5. In CB, Ruffe is at an earlier stage of invasion than in SLR and is in the “boom” phase of the invasion, indicated by the exponential population increase since 1993. In CB, the invasion stage is likely at a 4 (Colautti and MacIsaac 2004). The distinct pattern observed in CB CPUE is that every 5-7 years the population oscillates, underlying the overall increase in abundance (Figure 8). The cause of these oscillations are not known, but they could be due to an unreliable food source or inter- or intraspecific competition (Ruffe Task Force 1992).

Comparing these two systems, which include the two largest populations of Ruffe in the North America (Gutsch and Hoffman 2016), Ruffe partially conforms to the typical “boom-bust” invasion population patterns, but they are at different stages (Figure 6) (Simberloff and Gibbons 2004; Cooling and Hoffmann 2015). The “bust” in SLR has been slow (21 years), and the “boom” in CB is gradual and is in the 23rd year of the invasion. The “boom” in CB (r=0.196) is very different from the “boom” in SLR (r=0.634) after the Ruffe’s first detection (Figure 6). However, it is possible that Ruffe was present long before its first detection in SLR.

### *EXPLORATORY ANALYSIS OF COMPETITORS AND PREDATORS*

The strong correlations between Ruffe CPUE and competitor CPUE in SLR suggests that competition for food, space, or other resources may be contributing to the Ruffe decline. As Ruffe populations decrease, Trout Perch and Emerald Shiner may be outcompeting Ruffe and Yellow Perch. However, based on many sources, I could not find a mechanistic explanation for the Ruffe decline. The two lines of evidence I examine are spawning habitats (Beard and Carline 1991) and diet (Chapman 1966) because these factors are the most common causes of competition that may cause a population to decline.

Ruffe and Trout Perch may compete for spawning habitat. Trout Perch and Ruffe spawn at similar times (starting in early spring and continuing through the summer) (Muncy 1962; Magnuson and Smith 1963) and depths (less than 1.524 m) (Muncy 1962; Magnuson and Smith 1963; Gutsch and Hoffman 2016). Trout Perch spawns on silt or boulder bottoms at 4-10$^{o}$C (Lawler 1954). Ruffe spawns on almost any substrate at 5-18$^{o}$C (Gutsch and Hoffman 2016). Yellow Perch, however, spawns in submerged vegetation and brush at 36-44$^{o}$C (Muncy 1962). This is a feasible hypothesis, but more evidence is needed to claim that this is the reason for the Ruffe decline.

According to Ogle et al. (1995), Ruffe and Yellow Perch have similar diets, as do Trout Perch (Wells 1980), suggesting competition for food resources. Adult Ruffe eats midges, macrobenthos, burrowing mayflies, and caddisflies (Ogle *et al.* 1995). Adult Yellow Perch eats amphipods, fish eggs, Mysis (which are not in the river), and crayfish (Wells 1980), as well as small fish. Muncy (1962) found that Yellow Perch eats small crustaceans and insects, especially chironomids, one of the Ruffe's preferred food items (Gutsch and Hoffman 2016). Trout Perch eats amphipods, immature midges, and zooplankton (Wells 1980). Thus, the three species could compete for food resources owing to diet overlap. However, Hoffman et al. (2010) found that Ruffe and Yellow Perch in SLR are more isotopically similar to each other than Trout Perch, not suggesting competition for food resources. They found that Trout Perch had higher $\delta^{15}N$ values than Yellow Perch or Ruffe, suggesting they are feeding at different trophic levels. Moreover, Yellow Perch and Trout Perch are typically inactive at night (except during spawning) and feed during the day (Muncy 1962; Magnuson and Smith 1963); whereas, Ruffe is often most active at night, feeding in the shallow areas in the darkness (Ogle *et al.* 1995). Also, it uses different habitats (Peterson *et al.* 2011). Ruffe is primarily in the thalweg (highest CPUE in trawls), whereas Yellow Perch tends to be in littoral habitat (highest CPUE in fyke nets and electrofishing) (Peterson *et al.* 2011). So, while it is possible these fishes are depleting the same resources, the competition may not overlap temporally or spatially. Based on stable isotope evidence and likely diel habitat partitioning, it is unlikely that there is diet competition with Ruffe and Yellow Perch.

## *FACTORS THAT AFFECT FISH POPULATION DYNAMICS*

My findings suggest a recent divergence between SLR and CB. Bronte et al. (1998) found similar trends between SLR and CB among many fish species in the years 1989-1996; whereas I found Ruffe CPUE trends were quite different between the two systems. I found both positive and negative associations between Ruffe and other fishes in SLR, but no significant associations between Ruffe and other fishes in CB. The CPUE data indicate that the population dynamics of Ruffe, and possibly other fishes in CB and SLR are highly variable and are likely not influenced by the same variables. The data included in this study (CPUE of competitor and predator fishes) was not able to account for the observed oscillations of Ruffe CPUE in CB. There was possibly a divergence of these two systems since Bronte et al. (1998), and this topic is worthy of further investigation.

In CB, I found no indication that other fish populations have declined due to the invasion of Ruffe. Because CB is at an earlier invasion stage, it is possible the ecological effects of Ruffe has not yet been realized (i.e., Ruffe densities are too low or catch efficiency is too low); therefore, interactions with other fishes would not be measurable yet. The overall population size of Ruffe in CB since its introduction has been substantially less than the population size of Ruffe in SLR in the corresponding year of invasion (Figure 6). It is possible that CB is not as suitable of an environment for Ruffe. If that is the case, there may not be significant ecological changes to CB due to the invasion of Ruffe. Future research should examine and map Ruffe range expansion and contraction, which could provide new insights regarding changes from “boom” to “bust” and time-dependent patterns of invasion of a particularly prolific invasive species. Understanding these boom-bust cycles in invasive species is important to recognize for formulation of management decisions relating to invasive species control.

# Chapter 3: Using stable isotopes to characterize Ruffe (*Gymnocephalus cernua*) trophic pathways and movements in the St. Louis River and Chequamegon Bay, USA

## Abstract

Food webs have been altered by invasive species in ecosystems throughout the globe. Stable isotope ratios are commonly used to trace trophic pathways and study complex landscape inputs, and thereby understand how food webs are structured. The goals of this study were to identify energy sources contributing to Ruffe production and use habitat-specific stable isotope ratios to study life stage-specific movements. I measured Ruffe $\delta^{13}C$ and $\delta^{15}N$ values in the St. Louis River and Chequamegon Bay and estimated the diet contributions from various habitat-specific organic matter (OM) sources, including Lake Superior benthic periphyton, coastal wetland benthic periphyton, riverine matter derived from a mix of phytoplankton and terrestrial OM, and river sediment methane using a mass-balance mixing model. Further, I identified size-based or stage-based movements between Lake Superior and inshore habitats based on Ruffe $\delta^{13}C$ and $\delta^{15}N$ values. I found significant differences in $\delta^{13}C$ and $\delta^{15}N$ values between Ruffe captured in Lake Superior and those captured in the St. Louis River, but not among locations within the river. I found size-based differences, as well; medium-sized fish, 65-85 mm standard length (SL), had $\delta^{13}C_{lipid\ corrected}$ values of about -40‰ to -16‰, a spread of 24‰. However, small fish (<65 mm SL) had $\delta^{13}C_{lipid\ corrected}$ values of -50‰ to -24‰, shifted -10‰ with a spread of 26‰; and large fish (80-148 mm SL) had $\delta^{13}C_{lipid\ corrected}$ values of -54‰ to -14‰, which is a spread of 40‰, spanning the range of values measured in this study. Extremely depleted $^{13}C$ values (<-36‰ $\delta^{13}C$) indicate that some fish captured within coastal wetlands were feeding in a methane-based trophic pathway. The high $\delta^{13}C$ values of both small and large Ruffe indicate these fish were both swimming and

feeding in Lake Superior; the higher values of medium size Ruffe indicate coastal wetland dependence during the spawning period. The broad range in $\delta^{13}C$ values of large Ruffe indicate routine occupancy of both lake and wetland habitats; 59.7% of individuals were predominantly feeding in a wetland-dominated trophic pathway, whereas 40.3% were feeding in a lake-dominated trophic pathway. This observation is the first of wetland fish obtaining substantial energy from a methane-based food web, as well as the first observation of distinct, size-based diet shifts and movements among coastal habitats in Ruffe. This indicates Ruffe has the ability to occupy a novel trophic niche within coastal wetlands and is an obligate user of wetland habitat during spawning but otherwise facultative user of lake and wetland habitat.

## Introduction

Great Lakes coastal wetlands support many ecological, economic, and cultural ecosystem services (Sierszen *et al.* 2012). Coastal wetlands provide plant and animal habitat, hydrologic retention, nutrient cycling, shoreline protection, and sediment trapping, providing an important role in the Great Lakes ecosystem. They support a great biodiversity that drives the Great Lakes food web with up to one-third of the primary production originating in coastal wetlands (Brazner *et al.* 2000). Characterizing the food web of a coastal wetland is challenging because the organic matter supporting consumers comes from a variety of sources within the ecosystem (Hoffman *et al.* 2015). The landscape mosaic of a Great Lakes coastal wetland generally is composed of three ecosystems: terrestrial, coastal wetland (river and wetland), and lake. Within the aquatic ecosystems are littoral, benthic, and pelagic habitats, each supported by distinct energy sources.

Positioned between the land and the lake, coastal wetland food webs are fueled both by high photosynthetic production (i.e., autochthonous energy sources) and by inputs of energy and nutrients from these adjacent ecosystems (i.e., allochthonous inputs; (Hoffman *et al.* 2010)). Another potential source of energy to the food web is chemosynthetic production of methane within river sediments, which can contribute to higher trophic levels when primary consumers graze on a mix of particles and methane-oxidizing bacteria (MOB) in stratified sediments (Bastviken *et al.* 2004; Jones and Grey 2011). At the base of most food webs is phytoplankton. The autochthonous carbon from phytoplankton can be limited by nutrient availability, light, resident time, phytoplankton growth rate, and dissolved $CO_2$ (DIC) concentration and may be used by organisms like zooplankton and benthic macroinvertebrates (O'Leary 1981; Farquhar *et al.* 1982; Hoffman and Bronk 2006; Hoffman *et al.* 2010). Primary consumers, including zooplankton, benthic invertebrates, and fish, may also consume allochthonous organic matter, such as particulate organic matter derived from riparian or upland vegetation, which can potentially enhance overall productivity (Wallace *et al.* 1997; Cole and Caraco 2001; Hoffman *et al.* 2008, 2010). These allochthonous carbon and energy subsidies can supplement autochthonous primary production in both pelagic and benthic food webs (Jansson *et al.* 2007; Reynolds 2008; Jones and Grey 2011; Hoffman *et al.* 2015).

These same allochthonous carbon inputs can be processed by heterotrophic bacteria under oxic conditions, providing biomass for zooplankton grazers (Jones and Grey 2011). However, in anoxic conditions, which are common in the hypolimnion of stratified lakes and in aquatic sediments, carbon may originate by different microbial metabolic pathways, especially methanogenesis. Lake sediments are known for their high methane production and their significant contribution to the global methane budget (Bastviken *et al.* 2004). Some of this methane is available to methane-oxidizing bacteria (MOB), which oxidize it once it reaches an oxygenated sediment layer or water column (Rudd and Taylor 1980; Bastviken *et al.* 2003, 2004; Whalen 2005; Juutinen *et al.* 2009; Jones and Grey 2011). Not only does methane get added to the biogeochemistry of the lake, but it also becomes an important source of carbon and energy in freshwater trophic pathways, where it is readily available to benthic invertebrates (Bastviken *et al.* 2003; Jones and Grey 2011).

Across the globe, aquatic food webs have been greatly impacted by invasive species (Gurevitch and Padilla 2004). These food web impacts can have detrimental ecosystem-level effects, including modified habitat coupling, nutrient cycling, and ecosystem resilience (Eby *et al.* 2006; Britton *et al.* 2010; Pilger *et al.* 2010; Walsworth *et al.* 2013). Invasive species can have strong impacts on aquatic food webs owing to the competitive advantage invasive fish have over native fish (Cox and Lima 2006; Walsworth *et al.* 2013). Although it is challenging to detect or predict the impacts of invasive species on aquatic food webs, some of these interactions are still measureable (Polis 1991; Lodge 1993; Polis and Strong 1996). This is an even greater challenge at the landscape-scale because it requires consideration of inputs from multiple aquatic habitats and also adjacent ecosystems (Hoffman *et al.* 2015).

Stable isotopes of light elements such as hydrogen, carbon, nitrogen and sulfur are useful for tracing both autochthonous and allochthonous trophic pathways in coastal food webs (Hoffman 2016). For example, because there is little isotopic fractionation of carbon between a consumer and its diet (about 0.4‰) (Vander Zanden and Rasmussen 2001), carbon stable isotopes can be used to trace consumer diets, identify predator-prey relationships, and elucidate trophic pathways (i.e., the connection between a carbon source such as phytoplankton and a high-level consumer). In particular, where organic matter sources that are potentially contributing to a coastal food web have distinct carbon stable isotope ratios (i.e., $\delta^{13}C$ values), aquatic food webs can be reconstructed and major trophic pathways identified (Hecky and Hesslein 1995; Vander Zanden and Rasmussen 2001). Further, nitrogen stable isotope ratios can be used to estimate consumer trophic position because consumers exhibit a consistent and measurable enrichment in $^{15}N$ with each successive trophic level (Cabana and Rasmussen 1996; Vander Zanden and Rasmussen 1999, 2001). Typically, consumer $\delta^{15}N$ values are enriched by 3.4‰ on average above that of their prey (Vander Zanden and Rasmussen 2001; McCutchan *et al.* 2003). If both carbon and nitrogen stable isotope ratios are measured, trophic position, omnivory, energy sources and flows, and food chain length can be determined (Vander Zanden and Rasmussen 2001). Carbon and nitrogen stable isotopes have been shown to be particularly helpful in studying Great Lakes coastal wetland food webs because many of the available organic matter sources (e.g., phytoplankton, epiphytic periphyton, emergent vegetation, benthic periphyton, etc.) have distinct isotopic ratios (Keough *et al.* 1996; Hoffman *et al.* 2015).

I studied the trophic ecology of Ruffe, an invasive fish, in Lake Superior coastal wetlands. Ruffe is native to Europe and Asia and was accidentally introduced to the US through ballast water discharge (Simon and Vondruska 1991; Pratt *et al.* 1992b). Ruffe is a small, demersal percid that consumes benthic invertebrates and has been found to compete with other small forage fishes native to Lake Superior (Ruffe Task Force 1992; Evrard *et al.* 1998; Czypinski *et al.* 2002). In 1986, Ruffe was first discovered in the St. Louis River (SLR), a drowned river mouth coastal wetland in far western Lake Superior, and subsequently spread across the upper Laurentian Great Lakes (Bowen and Keppner 2013). Ruffe inhabits coastal wetlands throughout the year, but also inhabits Lake Superior waters up to 205 m depth (Gutsch and Hoffman 2016). The effects of Ruffe on Lake Superior coastal wetland food webs were studied in the mid-1990s during a period when Ruffe had become relatively abundant (Czypinski *et al.* 2002; Bowen and Keppner 2013) but not since. Over the past twenty years, these wetlands have undergone substantial change with respect to fish assemblages and environmental conditions (Angradi *et al.* 2015; Bellinger *et al.* 2016). My objectives for this study were to identify trophic pathways between basal energy sources and Ruffe using carbon and nitrogen stable isotope ratios (i.e., $\delta^{13}C$ and $\delta^{15}N$ values) and to use habitat-specific stable isotope ratios to trace movements of Ruffe between coastal wetlands and Lake Superior. First I measured $\delta^{13}C$ and $\delta^{15}N$ values in Ruffe in two large, coastal ecosystems in Lake Superior – St. Louis River and Chequamegon Bay. I used dual-isotope mixing models to estimate the contribution of both photosynthetic and chemosynthetic carbon sources to the food web. The photosynthetic sources included coastal wetland benthic periphyton, Lake Superior benthic periphyton, and riverine organic matter (itself a mix of freshwater phytoplankton and terrestrial-derived organic matter). The chemosynthetic source was methane from river sediments. I further identified movements of Ruffe based on mis-matches between where the individual fish was captured (i.e., Lake Superior or coastal wetland) and the fish's trophic pathway based on its $\delta^{13}C$ and $\delta^{15}N$ values.

## Methods

### *STUDY SITE*

In this study, I examined coastal wetland and lake ecosystems and benthic habitats in the landscape mosaic. My primary study sites were two Great Lakes coastal systems: St. Louis River, MN and WI, a drowned river mouth coastal wetland located in the western arm of Lake Superior, and Chequamegon Bay, WI, a large coastal embayment located in the southwestern part of Lake Superior (Figure 16). Both areas are biogeochemical mixing zones and are suitable for stable isotope food web studies because the variety of organic matter source inputs (i.e., Lake Superior phytoplankton or benthic periphyton, coastal wetland phytoplankton or periphyton, coastal wetland vegetation, terrestrial-derived organic matter) have distinct $\delta^{13}C$ and $\delta^{15}N$ values (Hoffman *et al.* 2015). Coastal wetlands in the Great Lakes are good examples of "transition zones," where one geochemically distinct water source flows into another, even though all the water is freshwater (as opposed to a marine estuary) (Hoffman *et al.* 2010). These geochemical transition zones are important for conducting stable isotope studies because they provide the basis for food webs along the transition zones to have distinct isotopic compositions owing to isotopic mixing. The St. Louis River is 288 km long, and the watershed has an area of 9,412 $km^2$ (Hoffman *et al.* 2010). The estuary is about 50 $km^2$ and lies between Minnesota and Wisconsin (Angradi *et al.* 2015). Water height varies daily by about 13 cm due to weak semi-diurnal tides and periodic seiche flows of about 8 hour duration (Trebitz 2006). There are several ecologically distinct regions within the St. Louis

River, including two turbid, clay-influenced bays (Allouez Bay, Pokegama Bay), two large lake influenced bays (Superior Bay, St. Louis Bay), a large river-influenced bay (Spirit Lake) and an upper section that, although bi-directional in flow, has a confined channel and for which the water chemistry is not influenced by lake exchanges (Figure 16). Water clarity is relatively low throughout the river owing to both high dissolved organic carbon concentrations and occasionally high suspended solids concentrations (Bellinger *et al.* 2016). The average depth is 3.0 m (maximum depth 16 m; (Angradi *et al.* 2015; Bellinger *et al.* 2016)).

Chequamegon Bay has a surface area of about 160 $km^2$. Water quality in Chequamegon Bay is much more lake-influenced than in the St. Louis River; influence of tributary waters is largely limited to the south end, at the mouth of Fish Creek, which is the largest tributary to Chequamegon Bay (Hoffman et al. 2012). The mean depth is about 9 m (maximum 23 m). Water clarity throughout Chequamegon Bay is generally higher than in the St. Louis River.

### ***FISH COLLECTIONS***

Fish were collected in the summer and fall of 2014, winter of 2014-2015, spring of 2015, and summer of 2015 using a mix of approaches, including by otter trawl, fyke net, or anglers ice fishing (Table 9, Figure 17). Once collected, Ruffe were placed in a clean, plastic bag, and then stored on ice to be transported back to the US EPA Mid-Continent Ecology Division, Duluth, MN, laboratory where they were frozen at -20° C until they were processed.

### ***LABORATORY METHODS***

Ruffe were thawed, measured (standard, fork, and total length ±1 mm), and weighed (± 0.1 g wet weight). Using a sterilized scalpel, I obtained a muscle sample from the dorsal side of each fish and removed the skin from the tissue sample. I rinsed the sample thoroughly with DI water, dried the tissue at 45ºC for 24 hours, and ground the tissue into a powder. I used a Costec 4010 EA and Therma Delta Plus XP isotope ratio mass spectrometer to analyze the fish tissue (US EPA Mid-Continent Ecology Division, Duluth, MN). Stable isotope ratios are reported in δ notation, $\delta X{:}\delta X = (R_{sample}/R_{standard} - 1) \times 10^3$, where X is the C or N stable isotope, R is the ratio of heavy to light stable isotopes, and Pee Dee Belemnite and air are the standards for $\delta^{13}C$ and $\delta^{15}N$, respectively. I normalized $\delta^{13}C$ value for lipid content using an arithmetic mass balance correction based on bulk C:N ($C{:}N_{bulk}$) values, with $C{:}N_{lipid\ free}$ of 3.5 (SD±0.3) and lipid isotopic discrimination of -6.5‰ (SD±0.4‰; (Hoffman *et al.* 2015)).

### ***ANALYTICAL METHODS***

To test whether there were significant differences in either $\delta^{13}C_{lipid\ corrected}$ or $\delta^{15}N$ values among capture areas (upper estuary, lower estuary, and Lake Superior), I used a Kruskal-Wallis One-Way Analysis of Variance on Ranks. The Upper estuary area included the St. Louis River and Spirit Lake; the lower estuary area included St. Louis Bay, Superior Bay, and Allouez Bay; and the Lake Superior area included both open waters and embayments (e.g., Cheqaumegon Bay).

I used Ruffe $\delta^{13}C$ and $\delta^{15}N$ data to build a dual isotope, three-source mixing model (Phillips and Gregg 2001) to quantify source contributions from

Lake Superior benthic periphyton, a mix of benthic and pelagic organic matter from lower estuary (the “bentho-pelagic” food web, which is mix of phytoplankton and river sediment that is isotopically difficult to separate; (Hoffman *et al.* 2010)), and a mix of phytoplankton and river sediment from the upper estuary. For the mixing model, the proportional contribution to the fish’s isotopic composition from each source must sum to 1 (Phillips and Gregg 2001). Following Blazer et al. (2014), I selectively fit $\delta^{15}N$ and $\delta^{13}C_{lipid\ corrected}$ values when either or both value fell outside the convex hull of the polygon defined by the $\delta^{13}C$ and $\delta^{15}N$ values of the three sources. The model fit was iterative, adjusting the $\delta^{15}N$ (or $\delta^{13}C$) until all source contributions were between 0 and 1. This is necessary because the model does accommodate variability in source stable isotope ratios. I preferentially adjusted the $\delta^{15}N$ value because small changes in the trophic level have a much larger effect on the fish’s $\delta^{15}N$ value than its $\delta^{13}C$ value. I had to adjust 133 (out of 220 fish) $\delta^{15}N$ values and 21 $\delta^{13}C_{lipid\ corrected}$ values to fit the fish to the model. The mean adjustment was 0.64‰ (range: 0‰ to 5.2‰) for $\delta^{15}N$ values and 1.0‰ (range: 0‰ to 1.9‰) for $\delta^{13}C_{lipid\ corrected}$ values.

I used available fish and invertebrate data to define the sources for the mixing model. These sources were used to represent spatially distinct trophic pathways within Lake Superior and coastal wetlands to facilitate the interpretation of the stable isotope data with respect to both diet and movements. The Lake Superior trophic pathway is based on benthic periphyton, which is an important carbon source in the nearshore of the lake (Keough *et al.* 1996; Sierszen *et al.* 1996). To define the source value, I used Ruffe captured in Lake

Superior that had an isotopic composition consistent with consuming nearshore benthic invertebrates ($\delta^{13}C$ <<-20‰; (Hoffman *et al.* 2015)): mean $\delta^{13}C_{lipid\ corrected\ Lake\ Superior}$ = -16.3‰, SD±2.17‰, and mean $\delta^{15}N_{Lake\ Superior}$ = 5.38‰, SD±0.78‰, N=74. The two estuarine trophic pathways are both based on a mix of river sediment and phytoplankton, but are distinguishable by location (upper estuary versus lower estuary) due to the longitudinal mixing of river and lake waters, which enriches the $^{13}C$ content of the food web at the river mouth (Hoffman *et al.* 2010), as well as the contribution of waste water treatment effluent, which enriches the $^{15}N$ content of the food web at the river mouth (Hoffman *et al.* 2012). To define the upper estuary source value, I used the mean $\delta^{13}C_{lipid\ corrected}$ and $\delta^{15}N$ values of White Sucker (*Catostomus commersonii*) captured in the river above Spirit Lake (i.e., associated with my upper estuary locations) from Blazer et al. (2016): mean $\delta^{13}C_{lipid\ corrected\ upper\ estuary}$= -34.0‰. SD±1.9‰, mean $\delta^{15}N_{upper\ estuary}$= 8.6‰, SD±1.3‰ (N=104). I used these values because White Sucker, like Ruffe, is a demersal fish that primarily consumes benthic invertebrates (Blazer *et al.* 2014; Gutsch and Hoffman 2016). The water near the Western Lake Superior Sanitary District (WLSSD) effluent, near the city of Duluth in the lower estuary, is typically $^{15}N$-enriched (Hoffman *et al.* 2012). To define the lower estuary source value, I used the mean $\delta^{13}C_{lipid\ corrected}$ and $\delta^{15}N$ values of two highly $^{15}N$-enriched benthic invertebrate samples taken adjacent to the effluent outfall of the WLSSD waste water treatment plant: $\delta^{13}C_{lipid\ corrected\ lower\ estuary}$= -30.2‰, SD±1.10‰, $\delta^{15}N_{lower\ estuary}$= 12.8‰, SD±0.33‰, N=2. This data was aquatic Mayfly data from Roesler (2016), processed using the methods described above.

A subset of the fish had substantially lower $\delta^{13}C$ and $\delta^{15}N$ values than my upper estuary source (i.e., Ruffe had $\delta^{13}C$ < -35‰ and $\delta^{15}N$ < 7‰), implying they were feeding in a trophic pathway based on an organic matter source not included in the three source model. To address this issue, I created a four source model. Because the solution of the four source model is mathematically underdetermined (i.e., two stable isotope ratios and four sources), I used an IsoSource model to estimate source contributions (IsoSource version 1.3). IsoSource is a Microsoft Visual Basic software package which iteratively calculates ranges and means of source proportional contributions to a mixture on stable isotope analyses when the number of sources is too large to permit a unique solution. The four sources I included in the model were upper estuary, lower estuary, Lake Superior, and methane contribution. I took a conservative approach with respect to this fourth source, assuming only fish with relatively low $\delta^{13}C$ values were obtaining some diet contribution from the source. I therefore only include Ruffe in the model that had a $\delta^{13}C_{lipid\ corrected}$ value less than -36‰. I chose this value because, based on the current literature, there are no fish ever recorded in SLR with a lower $\delta^{13}C_{lipid\ corrected}$ value (-36.6‰) (Sierszen *et al.* 1996; Hoffman *et al.* 2015). Very low $\delta^{13}C$ values in aquatic food webs occur when methane contributes to the food web (Bastviken *et al.* 2003; Ravinet *et al.* 2010; Jones and Grey 2011); methane $\delta^{13}C$ values typically range from -50‰ to -60‰ (Whiticar 1999). A small number of burrowing trichopterans had been sampled previously from the St. Louis River with very low $\delta^{13}C$ values, indicating the potential for a methane-based trophic pathway to contribute to production of higher-order consumers (J. Hoffman, unpublished data). To define the source value for the methane-based trophic pathway, I used the mean of five trichopterans samples with very low $\delta^{13}C$ values that were obtained from the upper estuary and correcting for the trophic enrichment factor (+0.4‰ $\delta^{13}C$, +3.4‰ $\delta^{15}N$; (Vander Zanden and Rasmussen 2001; McCutchan *et al.* 2003)): $\delta^{13}C_{lipid\ corrected\ methane}$= -72.0‰, SD±22.7‰, $\delta^{15}N_{methane}$= -3.1‰, SD±5.4‰.

## Results

I analyzed a total of 205 Ruffe captured in the St. Louis River, 2 captured in Lake Superior, 74 captured in Chequamegon Bay, and 16 captured from unknown (regions identifiable) locations (Table 10). The $\delta^{13}C_{lipid\ corrected}$ and $\delta^{15}N$ values were significantly different among the Ruffe captured in Lake Superior (including fish captured in Chequamegon Bay) compared to those captured in the upper estuary (ANOVA, df=2, Q<0.001, p<0.001) and lower estuary (ANOVA, df=2, Q<0.001, p<0.001), but there was no difference among the $\delta^{13}C_{lipid\ corrected}$ values (ANOVA, df=2, Q=1.481, p=0.416) or $\delta^{15}N$ values (ANOVA, df=2, Q=2.145, p=0.096) between Ruffe captured in the upper and lower estuary.

Ruffe captured in Chequamegon Bay and Lake Superior were isotopically similar (Figure 18), though the two fish caught in Lake Superior are slightly more $^{15}N$-enriched than the fish caught in Chequamegon Bay. Two Ruffe in Chequamegon Bay had $^{13}C_{lipid\ corrected}$ values indicating recent use of wetland habitat ($\delta^{13}C$ -29.81‰, $\delta^{15}N$ 7.68‰, and $\delta^{13}C$ -28.03‰, $\delta^{15}N$ 5.57‰) (Figure 18), which was unusual for Ruffe in Chequamegon Bay. Within the St. Louis River, some of the Ruffe captured had stable isotope ratios similar to either the upper estuary or lower estuary sources values, implying these fish were likely exclusively feeding in these areas. Ruffe were also captured with $\delta^{13}C$ and $\delta^{15}N$ values intermediate between these source values, implying these fish were likely feeding throughout the lower and upper estuary (Figure 18). Among fish caught in the lower estuary, there were two fish that are noticeably $^{15}N$-enriched ($\delta^{13}C$ -28.29‰, $\delta^{15}N$ 15.87‰) and ($\delta^{13}C$ -26.74‰, $\delta^{15}N$ 16.14‰). About half of the fish caught in the upper estuary and a quarter of the fish caught in the lower estuary had a $\delta^{13}C_{lipid\ corrected}$ value of -36‰ or less (Figure 18).

There were size differences associated with capture location and stable isotope ratios. Ruffe captured in Chequamegon Bay were the smallest among the capture locations (mean=59.53 mm, SD=25.04), but did not have either the smallest or the largest individual Ruffe (range: 33-117 mm) (Figure 19). This small mean size can be attributed to the abundance of juvenile Ruffe captured in Chequamegon Bay during summer of 2015 (Table 9). In the St. Louis River, fish size varied by capture location. The Ruffe captured in the lower estuary were an intermediate size (mean=70.70 mm, SD=17.21), but did include young-of-year (YOY; 25 mm total length). The Ruffe captured in the lower estuary ranged in size from 25-133 mm, which encompasses the larval, juvenile, and adult stages (Gutsch and Hoffman 2016). The Ruffe captured in Lake Superior also included YOY Ruffe. The Ruffe captured in the upper estuary included the majority of large, adult Ruffe (mean=88.57 mm and 83.1 mm, SD=16.47 and 16.27, respectively) (Figure 19). Moreover, among Ruffe captured within the estuary, those with a $\delta^{13}C$ <-36‰ included large adults (mean=83.1 mm, SD=16.27, range=37-119). Ruffe from unknown locations, which were all caught in the winter, were the largest fish captured (mean=113.75 mm, SD=18.01, range: 82-148 mm) (Figure 19).

Ruffe had an unprecedented range in $\delta^{13}C$ values: -52.2‰ to -14.2‰. Ruffe exhibited a remarkable size-based shift with respect to $\delta^{13}C_{lipid\ corrected}$ values (Figure 20a). Small Ruffe (<60 mm SL) generally had a $\delta^{13}C_{lipid\ corrected}$ value of about -20‰ to -35‰, indicating these fish have trophic pathways based in a mix of Lake Superior and estuarine organic matter sources. At lengths ranging from 60-80 mm, most Ruffe had a $\delta^{13}C_{lipid\ corrected}$ value of -25‰ to -45‰, indicating a marked shift away from Lake Superior habitat and towards a greater variety of estuarine organic matter sources, including the methane-based trophic pathway. The largest fish sampled, which ranged from 80-148 mm, had the largest range of $\delta^{13}C_{lipid\ corrected}$ values: -54‰ to -14‰ (Figure 20a). The range in $\delta^{15}N$ values was substantially less than the range in $\delta^{13}C$ values, generally 3‰ to 13‰ across the range of lengths. Two fish were $^{15}N$-enriched, with $\delta^{15}N$ values of about 16‰ (corresponding to fish of 45 mm and 95 mm total length; Figure 20b).

Upon closer examination, an interesting pattern is apparent (Figure 21). Ruffe less than 65 mm rely on both lake- and wetland-dominated trophic pathways. For fish that are 25-65 mm, the majority of the fish with higher than 50% lake-dominated trophic pathways are from Chequamegon Bay and Lake Superior and the ones with less than 50% are from everywhere else. There is a size-based shift after fish get larger than 65 mm. One hundred percent of fish 65-85 mm have a wetland-dominated trophic pathway. Then, once the fish are greater than 85 mm, they disperse again, almost equally, with 59.7% with a wetland-dominated trophic pathway and 40.3% with a lake-dominated trophic pathway (Figure 21).

Consistent with the wide range of stable isotope ratios observed, all three sources contributed to Ruffe nutrition (Figure 22). Among Ruffe captured in Chequamegon Bay, Lake Superior benthic periphyton was the dominant organic matter source (69 or 74 fish had >75% contribution from this Lake Superior trophic pathway). However, among Ruffe captured in the St. Louis River, many of the fish relied on upper estuary organic matter sources: 51 of the 205 fish had >75% contribution from the upper estuary source. In contrast, only 9 Ruffe had >75% contribution from the lower estuary source. There is also a notable break in the distribution of stable isotope ratios between the Ruffe caught in the estuary and the fish caught in Lake Superior (Figure 23). The remainder of the 76 fish relied on a mix of sources; 57 derived 25-75% of their nutrition from Lake Superior, 67 derived 25-75% of their nutrition from the lower estuary, and 71 derived 25-75% their trophic nutrition from the upper estuary. The standard deviations (SD) associated with the contribution estimates were source-dependent. For contributions <5% from the upper estuary source, the mean SD was 34%, and for contributions >95%, the mean SD was 27%. For the lower estuary source, contributions <5% had a mean SD of 22%; there were no contributions greater than 95%.  Contributions <5% from the Lake Superior source had a mean SD of 9% and contributions >95% had a mean SD of 33%.

Based on the four-source IsoSource model, the chemosynthetic trophic pathway based on MOB contributed an average of 23% (SD=10%, range= 13-53%) of nutrition to the subset of Ruffe captured in the St. Louis River with a $\delta^{13}C_{lipid\ corrected}$ <-36‰ (Figure 24a).  Among these same Ruffe, the upper estuary trophic pathway contributed an average of 48% (SD=35%, range= 0-101%) to their nutrition, whereas the Lake Superior trophic pathway only contributed an average of 26% (SD=29%, range= 0-100%) (Figure 24a).  In contrast, among the Ruffe captured in Chequamegon Bay, the Lake Superior trophic pathway contributed an average of 79% (SD=11%, range= 23-96%) to their nutrition, whereas the lower estuary trophic pathway (physically associated with the Fish Creek mouth and south end of Chequamegon Bay) contributed an average of 6% (SD=6%, range= 0-22%) (Figure 24b).

## Discussion

The flow of energy and nutrients among adjacent habitats and ecosystems is a defining character of coastal food webs (Hoffman *et al.* 2015). Evidence for both routine and episodic energy exchanges between coastal wetlands and riparian ecosystems, rivers and the adjacent open coast, and benthic and pelagic habitats is widespread (Vander Zanden and Vadeboncoeur 2002; Carpenter *et al.* 2005; Hoffman *et al.* 2015).  The results of this study stand apart because I found evidence for a novel source of energy to a coastal wetland food web: chemosynthetic methane-oxidizing bacteria. The data are remarkable in part because of the unusually large range in $\delta^{13}C$ values in Ruffe, but also because the very low $\delta^{13}C$ values indicate reliance on a methane-based trophic pathway. The data are also remarkable because they indicate a distinct, size-based shift in trophic pathways that are consistent with movements between coastal wetlands and the nearshore waters of Lake Superior. As such, the data indicate Ruffe – an invasive species - occupies a unique trophic niche within the Great Lakes. By occupying a unique niche, it allows Ruffe to reduce potential competition, and may also facilitate establishment of new wetland habitats. Within the context of the food web in an invaded coastal wetland, it also facilitates the emergence of novel trophic pathways. Here I discuss the role of MOB in the food web; limitations of the data and mixing models; and then habitat-specific, life cycle-based movements of Ruffe and implications for spread.

***METHANE CONTRIBUTION***

Ruffe were captured throughout St. Louis River and Chequamegon Bay, but Spirit Lake, a particular area within the St. Louis River, had a surprisingly high number of Ruffe (27) that were highly $^{13}C$-depleted. Sixty-two Ruffe had very low $\delta^{13}C$ values (>36‰); Ruffe captured in Spirit Lake composed nearly half of the $^{13}C$-depleted Ruffe. The lowest previously recorded $\delta^{13}C$ value for Ruffe is -43.6‰; the fish was captured in a temperate lake in Finland at a depth > 12 m, and it was estimated that within the lake methane contributed between 12% and 17% to Ruffe biomass (Ravinet *et al.* 2010). Those Ruffe were primarily consuming chironomid larvae (Ravinet *et al.* 2010). Sierszen et al. (1996) measured Ruffe that were $^{13}C$-depleted ($\delta^{13}C$ -36.6‰), which is the lowest $\delta^{13}C$ value previously recorded in the St. Louis River; the authors concluded that Ruffe was utilizing a carbon source not previously characterized in the river. Hoffman et al. (2010) measured Ruffe in the St. Louis River that was relatively $^{13}C$-enriched; the lowest $\delta^{13}C$ value was about -26‰ and the highest $\delta^{13}C$ value was -15‰. The Ruffe measured in this study is much more $^{13}C$-depleted than prior studies, with $\delta^{13}C$ values as low as -52.5‰. The highest $\delta^{13}C$ value for Ruffe was -14.2‰. The data show a remarkably wide range of stable isotope values, with a span of $^{13}C$ of 38.3‰ from the highest to lowest measured $\delta^{13}C$ value. Additionally, trichopterans captured in the St. Louis River had $\delta^{13}C$ values as low as -77.5‰.

To my knowledge, this is the first discovery of a higher consumer (Ruffe) having an extremely $^{13}C$-depleted signature in a coastal wetland. The carbon stable isotope ratio of the fish indicates it is feeding in a chemosynthetic trophic pathway, most likely based on methane production in anoxic sediment at the bottom of the river (Ravinet *et al.* 2010; Jones and Grey 2011). Trichopterans and chironomids most likely assimilate methane carbon by consuming methane-oxidizing bacteria (MOB). MOB are the source of extremely depleted $^{13}C$ because biogenic methane $\delta^{13}C$ values typically range from -60‰ to -50‰ (Whiticar 1999). MOB use of methane can result in further $^{13}C$ depletion with isotopic fractionation up to 20‰ (Summons *et al.* 1994; Jones and Grey 2011). Although chironomid larvae that are highly $^{13}C$-depleted can be consumed by higher consumers (Jones and Grey 2011), few studies have attempted to evaluate this (Harrod and Grey 2006; Ravinet *et al.* 2010), and there is little evidence for methane-derived carbon in higher consumers. Deines and Grey

(2006) found that demersal fish do not consume $^{13}C$-depleted chironomid larvae because they do not feed in oxygen-depleted water layers where the larvae are abundant. In the St. Louis River, only burrowing trichopterans captured in benthic dredge samples have been found to be sufficiently $^{13}C$-depleted to indicate feeding in a methane-based trophic pathway. It is not known how methane carbon is assimilated by these trichopterans. Presumably, their burrow intersects anoxic sediment within stratified sediment, and the overlying water has sufficient oxygen for these trichopterans to survive. It is plausible that the trichopterans are directly consuming MOB (i.e., feeding in the microbial food web) within their burrow, or consuming a mix of sediment and MOB. Little is known about this trophic pathway, and future research is needed.

Methane is an allochthonous carbon source, likely produced from terrestrial-derived organic matter, such a decaying litter and soil.  The methane-influenced food web is a donor system to the overall St. Louis River food web (recipient) through benthic invertebrates (trichopterans).  Methane is transferred to bacteria, and in turn the energy is transferred up several trophic levels and consumed by Ruffe.  Because trichopterans are intolerant to hypoxia, it is likely that the top layer of sediment is well-oxygenated, and their tubes are colonized by MOB at a depth within the sediment corresponding to a strong redox gradient. I found no methane fish outside of the St. Louis River, suggesting that Lake Superior is not a recipient of the methane-influenced food web.

The existence of a methane-based tropic pathway has not previously been demonstrated in a coastal wetland ecosystem.  These trichopterans are the only other organisms measured in SLR to have such a $^{13}C$-depleted stable isotope signal (Hoffman, personal comm., Gutsch, unpublished data). Both Bastviken *et al.* (2003) and Jones and Grey (2011) found chironomid larvae and zooplankton with depleted $^{13}C$ in other freshwater ecosystems, but these invertebrates have been analyzed in the St. Louis River and none has yet been measured with any unusually low $\delta^{13}C$ values (Keough *et al.* 1996; Sierszen *et al.* 1996; Hoffman *et al.* 2010; Blazer *et al.* 2014). Ruffe has the ability to feed in hypoxic environments at great depths (Bergman 1988b; Hölker and Thiel 1998), perhaps allowing it to forage in places other fish cannot (Jones and Grey 2011). Ruffe has been found previously to feed on chironomids in a methanogenic food web (Ravinet *et al.* 2010).

### ***SIZE-BASED HABITAT USAGE***

I found size-based habitat usage of Ruffe that corroborated the proposed Ruffe life-cycle from Gutsch and Hoffman (2016). Small fish have high $\delta^{13}C_{lipid\ corrected}$ values, medium-size fish have low $\delta^{13}C_{lipid\ corrected}$ values, and large-size fish have a wide range of $\delta^{13}C_{lipid\ corrected}$ values. The size-based patterns indicate a distinct connection between movement and life history. At small sizes, Ruffe disperses from wetland habitat and uses both the lake and the wetland as rearing habitat. At 65-85 mm, it is mature and moves into wetlands to spawn (Gutsch and Hoffman 2016). The associated shift in the stable isotope composition is noteworthy, as other migratory fishes are known to spawn and not feed, allowing researchers to use the stable isotope composition to track their origin (Groot and Margolis 1991). The shift implies either that the fish is moving into the wetland to feed prior to spawning (i.e., staging), or that it feeds in the wetland during the spawning period. After spawning, in mid- to late-summer it disperses again, moving into the lake or else remaining in the coastal wetland.

### *LAKE SUPERIOR VS WETLAND USAGE*

Although Ruffe is commonly found in wetlands, my results demonstrate it uses Lake Superior at multiple life stages. Its life-history is comparable to the native percid, Yellow Perch (*Perca flavescens*). Ruffe lays its eggs in the spring or early summer in a wetland, similar to Yellow Perch, which also lays its eggs in the spring (Schoen *et al.* 2016). Ruffe exclusively spawns in coastal wetlands but not Yellow Perch (Robillard and Marsden 2001). Ruffe spawns in shallow (<3 m) and relatively warm water (5-18$^{o}$C;Kiyashko and Volodin 1978; Brown *et al.* 1998; Gutsch and Hoffman 2016). Yellow Perch can spawn in deeper (14 m), cooler water (Huff *et al.* 2004) than Ruffe. As larvae, Ruffe is demersal and remains on the spawning grounds in the wetland (Disler and Smimov 1977; Gutsch and Hoffman 2016). In contrast, Yellow Perch larvae have a 40-day pelagic phase post-hatch to evade predation and begin feeding, after which they return to littoral vegetation (Whiteside *et al.* 1985). As Ruffe transitions to the juvenile stage, it moves into lake and coastal wetlands, as shown by the stable isotope ratio data. Juveniles and adults move freely from wetland to lake to access resources and overwinter (Gutsch and Hoffman 2016). Its use of the coastal wetland and lake is similar to Yellow Perch. Yellow Perch in the Great Lakes has a variety of life history strategies, including annual use of wetland habitat, bi-annual or year-round use of wetland habitat, and wetland habitat use as juveniles and movement to nearshore as adults, suggesting it can spawn in nearshore habitat (Schoen *et al.* 2016). In contrast, Ruffe migrates to and from the lake and wetland throughout the different stages of its life (some may stay in the wetland their whole life), but it only spawns in coastal wetlands (Gutsch and Hoffman 2016). As such, spawning habitat is a limiting factor for Ruffe (Gutsch and Hoffman 2016).

### *STABLE ISOTOPE AND MIXING MODEL OUTPUT UNCERTAINTY*

An important consideration for interpretation of stable isotope ratios of fish is the isotopic turnover, which has an allometric relationship to the size of the fish (Vander Zanden *et al.* 2015). Based on the allometric relationship (Vander Zanden et al. 2015, Eq. 2: constant 0.16, intercept 3.28), for all Ruffe (average weight 13.66 g) the estimated half-life is 40.4 days. For small Ruffe (25-60 mm), the half-life is 31.2 days, for medium Ruffe (60-85 mm), the half-life is 37.5 days, and for large Ruffe (85-148 mm) the half-life is 45.7 days. The ecological implication of these half-life estimates is that stable isotope ratios reflect diet and movement integrated over seasons in large fish, whereas they reflect within-season diet and movement in small fish. These long half-lives for large fish prevent a direct interpretation with respect to life history because seasonal movements are common (Gutsch and Hoffman 2016). However, these are still relatively short half-lives, less than a year, compared to other fish.

For mixing models, the error in source contribution is related to the isotopic differences among sources. For the mixing model, the methane and lake source $\delta^{13}C$ and $\delta^{15}N$ values are well-separated; however, the upper and lower estuary sources values were much more isotopically similar to each other. This similarity reduces the certainty attributing upper and lower estuary sources to fish production relative to lake and methane sources.

Second, among the Ruffe I sampled, a few had unusually high $\delta^{15}N$ values, the source of which is likely nitrogen in Western Lake Superior Sanitary District (WLSSD) effluent (Hoffman *et al.* 2012). It is likely that these Ruffe were feeding in the effluent near this facility, the outflow of which is located in the north corner of St. Louis Bay (Figure 16).

Finally, many of the Ruffe sampled had $\delta^{13}C$ and $\delta^{15}N$ values that were intermediate between the sources (Figure 22). This movement behavior of these fishes is difficult to interpret because the fish can either be moving between locations to feed, feeding in a location with an intermediate isotopic value associated with the food web, or feeding in a region but occasionally intercepting prey drifting from the other location. The model output cannot discern among these alternatives. The question arises as to whether the evidence for lake habitat could be acquired without feeding in Lake Superior. While it is possible that some lake signal could be acquired by consuming the eggs of potomadromous fishes, these fish generally spawn in a part of the river that is poor habitat for Ruffe, at the top of the estuary where there is gravel and cobble substrate and fast current. Alternatively, Ruffe feeding at the river mouth near the edge of transition zone may have a lake-influenced isotopic composition, but the river discharge is generally sufficient that the isotopic composition of the food web at the river mouth is more similar to the wetland than Lake Superior (Hoffman *et al.* 2010; Bellinger *et al.* 2016).

### ***CONCLUSIONS***

My goals were to identify movements linked to life-history and to identify trophic pathways supporting Ruffe in the St. Louis River. I found that Ruffe exhibit remarkable size-based movements throughout its life cycle, and it is dependent on coastal wetlands and demonstrates facultative use of Lake Superior. The landscape mosaic in this study included three ecosystems: river bottom, coastal wetland, and lake. The ecosystems are connected by a mixture of autochthonous and allochthonous inputs, including a chemosynthetic pathway. The stable isotope analysis revealed that some Ruffe were feeding in a methane-based trophic pathway, possibly reducing resource competition. No other study has found use of this trophic pathway in coastal wetlands. The role of methane in coastal wetlands merits further investigation because, based on this study, it potentially has important implications with respect to both carrying capacity and invasive species resource competition.

# Chapter 4: Lake Superior-scale species distribution modeling of Ruffe (*Gymnocephalus cernua*)

## Abstract

Species distribution modeling is an innovative way to predict suitable habitat of invasive species. My goal was to understand how using environmental data resolved to relatively fine spatial scales (i.e., 100m to 1000 m), as well as using different species occurrence data of varying temporal windows, would affect model performance with respect to predicting potential habitat of an invasive fish, Ruffe (*Gymnocephalus cernua*). I used 30-m-scale environmental variables to develop a Maxent species distribution model. To examine the effect of spatial data resolution, I developed and compared competing models at different spatial scales: 250-m, 500-m, 1000-m, 2000-m, and 2000-m selected model. In addition, I conducted two time-series analyses, comparing models developed from occurrence data broken into decade time blocks (1986-1996, 1997-2006, 2007-2014) and analyzed separately or cumulatively. I calculated percent suitable habitat for all of the models. I predicted that there would be an optimal spatial scale to model Ruffe—that very low and very high spatial scale models would not perform well, but a model at intermediate spatial scales would be the best model. Among the models constructed using environmental data from various spatial resolutions, the best performing model used 500-m data and the worst performing model used 2000-m data. The important geographic discrepancies in potential habitat occurred around the Apostle Islands, WI, Isle Royale, MN, Grand Marais, MI, Whitefish Point, MI, and Red Rock and Nipigon in Canada. I showed multiple models that performed similarly, according to area under the curve (AUC) scores but had different physical results with the suitable habitat prediction maps and percent area predicted. Differences in grid sizes of

100s of meters resulted in differences of thousands of square kilometers of predicted suitable habitat.  The Maxent model results from the separate and cumulative time-series analyses were similar.  I found minor differences in the environmental variable outputs. However, I found substantial differences in the AUC scores for the time-series analyses.  The separate time-series models all performed similarly well, but the performance of the cumulative models declined as data were added to subsequent models.  A 30-m-scale species distribution model for Ruffe in Lake Superior can be used for showing areas that are suitable habitat for them.  Maxent can be a powerful tool to model invasive species, using the precautions outlined in my methods.

## Introduction

There have been recent advances in the ability to model a species' geographic distribution based on their ecological niche (Elith 2002; Elith *et al.* 2006, 2010, 2013; Phillips and Dudík 2008; Khanum *et al.* 2013; VanDerWal *et al.* 2013; Yang *et al.* 2013; Guillera-Arroita *et al.* 2014; Matyukhina *et al.* 2014; Yi *et al.* 2016), an idea first introduced by Joseph Grinnell (Grinnell 1924; Guisan and Zimmerman 2000; Pearson and Dawson 2003; Peterson 2003, 2006; Soberon and Peterson 2004). Grinnell (1924) focused on an individual species' geographical confinement by its biotic and abiotic ecological needs and posited that understanding an organism's niche would better help us understand the evolution of that organism. Elton (1927) later expanded the niche concept to include a species' interaction within its community, not only its geographic location. Elton (1927) observed that organisms can have almost identical niches, such as a specific type of carnivory, in different communities even when they are geographically separated. Hutchinson (1957) later postulated that the niche could be conceived as a n-dimensional hypervolume, wherein the hypervolume is defined by all biotic and abiotic factors that affect the species in the community and represents the multi-dimensional space in which an organism can exist based on all of these factors. Hutchinson (1957) called the hypervolume an organism's fundamental niche. MacArthur (1972) quantified and integrated the two concepts of the individual and community ecological niches. According to Peterson (2003), the niche defined by Grinnell and MacArthur is: "the quantity [any ecological requirement] that limits geographic distributions of species." The fundamental niche is defined by all of the variables in which the organism can exist long-term. In contrast, the realized niche is usually within the fundamental niche and is the subset where it actually occupies (Hutchinson 1957; Phillips *et al.* 2006).

Species distribution models (SDMs) are used to predict suitable habitat (or fundamental niches) for species across a particular landscape. In the context of non-native species, they have been applied to identify likely places where non-native species could successfully establish if introduced, as well as locations to which they could spread (Peterson and Vieglais 2001; Peterson and Robins 2003; Thuiller *et al.* 2005; Chen *et al.* 2007; Ficetola *et al.* 2007; Broennimann *et al.* 2007; Jeschke and Strayer 2008; Jiménez-Valverde *et al.* 2011). For example, Drake and Lodge (2006) created a SDM that predicted suitable habitat for Rainbow Smelt (*Osmerus mordax*) and Ruffe (*Gymnocephalus cernua*) within North America; based on the model, Ruffe was likely to invade the Midwestern and Northeastern United States. However, because the model output had a relatively coarse geospatial resolution of 0.1 degree decimals, it had low predictive power at the "local" level.

Identifying locations at high risk for invasion requires some understanding of vectors for spread, relative propagule pressure, and the suitability of the chemical, physical, and biological conditions (Colautti and MacIsaac 2004). Species distribution modeling is used to predict whether or not chemical or physical (or both) conditions are suitable for an introduced species to establish and spread throughout a particular landscape (Peterson 2003). SDMs are cost effective because they can use existing data (Fielding and Bell 1997). However, these models have limitations based on how they are constructed. Typically, SDMs use global climate data, such as annual cloud cover, annual frost frequency, annual vapor pressure, annual precipitation, mean annual temperature, slope, etc., as their environmental component and occurrence data from the native range of the organism (Peterson and Vieglais 2001; Peterson *et al.* 2003; Phillips *et al.* 2006). Often the prediction maps are at such a large scale that the output gives only a vague idea (e.g., all of the Great Lakes) of where an invasive organism might be able to establish a population.

Within Lake Superior, Ruffe is an ideal model invasive species for constructing a SDM. It first invaded the St. Louis River estuary, MN, (Figure 25A) in 1986; there was a steady population increase until 1995, and then the population sharply declined, indicative of the typical "boom-bust" cycle of most invasive species (Chapter 2). Ruffe spread to Thunder Bay Harbor, Ontario, Canada, by 1991, Lake Huron by 1995, and Lake Michigan by 2002, most likely by inter-lake spread when eggs or larvae were introduced in ballast water from commercial ships (Ricciardi and MacIsaac 2000). Ruffe is a habitat generalist, spawns multiple times throughout the spawning season, and it has high fecundity (Gutsch and Hoffman 2016). Ruffe is highly competitive with native, benthic fishes (Ogle 1998). Despite these characteristics, Ruffe has yet to spread extensively through the upper Great Lakes (USEPA 2008; USGS 2014). Because it has not spread everywhere in Lake Superior, the opportunity exists to use available presence data within the Laurentian Great Lakes to model potential suitable habitat elsewhere in Lake Superior.

I developed a SDM using Ruffe as a model species. My lake-scale environmental variables were at a 30-m-scale instead of a global scale. To examine the effect of spatial data resolution, I developed and compared competing models at different spatial scales: 250-m, 500-m, 1000-m, 2000-m, and 2000-m selected model. In addition, I conducted two time-series analyses, comparing models developed from occurrence data broken into decade time blocks (1986-1996, 1997-2006, 2007-2014) and analyzed them separately and cumulatively. I predicted the area of suitable habitat within the buffer and Lake Superior for each model and for three habitat zones—offshore, nearshore, and in-shore. I predicted that there would be an optimal spatial scale to model Ruffe—that very fine and coarse spatial scaled models would not perform well, but a model with intermediate spatial scale would be the best model.

## Methods

### *STUDY AREA*

My study area was Lake Superior, USA (Figure 25). The lake has a surface area of 82,097 $km^2$ (maximum length 563 km, maximum width 257 km), and a shoreline length of 4,393 km (including islands). Its volume is 12,232 $km^3$ (maximum depth 406 m, average depth 149 m), with a retention time of 173 years (GLERL and NOAA 2000).

### *RUFFE OCCURRENCE DATA AND ENVIRONMENTAL DATA*

For my model, I used adult and juvenile Ruffe occurrence data (i.e., presence only, absences were excluded) from multiple sources (Table 11). I had a total of 362 occurrences (Figure 26). Most occurrences were within Lake Superior, but a few occurred in inland lakes or streams connected to Lake Superior for which I lacked corresponding environmental data. Assuming these fish at some time occupied the connecting water body, I associated the points with the nearest, connected shoreline location using shoreline data from the Great Lakes Aquatic Habitat Framework (GLAHF 2017). I found substantial clustering of occurrences in two locations, the St. Louis River (194 points) and Chequamegon Bay (74 points), which accounted for 74% of the total occurrences (Figure 26). Based on the variogram, the occurrence data were autocorrelated at a relatively fine spatial scale (range = 77.43 km, nugget = 13.22 km, sill = 13219.14) due to this clustering in Chequamegon Bay and St. Louis River. Model iterations to address this autocorrelation are described below.

I limited the spatial domain of the model using the occurrence data by setting a buffer around the Lake Superior shoreline (Figure 26). The limit of the buffer was set to either the maximum depth (205 m) or distance from shore (15 km) that Ruffe has been captured in Lake Superior, assuming these bounds represent a limit on suitable habitat. Several areas along the north shore on the US side that were excluded from the model because the bottom depth was too great (Figure 26).

The environmental data I included in all of the models were turbidity, depth, substrate type, wave height, and distance to the nearest wetland (Table 12). Light extinction is one of the most important variables to Ruffe. Ruffe lives in dark or turbid areas and is adapted to low-light conditions. It possesses both a tapeta lucidum and well-developed lateral line (Gutsch and Hoffman 2016). Ruffe is also often found in deep, dark water. However, it requires shallow water habitat, whether turbid or clear, for spawning (Gutsch and Hoffman 2016). Ruffe do not exhibit strong preferences for specific substrates and has been found in almost every kind of substrate. However, it may prefer mud or clay due to the turbid qualities (Gutsch and Hoffman 2016). Wave height was used a proxy for both depth and exposure. For example, in a deep, offshore, exposed location, waves are typically higher than in a shallow, inshore, protected location. Finally, distance to wetland was chosen because Ruffe is wetland-dependent. It is routinely captured in and requires coastal wetlands for spawning (Chapter 3). All data layers were resampled to a 30-m resolution.

Turbidity data came from the Michigan Tech Research Institute (http://www.mtri.org/). Turbidity was determined using MODIS imagery from NOAA and NASA at K490, which is the diffuse attenuation coefficient at 490 nm (Wang *et al.* 2009) (Figure 27). In essence, it measures the rate at which light at wavelength 490 nm is attenuated with depth. I retrieved turbidity data only for the summer months (June, July, and August) for 2010-2013 and averaged those images. June, July and August were chosen because they include both stratified (July and August) and unstratified (June) conditions and are ice-free months. I had a total of 12 images, 1 image for each month, 3 images for each year. Michigan Tech averaged the values of MODIS images of cloud-free pixels and provided the monthly averages, from which I estimated the annual averages. The original resolution for turbidity was 1 km x 1 km, but I resampled it to 30 m x

30 m so I could use it in the model. The range of turbidity within the model spatial domain was 0-12.7 nm (mean 0.16 nm, and one standard deviation [SD] 0.31 nm).

Depth, substrate type, wave height, and distance to wetland all came from the Great Lakes Aquatic Habitat Framework (GLAHF) data set (https://www.glahf.org/). Depths within the model domain ranged from 0-205 m, with an average of 85.9 m ± SD of 59 m (Figure 27). Available substrate types included mud, sand, hard, and clay (Figure 27). The percentage of each of these within my buffer varied: mud (21.0%), sand (9.3%), hard (43.2%), and clay (26.6%). Substrate types for the offshore (>100 m) were digitized from observations published in peer-reviewed publications, and in the coastal and nearshore areas (<100 m) were described by the Army Corp of Engineers (2012) and confirmed by researchers across the Great Lakes (GLAHF 2017). I calculated distance from occurrence points to coastal wetlands using Euclidean distance (mean 32,555 m ± SD 37,717 m, range 0 to 146,456 m). The coastal wetlands dataset published by GLAHF came from the Great Lakes Coastal Wetland Consortium (GLCWC) (GLAHF 2017). Wave height was retrieved from the GLAHF wave action section, developed by U. S. Army Corps of Engineers (USACEs) Wave Information Studies (Figure 27). WISWAVE is a model used to calculate wave height. WISWAVE is a discrete spectral wave model (Engineers 2010) that models wind wave generation and propagation and helps determine spatial and temporal changes in wave field as a function of wind (Dhanak and Xiros 2016). I derived wave height from GLAFH; I interpolated it using ArcGIS software.  Within my model domain, wave height averaged 0.324 m ± SD 0.009 m and ranged from 0.0985 to 0.530 m.  I used Pearson's coefficient to examine whether the environmental variables were correlated; correlations were generally weak (Table 13).

## SDM Model

For my SDMs, I used a maximum entropy algorithm and the Maxent software (Maxent, version 3.3.3k) (Phillips *et al.* 2006).  Maxent is a maximum entropy based machine learning program.  It is becoming increasingly popular to use for species distribution modeling due to its high performance (Elith *et al.* 2006; Hernández *et al.* 2006).  Maxent uses presence-only occurrence data and environmental data (continuous or categorical) in ArcGIS. It uses environmental constraints to estimate the probability distribution for a species' occurrence (Phillips *et al.* 2006).  Maxent uses the equation:

$$\Pr(y = 1|\, z) = f_1(z)\,\Pr(y = 1)\,/f(z),$$

that shows if I know the conditional density of the covariates at the presence sites ($f_1(z)$) and the unconditional density of the covariates across the study area, f(z), I then need to know the prevalence Pr(y=1) to calculate the conditional probability of occurrence.  Maxent first estimates the ratio $f_1(z)/f(z)$, which is the raw output and then estimates the logistic output: $\log(f_1(z)/f(z))$ (Elith *et al.* 2011). The output of Maxent is a relative probability estimate of presence of the species from 0 to 1, with 0 being low probability and 1 being high probability.  The prediction map shows suitable habitat.  For each of my models, I used the default settings in Maxent, which is standard practice.  Thirty percent of the occurrence data were kept out as test data; the other seventy percent were used as training data.

Ruffe occurrence data together with background data were used to determine the Maxent distribution. Background data are a random sample of points from the landscape (that may or may not be occupied by Ruffe). I created 6 different models for comparison and 5 additional models for the time series analysis. Different numbers of occurrences were assigned to test, training, and background data for each of the six models (Table 14).

For each model, I calculated percent suitable habitat using a logistic threshold at maximum test sensitivity plus specificity within my buffer and for Lake Superior. This was a value I used from the output as a cutoff to determine the percentage of suitable habitat; everything above that value was suitable and everything below the value was not suitable. To evaluate the ecological significance, I calculated the percent of suitable area within the model domain found within each of three depth zones (in-shore (<30 m), nearshore (<100 m), and offshore (>100 m)) commonly used for Lake Superior management plans (Figure 28). I used an ESRI Zonal statistics tool to calculate the suitable area and percent per zone. The 30 x 30-meter raster was then converted to meters squared to determine the final area that was occupied by each model and zone within the buffer and Lake Superior. All raw data and calculations are reported in Table A-3.

***MODEL VARIATIONS***

Because non-native species in the Great Lakes have generally been found most commonly in and around urban areas and shipping ports, spatial clustering of species presence data around urban areas and ports is typical (O'Malia *et al., in review*). I found substantial clustering in the St. Louis River and Chequamegon Bay, with significant autocorrelation as indicated by the associated variogram. Because of this autocorrelation, I created 6 different SDMs, each with a different distance buffering surrounding the occurrence points (focal point) to remove clustering. These buffers included 250-m, 500-m, 1000-m, 2000-m, 2000-m selected removal, and no point removal (all data). The buffering distances were chosen by analyzing variograms of all the data to identify the sill, and then choosing several distances surrounding the sill distance. The buffers were created in ArcGIS. Each point was buffered at 250-m, 500-m, 1000-m, and 2000-m and presence was recalculated at the specified buffer scale. For the 2000-m selected model, only points in St. Louis River and Chequamegon Bay were removed at 2000-meter distances and all the other points were left alone. The 250-m, 500-m, all data, and 2000-m selected models still had some autocorrelation. The autocorrelation effects the model covariates, but not the model outputs.

In addition, I conducted a time-series analysis on all of the Ruffe data from 1986-2014. I broke the data into approximate ten-year increments: 1986-1996, 1997-2006, and 2007-2014. First, I examined a cumulative time series analysis (i.e., sequentially adding the data by decade). Second, I examined a discrete time-series analysis (i.e., treating each decadal data set separately). My goal was to determine whether examining the time-series cumulatively would yield different results than the discrete analysis. The cumulative time-series analysis mimics the tracking of Ruffe movements through time, whereas the discrete analysis maintains the evolution of distribution through time. For the cumulative time-series analysis, I developed three Maxent models using all of the occurrence points in Lake Superior within the following calendar years: 1986-1996, 1986-2006 and 1986-2014. For the separate time-series analysis, I created separate Maxent models for each ten-year time frame: 1986-1996, 1997-2006, and 2007-2014. I compared models within each type of time-series analysis using area under the curve (AUC), and I compared the environmental variables of each model using several Maxent outputs: response curves (variable vs logistic output), percent contribution (percent the variable contributed to the model), and jackknife of the test gain (determines maximum likelihood with the variable in the model alone or without the variable in the model). I also produced a map of the predicted suitable habitat within Chequamegon Bay to illustrate differences among the models. Chequamegon Bay was of particular interest because Ruffe established there many years after first introduction into the lake and because it has diverse habitat.

I used the area under the receiver operating characteristic (ROC) curve AUC test statistic to evaluate model performance (Phillips and Dudík 2008). Phillips and Dudik (2008) described the AUC as the probability that a randomly chosen presence site will be ranked above a randomly chosen absence site. AUC on average is 0.5 and 1.0 is perfect; 0.75 is considered “potentially useful”

(Elith 2002). Without absence data, background or pseudo-absence data is used, as with my study, to perform the test. In this case, the AUC is described as being the probability that a randomly chosen presence site is ranked above a random background site (Phillips *et al.* 2006). I also compared models qualitatively using map outputs (i.e., prediction maps). I qualitatively compared environmental variables within each of the models using Maxent output response curves. Then I compared the variables in the models using two Maxent outputs - percent contribution and jackknife of test gain. The jackknife refers to the method of removing one variable at a time and rerunning the model without it. It allows the testing of the influence of the variable on “gain” which is basically a likelihood statistic that maximizes the probability of the presences in relation to the background data.

## Results and Discussion

### ***COMPARISON OF SDMS VARYING SPATIAL RESOLUTION***

All of the Maxent models showed high predictive power (AUC > 0.9). However, the best model, based on the AUC score using test data, was the 500-m model, with an AUC score of 0.977 (Figure 29). The model with all the data and the 2000-m selected model had AUC scores similar to the 500 m model (Figure 29). The 250-m and the 1000-m models had about the same AUC score, and the-2000 m model had the lowest AUC score using test data. However, all of the models were greater than 90% accurate based on their AUC scores, and all but one was greater than 95% accurate (Figure 29).

As the distance buffer increased, clustering decreased, and the AUC increased to its maximum (500-m model) and then began to decrease again (Figure 30). However, the 2000-m selected model, where I only removed points at a distance of 2000-m from the very clustered areas (Chequamegon Bay and St. Louis River), performed very well, almost as well as the best distance buffer model (Figure 29). But the 2000-m selected model also had the most spatial autocorrelation due to clustering at other locations.

The Maxent models showed similar habitat suitability among spatial variations of buffer distances. The all-data model and the 500-m model had very similar habitat suitability (Figure 31). The 2000-m selected model had the most predicted suitable habitat (Figure 31, Table 14). As expected, all models predicted that the St. Louis River and Chequamegon Bay were highly suitable (Figure 31). Similarly, all models predicted high habitat suitability for Ruffe along the south shore of the western arm of Lake Superior, across the central south shore, from L'Anse, MI, to Au Train, MI, along the southeast corner of the lake from Whitefish Bay to Sault Ste. Marie, MI, and by Hurkett and Thunder Bay in Canada. There were some notable differences as well, described here in counterclockwise order about the lake, starting at the Apostle Islands. Habitat suitability around the Apostle Islands is high in all models except the 250-m model (Figure 31). There is high habitat suitability for Ruffe around the Keweenaw Peninsula and through Portage Lake in all of the models except the 250-m model. There is high habitat suitability around Grand Marais, MI, and Whitefish Point, MI in the model with all data, 500-m, and 2000-m selected models (Figure 31). There is some suitable habitat in those locations in the other models as well. The 1000-m and 2000-m model show some suitable habitat for Ruffe near Red Rock and Nipigon in Canada (Figure 31). There is some unexpected prediction of suitable habitat for Ruffe along the north side of Isle Royale for the 500-m, 1000-m, 2000-m and 2000-m selected models; for the 500-m and the 2000-m selected models, the entire island was deemed suitable habitat. This is surprising because there is substantial angler effort and multiple fisheries-independent surveys in the nearshore habitat of Isle Royale, yet only one Ruffe has ever been captured and reported there (Figure 31). All models were in agreement that there is a lack of suitable habitat along the north shore of Lake Superior (Figure 31). This finding is likely due to the geomorphology of the areas. The north shores of Lake Superior are steep and rocky with very few wetlands, low turbidity, and high exposure. In addition, Ruffe requires wetlands at some point during its life cycle to reproduce (Gutsch and Hoffman 2016).

The percent area predicted to be suitable habitat within the buffer increased from the 250-m model to the 1000-m model, but then decreased slightly for the 2000-m model (Table 15). The model with the most predicted area was the 2000-m selected model. Whereas, the model with all the data had an intermediate percentage (Table 15). The same trends are true for prediction of suitable habitat in Lake Superior. The observation that the maximum suitable area was associated with the 1000-m and 2000-m models can potentially be explained by the methodology. By expanding the buffers to reduce autocorrelation, the proportion of habitat assigned to be suitable for Ruffe was increased. Plausibly, this is because as the buffer distance increases, the buffer extends farther into Lake Superior and includes both suitable and unsuitable conditions (turbidity, depth, wave height). This process causes the model to associate a presence with habitat that is less turbid, deeper, or with greater wave height than where the fish was actually captured because the shallow, turbid, protected nearshore composes a very small amount of habitat area in Lake Superior.

The predicted distribution of habitat within the inshore, nearshore, and offshore depths varied widely among models. For inshore habitat, there is an increase from the 250-m model (~6%) to the 1000-m model (~16%), then the percentage decreases again for the 2000-m model (15.5%). The model with all the data is intermediate. For the nearshore habitat, I found the same relative ranking. For the offshore habitat, however, there is no suitable area for the 250-m model, almost 0% for the 500-m and the 2000-m models, less than 1% for the all-data model and the 1000-m model, and almost 3% for the 2000-m selected model (Table 14).

The 500-m model and the 2000-m selected model have almost identical AUC scores, and yet the percent of suitable area predicted for the 2000-m selected model is almost three times that of the 500-m model (Table 14). Further, there is 6 times more offshore habitat predicted for the 2000-m selected model than the 500-m model, despite having the same accuracy. The 2000-m selected model was highly spatially autocorrelated, but the 1000-m model, with an AUC just below the two best models and no autocorrelation, also predicts almost 3 times the amount of suitable habitat for Ruffe than the 500-m model and about the same amount of offshore suitable habitat as the 2000-m selected model (Table 15). Similar issues arise when comparing the in-shore habitat among models. The 500-m model predicts about half as much suitable habitat than the 1000-m and the 2000-m selected models (Table 15).

It is important to examine model performance versus the model outputs from a management perspective. If a natural resource manager were to choose to use the 500-m model, he or she might greatly under-predict the amount of area to be monitored for Ruffe and might subsequently fail to detect the fish. However, if he or she used the 2000-m selected model, they might greatly over-predict the amount of area to be monitored for Ruffe and subsequently waste resources and lower the overall probability of detection. Ruffe is known to occupy in-shore habitats and some nearshore habitats, but less commonly known is that that it occupies so much offshore habitat (Chapter 1). Ruffe occupies deep waters of Lake Superior, but the ecological role of deep water habitat for Ruffe remains unknown. Both larvae and adults are captured in Lake Superior (Chapter 3).

Overall, the response curves (environmental variables vs. logistic output) are consistent across all models, except for one difference. For every model, when other environmental variables are held at their average value all categories (mud, sand, clay, and hard) are important; when substrate is by itself in the model, sand is most important. Overall, all environmental variables have high logistic output at low values and decrease as the values of the variables increase.

Based on percent contribution, depth and wave height were generally the most important variables in the model. Depth was the most important environmental variable for all models except the all-data and 250-m models, for which it was second most important next to wave height (Table 16). In all of the other models, wave height was second most important, except for the 2000-m selected model, whereas distance to wetland was the second most important variable. Substrate type was least important for the model with all data, 250-m, 500-m, and 1000-m models, whereas turbidity was least important for the 2000-m and the 2000-m selected model (Table 16).

Based on the jackknife of the test gain results, wave height was the best variable alone in all of the models, except for the 2000-m model, in which distance to wetlands was best by itself (Figure 32). It is unknown why these results are not wholly consistent with the previous results (i.e., depth generally was most important when all variables were considered together). In the all-data model, the likelihood of the model decreased if depth, turbidity, or wave height were removed (Figure 32). In the 250-m model, the likelihood of the model decreased when removing wave height and distance to wetland. In the 500-m model, wave height and depth decreased the likelihood of the model. For the 1000-m model and 2000-m selected model, wave height and turbidity decreased the likelihood of the model. In the 2000-m model, turbidity decreased the likelihood of the model (Figure 32).

As a whole, the results indicate that among the environmental variables considered, depth, wave height, and turbidity can explain much of the variation in Ruffe distribution in Lake Superior. Depth and wave height possibly have a high magnitude of change (or large range) and are spread across a large gradient, and so might be better predictors than substrate, distance to wetland, and turbidity. This is apparent in the jackknife analysis; turbidity was an important variable on its own, but not in combination with the other variables. However, when all of the variables are included in the model, depth and wave height have a higher percent contribution than turbidity. It is also possible that individual variable importance is affected by spatial autocorrelation. Note that for the 2000-m selected model, in which spatial autocorrelation was removed from the St. Louis River and Chequamegon Bay, no single variable contributes strongly to the model outcome, and there is little, if any, effect on the likelihood of the model when any single variable is removed from the model (Figure 32). That turbidity and depth were important model factors is consistent with Ruffe biology. Ruffe is a demersal species that prefers low light conditions of either turbid or deep waters (Gutsch and Hoffman 2016). To some extent, depth and wave height may be confounded in the models because depth and wave height are related (Pearson correlation coefficient = 0.418, p=0). In contrast, high turbidity generally only occurs in shallow waters. Given that Ruffe requires wetlands for spawning, it was surprising that distance to wetlands was only best alone in the 2000-m model. This is plausibly because at this large grid size (2000 m), there is loss of depth and turbidity resolution such that Ruffe that are in shallow and turbid locations near the shoreline are assigned to a greater depth and lower turbidity because the cell in which they are located includes more nearshore and less inshore habitat compared to the smaller grid sizes. In contrast, distance to wetlands becomes more important because the distances (generally >2000 m) are relatively unaffected by this scaling exercise.

### ***TIME SERIES ANALYSIS***

For the separate time-series analysis, the AUC scores and response curves were similar among models (Figure 33). Wave height had the greatest percent contribution in all three models. For 1986-1996, turbidity had the second highest percent contribution, whereas for 1997-2006 and 2007-2014, depth had the second highest percent contribution. In the jackknife analysis, for the 1986-1996 model, wave height and turbidity were most important when alone in the model, and there was only a small effect when any variable was removed from the model. For the other two stanzas, wave height was the most important variable when alone in the model. For the 1997-2006 model, there was only a small effect when any variable was removed from the model. For the 2007-2014 model, model likelihood was decreased the most by removing depth, turbidity, and wave height. The maps of Chequamegon Bay indicate an increase in suitable habitat as the time frames progress. By the 2007-2014 times, almost all of Chequamegon Bay is predicted to be suitable habitat (Figure 34). This result indicates that the model prediction of suitability of a given place (i.e., Chequamegon Bay) depends on the time-dependent habitat information provided as Ruffe spreads along the lake’s southern shore.

For the cumulative time-series model, the AUC scores decreased as data were added for each time period; 1986-1996 had the highest AUC score (0.979), and 1986-2014 had the lowest AUC score (0.954) (Figure 35). The response curves are similar among the three models, and, as in the separate time-series analysis, wave height had the highest percent contribution for all the models. For 1986-1996, turbidity had the next highest percent contribution, and for 1986-2006 and 1986-2014, depth had the next highest percent contribution, similar to the separate time-series analysis. In the jackknife analysis for 1986-1996, wave height and turbidity are about equally most important when alone in the model, and there is little change when any variable is removed from the model. For the 1986-2006 model, wave height is most important when alone in the model, and depth causes a decrease in likelihood when removed. For the 1986-2014 model, wave height is most important again, and wave height and depth cause a decrease in likelihood when removed. Overall, the results from the separate and cumulative time-series analyses were similar, with only minor differences in the relative rankings of the environmental variables. However, there were substantial differences in the AUC scores among the models (Figure 33 and 35). The separate time-series models all performed very well, but the cumulative models decreased in performance as data were added to subsequent models. One explanation for this finding is that as Ruffe spreads from west to east through time, it started to inhabit new locations that were deeper, with larger waves, and less turbid (Table 17), and the variability in the conditions increased, causing the AUC to decrease. For example, early in the invasion, perhaps it inhabited a drastically reduced subset of the current locations in terms of the overall distribution, thus also reducing the variability in the covariate needed to explain their habitat association. Through time, Ruffe has become widely distributed and has been captured in a wide array of locations (whether suitable habitat or not), increasing the variability in covariates and reducing the precision of the model.

## Conclusion

Overall, this study demonstrates the strong potential to apply Maxent at spatial scales that could be used in ecological risk assessment or monitoring design. I used lake-scale environmental and geographical data to produce a high-resolution (30 x 30 m grid) predictive distribution map for Ruffe in Lake Superior. However, my results demonstrate that there are some important considerations when developing a SDM at a lake-scale. I found multiple models performed similarly according to AUC scores but had ecologically different suitable habitat prediction maps. Indeed, relatively small differences in buffers (±100s of meters) resulted in differences of billions of square meters predicted. To identify the degree to which data aggregation effect model results, using multiple buffers is necessary. Additionally, with the time-series analysis, the AUC scores decreased as more data was added to the models through time; this suggests that the ability to predict suitable declines as an invasion progresses if presence data are combined through time. If only one data aggregation method is chosen or all of the data is used and autocorrelation is ignored, there is potential for substantial over- or under-prediction of suitable habitat in the model (Table 15).

# Dissertation Summary

## Chapter 1 summary

Invasive Ruffe (*Gymnocephalus cernua*) has caused substantial ecological damage in North America, parts of Western Europe, Scandinavian countries, and the United Kingdom (Maitland and East 1989; Adams and Tippett 1991; Selgeby and Edwards 1993; Adams 1994; Kalas 1995; Ogle *et al.* 1996; Selgeby 1998; Lorenzoni *et al.* 2009). Once it invades a new waterbody, it is nearly impossible to eradicate, in part, due to its adaptability. In each life stage, Ruffe exhibits plasticity with regard to chemical, physical, biological, and habitat requirements. Adult Ruffe has characteristics that allow it to adapt to a range of environments, including rapid maturation, relatively long life and large size (allowing it to reproduce many times in large batches), batch spawning, genotype and phenotype (having plasticity in their genetic expression), tolerance to a wide range of water quality, broad diet, and multiple dispersal periods. There is, however, variability among these characteristics between the native, non-native North American, and European non-native populations, which presents a challenge to managing populations based on life history characteristics. Monitoring and spread prevention strategies are important because, based on Ruffe's variable life history strategies and its recent range expansion, all of the Laurentian Great Lakes and many other water bodies in the UK, Europe, and Norway are vulnerable to Ruffe establishment.

## Chapter 2 summary

Invasive species often show a period of rapid initial increase (boom) followed by a population crash (bust) before achieving a relatively stable, equilibrium population size. My study was located in St. Louis River, MN/ WI, and Chequamegon Bay, WI. I used these two systems to compare because they were invaded by Ruffe at different time periods, and they have similar fish communities. The timeline of population growth and spread of an introduced species can be conceptualized as a series of invasion stages (Sakai *et al.* 2001; Colautti and MacIsaac 2004; Simberloff and Gibbons 2004). In stage 0, propagules of the introduced species are in the donor region; in stage 1, the introduced species is transported outside of its current range; in stage 2, individuals are released and introduced into a new region. In stage 3, the species becomes established, distributed in a small area and is numerically rare. In stage 4, the species' population is either spatially widespread but numerically rare, or localized but abundant. Finally, in stage 5, organisms are widespread and dominant (Colautti and MacIsaac 2004). Stage 4-5 is typically when the population "booms," and following stage 5, there is often a population crash that is referred to as the "bust."

I found that Ruffe populations in both the St. Louis River and Chequamegon Bay are at different invasion stages. In the St. Louis River, the population increased from the initial invasion in 1986 up to 1995 and has been in decline for the past two decades (1996-2015). In Chequamegon Bay, the overall population is increasing, but is doing so by oscillating every 5-7 years. I conclude that Ruffe populations in both systems partially conform to the typical "boom-bust" patterns seen with other invasive fish species. I also found many differences in the fish population trends, in addition to Ruffe, between the St. Louis River and Chequamegon Bay. Understanding patterns of invasive species can be helpful to natural resource managers who are interested in population trends of invasive species.

## Chapter 3 summary

Food webs have been greatly impacted by invasive species in ecosystems across the globe. My goal was to study the role of Ruffe in the St. Louis River food web, using carbon and nitrogen stable isotope analysis to characterize its associated trophic pathways (i.e., the various organic matter sources and associated habitats supporting Ruffe's diet). I found significant differences in $\delta^{13}C$ and $\delta^{15}N$ values between Ruffe captured in Lake Superior and those captured in the St. Louis River but not among locations within the river. I found size-based differences as well; medium-sized fish, 60-80 mm total length (SL), had a $\delta^{13}C_{lipid\ corrected}$ of about -25‰ to -45‰, lower than either small (<60 mm SL) or large (80-148 mm SL) Ruffe (-38.2‰ to -14.2‰). Extremely depleted $^{13}C$ values (<-36‰ $\delta^{13}C$) indicate that some fish captured within coastal wetlands were feeding in a methane-based trophic pathway. The high $\delta^{13}C$ values of both small and large Ruffe indicate these fish were both swimming and feeding in Lake Superior; the higher values of medium size Ruffe suggest coastal wetland dependence during the spawning period. The broad range in $\delta^{13}C$ values of large Ruffe indicate routine occupancy of both lake and wetland habitats; 59.7% of individuals were predominantly feeding in a wetland-dominated trophic pathway, whereas 40.3% were feeding in a lake-dominated trophic pathway. This is the first observation of wetland fish obtaining substantial energy from a methane-based food web, as well as the first observation of distinct, size-based diet shifts and movements among coastal habitats in Ruffe. This indicates Ruffe has the ability to occupy a unique trophic niche within coastal wetlands, and it is an obligate user of wetland habitat during spawning but otherwise a facultative user of lake and wetland habitat.

## Chapter 4 summary

My goal was to understand how using environmental data resolved to relatively fine spatial scales (i.e., 100m to 1000 m), as well as using different species occurrence data of varying temporal windows, would affect species distribution model performance with respect to predicting potential habitat of an invasive fish, Ruffe. I used Lake Superior catch data and environmental variables to develop a Maxent species distribution model. To examine the effect of spatial data resolution, I developed and compared competing models at different spatial scales: 250-m, 500-m, 1000-m, 2000-m, and 2000-m selected model. In addition, I conducted two time-series analyses, comparing models developed from occurrence data broken into decade time blocks (1986-1996, 1997-2006, 2007-2014) and analyzed separately or cumulatively. I calculated percent suitable habitat for all of the models. I predicted that there would be an optimal spatial scale to model Ruffe—that very low- and very high-spatial scale models would not perform well, but an intermediate model would be the best. Among the models constructed using catch and environmental data from various spatial resolutions, the best performing model used 500 m data and the worst performing model used 2000 m data. The important geographic discrepancies in potential habitat occurred around the Apostle Islands, WI, Isle Royale, MN, Grand Marais, MI, Whitefish Point, MI, and Red Rock and Nipigon in Canada. I showed multiple models that performed similarly according to the area under the curve (AUC) scores, but had different results with respect to the area and distribution of suitable habitat predicted. The Maxent model results from the separate and cumulative time-series analyses were similar. I found minor differences in the environmental variable outputs. However, I found substantial differences in the AUC scores. The separate time series models all performed similarly well, but the performance of the cumulative models declined as data were added to subsequent models. Maxent can be a powerful tool to model invasive species, using the precautions I provide.

## Synthesis

Species distribution models produce a variety of maps with varying accuracy. Statistically, these maps are “accurate” and illustrate “suitable” habitat based on occurrence and environmental data. However, it is worthwhile to critically evaluate the model output from a biological perspective. For example, some of the models predicted suitable habitat around the entire Isle Royale, a large island in the northwest corner of Lake Superior about 38.9 km from the Minnesota shore. Isle Royale is a rocky island with a few small wetlands. It has very little sheltered habitat. The question arises as to whether the habitat is suitable based on the biology and ecology of Ruffe.

Based on the results presented in Chapter 1, Ruffe has specific life-history requirements that might prevent it from establishing a population on Isle Royale. Isle Royale is in the north shore of Lake Superior. It is very rocky, and conditions can be harsh. It has 19 embayments surrounding the island adjacent to Lake Superior (Gorman *et al.* 2008). Average depth in the embayment is highly variable, but the mean is 1.28 m (Gorman *et al.* 2008). In Lake Superior, Ruffe has been captured from 0.2-205 m (USGS, personal comm., 2014), but they prefer deep, dark habitats (Gutsch and Hoffman 2016). Substrate is between silty sand and bedrock but predominantly bedrock (Gorman *et al.* 2008). Ruffe prefers sandy, silty, well-aerated, slow-moving water with little or no vegetation (Kontsevaya and Frantova 1980; Popova *et al.* 1998; Ogle 1998). There is very little organic matter, only 5%. Of the existing organic matter, 60% of it is woody debris (Gorman *et al.* 2008). Organic matter is very important for Ruffe because it provides nutrient-rich food for invertebrates (Pinder 1995), so Ruffe can feed on them (Ogle *et al.* 1995). Very little of the shoreline is protected by wave action and ice scouring, which is why there is so little organic matter. The wave action has also caused there to be a lack of overhead shade, logs, and emergent vegetation (Gorman *et al.* 2008). Ruffe requires protected, slow-moving water in which to reside (Gutsch and Hoffman 2016). It lives in waters ranging from oligotrophic to eutrophic but prefer eutrophic waters (Fedorova and Vetkasov 1974; Disler and Smimov 1977; Leach *et al.* 1977; Hansson 1985; Johansson and Persson 1986; Bergman 1988a, 1990, 1991; Bergman and Greenberg 1994; Rösch *et al.* 1996; Popova *et al.* 1998; Lehtonen *et al.* 1998; Brown *et al.* 1998). Although some of the embayments and wetlands on Isle Royale might be eutrophic, most of them are likely lake influenced so they are probably more oligotrophic. It prefers turbid, dark conditions because Ruffe possesses a tapeta lucidum and sensitive lateral line systems, allowing it to forage in low-light conditions (Hölker and Thiel 1998).

In addition to the life history characteristics being a barrier to establishing a population on Isle Royale, there are also movement barriers (Chapter 3). Based on stable isotope findings, when Ruffe is small (25-65 mm), it uses both the lake and the wetland to feed, but when it is 65-85 mm, it is restricted to the wetland. Thus, there is limited opportunity to complete its life cycle on or around Isle Royale, which only has a few, small coastal wetlands. Once the fish is greater than 85 mm, it disperses again to lake and wetland. Although Ruffe could go out to Isle Royale, as the SDM predicts, based on what I know about movements and biology, it is very unlikely that Ruffe would establish a population out there.

Other inconsistencies between the ecology of Ruffe and the SDM model outputs arose. Notably, the SDM predicted that the St. Louis River is a suitable Ruffe habitat; however, based on the population dynamics modeling (Chapter 2), the Ruffe population is exponentially decreasing in the St. Louis River and has been since 1995. Given the population is declining, this could indicate the habitat is not suitable for Ruffe though Ruffe remain abundant, despite the significant decline of Ruffe in the St. Louis River. At present, the St. Louis River catch per unit effort (CPUE) is comparable to Ruffe CPUE in Chequamegon Bay. Given their known life history characteristics and habitat preferences (i.e., turbid, deep, cool, organic, sandy, silty substrate, slow-moving water, and shelter). Based on the scientific literature, the St. Louis River is high-quality habitat for Ruffe (Chapter 1).

Further, the SDM indicates that both Chequamegon Bay and the St. Louis River are equally suitable habitats for Ruffe, though there are ecologically-relevant differences in habitat quality. Coastal wetlands are less extensive in Chequamegon Bay than in the St. Louis River, the open water is turbid (30.6 NTU (16.8 SD)), cool ($23^{o}C$) (Hoffman *et al.* 2015), deep (maximum depth = 23 m) (Bronte *et al.* 1998), and the bottom is mostly clay, sand, and silt. Because of the substrate and the cold water, the abundance of benthic invertebrates is lower than in coastal wetlands (DUAN *et al.* 2009). The Ruffe caught in Chequamegon Bay are small to medium in size (<100 mm). The water in the St. Louis River is also turbid (67.8 (30.1 SD)), warmer ($29^{o}C$) (Hoffman *et al.* 2015), shallower (maximum depth = 16 m) (Angradi *et al.* 2015), and the bottom is mostly organic matter because it is a drowned river mouth. Because of the warmer water and the productive organic matter substrate, benthic invertebrates are abundant in the St. Louis River (DUAN *et al.* 2009). As a result, there is high-quality foraging fish habitat. A SDM that incorporates this more extensive habitat quality data could possibly distinguish the suitability of these two systems but georeferenced data to support such a SDM are not available, and these two systems are among the most well-studied in Lake Superior.

Species distribution models can produce maps of suitable habitat for invasive species to help predict introduction, spread, and movement. However, when using finely-scaled (i.e., lake scale) environmental variables, one must use caution when examining the results, both from a management and a biological perspective. I found a variety of inconsistencies between the SDM model output and the biological traits of Ruffe.  From this study, I conclude that biological data (e.g., habitat preference, environmental tolerance, life history) are needed along with model performance statistics to evaluate model success from plausibility. Such evaluation should be ongoing and iterative, because as new detections of an invasive species arise, both the predicted suitability will and the SDM accuracy will change. This challenge highlights the fundamental challenge to predicting invasive species habitat – that both the species and its new environment will change through time, yielding a dynamic understanding of suitable habitat. In this context, it is necessary to continually study an invasive species habitat preferences and distribution to obtain the most accurate depiction of the animal’s suitable habitat or fundamental niche.

# Illustrations

## Tables

Table 1. Life history traits of Ruffe (Gymnocephalus cernua) throughout each main life stage. 1. (Collette *et al.* 1977); 2. (Balon *et al.* 1977); 3. (Kolomin 1977); 4. (Lorenzoni *et al.* 2009); 5. (Neja 1988); 6. (Kovac 1998); 7. (Maitland 1977); 8. (Craig 1987); 9. (Kiyashko and Volodin 1978); 10. (Brown et al. 1998); 11. Fedorova and Vetkasov, 1974; 12. (Popova *et al.* 1998); 13. (French and Edsall 1992); 14. (Disler and Smimov 1977); 15. (Hokanson 1977); 17. (Eckmann 2004); 18. (Selgeby 1998); 19. (Kangur *et al.* 1999); 20. (Jamet and Lair 1991); 21. (Ogle *et al.* 1995); 22. (Lind 1977); 23. (Van Densen and Hadderingh 1982); 24. (Nilsson 1979); 25. (Sandlund et al. 1985); 26. (USGS, personal comm. 2014); 27. (Volta *et al.* 2013); 28. (Kalas 1995); 29. (Bastl 1988).

| Stage | Habitat | Size (mm) | Duration | Diet | Movements | Depth (m) | Temperature (ºC) | Special Requirements |
|---|---|---|---|---|---|---|---|---|
| **Egg** | On bottom attached to sand, gravel, clay[1], plants, branches, rocks, or logs[2] | 0.50-1.0[1]; 0.90-1.213; 0.71-1.59[4]; 0.64–0.98[5]; 0.97-1.07[6] | 5-12 days[,7,8], 4-6 days[2,6] | Yolk | Stationary | ≤ 5[9] | 5 – 18[10] | pH=6.5-10.5[9] |
| **Larvae** | On bottom at spawning grounds[14] | 3.35-4.40 at hatch[6,11], 6-8 one week after hatch[12], 16-18 end of larval stage[12] | 20 days[6] | Yolk sac, exogenous[13] | Stationary[14], passive drift | 0.5-5[10] | 16.2 to 23[6], growth optimum: 25-30[15] | |
| **Juvenile** | Benthic littoral[6,10] | 14-110[6,8] | 28 days post-hatch[6] | Mainly benthic invertebrates[10,12] | Diel feeding, possibly migrating to overwintering grounds[17,18] | 0-15+[18] | ≥0.2[19], thermal max: 34.5[15] | Apparently feeds at overwintering habitat[18] |
| **Adult** | Benthic, sandy, silty areas[12,20,21] | 57-90[29], 110+[22] | 2-3 years (some systems | Mainly benthic invertebrates, | Diel feeding, migration to spawning | 0.2-205.0[17,23,24,25,26,27,28], | ≥0.2[19], spawning | Apparently feeds at |

| | | | | | | |
|---|---|---|---|---|---|---|
| | Ruffe can mature in 1 year)[22] | some zooplankton | grounds, migration to overwintering grounds[17,18] | spawning grounds < 3[9], overwintering grounds 15+[17,18] | grounds are 5-18[10] | overwintering habitat[18] |

Table 2. Differences between native, non-native North American, and other non-native Ruffe (*Gymnocephalus cernua*) populations with respect to habitat usage, depths inhabited, feeding habits, age and size at maturity, maximum age and length acquired, and reproduction (if batch spawning occurs). 1. (Hölker and Thiel 1998); 2. (Pratt 1988); 3. (Sierszen *et al.* 1996); 4. (Fairchild and McCormick 1996); 5. (Brown *et al.* 1998); 6. (Stepien *et al.* 1998); 7. (Selgeby 1998); 8. (Ogle *et al.* 2004); 9. (Ogle 2009); 10. (Peterson *et al.* 2011); 11. (USGS 2014); 12. (USEPA, personal comm. 2006-2007); 13. (USFWS, personal comm. 2014); 14. (USGS, personal comm. 2014); 15. (Maitland and East 1989), 16. (Eckmann 2004); 17. (Volta *et al.* 2013); 18. (Wootten 1974); 19. (Winfield *et al.* 2004); 20. (Kalas 1995); 21. (Lorenzoni *et al.* 2009); 22. (Duncan 1990); 23. (Nilsson 1979); 24. (Van Densen and Hadderingh 1982); 25. (Sandlund *et al.* 1985); 26. (Johnsen 1965); 27. (Polivannaya 1974); 28. (Kozlova and Panasenko 1977); 29. (Boikova 1986); 30. (Nagy 1988); 31. (Jamet and Lair 1991); 32. (Kangur and Kangur 1996); 33. (Werner *et al.* 1996); 34. (Kangur *et al.* 2000); 35. (Ogle *et al.* 1995); 36. (Adams and Tippett 1991); 37. (Neja 1988); 38. (Fedorova and Vetkasov 1974); 39. (Craig 1987); 40. (Lappalainen and Kjellman 1998); 41. (Lind 1977); 42. (Ogle 1998); 43. (Maitland 1977).44. (Bastl 1988); 45. (Kolomin 1977); 46. (Devine *et al.* 2000); 47. (Crosier and Molloy 2007); 48. (Popova *et al.* 1998); 49. (Berg 1949); 50. (Sanjose 1984); 51. (Koshelev 1963); 52. (Hokanson 1977).

| | **Native** | **Non-native North America** | **Non-native Europe** |
|---|---|---|---|
| **Habitats** | Lakes, rivers, ponds, bays, brackish waters, tidal coastal wetlands, non-tidal coastal wetlands, and reservoirs[1] | Lakes, rivers, non-tidal coastal wetlands[2,3,4,5,6,7,8,9,10,11,12,13,14] | Lakes, reservoirs[15,16,17,18,19,20,21,22] |
| **Depths** | 0.25 - 85 m[23,24,25] | 0.2 - 205 m[14] | 4.9[20] - 50 m[17], 30 - 70 m[16] |
| **Feeding Habits** | Primarily chironomid larvae or pupae[26,27,28,29,30,31,32,33,34] | Zooplankton (age-0); caddisflies and burrowing mayflies (>12 cm Ruffe)[35]; *Mysis* shrimp and Cisco eggs (winter diet)[7] | Loch Lomond, Scotland- Powan ova and caddisflies[36]; Lake Mildevatn, Norway- zooplankton[20] |
| **Age at maturity** | 1-4 years[31,37,38,39]; Finland- <1[40,41]; Nadym basin- mainly 3-4[444] | 2-3 years[42] | 2-3 years[42,43];Lake Piediluco, Italy- age 1 for both sexes[21] |
| **Size at maturity** | Females: 57-90 mm, Males: 80+ mm[44]; Nadym basin- 20-30 g and 110-120 mm[45] | 110-120 mm[42] | Loch Lochmond, Scotland- ~11.67 g/ ~7.5 g[46]; Britain/ Europe- 110-120 mm[42,52]; Lake Piediluco, Italy- females: 78.74±0.83 mm, males: 69.42±1.91 mm[21] |
| **Maximum age** | Females: 11 yrs, males: 7 yrs[41,47,43]; Bay of Ob' River, Russia- 20 yrs.[48]; 8-10[48] | 10 years | 8-10[47]; 6 years [21] |

| | | | |
|---|---|---|---|
| **Maximum length** | 290 mm[1]; Siberia- 500 mm[49] (unconfirmed)[50], 200 mm[41] in Finland | 184 mm[13] | Lake Constance, Germany- 124 mm[16]; Lake Piediluco, Italy – 191 mm, 141 g[21] |
| **Reproduction** | Multiple clutches of eggs throughout spawning season[38,45,51,52] | Prolonged spawning season but no evidence of multiple clutches[5] | Lake Piediluco, Italy- no information about batch spawning but small relative and absolute fecundities[21] |

Table 3. Data comparison by location and vessel for the St. Louis River (SLR), WI/ MN, USA, and Chequamegon Bay (CB), WI, USA, for 1993-2015. Game fish are potential predators: Walleye, Smallmouth Bass, Muskellunge, and Northern Pike. Forage fish include Ruffe, as well as Emerald Shiner, Round Goby, Spottail Shiner, Johnny Darter, Trout Perch, and Yellow Perch.

| Date | Data Type | Location | Survey Type | Vessel/Gear | Vessel Sweep (hec/hr) | Agency |
|---|---|---|---|---|---|---|
| 1993-2015 | Game Fish | SLR | Gill net | NA | NA | MN DNR |
| 1993-2015 | Game Fish | CB | Creel survey | NA | NA | WI DNR |
| 2006, 2007, 2010-2015 | Forage Fish | SLR | Bottom trawl | 4.9 m otter trawl | 0.7964 | USWFS, USEPA, 1854 Treaty Authority |
| 1993-1999 | Forage Fish | CB/ SLR | Bottom trawl | R/V *Kiy*i or R/V *Grayling*- 11.9 m trawl | 2.05 | USGS, USFWS |
| 1993-2004 | Forage Fish | CB/ SLR | Bottom trawl | USGS *R/V* Coaster- 4.9 m trawl | 0.785 | USGS, USFWS |

Table 4. Pearson correlation matrix for fish in the St. Louis River, MN/ WI, USA, from 1993-2015. Above the shaded region are the r values and below the shaded region are the p values. The bolded r values are classified “strong” or “very strong” correlations according to Evans (1996). The bolded p values are those considered significant (<0.05). Fish species are abbreviated as follows: RUF = Ruffe, EMS = Emerald Shiner, JOD = Johnny Darter, ROG = Round Goby, SPS = Spottail Shiner, TRP = Trout Perch, YEP = Yellow Perch, MUS = Muskellunge, SMB = Smallmouth Bass, NOP = Northern Pike, WAL = Walleye.

| | RUF | EMS | JOD | ROG | SPS | TRP | YEP | MUS | SMB | NOP | WAL |
|---|---|---|---|---|---|---|---|---|---|---|---|
| RUF | | -0.517 | 0.277 | -0.255 | 0.269 | **-0.604** | **0.644** | 0.014 | 0.380 | 0.540 | -0.033 |
| EMS | **0.012** | | -0.369 | 0.019 | 0.046 | 0.427 | -0.466 | 0.136 | -0.130 | -0.078 | 0.294 |
| JOD | 0.201 | 0.083 | | 0.597 | 0.455 | 0.200 | -0.042 | -0.082 | 0.243 | -0.342 | 0.017 |
| ROG | 0.240 | 0.930 | **0.003** | | 0.496 | 0.490 | -0.433 | -0.103 | 0.365 | -0.483 | 0.478 |
| SPS | 0.215 | 0.834 | **0.029** | **0.016** | | 0.229 | 0.046 | 0.195 | 0.203 | 0.085 | **0.617** |
| TRP | **0.002** | **0.042** | 0.361 | **0.018** | 0.293 | | -0.326 | 0.048 | -0.359 | -0.522 | 0.166 |
| YEP | **0.001** | **0.025** | 0.851 | **0.039** | 0.835 | 0.129 | | 0.177 | 0.076 | **0.603** | -0.289 |
| MUS | 0.948 | 0.537 | 0.708 | 0.639 | 0.374 | 0.829 | 0.419 | | -0.267 | 0.189 | 0.254 |
| SMB | 0.074 | 0.556 | 0.264 | 0.087 | 0.352 | 0.093 | 0.731 | 0.218 | | 0.121 | 0.221 |
| NOP | **0.008** | 0.722 | 0.110 | **0.020** | 0.699 | **0.011** | **0.002** | 0.387 | 0.581 | | 0.040 |
| WAL | 0.881 | 0.173 | 0.939 | **0.021** | **0.002** | 0.448 | 0.182 | 0.242 | 0.311 | 0.855 | |

Table 5. Pearson correlation matrix for fish in Chequamegon Bay, WI, USA, from 1993-2015. Above the shaded region are the r values and below the shaded region are the p values. The bolded r values are those classified as “strong” or “very strong” correlations according to Evans (1996). The bolded p values are those considered significant (below 0.05). Fish species abbreviations follow Table 4.

| | RUF | EMS | JOD | SPS | TRP | YEP | NOP | WAL |
|---|---|---|---|---|---|---|---|---|
| RUF | | 0.272 | 0.283 | 0.239 | 0.130 | 0.275 | -0.312 | -0.016 |
| EMS | 0.209 | | 0.551 | **0.829** | 0.419 | 0.517 | -0.018 | 0.237 |
| JOD | 0.190 | **0.006** | | **0.650** | 0.047 | **0.749** | -0.358 | -0.112 |
| SPS | 0.273 | **0.000** | **0.001** | | 0.504 | **0.635** | -0.205 | 0.050 |
| TRP | 0.555 | **0.047** | 0.830 | **0.014** | | 0.053 | 0.068 | 0.180 |
| YEP | 0.205 | **0.011** | **0.000** | **0.001** | 0.809 | | -0.544 | -0.270 |
| NOP | 0.147 | 0.934 | 0.094 | 0.348 | 0.759 | **0.007** | | 0.564 |
| WAL | 0.942 | 0.276 | 0.612 | 0.819 | 0.410 | 0.213 | **0.005** | |

Table 6. Univariate linear models of Ruffe (*Gymnocephalus cernua*) CPUE over 23 years in the St. Louis River, MN/ WI, USA, from 1993-2015 ranked by Akaike Information Criteria (AIC). For each model, ln(Ruffe CPUE + 1) is the response variable, k is the number of estimable parameters, including the intercept, and the parameter listed is the predictor variable. Corrected AIC ($AIC_c$) was used to account for my small sample size. Δ $AIC_c$ is the difference in $AIC_c$ from the smallest $AIC_c$ (0.00 is the smallest). $AIC_c$ weight represents conditional probabilities for each model.

| Models | k | Log-likelihood | $AIC_c$ | $\Delta AIC_c$ | $AIC_c$ weight ($w_i$) | Interaction |
|---|---|---|---|---|---|---|
| **Yellow Perch** | 2 | -25.45883 | 57.52 | 0.00 | 0.62 | Competitor |
| **Trout Perch** | 2 | -26.41271 | 59.43 | 1.91 | 0.24 | Competitor |
| **Northern Pike** | 2 | -27.65816 | 61.92 | 4.40 | 0.07 | Predator |
| **Emerald Shiner** | 2 | -28.04333 | 62.69 | 5.17 | 0.05 | Competitor |
| **Smallmouth Bass** | 2 | -29.82926 | 66.26 | 8.74 | 0.01 | Predator |
| **Null (intercept only)** | 1 | -31.62196 | 67.43 | 10.33 | 0.00 | NA |
| **Johnny Darter** | 2 | -30.70602 | 68.01 | 10.49 | 0.00 | Competitor |
| **Spottail Shiner** | 2 | -30.75934 | 68.12 | 10.60 | 0.00 | Competitor |
| **Round Goby** | 2 | -30.84895 | 68.30 | 10.78 | 0.00 | Competitor |
| **Walleye** | 2 | -31.60948 | 69.82 | 12.30 | 0.00 | Predator |
| **Muskellunge** | 2 | -31.61955 | 69.84 | 12.32 | 0.00 | Predator |

Table 7. Parameter estimates for the top four models that explain 99% of the model weights for predicting Ruffe (*Gymnocephalus cernua*) densities in the St. Louis River, MN/ WI, USA, from 1993-2015 (CI = confidence interval). Parameter estimates with intercepts can be found in the supplemental tables.

| Parameter or Variable | Lower 95% CI | Parameter Estimate | Upper 95% CI | P value |
|---|---|---|---|---|
| Yellow Perch | 0.320 | 0.650 | 0.979 | 0.001 |
| Trout Perch | -0.920 | -0.588 | -0.256 | 0.002 |
| Northern Pike | 0.550 | 1.649 | 2.749 | 0.010 |
| Emerald Shiner | -0.862 | -0.505 | -0.147 | 0.010 |

Table 8. Univariate linear models of Ruffe (*Gymnocephalus cernua*) CPUE over 23 years in Chequamegon Bay, WI, USA, from 1993-2015 ranked by Akaike Information Criteria (AIC). For every model, ln(Ruffe CPUE + 1) is the response variable, k is the number of estimable parameters, including the intercept, and each parameter listed is the predictor variable. Each predictor variable for all models was natural log transformed. Corrected AIC ($AIC_c$) was used to account for my small sample size. $\Delta AIC_c$ is the difference in $AIC_c$ from the smallest $AIC_c$ (0.00 is the smallest). $AIC_c$ weight represents conditional probabilities for each model. There were no significant variables—all of them encompass zero.

| Models | k | Log-likelihood | $AIC_c$ | $\Delta AIC_c$ | $AIC_c$ weight ($w_i$) | Interaction |
|---|---|---|---|---|---|---|
| **Null (intercept only)** | 1 | -46.07433 | 96.34 | 0.00 | 0.18 | NA |
| **Northern Pike** | 2 | -44.89592 | 96.39 | 0.05 | 0.18 | Predator |
| **Johnny Darter** | 2 | -45.11274 | 96.83 | 0.49 | 0.14 | Competitor |
| **Yellow Perch** | 2 | -45.17265 | 96.95 | 0.61 | 0.13 | Competitor |
| **Emerald Shiner** | 2 | -45.19094 | 96.98 | 0.64 | 0.13 | Competitor |
| **Spottail Shiner** | 2 | -45.39926 | 97.40 | 1.06 | 0.11 | Competitor |
| **Trout Perch** | 2 | -45.87891 | 98.36 | 2.02 | 0.07 | Competitor |
| **Walleye** | 2 | -46.07135 | 98.74 | 2.40 | 0.05 | Predator |

Table 9. Summarized sampling methods for Ruffe (*Gymnocephalus cernua*) from 2014-2015. Sampling was completed from Summer 2014 – Summer 2015 by various methods of capture.

| **Season of Sampling** | **Year of Sampling** | **Survey Gear and Specs** | **Vessel** | **Method of Capture** | **Survey Design** | **Number of Fish Captured** |
|---|---|---|---|---|---|---|
| Summer | 2014 | 18'-otter trawl, 5 minutes | FWS small trawling boat | Bottom trawl | 1854 Treaty Authority: random three strata (depth ranges that cover dredged, original river channels, shallow/ floodplains), 40/ yr. USFWS: (Stevens, L and Olsen 1999; Stevens and Olsen 2004), 10 SLR, 15 Cheq. Bay/ yr. | 221 |
| Fall | 2014 | 18'-otter trawl, 5 minutes in SLR, 10 min in Lake Superior | R/V Blue Heron | Bottom trawl | Random 9 SLR and 5 Lake Superior based on Selgeby (1998) | 26 |
| Winter | 2014-2015 | Angler fishing | N/A | Fishing pole | Opportunistic citizen science fish collection project to collect Ruffe in the SLR and Cheq. Bay from ice anglers, anglers instructed to collect Ruffe and place them in the bag and mark on the map where they caught them | 34 |
| Spring | 2015 | Fyke nets (4.76 mm mesh, front opening 0.9m x 1.2m, lead-to-lead length 15 m), 12 hr overnight set | Mudlark (small EPA vessel) | Fyke nets | Non-randomly chose 4 locations in Superior Bay, St. Louis Bay, and Allouez Bay during spring spawning, set paired fyke nets parallel to shore | 0 |
| Summer | 2015 | 18'-otter trawl, Fyke nets, Windermere traps | FWS small trawling | Bottom trawl, fyke nets, | Non-random; selected habitat that was gear-appropriate in Amnicon River, Brule River, Flag River | 51 (bottom trawling), 0 fyke and Windermere |

| | | | |
|---|---|---|---|
| (length 1.22 m, width 0.71 m, diameter 0.60 m, conical entrances 50.8 mm | boat, Mudlark | Windermere traps | complex, Bark Bay Slough using 8 paired fyke nets and 4 baited Windermere traps |

Table 10. Summary of stable isotope data. Data was collected and analyzed between the summer of 2014 and the summer of 2015. We used $\delta^{13}C_{lipidcorrected}$ values. SL is standard length. SD is standard deviation. 25 and 75 quartiles are the 25th and 75th percentile, respectively.

| Year | Area | Location | Mean (SD) $\delta^{13}C$ lipid corr | Range $\delta^{13}C$ lipid corr | Median (25th -75th Quartiles) $\delta^{13}C$ lipid corr | Mean (SD) $\delta^{15}N$ | Range $\delta^{15}N$ | Median (25th -75th Quartiles) $\delta^{15}N$ | Mean SL (mm) | Mean Weight (g) | Range C:N | Sample Size |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 2014 | Cheq. Bay | Cheq. Bay | -21.0 (3.1) | -28.0 - -17.2 | -20.5 (-22.6 - -18.4) | 4.9 (0.9) | 4.0 – 7.0 | 4.6 (4.2 - 5.7) | 63.9 | 8.9 | 3.7 – 4.2 | 10 |
| 2015 | Cheq. Bay | Cheq. Bay | -19.8 (1.8) | -29.8 - -16.8 | -19.8 (-20.5 - -18.9) | 6.3 (0.5) | 5.2 – 7.7 | 6.3 (6.0 – 6.6) | 48.4 | 2.6 | 3.6 – 6.5 | 53 |
| 2014-2015 | Cheq. Bay | Cheq. Bay | -18.8 (2.4) | -22.8 - -14.2 | -18.8 (-19.6 - -17.4) | 6.1 (0.4) | 5.5 – 6.9 | 6.1 (5.8 – 6.4) | 108.4 | 29.6 | 3.7 – 4.2 | 11 |
| 2014 | Lake Superior | Lake Superior | -19.8 (0.2) | -19.9 - -19.7 | -19.8 | 8.0 (0.02) | 8.0 – 8.0 | 8.0 | 29 | 0.4 | 3.8 – 4.3 | 2 |
| 2014-2015 | Lake Superior | Lake Superior | -18.1 | NA | NA | 6.6 | NA | NA | 127 | 49.1 | 4.0 | 1 |
| 2014 | Lower | Lower St. Louis Bay | -32.5 (5.1) | -45.0 - -23.9 | -32.3 (-36.7 - -28.5) | 8.9 (1.3) | 5.8 – 12.2 | 8.9 (8.0 – 9.7) | 83.1 | 16.6 | 3.5 – 4.6 | 75 |
| 2014 | Lower | Pokegama Bay | -35.0 (2.5) | -37.7 - -31.6 | -35.3 (-37.1 - -32.5) | 7.4 (0.7) | 6.5 – 7.9 | 7.6 (6.7 – 7.9) | 43 | 1.9 | 3.6 – 3.8 | 4 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 2014 | Lower | Superior Bay | -31.5 (4.2) | -48.0 - -22.3 | -31.2 (-33.9 - -28.6) | 9.9 (2.0) | 4.0 – 16.1 | 9.8 (9.0 – 11.2) | 67.7 | 8.7 | 3.7 – 4.3 | 72 |
| 2014 | Lower | Upper St. Louis Bay | -32.4 (6.6) | -40.3 - -24.4 | -32.0 (-39.7 - -26.0) | 8.1 (1.4) | 6.7 – 10.6 | 7.7 (7.2 – 9.1) | 78 | 11.3 | 3.7 – 3.8 | 6 |
| 2014-2015 | Lower | SLR Winter | -32.5 (4.8) | -36.7 - -26.4 | -35.2 (-36.4 - -27.3) | 9.3 (2.0) | 7.3 – 12.3 | 9.1 (7.5 – 11.2) | 104.2 | 28.7 | 3.7 – 4.2 | 5 |
| 2014-2015 | Unspec. SLR | Unspec. SLR | -20.1 (4.0) | -31.8 - -16.3 | -18.7 (-21.6 - -17.5) | 5.8 (0.7) | 4.1 – 6.8 | 5.9 (5.4 – 6.3) | 113.8 | 39.4 | 3.6 – 4.3 | 16 |
| 2014 | Upper | Spirit Lake | -38.0 (6.3) | -52.4 - -25.8 | -37.4 (-41.7 - -33.8) | 7.3 (1.8) | 2.8 – 9.8 | 7.7 (6.4 – 8.5) | 88.6 | 17.4 | 3.5 – 5.6 | 43 |

Table 11. Description of occurrence data for Maxent model. Data ranged from 2005-2015 and came from literature or agencies. Data was collected a variety of ways (Gear).

| Organization/ Source | Years | Design | Gear | Habitat | Location Sampled | Number of Occurrence Points |
|---|---|---|---|---|---|---|
| Peterson et al. 2011, Lindgren et al. 2006 | 2007 | Total catch | Fyke net, electrofish, trawl, seine | Coastal wetland | Lower St. Louis River, MN | 1 |
| USEPA 2006-2007 | 2006-2007 | Total catch | Fyke net, electrofish, trawl, seine | Coastal wetland | St. Louis River estuary, MN | 109 |
| USFWS, personal comm. 2014-2015 | 2005-2014 | Total catch | Bottom trawl, fyke net | Rivers and coastal wetlands | Amnicon River, Flag River, Iron River, Marquette Lower Harbor, Ontonogon river, Pike Bay, Grand Marais, Keeweenaw Lower Entry, Portage Lake, St. Louis River, Chequamegon Bay, Thunder Bay | 180 |
| USFWS/ USEPA 2008 | 2008 | Total catch | Bottom trawl, fyke net | Coastal wetland | St. Louis River estuary, MN | 23 |
| USGS 2014 | 2005-2011 | Total catch/ sightings | NA | NA | Amnicon River, WI, Beartrap-Nemadji, WI, Kaministiquia River, Ontario, Canada, West bay at Grand Marais, MI, Little Lake, MI, Misery River, Keweenaw Peninsula, MI, Sturgeon River Slough, Keweenaw Peninsula, MI, Squaw Bay, Beartrap-Nemadji, WI, St. Louis River/ Estuary/ Bay/ Harbor, MN/ WI, Chequamegon Bay, WI, near Tahquamenon Bay, MI, north of Whitefish Point, MI | 11 |
| USGS, personal comm. 2014-2015 | 2005-2014 | Total catch | Bottom trawl | Lakes, wetlands, rivers | Chequamegon Bay, Preq I Bay- Stockton Isl, Bear Island, Superior Entry, Apostles inshore, E. Madeline Island, Is Royale (Lk. Desor Reef), Mawikwe (Squaw) Pt., Port Wing, Duluth-Superior grid 1402, | 64 |

Tahquaenon Is, Whitefish Pt., Basswood Island, NE
Herbster (Bark Point), Raspberry Island (PT.DET),
Lake Superior, MN, USA/ Canada

Table 12. Description of environmental data for Maxent model. There are five environmental layers, four came from the Great Lakes Aquatic Habitat Framework (GLAHF). Data was collected and analyzed a variety of ways. For the best description of the data, see the metadata on the GLAHF and MIT websites.

| Data | Source | Resolution, as obtained | Years/ Seasons/ Dates | Link to data | Link to meta data |
|---|---|---|---|---|---|
| Turbidity | Michigan Tech Research Institute | 1 km x 1 km | 2010-2013/ June-August | http://www.mtri.org/ | http://spatial.mtri.org/static/greatlakeswaterquality/ |
| Depth | GLAHF | 30 m x 30 m | See metadata | https://www.glahf.org/ | Metadata is in the link to data |
| Substrate | GLAHF | 30 m x 30 m | 1968-present | https://www.glahf.org/ | Metadata is in the link to data |
| Wave height | GLAHF | 30 m x 30 m | Hourly time step, 1979-2012 | https://www.glahf.org/ | Derived from GLAHF-interpolated using Arc software |
| Distance to wetland | GLAHF | 30 m x 30 m | See metadata | https://www.glahf.org/ | Calculated using the GLAHF “Coastal Wetlands” data-metadata within |

Table 13. Number of points for Maxent modeling for all six models. We applied distance buffers to remove clustering of occurrences points. This tables shows the number of points that resulted from the cluster-removal and ended up in each of the six models, as well as the resulting background points assigned by the Maxent program.

| | # occurrences | # training data | # test data | # points used to determine Maxent distribution (background and presence points) | # points in CB | # points in SLR |
|---|---|---|---|---|---|---|
| All data | 362 | 254 | 108 | 10249 | 74 | 194 |
| 250 m | 233 | 164 | 69 | 10162 | 53 | 127 |
| 500 m | 168 | 118 | 50 | 10117 | 44 | 76 |
| 1000 m | 109 | 77 | 32 | 10075 | 34 | 36 |
| 2000 m | 69 | 49 | 20 | 10047 | 19 | 15 |
| 2000 m selected removal | 129 | 91 | 38 | 10089 | 20 | 15 |

Table 14. Percent area predicted from buffer and from Lake Superior, as well as for different zones from the Maxent model for all six models using a logistic threshold at maximum test sensitivity plus specificity. Maximum test sensitivity plus specificity is an output from Maxent used as a threshold so everything above it was suitable habitat and everything below it was unsuitable habitat. I calculated percentages from the Maxent output predictive maps.

| | Percent area predicted from buffer | Percent area predicted from Lake Superior | Depth (m) | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | In-shore <30 | | Nearshore <100 | | Offshore >100 | |
| | | | Percent of Buffer | Area ($km^2$) | Percent of Buffer | Area ($km^2$) | Percent of Buffer | Area ($km^2$) |
| **Full adult model** | 14% | 5.76% | 13.22% | 4330 | 1.00% | 329 | 0.21% | 68.7 |
| **250 m** | 6% | 2.38% | 5.93% | 1940 | 0.020% | 6.50 | | |
| **500 m** | 8% | 3.45% | 8.49% | 2780 | 0.16% | 51.8 | 2.75e-6% | 0.0009 |
| **1000 m** | 20% | 8.17% | 16.38% | 5370 | 3.50% | 1150 | 0.58% | 191 |
| **2000 m** | 17% | 7.03% | 15.51% | 5080 | 2.11% | 692 | 1.18e-4% | 0.0387 |
| **2000 m selected removal** | 22% | 8.70% | 15.46% | 5070 | 3.38% | 1110 | 2.96% | 969 |

Table 15. Percent contribution of environmental variables for all six models. Percent contribution is a Maxent output that explains how much the environmental variables contribute to the prediction of suitable habitat.

| | **Depth** | **Wave Height** | **Distance to Wetland** | **Turbidity** | **Substrate** |
|---|---|---|---|---|---|
| Full model (all data) | 31.5 | 36.2 | 16.8 | 13.5 | 2.1 |
| 250 m | 38.7 | 40.1 | 5.4 | 13.6 | 2.2 |
| 500 m | 33.3 | 31.1 | 16.8 | 15.6 | 3.1 |
| 1000 m | 49.2 | 36.7 | 4.6 | 5.1 | 4.4 |
| 2000 m | 55 | 28.6 | 5.2 | 5 | 6.2 |
| 2000 m selected model | 48.5 | 14.5 | 29.9 | 2.6 | 4.6 |

Table 16. Pearson correlation r and p values of environmental variables from the Maxent models. The p values are shaded, r values are in white. A rho value of 0.65 or higher is typically considered correlated.

| | Depth | Wave Height | Turbidity | Substrate Type | Distance to Wetland |
|---|---|---|---|---|---|
| Depth | | 0 | 8.19E-10 | 0 | 4.70E-06 |
| Wave Height | 0.418 | | 0 | 2.61E-09 | 0.244 |
| Turbidity | -0.264 | -0.481 | | 0.007 | 4.44E-16 |
| Substrate Type | 0.496 | 0.256 | -0.117 | | 0 |
| Distance to Wetland | 0.198 | 0.051 | -0.343 | 0.155 | |

Table 17. Average value of environmental variables for the Maxent model east and west of the Keweenaw Peninsula (longitude -88.51). Ruffe (*Gymnocephalus cernua*) reached the Keweenaw Peninsula in 2002 and continued to spread east from there.

| | **East of Keweenaw** | **West of Keweenaw** |
|---|---|---|
| Turbidity | 0.222 | 0.404 |
| Depth | 35.259 | 5.153 |
| Wave Height | 0.219 | 0.149 |
| Substrate | 3 | 3 |
| Distance to Wetland | 3980.374 | 4068.116 |

## Figures

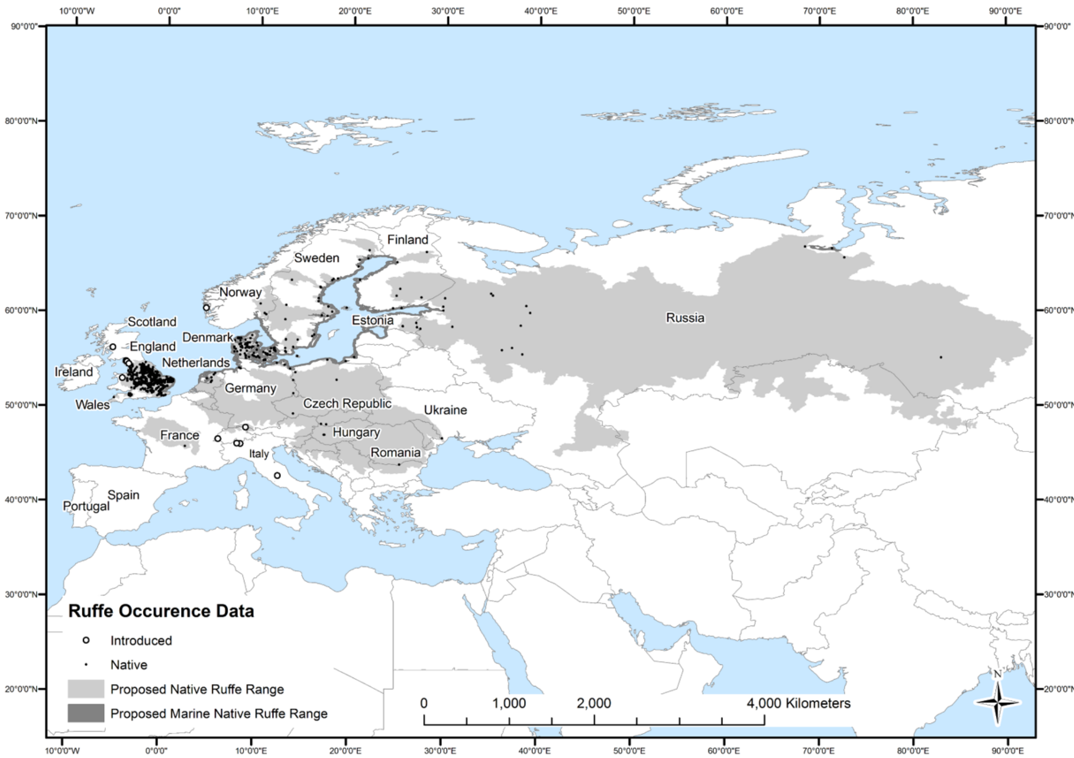



Figure 1. Proposed range map for Ruffe (*Gymnocephalus cernua*). Points include both native (N=229) and non-native (N=16) populations. Occurrence points were plotted in ArcGIS using latitudes and longitudes from Ruffe data in the literature (Johnsen 1965; Nygren *et al.* 1968; Travkina 1971; Wootten 1974; Nyman 1975; Kolomin 1977; Kozlova and Panasenko 1977; Willemsen 1977; Biro 1977; Dykova and Lom 1978; Doornbos 1979; Neuman 1979; Nilsson 1979; Pihu and Maemets 1982; Van Densen and Hadderingh 1982; Logvinenko *et al.* 1983; Hansson 1984, 1987; Sterligova and Pavlovskiy 1984; Bagge and Hakkari 1985; Sandlund *et al.* 1985; Vollestad *et al.* 1986; Boikova 1986; Bakanov *et al.* 1987; Matkovskiy 1987; Mayr *et al.* 1987; Peters *et al.* 1987; Boron and Kuklinska 1987; Bastl 1988; Nagy 1988; Neuman and Karas 1988; Bergman 1988a; Parmanne 1988; Bergman 1991; Eklov and Hamrin 1989; Maitland and East 1989; Neja 1989; Appelberg 1990; Duncan 1990; Lindesjoo and Thulin 1990; Tellervo Valtonen *et al.* 1990; Urho *et al.* 1990; Bonsdorff and Storberg 1990; Jamet and Lair 1991; Jokela *et al.* 1991; Mattila 1992; Kalas 1995; Werner *et al.* 1996; Popova *et al.* 1998; Hölker and Thiel 1998; Lehtonen *et al.* 1998; Stepien *et al.* 1998; Pietrock *et al.* 1999; Kangur 2000; Kangur *et al.* 2000, 2003; Lilja *et al.* 2003; Winfield *et al.* 2004; Lorenzoni *et al.* 2009; Peterson *et al.* 2011; Volta *et al.* 2013).

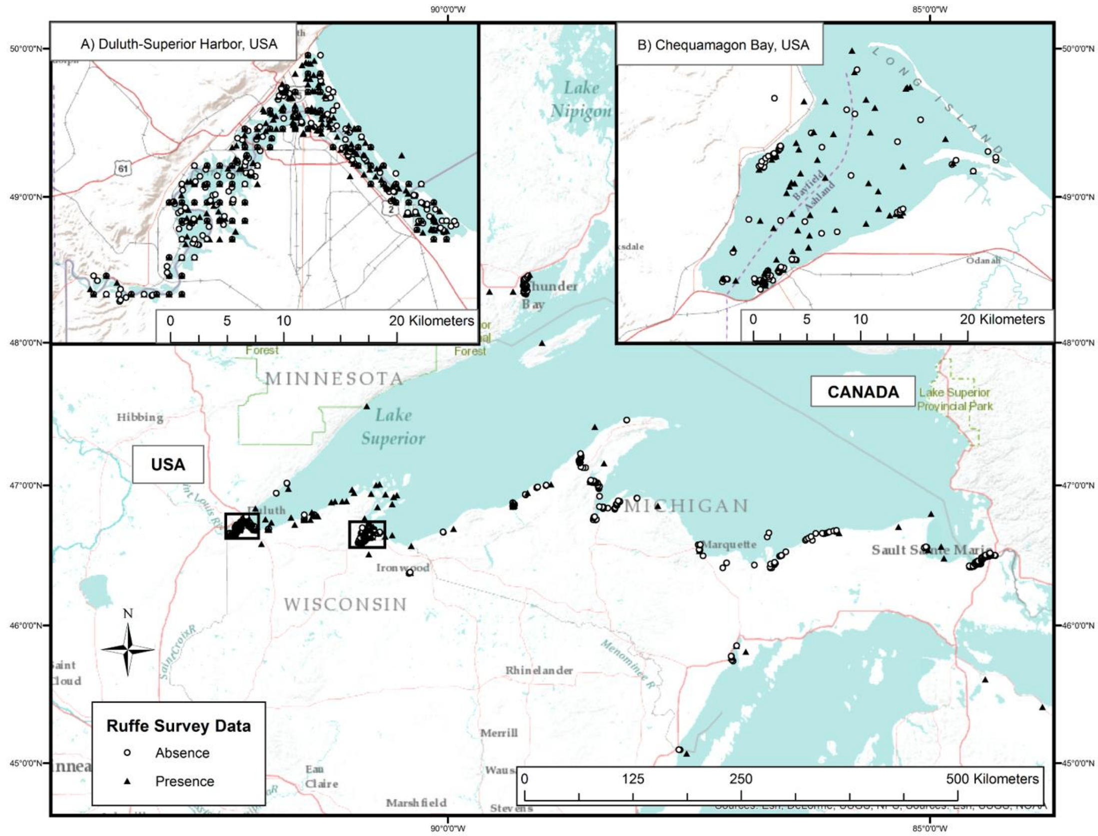



Figure 2. Occurrence data of Ruffe (*Gymnocephalus cernua*) in the Laurentian Great Lakes, North America. Points (N=5,898) include surveyed areas for Ruffe, monitoring presence (solid triangles) and absence (open circles) near the invaded regions in the Great Lakes (Brown et al., 1998; Eckmann, 2004; Fairchild and Howard McCormick, 1996; Lorenzoni et al., 2009; Maitland and East, 1989; Ogle, 2009; Ogle et al., 2004; Peterson et al., 2011; Pratt, 1988; Selgeby, 1998; Sierszen et al., 1996; Stepien et al., 1998; Volta et al., 2013) and from personal communication with several agencies, including United States Geological Survey- Lake Superior Biological Station, United States Fish and Wildlife Service- Ashland FWCO, and Environmental Protection Agency-MED.

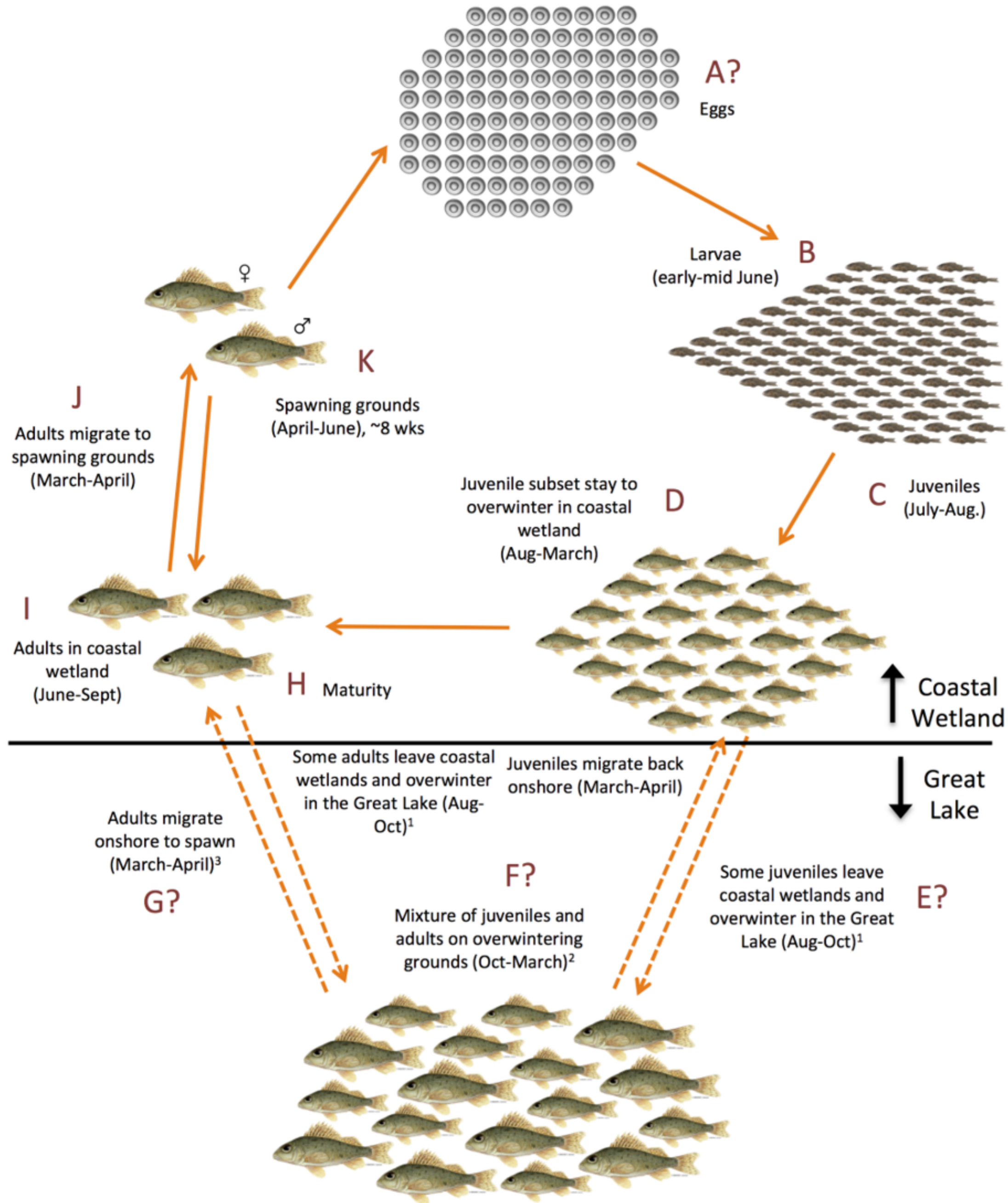



Figure 3. Proposed Ruffe (*Gymnocephalus cernua*) life cycle in the Laurentian Great Lakes. Letters with question marks indicate stages with some incomplete information; letters without question marks indicate there is a thorough understanding of that life stage in the literature. Solid lines between stages indicate a known life stage path; dotted line indicates a hypothesized life stage path. The lettering increases in order from egg to spawning pair in a clockwise fashion. Relative fish abundances at any stage are for illustration purposes only. 1. (Eckmann 2004); 2. (Selgeby 1998); 3. (Popova *et al.* 1998).

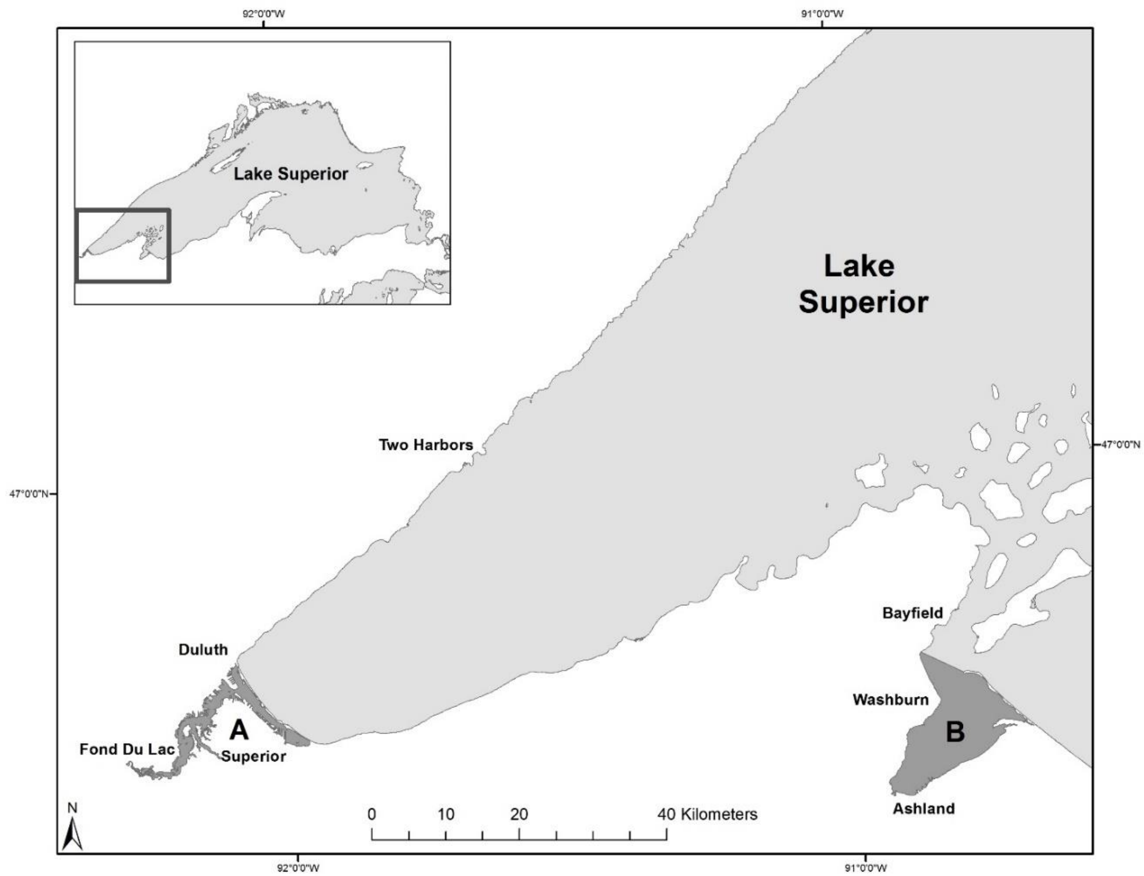



Figure 4. Map of western Lake Superior with study sites St. Louis River, MN/ WI, USA (A) and Chequamegon Bay, WI, USA (B).

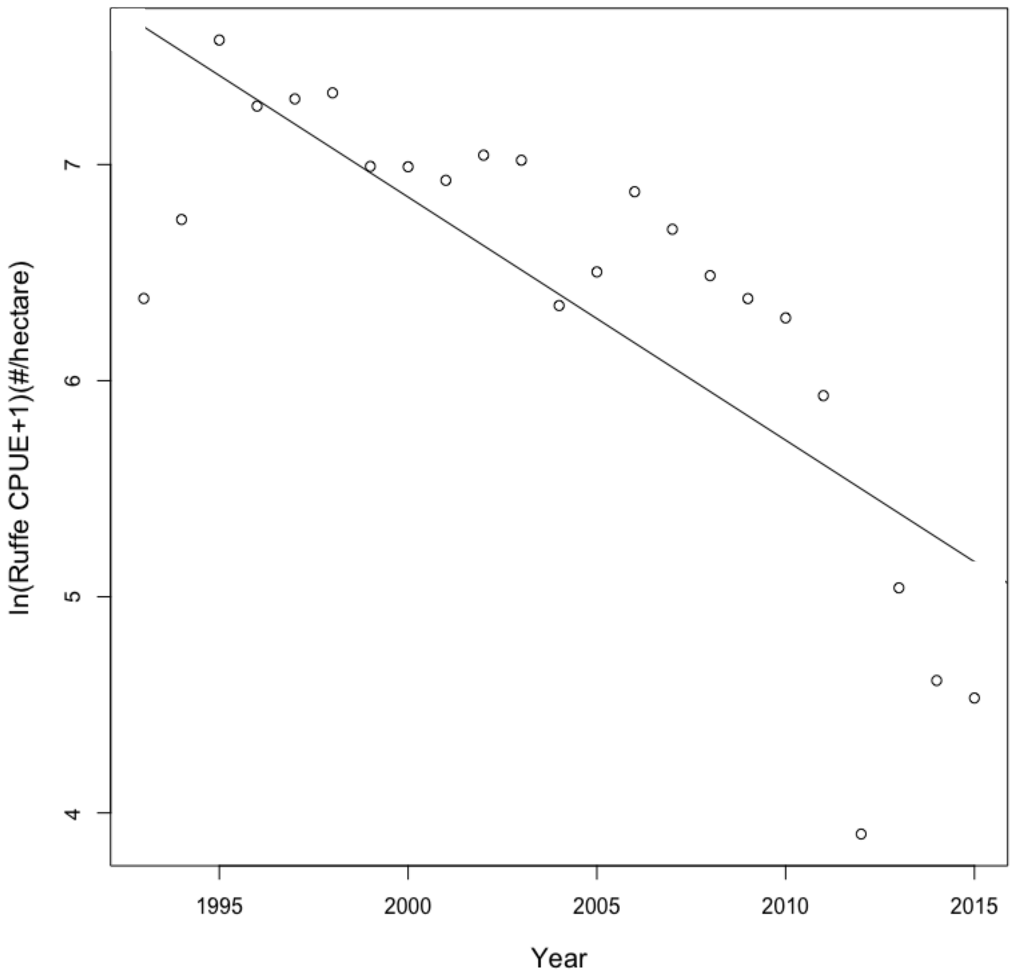



Figure 5. Ruffe (*Gymnocephalus cernua*) catch per unit effort (CPUE) in the St. Louis River, MN/ WI from 1993-2015. CPUE is represented in ln(Ruffe CPUE +1) in #/hectare. The points are annual mean CPUE, and the line is the linear model fit.

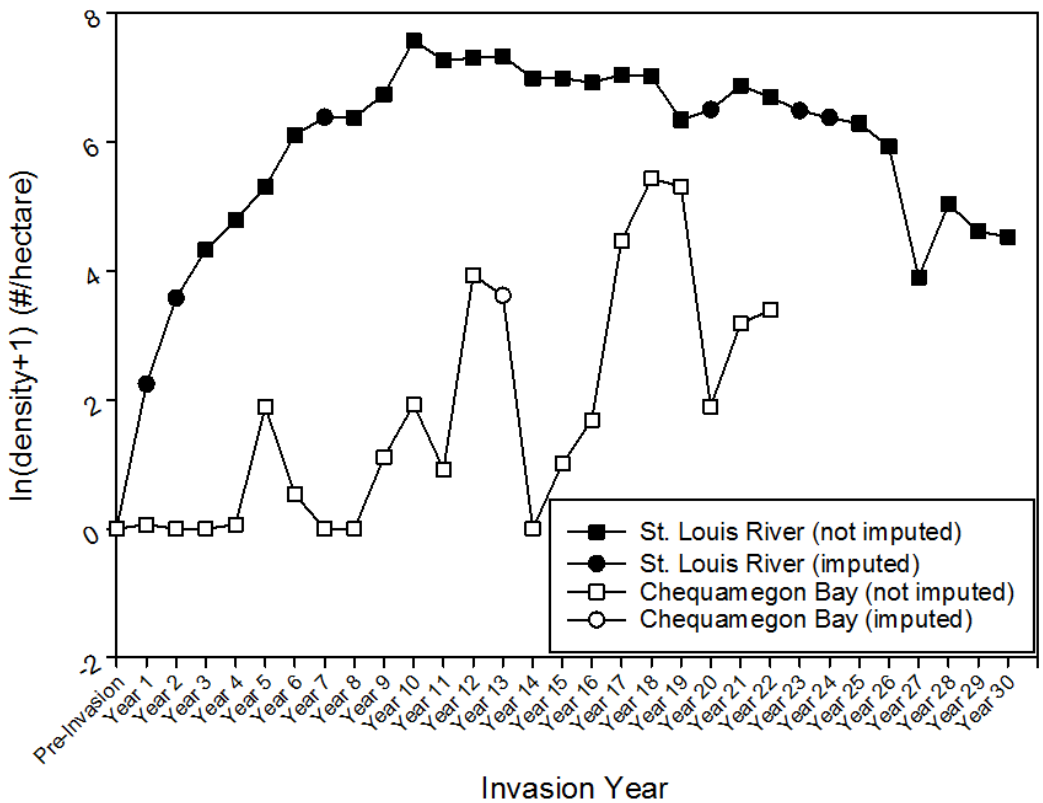



Figure 6. Annual mean catch per unit effort (CPUE) of Ruffe (*Gymnocephalus cernua*) in the St. Louis River, MN/WI and Chequamegon Bay, WI beginning from one year prior to the first Ruffe detection in each system (St. Louis River: 1985; Chequamegon Bay: 1993) to 2015. St. Louis River data from 1985-1992 was borrowed from Pratt (1988), Ruffe Task Force (1992), and USGS and missing data (St. Louis River: 2005, 2008, 2009; Chequamegon Bay: 2006) were imputed using the same spline curve technique described in the methods. The circles are the imputed values, and the squares are the known values.

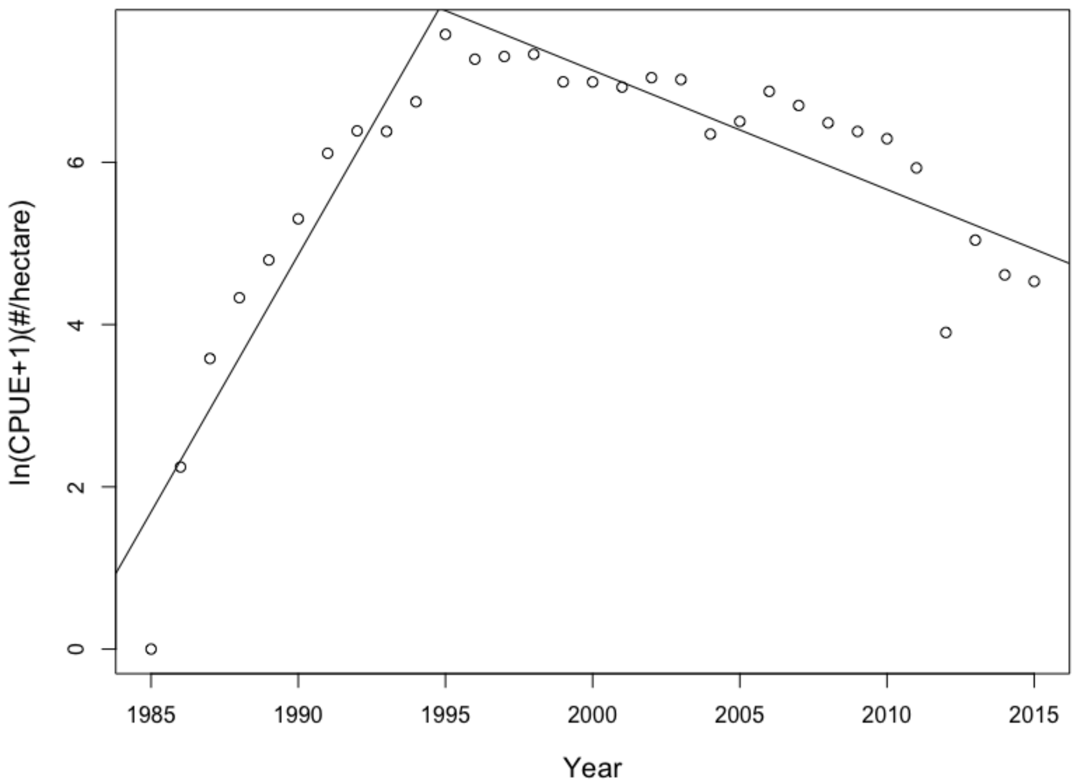



Figure 7. Annual mean catch per unit effort (CPUE) of Ruffe (*Gymnocephalus cernua*) in the St. Louis River, MN/WI from 1985-2015. Data from 1985-1993 was extrapolated from Ruffe Task Force literature and missing data (2005, 2008, 2009) were imputed using the same spline curve technique described in the methods (Ruffe Task Force 1992). Two linear models were fit: 1983-1995, and 1996-2015.

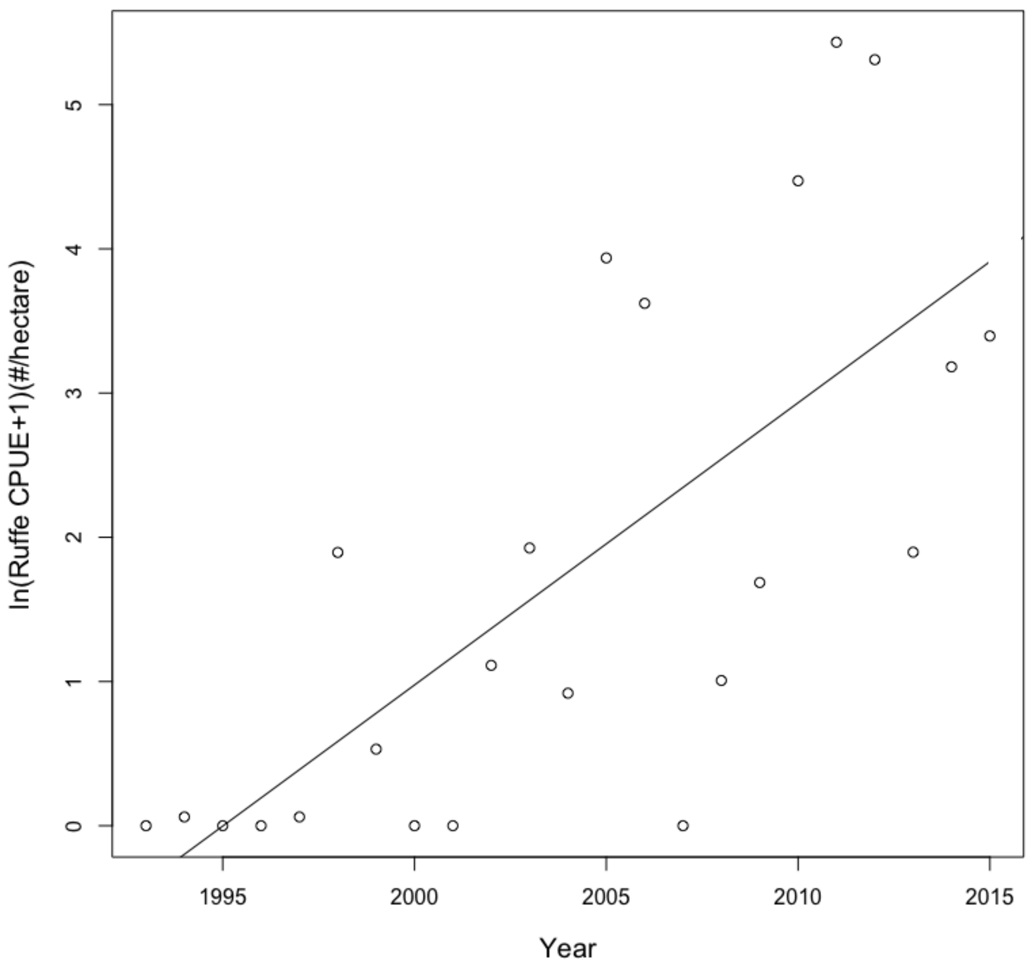



Figure 8. Ruffe (*Gymnocephalus cernua*) catch per unit effort (CPUE) in Chequamegon Bay, WI from 1993-2015. CPUE is represented in ln(Ruffe CPUE +1) in #/hectare. The points are the annual mean CPUE, and the line is the linear model fit.

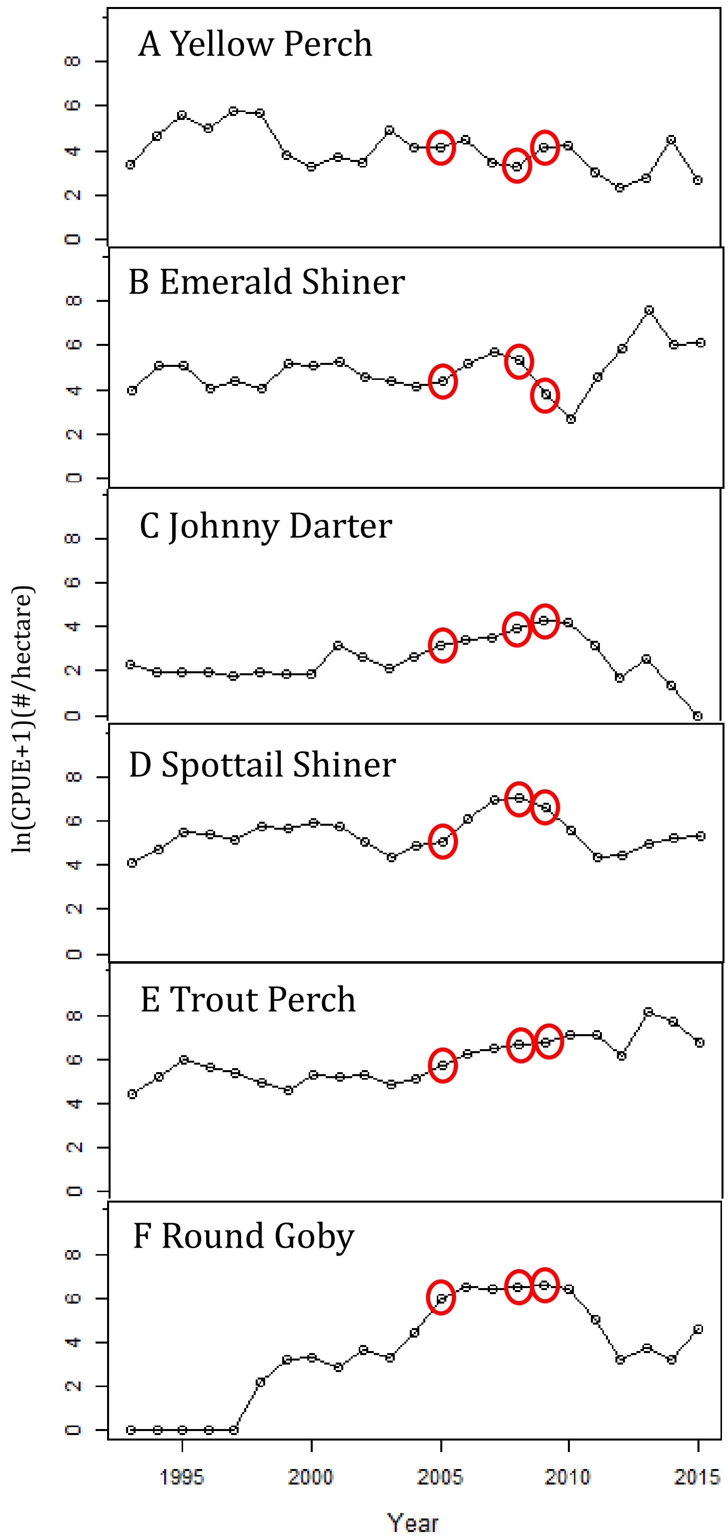



Figure 9. Annual mean catch per unit effort (CPUE) of potential Ruffe (*Gymnocephalus cernua*) competitors, including a) Yellow Perch (*Perca flavescens*), b) Emerald Shiner (*Notropis atherinoides*), c) Johnny Darter (*Etheostoma nigrum*), d) Spottail Shiner

(*Notropis hudsonius*), e) Trout Perch (*Percopsis omiscomaycus*), and f) Round Goby (*Neogobius melanostomus*) in the St. Louis River, MN/ WI from 1993-2015. The points circled are the imputed values.

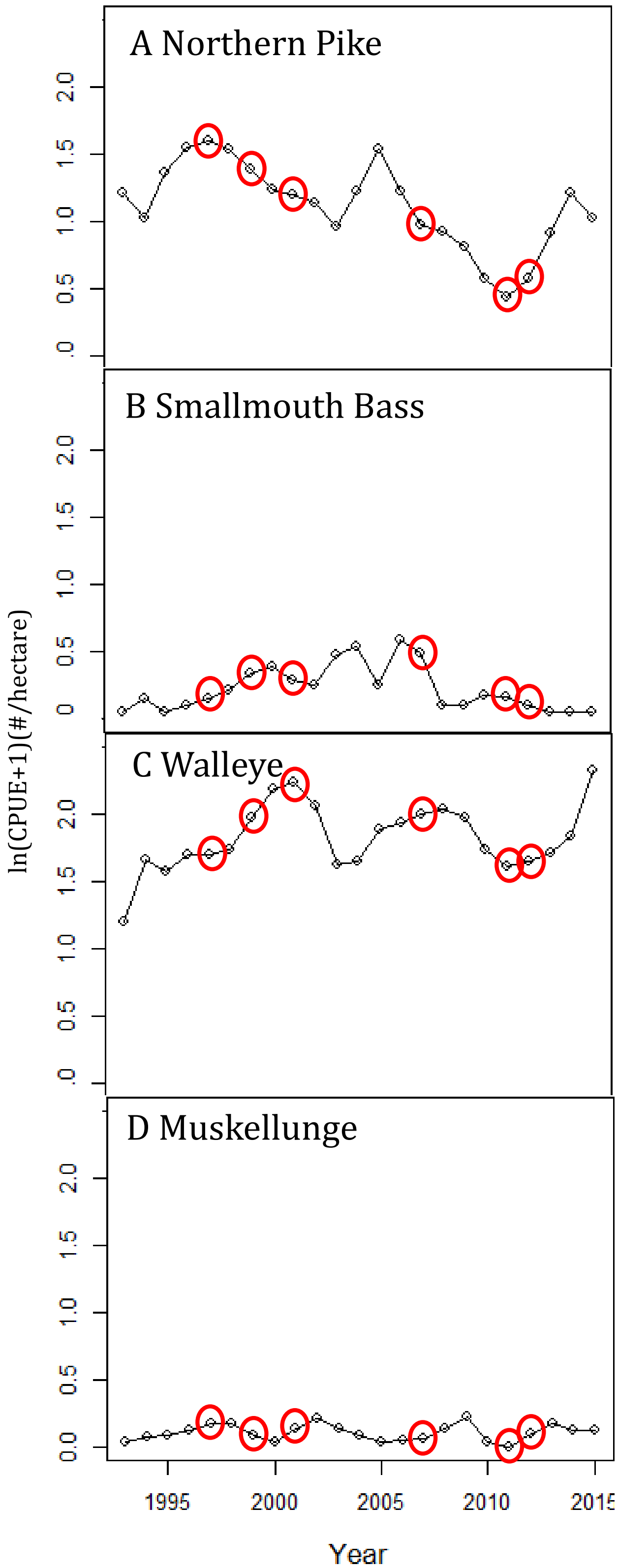

A Northern Pike
B Smallmouth Bass
C Walleye
D Muskellunge
ln(CPUE+1)(#/hectare)
0.0
0.5
1.0
1.5
2.0
1995
2000
2005
2010
2015
Year

Figure 10. Annual mean catch per unit effort (CPUE) of potential Ruffe (*Gymnocephalus cernua*) predators, including a) Northern Pike (*Esox lucius*), b) Smallmouth Bass (*Micropterus dolomieu*), c) Walleye (*Sander vitreus*), and d) Muskellunge (*Esox masquinongy*) in the St. Louis River, MN/ WI from 1993-2015. The points circled are the imputed values.

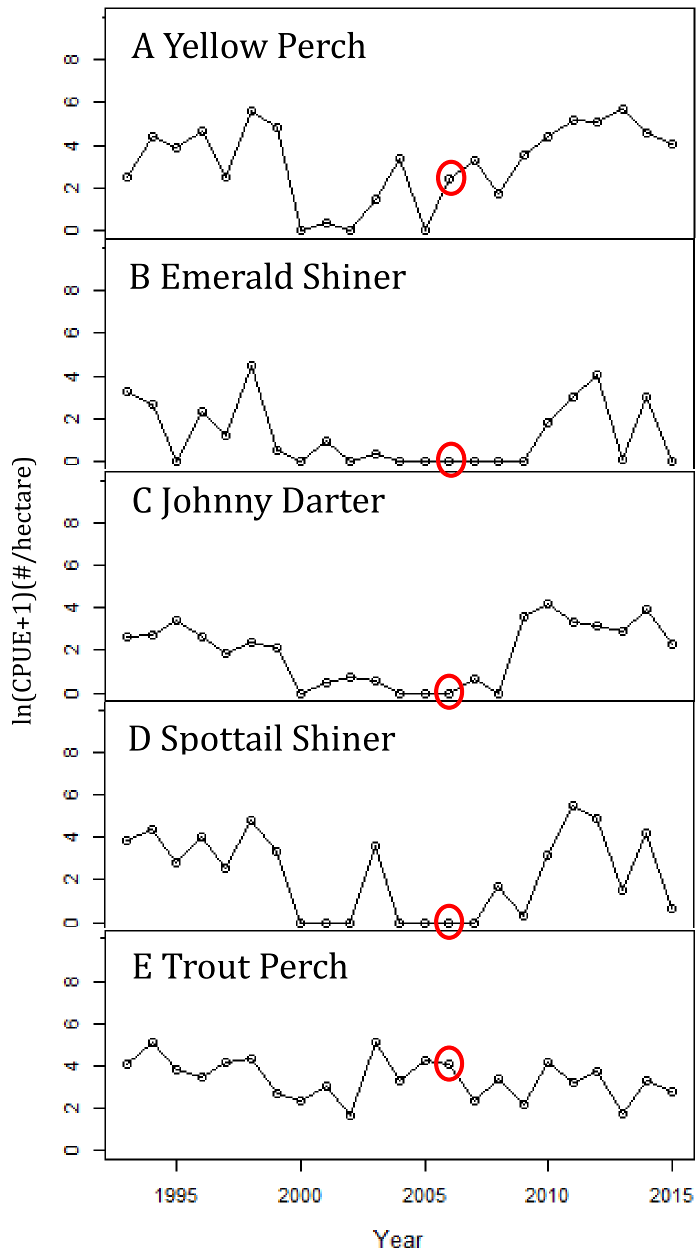



Figure 11. Annual mean catch per unit effort (CPUE) of potential Ruffe (*Gymnocephalus cernua*) competitors, including a) Yellow Perch (*Perca flavescens*), b) Emerald Shiner (*Notropis atherinoides*), c) Johnny Darter (*Etheostoma nigrum*), d) Spottail Shiner (*Notropis hudsonius*), and e) Trout Perch (*Percopsis omiscomaycus*) in Chequamegon Bay, WI from 1993-2015. The points circled are the imputed values.

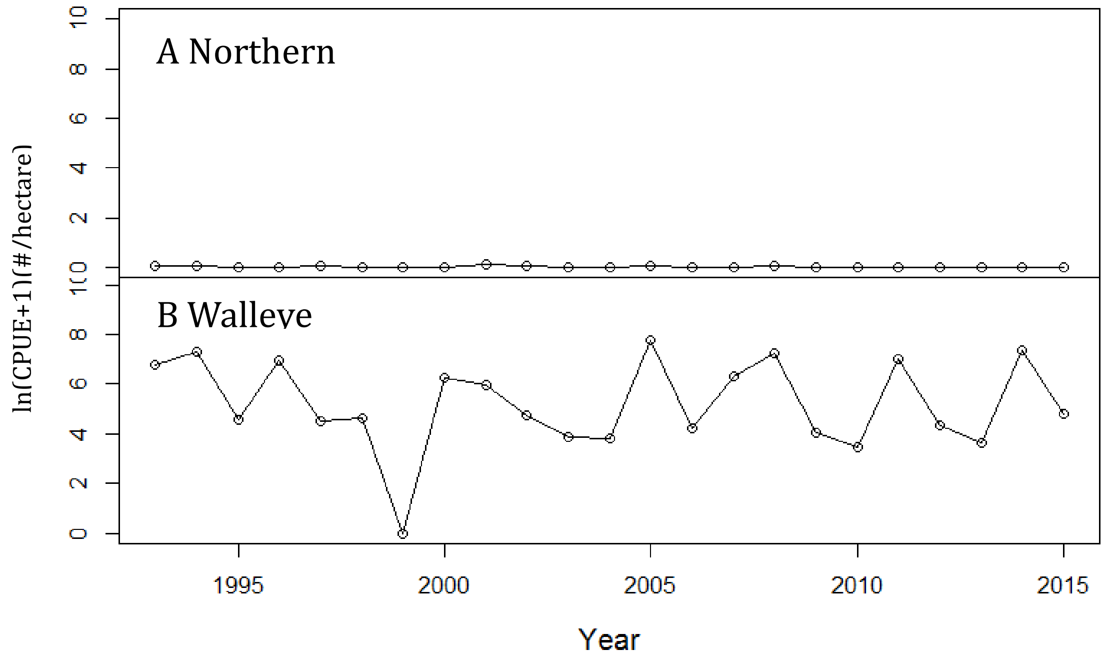



Figure 12. Annual mean catch per unit effort (CPUE) of potential Ruffe (*Gymnocephalus cernua*) predators, including a) Northern Pike (*Esox lucius*) and b) Walleye (*Sander vitreus*) in Chequamegon Bay, WI from 1993-2015. The points circled are the imputed values.

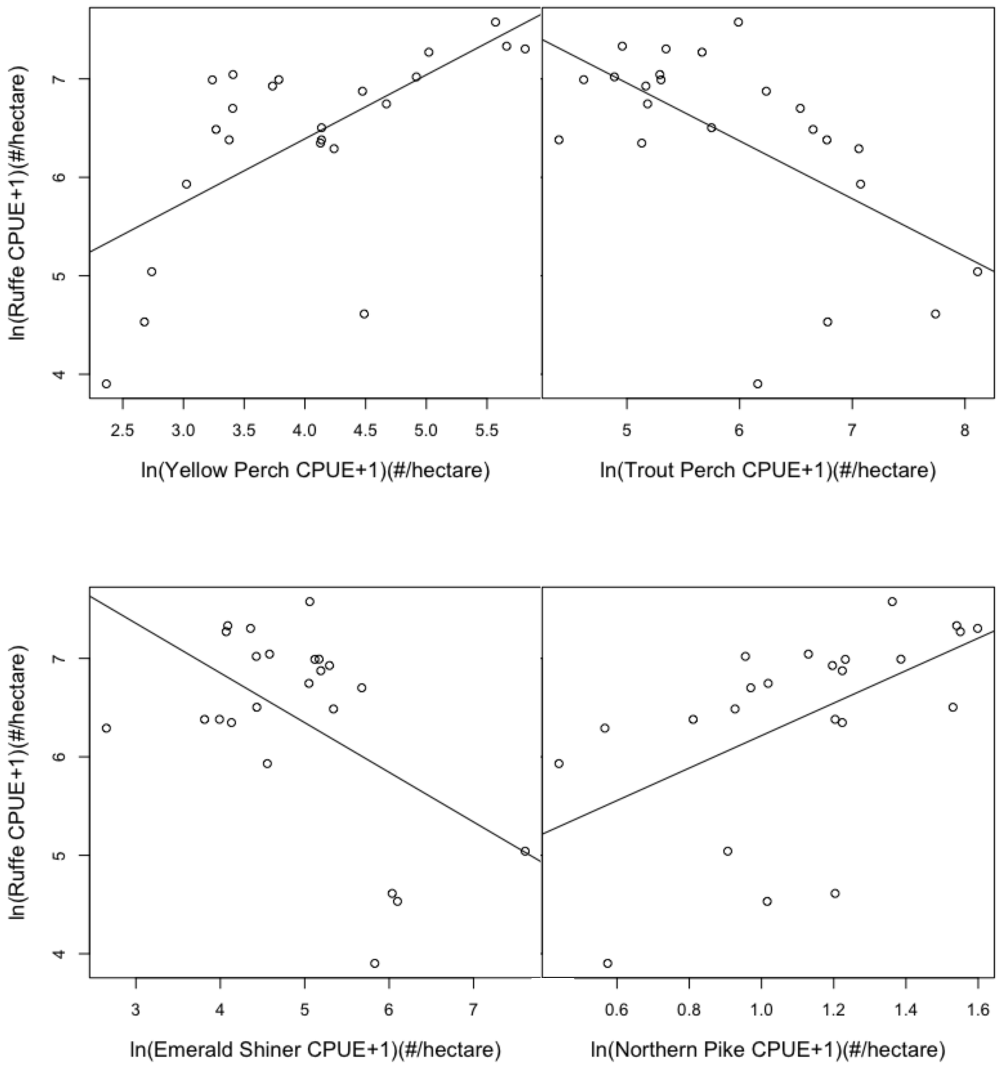



Figure 13. Best fit models of Ruffe (*Gymnocephalus cernua*) catch per unit effort (CPUE) in the St. Louis River, MN/ WI. These four models contain 99% of the model weight.

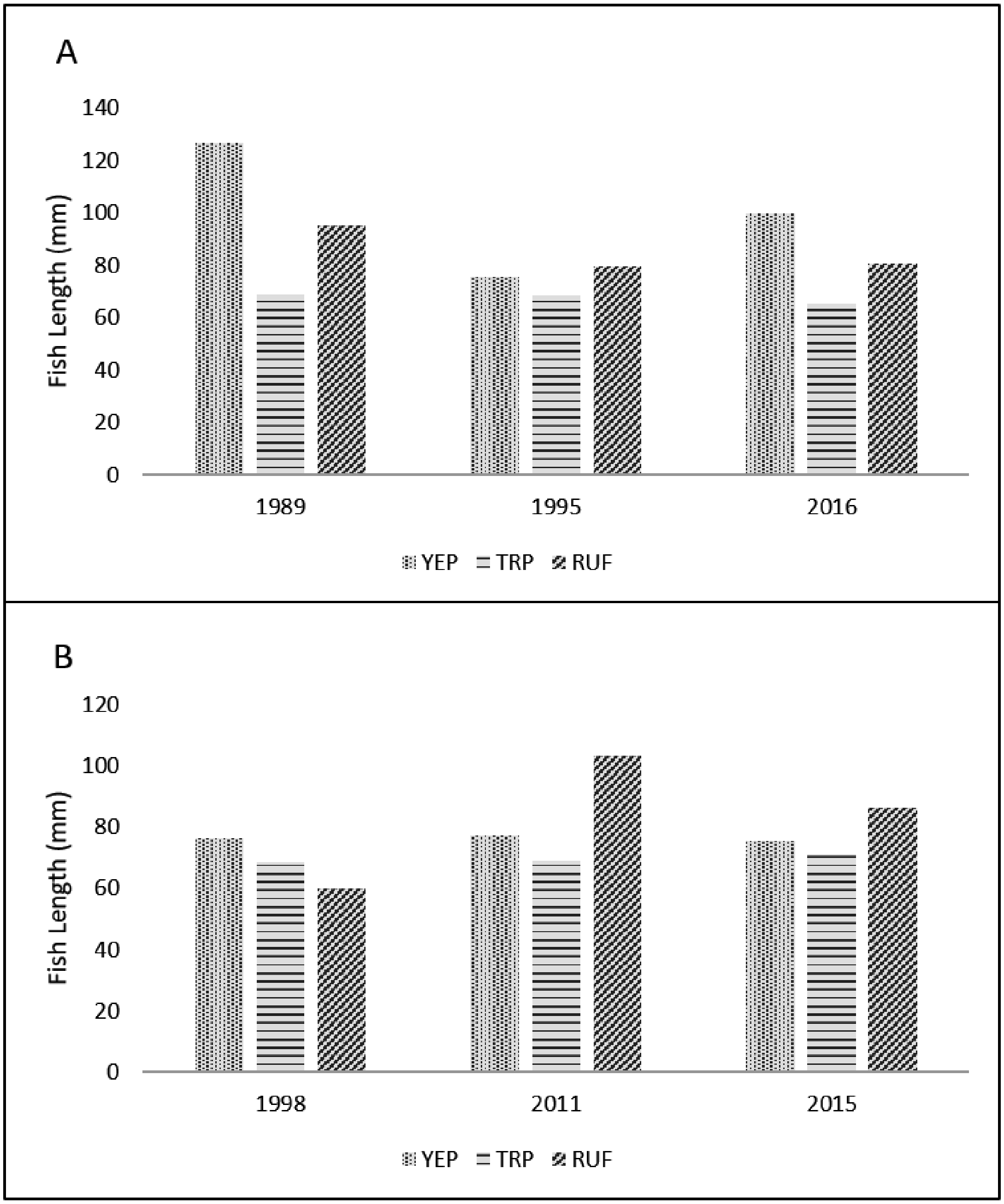
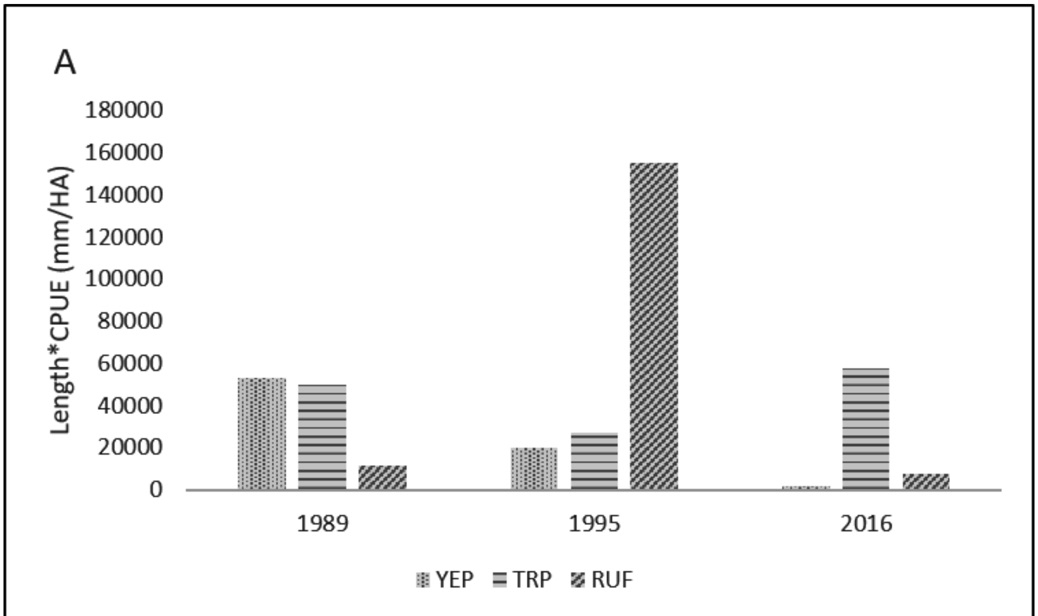



Figure 14. Fish lengths between systems (A) St. Louis River and (B) Chequamegon Bay for Yellow Perch (YEP), Trout Perch (TRP), and Ruffe (RUF). Years chosen represent a year close to initial Ruffe invasion, the year of peak Ruffe CPUE so far, and the most current data year we have.



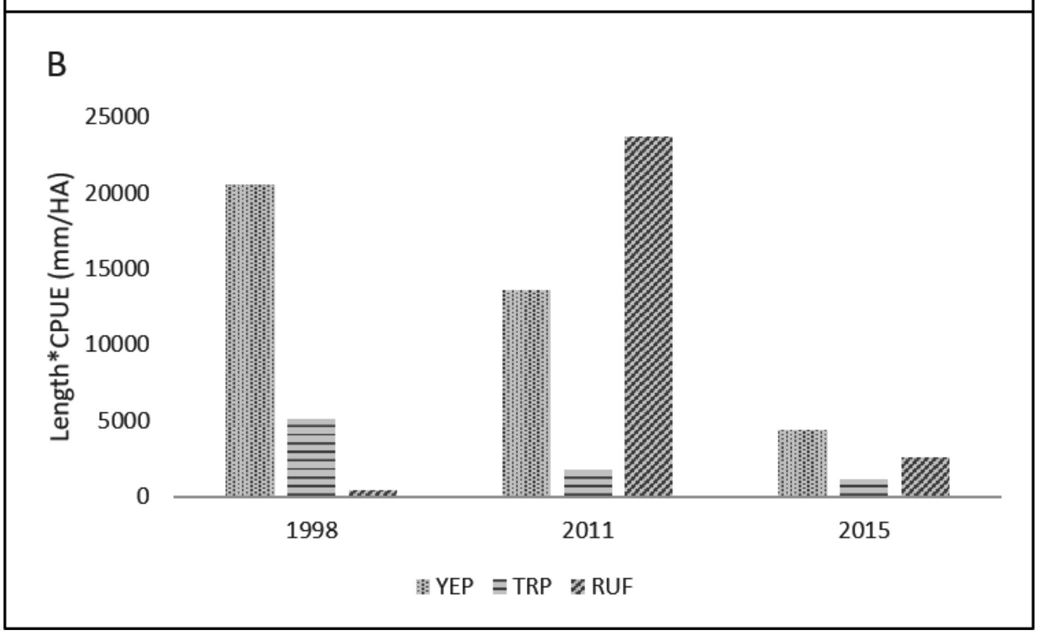




Figure 15. Fish length*CPUE, a surrogate for biomass, in each system (A) St. Louis River and (B) Chequamegon Bay for Yellow Perch (YEP), Trout Perch (TRP), and Ruffe (RUF). Years chosen represent a year close to initial Ruffe invasion, the year of peak Ruffe CPUE so far, and the most current data year we have.

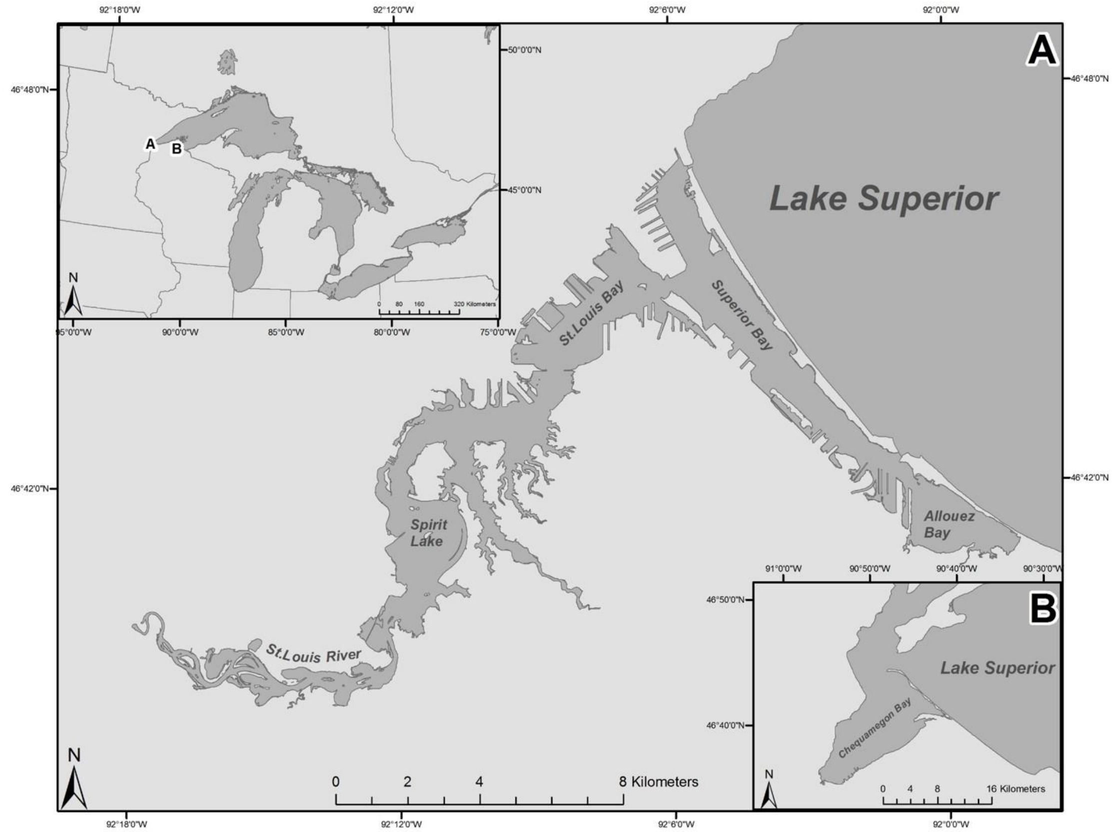



Figure 16. Map of primary study sites for stable isotopes study. **A** is the St. Louis River watershed, MN/WI, and **B** is Chequamegon Bay, WI.

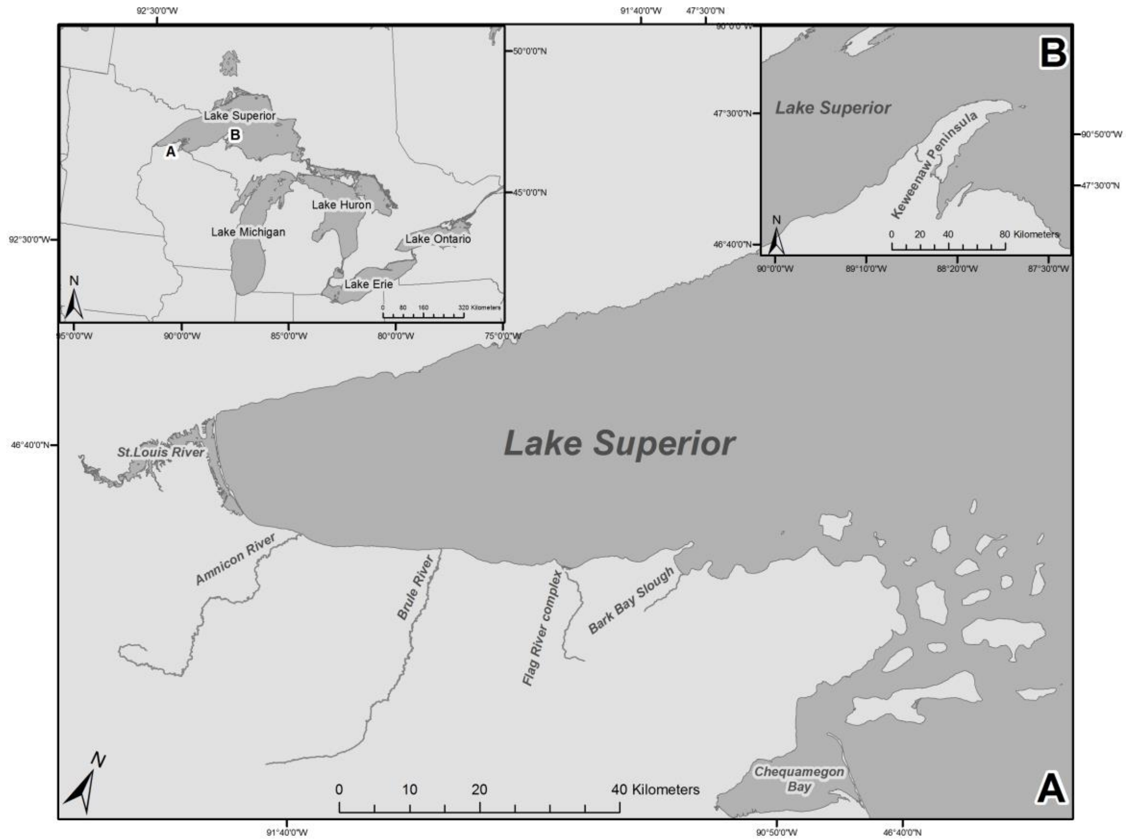



Figure 17. Map of south shore and over-winter study sites. **A** includes Amnicon River, Brule River, Flag River complex, and Bark Bay slough, WI. **B** includes Keweenaw Peninsula, MI.

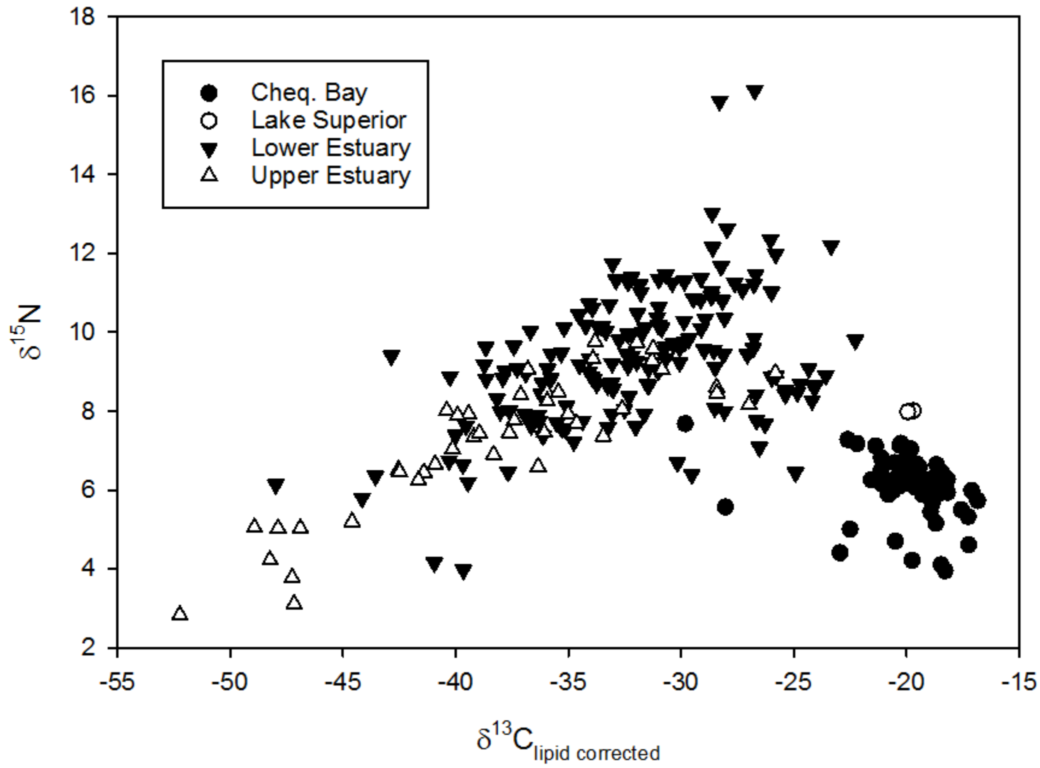



Figure 18. $\delta^{13}$Clipid corrected and $\delta^{15}$N values by capture location. Points represent individual fish in different locations based on stable isotope composition.

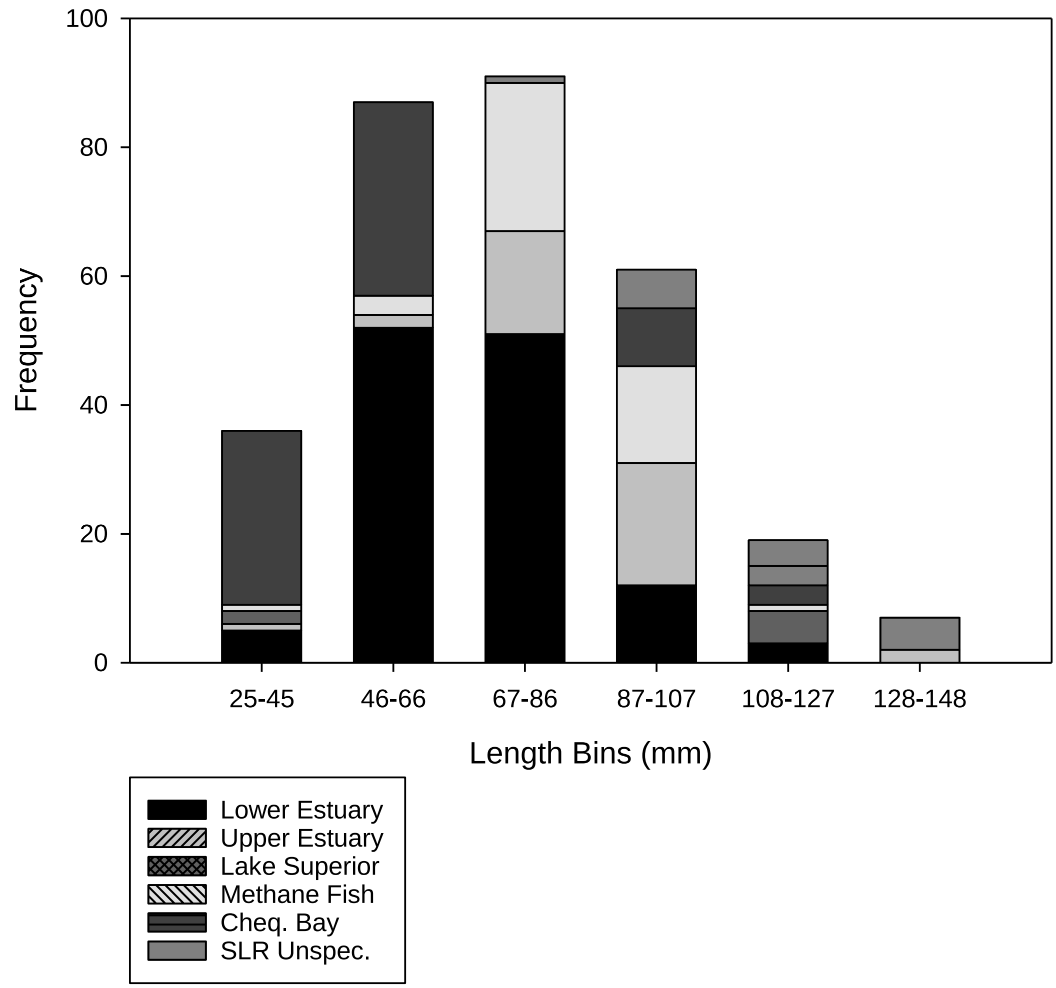



Figure 19. Length frequency of Ruffe (*Gymnocephalus cernua*) by capture location. Ruffe were binned into six different length classes, each encompassing approximately 20 mm. Ruffe locations are identified based on their length.

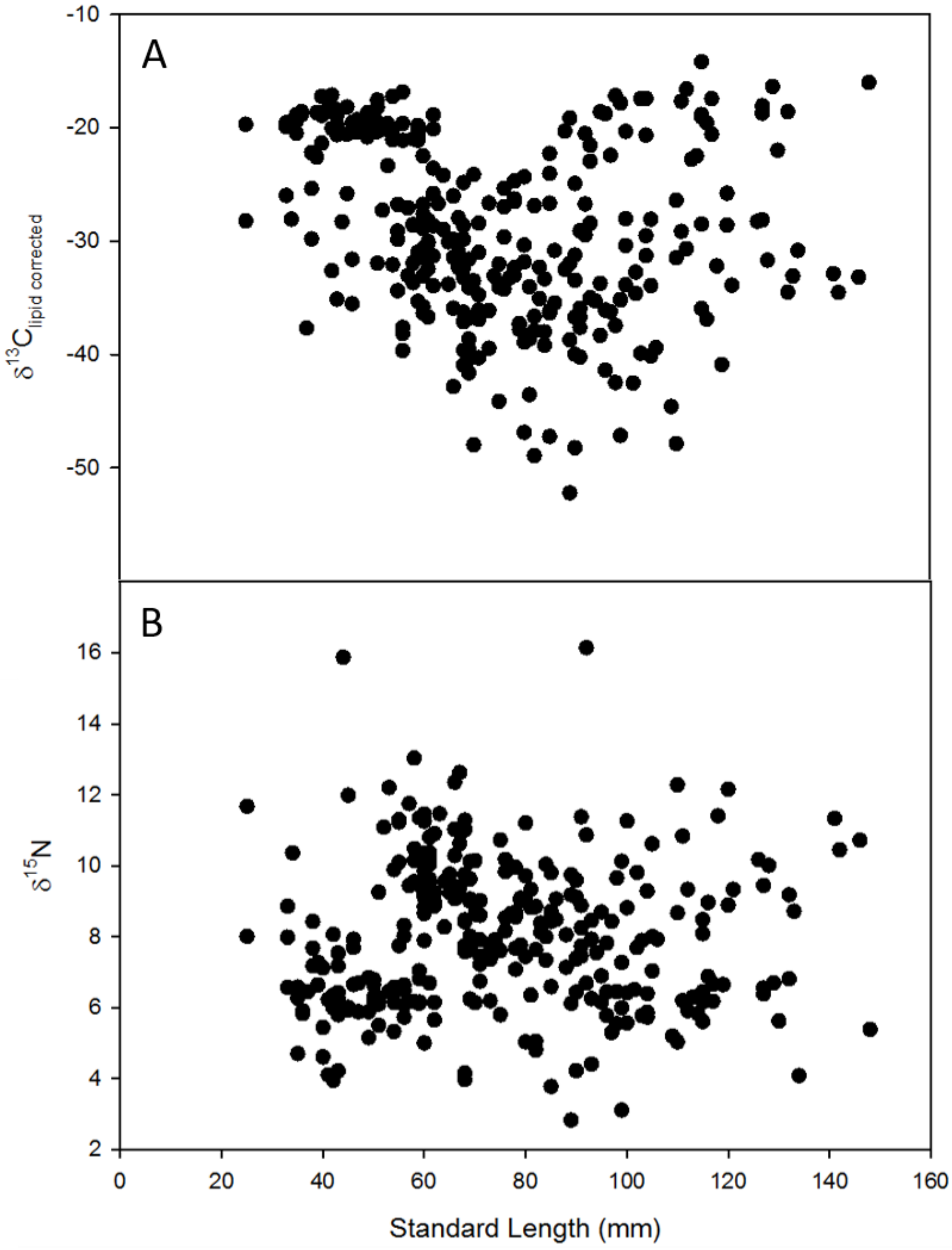



Figure 20. A) $\delta^{13}$Clipid corrected and B) $\delta^{15}$N values by standard length (mm). Points represent individual fish raw isotope values.

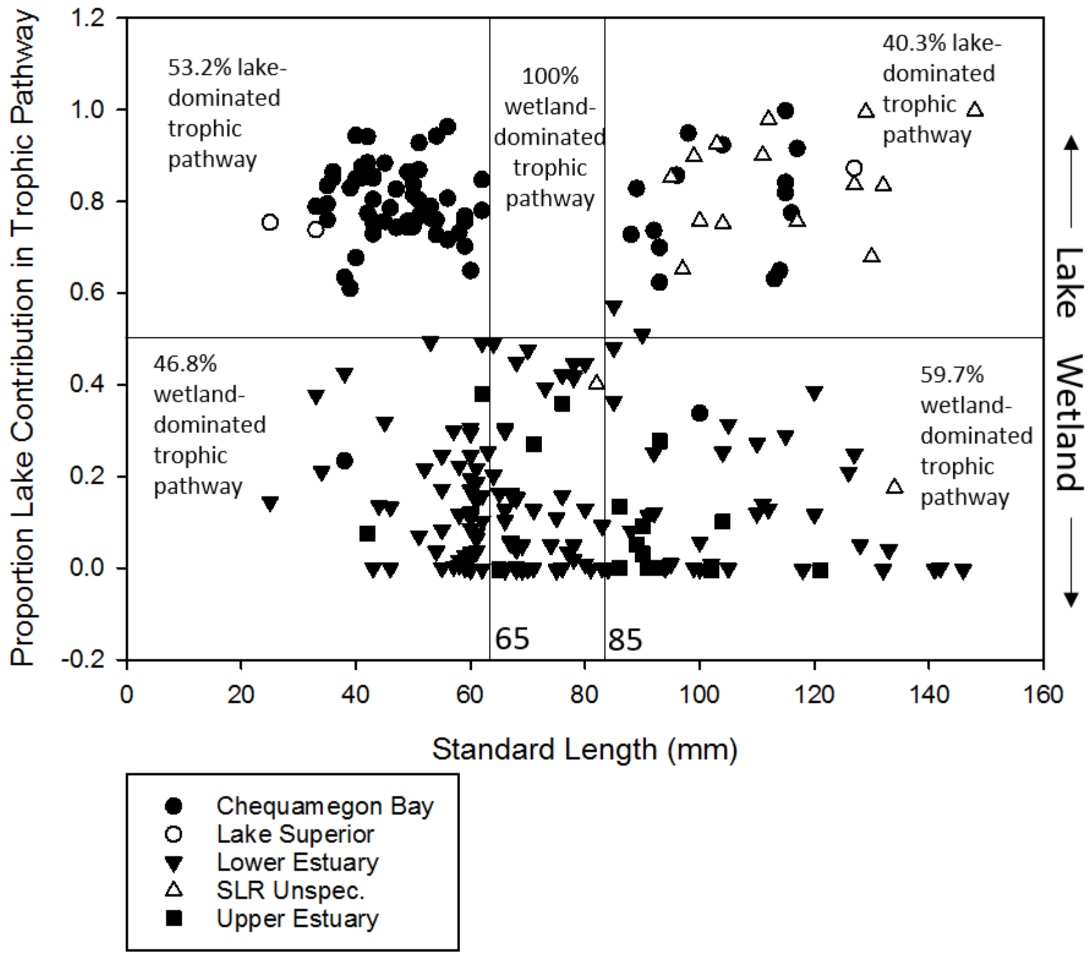



Figure 21. Proportion Lake Superior contribution in trophic pathway. Vertical lines delineate size groupings of Ruffe (*Gymnocephalus cernua*), the horizontal line delineate “lake” vs “wetland” dominated trophic pathway, and shapes indicate capture location. “SLR Unspec.” is in the St. Louis River but in an unspecified location.

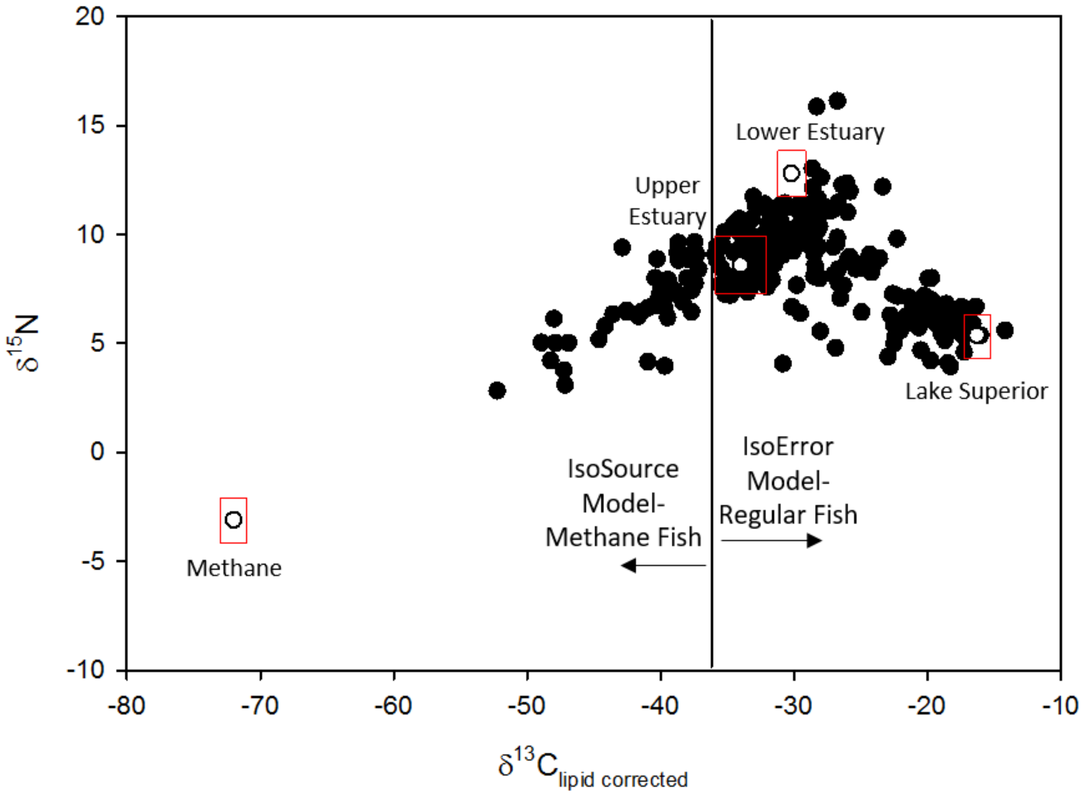



Figure 22. Unfitted stable isotopes model. Open circles are the sources for the IsoError and IsoSource models. The rectangles represent the standard deviation around the sources. Solid points represent fish stable isotopes in the models. The line is drawn at -36‰ $\delta^{13}C$ to differentiate the two models.

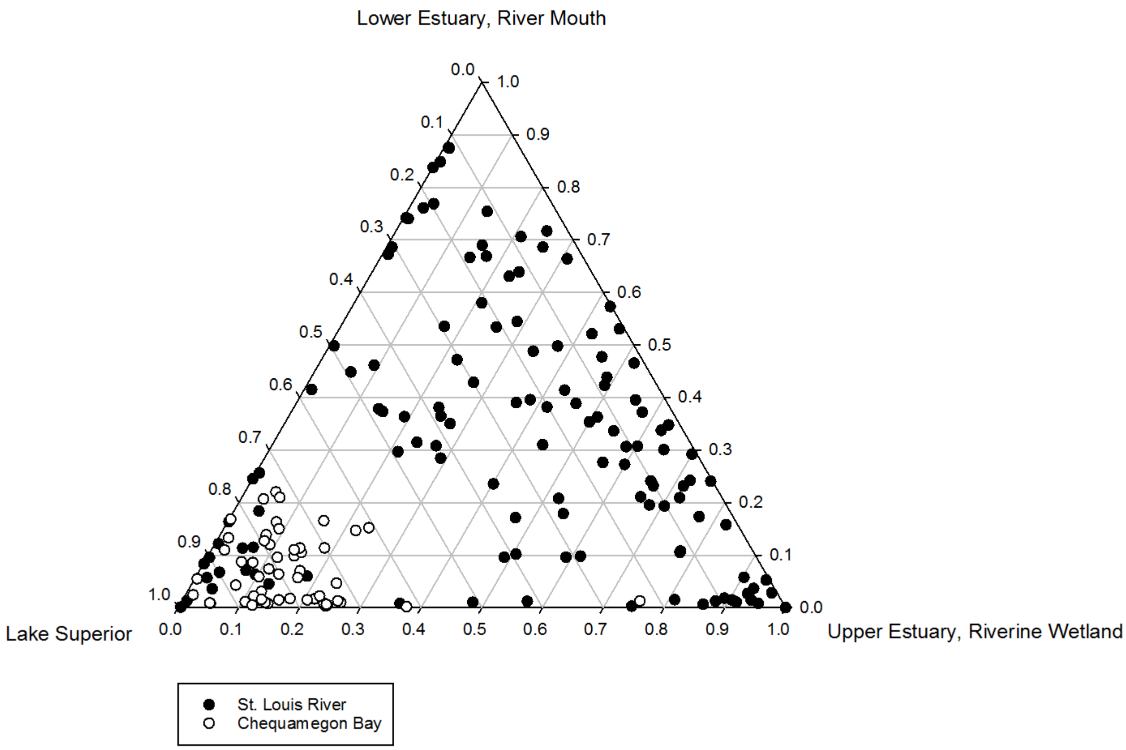



Figure 23. Triangle plot with points from St. Louis River and Chequamegon Bay. The three points of the triangle represent the sources in the IsoError model, and the points show where the individual fish fall along the axes by proportion contribution. At the points of each triangle, St. Louis River sources are listed first, and then a description of the equivalent habitats are listed second.

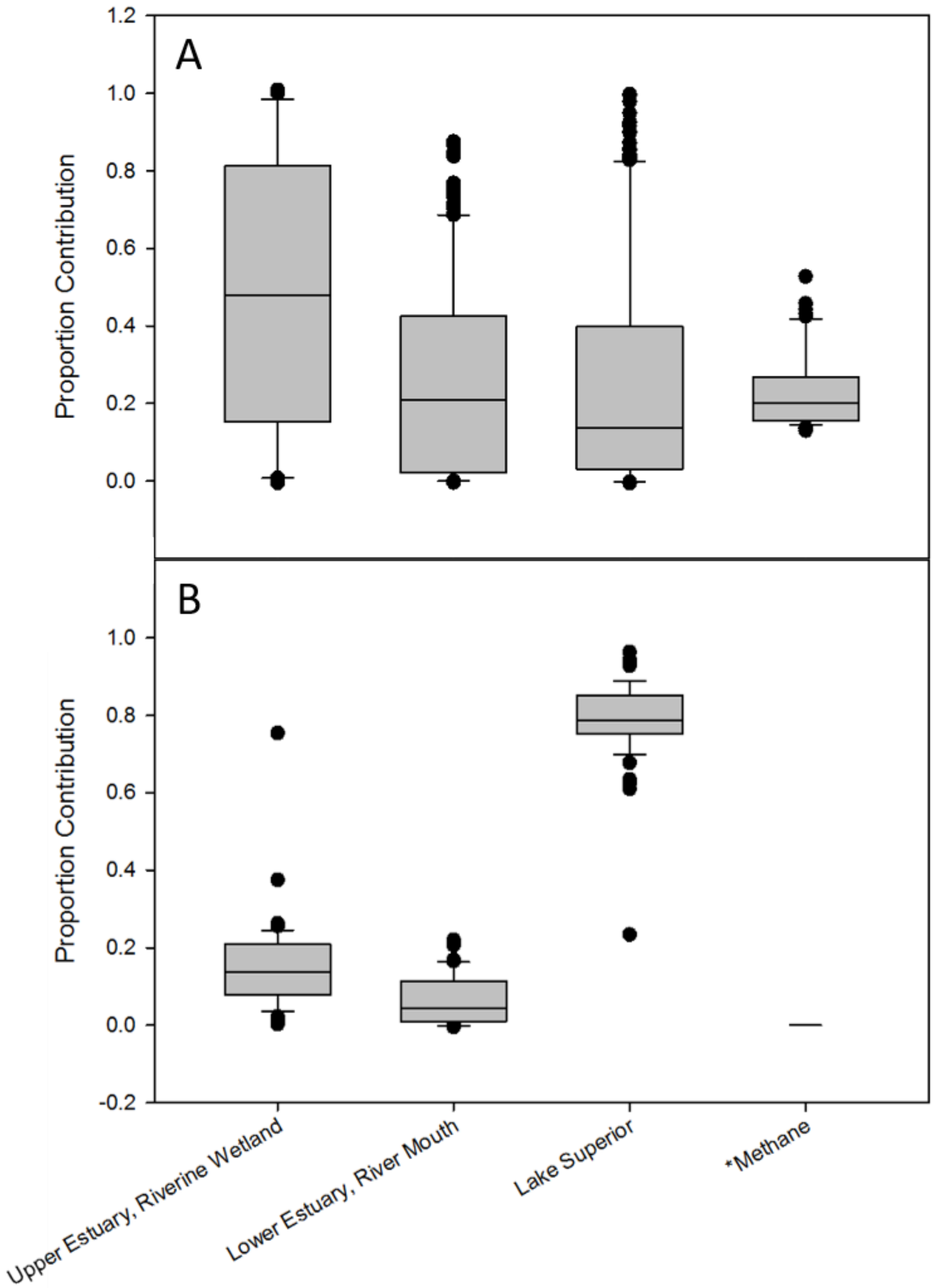



Figure 24. Boxplots of proportion contribution for A) St. Louis River and B) Chequamegon Bay Ruffe (*Gymnocephalus cernua*). At the x-axis major ticks, St. Louis River sources are listed first, and then a description of the equivalent habitats are listed second. Methane-influenced fish were not measured for Chequamegon Bay Ruffe.

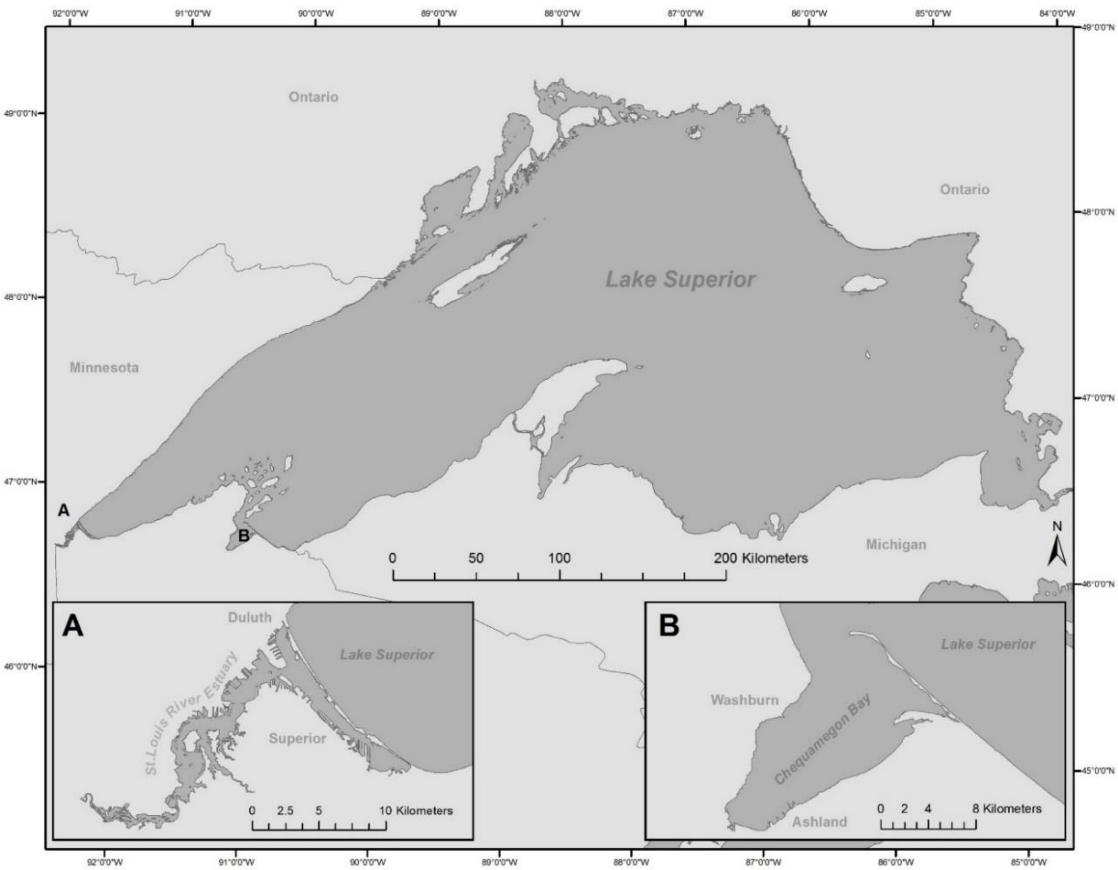



Figure 25. Site map of Lake Superior for Maxent model. Panel A is St. Louis River, WI/MN, and panel B is Chequamegon Bay, WI.

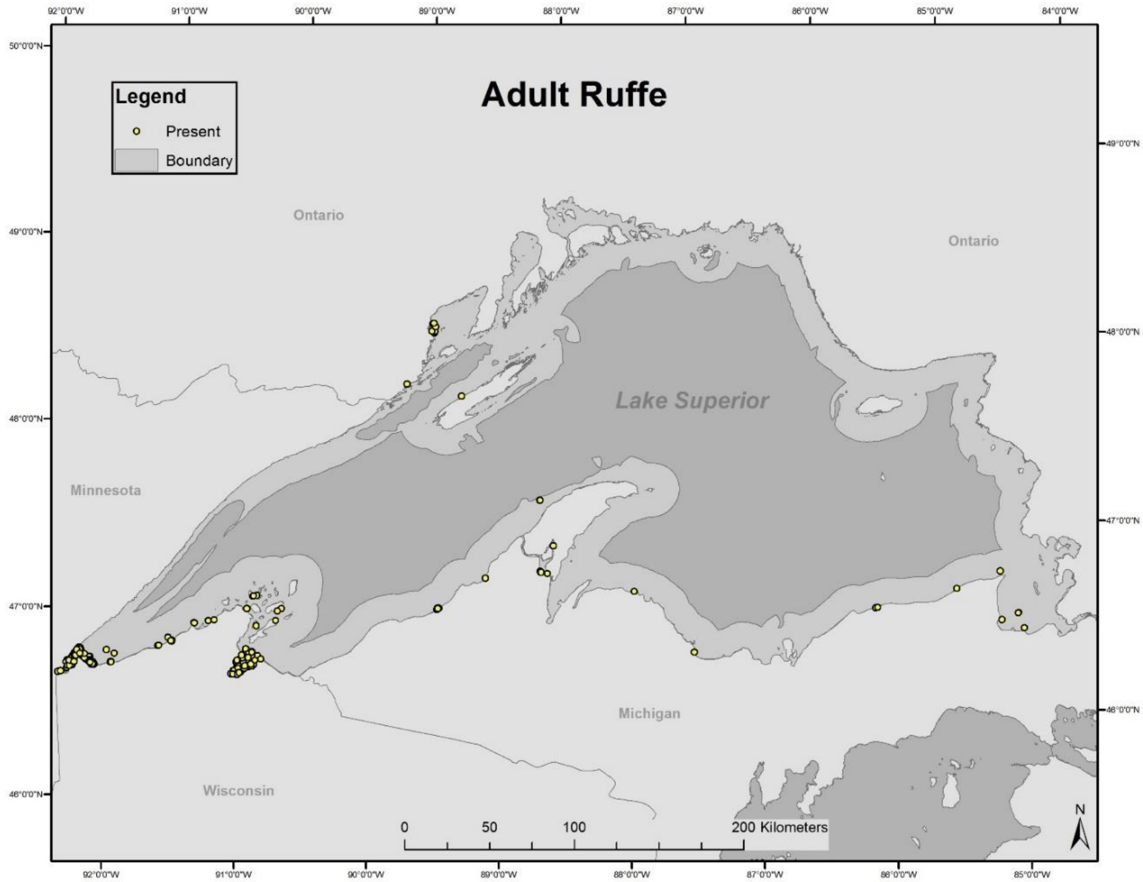



Figure 26. Occurrence points of Ruffe (*Gymnocephalus cernua*) with model buffer at 15 km distance/ 250 m depth. Buffer was created to represent the area where Ruffe can exist. Points represent Ruffe presences from 2005-2015. All overlapping points were removed. Data was gathered from US Geological Survey, US Fish and Wildlife Service, US Environmental Protection Agency, and 1854 Treaty Authority.

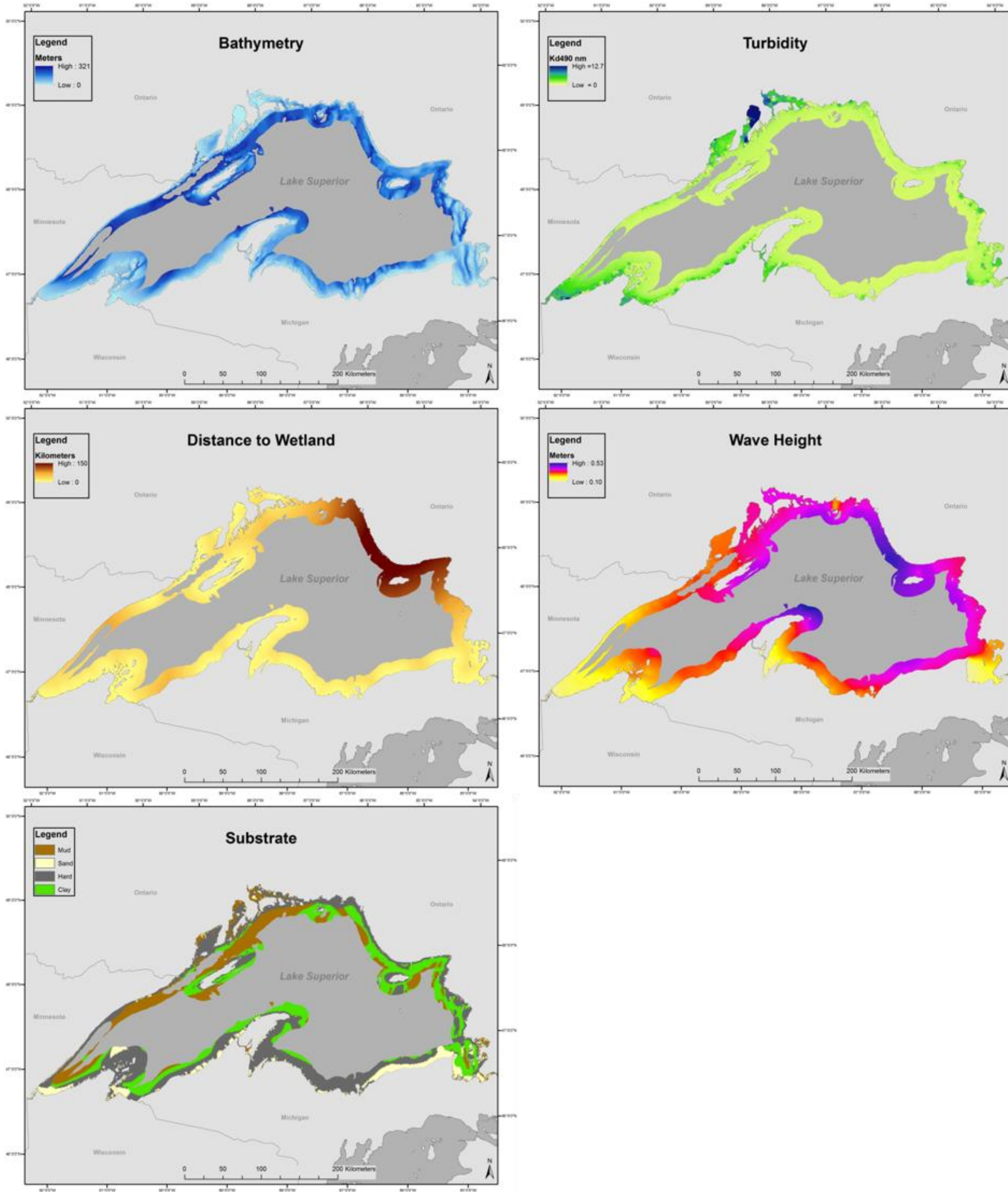



Figure 27. Environmental layers for the Maxent model. Environmental data was created in ArcGIS. It was collected from Michigan Tech Research Institute and Great Lakes Aquatic Habitat Framework. All data was resampled to 30-m resolution.

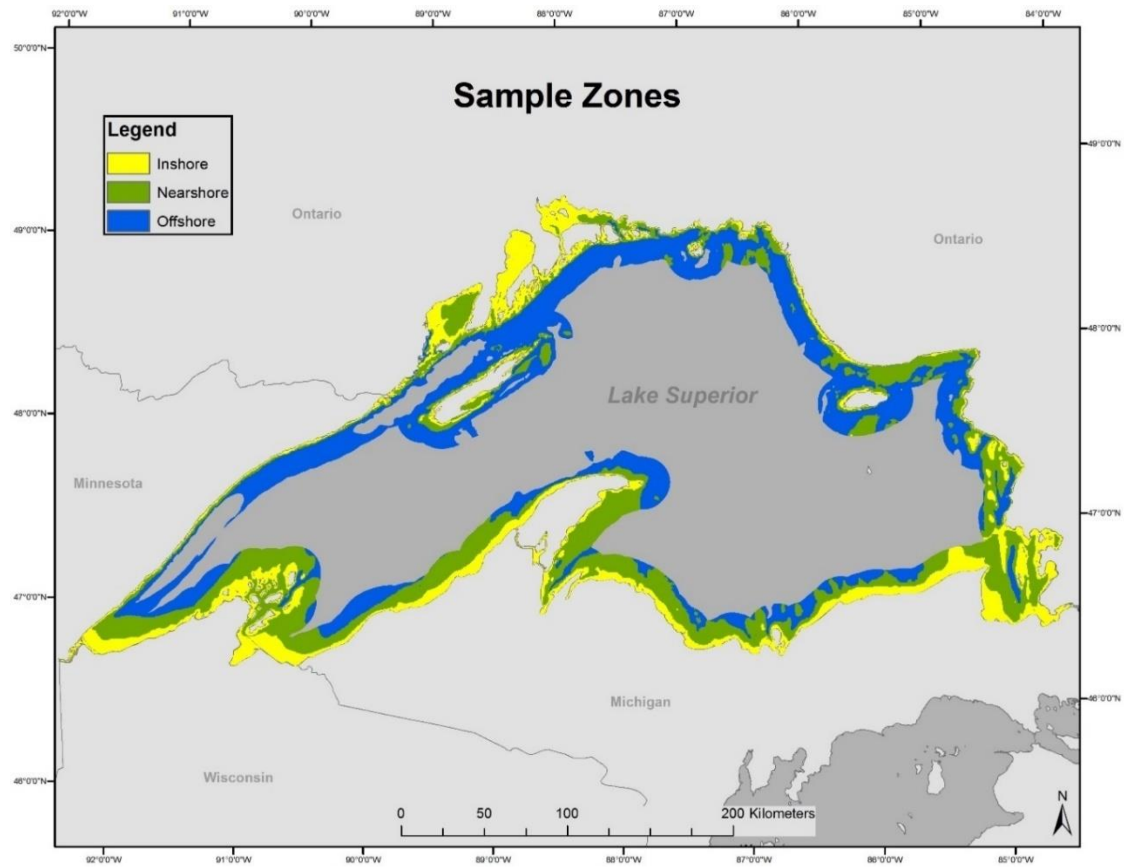



Figure 28. Proportion of suitable area within the buffer of three zones: in-shore (<30 m), nearshore (<100 m), and offshore (>100 m). Area was predicted using Maxent and ArcGIS. Area ($km^2$) and percent of buffer can be found in Table 14.

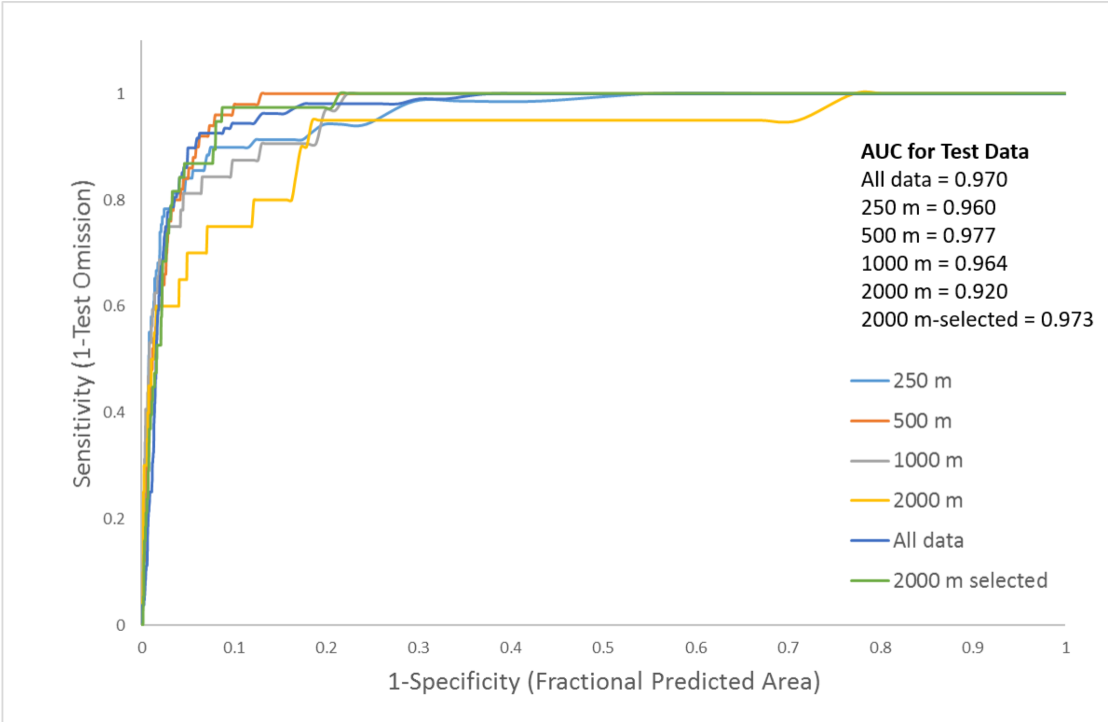



Figure 29. Receiver operator characteristic (ROC) plot for the six distance-buffered Maxent models. Area under the curve (AUC) scores are displayed to compare the six models. An AUC score above 0.75 is “potentially useful” (Elith 2002).

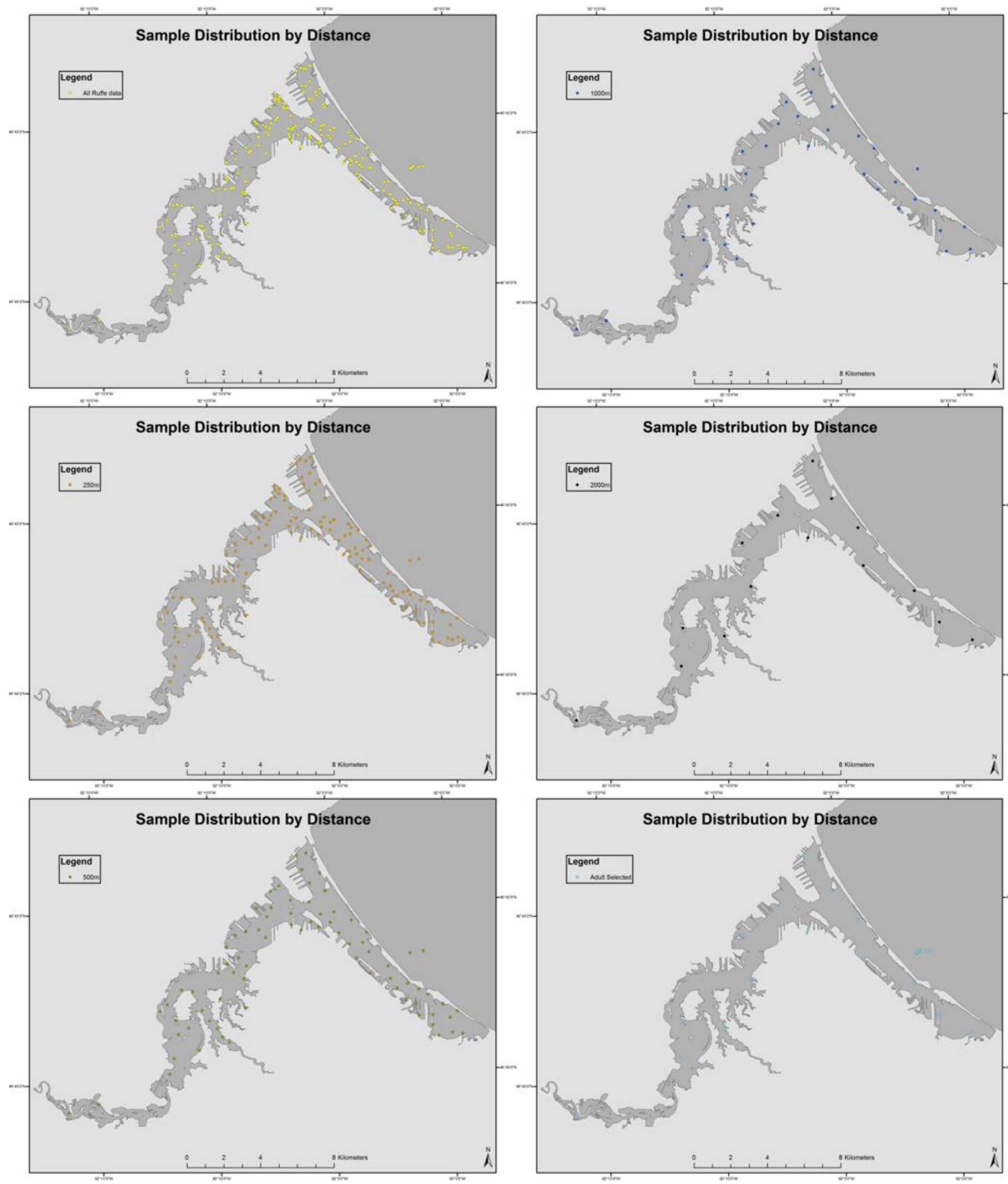



Figure 30. Example of cluster removal for each model distance buffer in the St. Louis River, WI/MN. Occurrence points for Ruffe (*Gymnocephalus cernua*) were very clustered in the St. Louis River and Chequamegon Bay, WI. We applied these distance buffers to the data to remove the clustering. There was a 250, 500, 1000, 2000, and 2000-m selected buffer. The 2000-m selected buffer removed clustering only in St. Louis River and Chequamegon Bay, while the other buffers removed clustering in all of Lake Superior. Points were removed using a computer algorithm that chose a point and removed that points within the chosen distance surrounding that point.

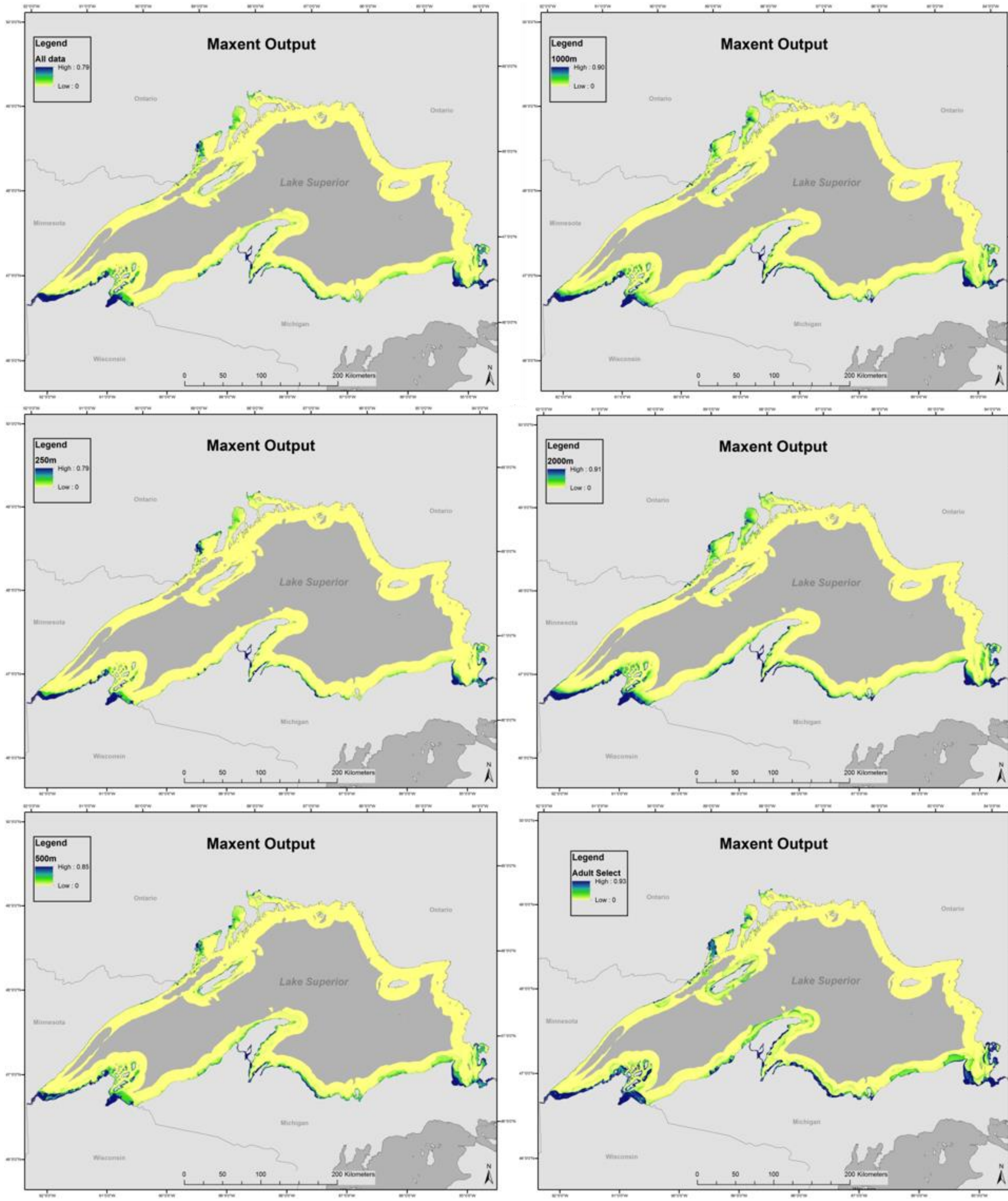



Figure 31. Maxent prediction maps of suitable habitat for Ruffe (*Gymnocephalus cernua*). These maps are an output of the Maxent model. The dark regions represent high suitability and the light regions represent no suitability.

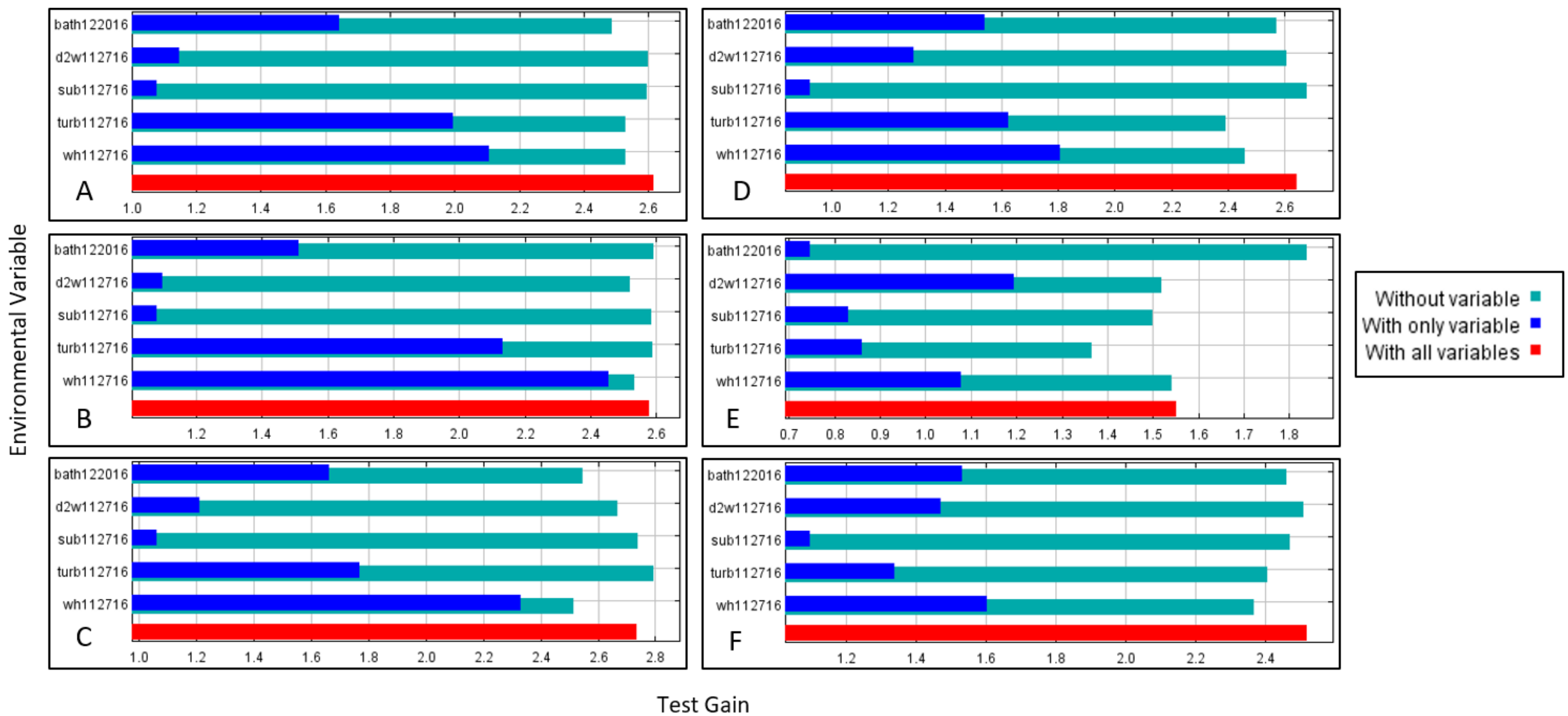



Figure 32. Jackknife test gain outputs from each model. A) model with all data, B) 250-m model, C) 500-m model, D) 1000-m model, E) 2000-m model, and F) 2000-m selected model. These are an output from Maxent that determine which environmental variables are most important to the model.

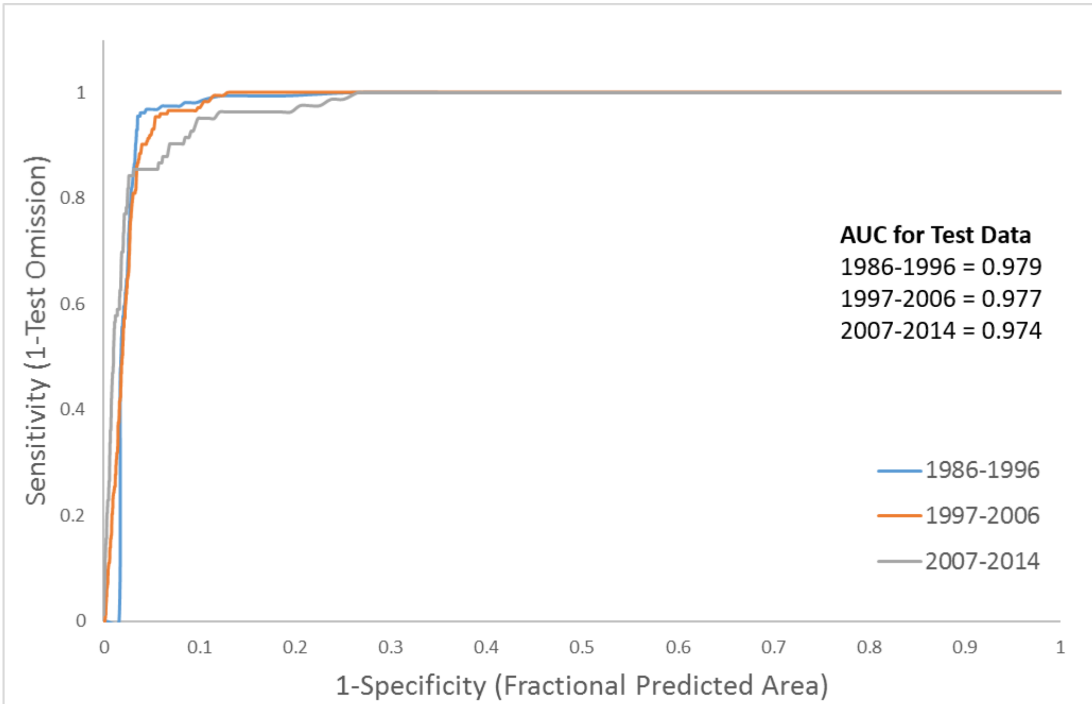



Figure 33. Receiver operator characteristic (ROC) plot for separate time series. Area under the curve (AUC) scores are displayed to compare the three time frames. An AUC score above 0.75 is "potentially useful" (Elith 2002).

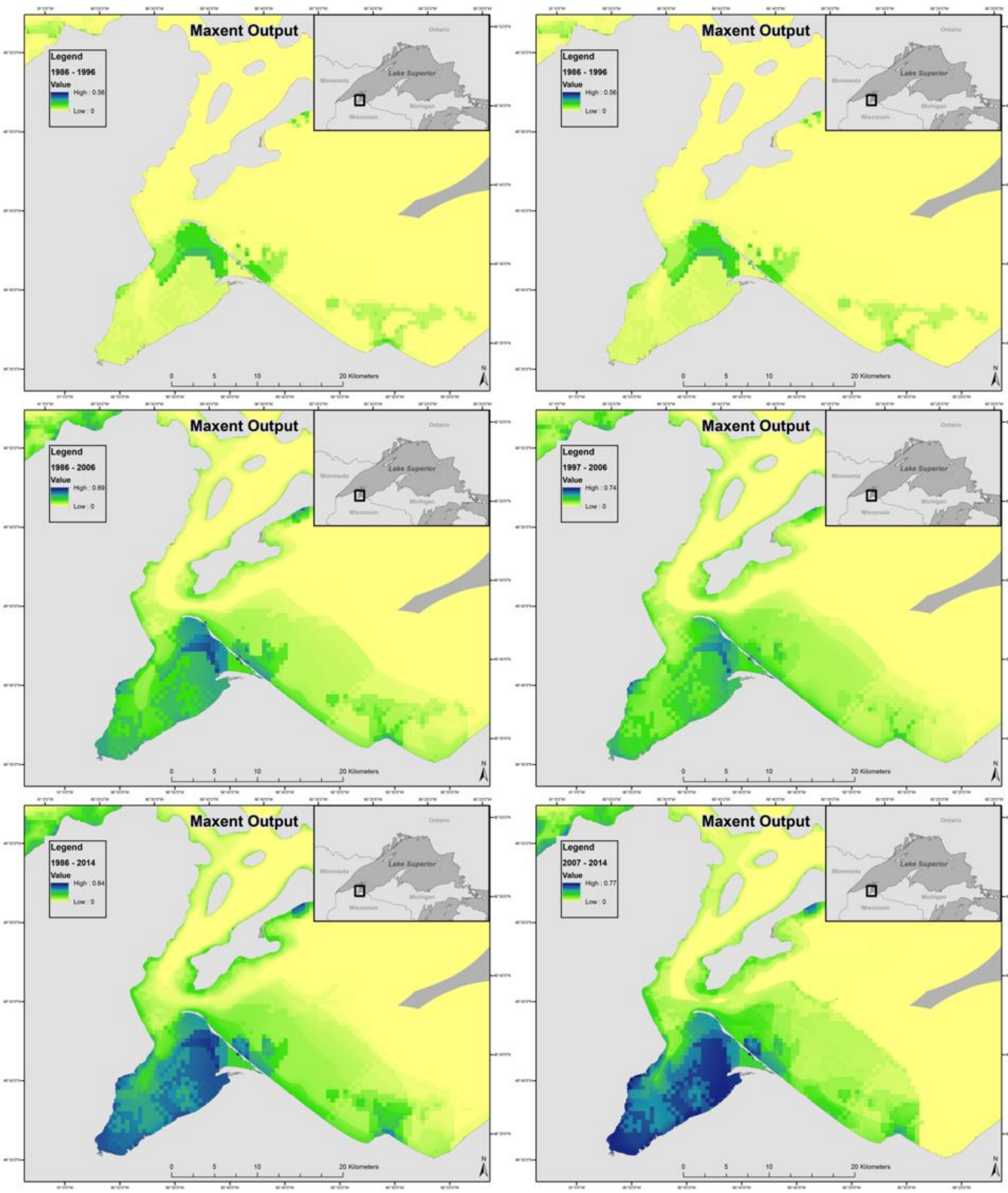



Figure 34. Chequamegon Bay for the cumulative (column 1) and separate (column 2) time series analyses. These are Maxent output predictive maps of suitable habitat. The dark regions represent high suitability and the light regions represent no suitability.

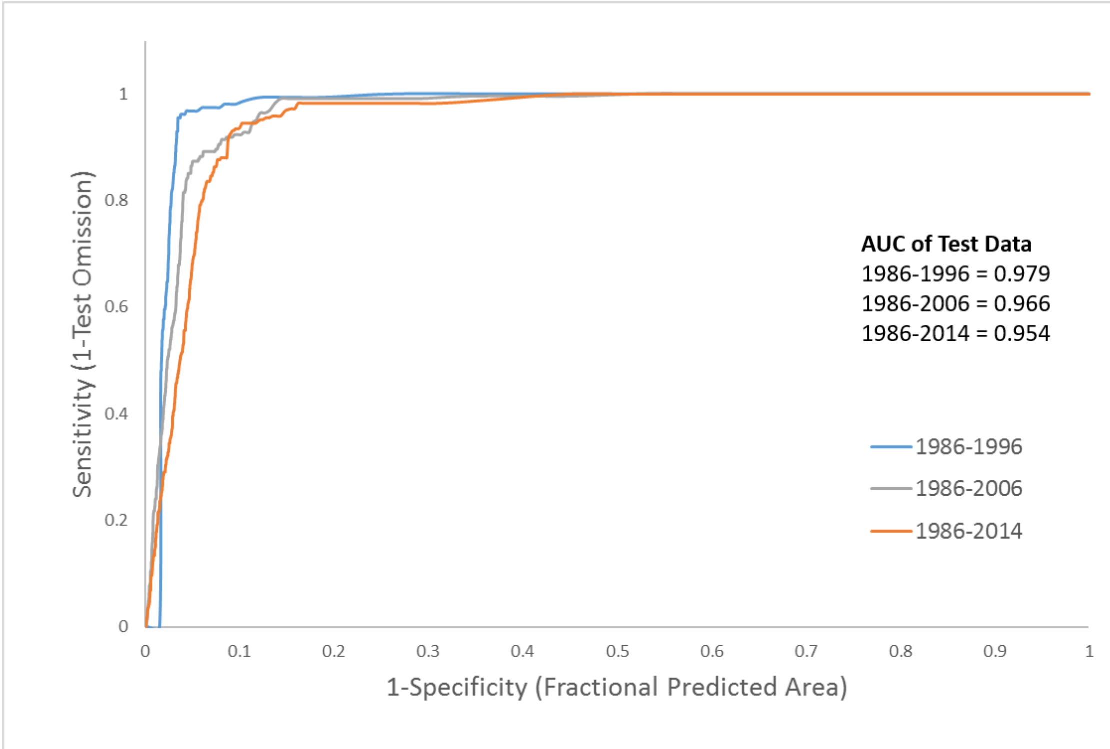



Figure 35. Receiver operator characteristic (ROC) plot for cumulative time series. Area under the curve (AUC) scores are displayed to compare the three time frames. An AUC score above 0.75 is “potentially useful” (Elith 2002).

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# Appendices

## Tables

Table A-1. Time-series catch per unit effort (CPUE) data, natural logarithm-transformed (ln(CPUE+1)), for St. Louis River, MN/WI from 1993-2015 (Chapter 2). Competitor and Ruffe (*Gymnocephalus cernua*) data was collected via a bottom trawl (USFWS) and predator data was collected via gill nets (MNDNR) (see methods of Chapter 2 for details). Gaps in the data were imputed using a cubic spline method.

| Year | Ruffe | Emerald Shiner | Johnny Darter | Round Goby | Spottail Shiner | Trout Perch | Yellow Perch | Muskellunge | Smallmouth Bass | Northern Pike | Walleye |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1993 | 6.381 | 3.990 | 2.263 | 0.000 | 4.110 | 4.397 | 3.377 | 0.047 | 0.047 | 1.204 | 1.204 |
| 1994 | 6.746 | 5.050 | 1.992 | 0.000 | 4.734 | 5.183 | 4.672 | 0.074 | 0.143 | 1.019 | 1.669 |
| 1995 | 7.576 | 5.060 | 1.986 | 0.000 | 5.461 | 5.989 | 5.570 | 0.091 | 0.047 | 1.362 | 1.580 |
| 1996 | 7.271 | 4.069 | 1.945 | 0.000 | 5.370 | 5.667 | 5.021 | 0.134 | 0.091 | 1.551 | 1.700 |
| 1997 | 7.304 | 4.358 | 1.794 | 0.000 | 5.172 | 5.347 | 5.814 | 0.179 | 0.140 | 1.598 | 1.708 |
| 1998 | 7.332 | 4.089 | 1.907 | 2.154 | 5.726 | 4.959 | 5.661 | 0.174 | 0.214 | 1.540 | 1.743 |
| 1999 | 6.992 | 5.170 | 1.823 | 3.206 | 5.668 | 4.617 | 3.785 | 0.091 | 0.341 | 1.386 | 1.977 |
| 2000 | 6.990 | 5.120 | 1.821 | 3.260 | 5.905 | 5.303 | 3.238 | 0.047 | 0.389 | 1.232 | 2.192 |
| 2001 | 6.927 | 5.293 | 3.194 | 2.887 | 5.782 | 5.168 | 3.734 | 0.145 | 0.283 | 1.196 | 2.244 |
| 2002 | 7.043 | 4.583 | 2.624 | 3.647 | 5.029 | 5.293 | 3.408 | 0.214 | 0.251 | 1.130 | 2.067 |
| 2003 | 7.020 | 4.426 | 2.125 | 3.327 | 4.369 | 4.889 | 4.918 | 0.140 | 0.470 | 0.956 | 1.629 |
| 2004 | 6.348 | 4.132 | 2.670 | 4.434 | 4.894 | 5.132 | 4.128 | 0.095 | 0.531 | 1.224 | 1.649 |
| 2005 | 6.503 | 4.432 | 3.190 | 5.961 | 5.080 | 5.751 | 4.137 | 0.047 | 0.251 | 1.530 | 1.897 |
| 2006 | 6.874 | 5.190 | 3.402 | 6.460 | 6.101 | 6.237 | 4.474 | 0.049 | 0.588 | 1.224 | 1.946 |
| 2007 | 6.701 | 5.676 | 3.470 | 6.402 | 6.978 | 6.538 | 3.406 | 0.070 | 0.479 | 0.970 | 2.001 |
| 2008 | 6.486 | 5.340 | 3.970 | 6.506 | 7.034 | 6.652 | 3.268 | 0.147 | 0.100 | 0.927 | 2.046 |
| 2009 | 6.380 | 3.813 | 4.313 | 6.622 | 6.585 | 6.775 | 4.137 | 0.223 | 0.095 | 0.811 | 1.981 |
| 2010 | 6.291 | 2.650 | 4.205 | 6.387 | 5.584 | 7.058 | 4.240 | 0.047 | 0.174 | 0.566 | 1.743 |
| 2011 | 5.931 | 4.557 | 3.122 | 5.064 | 4.408 | 7.073 | 3.025 | 0.001 | 0.163 | 0.439 | 1.615 |
| 2012 | 3.902 | 5.829 | 1.709 | 3.223 | 4.454 | 6.161 | 2.365 | 0.104 | 0.100 | 0.574 | 1.648 |
| 2013 | 5.041 | 7.611 | 2.569 | 3.693 | 4.937 | 8.112 | 2.739 | 0.174 | 0.047 | 0.907 | 1.718 |
| 2014 | 4.613 | 6.036 | 1.312 | 3.187 | 5.264 | 7.738 | 4.489 | 0.134 | 0.047 | 1.204 | 1.838 |
| 2015 | 4.532 | 6.099 | 0.000 | 4.564 | 5.283 | 6.781 | 2.678 | 0.134 | 0.047 | 1.016 | 2.326 |

Table A-2. Time-series catch per unit effort (CPUE) data, natural logarithm-transformed (ln(CPUE+1)), for Chequamegon Bay, WI, USA from 1993-2015 (Chapter 2). Competitor and Ruffe (*Gymnocephalus cernua*) data was collected via a bottom trawl (USFWS) and predator data was collected via creel surveys (see methods of Chapter 2 for details). Gaps in the data were imputed using a cubic spline method.

| Year | Ruffe | Emerald Shiner | Johnny Darter | Spottail Shiner | Trout Perch | Yellow Perch | Northern Pike | Walleye |
|---|---|---|---|---|---|---|---|---|
| 1993 | 0.000 | 3.291 | 2.634 | 3.865 | 4.065 | 2.468 | 0.045 | 6.772 |
| 1994 | 0.061 | 2.703 | 2.721 | 4.328 | 5.077 | 4.430 | 0.051 | 7.321 |
| 1995 | 0.000 | 0.000 | 3.449 | 2.785 | 3.857 | 3.837 | 0.030 | 4.554 |
| 1996 | 0.000 | 2.288 | 2.657 | 3.999 | 3.507 | 4.691 | 0.038 | 6.928 |
| 1997 | 0.061 | 1.201 | 1.895 | 2.525 | 4.142 | 2.479 | 0.049 | 4.522 |
| 1998 | 1.895 | 4.457 | 2.379 | 4.794 | 4.314 | 5.594 | 0.024 | 4.644 |
| 1999 | 0.531 | 0.531 | 2.122 | 3.300 | 2.715 | 4.862 | 0.000 | 0.000 |
| 2000 | 0.000 | 0.000 | 0.000 | 0.000 | 2.323 | 0.000 | 0.033 | 6.279 |
| 2001 | 0.000 | 0.963 | 0.462 | 0.000 | 3.057 | 0.375 | 0.124 | 5.991 |
| 2002 | 1.112 | 0.000 | 0.732 | 0.000 | 1.702 | 0.000 | 0.048 | 4.727 |
| 2003 | 1.926 | 0.331 | 0.536 | 3.615 | 5.097 | 1.455 | 0.009 | 3.892 |
| 2004 | 0.919 | 0.000 | 0.000 | 0.000 | 3.279 | 3.352 | 0.017 | 3.807 |
| 2005 | 3.937 | 0.000 | 0.000 | 0.000 | 4.274 | 0.000 | 0.059 | 7.771 |
| 2006 | 3.622 | 0.000 | 0.000 | 0.000 | 4.044 | 2.418 | 0.015 | 4.248 |
| 2007 | 0.000 | 0.000 | 0.681 | 0.000 | 2.362 | 3.234 | 0.015 | 6.297 |
| 2008 | 1.007 | 0.000 | 0.000 | 1.677 | 3.383 | 1.754 | 0.089 | 7.265 |
| 2009 | 1.686 | 0.000 | 3.610 | 0.307 | 2.157 | 3.510 | 0.013 | 4.043 |
| 2010 | 4.472 | 1.821 | 4.194 | 3.166 | 4.187 | 4.404 | 0.009 | 3.466 |
| 2011 | 5.433 | 3.020 | 3.312 | 5.513 | 3.258 | 5.168 | 0.028 | 7.011 |
| 2012 | 5.312 | 4.042 | 3.200 | 4.844 | 3.708 | 5.053 | 0.015 | 4.369 |
| 2013 | 1.897 | 0.107 | 2.902 | 1.504 | 1.736 | 5.697 | 0.010 | 3.664 |
| 2014 | 3.182 | 3.029 | 3.951 | 4.217 | 3.261 | 4.573 | 0.032 | 7.375 |
| 2015 | 3.396 | 0.000 | 2.336 | 0.646 | 2.798 | 4.062 | 0.023 | 4.804 |

Table A-3. Raw time-series catch per unit effort (CPUE) data for St. Louis River, MN/WI, USA from 1993-2015. Competitor and Ruffe (*Gymnocephalus cernua*) data was collected via a bottom trawl (USFWS) and predator data was collected via gill nets (MNDNR) (see methods of Chapter 2 for details). Data has not been imputed or log-transformed (Chapter 2).

| Year | Emerald Shiner | Johnny Darter | Round Goby | Ruffe | Spottail Shiner | Trout Perch | Yellow Perch | Muskellunge | Northern Pike | Smallmouth Bass | Walleye |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1993 | 53.067 | 8.616 | 0.000 | 589.258 | 59.917 | 80.214 | 28.269 | 0.048 | 2.333 | 0.048 | 2.333 |
| 1994 | 155.065 | 6.329 | 0.000 | 849.555 | 112.756 | 177.298 | 105.886 | 0.077 | 1.769 | 0.154 | 4.308 |
| 1995 | 156.537 | 6.285 | 0.000 | 1950.291 | 234.370 | 398.129 | 261.475 | 0.095 | 2.905 | 0.048 | 3.857 |
| 1996 | 57.514 | 5.995 | 0.000 | 1436.605 | 213.848 | 288.040 | 150.575 | 0.143 | 3.714 | 0.095 | 4.476 |
| 1997 | 77.138 | 5.016 | 0.000 | 1485.162 | 175.334 | 208.966 | 333.943 | | | | |
| 1998 | 58.653 | 5.733 | 7.622 | 1527.257 | 305.746 | 141.441 | 286.525 | 0.190 | 3.667 | 0.238 | 4.714 |
| 1999 | 174.965 | 5.191 | 23.684 | 1086.615 | 288.419 | 100.182 | 43.044 | | | | |
| 2000 | 166.263 | 5.177 | 25.048 | 1084.519 | 366.047 | 199.866 | 24.475 | 0.048 | 2.429 | 0.476 | 7.952 |
| 2001 | 197.962 | 23.397 | 16.935 | 1018.504 | 323.384 | 174.533 | 40.829 | | | | |
| 2002 | 96.809 | 12.790 | 37.370 | 1144.294 | 151.798 | 197.850 | 29.204 | 0.238 | 2.095 | 0.286 | 6.905 |
| 2003 | 82.630 | 7.373 | 26.867 | 1117.847 | 77.971 | 131.831 | 135.726 | 0.150 | 1.600 | 0.600 | 4.100 |
| 2004 | 61.319 | 13.434 | 83.260 | 570.108 | 132.511 | 168.299 | 61.061 | 0.100 | 2.400 | 0.700 | 4.200 |
| 2005 | | | | | | | | 0.048 | 3.619 | 0.286 | 5.667 |
| 2006 | 178.514 | 29.034 | 637.747 | 966.136 | 445.420 | 510.100 | 86.732 | 0.050 | 2.400 | 0.800 | 6.000 |
| 2007 | 290.642 | 31.140 | 601.875 | 811.931 | 1071.823 | 690.106 | 29.132 | | | | |
| 2008 | | | | | | | | 0.158 | 1.526 | 0.105 | 6.737 |
| 2009 | | | | | | | | 0.250 | 1.250 | 0.100 | 6.250 |
| 2010 | 13.159 | 65.997 | 593.018 | 538.523 | 265.244 | 1161.175 | 68.408 | 0.048 | 0.762 | 0.190 | 4.714 |
| 2011 | 94.325 | 21.698 | 157.208 | 375.490 | 81.065 | 1179.106 | 19.588 | | | | |
| 2012 | 339.026 | 4.521 | 24.109 | 48.518 | 84.983 | 472.828 | 9.643 | | | | |
| 2013 | 2018.484 | 12.055 | 39.177 | 153.692 | 138.322 | 3333.300 | 14.465 | 0.190 | 1.476 | 0.048 | 4.571 |
| 2014 | 417.077 | 2.712 | 23.205 | 99.749 | 192.265 | 2293.922 | 87.997 | 0.143 | 2.333 | 0.048 | 5.286 |
| 2015 | 444.500 | 0.000 | 94.927 | 91.914 | 195.881 | 879.960 | 13.561 | 0.143 | 1.762 | 0.048 | 9.238 |

Table A-4. Raw time-series catch per unit effort (CPUE) data for Chequamegon Bay, WI, from 1993-2015. Competitor and Ruffe (*Gymnocephalus cernua*) data was collected via a bottom trawl (USFWS) and predator data was collected via creel surveys (see methods of Chapter 2 for details). Data has not been imputed or log-transformed (Chapter 2).

| Year | Walleye | Northern Pike | Smallmouth Bass | Muskellunge | Emerald Shiner | Johnny Darter | Ruffe | Spottail Shiner | Trout Perch | Yellow Perch |
|---|---|---|---|---|---|---|---|---|---|---|
| 1993 | 872.000 | 149.000 | 421.000 | 0.000 | 25.871 | 12.936 | 0.000 | 46.719 | 57.269 | 10.801 |
| 1994 | 1511.000 | 470.000 | 0.000 | 0.000 | 13.919 | 14.192 | 0.063 | 74.809 | 159.372 | 82.909 |
| 1995 | 94.000 | 0.000 | 5.000 | 0.000 | 0.000 | 30.456 | 0.000 | 15.196 | 46.342 | 45.401 |
| 1996 | 1019.000 | 137.000 | 0.000 | 0.000 | 8.854 | 13.251 | 0.000 | 53.564 | 32.339 | 108.006 |
| 1997 | 91.000 | 66.000 | 0.000 | 0.000 | 2.323 | 5.652 | 0.063 | 11.492 | 61.915 | 10.926 |
| 1998 | 103.000 | 43.000 | 0.000 | 0.000 | 85.212 | 9.796 | 5.652 | 119.749 | 73.721 | 267.755 |
| 1999 | 0.000 | 0.000 | 0.000 | 0.000 | 0.700 | 7.348 | 0.700 | 26.123 | 14.110 | 128.280 |
| 2000 | 532.000 | 275.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 9.209 | 0.000 |
| 2001 | 399.000 | 97.000 | 0.000 | 0.000 | 1.620 | 0.588 | 0.000 | 0.000 | 20.261 | 0.455 |
| 2002 | 112.000 | 72.000 | 0.000 | 0.000 | 0.000 | 1.079 | 2.041 | 0.000 | 4.486 | 0.000 |
| 2003 | 48.000 | 5.000 | 0.000 | 0.000 | 0.392 | 0.709 | 5.863 | 36.135 | 162.533 | 3.284 |
| 2004 | 44.000 | 9.000 | 0.000 | 0.000 | 0.000 | 0.000 | 1.507 | 0.000 | 25.553 | 27.556 |
| 2005 | 2370.000 | 312.000 | 0.000 | 0.000 | 0.000 | 0.000 | 50.270 | 0.000 | 70.840 | 0.000 |
| 2006 | 69.000 | 101.000 | 0.000 | 0.000 | | | | | | |
| 2007 | 542.000 | 88.000 | 0.000 | 0.000 | 0.000 | 0.975 | 0.000 | 0.000 | 9.615 | 24.385 |
| 2008 | 1428.000 | 57.000 | 0.000 | 0.000 | 0.000 | 0.000 | 1.738 | 4.348 | 28.470 | 4.780 |
| 2009 | 56.000 | 10.000 | 0.000 | 0.000 | 0.000 | 35.966 | 4.398 | 0.360 | 7.644 | 32.434 |
| 2010 | 31.000 | 0.000 | 0.000 | 0.000 | 5.180 | 65.269 | 86.554 | 22.707 | 64.850 | 80.791 |
| 2011 | 1108.000 | 329.000 | 0.000 | 0.000 | 19.493 | 26.448 | 227.729 | 246.788 | 25.006 | 174.574 |
| 2012 | 78.000 | 272.000 | 0.000 | 0.000 | 55.966 | 23.525 | 201.809 | 125.999 | 39.756 | 155.445 |
| 2013 | 38.000 | 57.000 | 0.000 | 0.000 | 0.112 | 17.204 | 5.667 | 3.501 | 4.675 | 297.057 |
| 2014 | 1595.000 | 394.000 | 0.000 | 0.000 | 19.675 | 50.967 | 23.086 | 66.829 | 25.064 | 95.819 |
| 2015 | 121.000 | 37.000 | 0.000 | 0.000 | 0.000 | 9.338 | 28.853 | 0.908 | 15.412 | 57.065 |

Table A-5. Calculations of predicted percent area from the Maxent model (Chapter 4). Area is in meters divided by meters of the buffer and multiplied by 100 to get the percentage.

| | **Percent area predicted from buffer** | **Percent area predicted from Lake Superior** | **Depth (m)** | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | <30 | | <100 | | >100 | |
| | | | Count | Area ($m^2$) | Count | Area ($m^2$) | Count | Area ($m^2$) |
| **Full adult model** | (5254259/ 36413732)*100 = **14%** | (4728833100/82 097000000)*100 = **5.76%** | (4812688/36413732 )*100 = **13.22%** | 43314192 00 | (365269/36413732)* 100 = **1.00%** | 32874210 0 | (76302/36413732 )*100 = **0.21%** | 686718 00 |
| **250 m** | (2168114/ 36413732)*100 = **6%** | (1951302600/82 097000000)*100 = **2.38%** | (2160894/36413732 )*100 = **5.93%** | 19448046 00 | (7220/36413732)*10 0 = **0.020%** | 6498000 | | |
| **500 m** | (3147913/ 36413732)*100 = **8%** | (2833121700/82 097000000)*100 = **3.45%** | (3090323/36413732 )*100 = **8.49%** | 27812907 00 | (57589/36413732)*1 00 = **0.16%** | 51830100 | (1/36413732)*10 0 = **2.75e-6%** | 900 |
| **1000 m** | (7452951/ 36413732)*100 = **20%** | (6707655900/82 097000000)*100 = **8.17%** | (5964854/36413732 )*100 = **16.38%** | 53683686 00 | (1275697/36413732 )*100 = **3.50%** | 11481273 00 | (212400/3641373 2)*100 = **0.58%** | 191160 000 |
| **2000 m** | (6415621/ 36413732)*100 =**17%** | (5774058900/82 097000000)*100 = **7.03%** | (5646215/36413732 )*100 = **15.51%** | 50815935 00 | (769363/36413732)* 100 = **2.11%** | 69242670 0 | (43/36413732)*1 00 = **1.18e-4%** | 38700 |
| **2000 m selected removal** | (7935392/ 36413732)*100 = **22%** | (7141852800/82 097000000)*100 = **8.70%** | (5628547/36413732 )*100 = **15.46%** | 50656923 00 | (1230191/36413732 )*100 = **3.38%** | 11071719 00 | (1076654/364137 32)*100 = **2.96%** | 968988 600 |

## Figures

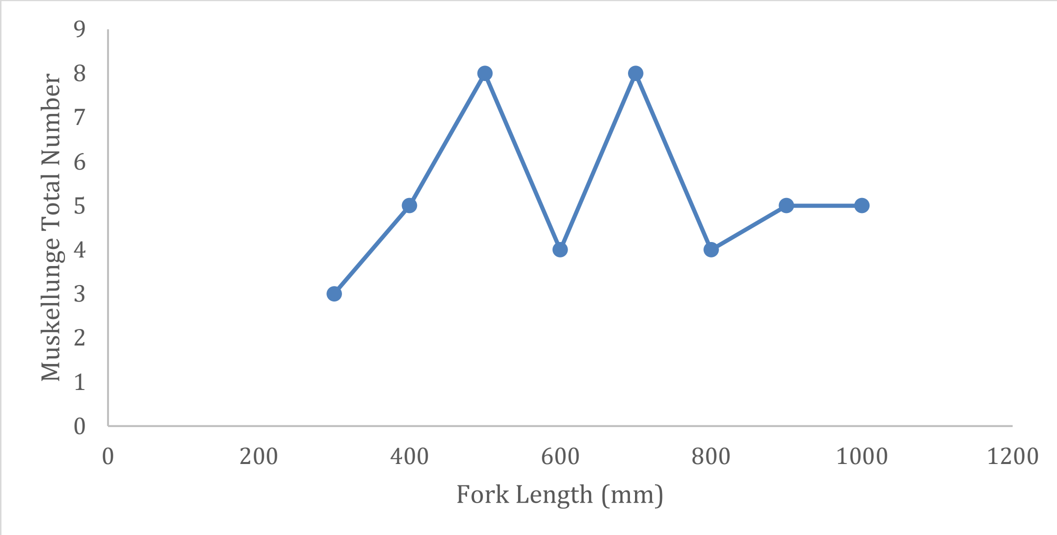


Figure A-1. Catch curve for Muskellunge (*Esox masquinongy*) in Minnesota Department of Natural Resources' gill nets from 1993-2015 (Chapter 2). Data was calculated using Microsoft Excel.

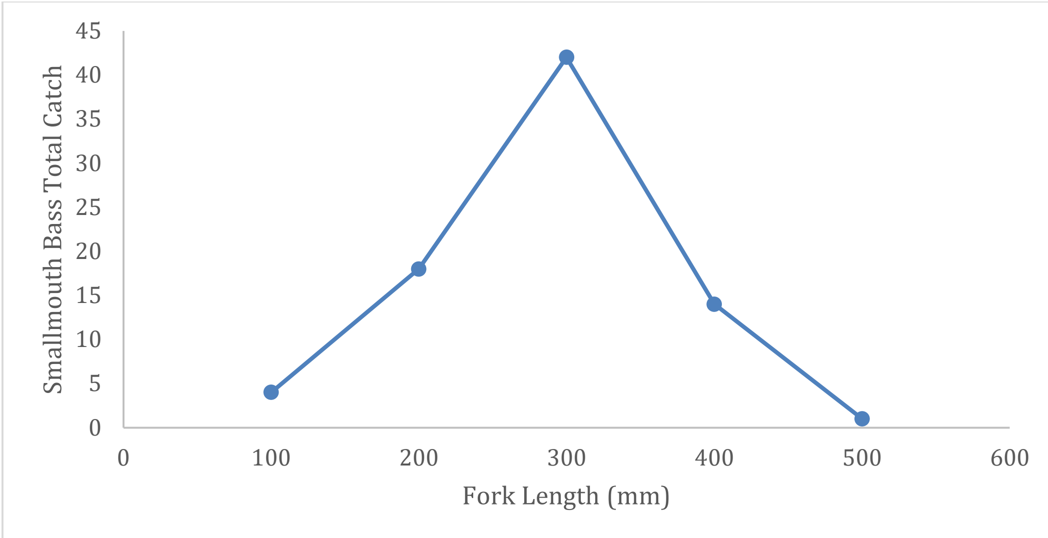


Figure A-2. Catch curve for Smallmouth Bass (*Micropterus dolomieu*) in Minnesota Department of Natural Resources' gill nets from 1993-2015 (Chapter 2). Data was calculated using Microsoft Excel.

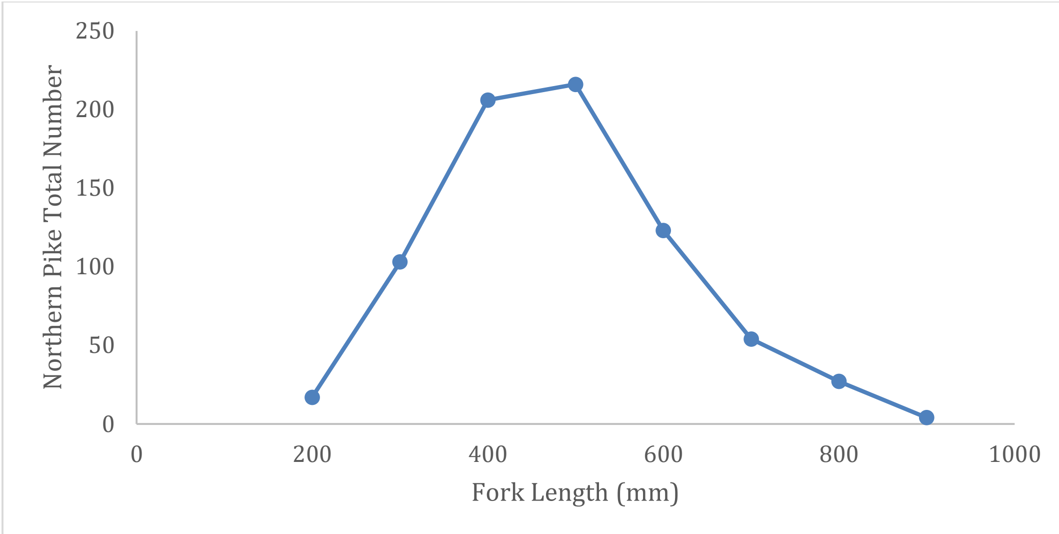


Figure A-3. Catch curve for Northern Pike (*Esox lucius*) in Minnesota Department of Natural Resources' gill nets from 1993-2015 (Chapter 2). Data was calculated using Microsoft Excel.

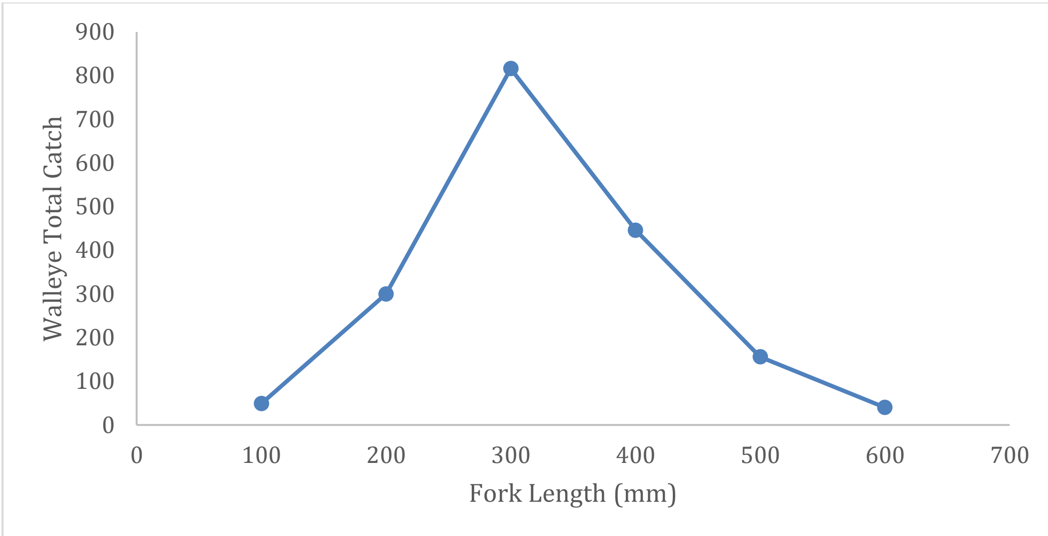


Figure A-4.  Catch curve for Walleye (*Sander vitreus*) in Minnesota Department of Natural Resources' gill nets from 1993-2015 (Chapter 2). Data was calculated using Microsoft Excel.

# Appendix F

# Evans (2016) St. Louis River AOC Semi-aquatic Mammal Report

# REPORT: STATUS OF SEMI-AQUATIC MAMMALS IN THE ST. LOUIS RIVER AREA OF CONCERN

SEPTEMBER 2016

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Department of Forest & Wildlife Ecology
University of Wisconsin – Madison

Beaver (*Castor canadensis*)

River otter (*Lutra canadensis*)

Mink (*Neovison vison*)

Muskrat (*Ondatra zibethicus*)

## Abstract

This document summarizes the findings from research conducted to address Beneficial Use Impairments in the United State EPA designated St Louis River Area of Concern, specifically *BUI 2: Degraded Fish and Wildlife Populations* and *BUI 9: Loss of Wildlife Habitat* in regards to native mammals. Methods include aerial surveys for beaver, muskrat and otter sign, and trail camera surveys for beaver, muskrat, otter and mink.

The research was funded by a Great Lakes Restoration Initiative grant #GL-00E01312 Sub 3. It is a collaboration between the University of Wisconsin-Madison and the Wisconsin Department of Natural Resources, along with the Minnesota Pollution Control Agency, the Fond du Lac Band of Lake Superior Chippewa, and numerous local, state and federal agencies and staff. The data are also the focus of a master's thesis in the Department of Forest and Wildlife Ecology at UW Madison.

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# 1 Introduction

Expansion of human populations and industrial activity have impacted ecosystems and compromised their ability to support both natural and human wellbeing (Mills 2013). Functional riverine and wetland systems provide numerous ecosystem services, including water purification, flood control, wildlife habitat and aesthetic value, but since the turn of the twentieth century 50% of North American wetlands have undergone moderate to severe modification (Millennium Ecosystem Assessment 2005, Thorp et al. 2010). The St. Louis River, which flows into Lake Superior and forms the harbor between Duluth, Minnesota and Superior, Wisconsin, is significantly degraded as a result of human activities (MPCA and WDNR 1992*a*). Land use changes beginning in the 1860's included timber clearing and milling and installing railroad facilities (Kellner et al. 2000). The expansion of the transshipment industry created incentive for extensive modifications of the harbor and estuary, and as early as 1873 dredging for channels contributed to habitat degradation and sediment loading that are cause for concern to date (Kellner et al. 2000, MPCA 2015). Effects of industrialization in the area include extensive loss of wetlands, transformation of the benthic environment, and physical and chemical pollutants which damaged both the physical structure and ecological functioning of the river and estuary (MPCA and WDNR 1995, MPCA 2013).

In 1987, the St. Louis River was designated as an Area of Concern (AOC) in the Great Lakes Water Quality Agreement between the United States and Canada MPCA and WDNR 1992). Nine Beneficial Use Impairments (BUIs) were listed, and both rehabilitating human use and returning natural stability to the system were identified as management goals. Two BUIs pertain specifically to wildlife management: BUI 2 *Degradation of Fish and Wildlife Populations*, and BUI 9 *Loss of Fish and Wildlife Habitat* (MPCA 2013). There have been numerous collaborative remediation projects within the AOC, including work targeted to restore spawning habitat for Lake sturgeon and beach improvements for Piping plover (MPCA 2013), but to date there has been no direct assessment of the status of native mammals in the system.

The primary objective of this research was to determine if the St. Louis River AOC currently supports populations of native mammal species in similar abundances as areas with less extensive impairment. Complete population censuses are generally not feasible in wildlife research; but by performing surveys and estimating a probability of detection, relative occupancy estimates can be generated to answer similar ecologically relevant questions (MacKenzie et al. 2002, 2006). For this study, the state variables addressed by occupancy modeling (primarily the proportion of sampling units occupied by species of interest, MacKenzie et al. 2006) will be sufficient to meet the objectives set in place by the Remediation Action Plan (RAP, MPCA 2013). Regarding the Degradation of Fish and Wildlife Populations BUI the RAP states that “Removal of this BUI is not dependent on specific small aquatic mammal population numbers. However, to support development of concurrence among state resource management agencies, a small mammal survey will be conducted in the estuary to verify that populations are not limited by physical habitat, food sources, water quality, or contaminated sediments” (MPCA 2013).

To thoroughly assess the status of mammal populations in the area, four native species dependent on aquatic resources were selected to study: river otter (*Lontra canadensis*), mink (*Neovison vison*), beaver (*Castor canadensis*) and muskrat (*Ondatra zibethicus*). I collected data on occurrence of these species using both motion triggered camera surveys and aerial sign surveys, with the two goals of 1) estimating relative occupancies of all four species to provide adequate information for assessing the BUIs and 2) comparing between survey methods for relative costs, efficacy and potential biases.

## 2 Semi-aquatic Mammals

The four target species for this research span several trophic levels and have distinct roles within aquatic ecosystems, which presents both unique risks for species decline linked to industrialization as well benefits linked to species recovery. Both river otter and mink are carnivorous and therefore dependent upon access to reliable prey resources to occupy an area permanently (Buskirk and Zielinski 2003). They are two of the three mammals listed as representative species by the Great Lakes Water Quality Initiative wildlife criteria (United States Environmental Protection Agency 1995) because of their sensitivity to heavy metals and polychlorinated biphenyls (PCBs) in the environment. In an area with a history of chemical pollution such as the AOC, these compounds can bioaccumulate and lead to decreased survival and reproduction (Wren 1987, Poole et al. 1998, Mayack and Loukmas 2001). Buskirk and Zielinski (2003) compiled multiple lines of evidence (lesions, absence from areas with contaminants) showing that carnivores in aquatic systems are at potentially great risk from chronic exposure pesticides, heavy metals and other pollutants, and diminished populations have been demonstrated within other Areas of Concern (Letteros et al. 2008, Strom 2013). Otters are piscivores primarily, although supplementing with amphibians or small mammals, and incorporating crayfish in potentially large proportions seasonally (Roberts et al. 2008). The smaller mink eat a wider variety of prey including fish, crustaceans, and mammals, with muskrats being especially important during the winter (Kurta 1952, Melquist et al. 2003). Mink can also provide an indication of contamination in aquatic ecosystems, being high on the trophic chain, yet relatively short lived and occupying small home ranges increases their sensitivity (Larivière 2003, New York State 2010).

All of the four target species are regulated as furbearers, and experienced extensive reductions in population size following European settlement of the area (WDNR 2012). Increasing regulation of trapping began to emerge in the late 1800’s, and species have recovered in many areas and are routinely monitored using multiple methodologies (Kohn and Ashbrenner 1984, Rolley and MacFarland 2012, WDNR 2012).

Beavers were nearly extirpated from much of the great lakes region by 1900 due to unregulated trapping for the fur trade (WDNR 2012). Once trapping was regulated, populations throughout the area rebounded without human intervention; by 1990 beavers in northern Wisconsin were abundant enough that subsidies were offered to trappers to assist the Wisconsin DNR in reducing the population size (WDNR 2015). Beaver management zones were established to balance between the negative consequences of their habitat modifying behavior on trout populations and flooding of roadways, crops and private property with the benefits provided, particularly for waterfowl species (WDNR 1990). Beaver populations are monitored using specialized questionnaire replies from resident trappers to provide harvest information (Dhuey and Olson 2014a) and in the in the northern management zones, which have higher beaver densities, by using aerial surveys (Rolley et al. 2011). In recovering ecosystems beaver reintroductions have been used as a restoration tool to increase habitat complexity, connectivity and retention of water during drought conditions (Hood and Larson 2015).

Muskrats and mink are now also trapped extensively, with no bag limit lengthy trapping seasons in Wisconsin (late October/early November to early March, WDNR 2014). Populations within the state are tracked using questionnaires from registered trappers, which indicate that around half of surveyed trappers target muskrats, resulting in annual harvests around two to three hundred thousand, while fewer trappers target mink and harvest between ten and twenty thousand (Dhuey and Olson 2013, Dhuey and Olson 2014b). Muskrats are a major food source for mink and are prey for raccoons and terrestrial predators - as a prey species they reproduce more swiftly than the other species of interest to this research and are capable of bearing litters of young each year (Erb and Perry Jr. 2003).

## 3 Study Areas

The St. Louis River AOC in total encompasses the lower 63 kilometers (39 miles) of the St. Louis River, the associated watershed containing multiple tributaries and adjacent streams, and the Nemadji river watershed. The majority of remediation work has occurred on the stretch of the St. Louis River below the Fond du Lac dam, and in the Nemadji River watershed, where the Koppers Company lumber processing plant at Crawford Creek caused point source pollution from as early as 1928 to today, though since 1991 waste water has been transported offsite for treatment (MPCA 1992, 2013; MPCA and WDNR 1992). Therefore this research focused on those areas, specifically the St. Louis River from the Fond du Lac dam to the Bong Bridge, and the Nemadji River from six miles above Crawford Creek to its outlet in Allouez Bay (see Figure 1).

Because there is no information available on semi-aquatic mammal population status in the St. Louis River estuary prior to degradation, this study design uses reference areas in order to determine if the current population status for each target species meets the recovery requirements designated by the EPA. I and fellow researchers located two distinct reference sites to adequately reflect the diversity of habitats and flow regimes present in the St. Louis AOC: the Boulder Lake Reservoir in northeastern Minnesota, as an example of a relatively unimpaired lentic system, and the St. Croix River on the north central border between Wisconsin and Minnesota, to represent a relatively unimpaired lotic system. Both of these areas were deemed sound representations of the ecological potential of the AOC by meeting the criteria that they 1) possess similar habitat types to the St. Louis River estuary and are likely to support populations of the target species, based on expert opinion, 2) have minimal anthropogenic impacts including development and point sources of pollution along the shoreline, 3) are at least partially open to public trapping and are accessible by several means and 4) are geographically close to the AOC (<100 kilometers maximum linear distance) without being contained within it. Because the St. Louis AOC has an industrial component that will not be removed or restored, the reference areas populations will not be used as specific goals for AOC populations, but rather provide points for comparison and ultimately establish a basis for the consensus decision by resource managers about the status of semi-aquatic mammal populations in the AOC.

## 3.1 St. Louis River and Estuary

To ensure consistent effort and draw accurate inferences, I defined the boundaries for semi-aquatic mammal surveys within the AOC to extend from the Fond du Lac dam downstream to the Richard I. Bong Memorial Bridge. This area encompasses the diversity of flow regimes and habitat types in the AOC, and corresponds to several of the remediation projects (e.g sediment removal and wetland restoration at Mud Lake; shoreline restoration at Chamber's Grove. MPCA 2013). Immediately below the dam the St Louis river follows a narrow channel with relatively faster flow (channel width ranging 50-400m). The river then widens into a slower, shallower channel and forms Mud Lake and Spirit Lake (maximum width over 2,000m). On the Wisconsin (southeast) side several small tributaries form Pokegama Bay, an estuary system with meandering channels and dense vegetation. North of Pokegama Bay, the habitat transitions into moderate residential development and progressively becomes more industrialized as one approaches the Bong Bridge. Beyond this point the shoreline is intensively modified and provides very little natural habitat suitable for semi-aquatic mammals.

## 3.2 Nemadji Watershed and Allouez Bay

The Nemadji River water lies just southeast of the St. Louis River, and enters Lake Superior adjacent to Wisconsin Point and Allouez Bay. The river channel is relatively sinuous and narrow (typically 25-50m) and may provide habitat for wildlife that is distinct from the St Louis River. To investigate the potential impact of industrial point source contamination this study area extends from 10 kilometers (roughly 6 miles) upstream of the Crawford Creek confluence and downstream to the river's mouth, approximately 12 kilometers. Allouez Bay (including the interior, southern shoreline of Wisconsin Point) is also included in this portion of the study area. The bay is also vulnerable to industrial pollution and has been targeted for wildlife habitat restoration (MPCA 2013).

## 3.3 Boulder Lake Reservoir

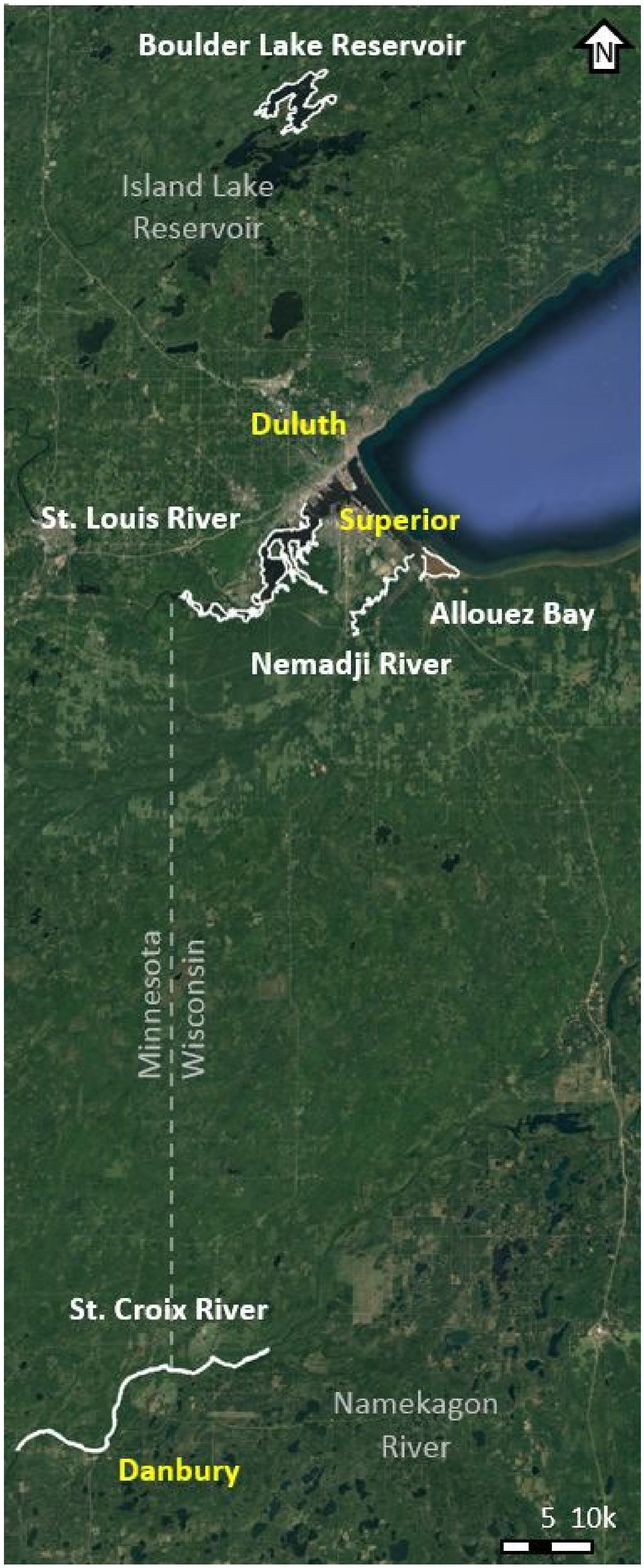
The Boulder Lake Reservoir, in northeastern Minnesota, is owned by Minnesota Power, ALLETE Inc and supports one small dam, but is otherwise minimally disturbed. The land is managed by the Boulder Lake Environmental Learning Center through the University of

Minnesota-Extension (www.boulderlake.org). Boulder Lake Reservoir was selected as a reference site out of several lakes in northeastern Minnesota because it has low levels of private development along the shoreline and can be assumed to represent minimally degraded habitat. Furthermore, Boulder Lake has open public access for boating, camping, fishing and hunting recreation which both mimics the St. Louis study area and facilitates access for this research. Specific access and logistic support for the project was granted through the Boulder Lake Environmental Learning Center by the program director.

### 3.4 St. Croix River

The St. Croix River runs from northwestern Wisconsin, and forms the border between Minnesota and Wisconsin until joining the Mississippi River below Minneapolis. It is designated as a National Scenic Riverway, and as such there is public access to a relatively undisturbed shoreline both by foot and by canoe. It is managed by the National Park Service (NPS), and while hunting is allowed trapping is prohibited on portions owned by the park (NPS 2006, WDNR 2014). However, trapping is allowed on other publically owned segments in both Minnesota and Wisconsin, and in Minnesota water sets between the mean high tide mark and the center of the channel(NPS 2006, MDNR 2016). Trapping is also allowed for privately held land with the owner’s permission and for tribal trappers exercising treaty rights. Furthermore, the removal of nuisance beaver is conducted by USDA Animal and Plant Health Inspection Service for safety concerns along the waterway as well as to enhance trout habitat (NPS 2006, WDNR 2012). As such, the trapping regime should be similar to that of the St. Louis River estuary and does not impede comparison between the two sites. This reference area includes the stretch of river north of Danbury, Wisconsin from the confluence of the Namekagon River downstream approximately 26 kilometers to Thayer’s Landing at the HWY 77 bridge / MN 173 junction. This portion of the river offers several points of access both on foot and by canoe, and is sufficient in size for comparable survey effort as the Boulder Lake reference area.

Figure 1. Map of AOC study areas (St. Louis River, Nemadji River, Allouez Bay) and reference sites (Boulder Lake Reservoir, St. Croix River)



# 4 Field Methods

Wildlife species are often cryptic and elusive, which make their populations difficult to quantify (O'Brien 2011). Complete counts of all individuals of a species are nearly impossible to obtain, and partial count data must be adjusted to account for imperfect detection (White 2005). For species that are harvested, some relevant data can be gathered through registration of carcasses or reports from hunters and trappers, but such data can be unreliable due to variation in harvest effort as recreational interest and market values change (Kohn and Ashbrenner 1984) and the often non-linear relationship between density and harvest (Van Deelen and Etter 2003). The natural resource management agencies of both Wisconsin (WDNR) and Minnesota (Minnesota Department of Natural Resources, MDNR) have used various techniques to monitor population trends for beaver, otter, mink and muskrat. Current monitoring techniques for these species typically combine harvest reports and related demographic data (Dhuey et al. 2015, Erb 2016) and winter sign surveys both on the ground and from aircraft for beaver and otter (Erb 2013, Rolley et al. 2011, Rolley et al. 2013). While historic and contemporary harvest information can be useful for tracking population trends around the AOC, more detailed and precise information was needed to meet the mandate of this research. Because aerial surveys are already used by management agencies for beaver and otter, aerial surveys in fall and winter were included in the study design to collect data on the status of these two species within the St. Louis AOC.

However, while muskrat sign is also visible from the air, there is no established protocol to collect information on muskrat or mink by aerial surveys. Furthermore, relying on a single methods for which estimates of precision and bias are not available could lead to flawed inference, therefore I investigated the use of multiple survey methods. Using more than one approach allows for simultaneous investigation of all four species, and offers quality control against potential flaws in a single approach, as well as providing information to management agencies on efficacy should future surveys be desired. Motion triggered cameras are increasingly popular in wildlife research, and rigorous methods for study design and analysis are widely accepted (O'Connell et al. 2011). Using trail cameras is effective for collecting data on multiple species simultaneously (Lesmeister et al. 2015), enables data collection beyond the seasonal restrictions for aerial surveys for beaver and otter, and can circumvent some potential biases present in other non-invasive methods. For example, other methods for surveying otters include scat surveys (Jeffress et al. 2011) which typically assess only one species at a time (but see Williamson and Clark 2011), can be impacted by seasonal differences in detection probability (Fusillo et al. 2007, Parry et al. 2013), and may have bias introduced by distance from anthropogenic structures or limited area coverage to account for false absences (Swimley et al. 1998, Crimmins et al. 2009). Cameras are also able to collect data at a fine spatial scale and can incorporate information on habitat features at microsites within the area of interest. Several dozen Reconyx brand trail cameras (Reconyx, Holman WI) were available for research use from the WDNR, and based on the above benefits I selected this approach for semi-aquatic mammal surveys.

Other non-invasive techniques considered were baited track plates (which have been used for many non-aquatic species, Zielinski et al. 1995) and floating rafts (often used for mink surveys in Europe, Schooley et al. 2012). After considering these techniques I determined that track plates would not encompass all the species of interest and could have complications in close proximity to water, and that although there is interest in floating rafts to assess recovery of mink in the Sheboygan Area of Concern (Natalie Miller, personal communication), that approach would be logistically infeasible to meet the objectives of this project.

### 4.1 Aerial survey methods

In order to efficiently track beaver populations on the landscape, many agencies in both North America and northern Europe have used aerial surveys (Payne 1981). Differences in detection success vary between fixed wing, helicopter and ground survey methods (e.g. Robel and Fox 1993). To conduct a thorough comparison of census-style flight surveys, Payne (1981) used both fixed wing (Super Cub) and helicopter surveys flown in a single day, and tested the results against "ground truthed" data from reliable trapper surveys. The results indicated that helicopters missed 19% of beaver lodges and fixed wings missed 39%. Aerial surveys within Wisconsin are included in the current ten-year beaver management plan, and include both helicopter surveys of selected quadrats and fixed wing flights over trout streams (WDNR 2015). However, no data exists on the survey efficacy for this approach in Wisconsin.

For this study, flight surveys used a Cessna fixed-wing four person aircraft provided by WDNR and flown by an agency pilot with experience conducting wildlife surveys. Although many wildlife surveys use fixed wing aircraft (Walsh et al. 2010, Jacques et al. 2014) they do have certain flight speed and height limitations that could theoretically impact detection rates for such surveys. To account for any resulting inadequacies with the data gathered from fixed wing surveys, one comparison survey was completed in a helicopter. Helicopters have increased maneuverability and the ability to fly slowly and hover, which allows for more precise surveys.

However, helicopters are significantly more expensive than fixed wing aircraft and as such specific insight into the feasibility of less expensive techniques is desirable.

Ideal timing for aerial surveys of beaver lodges is in the window between leaf-off and ice up, ranging in October and November, while otter sign surveys require ice cover and recent snow fall (WDNR 2015). The majority of our fixed wing flights occurred in synchrony with fall camera trapping sessions, which also corresponds to the protocol followed by Wisconsin DNR for beaver and river otter population estimates (Rolley et al 2011, Rolley et al 2013). An additional late flight in February 2016 (after closing camera stations) was conducted due to poor conditions for otter sign on earlier winter 2015/2016 flights.

Data collected on the flights included a GPS track logs of the flight path; starting and stopping locations and times for each segment of the surveys; and waypoints for any sign recorded. Observable sign for beaver consisted of lodges, food caches, and wood chips and downed trees as a result of chewing activity. Muskrat sign consisted of "push-ups" in the fall, which were generally not observable after snow had fallen. Otter sign consisted of tracks in snow, which were only observable during periods of ice cover and were discernable from other animal tracks by the distinctive sliding pattern in the snow.

## 4.2 Camera survey methods

Reconyx brand HyperFire HC500, PC800 and PC900 cameras (Reconyx, Holman WI) were placed within the AOC and the two reference sites for two month-long pilot deployments in fall/winter 2014/2015 and one eight-month full deployment from summer 2015 to winter 2016. Sampling in multiple seasons should maximize detection probabilities of species more active on land prior to freeze (spring/summer 2015), enabling analysis of best management practices for using these techniques in the future, while also collecting data concurrent with the aerial surveys conducted by the Wisconsin Department of Natural Resources (fall/winter 2015).

The number of cameras allocated to each study area was proportional to the total miles of shoreline in each. Within the larger and more complex Area of Concern, distinct habitat types were stratified and effort was allocated accordingly to ensure complete, unbiased coverage (O'Connell et al. 2011). The habitat strata designated within the AOC were a) the upper river section, b) the central channel and Mud Lake, c) Pokegama bay, d) the suburban eastern shore, e) Allouez bay, and f) the Nemadji river (see appendix A.1 for a detailed map).The more homogenous reference areas were treated as a single habitat type.

To adequately space cameras for independent sampling and to avoid selection bias in the field, target locations were established before placing cameras in the field. First the shoreline of each study area was broken into 1 kilometer long segments, which were then randomly ranked. The first ranked segment was assigned a “target location” in the center of the segment. The process continued down the ranked segments, with either a point being assigned or censured such that no two adjacent segments were sampled simultaneously. During the first iteration for cameras deployed in November 2014, the minimum distance between two target locations was set at 1.5 kilometers. When randomizing for the second deployment of cameras in December 2014, target locations were at minimum 1.5 kilometers from another target site and 1 kilometer of any previously sampled camera site. For the third deployment in summer 2015, additional cameras were available such that I was not required to move them to ensure adequate coverage. I prioritized resampling prior locations, and followed the same process to select target locations for the remaining cameras in segments a minimum of 1 kilometer apart.

Cameras were deployed in the field by navigating to the target location as closely as possible, contingent on access and that the habitat provided at least some adequate structure for securing the camera. The area was then searched for sign of animal passage to the water and specific indications of use by target species, such as scat, tracks, lodges or feeding sign. Data collected at the selected site included GPS coordinates; terrestrial and emergent aquatic vegetation species and percent cover; diameter at breast height of the camera tree; bearing, angle and height above the ground for the camera; distance from the camera to the water; and camera make, model, and battery status. Cameras were revisited every 1-2 months to collect data cards, exchange batteries, and rectify any issues with the camera or site. In the event that site characteristics had changed substantially, either due to vegetation growth or rises in water level, cameras were re-angled or moved entirely (<30m) to compensate, and new covariate data were collected if applicable.

# 5 Statistical Methods

## 5.1 Aerial survey Kruskal Wallis tests

Analyses of aerial data were conducted by compiling all data points collected on each flight into species detections: beaver sign (lodges, food caches, and chewing sign), muskrat sign (push-ups), and otter tracks (tracks and slides in snow). A confidence score was assigned for potentially ambiguous points and two datasets were maintained of only *certain* data points or *combined* with lower confidence points. All counts were then divided by either the total kilometers of flight distance or the minutes of flight time for each study area completed in the survey to account for variable sampling effort. For estimated relative abundance, only a single observer present on all surveys was used for consistency. These data were then assessed on a single species basis with a Kruskall Wallis test against the null that median values would be equal between the AOC study areas and the reference sites. The non-parametric test does not require data to be normally distributed (Zar 2010), and although a Poisson distribution can be used for aerial data (Hodgson et al. 2016) the transformation was not universally helpful for these data.

## 5.2 Occupancy modeling framework

To accurately assess overall status across multiple camera stations, the pattern of detections (positive identified images) for each species can be modeled to inform both detection probabilities and occupancy estimates (MacKenzie et al. 2002, 2006, Royle and Nichols 2003). Each species of interest was recorded as either detected (1) or not detected (0) for every 24-hour period that a camera was functioning, to create a detection history. Even for sites that detected species, there were typically many days of non-detection, resulting in detection histories with many 0 events. To increase the modeling power I collapsed all daily data into one-week observation periods, such that a week in which a species was detected once or more is observed as 1, and never detected is observed as 0. These 0-1 observations result in 11 weekly detection history in each season, for each species and camera site.

Example: Mink detection history at camera site SLE3-1 in Season 1: $h$ = 00001011110

Occupancy modeling leverages several ecological and mathematical features of the detection histories to obtain robust estimates of their underlying biological processes (MacKenzie et al. 2002, 2006, Royle and Nichols 2003). The first of these is that an observation period recorded as 1 must result from two conditions being met: the site is occupied by the species, and the species is detected successfully. The true occupancy state of a site, denoted $z_i$, is either 0 (unoccupied) or 1 (occupied). The probability of a site being occupied is then denoted as $\Psi = P(z_i = 1)$, and this is the value which can be modeled to infer species occupancy patterns. Because successful detection must also occur, the probability of any single observation *j* at site *i* recording a 1 is given as the probability of occupancy (Ψ) multiplied by the probability of detection (*p*):

$$P(obs_{ji} = 1) = \Psi p$$

The second characteristic is that a non-detection, or 0, observation can result from two different scenarios: either the site is truly unoccupied, or the site is occupied but the species was not successfully detected (false-absence). Because both these potential states are dichotomous, the probability of one event is simply one-minus the probability of the alternative (ie non-detection is one-minus-detection, or 1-*p*). Thus the probability of an observation being 0 can be modeled accounting for both scenarios:

$$P(obs_{ji} = 0) = (1-\Psi) + \Psi(1-p)$$

(Royle and Nichols 2003, Royle and Dorazio 2008)

The uncertainty in non-detections can be considered as a nuisance parameter and ignored, but this can result in severely biased conclusions (see White 2005). Instead, explicitly incorporating the underlying source of uncertainty in any wildlife research that depends on imperfect observations will improve the accuracy of the state variable of interest and strengthen inferences of the ecological processes involved (MacKenzie et al. 2002, 2006).An important assumption for these models is that the true occupancy state of a site, $z_i$, does not change over the monitoring period. I subdivided the full 2015/2016 season data into three 11-week seasons to ensure that this assumption was met. Models can be further strengthened by incorporating habitat and observations covariates that can influence both detection and occupancy probabilities. Because detection and occupancy state variables are binomial (one of two options), and informative parameters are typically continuous or categorical, they are linked with a logit transformation. This allows linear combinations of explanatory factors, but means resulting estimates must be back transformed if values in the original units are of interest. Modeling occupancy probably with covariates 1 to *x* gives

$$\text{logit}(\Psi) = \ln\left(\frac{\Psi}{1\text{-}\Psi}\right) = \beta_0 + \beta_1 + \ldots\ \beta_x$$

(MacKenzie et al. 2006)

### 5.3 Detection probability modeling

I used a sequential modeling approach (Burnham and Anderson 2002, MacKenzie et al. 2012, Lesmeister et al. 2015) to first select only the parameters which were informative for the detection process. I speculated *a priori* that the following seven site-level factors might influence detection rates: study area in which the camera was located (AREA), distance from camera to water (DIST.WATER), percentage of canopy closure on land near the camera (TREE.COVER), percentage of emergent aquatic vegetation cover (MARSH.COVER), slope of the camera set (SLOPE), height of the camera from the ground (CAM.HEIGHT) and diameter at breast height of the focal tree (CAM.DBH). For modeling the effect of AREA, I differentiated between the St Louis river estuary (SLE) and the Nemadji river and Allouez bay (ALZNMJ), and between the two reference sites, Boulder Lake reservoir (BLR) and the St Croix river (SCR). The St Louis river estuary was modeled as the intercept (baseline for comparison), and the other three areas were incorporated as alternative states. This increases the number of variables (k) from only one (for the other site-level covariates) to three, which will play a role in the eventual AIC model ranking process.

In additional to site-level features, observation-level covariates could affect the probability of detecting species at cameras during each week-long period. These included temperature data which were collected twice each day at 2:00AM and 2:00PM when timelapse images were automatically recorded. Temperature measurements were then either averaged over each observation period (AM.Ave, PM.Ave), or sorted for weekly maximum and minimum temperatures (AM.Max, AM.Min, PM.Max and PM.Min). Lastly, the total number of days within the observation period for which each camera was operational was calculated (ACTIVE). Differences in the active period for cameras are the result of logistic realities deploying and checking scores of stations over three distinct areas, and of camera failures caused by filled SD cards, depleted batteries, water and ice cover, or other circumstances.

Table 1. Detection Model Suite

| Site Covariates | | Observation Covariates | |
|---|---|---|---|
| NULL | TREE.COVER | ACTIVE | AM.Min |
| AREA | MARSH.COVER | AM.Ave | PM.Min |
| DIST.WATER | SLOPE | PM.Ave | AM.Max |
| CAM.HEIGHT | | | PM.Max |

These site-level and observation-level covariates were each run independently as the only explanatory factor for detection, with occupancy held null, in a suite of models for each species within each season. I used Program R (R Core Team 2014), the package unmarked (Fiske and Chandler 2011, Fiske et al 2016), and the function *occu* to fit the single season occupancy model described by MacKenzie et al. 2002. By using Aikaike's Information Criterion model ranking (Burnham and Anderson 2002) I determined which covariates may have meaningful influence on detection probabilities by sequentially including only the highest ranked models until I obtained a cumulative model weight ≥ 0.90. I noted the ΔAIC at each step, but did not follow a strict cut off at Δ = 2 (see Arnold 2010), because of both the non-nested nature of these models and the penalty assigned for additional parameters, which could put models including AREA at a disadvantage due to the dummy variables differentiating among study areas. This approach generated a set of detection covariates that were specifically relevant to each species and season, and only these were then included in the suite of models to investigate occupancy itself (MacKenzie et al. 2012).

## 5.4 Occupancy probability modeling and weighted estimates

To assess occupancy differences between camera sites and study areas, I examined the following variables in the second phase of sequential modeling to determine their influence on the presence or absence of a species: AREA, TREE.COVER, and MARSH.COVER. These variables were assembled into a full model suite (eight models), where each was present individually and in additive combination. I did not consider interactions due to the large number of competing models this approach would generate, and for each species and season the detection parameters indicated in the first phase were used.

Table 2. Occupancy Model Suite

| Without AREA | With AREA |
|---|---|
| NULL | AREA |
| TREE.COVER | AREA + TREE.COVER |
| MARSH.COVER | AREA + MARSH.COVER |
| TREE.COVER + MARSH.COVER | AREA + TREE.COVER + MARSH.COVER |

Top models were selected following similar rules for inclusion as for top detection models: keeping top ranked models up to a cumulative weight > 0.90, but not discarding suites with a highest model weight < .50. Those ranked as top models were then assessed with parametric bootstrapping to avoid incorrectly assuming that inclusion in top models indicated a good model (MacKenzie and Bailey 2004). I used *parboot* to refit 1000 simulated data sets back to the model and calculating the chi-square test statistic for goodness of fit and the $\hat{c}$ overdispersal factor (MacKenzie and Bailey 2004, Fiske et al. 2016).

Each of the occupancy models, by incorporating information from all of the included covariates, calculates a fitted occupancy estimate ($\hat{\Psi}$) for each observation period of each camera site. Using a model averaging approach (Burnham and Anderson 2002) I took the occupancy probability generated by each of the top models and averaged across observation periods within a season to obtain the predicted overall occupancy for each site for each model, $\hat{\Psi}_{MODEL}$. Finally, I multiplied that average for each site by the proportion of the cumulative AIC weight which that model accounted for, and then summed all top models together, giving

$$\hat{\Psi}_{WEIGHTED} = \text{Sum}\left(\frac{\hat{\Psi}_{MODEL} \times \text{ AIC weight}_{MODEL}}{\text{AIC weight}_{CUMULATIVE}}\right)$$

In this way the variation of the results from different covariates being present in different models is preserved but the final outcome is weighted towards those with the greatest explanatory power. This generated a data set of weighted occupancy estimates for each site, allowing comparison between the AOC and the reference areas (appendix).

## 5.5 Equivalency tests

Equivalency tests operate under the null assumption that two populations being compared will have different distributions or mean values (Wellek 2010). This is a departure from traditional hypothesis testing in which the null assumption is that there is no difference, and this emphasis makes it a potentially preferable method for assessing situations such as environmental remediation (Manly 2001). Placing the burden of proof on demonstrating the equality of a degraded system and a control system, rather than failing to find evidence against equality, is recommended by the US Environmental Protection Agency (EPA 1994) and there are examples of statistical exploration in the natural sciences (Robinson and Froese 2004, Robinson et al. 2005).

I used the R package *equivalence* version 0.7.2 (Robinson 2016) to test for differences in weighted occupancy estimates for each species and season between AOC and reference areas. The test *Rtost* is a robust two one-sided test that is appropriate even if assumptions of normally distributed data cannot be met.

# 6 Field Results

## 6.1 Aerial survey results

A total of six fixed wing flights and one helicopter flight were successfully conducted within the St Louis AOC and reference sites (table 3). Of the six fixed wing flights one was a training flight, and another was forced to terminate before completion due to mechanical problems with the aircraft. Numerous additional flights were attempted but either canceled or terminated prior to completion due to inclement weather or scheduling conflicts. Of the four complete fixed wing surveys three were conducted with the pilot and a single observer (BE, similar protocol to Johnston and Windels 2015). One fixed wing and the helicopter survey were conducted with multiple observers to help assess detection rates (Walsh et al. 2010).

One test flight was conducted on September 25 2014 to assess visibility of sign at typical fixed wing speed and height, and flight protocol was finalized based on those observations. In the 2014 season, one aerial survey was completed on November 20, and a second survey was partially performed on December 18. However, the second survey was terminated prior to completion due to safety concerns over a malfunction of aircraft altimeter equipment, and additional attempts were canceled due to weather. During the 2015 season similar conflicts with weather and prior commitments for flight time restricted the number of flights performed, with one conducted on November 13 and another on December 4. No additional flights in December were possible despite several attempts, and due to the early timing the snow cover was not consistent between all three study areas. To obtain more complete coverage of the study areas a later winter flight was completed on February 17 2016.

The timeline for contracting with a new helicopter service provider delayed the comparison helicopter survey until April, 2016. Once a contract was in place with Brainerd Helicopters Services Inc, a complete survey was conducted with three observers in a Bell 206B3 Jet Ranger helicopter. Total flight time was six hours and survey conditions were excellent for detecting beaver sign, however this survey occurred outside of the typical sampling season.

Table 3: Aerial surveys completed from fall 2014 to spring 2016 recorded the greatest number of observations occurred in early winter, with a thin layer snow on the ice to facilitate identification of otter tracks without obscuring beaver lodges. Fall fixed wing flights and the spring helicopter survey conditions also allowed for recording of beaver food caching and chewing activity.

| Date Observers | Flight time (h:mm) | Distance (km) | Data Points | Conditions |
|---|---|---|---|---|
| 09/24/2014 BE, NR | N/A | N/A | N/A | Beaver sign was abundant, but no snow for otter tracks. BE trained with NR. Data not included into analyses. |
| 11/20/2014 BE | 2:03 | 341 | 174 | Light snow present for otter sign, larger beaver sign detectable but decreased visibility for muskrat sign. |
| 12/18/2014 BE | (1:44) | (202) | (88) | PARTIAL FLIGHT: Allouez Bay, Nemadji River and St. Louis study areas flown, then error with the flight altimeter required landing early. |
| 11/13/2015 BE | 2:18 | 340 | 254 | Excellent conditions for beaver and muskrat sign. |
| 12/4/2015 BE | 2:33 | 393 | 323 | Partial snow conditions allowed for otter sign detection at Boulder Lake Reservoir consistently, in some portions of the St. Louis study area, and only in patches along the St. Croix River. |
| 02/17/2016 BE, NF | 2:40 | 365 | 198 | Consistent snow cover for otter sign in all study areas. NF experienced observer. |
| | | Helicopter Survey | | |
| 04/26/2016 BE, NF, MW | 2:55 | 254 | 174 | All ice cover melted, signs remaining of winter beaver food caches and fresh spring activity detectable as well as muskrat sign. MW trained. |

## 6.2 Camera survey results

Motion triggered trail camera surveys were conducted from fall 2014 to early winter 2016 in three distinct deployments. Deployments 1 and 2 each spanned approximately one month from November to December 2014, and December 2014 to January 2015, and consisted of 28 and 29 cameras. In deployments 1 and 2 a total of 8,244 and 27,594 images were recorded, of which 369 and 9,706 were classified as false triggers, leaving 7,875 and 17,888 images triggered by people or animals. These deployments together are considered the *pilot season*, and although detections of target species were sparse they were valuable to assess proof of concept and establish successful research protocol. While all of the four target species were detected, only otter were detected at enough sites for preliminary analyses, and due to the short time span these data are not considered in further detail for the purposes of this report.

Deployment 3 spanned a total of eight months from June 2015 to February 2016, with up to 65 cameras, and these cameras were able to remain active at a site continuously unless changes in habitat characteristics (especially fluctuations in water level) resulted in the camera being moved to a new “site” within 30m of the original location. Some sites were pulled early if access would be unsafe following ice formation, but the majority remained deployed and usable data were collected through January 2016 (the cameras left in the field later into 2016 were not representative of all study areas and although some target species were detected, data analysis only include timeframes for which all areas can be compared). 585,106 total images were recorded, of which 388,164 were false triggers, leaving 196,942 images triggered by people or animals. These data comprise the *full season* of camera research, and to enable analyses over the extended time period deployment 3 was further divided into three roughly equal 11-week long ‘seasons’ (table 4).

During the pilot season, cold weather and precipitation caused some cameras to fail in the field, although the majority remained active and unobstructed. Over deployment 3, cameras were checked on a monthly basis whenever possible to reduce data loses from camera failures. Despite frequent revisits, SD cards were in some cases filled by false triggers (either moving vegetation or waves) which created gaps in the opportunities for a camera to detect target species. This potential difference in trap nights of effort was addressed in the occupancy modeling step. Despite camera failures, numerous species were detected throughout the three deployments with a high diversity of mammalian species (see appendix A.5).

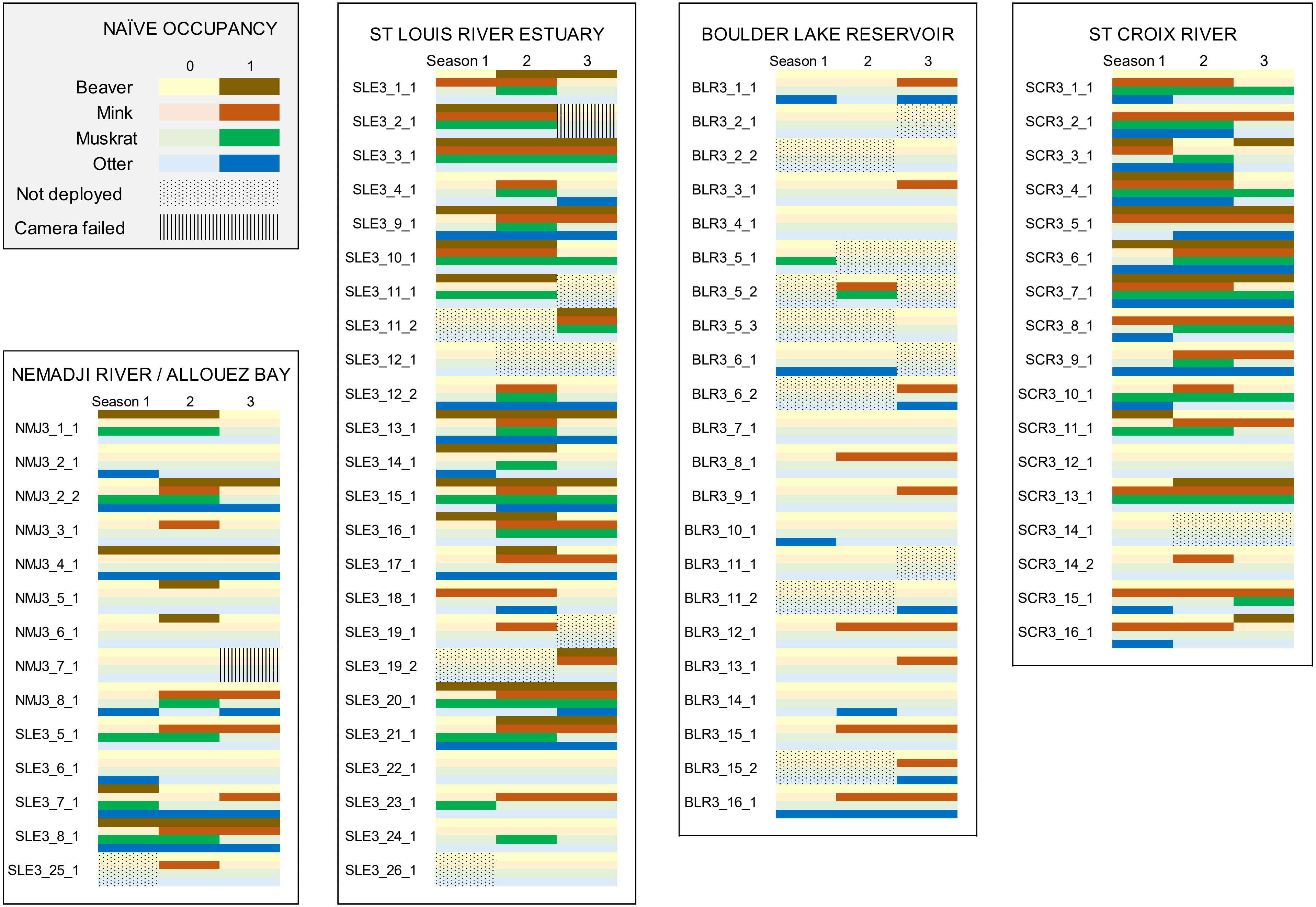
Table 4: Camera deployments from fall 2014 to winter 2016

| Deployment | Start | End | Cameras |
|---|---|---|---|
| 1 | 11/4-14/2014 | 12/2-12/8/2014 | 28: two malfunctioned; one did not record images. |
| 2 | 12/2-7/2014 | 1/10-1/13/2015 | 29: two failed due to battery/ weather problems; two did not record images. |
| 3.1 | 5/29/2015 | 8/14/2015 | 66: 4 moved to accommodate spring vegetation growth. |
| 3.2 | 8/15/2015 | 10/30/2015 | 65: one was stolen by a beaver and no data was retrievable. |
| 3.3 | 10/31/2015 | 1/15/2016** | 57 total due to winter access concerns. |

* *Setting, checking and adjusting cameras as-needed basis*
**Numerous cameras were not retrieved until early February 2016, and one camera remained deployed until 4/27/2016 due to unsafe ice conditions. Although these time frames cannot be included in analyses without other cameras to reference against (not all study areas are represented), target species were detected during the mid-winter and even early spring months, and the data are maintained for future reference.

Camera sites varied in the detection of target species between seasons, and the variation in weekly detections is further modeled in the occupancy analyses. Figure 2 shows naïve occupancy, simply the successful detection of each species at a camera site, over the course of the study.

Figure 2. Detections / non-detection history (naïve occupancy) of each target species at each camera site over all three seasons.



# 7 Statistical Results

## 7.1 Aerial survey results

Kruskal Wallis non-parametric tests found that there was no combination of species, area (reference or AOC), and method (detections per kilometer or per minute) which provides evidence against the null that aerial survey results are equal between the AOC study areas and the reference sites. Both beaver sign and muskrat sign were recorded more frequently in the AOC versus reference sites (359 versus 142 and 311 versus 108 detections) while otter tracks were less frequently recorded in the AOC (69 versus 134), likely a result of excellent visibility and large areas of open ice on Boulder Lake Reservoir.

Due to the somewhat sparse data, tests were pooled across the two AOC areas and the two reference areas. I found no evidence of significant ($p < 0.05$) differences in the relative abundance of beaver or otter sign between the AOC and the reference sites. Flight sample sizes may be insufficient to further model any heterogeneity in detection linked to season, weather or observers, but the data available do not indicate any reduced abundance in the Area of Concern.

Table 5: Results of Kruskal Wallis non-parametric tests for aerial survey data.

| | Detections per kilometer | | | Detections per minute | | |
|---|---|---|---|---|---|---|
| | AOC Median | Reference Median | Kruskall Wallis p-value | AOC Median | Reference Median | Kruskall Wallis p-value |
| Beaver sign | 0.563 | 0.404 | 0.1984 | 0.463 | 0.232 | 0.4097 |
| Muskrat sign | 0.387 | 0.121 | 0.3342 | 0.425 | 0.126 | 0.4665 |
| Otter tracks | 0.281 | 0.631 | 0.2996 | 0.357 | 0.407 | 0.3640 |

## 7.2 Detection probability modeling results

The AIC top model ranking results for testing all parameters which could impact detection by each species and season are shown in the appendix (A.6). With the exception of otter in season 2, the AIC ranking process selected one or more top models, and the variables present in them were concluded to be informative parameters for the underlying biological process of species detection (table 6). The sign of the β slope value for temperature was typically positive, indicating that warmer measurements increased detection probability, while distance to water was negative, suggesting that for species which included that parameter cameras closer to water would have greater detection success. The slope for the camera height variable was overall negative, which may reflect the quality of camera sites in which a lower attachment location is possible. However, speculation on detection processes is not the focus of this research. Inclusion of only relevant parameters strengthens the models used to assess occupancy, but are not otherwise examined further.

Table 6. Selected detection parameters by species and season

| | Beaver | | Muskrat |
|---|---|---|---|
| 1 | AMave + PMmin | 1 | DIST.WATER |
| 2 | ALZNMJ + BLR + SCR | 2 | DIST.WATER |
| 3 | AMave + AMmin | 3 | AMmax + PMave + AMave |
| | **Mink** | | **Otter** |
| 1 | ALZNMJ + BLR + SCR + DIST.WATER | 1 | CAM.HEIGHT |
| 2 | DIST.WATER + AMmin | 2 | *Null* |
| 3 | AMmax + AMave | 3 | PMmax + PMave |

## 7.3 Occupancy probability modeling results and weighted averages

Results of AIC model ranking for assessing occupancy processes are included in the appendix (A.7). Top model weights were more evenly distributed than for detection model ranking, so more models were included to obtain the cumulative weight cut off of 0.9. I conducted parametric bootstrapping for all top models to assess model fit, because ranking among top models cannot be assumed to mean that good models are present (MacKenzie and Bailey 2004). Overall I found excellent fit in terms of chi-square tests, with the mean probability of $p(\chi^2) = 0.635 \pm 0.272$, which does not raise concerns that observed values are abnormally far from expected values. The overall overdispersion factors $\widehat{C} = 1.101 \pm 0.163$ are also very close to 1, indicating the variance within the data are neither much higher nor lower than expected. Only the top two models for otter in season 2 showed an indication of fitting poorly, the chi-square tests bordered on an extreme value at the alpha = 0.05 level (highest model $p(\chi^2) = 0.065$ and second highest model $p(\chi^2) = 0.045$), and there was some evidence that the observed variation was lower than that expected by the model ($\widehat{C} = 0.877$ and $\widehat{C} = 0.834$) although these are not extreme departures from 1 and clear cutoffs are not defined (Zar 2010). The next three models included in the suite demonstrated slightly better fit, and by using a weighted average approach conclusions should be reasonably good. The data for otters in season 2 were the only case were the detection model ranking rules I followed produced a null result, suggesting unexplained variation in the detection process may contribute to the poorer fitting occupancy models.

In regards to the ranking of the different study area parameters among top models for occupancy, the trend is that the AREA variables are rarely included if they are already present in the detection formula. Looking at the structure of the models when the study area is explicitly modeled for occupancy, in beaver season 2 and mink season 1, the β coefficients are slightly negative for the Allouez bay/Nemadji river factor, very negative for Boulder Lake, and positive for St Croix River. This indicates that the occupancy values St Louis estuary, modeled as the intercept, fall between either extreme. For more rigorous assessment of the differences between study areas, I used the weighted site occupancy values to statistically assess the equivalency between AOC and reference areas for all species and seasons.

While the full weighted occupancy output is extensive, an average value across all top models is presented below (table 7) and indicates the overall high occupancy probabilities within the AOC. The range shown is the minimum to maximum for all models and all sites with a study area, regardless of AIC weighting.

Table 7. The average, weighted occupancy value for each species at all sites in a study area (by AIC model weight), and the range of values at sites prior to averaging (all models).

BEAVER

| | Season 1 | | Season 2 | | Season 3 | |
|---|---|---|---|---|---|---|
| | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range |
| BLR | $1.38 \times 10^{05}$ | ($9.65 \times 10^{7}$, .147) | $6.97 \times 10^{6}$ | ($5.03 \times 10^{7}$,$1.98 \times 10^{5}$) | $9.16 \times 10^{6}$ | ($1.21 \times 10^{6}$, .217) |
| SCR | 0.103 | (.019, .160) | 0.126 | (.035, .175) | 0.142 | (.047, .258) |
| SLE | 0.159 | (.019, .252) | 0.219 | (.094, .364) | 0.124 | (.041, .227) |
| ALZNMJ | 0.098 | (.042, .164) | 0.156 | (.073, .218) | 0.105 | (.055, .227) |

MINK

| | Season 1 | | Season 2 | | Season 3 | |
|---|---|---|---|---|---|---|
| | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range |
| BLR | $3.67 \times 10^{6}$ | ($4.59 \times 10^{12}$, $2.27 \times 10^{4}$) | 0.072 | (.002, .283) | 0.107 | (.057, .302) |
| SCR | 0.140 | (.008, .293) | 0.278 | (.103, .416) | 0.138 | (.088, .276) |
| SLE | 0.056 | (.004, .190) | 0.216 | (.099, .306) | 0.108 | (.043, .199) |
| ALZNMJ | $2.10 \times 10^{6}$ | ($1.10 \times 10^{11}$, $1.31 \times 10^{5}$) | 0.170 | (.058, .322) | 0.135 | (.056, .265) |

MUSKRAT

| | Season 1 | | Season 2 | | Season 3 | |
|---|---|---|---|---|---|---|
| | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range |
| BLR | 0.026 | ($8.77 \times 10^{8}$, .252) | 0.034 | ($9.46 \times 10^{7}$, .458) | $2.16 \times 10^{06}$ | ($1.33 \times 10^{8}$, .189) |
| SCR | 0.117 | (.007, .280) | 0.160 | (.023, .480) | 0.164 | (.031, .345) |
| SLE | 0.086 | (.007, .256) | 0.209 | (.023, .469) | 0.065 | (.033, .201) |
| ALZNMJ | 0.095 | (.003, .230) | 0.191 | (.007, .411) | $1.22 \times 10^{05}$ | ($4.45 \times 10^{7}$, .178) |

OTTER

| | Season 1 | | Season 2 | | Season 3 | |
|---|---|---|---|---|---|---|
| | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range | $\bar{\Psi}$ | Range |
| BLR | 0.058 | (.009, .183) | 0.076 | (.013, .135) | 0.099 | (.040, .220) |
| SCR | 0.182 | (.037, .315) | 0.115 | (.032, .166) | 0.150 | (.035, .323) |
| SLE | 0.077 | (.030, .269) | 0.098 | (.031, .147) | 0.123 | (.044, .222) |
| ALZNMJ | 0.169 | (.003, .327) | 0.096 | (0.0468, .135) | 0.158 | (.072, .321) |

## 7.4 Equivalency tests

Weighted estimates for the occupancy probability at each camera site ranged widely from <0.001 for some estimates of beaver at Boulder Lake to 0.416 for muskrat at several sites along the St Croix river. All site estimates for each species and season were grouped by either REF if in a reference area or AOC if in the area of concern, and treated as the sample of populations of interest for assessing any significant difference. The equivalency test calculates the mean difference $\bar{d}$ and then uses the standard error of the reference site to determine the region of equivalence. If the sample distributions fall within that region, than the null assumption that they are different can be rejected (table 8).

Table 8: Results of robust TOST (two one-sided tests) against the null that the means of weighted occupancy at AOC sites and at reference sites are different ($\bar{d} \neq 0$). Alpha = 0.05, region of equivalence = 20% SE of reference data. All tests reject the null.

| | $\bar{d}$ | SE | 90% CI | df | P-value |
|---|---|---|---|---|---|
| Beaver 01 | -0.086 | 0.016954 | (-0.11495, -0.05779) | 38.3 | $4.10\times10^{-16}$ |
| Beaver 02 | -0.136 | 0.019661 | (-0.16921, -0.10224) | 27.0 | $8.86\times10^{-10}$ |
| Beaver 03 | -0.065 | 0.017592 | (-0.09501, -0.03538) | 31.5 | $2.71\times10^{-15}$ |
| Mink 01 | 0.019 | 0.018976 | (-0.01370, 0.051049) | 25.8 | $8.65\times10^{-15}$ |
| Mink 02 | -0.030 | 0.033653 | (-0.08794, 0.027254) | 23.8 | $8.50\times10^{-09}$ |
| Mink 03 | 0.005 | 0.010463 | (-0.01344, 0.022374) | 23.8 | $1.92\times10^{-20}$ |
| Muskrat 01 | -0.027 | 0.019957 | (-0.06115, 0.006271) | 35.3 | $2.00\times10^{-16}$ |
| Muskrat 02 | -0.091 | 0.041396 | (-0.16090, -0.02018) | 28.7 | $5.65\times10^{-06}$ |
| Muskrat 03 | 0.011 | 0.017577 | (-0.01857, 0.041031) | 31.1 | $1.39\times10^{-17}$ |
| Otter 01 | 0.010 | 0.021818 | (-0.02665, 0.047184) | 33.1 | $1.55\times10^{-15}$ |
| Otter 02 | 0.002 | 0.008095 | (-0.01191, 0.015366) | 38.9 | $1.02\times10^{-32}$ |
| Otter 03 | -0.016 | 0.007827 | (-0.02908, -0.00271) | 39.3 | $1.04\times10^{-32}$ |

The successful rejection of the null hypothesis for all species and season tests indicate that no statistically significant differences are detectable between the AOC and the reference sites. Visual representation via boxplots show that although there are outliers the majority of site estimates do overlap (see appendix A.8).

# 8 Discussion

## 8.1 Methods to compare between Area of Concern and reference study areas

The use of two distinct survey methods enabled comparison between the study areas despite challenges presented by the diversity of habitat types, logistic constraints and year to year variation experienced. This is illustrated especially by the low occupancy estimates from camera surveys for Boulder Lake reservoir for the third deployment, which I speculate were ultimately caused by unusually low water levels altering space use by target species. While cameras obtained fewer detections overall, and none of beaver, the aerial surveys were less susceptible to this short-term fluctuation. Because trail camera success is closely tied to the quality of the site selected, during periods of time when animal movement patterns may have shifted (e.g. due to weather anomalies or disturbance) the boarder spatial scale provided by aerial surveys could prevent complete failure to detect animal sign.

However, the effort and cost required to conduct thorough fixed wing or helicopter surveys is extensive, and with the limited number of flights I was able to conduct there are limitations on the rigor of conclusions that can be drawn. Even with dedicated staff to ensure that every good weather day could be taken advantage of, it is still only possible to collect information on beaver, otter and muskrat, and quality observations are restricted to only a specific season each year. Cameras have the advantage for monitoring a broad suite of species, and with the caveat that multiple sites must be maintained and monitored to account for heterogeneous detection, they were able to collect sufficient data for more rigorous analyses.

## 8.2 Toxicant analyses

Legacy and ongoing contaminants can be a cause for concern throughout aquatic ecosystems, and therefore the scope of this project potentially included investigating levels from tissue samples within the AOC and the reference areas should evidence of suppressed populations arise. None of the data available from these surveys indicates that otter, mink, muskrat or beaver are abnormally restricted within the AOC, thus toxicant analyses are not considered to be a high priority at this time.

Over the course of this research myself and collaborators made efforts to collect carcasses of otters and mink from within the study areas, in the event toxicant analyses were deemed valuable and funds were available. While agency staff in both Minnesota and Wisconsin, as well as members of the trapping communities, were obliging, the sample sizes of animals that can be confirmed as harvested within the study areas are very small. Only two otters known to come out of the AOC were successfully collected by WDNR staff, and of those only one liver is available for analyses. Three otters were taken by a tribal trapper near the St Croix reference area, though the exact location is unknown. Additional livers were extracted from otters with the harvest county listed as either Burnett, Douglas or Washburn, but precise locations are unknown. No samples were available from the AOC and reference sites on the Minnesota side, but a trapper volunteered four otter and eight mink livers from Brimson MN. All samples that could be of potential value are currently frozen and stored at UW Madison.

## 8.4 Degraded Fish and Wildlife Populations BUI Status and Recommendations

From the analyses of aerial and camera surveys conducted for this report, I have not found evidence that beaver, mink, muskrat or otter detections within the St Louis River Area of Concern are significantly lower than those in reference sites. It is important to note that wildlife surveys are not the same as a population census, so while these data cannot provide an estimate of the number of individuals with the AOC, the surveys were designed to assess any differences in relative abundances of species. By assessing at the scale of detections per unit of survey effort in the flight data, and by modeling camera site data in an occupancy framework, results illustrate the presence of species and use of habitat resources throughout the areas of interest. A key assumption is that reference sites share many of the ecological features of the AOC, with only the history of degradation and remediation substantially differentiating them. From careful selection of reference sites, qualitative observations, and the quantitative results of the surveys, this appears to be a valid assumption despite weather anomalies.

The removal target for BUI 2 states *In consultation with their federal, tribal, local, and nonprofit partners, state resource management agencies concur that diverse native fish and wildlife populations are not limited by physical habitat, food sources, water quality, or contaminated sediments*. (MPCA 2013). This research assessed the status of one target group under the BUI; four native mammal species that are assumed to have been rare or extirpated during the most extreme degradation of the St Louis River ecosystem. This study has found that the removal objective for small semi-aquatic mammals is being met as no evidence that a current lack of suitable habitat, resources or pollution was impeding their ability to naturally repopulate the area. The data cannot ascertain if aspects of habitat, food availability, or water quality are sub-optimal, but there is support that the ecosystem is healthy to the degree required for these species to meet their life requirements at levels similar to areas without the same history of degradation.

Similarly, BUI 9 addresses the loss of habitat features required for wildlife to exist within the AOC, and while this research was not a direct assessment of habitat quality, the four target species span several trophic levels and have diverse habitat needs. The detection of beaver, muskrat, mink and otter in the St. Louis AOC indicate that sufficient resources are available to them. Although this research is not an exhaustive assessment of all wildlife species that should be present within the fully restored estuary, there is no indication that specific features require additional remediation attention before recovery can continue.

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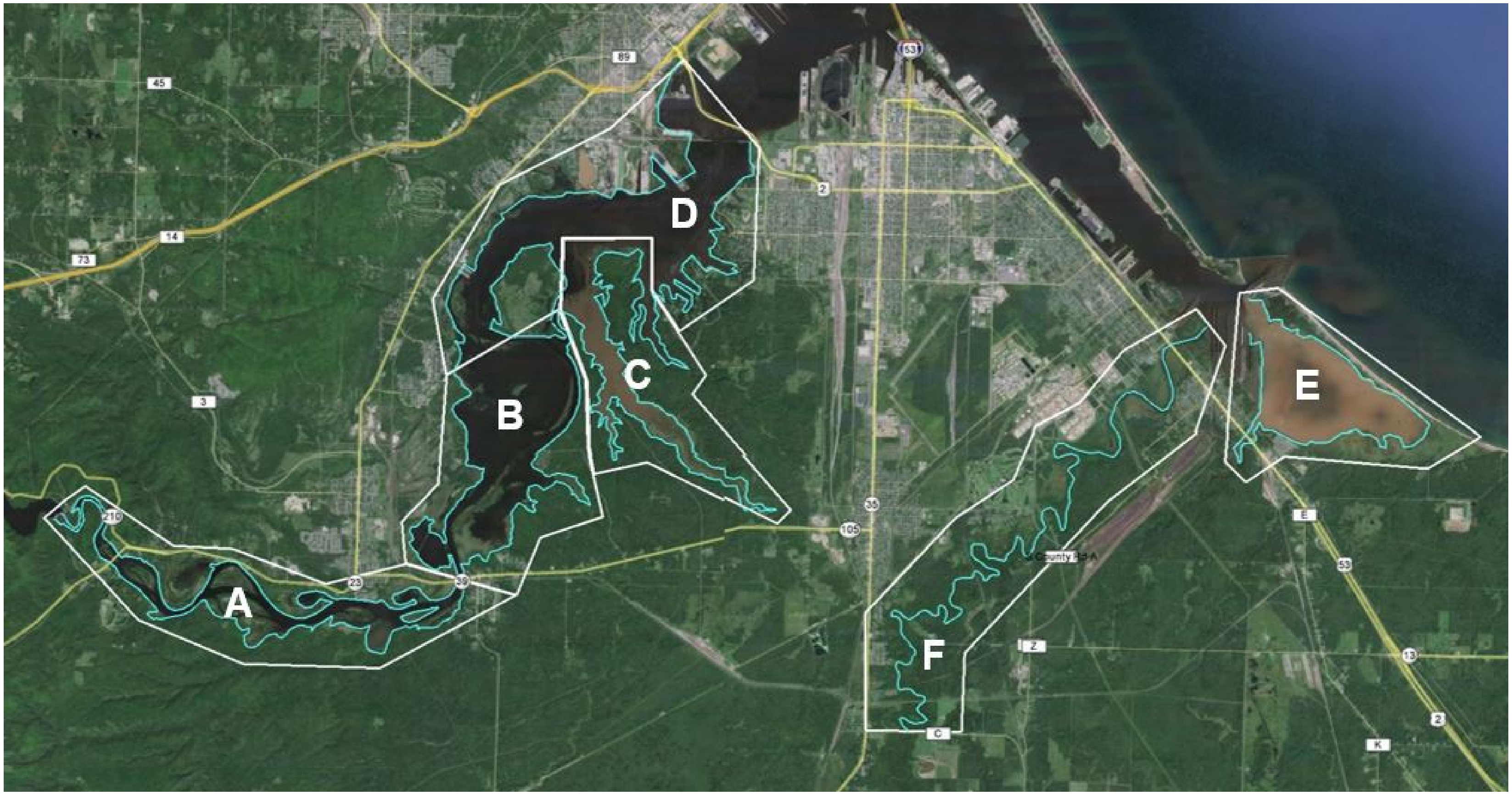
# A Appendix

## A.1 Map of zone differentiation for stratification by habitat type in the St Louis AOC and Nemadji / Allouez Bay study areas




Zone A: Riverine. Faster flowing narrowing channel from Fond du Lac Dam to Oliver Bridge near Mud Lake

Zone B: Undeveloped estuary. From Oliver Bridge to Spirit Lake, area includes remediation sites

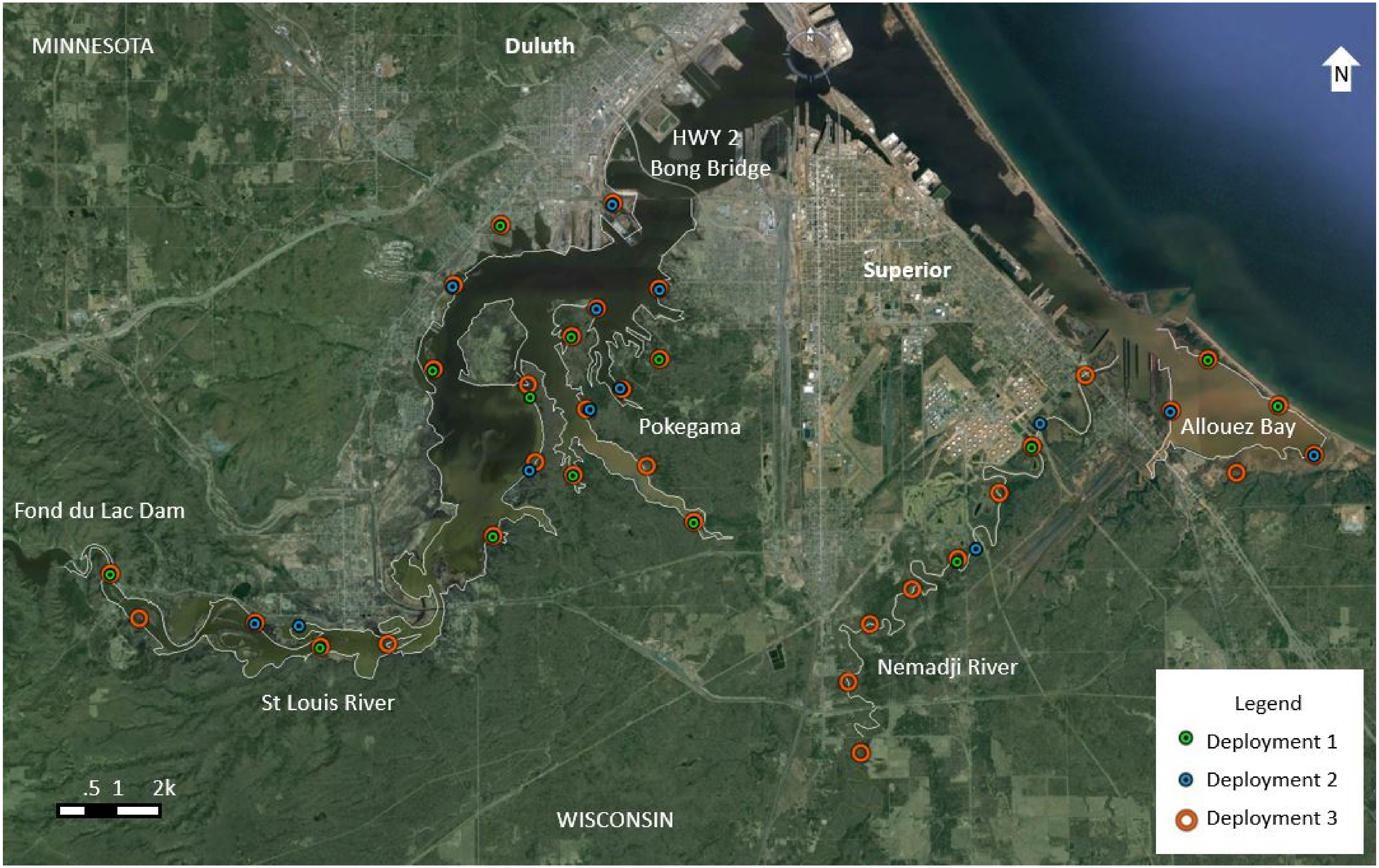
Zone C: Developed estuary. Morgan Park to Bong Bridge (west) and Pokegama estuary (east). Moderate to extensive industrialization

Zone D: Pokegama estuary. Shallow water, dense vegetation

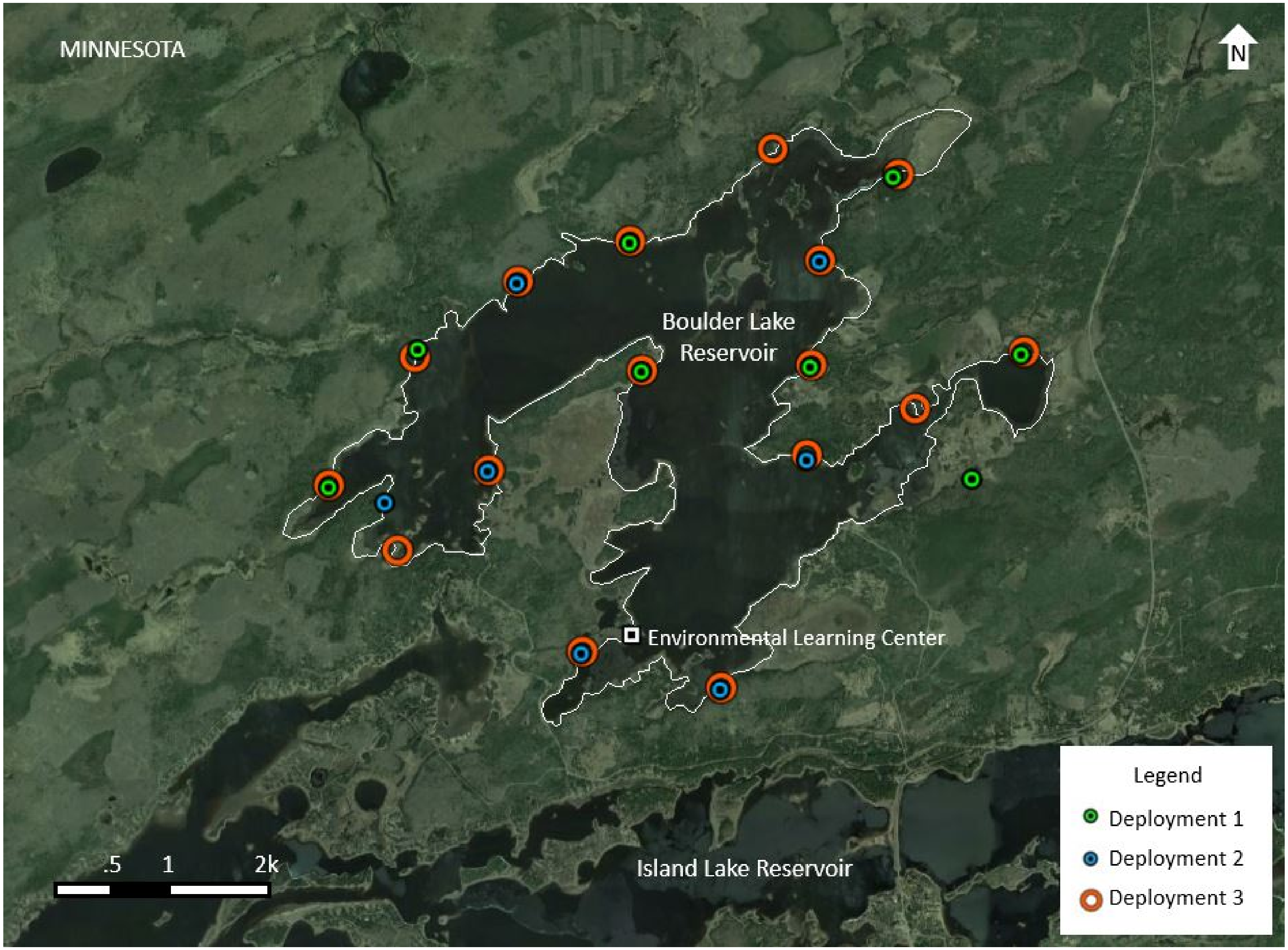
Zone E: Allouez Bay. Inside perimeter of Wisconsin Point, eastern habitat including extensive marsh

Zone F: Nemadji River. From mouth up 12 miles to bridge at E County Road CS

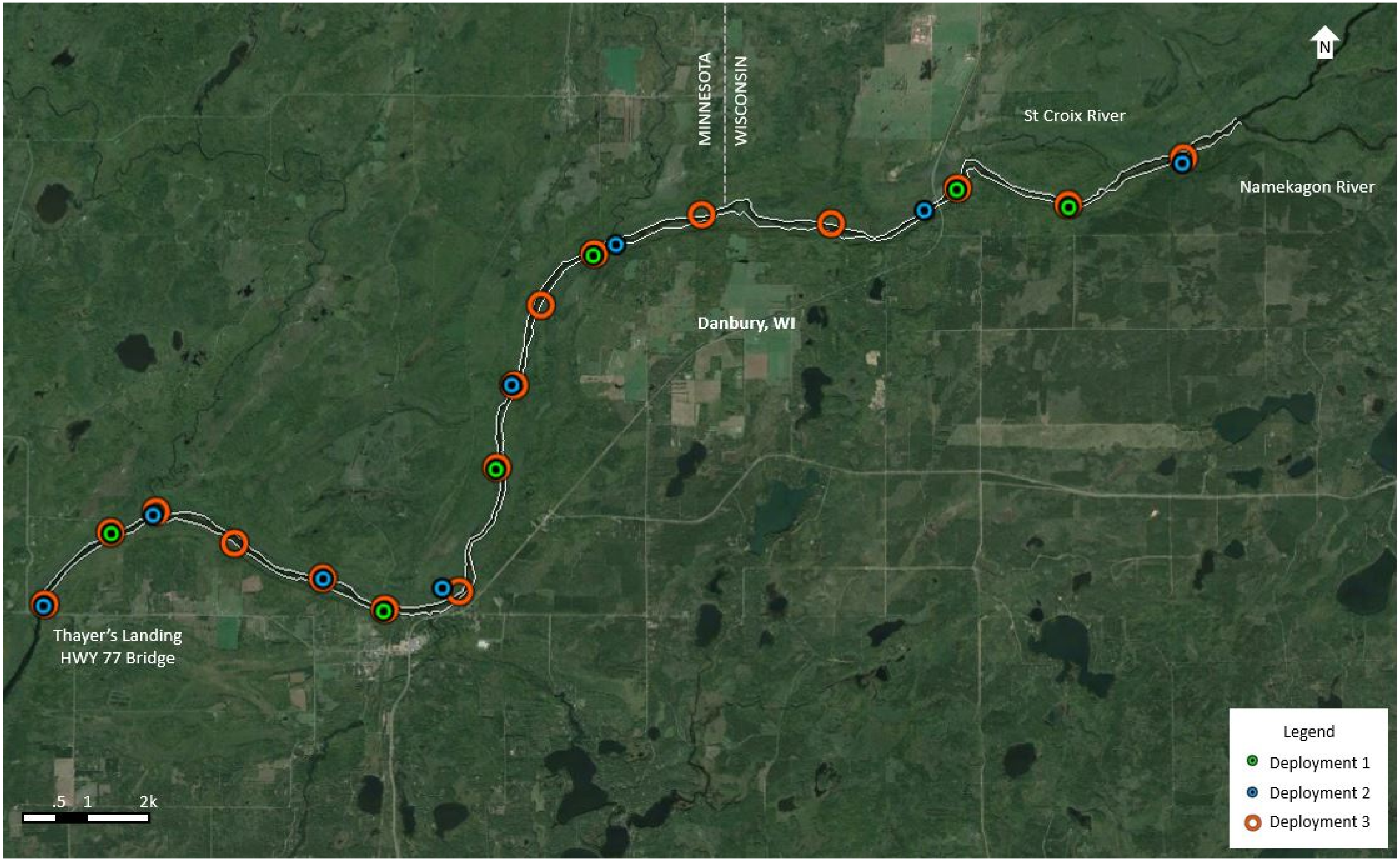
A.2 Camera locations for all deployments in the St Louis River AOC and Nemadji River / Allouez Bay study areas



## A.3 Camera locations for all deployments in the Boulder Lake reference area



## A.4 Camera locations for all deployments in the St. Croix River reference area



A.5 Pilot Season Species Detections: Early results indicated that all four target species could be detected with trail cameras, as well as a diversity of other species. There was a drop in sites with confirmed detections of target species in the second deployment.

Deployment one: 28 Cameras

| Species | # Sites | Species | # Sites |
|---|---|---|---|
| Bird *all species* | 11 | People | 3 |
| **Otter** | **11** | **Beaver** | **3** |
| Squirrel | 10 | Black bear | 2 |
| Mouse | 9 | **Mink** | **3** |
| Coyote | 8 | Weasel | 2 |
| Deer | 8 | Domestic cat | 1 |
| Fox | 8 | Domestic dog | 1 |
| Rabbit | 7 | Fisher | 1 |
| Raccoon | 6 | Marten | 1 |
| Bobcat | 4 | **Muskrat** | **1** |

Deployment two: 29 Cameras

| Species | # Sites | Species | # Sites |
|---|---|---|---|
| Bird *all species* | 6 | People | 7 |
| **Otter** | **1** | **Beaver** | **0** |
| Squirrel | 10 | Black bear | 0 |
| Mouse | 10 | **Mink** | **1** |
| Coyote | 9 | Weasel | 1 |
| Deer | 2 | Domestic cat | 0 |
| Fox | 10 | Domestic dog | 7 |
| Rabbit | 5 | Fisher | 0 |
| Raccoon | 15 | Marten | 0 |
| Bobcat | 8 | **Muskrat** | **0** |

Deployment 3 Detections: More cameras were deployed for an extended period of time, allowing sites to be modified as needed and providing more detections of target species.

| Species | Season 1 *70 sites* | Season 2 *66 sites* | Season 3 *66 sites* |
|---|---|---|---|
| **Beaver** | 20 | 24 | 18 |
| **Mink** | 15 | 41 | 31 |
| **Muskrat** | 21 | 32 | 14 |
| **Otter** | 28 | 21 | 22 |

Deployment 3 non-target species: Those in bold were not detected in the pilot year

| Species detected at one or more camera sites across all three seasons | | | | | | |
|---|---|---|---|---|---|---|
| **Badger** | Coyote | Duck / Geese | Groundhog | Mouse | Rabbit | Striped Skunk |
| Bald Eagle | Crane / Heron | Fisher | Grouse | Owl / Raptors | Raccoon | Turkey |
| Birds | Deer | Fox - Gray | Gull / Pelican | People | **Reptile** | Weasel |
| Black Bear | Dom. Dog / Cat | Fox - Red | Raven /Jay | Porcupine | Squirrel | **Wolf** |

A.6 Ranking top detection models with site- and observation-level covariates: Of the seven site-level variables tested, four (TREE.COVER, MARSH.COVER, SLOPE, and TREE.DBH) were never present in a top model. Of the observation-level variables, ACTIVE was never present in a top model.

**BEAVER**

| Season | Model Equation | AIC | Δ AIC | AIC Weight | Cumulative Weight |
|---|---|---|---|---|---|
| 1 | Ψ(.), p(AM.Ave) | 313.2184 | 0 | 0.577113 | 0.577 |
| | Ψ(.), p(PM.Min) | 314.5877 | 1.369323 | 0.291015 | 0.868 |
| 2 | Ψ(.), p(AREA) | 405.57 | 0 | 0.988655 | 0.989 |
| 3 | Ψ(.), p(AM.Ave) | 234.0961 | 0 | 0.751067 | 0.751 |
| | Ψ(.), p(AM.Min) | 237.0452 | 2.94904 | 0.171911 | 0.923 |

**MINK**

| Season | Model Equation | AIC | Δ AIC | AIC Weight | Cumulative Weight |
|---|---|---|---|---|---|
| 1 | Ψ(.), p(AREA) | 225.5965 | 0 | 0.819405 | 0.819 |
| | Ψ(.), p(DIST.WATER) | 229.0399 | 3.443387 | 0.146479 | 0.966 |
| 2 | Ψ(.), p(DIST.WATER) | 604.6594 | 0 | 0.711929 | 0.712 |
| | Ψ(.), p(AM.Min) | 606.8938 | 2.234401 | 0.232939 | 0.945 |
| 3 | Ψ(.), p(AM.Max) | 336.52 | 0 | 0.578023 | 0.578 |
| | Ψ(.), p(AM.Ave) | 337.3379 | 0.817878 | 0.384012 | 0.962 |

**MUSKRAT**

| Season | Model Equation | AIC | Δ AIC | AIC Weight | Cumulative Weight |
|---|---|---|---|---|---|
| 1 | Ψ(.), p(DIST.WATER) | 280.9117 | 0 | 0.999895 | 0.9999 |
| 2 | Ψ(.), p(DIST.WATER) | 481.4915 | 0 | 1 | 1 |
| 3 | Ψ(.), p(AM.Max) | 141.7617 | 0 | 0.5223 | 0.522 |
| | Ψ(.), p(PM.Ave) | 143.0719 | 1.310182 | 0.27128 | 0.794 |
| | Ψ(.), p(AM.Ave) | 144.0001 | 2.238425 | 0.17055 | 0.964 |

**OTTER**

| Season | Model Equation | AIC | Δ AIC | AIC Weight | Cumulative Weight |
|---|---|---|---|---|---|
| 1 | Ψ(.), p(CAM.HEIGHT) | 373.5132 | 0 | 0.958728 | 0.959 |
| 2* | *Ψ(.), p(CAM.HEIGHT) – NULL* | | *0* | *0.324404* | *0.324* |
| 3 | Ψ(.), p(PM.Min) | 316.7077 | 0 | 0.658745 | 0.659 |
| | Ψ(.), p(PM.Ave) | 318.4363 | 1.728654 | 0.277553 | 0.936 |

* The top model for otter season 2 detection falls below the 0.5 AIC Weight cut off for consideration as a top model. The null detection model is therefore selected.

A.7 Top model AIC ranking for occupancy estimation, using the detection parameters determined to be significant for each species and season. Parametric bootstrapping found excellent model fit with chi-square and overdispersal factors except the otter season 2 suite.

**BEAVER**

| Season | Model # | Model Equation | k | n | AIC | Δ AIC | AIC Weight | $R^2$ | Cumulative Weight | $p(\chi^2)$ | $\hat{c}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 6 | p(AMave+PMmin), Ψ(AREA+MARSH.COVER) | 8 | 70 | 303.2 | 0 | 0.40 | 0.224 | 0.40 | 0.37 | 0.98 |
| | 2 | p(AMave+PMmin), Ψ(AREA) | 7 | 70 | 304.0 | 0.80 | 0.27 | 0.192 | 0.67 | 0.33 | 0.98 |
| | 8 | p(AMave+PMmin), Ψ(AREA+TREE.COVER+MARSH.COVER) | 9 | 70 | 304.7 | 1.45 | 0.20 | 0.230 | 0.87 | 0.28 | 0.96 |
| | 5 | p(AMave+PMmin), Ψ(AREA+TREE.COVER) | 8 | 70 | 305.7 | 2.50 | 0.12 | 0.196 | 0.98 | 0.25 | 0.96 |
| 2 | 1 | p(AREA), Ψ(.) | 5 | 66 | 405.6 | 0 | 0.29 | 0.000 | 0.29 | 0.65 | 1.02 |
| | 4 | p(AREA), Ψ(MARSH.COVER) | 6 | 66 | 406.4 | 0.84 | 0.19 | 0.018 | 0.48 | 0.89 | 1.97 |
| | 3 | p(AREA), Ψ(TREE.COVER) | 6 | 66 | 406.7 | 1.18 | 0.16 | 0.012 | 0.64 | 0.67 | 1.02 |
| | 7 | p(AREA), Ψ(TREE.COVER+MARSH.COVER) | 7 | 66 | 407.2 | 1.65 | 0.13 | 0.035 | 0.76 | 0.87 | 1.08 |
| | 2 | p(AREA), Ψ(AREA) | 8 | 66 | 408.2 | 2.68 | 0.08 | 0.049 | 0.84 | 0.44 | 0.99 |
| | 6 | p(AREA), Ψ(AREA+MARSH.COVER) | 9 | 66 | 408.3 | 2.71 | 0.07 | 0.077 | 0.91 | 0.61 | 1.03 |
| | 8 | p(AREA), Ψ(AREA+TREE.COVER+MARSH.COVER) | 10 | 66 | 409.1 | 3.56 | 0.05 | 0.093 | 0.96 | 0.55 | 1.03 |
| 3 | 2 | p(AMave+AMmin), Ψ(AREA) | 7 | 66 | 228.4 | 0 | 0.46 | 0.194 | 0.46 | 0.89 | 1.24 |
| | 6 | p(AMave+AMmin), Ψ(AREA+MARSH.COVER) | 8 | 66 | 229.5 | 1.13 | 0.26 | 0.205 | 0.71 | 0.89 | 1.26 |
| | 5 | p(AMave+AMmin), Ψ(AREA+TREE.COVER) | 8 | 66 | 230.3 | 1.98 | 0.17 | 0.194 | 0.88 | 0.87 | 1.26 |
| | 8 | p(AMave+AMmin) ~ ALZNMJ + BLR + SCR + ZTREE.COVER + ZMARSH.COVER | 9 | 66 | 231.5 | 3.13 | 0.10 | 0.205 | 0.98 | 0.89 | 1.30 |

**MINK**

| Season | Model # | Model Equation | k | n | AIC | Δ AIC | AIC Weight | $R^2$ | Cumulative Weight | $p(\chi^2)$ | $\hat{c}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 4 | p(AREA+DIST.WATER), Ψ(MARSH.COVER) | 7 | 70 | 217.5 | 0 | 0.48 | 0.056 | 0.48 | 0.83 | 1.21 |
| | 1 | p(AREA+DIST.WATER), Ψ(.) | 6 | 70 | 219.3 | 1.81 | 0.19 | 0.000 | 0.67 | 0.96 | 1.23 |
| | 7 | p(AREA+DIST.WATER), Ψ(TREE.COVER+MARSH.COVER) | 8 | 70 | 219.4 | 1.97 | 0.18 | 0.056 | 0.85 | 0.75 | 1.15 |
| | 3 | p(AREA+DIST.WATER), Ψ(TREE.COVER) | 7 | 70 | 221.2 | 3.71 | 0.07 | 0.002 | 0.92 | 0.91 | 1.20 |
| 2 | 2 | p(DIST.WATER+AMmin), Ψ(AREA) | 7 | 66 | 591.4 | 0 | 0.25 | 0.087 | 0.25 | 0.91 | 1.10 |
| | 1 | p(DIST.WATER+AMmin), Ψ(.) | 4 | 66 | 591.5 | 0.03 | 0.25 | 0.000 | 0.50 | 0.91 | 1.09 |
| | 6 | p(DIST.WATER+AMmin), Ψ(AREA+MARSH.COVER) | 8 | 66 | 593.0 | 1.58 | 0.12 | 0.093 | 0.62 | 0.89 | 1.10 |
| | 4 | p(DIST.WATER+AMmin), Ψ(MARSH.COVER) | 5 | 66 | 593.2 | 1.72 | 0.11 | 0.005 | 0.73 | 0.91 | 1.10 |
| | 5 | p(DIST.WATER+AMmin), Ψ(AREA+TREE.COVER) | 8 | 66 | 593.4 | 1.97 | 0.09 | 0.088 | 0.82 | 0.89 | 1.09 |
| | 3 | p(DIST.WATER+AMmin), Ψ(TREE.COVER) | 5 | 66 | 593.5 | 2.03 | 0.09 | 0.000 | 0.91 | 0.92 | 1.09 |
| | 8 | p(DIST.WATER+AMmin), Ψ(AREA+TREE.COVER+MARSH.COVER) | 9 | 66 | 594.9 | 3.42 | 0.05 | 0.095 | 0.96 | 0.86 | 1.10 |
| 3 | 1 | p(AMmax+AMave), Ψ(.) | 4 | 66 | 338.1 | 0 | 0.38 | 0.000 | 0.38 | 0.96 | 1.12 |
| | 3 | p(AMmax+AMave), Ψ(TREE.COVER) | 5 | 66 | 339.1 | 1.02 | 0.23 | 0.015 | 0.61 | 0.92 | 1.11 |
| | 4 | p(AMmax+AMave), Ψ(MARSH.COVER) | 5 | 66 | 339.9 | 1.80 | 0.16 | 0.003 | 0.77 | 0.92 | 1.11 |
| | 7 | p(AMmax+AMave), Ψ(TREE.COVER+MARSH.COVER) | 6 | 66 | 340.5 | 2.39 | 0.12 | 0.024 | 0.88 | 0.82 | 1.08 |
| | 2 | p(AMmax+AMave), Ψ(AREA) | 7 | 66 | 342.5 | 4.33 | 0.04 | 0.025 | 0.93 | 0.91 | 1.12 |
| | 5 | p(AMmax+AMave), Ψ(AREA+TREE.COVER) | 8 | 66 | 342.8 | 4.66 | 0.04 | 0.050 | 0.96 | 0.77 | 1.09 |

**MUSKRAT**

| Season | Model # | Model Equation | k | n | AIC | Δ AIC | AIC Weight | $R^2$ | Cumulative Weight | $p(\chi^2)$ | $\hat{c}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 1 | p(DIST.WATER), Ψ(.) | 3 | 70 | 280.9 | 0 | 0.41 | 0.000 | 0.41 | 0.63 | 1.15 |
| | 3 | p(DIST.WATER), Ψ(TREE.COVER) | 4 | 70 | 282.3 | 1.39 | 0.20 | 0.009 | 0.61 | 0.60 | 1.12 |
| | 4 | p(DIST.WATER), Ψ(MARSH.COVER) | 4 | 70 | 282.3 | 1.42 | 0.20 | 0.008 | 0.81 | 0.61 | 1.15 |
| | 7 | p(DIST.WATER), Ψ(TREE.COVER+MARSH.COVER) | 5 | 70 | 283.2 | 2.28 | 0.13 | 0.025 | 0.94 | 0.54 | 1.10 |
| 2 | 1 | p(DIST.WATER), Ψ(.) | 3 | 66 | 481.5 | 0 | 0.28 | 0.000 | 0.28 | 0.82 | 1.27 |
| | 3 | p(DIST.WATER), Ψ(TREE.COVER) | 4 | 66 | 482.6 | 1.11 | 0.16 | 0.013 | 0.44 | 0.81 | 1.25 |
| | 4 | p(DIST.WATER), Ψ(MARSH.COVER) | 4 | 66 | 482.6 | 1.12 | 0.16 | 0.013 | 0.60 | 0.90 | 1.27 |
| | 7 | p(DIST.WATER), Ψ(TREE.COVER+MARSH.COVER) | 5 | 66 | 483.2 | 1.72 | 0.12 | 0.034 | 0.71 | 0.81 | 1.25 |
| | 2 | p(DIST.WATER), Ψ(AREA) | 6 | 66 | 483.6 | 2.12 | 0.10 | 0.057 | 0.81 | 0.85 | 1.28 |
| | 5 | p(DIST.WATER), Ψ(AREA+TREE.COVER) | 7 | 66 | 484.1 | 2.56 | 0.08 | 0.079 | 0.89 | 0.85 | 1.29 |
| | 8 | p(DIST.WATER), Ψ(AREA+TREE.COVER+MARSH.COVER) | 8 | 66 | 484.6 | 3.10 | 0.06 | 0.099 | 0.95 | 0.84 | 1.33 |
| 3 | 2 | p(AMmax+AMave+PMave), Ψ(AREA) | 8 | 66 | 131.8 | 0 | 0.48 | 0.278 | 0.48 | 0.45 | 1.10 |
| | 5 | p(AMmax+AMave+PMave), Ψ(AREA+TREE.COVER) | 9 | 66 | 133.1 | 1.29 | 0.25 | 0.288 | 0.73 | 0.34 | 0.96 |
| | 6 | p(AMmax+AMave+PMave), Ψ(AREA+MARSH.COVER) | 9 | 66 | 133.8 | 2.00 | 0.18 | 0.278 | 0.91 | 0.45 | 1.05 |
| | 8 | p(AMmax+AMave+PMave), Ψ(AREA+TREE.COVER+MARSH.COVER) | 10 | 66 | 135.0 | 3.28 | 0.09 | 0.288 | 1.00 | 0.33 | 1.02 |

**OTTER**

| Season | Model # | Model Equation | k | n | AIC | Δ AIC | AIC Weight | $R^2$ | Cumulative Weight | $p(\chi^2)$ | $\hat{c}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 2 | p(CAM.HEIGHT), Ψ(AREA) | 6 | 70 | 368.2 | 0 | 0.41 | 0.151 | 0.41 | 0.47 | 1.00 |
| | 6 | p(CAM.HEIGHT), Ψ(AREA+MARSH.COVER) | 7 | 70 | 369.4 | 1.27 | 0.22 | 0.159 | 0.63 | 0.31 | 0.98 |
| | 5 | p(CAM.HEIGHT), Ψ(AREA+TREE.COVER) | 7 | 70 | 369.7 | 1.57 | 0.19 | 0.156 | 0.82 | 0.44 | 1.01 |
| | 8 | p(CAM.HEIGHT), Ψ(AREA+TREE.COVER+MARSH.COVER) | 8 | 70 | 370.7 | 2.56 | 0.11 | 0.168 | 0.93 | 0.42 | 1.00 |
| 2 | 4 | p(.), Ψ(MARSH.COVER) | 3 | 66 | 355.9 | 0 | 0.43 | 0.059 | 0.43 | 0.07* | 0.88 |
| | 7 | p(.), Ψ(TREE.COVER+MARSH.COVER) | 4 | 66 | 357.5 | 1.58 | 0.19 | 0.065 | 0.62 | 0.05** | 0.83 |
| | 1 | p(.), Ψ(.) | 2 | 66 | 357.9 | 1.98 | 0.16 | 0.000 | 0.78 | 0.18 | 0.97 |
| | 3 | p(.), Ψ(TREE.COVER) | 3 | 66 | 358.7 | 2.83 | 0.10 | 0.017 | 0.89 | 0.20 | 0.97 |
| | 6 | p(.), Ψ(AREA+MARSH.COVER) | 6 | 66 | 360.2 | 4.33 | 0.05 | 0.082 | 0.93 | 0.22 | 0.93 |
| 3 | 4 | p(PMmax+PMave), Ψ(MARSH.COVER) | 5 | 66 | 318.2 | 0 | 0.36 | 0.038 | 0.36 | 0.62 | 0.88 |
| | 1 | p(PMmax+PMave), Ψ(.) | 4 | 66 | 318.7 | 0.54 | 0.27 | 0.000 | 0.63 | 0.18 | 0.97 |
| | 7 | p(PMmax+PMave), Ψ(TREE.COVER+MARSH.COVER) | 6 | 66 | 319.7 | 1.52 | 0.17 | 0.045 | 0.80 | 0.68 | 1.05 |
| | 3 | p(PMmax+PMave), Ψ(TREE.COVER) | 5 | 66 | 320.6 | 2.48 | 0.10 | 0.001 | 0.91 | 0.20 | 0.97 |
| | 6 | p(PMmax+PMave), Ψ(AREA+MARSH.COVER) | 8 | 66 | 322.7 | 4.56 | 0.04 | 0.059 | 0.94 | 0.38 | 0.98 |

* Significant at the alpha = 0.10 level
** Significant at the alpha = 0.05 level

A.8 Boxplots of mean value and outliers for weighted occupancy estimates by AOC and reference areas; species, and season.

# Appendix G

# Piping Plover Habitat Restoration

# Project Summary

**Project Name:**
Wisconsin Point Bird Sanctuary Piping Plover Nesting Habitat Restoration

**SLRAOC Management Action: 2.05**

**History:**

Piping Plovers were added to Wisconsin's endangered species list in 1979 and listed as federally endangered in 1986. In the Great Lakes region, Piping Plovers use sparsely vegetated beaches, cobble pans, and sand spits to breed and raise their young for a period of approximately three to four months, annually. Wintering grounds range from North Carolina to Florida and along the Florida Gulf Coast to Texas, Mexico, and the Caribbean Islands. Threats to Piping Plovers include the following: habitat destruction and degradation, human disturbance, and contaminants. Plovers are also impacted by the genetic and geographic consequences of their small population size (U.S. Fish and Wildlife Service, 2003).

**Plover habitat management efforts at Wisconsin Point Bird Sanctuary and Shafer Beach**

The Wisconsin Point Bird Sanctuary is in an easement for common tern (endangered in Wisconsin) and piping plover (federally endangered) habitat. In 2014, there were just 70 breeding pairs of piping plover in the Great Lakes, with most pairs nesting in the Lake Michigan basin (43 pairs) and only 12 pairs in the Lake Superior basin. The Wisconsin Department of Natural Resources (WDNR) has actively managed for piping plover in the St. Louis River estuary since 1980. In 1980, the Barker's Island Bird Sanctuary was established cooperatively by the WDNR and City of Superior at the east end of Barker's Island. Piping plover nested there from 1957 through 1971. Beginning in 1982, WDNR installed up to 10 plover decoys and a sound system that used adult vocalizations on a continuous loop at the Barker's Island Bird Sanctuary in an unsuccessful attempt to attract breeding adults. In 1989, the City of Superior designated the Bird Sanctuary on the Allouez Bay-side of Wisconsin Point as mitigation for developing the property on Barker's Island and losing that habitat. The Wisconsin Point property was given to WDNR, cleared with a rotovator, and a chain-link fence was installed around the perimeter to deter traffic. The St. Louis River Alliance (SLRA) conducted a habitat restoration project in the 1990's at the Bird Sanctuary with weed fabric and tree removal. The WDNR actively managed this property for common terns and piping plover until 2005, including vegetation control and monitoring. None of the habitat management were successful in attracting either terns or plovers to nest on the property. Interstate Island was identified as a preferred alternative tern nesting area. After 2005, WDNR discontinued management at the Bird Sanctuary, and common tern management efforts were re-directed at Interstate Island.

Management was started again in 2011, when SLRA received a grant from the US. Fish and Wildlife Service (USFWS) to undertake habitat restoration for piping plover on the Bird Sanctuary property (10 acres) and at Shafer Beach (~25 acres). The project managed habitat from 2012 – 2017. Through SLRA's partnership with Douglas County, USFWS, WDNR, and City of Superior, restoration at the Bird Sanctuary and Shafer beach has included eradicating invasive species, excavating and sloping the beaches, and clearing wood and debris. At Shafer Beach, the County cleared shrubs along the bluff adjacent to the beach to increase the distance between the waterline and the treeline. SLRA developed curriculum and outreach materials to educate over 200 children and adults about piping plover and also trained volunteers to monitor the beach for piping plover and educate beach-goers in an effort to minimize human and dog disturbances at the beach. Ten plover decoys were made in 2013 and were used in conjunction with a playback system in an attempt to attract breeding piping plover to the

Wisconsin Point and Shafer Beach sites. Typically, 1 – 3 piping plover are observed each year on Wisconsin Point and MN Point.

Management action 2.05 was determined necessary to restore historically-lost nesting habitat for the endangered Piping Plover as part of the BUI and support the 2003 USFWS Great Lakes Recovery Plan for the Piping Plover (U.S. Fish and Wildlife Service, 2003). The 2003 Recovery Plan's ultimate objective is to remove the Great Lakes population from the list of Threatened and Endangered Species, requiring that specific recovery criteria for population size, reproduction, habitat, and long-term protection are met. Management action 2.05 was officially included in the SLRAOC "Roadmap to Delisting" in 2013 (Minnesota Pollution Control Agency and Wisconsin Department of Natural Resources, 2013).

**Project Goal:**
The project goal was to increase available Piping Plover stopover and nesting habitat as well as to create habitat that is:

- long lasting and requires minimal maintenance
- resilient to changing water levels and storm events, and
- beneficially uses clean sand dredged from Duluth/Superior Harbor

**Project Outcome:**

- Created 14 acres of nesting and foraging habitat with 3 cobble nesting pans
- Utilized 87,485 CY of dredge material (sand) from the Duluth/Superior Harbor

**Project Coordinates (UTM center-point):** 46.700833, -92.011866

**Start and End Dates:** 2017-2019 design and implementation; 2020-2023+ establishment phase

**Lead Agency:** Wisconsin Department of Natural Resources

**Project Manager:** Cherie Hagen, Lake Superior Basin Supervisor, Wisconsin DNR

**Design Metrics:**

- 14 acres of nesting and foraging habitat for Piping Plover including specific metrics in Table 1.
- Utilized 87,485 cubic yards of dredge material from Duluth/Superior Harbor

**Funding:**

| **Funding Source** | **Amount** |
|---|---:|
| EPA Administered Great Lakes Restoration Initiative | $4,000,000 |
| | |
| **TOTAL COST** | **$4,000,000** |

**Project Partners:**

- U.S. Fish and Wildlife Service
- University of Wisconsin Sea Grant
- University of Wisconsin Superior- Lake Superior Research Institute
- WI Department of Natural Resources

**Project Summary:**

WDNR coordinated the project with support from a Restoration Site Team (RST) of local and regional species experts. The RST evaluated SLRE sites for their potential to attract and retain a Piping Plover colony, ultimately choosing the Wisconsin Point Bird Sanctuary (Bird Sanctuary). The Bird Sanctuary site is a fenced-in area owned by the WDNR. WDNR, USWFS, Douglas County, and the St. Louis River Alliance (SLRA) have all supported efforts to actively manage the Bird Sanctuary site since the 1980s for both Piping Plovers and Common Terns. Past management activities included grading, vegetation control, signage, fencing, removal of large woody debris, monitoring, and public outreach. The area is closed seasonally from April 1 through August 1 in order to reduce human impacts during migratory bird season.

The USACE evaluated historic water levels, resulting in a design incorporating target elevations that minimize impacts of water variability and shoreline erosion in the establishment of plover habitat. Beach widths, slopes, and open areas for breeding, nesting, and foraging were designed based on recommendations from RST species experts. The RST worked closely with the USACE to move the project from concept to final design, with the final design completed in late 2018.

The final design created approximately 14 acres of open sand and cobble beach suitable for Piping Plover nesting and foraging habitat. The sand required to construct the beach was obtained through annual Operations and Maintenance dredging of the federal navigational channel by USACE. WDNR developed physical and chemical criteria for these construction materials to ensure their suitability for Piping Plover. The USACE awarded a contract for project construction . In 2019, approximately 87,485 CY of approved dredged materials were placed to extend the existing shoreline and stabilize the slope from erosion, creating 14 acres of habitat. The existing spit feature was widened to encourage long-term connectivity, and a new beach was created. Following sand placement, the habitat was enhanced with cobbles, native dune grass restoration and a fence upgrade to deter predators. The project was completed in late 2019.

**Detailed Project Description:**

Planning and scoping of plover habitat enhancements has been ongoing since 2011 with design for management action 2.05 beginning in 2017. Partners worked closely with USACE engineers, researchers and species experts to design Piping Plover habitat with the following considerations:

- Established minimum habitat design criteria for species
- Using historic high water levels 604’ IGLD85, targeting a minimum of 30 year habitat lifespan
- Installed buoys in Allouez Bay monitoring wave heights in various conditions
- Conducted hydrodynamic and sediment transport modeling
- Analyzed sediment grain size and chemistry of dredge material from potential source locations in the Duluth/Superior Harbor
- Established WDNR criteria for acceptable dredge material for project
- Identified acceptable dredge material in East Gate Basin to construct beach

The design process recognized that restoring plover nesting habitat would provide ancillary benefits to the estuary such as manoomin (wild rice), fish spawning, and dual use habitat for other shorebirds (USACE - Detroit District, 2017).

After several years of planning, construction of the beach habitat was completed in just three months, from August through October 2019. The US Army Corps of Engineers (USACE) contracted with the Roen Salvage Company to lead construction with continued involvement and communication with the partner team. After the existing site was prepped and tern island partially demolished, beach construction began, which involved sand (dredged material) being hydraulically placed both below and above the water to obtain the design slope and elevations. Substantial construction was completed in late 2019.

This is the first Wisconsin project in the estuary that beneficially uses dredge material to restore fish and wildlife habitat. This is an excellent example of how we can collaborate in a working harbor to restore endangered species habitat and ensure navigation channels are open for shipping and commerce. The St. Louis River Area of Concern Remedial Action Plan (SLRAOC RAP) identifies Piping Plover as a target species in the Degraded Fish and Wildlife Populations Beneficial Use Impairment (BUI) with a goal to increase available Piping Plover stopover and nesting habitat. Biologists are documenting increasing numbers of this tiny shorebird, so now is a perfect time to restore additional habitat as they seek out places to rest and nest. Not only does habitat restoration achieve AOC BUI goals, but it also increases critical habitat needed to help meet the USFWS Great Lakes Piping Plover recovery population goal.

Partners will continue teaming up for Piping Plover, focusing efforts from 2020 through 2023+ on habitat establishment and management, outreach and education, and monitoring.

**Post-construction monitoring and establishment:**

Goals and outcomes of the establishment phase- 2020-2023:

- Assess habitat twice yearly to identify actions necessary to maintain suitable habitat while maintaining minimum habitat features (Table 1)
- Conduct actions to maintain suitable habitat (i.e. remove unwanted vegetation/wood, maintain slopes)
- Develop education and outreach materials to protect Piping Plover habitat from human activity at the site
- Monitor for Piping Plovers, document behavior, nesting, fledgling survival and success
- Nest protection and predator control in coordination with USFWS

Outlined below is the proposed work that will be conducted as part of the Establishment Phase. The existing federal and state permits issued for the project construction will apply for 5 to 3 years respectively. Depending on the method selected for woody debris removal (burning) or invasive plant control (herbicide) additional state and local permits may be required. We will be applying standard USFWS and WDNR methods for habitat assessment and management.

1. Native dune grass will be planted near foraging areas (up to 1.5 acres).
2. Habitat assessments will be conducted by WDNR, USFWS and St. Louis River Alliance species experts to verify if the habitat meets the established criteria and to identify any actions needed
3. Habitat management actions may include the following:
    - Woody debris removal
    - Invasive plant control
    - Unwanted vegetation management to maintain open beach in designated nesting area
    - Shoreline grading to ensure desired slope for species is maintained, as needed
    - Cobble supplement for nesting pans, as needed
4. Property management actions below will be conducted by WDNR staff:
    - Access road gate installation
    - Wildlife area closure sign installation and removal before and after the nesting season

**Table 1: Wisconsin Point Bird Sanctuary Minimum Piping Plover Habitat Feature Criteria**

| **Habitat Feature** | **Design Metric** | **Baseline Measurements March, 2020** |
|---|---|---|
| 1. Total Beach Length | ≥ 0.52 km (0.32 mile) | 0.98 km (0.61 mile) |
| 2. Beach Slope | ≤ 10% | <10% |
| 3. Available Nesting Habitat | ≥1 ha (2.5 ac) | 5.68 ac |
| 4. Nesting Habitat Width | ≥50 m (164 ft) | 369ft |
| 5. Tree Line Distance | ≥91 m (300 ft) | N/A |
| 6. Vegetation Cover | ≤5% | <5% |
| 7. Woody Debris Cover | ≤5% | <5% |
| 8. Cobble Pan Dimensions | (1) 50m (164 ft) x 30 m (98 ft) | 15, 432 sq. ft. |
| | (2) 50m (164 ft) x 30 m (98ft) | 20,337 sq. ft. |
| | (3) 50 m (164ft) x30 m (98 ft) | 9,110 sq. ft. |
| Total cobble pan area | 48,216 sq. ft. | 44, 879 sq. ft. |
| 9. Cobble Pan Composition | 60% Cobble, 40% Gravel | 60/40 |
| 10. Cobble Pan % Cover | 30-60% Cobble, 70-40% Sand | 40% Cobble/Gravel 60% Sand |

**Reports, Resources, Documents**: Project documents listed are archived with USACE and WDNR

- Project fact sheet https://widnr.widen.net/view/pdf/y07izhkltm/undefined

- AOC story map: https://mpca.maps.arcgis.com/apps/MapJournal/index.html?appid=d60723ef1a4042d7932bb95208b7a1a6
- AOC video highlighting Plover habitat restoration: https://youtu.be/5XZJPMOLqFk
- 2017 USACE technical memo
- USACE plans and specifications as awarded (June 21, 2019)
- Establishment phase habitat assessment protocol and datasheet
- WDNR site management plan (2022)

**Date Prepared:**
3-7-2022

**Attachment A. Before Photos:**

Pre-project vegetation and debris

Pre-project aerial view

**Attachment B Construction Photos:**

Material placement progress aerial images

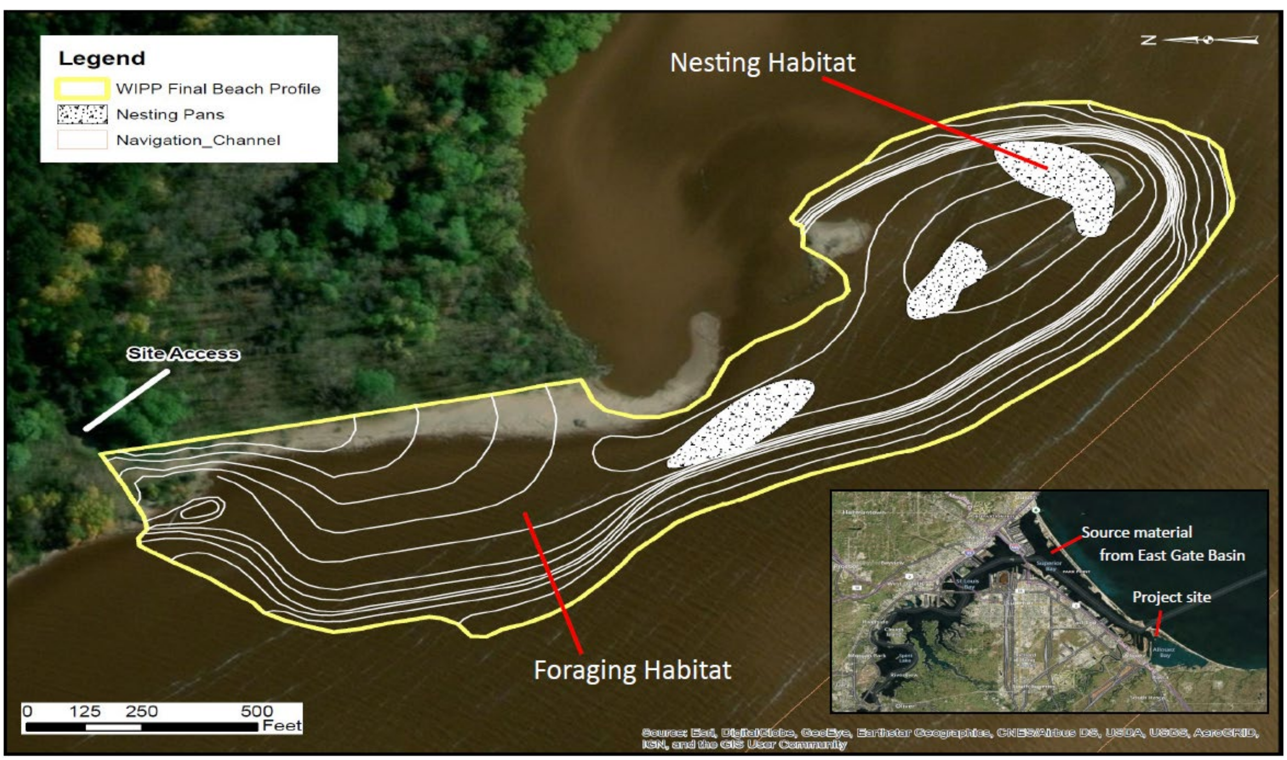
**Attachment C After Photos**:

completed cobble pan

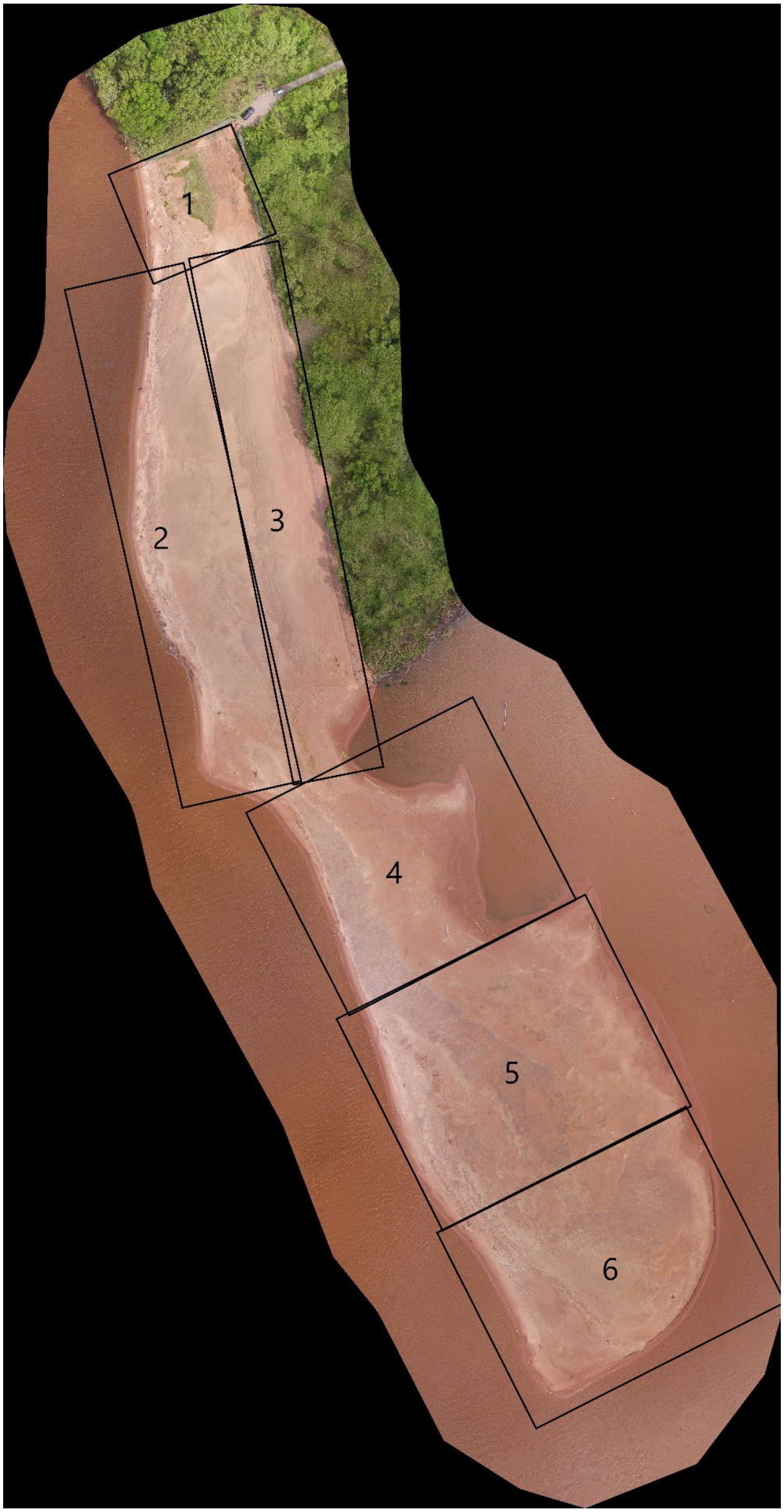

Completed site and fence (2020)

**Attachment D: Project Maps:**


Source material location




Concept design map



Establishment phase management zones

# Appendix H

# Interstate Island Habitat Restoration

# Project Summary

**Project Name:** Interstate Island Avian Habitat Restoration, Minnesota and Wisconsin

**SLRAOC Management Action:** 2.06

**History:** When Interstate Island was created from dredged material in the mid-1930s it was nearly 33 acres in size. In the 1960s, part of the island was mined for sand fill. The entire population of Common Terns in the Estuary, approximately half of the Lake Superior tern population, has nested on the island since 1990. High water levels in Lake Superior have caused erosion, flooding, and scour of the island's environment for the past several years.

**Project Goal:** Restore and protect critical nesting habitat for Common Tern and stopover habitat for Piping Plover.

**Project Outcome:** Colonial waterbird nesting habitat was protected by raising the elevation of Interstate Island, ensuring that at least 5.5 acres of nesting habitat could be sustained at high water levels. In addition, gull exclusion fencing was replaced and 2,934 feet (at OHWL 602.8' IGLD85) of natural shoreline was enhanced or restored, providing habitat for migratory shorebirds. The project resulted in the beneficial reuse of 52,624 cubic yards of dredge materials excavated from the federal navigation channel.

**Project Coordinates (UTM center-point):** 15 T 568033 E 5177682 N

**Start and End Dates:** 2020-2021

**Lead Agencies:** Minnesota Department of Natural Resources (MNDNR) and Minnesota Land Trust (MLT)

**Project Managers:** Melissa Sjolund (MNDNR) and Virginia Breidenbach (MLT)

**Metrics:**

- 6.7 acres of colonial water bird nesting habitat restored or enhanced above high-water design elevation (605.5 feet above sea level - IGLD85)
- 8.4 acres of habitat restored or enhanced above the OHWL (602.8 feet - IGLD85)
- 2,934 feet of natural shoreline restored or enhanced at the OHWL (602.8 feet - IGLD85)
- 52,624 cubic yards or dredged materials beneficially reused

**Federal/Nonfederal Cost-Share (%):** 83:17

**Funding:** [alphabetical listing]

| **Funding Source** | **Amount** |
|---|---:|
| Great Lakes Coastal Program (USFWS) | $ 200,000 |
| Great Lakes Fish and Wildlife Restoration Act (USFWS) | $ 145,000 |
| Great Lakes Restoration Initiative (US EPA) | $ 834,650 |
| Harbor Management Trust Fund (US ACE) | $ 1,149,290 |
| Minnesota's Lake Superior Coastal Program (NOAA) | $ 20,200 |
| Minnesota's Outdoor Heritage Fund | $ 487,000 |
| **TOTAL COST** | **$ 2,836,140** |

**Project Partners:** [alphabetical listing; for design consultants and construction contractors, note their role in parentheses following their firm names; avoid acronyms]

- JF Brennan Company (Construction contractor)
- Minnesota Department of Natural Resources
- Minnesota Land Trust (Project Management, oversight of design and construction)
- Natural Resources Research Institute, University of Minnesota, Duluth
- Roen Salvage (Dredging and construction contractor)
- SEH Engineering (Engineering and construction oversight contractor)
- U.S. Army Corps of Engineers
- U.S. Fish and Wildlife Service
- Wisconsin Department of Natural Resources

**Project Summary:** Common Terns nesting habitat was restored at Interstate Island Wildlife Management Area in the St. Louis River Estuary. Straddling the Minnesota-Wisconsin border, the island is one of only two nesting colonies in the Lake Superior watershed. The $2.8M project used clean sand dredged from the Duluth-Superior Harbor to elevate the island and restore habitat lost to erosion and high water, providing for long-term sustainability of the nesting habitat. Coarse sand and gravel were added to provide high-quality tern nesting substrate. In addition, 2,934 feet of natural shoreline was restored, enhancing migratory habitat for the endangered Piping Plover and other shorebirds.

**Detailed Project Description:** Interstate Island, a Wildlife Management Area jointly administered by the Minnesota and Wisconsin Departments of Natural Resources for nesting colonial waterbirds is an approximately 5.5-acre island within the Duluth-Superior Harbor. Interstate Island is one of the two remaining Common Tern (*Sterna hirundo*) nesting sites in the Lake Superior watershed and is the only federally-listed critical habitat for Piping Plover (*Charadrius melodus*) in Minnesota. In the 1980s, the site became of interest as habitat for Common Terns as human disturbance and site development in other nesting locations in the estuary made those places no longer viable for the species. A restoration project was conducted on Interstate Island in 1989 to clear all vegetation completely to expose sand substrate in an effort to attract Common Terns. The entire breeding population of the SLRE was subsequently attracted to the site in 1989 and 1990.

Emergency habitat protection was completed in 2015 to protect from high water levels by adding sand and gravel to elevate the core nesting area and by constructing a riprap berm. To provide long-term sustainability of the nesting habitat approximately 14,800 $yd^3$ of additional sand was placed on the island in the spring of 2020. This stabilized the island and raised the elevation further. In the fall of 2020, approximately 46,000 $yd^3$ of sand from regular navigation channel maintenance dredging and 7,000 $yd^3$ of sand from strategic navigation dredging was used to reach final design elevations and expand the size of the island to ensure that at least 5.5 acres of area was available for colonial waterbird nesting habitat and that the habitat would be resilient to fluctuating water levels. The project also enhanced the riprap berm around the nesting area, replaced fencing designed to exclude gulls from the 30,000 $ft^2$ core tern nesting area, placed 2 feet of coarse sand with scattered gravel, cobble, and driftwood in the core tern area, and created an additional 1,320 feet of natural shoreline suitable for use by migratory shorebirds, including Piping Plover.

The design incorporates a conservative reference elevation of 605.5’ IGLD85. This is equal to the historic Lake Superior high elevation of 604.5’ IGLD85, with an additional foot incorporated to ensure island protection and resiliency to increasingly unpredictable Lake Superior water levels and storm surges. For the purpose of this project, the 605.5’ IGLD85 elevation identifies the area of Interstate Island that will be considered upland in the long-term. Shoreline slopes were restored to a maximum of 10%to provide for use by both nesting and migrating shorebirds.

Annual monitoring following spring 2020 construction is being conducted to determine the restoration’s effects on tern nesting productivity and migratory bird use. MNDNR will evaluate restoration effectiveness by examining pre- and post-restoration breeding success, juvenile survival, and habitat use. University of Minnesota’s Natural Resources Research Institute (NRRI) maintains a long-term dataset on the island’s tern colony; NRRI researchers will contribute to the analysis of restoration effectiveness and sustainability of the colony.

**Post-Construction Sampling:** MNDNR and partners developed a comprehensive Long-term Monitoring and Maintenance Plan (Plan) to assess and address ongoing and cumulative effects of wind scour, ice, storm surge, wave wash, woody vegetation, and invasive species that may negatively affect habitat quality. MNDNR and WDNR are responsible for jointly implementing the Plan, which focuses on maintaining suitable vegetation, infrastructure, and substrate required to provide high-quality habitat. The Plan addresses habitat maintenance via periodic substrate replacement within the footprint and elevations established by the restoration project. Future maintenance activities are anticipated to occur outside the Common Tern nesting season (March 1 to August 30).

**Date Prepared:** 06/15/2021

**Reports:**


MNDNR. 2020. Phase III: Long-Term Monitoring and Maintenance Plan. Interstate Island Habitat Restoration. Prepared by Short Elliott Hendrickson Inc., Duluth, MN, under contract with Minnesota Land Trust.

MNDNR. 2020. Quality Assurance Project Plan, St Louis River Area of Concern, Interstate Island WMA Avian Habitat Restoration Project. EPA Grant Number: GL00E02466. Submitted to Environmental Protection Agency, Great Lakes National Program Office, Jackson Blvd., Chicago, Illinois.

Matteson, S. W. 1988. Wisconsin Common Tern Recovery Plan. Wisconsin Endangered Resources Report 41. Bureau of Endangered Resources. Wisconsin Department of Natural Resources. Madison WI.

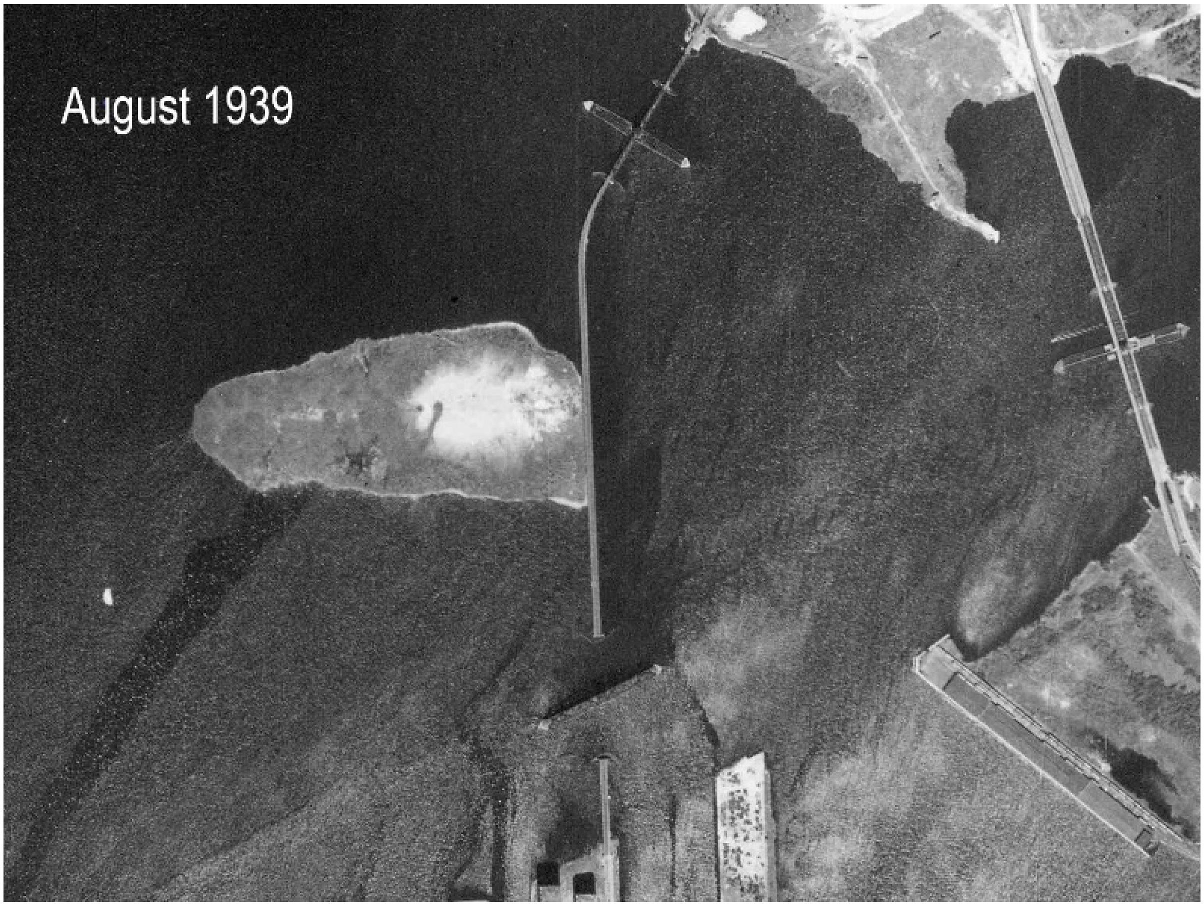
SEH. 2019. Construction Plans for Spring Interstate Island Improvements. 100% Design Plan Set.

SEH. 2019a. Interstate Island Design Memo. SEH No. 150297 14.00.

SEH. 2020. Construction Plans for Fall 2020 Improvements, Duluth/Superior Harbor. 100% Design Plan Set.

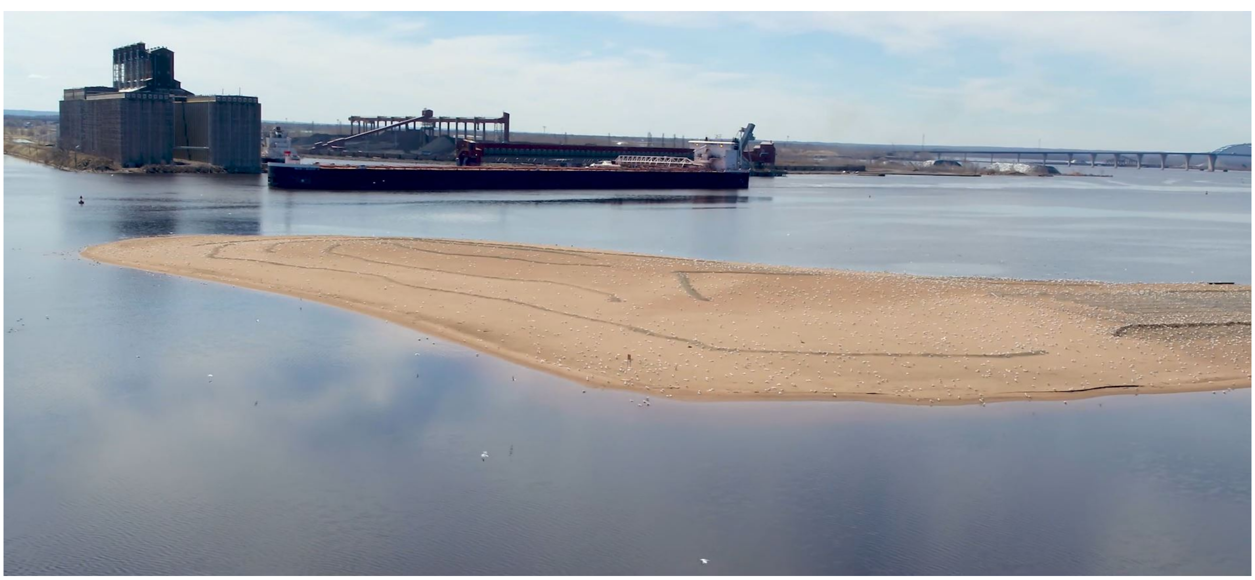
USACE. 2020. St. Louis & Douglas County, Duluth – Superior MN/WI, Duluth – Superior Harbor, FY20 Maintenance Dredging. Certified Final Plan Set. U.S. Army Corps of Engineers, Detroit District, Detroit, MI.

Attachment A. Before Photos:




Attachment B. After Photos:

April 2021. Photo Credit: J.F. Brennan



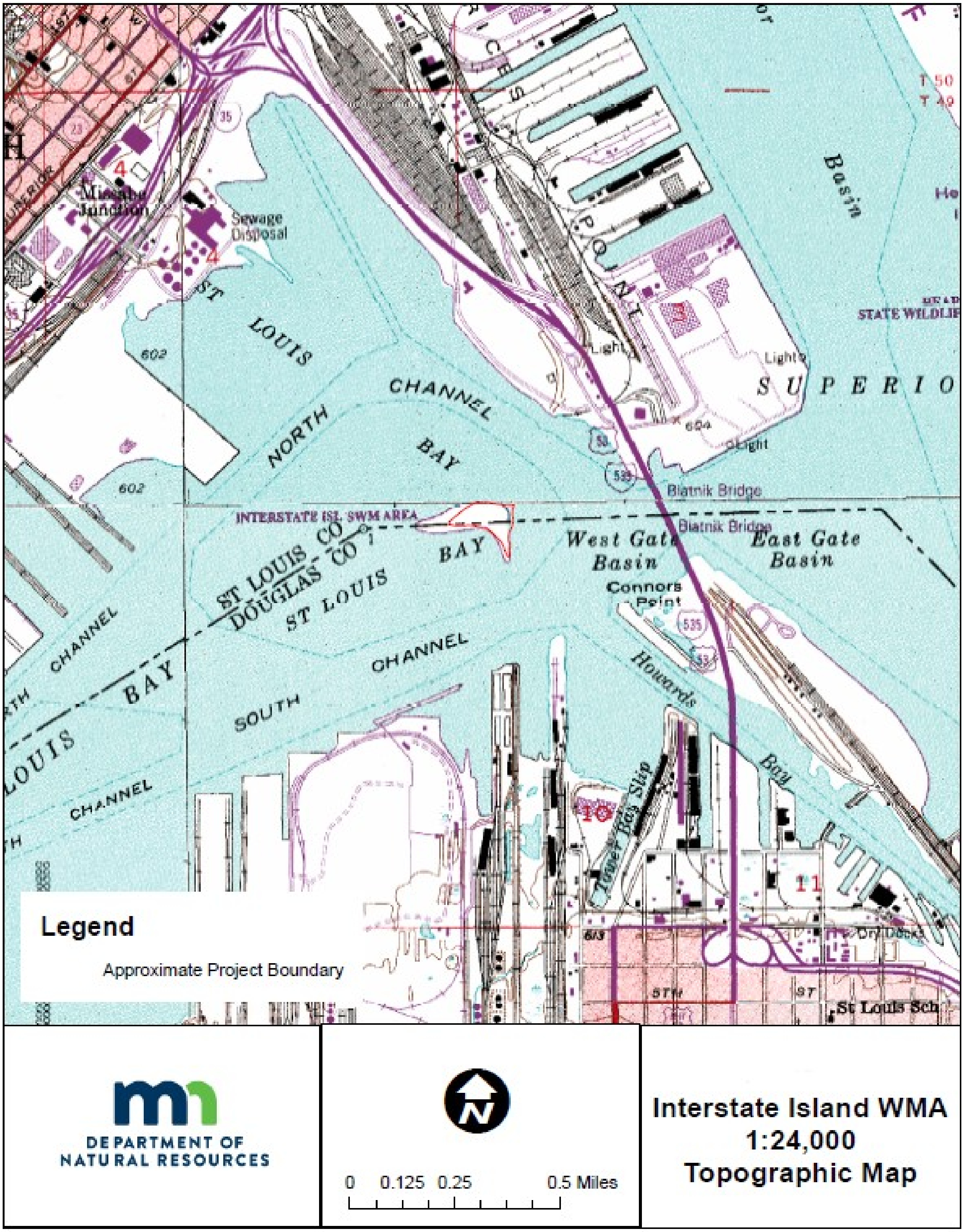
April 2021: Photo Credit: SEH, Inc.

April 2021: Photo Credit: SEH, Inc

April 2021: Photo Credit: SEH, Inc


May 2021: Photo Credit: Fred Strand

Attachment C. Project Maps:



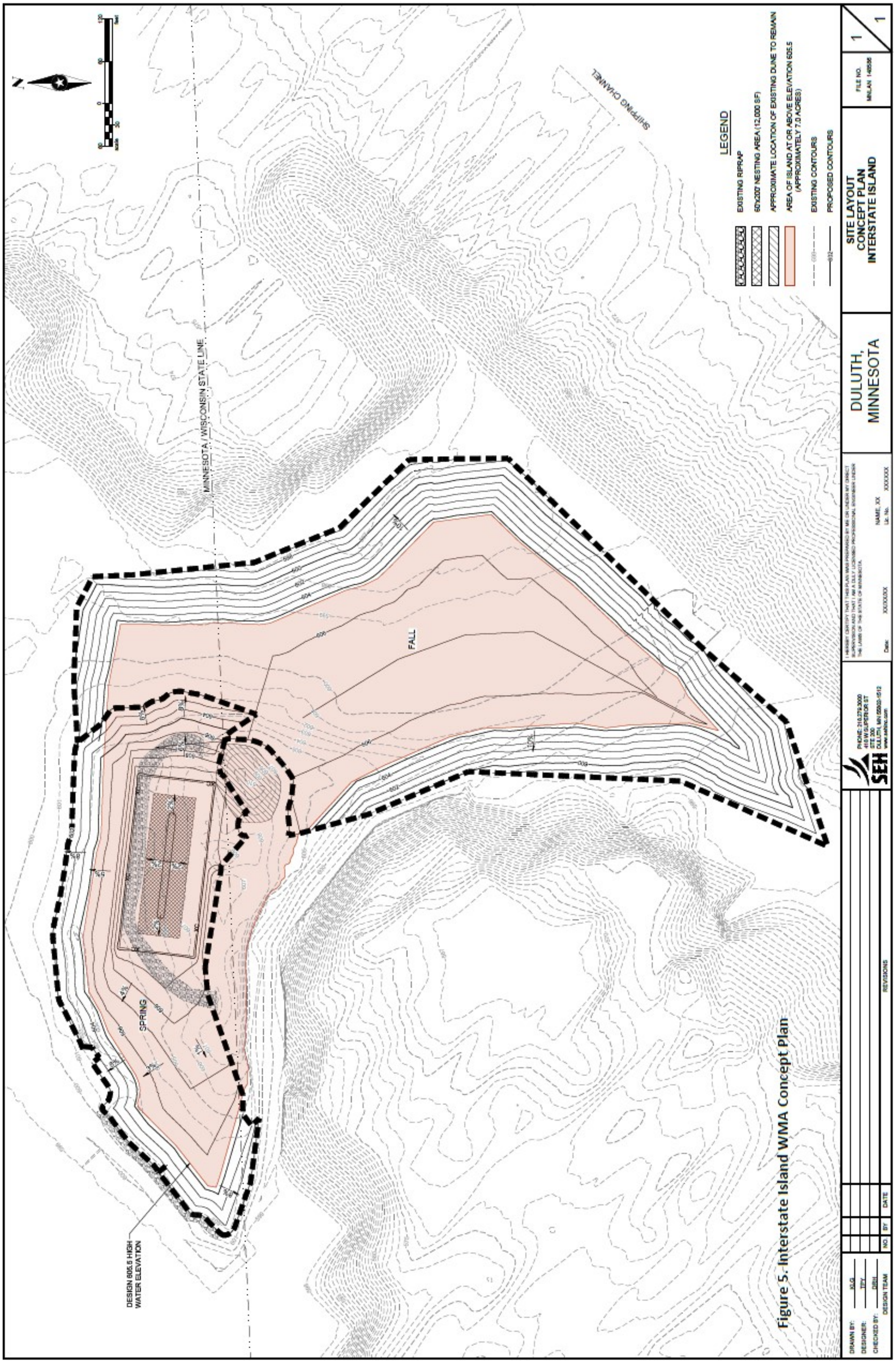



Figure 5. Interstate Island WMA Concept Plan

# Appendix I

# Public Involvement Process

Working together to protect, restore, and enhance the St. Louis River

St. Louis River Alliance
394 Lake Avenue S, Suite 208
Duluth, Minnesota 55802-2338

November 22, 2022

Melissa Sjolund MNDNR
Matt Steiger, WDNR
SLRAOC Coordinators

Re: Support for Proposal to remove the St Louis River Area of Concern Degraded Fish and Wildlife Populations

Dear Melissa Sjolund,
On behalf of the Board of Directors (BOD) of the St. Louis River Alliance (SLRA) I would like to inform you that we have reviewed the Beneficial Use Impairment Removal Package in great detail.

As stated in the BUI removal recommendations, the removal target for degraded fish and wildlife populations will have been met when: *"diverse native fish and wildlife populations are not limited by physical habitat, food sources, water quality, or contaminated sediments."* As you well know, this is a tremendously difficult goal to quantify, with great challenges around the lack of baseline data, the attribution of population limitations to legacy impacts, the ongoing emergence of new climate and chemical impacts on the estuary, and the basic ecosystem dynamics that influence fish and wildlife population equilibria.

In the face of these challenges, the AOC has developed and executed a Remedial Action Plan to address the limitations to fish and wildlife populations specifically posed by legacy pollutants and legacy habitat degradation. The SLRA has been involved in this process from early days, and we have appreciated the rigor with which the Remedial Action Plan has been developed and evolved in response to better information over the years.

In reviewing the BUI removal package, the SLRA BOD remains concerned that according to the data presented, some estuary fish and wildlife populations have not reached full recovery, and that certain pollutants with both legacy and modern sources such as mercury may indeed have an increasing impact on fish and wildlife populations in the estuary. We also recognize of course that full recovery of estuary fish and wildlife is outside of the limited scope of the AOC program. And we acknowledge the impossibility of measuring and achieving the BUI removal target as originally written.

At the same time, we agree and indeed celebrate that the process established by the Remedial Action Plan to address and remove this impairment has been completed. Having recently led establishment of the St. Louis River National Water Trail, we are seeing the larger community benefit from the tremendous work that has been done to restore the home of our native fish and wildlife. It is a truly exciting time as we see the hard work and investment of decades coming to fruition.

While we join the AOC in marking this excellent step, we also take two primary lessons from this deliberation. The first is the critical importance of measurable targets, and a data driven approach to the removal of the remaining BUIs. As much as possible, future removals must be based on measurable outcomes achieved, in addition to actions completed. The second is that we must remain vigilant in tracking and addressing ongoing threats from legacy pollutants. The SLRA supports continued recovery through completing remaining AOC management actions, removing remaining Beneficial Use Impairments, and implementing recommended future actions outside of the AOC program.

With all of this in mind, and after considerable discussion, the BOD took formal action on this matter on November 22nd, 2022, passing a resolution supporting the removal of the Degraded Fish and Wildlife Populations Impairment in the St. Louis River Area of Concern. We agree with the recommendation put forward by the Wisconsin Department of Natural Resources (WDNR), Minnesota Department of Natural Resources (MNDNR), Minnesota Pollution Control Agency (MPCA) and the Fond du Lac Band of Lake Superior Chippewa to request to the United States Environmental Protection Agency (USEPA) Great Lakes National Program Office's (GLNPO) to approve removal of the St. Louis River Area of Concern Degraded Fish and Wildlife Populations.

The St. Louis River Alliance has been actively involved in the Area of Concern Process and has been participating in the discussions of the specific actions that have been completed by the WDNR, the MPCA, and the MNDNR staff. We look forward to continuing our work together to remove the remaining beneficial use impairments and to the eventual delisting of the St. Louis River Area of Concern. We will remain vigilant in ensuring that the future actions starting on page 41 of the removal package are completed through programs outside the AOC and trust that the agencies responsible for managing these actions will diligently work towards the best interests of the St. Louis River. Completion of this work and documentation that all actions have been taken is a tangible milestone for the delisting of the St. Louis River Area of Concern. This is a major accomplishment, and we thank you for your work and commitment to this process.

Sincerely,

Kristi S Eilers
Executive Director
St. Louis River Alliance

**NRRI Duluth**
5013 Miller Trunk Highway
Duluth, Minnesota 55811
218-788-2694

**NRRI Coleraine**
One Gayley Ave / PO Box 188
Coleraine, Minnesota 55722
218-667-4201

April 22, 2022

To: Minnesota Department of Natural Resources
Minnesota Pollution Control Agency
Wisconsin Department of Natural Resources

From: Annie Bracey, Avian Ecologist, Natural Resources Research Institute
Fred Strand, Avian Expert
Sumner Matteson, Avian Ecologist, Bureau of Natural Heritage Conservation, WI DNR
Martha Minchak, Assistant Area Wildlife Manager, MN DNR – Wildlife
Gaea Crozier, Nongame Wildlife Specialist, MN DNR - EWR
Alexis Grinde, Wildlife Ecologist, Research Program Manager – Avian Ecology Laboratory, Natural Resources Research Institute

Re: Comments on the proposal to remove an impairment designation from the St. Louis River Estuary Area of Concern (AOC) for Degraded Fish and Wildlife Populations Beneficial Use

Dear AOC resource managers,

In reviewing the target removal goals set for delisting Common Tern (*Sterna hirundo*), we cannot support the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary. The primary metrics for assessing whether recovery goals are successfully met are based on quantitative measurements of the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair). The Common Tern population in the St. Louis River has not met these recovery goals.

The Common Tern colony in Ashland, WI provides an excellent natural control for comparing breeding site conditions and population dynamics with the St. Louis River colony (Interstate Island). Both colonies have been monitored long-term (>35 yrs.) using the same survey methodology and chiefly documented by the same individual. Therefore, between colony comparisons are sound.

It has been suggested that factors occurring during the non-breeding season (i.e. outside the scope of the AOC program) could influence tern survival, which is a commonly argued point that is easy to default to when causes are uncertain. However, understanding and accounting for full life-cycle population dynamics is fundamental to studying migratory animals. Given birds from both colonies are part of the same regional breeding population and knowing they share the same migratory routes and wintering locations implies that any factors affecting survival or fitness during the non-breeding season should affect survival and productivity at both colonies in the same relative magnitude and direction. This has not been shown to be the case. The Ashland colony is a younger aged population that is increasing in size while the Interstate colony is an older aged population that is decreasing in size. Although the causes associated with colony declines are unknown, determining whether they are legacy related issues will require more time and dedicated research.

Given the initial stated recovery goals have not been met for this species and that there is evidence to suggest the Interstate Island population is being limited by conditions on the breeding grounds (whether legacy related or not), removal of the BUI 2 is premature at this time based on these limitations and data gaps.

Of behalf of members of the Avian Technical Team,

Annie Bracey

Annie Bracey

To:
Annie Bracey, brace005@d.umn.edu
Fred Strand, fcstrand@gmail.com
Sumner Matteson, Sumner.Matteson@wisconsin.gov
Marth Minchak, martha.minchak@state.mn.us
Gaea Crozier, gaea.e.crozier@state.mn.us
Alexis Grinde, agrinde@d.umn.edu

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Annie Bracey, Fred Strand, Sumner Matteson, Martha Minchak, Gaea Crozier, and Alexis Grinde,

Thank you for your contributions to St. Louis River Area of Concern (AOC) program, your participation on the Avian Technical Team, and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

We understand your commitment to the recovery of avian species and passion for Common Terns and can assure you that BUI 2 removal will not result in wavering of the commitment of states' natural resource management programs.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern. The objective has changed twice, in 2017 and 2019. These changes are reflected in the annual RAP updates. Both changes were made in consultation with the Avian Technical Team and were part of an annual RAP public comment process. A comparison of the Interstate Island breeding colony with other colonies has not been a requirement of the AOC removal target or objectives.

The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. This has been a common topic in Technical Team meetings and discussions with the conclusion that the scope of the AOC program is more limited than states' endangered resources recovery and wildlife management programs. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

By acknowledging the migratory status of Common Terns, we are not dismissing the fact that local populations are below the WI recovery goal and research indicates these shortcomings are likely due to local factors. While there is no question that Common Terns complete critical life stages in the estuary, the fact is that most of their time is spent elsewhere. The responsibility of meeting or sustaining Common Tern recovery goals lies with state and federal threatened and endangered resource and wildlife management programs.

Through the AOC program, it is appropriate to addresses legacy contamination and habitat loss by prioritizing and implementing management actions that will have the greatest positive impact on estuary fish and wildlife. For the Common Tern, Interstate Island habitat restoration was identified by the Avian Tech Team as the priority and has been completed with many upgrades to the nesting area. The AOC program is committed to decreasing legacy contaminant exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

Based on your submitted comments and follow-up conversations, additional text was suggested by the Avian team commenters to highlight continuing concerns for avian species in the St. Louis River. This text was incorporated into the final removal package (see p. 42).

Thank you for your participation in this process and for attending many Technical Team meetings, discussions, and providing multiple comments and edits to the removal package during the process. We look forward to continued avian recovery as we gain knowledge and implement management actions under other BUIs and beyond the AOC program.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

cc:

Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

*Equal Opportunity Employer*

## Sjolund, Melissa (DNR)

**From:** Jasmine Baerg <baerg021@d.umn.edu>
**Sent:** Saturday, April 23, 2022 4:53 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Comment on the removal of BUI 2



---

Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I am against this recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

Thank you for listening to your community,

-Jasmine Baerg

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Jasmine Baerg (baerg021@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

## Sjolund, Melissa (DNR)

**From:** Erin Bergen <berg2838@d.umn.edu>
**Sent:** Friday, April 22, 2022 8:06 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Concerning the BUI, Common Terns, and the St. Louis River Estuary



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Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

Sincerely,
Erin Bergen

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Erin Bergen (berg2838@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

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|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

## Sjolund, Melissa (DNR)


**From:** Ryan Carlson <carl5459@d.umn.edu>
**Sent:** Tuesday, April 26, 2022 1:11 PM
**To:** Sjolund, Melissa (DNR)

**Subject:** Opposition to remove the BUI



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Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

Ryan Carlson

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Ryan Carlson (carl5459@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc: Rick Gitar, Cherie Hagen, Darrell Schindler, Barb Huberty, Matt Steiger, Pam Anderson, Neil Vanderbosch

## Sjolund, Melissa (DNR)

**From:** pat.t.collins@gmail.com
**Sent:** Monday, April 25, 2022 4:46 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Comment on BUI 2 Removal Package Draft



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Ms. Sjolund,
I am pleased to see the removal package for BUI 2 out for public comment. This represents a big milestone for the AOC delisting process. Congratulations to all the people that worked to get to this stage in the process.

Overall, I believe that the BUI 2 removal package presents a compelling case for removal of the Impaired Fish and Wildlife Populations BUI and I support the recommendation. The recommendation to remove this BUI includes well articulated rationale and sufficient documentation of the studies and work done to justify removal of this BUI.

I have the following specific comments about the content of the package:

1. I appreciate the discussion of actions in response to the recommendations of the 1995 RAP update. Those recommendations, and the actions they generated, represented the work of a diverse and committed group of local experts and resource professionals over the course of many years. While this work was done prior to the establishment of the roadmap for delisting, those actions were determined to be necessary for delisting.
2. While the wording of management action 2.01 may be narrower, I believe the intent was to not simply to conduct an inventory but to develop and use bird population inventory data to assess the status and trends of bird populations in the estuary over time. I note that a comparison to prior surveys from the 1970s and an evaluation of local versus larger scale patterns was part of the report. However, the report notes that a few bird species exhibited population trends in the estuary that differ from other regional trends. It may be that this was due to species rarity in the estuary as implied by the BUI removal report but clarifying this to document that the differences are not due to legacy AOC issues may be warranted. Additionally, the results point out the need for both post restoration monitoring and on-going, long-term monitoring in the estuary outside of the AOC delisting effort. This post-AOC monitoring need should be identified in the future actions section.
3. On page 18, the finding related to Bald Eagle populations is accurate with respect to the increasing trend for nesting in the estuary. Factors outside the AOC certainly are drivers of population recovery since the 1970s. However, it should not be overlooked that improvements in AOC fish populations and habitat quality are likely important for nest site selection. Without improvements of environmental conditions in the AOC beginning in the 1970s, eagles may not have recolonized the estuary as a nesting area. Overall, the recovery of eagle nesting in the estuary is a positive sign that actions taken at national and local scales have both made a positive difference for this species.
4. While Tree Swallows are not specifically mentioned in a management action or objective, it is surprising that no mention was made of the research conducted by Christin and Thomas Custer on Tree Swallow nest success and reproduction as an indicator of wildlife populations (see for example Custer et. al. 2018 Ecotoxicology 27, 457-476.) Investigations done by the Custers in the St. Louis River AOC may be relevant to this BUI as they could provide additional evidence of the impact, or lack of impact, of legacy contaminants on wildlife populations in the AOC.
5. The removal package would be strengthened by a more explicit discussion of how the St. Louis River estuary will fit as a component of Minnesota's water quality management framework. In particular, a discussion of how the estuary is, or will be, incorporated into efforts such as the St. Louis River One Watershed One Plan, the St. Louis

River Landscape Stewardship Plan, and other agency-based watershed management efforts would help clarify how the post-AOC conditions will be addressed in a way that protects the past investments in delisting the AOC and prevents future degradation of water quality and wildlife health that could lead to having to identify the area again in the future as an area of excessive beneficial use impairments.

Congratulations again on making significant progress leading to a recommendation to remove BUI 2. I look forward to continued success and the eventual delisting of the St. Louis River AOC.

Sincerely,
Pat Collins

DEPARTMENT OF NATURAL RESOURCES

Pat Collins
pat.t.collins@gmail.com

Re: BUI 2 Impairment Removal – Public Comment Acknowledgement

Dear Mr. Collins,

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). The AOC coordinators leading the BUI 2 removal process have reviewed your comments received during the formal public comment period and have provided the following responses to your comments.

You commented that the report failed to capture that an objective of the bird inventory completed under Management Action 2.01 was to have a baseline study for future comparison. This objective is acknowledged in the 2016 study report and relevant to the removal recommendation, therefore, the final Removal Recommendation has been edited with a statement on page 15 discussing additional applications of the survey data. To support these future comparisons, a recommendation for post-restoration and long-term monitoring was added to the Future Actions section (p. 43). Regarding Management Action 2.01, a statement was also added to page 17 concluding that due to overall rarity of the water-obligate species surveyed, local conditions are not a likely explanation for certain species being observed in historical, but not contemporary, surveys.

We concurred with your recommendation to reference the in-progress "One Watershed, One Plan" as an important tool for ongoing management of the St. Louis River watershed. A brief description of One Watershed, One Plan was incorporated into the discussions of continued management of water quality and physical habitat needs outside of the scope of the AOC program (see final Removal Recommendation report pages 46 and 47).

Your recommendation to reference tree swallow research conducted by Christine Custer was considered but did not result in changes to the final Removal Recommendation. Dr. Custer's work is relevant to the St. Louis River AOC as it provides a comparison between different AOCs but was not chosen as a line of evidence for BUI 2 removal.

Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc:

| | | | |
|---|---|---|---|
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |
| Rick Gitar | Cherie Hagen | Darrell Schindler | |

## Sjolund, Melissa (DNR)

**From:** Maya Enriquez <enriq074@d.umn.edu>
**Sent:** Sunday, April 24, 2022 4:23 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Conservation of Common Terns - Public Comment

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To whom it may concern at the AOC,

In regards to the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition to the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to this document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met recovery goals. I understand additional habitat has been created, however there are apparently continuing issues, and monitoring should be done in order to prevent any future harm to this specific population. Furthermore, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. As a practicing biology scientist and resident of St. Louis County, I would like to voice my opinion that **the BUI should not be delisted**. Thank you for your time and consideration.

Sincerely,
Maya Enriquez

--
Maya S. Enriquez, M.S.
*She/Her*
Lab Technician
University of Minnesota - Duluth

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Maya Enriquez (enriq074@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc:
Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

## Sjolund, Melissa (DNR)

**From:** Brett Howland <yellowquail04@gmail.com>
**Sent:** Friday, April 22, 2022 10:10 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** comment on proposal to remove impairment designation for St. Louis River Estuary Area of Concern



Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition to the recommendation to remove the impairment designation from the St. Louis River Estuary Area of Concern for Degraded Fish and Wildlife Populations based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for terns. The removal of the impairment designation seems premature, and it should not be delisted until additional information is available.

These birds are an incredibly beautiful species with immense ecological importance. I have enjoyed watching them nest and feeding in the area and would hate to see this species decline even more.

Brett Howland

**DEPARTMENT OF NATURAL RESOURCES**

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Brett Howland (yellowquail04@gmail.com),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc:
Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

*Equal Opportunity Employer*

## Sjolund, Melissa (DNR)

**From:** Audrey Huff <huff0114@umn.edu>
**Sent:** Friday, April 22, 2022 10:41 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Comment on the Removal of the Beneficial Use Impairment (BUI)



---

Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

Thank you for your time,

Audrey Huff

--
Audrey Huff

PhD Candidate.
Large Lakes Observatory
Water Resources Science - Limnology and Oceanography
University of Minnesota

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Audrey Huff (huff0114@umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

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The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

## Sjolund, Melissa (DNR)

**From:** Halle Lambeau <lambe385@d.umn.edu>
**Sent:** Friday, April 22, 2022 4:15 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** St Louis River Restoration Public Comment



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Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), **I want to voice my opposition for the recommendation to remove BUI 2** (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for terns. The BUI should not be delisted.

Thank you!
Halle Lambeau

--
**Halle Lambeau (she/her)**
M.S. Student
Integrated Biosciences
University of Minnesota Duluth | d.umn.edu
lambe385@d.umn.edu

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Halle Lambeau (lambe385@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

# COMMENT FORM
## for
### St. Louis River Area of Concern
### Removing the Degraded Fish and Wildlife Populations Beneficial Use Impairment

Your feedback is very important to the Minnesota Department of Natural Resources, Fond du Lac Band of Lake Superior Chippewa, and Wisconsin Department of Natural Resources. In the space below, please provide your comments regarding the proposal to remove the Degraded Fish and Wildlife Populations impairment. You may fill out this form and mail it to Melissa Sjolund (MNDNR) at the mailing address on the back or email your comments directly to melissa.sjolund@state.mn.us. Comments must be received on or before 4:30 pm **April 26, 2022**. You may attach additional pages if needed.

*To submit comments or petitions to the AOC agencies through the mail or email, you must state:
(1) Name and address
(2) The action you wish the AOC agencies to take, including specific references to the section of the draft BUI removal you believe should be changed.
(3) The reasons supporting your position, stated with sufficient specificity as to allow the AOC agencies to investigate the merits of the position.

*Note: Because all comments and related information are part of the public record for this proposal, commenters' names and email or postal addresses will be published and publicly available as they appear in the materials they submit.*

Please print clearly:

*Name: WILLIAM C. MAJEWSKI
*Mailing address: 3603 - 95TH AVE W, DULUTH, MN 55808
Email address: bsmajewski@aol.com

Actions desired (please reference pages of the BUI removal document that pertain) & supporting reasons:

WHEN ATTENDING THE BUI 2 SESSION, I WAS DISAPPOINTED TO LEARN THE METRICS FOR PIPING PLOVER & COMMON TERNS HAD BEEN DROPPED FROM THE ORIGINAL MEASURES., WITH RECENTLY COMPLETED HABITAT PROJECTS FOR EACH, IT SEEMS THAT DELAYING THE DELISTING FOR A YEAR OR TWO WOULD BE ADVISEABLE.

Additional Comments: ALSO, THE STURGEON PROJECT HAS NOT PROVEN TO BE SUCCESSFUL YET

Thank you for your feedback!

DEPARTMENT OF NATURAL RESOURCES

William C. Majewski
3603 95th Ave W
Duluth MN 55808
bsmajewski@aol.com

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Mr. Majewski,

Thank you for your interest in the St. Louis River Area of Concern (AOC), your important contributions to the AOC program, and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

You shared a concern that the metrics for the Common Tern and Piping Plover had been "dropped from the original measures." Please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern. The objective has changed twice, in 2017 and 2019. These changes are reflected in the annual RAP updates. Both changes were made in consultation with the Avian Technical Team and were part of an annual RAP public comment process. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The objective for the Piping Plover is similar, with numeric targets referenced, but never required for BUI removal. Habitat restoration projects were completed to support both the Common Tern and Piping Plover, and monitoring and maintenance plans are in place. Delaying BUI 2 removal to continue to monitor progress is not required.

You also expressed a concern that the Lake Sturgeon project has not been successful yet. The BUI 2 Removal Recommendation details the progress and limitations associated with Lake Sturgeon recovery (see discussion of Management Action 2.02 beginning on p. 25). Management actions and objectives associated with Lake Sturgeon have been addressed to the extent possible under the scope of the AOC. Restoration projects targeting sturgeon spawning habitat have been successful and natural reproduction is occurring. Resource managers working with the species concur that while sturgeon are on a trajectory for recovery, more time is likely needed. Future actions supporting continued recovery of Lake Sturgeon are contained in the report and will be pursued by resource managers working in the estuary outside of the AOC program.

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process and advocacy for fish and wildlife populations in the St. Louis River estuary.
Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

cc: Barb Huberty Rick Gitar

*Equal Opportunity Employer*

April 26, 2022

To: Melissa Sjolund
Area of Concern Coordinator
Minnesota Department of Natural Resources
Melissa.sjolund@state.mn.us

From: Willis Mattison, Professional Ecologist, Osage Minnesota

**Subject: Comments and Recommendations on State Agency Proposal to remove the Degraded Fish and Wildlife Populations Beneficial Use Impairment Designation from the St. Louis River Area of Concern (AOC).**

Dear Ms Sjolund,

Thank you for the opportunity to review, comment and make recommendations on the subject proposal.

My qualifications and interests for performing this review and for preparing these comments are that I hold a Masters Degree in Ecology and have 28 years experience as a Regional Director and Water Quality Specialist with the Minnesota Pollution Control Agency (MPCA) stationed in Rochester and Detroit Lakes. I also served on mobility assignment as MPCA's Duluth Regional Director where I had opportunity to become familiar with issues and various actors affecting the community, Lake Superior and the St. Louis Estuary. Since retirement I have served as volunteer technical advisor/advocate for many citizen groups around the state.

Besides my historical experience in the Duluth MPCA office, my more recent volunteer efforts have involved an increasing amount of work with various citizen groups interested in the Duluth/Superior harbor and the St. Louis River estuary area. Citizen's concerns for dredge spoil disposal practices have led to my advising the Protect Lake Superior Coalition (PLSC) that includes a number of state-wide and area environmental and user groups like fishing and boating organizations. My recruitment by the Park Point Community Club's MP-50 Long Range Planning Committee to help address concerns for beach erosion and lake water quality have allowed me to both advise and advocate this group as well.

Consequently, I've had the opportunity to participate in the U.S. Corps of Engineers' recent Section 111 charrette that is presently forming a multi-agency/city/citizen partnership to assess adverse impacts of the Corp's Federal Navigation facility on public and private property on Lake Superior and in the Estuary. And even more recently another citizen group, the Northland Climate Policy Team invited me to learn more about Duluth's Climate Action Work Plan. Meanwhile, the City of Duluth has embarked on its own long range coastal zone protection plan that involves other state and federal agencies like FEMA, NOAA and several emergency management agencies. These activities have led to still more interaction with other public and private educational and research institutions that are gathering data and producing useful information. Each and every one of these groups has slightly different objectives, is viewing the estuary from rather different perspectives and at times, in my experience, have appeared to have somewhat conflicting goals.

So I believe there exists a great opportunity to bridge all the various local, state, federal and NGO activities impacting the watershed, the estuary and Lake Superior under a larger umbrella effort. The AOC team, acting under the auspices of the Great Lakes Initiative may be the most well positioned entity to convene these groups and provide or help establish the needed coordinating and collaborating services.

Each of these volunteer endeavors has led me to become aware of and now review and prepare these comments on the Great Lakes Initiative, St. Louis River Area of Concern draft report.

Understandably, my perspectives in preparing these comments will be considerably different from local or even regional perspectives. A portion of my view will necessarily be quite retrospective in stance and is informed by my 28 year experience employed by a state natural resource management agency (MPCA) and by 20 more years dealing with other state and federal agencies as a volunteer citizen advocate/adviser in my retirement status. And my approach has morphed into a more global view very much informed by recent global climate, biodiversity and ecosystem condition reports from the United Nation’s Intergovernmental Panel on Climate Change (IPCC) and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES).

These United Nation’s reports are clearly providing the basis for my requesting a pause and a re-evaluation of the Great Lakes Initiative’s AOC program as applied to the estuary. The scope, scale and schedules for remedial AOC measure implementation in the St. Louis River estuary region has taken on a much greater global significance than could have been imagined at its outset in 1987, some 35 years ago. The necessity for immediately preserving and restoring vital land, stream, lake and marine ecosystem services has become paramount for combating both climate change and curbing global losses of biodiversity.

Even locally, the Area of Concern (AOC) research findings and implemented remediation of estuary beneficial use impairments (BUIs) may have significant bearing on and can be meaningfully informed by several other studies, planned projects, other activities by various other government resource management entities in the area. The U.S. Army Corps’ Section 111 investigation, the alternatives being considered and their ultimate implementation will undoubtedly impact the Area of Concern.

The Corps' dredge spoil disposal practices are currently being or will be challenged by the Coalition to Protect Lake Superior and dredge spoil management changes are being sought. These changes will very likely impact the AOC team's interests. Duluth's Climate Action Plan will likely produce changes in infrastructure that will have impacts on the estuary. And the Park Point's MP-50 planning will need to be informed by each of these other plans by the AOC team. This is obviously because there are significant overlaps of jurisdiction, ecological resources and geographic area of interest in these several efforts. Deliberate collaboration and cooperation including sharing of gathered data, analysis, goal setting, strategy implementation and outcome monitoring will be vital to the success of each of these efforts and to avoid either duplication of effort or possibly working at cross-purposes.

**My Requested change In Proposed Action:**

1. The removal targets have not been met and Beneficial Use Impairment (BUI) removal for fish and wildlife is not supported by the findings presented in this report therefore the proposed removal has not been justified.

2. The AOC current approach to individual and somewhat siloed (BUI) removals would benefit by shifting from the current rear-facing, single (or few) variant, geographically-defined site approach to a more forward-looking, multi-variant and regional-scaled and more comprehensive/collaborative restoration approach. This proposal has three interconnected aspects:

    A. Adjusting expectations - This would entail attempting to deal with the existing and likely (reasonably predictable) future conditions of the estuary rather than attempting to recreate past fish and wildlife conditions or populations which may no longer be possible. Conversion of the estuary from a richly biodiverse bay-mouth estuary ecosystem to a federal navigation facility has clearly produced a permanently deformed and impoverished (biologically) ecosystem. The estuary simply cannot be restored to what it once was and remain a full service commercial navigation facility. This type of readjustment of expectations would allow the setting of more realistic (achievable) targets.

    B. Readjusting to Regional Geographic Scale- Readjusting the AOC goals toward Identifying and protecting existing biodiverse areas by a more thorough ecological assessment of the region while restoring, as much as possible, specific geographically-defined degraded areas with the expressed goal for increasing their biodiversity would offer a wider array of metrics for judging success (or lack thereof) than the species specific indicators used in the current AOC approach. Existing, well established metrics of biodiversity (such as fish and macroinvertebrate IBI's) already in use in the St. Louis River watershed (MPCA WRAPs) and all around the state might be a better fit for setting objective goals and for monitoring/demonstrating target condition achievements. Such achievements could provide a solid evidence-driven basis for BUI removals.

C. Expanding Inter-agency, Cross-disciplined Coordination & Collaboration – This requires broadening the AOC scope to address both legacy and modern issues by purposeful inclusion of on-going factors that this report suggests may be limiting the achievement of restoration targets. This is necessary because many legacy and modern issues interact geographically, cumulatively and synergistically at the regional scale. And these issues are being managed (or are proposed to be managed) by multiple government jurisdictions such that the current AOC restoration efforts and targets are both obscured and may out of reach of the available AOC resources and authority. Interagency and inter-discipinary coordination won't resolve all these problems but doing so could reduce risk of overlap and counterproductive activities by the various entities acting independently. And there is great potential for leveraging the disparate resources and planned activities to multiply mutually consistent strategies. The Corps's Section 111 authority strongly urges (if not actually mandating) this very level of coordination with other federal, state and local entities proposing to alter components parts of an ecological community. The Corps Section 111 planning team should be contacted with offers to initiate this coordination.

3. Various beneficial uses of harbor sediments currently being used to restore lost or degraded ecosystem functions should be carefully re-examined for efficacy and durability. The Section 111 study has potential for furnishing much of this retrospective for certain parts of the estuary. The public (Park Point Community Club and LSPC have become quite aware of the shortcomings and downsides of using harbor sediments to address shore erosion and for restoring or protecting certain dune and island forest (SNA) ecosystems (actually microcosms) on Minnesota Point. This AOC report now reveals that using harbor sediments to restore aquatic, shore and terrestrial habitat is also falling short of expectations.

   The use of harbor sediments to encapsulate (cap) more contaminated sediments appears to be largely experimental and performance (efficacy) monitoring may reveal less that satisfactory results. State and Federal Agencies risk losing the public trust if they persist in declaring success where little or no success can be demonstrated by long-term post-project monitoring.

4. Expansion of Agency Perspectives and Peer Review - The present AOC team efforts as reflected in the draft report appear constrained by a agency program specialty and administrative influence situation occurs where government professionals are overly reliant on their agency's narrower program objectives and overall missions and have difficulty stepping back to take the broader global perspective needed. Also, strong political desires by upper level management (and local leaders) for removing all BMIs in the shortest time possible for public image purposes may explain the tendencies to lower the bar for objectives or to move the target goal posts as was done here. AOC manager's claim that BMI removals are justified when they clearly have not could be evidence of these influences. Adding a team of outside experts—a science advisory panel—should be considered to offset these influences and protect the professional integrity of the AOC team. Such a panel or counsel, that is actually mandated by provisions of the Minnesota Environmental Policy Act for all state agencies but has seldom, if ever been used would seem highly appropriate and is thus recommended here. Such a panel would be tasked with providing disinterested (lack of interest conflicts) peer review of the applied sciences so important to AOC activities and reassure the public that the proposed actions are well grounded in the applied sciences.
   This recommendation should not be considered as either a fault or a criticism of the AOC team or its advisors. This phenomenon is present in most if not all regulatory agency activities at all levels. The legislative requirement for such advisory councils was in recognition of this phenomenon and is designed to minimize any negative effects.

5. Public Engagement should be geographically and more meaningfully inclusive- The public involvement section of the report lists technical team member activity, the current comment period and St. Louis River summits as key opportunities for the public to learn about and have input into the report. Practically speaking the technical team can not reasonably be said to public involvement, the team consists primarily of government agency specialists. A review of the Summit agendas show a wide variety of presentations so the AOC report was apparently not the single declared purpose of the Summit where the public could go for this purpose. And lastly, the St. Louis River Alliance is listed as a primary focal point and vessel for public engagement. The Alliance presents as an excellent organization with local, possibly even regional sphere of influence and member representation and highly qualified staff. The Alliance's letter of

support is assumed but is not yet confirmed. The Alliance apparently was only recently presented with the AOC's draft removal recommendations and it remains to be seen whether or not this support is forthcoming. It will be important to note whether the Alliance's support represents the informed decision of the staff or of the general membership a large. The latter would be much desired as evidence of robust public engagement.

Beyond that, the Great Lakes in general and Lake Superior and the St Louis River in specific are national and state resource treasures of global significance. This could not be clearer both climate and biodiversity wise in view of the findings by the U.N. IPCC's and IPBES's latest reports. Earlier and more meaningful involvement by state and national (if not global) level individuals and organizations should be actively pursued.

6. Restrictions on Dredging Activities – The fifth of nine BUIs identified at Stage 1 includes "restrictions on dredging activities". But no clear definition of what this impairment was intended to address was found in this report. If these restrictions refer to regulatory or environmental restrictions it raises an important and yet unresolved issue. The existing utilization of harbor dredge material for various BUI remediation and Minnesota Point erosion mitigation may be challenged by the Izzak Walton League and the LSPC for performing activities which, at least historically have required several state and/or federal permits. And some of these permits such as MDNR fill in public waters may require mandatory environmental review under the Minnesota Environmental Policy Act (MEPA) and EQB rules. MPCA Antidegradation review my be required under provisions of the Clean Water Act and federal environmental review may be an appropriate comprehensive tool for umbrella assessment under NEPA. Preparation of a state or federal EIS (or combined state/federal) discretionary EIS may offer one of the best mechanisms for the coordination and collaboration of all public and private activities impacting the watershed, the estuary and Lake Superior. It would be an important gesture on the part of AOC involved agencies if these permits and environmental review were conducted voluntarily instead of adversarily through court or other action-forcing strategies.

This concludes my comments and recommendation.
Thank you.
Willis Mattison

DEPARTMENT OF NATURAL RESOURCES

Willis Mattison
mattison@arvig.net

Re: BUI 2 Impairment Removal – Public Comment Acknowledgement

Dear Mr. Mattison,

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the removal recommendation for BUI 2, you made a general statement that the removal targets have not be met. The AOC's Remedial Action Plan (RAP) is structured to allow BUI 2 removal when management actions are complete, objectives addressed, and targets reached. As described on p. 9 of the BUI 2 Removal Recommendation report, the removal target requires concurrence from resource managers to trigger BUI removal. This concurrence has been achieved and is based on meeting the requirements of the RAP.

Public engagement is an important part of the AOC work, and we are proud to support the St. Louis River Alliance in their mission of connecting citizens with the river. The Alliance assists the AOC program in distributing information, but it is the decision of their Board of Directors if the organization supports BUI removal.

You provided a substantial number of comments directed at the overall organization of the AOC program. The AOC program is administered by the US EPA under the Great Lakes Water Quality Agreement and structural or scoping changes to the program that are suggested do not meet the legacy impairment goals of the current program.

Dredge material management, beach nourishment and permitting processes comments fall outside of the scope of this request for public comment. No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

## Sjolund, Melissa (DNR)

**From:** Elise Miller <mill9104@d.umn.edu>
**Sent:** Monday, April 25, 2022 10:30 AM
**To:** Sjolund, Melissa (DNR)
**Subject:** Conservation of Common Terns



---

Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

Elise Miller

she/her/hers
Master's Student
University of Minnesota-Duluth

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Elise Miller (mill9104@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

April 21, 2022

To: Minnesota Department of Natural Resources
Minnesota Pollution Control Agency
Wisconsin Department of Natural Resources

From: Gerald J. Niemi, Ph.D., Professor Emeritus, University of Minnesota-Duluth
Lee A. Pfannmuller, Retired MNDNR Manager and Audubon Minnesota Ornithologist

Re: Comments on "Beneficial Use Impairment Removal Recommendations for Degraded Fish and Wildlife Populations

We completely reject the recommendation that the BUI removal target has been met for the Common Tern. We applaud the relevant management agencies efforts thus far to consider these species in the St. Louis River Area of Concern (SLRAOC) but faulty logic has been applied to suggest removal of these targets from this Area of Concern.

Gerald J. Niemi is currently an Emeritus Professor in the Department of Biology at the University of Minnesota, Duluth. He has spent over 40 years observing or studying birds in the St. Louis River ecosystem and has published many peer-reviewed papers and reports on its birds. Niemi has published over 160 peer-reviewed papers and over 200 reports and articles on birds in Minnesota, the upper Midwestern United States, and North America, including the Great Lakes. In addition, Niemi was on the original BUI wildlife technical team for the SLRAOC prior to his retirement in 2019.

Lee A. Pfannmuller served on the original Piping Plover Federal Recovery Team, oversaw and provided financial support to numerous field research projects on Minnesota's endangered and threatened species during her career at the DNR, co-edited the book, Minnesota's Endangered Flora and Fauna, and prepared an Implementation Blueprint for Minnesota Bird Conservation and a Common Tern Minnesota Conservation Plan for Audubon Minnesota.

Pfannmuller and Niemi are co- authors of an extensive website on the Breeding Birds of Minnesota (Pfannmuller, L., G. Niemi, J. Green, B. Sample, N. Walton, E. Zlonis A. Bracey, G. Host, J. Reed, K Rewinkel, and N. Will. 2017. The Minnesota Breeding Bird Atlas. Website - (http://www.mnbirdatlas.org) and coauthors of a major book on the Breeding Birds of Minnesota: Their History, Ecology, and Conservation (2022, University of Minnesota Press, Minneapolis MN, *in press*). The Common Tern and Piping Plover both figure prominently in both sources of information.

Common Tern

There are two primary areas where the activity to remove the Common Tern from the BUI are problematic: habitat management and population criteria. As noted in the SLRAOC BUI removal document, "the most critical support required to recover the SLRE Common Tern population was restoration of habitat at Interstate Island." It is appreciated that a project was initiated to "begin" restoration at Interstate Island, but this represented a minimal investment in solving the problem for Common Terns nesting in the SLRAOC. The Common Tern needs ongoing management at Interstate Island and additional potential habitat to survive long-term in the SLRAOC. Interstate Island has provided an excellent nesting site for the birds (although it still requires intensive field efforts to manage and monitor each year), but investment in one site is inadequate because of the potential risk for one event to decimate the population. The risk to the population needs to be spread out over several sites.

Regarding population concerns, we strongly support and endorse the comments and concerns identified by the Avian Technical Review Team. These concerns are summarized here in bold.

**General Comments/Concerns discussed by the Avian Technical Review Team**

**Acknowledging that the common tern nesting habitat at Interstate Island objective has been met does not mean population goals have been achieved.**

**The common tern population is the St. Louis River estuary (SLRE) has not met the recovery goal of a 10-year average of 200 nesting pairs with an average reproductive rate of 0.8 to 1.1 young fledge/breeding pair.**

**The cause/s of the common tern population not meeting the recovery goals has not been determined. The cause/s may or may not be legacy issues.**

**There are two common tern colonies in Lake Superior: one at Interstate Island in the SLRE and the other in Ashland, WI. Habitat at both colonies has been restored/renovated and is annually maintained and managed. Populations at both colonies have been consistently monitored.**

**The Ashland common tern population and the Interstate Island population are part of the same metapopulation. However, the Ashland colony is increasing in size with a decreasing average age. It is a growing population. The Interstate population is decreasing in size with increasing average age. It is a declining population.**

**Bracey et al.'s work found that both common tern colonies have the same migration routes and wintering use areas. Since both populations have the same migration and wintering use areas there is no information to support or to speculate that the Interstate colony is impacted by factor outside of the SLRE while the Ashland colony is not.**

**Differences between the two colonies: the Interstate colony is located in an Area of Concern (AOC) and has gull nesting habitat competition and egg and chick predation by gulls. The Ashland colony is not located in an AOC nor does it have a competition with or predation by gulls. However, the Ashland colony has had predation by mink, Great-Horned Owls and Peregrine Falcons which has not been observed at Interstate.**

**Gull nesting habitat competition and egg and chick predation at Interstate is managed to reduce the impact, but there is an adverse impact. There is no information to support that the gull impact alone is the cause or the only cause for the decreasing Interstate population. Other factor/s must also be negatively impacting the Interstate population. These may be**

**legacy issues. The work by Bracey et al. has shown higher concentrations of mercury in chicks and adults nesting at Interstate Island compared to Ashland Island, with concentrations often exceeding published toxicological risk thresholds.**

**Until the cause/s for the decreasing Interstate population can be determined and the population has met the recovery goal it is not appropriate to state that legacy issues are not the cause and to delist BUI 2.**

To summarize, these issues are particularly troubling with respect to the removal of the Common Tern from SLRAOC. (1) The original goal for the Common Tern was based on the Wisconsin Common Tern Recovery Plan which "*establishes a goal of a 10-year average of 200 nesting pairs with sufficient production of 0.8-1.1 young per breeding pair to maintain population stability in the St. Louis River Estuary"* (SLRAOC Public Comment Draft 2022). Since Niemi was on the original wildlife technical team, he remembers that this was clearly the original goal. Therefore, why was it decided during the process that this goal no longer needed to be met to remove the BUI for wildlife populations? In addition, our understanding is that the experts, who have been researching the Common Tern for years in the SLRAOC, had no say in the removal of this objective. It seems clear that the administrators of the AOC are not listening to the science and simply changing the rules at their convenience.

(2) It is also troubling that the AOC process has not adequately considered the big picture in the conservation of the Common Tern in the western Great Lakes region (e.g., see https://mnbirdatlas.org/species/common-tern/). The SLRAOC is a very important breeding area for the Common Tern in the Western Great Lakes; it is one of only four in the state of Minnesota and one of only two in all of Lake Superior. We believe management and conservation of the Common Tern in the SLRAOC is critical and it is the responsibility of both the MN DNR and WI DNR to be advocates for its long-term success in the SLRAOC. The conclusions for removal of the BUI for Common Tern and the statement that "*Achieving the numeric population goals is not required to remove BUI 2"* (Table 6, SLRAOC Public Comment Draft 2022) is not warranted. Neither the scientific data support this conclusion nor do the activities by the SLRAOC administration justify this conclusion.

Finally, adding *post hoc* a future needs section in the "hopes" that the issues of the Common Tern or all the other wildlife issues that also have been neglected makes the whole AOC process a complete sham. Why has all this time and money been spent on the AOC and BUI delisting process when the goals at the end are simply changed without merit, yet the problems and solutions persist.

DEPARTMENT OF NATURAL RESOURCES

Gerald Niemi, gniemi@d.umn.edu
Lee Pfannmuller, leepfann@msn.com

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Mr. Niemi and Mr. Pfannmuller,

Thank you for your interest in the St. Louis River Area of Concern (AOC), and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning your statement that restoring habitat at Interstate Island is a "minimal investment" that must be expanded to include on-going habitat management and creation of additional Common Tern habitat elsewhere in the estuary, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Through the AOC program, it is appropriate to addresses legacy contamination and habitat loss by prioritizing and implementing management actions that will have the greatest positive impact on estuary fish and wildlife. For the Common Tern, Interstate Island habitat restoration was identified by the Avian Tech Team as the priority and has been completed with many upgrades to the nesting area. In addition, a long-term maintenance and management plan was developed. The long-term plan is jointly implemented by Minnesota and Wisconsin DNRs, the agencies with current and on-going obligations for managing the habitat. It is not appropriate for the AOC program to assume long term responsibility for managing Interstate Island.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern. The objective has changed twice, in 2017 and 2019. These changes are reflected in the annual RAP updates. Both changes were made in consultation with the Avian Technical Team and were part of an annual RAP public comment process. A comparison of the Interstate Island breeding colony with other colonies has not been a requirement of the AOC removal target or objectives. Contrary to Mr. Niemi's recollection, this objective has never required that the WI recovery goal of 200 nesting pairs be met to remove BUI 2 and was not changed "at the end" or as an act of convenience.

The scope of the AOC program is limited to addressing legacy impacts and does not replace existing wildlife management or endangered resource programs. It is not appropriate for the AOC program to be held responsible for fully recovering threatened and endangered species, particularly migratory species vulnerable to impacts outside of the estuary that we cannot control. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover. Based on this understanding, we do not believe it is appropriate for the local AOC program to ensure "big picture" conservation of the Common Tern in the Western Great Lakes Region as you suggest. We agree that Common Tern management and conservation in the estuary is critical and remains the responsibility of state and federal resource managers within existing programs.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and the potential for ongoing limitations that are beyond the scope of the AOC management actions. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

We are sorry that the Future Actions section is not seen as valuable by the commenters. Based on our knowledge of the scope, obligations, and limitations of the AOC program, AOC coordinators have voluntarily included this section to help guide future work. We believe that capturing these recommendations adds guidance and connections to work beyond the AOC scope, work that is already underway.

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process and advocacy for bird populations in the St. Louis River estuary.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc:
Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

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**W.J. McCABE (DULUTH) CHAPTER**
**IZAAK WALTON LEAGUE OF AMERICA**
P. O. BOX 3063. • DULUTH, MN 55803

April 26, 2022

To: Melissa Sjolund
Area of Concern Coordinator
Minnesota Department of Natural Resources
Melissa.sjolund@state.mn.us

**RE: Recommendation to remove the Degraded Fish and Wildlife Populations Beneficial Use Impairment Designation from the St. Louis River Area of Concern (AOC)**

These comments are submitted on behalf of the McCabe Chapter of the Izaak Walton League of America (League), representing approximately 120 members in Duluth, MN and the surrounding area of northeastern Minnesota. Our chapter has long been an advocate for the improvement of the water quality, fish and wildlife habitat, public lands, and outdoor recreation values of northeastern Minnesota. The League recognizes the St. Louis River Estuary as the special treasure that it is in this region and our members value the many benefits and recreational opportunities it provides. The League has been active in the Area of Concern (AOC) designation process, has supported clean up activities on the St. Louis River, advocated for funding for Great Lakes restoration projects at the federal, state, and local level, and participated in restoration projects. While encouraged by the investments being made in the AOC and the progress towards addressing beneficial use impairments with the eventual goal of delisting, the League is concerned that the recommendation for removal of the Degraded Fish and Wildlife Populations Beneficial Use Impairment Designation (BUI 2) is premature. The reasons for our concern are outlined below.

The Public Comment Draft contains two contradictory statements in the Executive Summary:

The removal target will have been met when "in consultation with their federal, tribal, local, and nonprofit partners, state resource management agencies concur that diverse native fish and wildlife populations **are not limited by physical habitat, food sources, water quality, or contaminated sediments**."

and:

"Removal is recommended while acknowledging that **St. Louis River fish and wildlife populations may continue to face limitations caused by physical habitat, food sources, water quality, or contaminated sediments**. Limitations caused by legacy contamination and habitat loss will be further addressed through remaining AOC management actions and natural resource improvements outside of the AOC program."

This recommendation focuses on completion of management actions, as stated on page 51 of the Public Comment Draft: "While BUI 2 removal is based on the successful completion of its listed management actions, continued benefits to fish and wildlife populations will also be realized through activities associated with other SLRAOC BUIs. " The list of restoration activities completed, underway, and planned represents real progress in the AOC, but this does not justify removal of this BUI impairment. Our specific concerns are detailed below.

The sturgeon recovery metric of documenting an increasing trend in 2 – 5 year-old fish captured in summer index nets, and measuring at least two index values greater than 2.0 fish per lift has not been met. It may be true that more time is required for Lake Sturgeon recovery based on generation time and other species habits, but we do not agree with the conclusion that "the SLRAOC program has completed the key actions needed to address legacy impacts to the species and BUI removal is justified despite failure to meet removal targets." (page 31)

The Common Tern population in the St. Louis River Estuary has not met the recovery metric of a 10-year average of 200 nesting pairs with an average reproductive rate of 0.8 to 1.1 young fledge/breeding pair (adopted in 2017). While the cause of the poor Common Tern recovery is not well understood, it's premature to say that legacy issues are not a factor, particularly since the other Common Tern population in the region, in Ashland, Wisconsin, where habitat has also been restored, is growing.

The Public Comment Draft states that "emerging research conducted by Technical Team members independent of the AOC program indicates that mercury pollution may be an issue currently impacting the Interstate Island Common Tern population." (page 11). In fact, the research, which is ongoing, provides direct evidence that exposure to mercury from sediments poses a risk to survivorship of young terns (Bracey et al, *Integrated Environmental Assessment and Management* 2021;17:398–410.
https://setac.onlinelibrary.wiley.com/doi/full/10.1002/ieam.4341

Water quality is a critical factor impacting fish and wildlife, and also a factor in several of the BUIs that remain in place. The Minnesota Pollution Control Agency's recently released list of Impaired Waters for 2022 identifies the following impairments for the St. Louis Bay of the St. Louis River: DDT, Dieldrin, Dioxin, Mercury, PCBs and Toxaphene. These are all legacy pollutants. Scientists working with the Environmental Protection Agency's Duluth laboratory documented sediment mercury levels in the St. Louis River Estuary ten times higher than those in the Bad River in a study published in 2021 and identified legacy sources as the cause. (Janssen et al, *Science of the Total Environment* 2021; 779:146284).
https://www.sciencedirect.com/science/article/pii/S0048969721013528
https://doi.org/10.1016/j.scitotenv.2021.146284

The recommendation states that removing BUI 2 does not require removing the remaining BUIs. While this may make sense from an administrative or policy standpoint, it does not make sense biologically speaking. Fish and wildlife populations are directly impacted by all the remaining BUIs – habitat, water quality, contaminated sediments and food sources. In the absence of numeric metrics for fish and wildlife population recovery, it seems the only meaningful way to measure success is to determine if the target species are present and whether or not their habitat has been cleaned up and restored to the greatest extent possible. Much of this clean-up and restoration work is still underway or not yet begun, as detailed in the Public Comment Draft.

The U.S. Steel Superfund site clean-up is still in progress at this time. This is the largest Superfund site in the Great Lakes and involves sediments contaminated with legacy pollutants: polycyclic aromatic hydrocarbons and metals including copper, lead, and zinc. Remediation activities provide an opportunity for exposure of contaminated sediments and redispersal, in the water and surface sediments through redispersal of contaminants and presents a health risk to fish and wildlife. As we expressed in our 2018 comments on the removal of BUI 3 (fish tumors and other deformities), the League remains concerned that until sediment remediation at this and other sites is completed and monitoring demonstrates that the problem is resolved, legacy sediments continue to pose a threat to fish and wildlife health.

Removal of the BUI designation because management actions have been completed, even though impairments have not been resolved, is unacceptable. Premature removal of the BUI designation will also threaten funding for the ongoing and future work that is required to fully restore fish and wildlife populations in the St. Louis River Estuary, as well as possibly impacting prioritization of projects that address fish and wildlife in the estuary. While the League looks forward to the eventual delisting of the St. Louis River AOC and recovery of fish and wildlife populations, we do not agree with the recommendation to remove the BUI 2 designation at this time.

Recommendation:

- The Degraded Fish and Wildlife Populations Beneficial Use Impairment Designation should remain in place.
- The AOC program should consider adding a management action that directly addresses the risk to sensitive species from contaminated sediments.

Sincerely,

Julie O'Leary

Julie O'Leary
President, McCabe Chapter
Izaak Walton League of America
P.O. Box 3063
Duluth, MN 55803

DEPARTMENT OF NATURAL RESOURCES

Julie O'Leary
McCabe Chapter of the Izaak Walton League's
jloinduluth@gmail.com

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Ms. O'Leary,

We thank the McCabe Chapter of the Izaak Walton League for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Your members are concerned that the BUI 2 Removal Recommendation acknowledges that while we have completed the removal requirements contained in the AOC's Remedial Action Plan (RAP), limitations to estuary fish and wildlife populations remain. As detailed on pages 43-47 of the report, limitations to fish and wildlife populations in the estuary come from many sources and cannot be eliminated through the AOC program alone. We acknowledge this not to be contradictory, but to increase transparency regarding the role and limitations of the AOC program. To clarify, the following statement was added to the Executive Summary (p. ii), "*Limitations are at levels that cannot be addressed by additional management actions under BUI 2. Continued efforts to manage critical nesting habitat, remediate contaminated sediments and restore habitat under other BUIs as well as actions outside of the AOC program, will further benefit native fish and wildlife populations in the estuary*."

Your members request that all AOC projects that restore habitat and remediate contamination must be completed before removing BUI 2. The path towards AOC delisting is unique to each AOC. For the St. Louis River AOC, this path is defined by the RAP; our RAP is structured to permit the removal of BUI 2 prior to remaining BUIs. Removing BUI 2 does not prevent resource managers from implementing the entire RAP and the AOC program is not a substitute for the current and continued obligations of resource managers. The RAP has outlined this systematic approach to BUI removal since 2013 and is open for public input during the yearly update period. Fish and wildlife surveys show a system that is in recovery following a history of degradation. Complete recovery of fish and wildlife populations in the estuary is beyond the scope of the AOC program and not required to remove BUI 2. We acknowledge that work that further benefits fish and wildlife populations continues both within and outside the AOC program.

Your members share an opinion that the AOC program has not completed actions required to address legacy impacts to Lake Sturgeon. The BUI 2 Removal Recommendation details the progress and limitations associated with Lake Sturgeon recovery (see discussion of Management Action 2.02 beginning on p. 25). Management actions and objectives associated with Lake Sturgeon have been addressed within the scope of the AOC program. Restoration projects targeting sturgeon spawning habitat have been successful and natural reproduction is occurring. Resource managers working with the species concur that while sturgeon are on a trajectory for recovery, more time is likely needed. Future actions supporting continued recovery of Lake Sturgeon are contained in the report and will be pursued by the robust fisheries resource managers from many agencies working in the estuary outside of the AOC program.

Regarding the Interstate Island Common Tern colony population, The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. This has been a common topic in Technical Team meetings and discussions with the conclusion that the scope of the AOC program is more limited than states' endangered resources recovery and wildlife management programs. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Your members are concerned that the state's 2022 impaired waters list contains pollutants that have legacy sources and research led by Sara Janssen identifies elevated mercury in the St. louis River. Beneficial use impairments identified through the AOC program are not synonymous with 303(d) water quality impairments listed through the Clean Water Act. The AOC Program addresses impacts to water quality caused by legacy contamination by identifying and remediating hot spots through the Restrictions on Dredging BUI (BUI 5). These projects are in progress and will complete the AOC program's remedial obligations. The AOC program is not the mechanism by which 303(d) impairments are addressed or removed. Ongoing and future water quality management is the existing and continued responsibility of programs within local, state, federal, and tribal agencies.

mercury research is still being conducted by Sarah Janssen to better understand elevated mercury in the St. Louis River for AOC and beyond AOC purposes. The removal document references Sarah Janssen's research (see p. 22, 42, and 47) and recognizes that mercury from current and legacy sources exists within estuary sediments. Sediments with mercury concentrations exceeding remedial action levels are being remediated through the Restrictions on Dredging BUI (BUI 5). Mercury is a contaminant of concern for the Restrictions on Fish and Wildlife Consumption BUI (BUI 1) and management actions are in progress.

Your members worry that premature removal of BUI 2 will threaten funding for ongoing and future work required to fully restore the estuary. Removing BUIs and delisting an AOC signal that key actions required to address legacy impairments have been completed but do not signal a fully restored system. The AOC program is not a surrogate for the current and future management of the estuary resource through existing agencies and programs. Through groups like the Lower St. Louis River Habitat Work Group and Lake Superior Headwaters Sustainability Partnership, resource managers and stakeholders are working hard to use the momentum and structure established by the AOC program to keep the good work going and benefit from current and future funding opportunities. There are many funding opportunities available and currently being utilized outside of the AOC program to support continued work in the estuary.

Thank you for your participation in this process and advocacy for fish and wildlife populations in the St. Louis River estuary.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc:
Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

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## Sjolund, Melissa (DNR)

**From:** Emily Pavlovic <pavlo043@d.umn.edu>
**Sent:** Saturday, April 23, 2022 7:48 AM
**To:** Sjolund, Melissa (DNR)
**Subject:** Opposition to the removal of the Beneficial Use Impairment (BUI)



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Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

**I strongly urge that the BUI 2 is not delisted at this time!**

Regards,
Emily
--
**Emily Pavlovic**
M.S. Student
Graduate Teaching Assistant
Integrated Biosciences Program
University of Minnesota Duluth | d.umn.edu
pavlo043@d.umn.edu (Pronouns: She/Her)

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Emily Pavlovic (pavlo043@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

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|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

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## Sjolund, Melissa (DNR)

**From:** Alexandra Pesano <ampesano@d.umn.edu>
**Sent:** Saturday, April 23, 2022 1:45 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** BUI 2 Public Comment



Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition to the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

The information I've presented above comes second-hand from two trusted scientists, Dr. Alexis Grinde and Dr. Annie Bracey, I know. Drs. Grinde and Bracey have worked very closely with this Common Tern population for a number of years. I support their recommendation for the removal of this BUI.

Sincerely,
Allie Pesano

--
Alexandra (Allie) Pesano
She/Her/Hers
Masters Student | Integrated Biosciences Program
Graduate Teaching Assistant | Biology Department
University of Minnesota Duluth
ampesano@d.umn.edu

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Allie Pesano (ampesano@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

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## Sjolund, Melissa (DNR)

**From:** Nathan Pollesch <nate.pollesch@gmail.com>
**Sent:** Friday, April 22, 2022 8:51 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Opposition to delisting BUI 2



Dear AOC resource manager,

Colleagues who are ornithological experts have brought to my attention the plan to delist BUI 2 and their opposition to it. I echo their opposition.

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

From,
Nathan Pollesch, PhD
Duluth, MN USA

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Nathan Pollesch (nate.pollesch@gmail.com),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

| | | | |
|---|---|---|---|
| Cc: | Rick Gitar | Cherie Hagen | Darrell Schindler |
| Barb Huberty | Matt Steiger | Pam Anderson | Neil Vanderbosch |

## Sjolund, Melissa (DNR)

**From:** prattd@charter.net
**Sent:** Tuesday, April 26, 2022 10:10 AM
**To:** Sjolund, Melissa (DNR)
**Subject:** Comments on the Removal of BUI 2



Melissa,
Thanks for the opportunity of reviewing the Degraded Fish and Wildlife BUI 2 removal document. My comments follow:

My area of expertise is limited to the fishery portion of this BUI 2 removal. I was the Senior Fisheries Biologist for the Wisconsin Department of Natural Resources and co-managed with Minnesota fisheries professionals, fishery strategies on the St. Louis River WI/MN boundary waters from 1985 until my retirement in 2011.
After reviewing the information presented in the St. Louis River Area of Concern Beneficial Use Impairment Removal Recommendation Document, Degraded Fish and Wildlife Populations I fully agree that the key native fish populations are present and not limited by the legacy impairments referenced in the document.
During my time working along with Minnesota biologists, I feel that we were able to both restore the native fish structure of the St. Louis river fishery and bolster that fishery from the impacts of invasive exotics like ruffe and gobies. The health of the fishery is greatly improved from the fishery that I first observed in the latter part of the 1970's as part of a Wisconsin Department of Natural Resources sampling crew. I believe that we've come a very long way in achieving that original fisheries management goal.
Our original strategy of restoring the severely pollution impacted fishery of the 1970's has been very successful. The most important fishery impairment was overcome by improving the estuaries living environment with the enactment of the strategies to restore water quality coming from the Federal Water Quality Act of 1972. Secondly, the two states worked in synchrony to enhance the species that had been impacted by the water quality limitations. Muskellunge, Walleye, Northern Pike, Smallmouth Bass, Black Crappie and other Centrarchids were stocked to boost their population recovery. Lake Sturgeon were also stocked with the long-term goal of eventually reestablishing a self-sustaining population.
Ballast transferred invasives established in the mid to late 1980's triggered an emergency response to bolster the native predator populations even more by enacting more restrictive predator angling regulations including limiting the fishing season, reducing bag limits and size limits, and increasing predator stocking rates. Based on the Ruffe research presented in the document it appears that we have also achieved the goal of limiting the impacts of this invasive on the health of the St. Louis River estuaries fishery.

I thank you for the opportunity of commenting on this beneficial use impairment removal.
Sincerely
Dennis Pratt – Retired WI DNR fisheries biologist and Superior Wisconsin Resident

DEPARTMENT OF NATURAL RESOURCES

Dennis Pratt
prattd@charter.net

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Mr. Pratt,

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and thank you for sharing your firsthand experience and knowledge of the efforts to restore the St. louis River fisheries and their positive outcomes. We agree that we have come a long way since the 1970s and acknowledge the progress achieved through programs such as yours at the Wisconsin DNR.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

cc:
Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

| | |
|---|---|
| **From:** | tr smith |
| **To:** | Sjolund, Melissa (DNR) |
| **Cc:** | Nancy Larson; Bill Majewski; Rick Gitar; Hagen, Cherie L - DNR; Minchak, Martha J (DNR); Hendrickson, Deserae L (DNR); Piszczek, Paul P - DNR |
| **Subject:** | proposal to remove impairment designation |
| **Date:** | Saturday, April 2, 2022 6:36:24 PM |



Please consider these comments on the proposal to remove the Fish & Wildlife Impairment Designation for the St. Louis River Estuary Area of Concern (SLRAOC).

Established goals for the SLRAOC are focused on restoration of uses that have been determined to be impaired. Removal of an impairment designation is not a goal but should be the consequence of achieving the restoration goal.

Your proposal to remove a designation, rather than demonstrating restoration of an impaired use, misrepresents the goals and objectives associated with the SLRAOC. Such a proposal falsely suggests a claim that fish and wildlife populations have been fully restored.

It is admirable that good projects have been planned, funded, and completed, and it is evident that progress has been achieved. However, significant impairment of fish and wildlife populations continues. This ongoing impairment may result from legacy pollutants that remain, or from habitat destruction that has yet to be replaced or restored. It should be clear that much more funding, time, and effort will be required to achieve the fish and wildlife potential of the SLRAOC.

My concerns expressed here are actually verified in the Public Comment Draft recently released:

"Removal is recommended while acknowledging that St. Louis River fish and wildlife populations may continue to face limitations caused by physical habitat, food sources, water quality, or contaminated sediments. Limitations caused by legacy contamination and habitat loss will be further addressed through remaining AOC management actions and natural resource improvements outside of the AOC program."

Recommendations:

The impairment designation should remain visible to the public until the impaired use is restored.

The public should not be told that the goal has not been achieved but they should trust that the problems will be taken care of in the future through other unspecified programs.

SLRAOC impairment designations should be removed only after the corresponding use is restored.

Unmet restoration needs in the St Louis Estuary AOC should be clearly designated and identified to make them eligible for Federal, State, and Private funding opportunities. Premature removal of impairment designations will adversely affect funding for important fish and wildlife restoration initiatives.

Ted

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Ted R Smith

715-296-3905

DEPARTMENT OF NATURAL RESOURCES

Ted Smith, tr.smith.54801@gmail.com

Re: BUI 2 Impairment Removal – Public Comment Acknowledgement Greeting,

Dear Mr. Smith,

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

You share a concern that the BUI 2 Removal Recommendation acknowledges that while we have completed the removal requirements contained in the AOC's Remedial Action Plan (RAP), limitations to estuary fish and wildlife populations remain. As detailed on pages 43-47 of the report, limitations to fish and wildlife populations in the estuary come from many sources and cannot be eliminated through the AOC program alone. We acknowledge this not to be contradictory, but to increase transparency regarding the role and limitations of the AOC program. To clarify, the following statement was added to the Executive Summary (p. ii), *"Limitations are at levels that cannot be addressed by additional management actions under BUI 2. Continued efforts to manage critical nesting habitat, remediate contaminated sediments and restore habitat under other BUIs as well as actions outside of the AOC program, will further benefit native fish and wildlife populations in the estuary."*

You also express an opinion that fish and wildlife in the St. Louis River continue to be "significantly impaired" and that additional work is required. Fish and wildlife data collected in the estuary show a system that is in recovery following a history of degradation. Complete recovery of fish and wildlife populations in the estuary is beyond the scope of the AOC program and not required to remove BUI 2. The RAP is structured to allow BUI 2 removal when management actions are complete and independently of other AOC work focused on other BUIs. We acknowledge that work that further benefits fish and wildlife populations continues both within and outside the AOC program. Removing BUI 2 does not prevent resource managers from implementing the entire RAP.

The AOC program is a temporary designation and a catalyst for long term betterment of the benefits of healthy natural resources. We do not intend for this temporary designation to fully recover the estuary, be directed at issues outside of its intended scope, or be relied on for long term resource management. Specific management actions have been implemented to reach the goals of the AOC work under this BUI. Agencies coordinating the AOC program are preparing for a post-delisting hand off, where resource management remains the responsibility of existing programs. There are many funding opportunities available and currently being utilized outside of the AOC program to support continued work in the estuary.

Sincerely,
Melissa Sjolund

Melissa Sjolund

Minnesota DNR Area of Concern Coordinator
(218) 302-3245

cc:
Barb Huberty
Rick Gitar
Matt Steiger
Cherie Hagen
Pam Anderson
Darrell Schindler
Neil Vanderbosch

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## Sjolund, Melissa (DNR)

**From:** Hannah Toutonghi <touto003@d.umn.edu>
**Sent:** Friday, April 22, 2022 7:07 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Public comment on BUI 2 for Common Terns



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Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for terns. The BUI should not be delisted.

Thank you,
Hannah Toutonghi

--
Hannah Toutonghi (she/her)
Integrated Biosciences Graduate Student
University of Minnesota-Duluth
Email: touto003@d.umn.edu

DEPARTMENT OF NATURAL RESOURCES

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear Hannah Toutonghi (touto003@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

Concerning the recovery of the Common Tern, please see p. 11 of the BUI 2 Removal Recommendation for a detailed description of the Common Tern objectives contained in the AOC's Remedial Action Plan (RAP). Since 2013, the RAP has contained an objective for the Common Tern and has been available for public comment during the yearly update. The WI recovery goal of 200 nesting pairs has been a guide for the AOC program, but contrary to the provided comment, this objective has not been a requirement for BUI 2 removal. The AOC program has been able to support this recovery goal by completing management actions to provide much needed habitat work for specific species listed in the RAP. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

Instead of requiring a numeric population target for BUI removal, the habitat restoration management action was added to this BUI as recommended by local avian experts to contribute to the species recovery in a meaningful way while staying within the scope of the AOC program. The scope of the AOC program is limited to addressing legacy impacts and is better explained in the RAP. State and federal endangered resources and wildlife management programs are better suited to provide long term management of migratory species vulnerable to impacts outside of the estuary. This understanding is similarly reflected in the BUI 2 objectives for both the Common Tern and Piping Plover.

The BUI 2 Removal Recommendation acknowledges that Common Terns may face unique limitations in the St. Louis River estuary. Please see p. 11, 22, and 23 of the BUI 2 Removal Recommendation for a discussion of emerging research and mercury. It is important to acknowledge that mercury sources in the estuary are varied with only legacy sources falling under the scope of the AOC program while current sources are regulated by other programs under the Clean Water Act. The AOC program is committed to decreasing legacy mercury exposure to fish and wildlife (including Common Terns) by completing contaminated sediment remediation at hot spots in the estuary under the Restrictions on Dredging BUI (BUI 5). Many of these projects are located where terns forage. Future study of mercury and Common Terns at Interstate Island is acknowledged as valuable and included in the BUI 2 Removal Recommendation report as a future action to be pursued outside of the AOC program (see p. 41).

No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc: Rick Gitar, Cherie Hagen, Darrell Schindler, Barb Huberty, Matt Steiger, Pam Anderson, Neil Vanderbosch

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## Sjolund, Melissa (DNR)

**From:** K Wolf <wolf0616@d.umn.edu>
**Sent:** Sunday, April 24, 2022 3:50 PM
**To:** Sjolund, Melissa (DNR)
**Subject:** Opposition of BUI Removal



---

Dear AOC resource manager,

In reviewing the documents that are available for public comment (https://www.dnr.state.mn.us/st-louis-river-restoration/public-comment-opportunity.html), I want to voice my opposition for the the recommendation to remove BUI 2 (Degraded Fish and Wildlife Populations) based on the current status of the Common Tern population in the St. Louis River Estuary.

According to the document, the metrics related to the number of breeding pairs and colony productivity (i.e., 10-year average of 200 nesting pairs and productivity rates of 0.8 – 1.1 young fledged per breeding pair) were not met. Based on the data, the Common Tern population in the St. Louis River has not met these recovery goals. I understand additional habitat has been created, however there are apparently continuing issues and monitoring that may need to be mitigated in the future. Further, based on the information in this document, there is evidence that mercury may be an issue for Common Terns. The BUI should not be delisted.

Cheers,
K

Re: Beneficial Use Impairment Removal – Public Comment Acknowledgement

Dear K Wolf (wolf0616@d.umn.edu),

Thank you for your interest in the St. Louis River Area of Concern (AOC) and for your time reviewing and commenting on the Draft Removal Recommendation for the Degraded Fish and Wildlife Populations Beneficial Use Impairment (also known as BUI 2). We have reviewed your comments received during the formal public comment period and provide the following responses.

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No modifications to the final BUI 2 Removal Recommendation were made based on your submitted comments. Thank you for your participation in this process.

Sincerely,

Melissa Sjolund

Melissa Sjolund
Minnesota DNR Area of Concern Coordinator
(218) 302-3245

Cc: Rick Gitar, Cherie Hagen, Darrell Schindler, Barb Huberty, Matt Steiger, Pam Anderson, Neil Vanderbosch

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