Fish Spawning Habitat Restoration Challenges and Management Options for the Laurentian Great Lakes

by

Kerri Koszíwka

B.Sc., McMaster University, 2015

A thesis submitted to the Faculty of Graduate and Postdoctoral Affairs in partial fulfillment of the requirements for the degree of

Master of Science

in

Biology

Carleton University

Ottawa, Ontario

© 2021

Kerri Koszíwka

Abstract

Freshwater fish populations are facing numerous threats including habitat loss and degradation. As such, much effort is devoted to restoring degraded habitats and restoring access to spawning habitats. In this thesis, I conducted a narrative review on sturgeon spawning habitat characteristics and biological productivity outcomes in support of functional monitoring for mandated offsetting measures. While sturgeon life history characteristics limit the feasibility of functional monitoring, this approach can be applied to other fishes. Biotelemetry was used to guide efforts for restoration of fish populations in a river system fragmented by a dam. The dam was a barrier to migration for Lake Sturgeon (*Acipenser fulvescens*), but Walleye (*Sander vitreus*) did not fully migrate to the dam during the spawning season. These findings provide evidence for benefits of restoring river connectivity by removal of the dam but also consider impacts that dam removal would have with further invasion of non-native species.

Acknowledgments

I would like to thank my co-supervisors, Dr. Steven Cooke and Dr. Karen Smokorowski, as well as Dr. Tom Pratt, for their immense support and guidance throughout this project. You have all shared your collective knowledge with me and have helped shape me into the scientist I am today. There were many unavoidable curveballs with this project, from changes in study sites in 2019 due to the flooding of the Ottawa River to a cancelled field season and starting an entirely new project because of the COVID-19 pandemic. The resiliency and determination I have gained in the past year is a credit to your continuous mentorship and patience. Thank you to my committee members Dr. Gabriel Blouin-Demers and Dr. John Dettmers, for your always appreciated insights and advice. Funding for this project was provided by Fisheries and Oceans Canada.

I especially want to thank Fritz Fischer from the OMNRF and Tom Pratt for allowing me to work on the Black Sturgeon River dataset. I gained many skills throughout this process and could not be more appreciative. Thank you also to Kyle Stratton, Eric Berglund, and Tim Cano from the OMNRF and Lisa O'Connor and Bill Gardner from the DFO for their guidance and support compiling tagging methodologies and through various levels of telemetry analysis. I am also so grateful to Ben Hlina and Jill Brooks from the Cooke Lab for their mentorship through this process.

I want to thank the entire Sault Ste. Marie DFO office and their summer students for all of their support during the 2019 field season. A special thank you to Bill Gardner, who was immensely supportive at every turn and always had the answer, no matter the question. Thank you to Jess Taylor and Trina Rytwinski for their guidance with all things systematic review. I also want to acknowledge the researchers that sent additional data for their papers to be used in the narrative review. Thank you to the entire Cooke Lab for your camaraderie, community, and friendship in person in the lab and then virtually on Slack.

Lastly, I want to thank my family and friends for believing in and supporting me. To my parents, George and Charlene, thank you for teaching me the value of hard work and for supporting me on my path. Gabby, thank you for the edits, the insights, and the support every step of the way. And a special thank you for helping me get through the stressful final stages of analysis and writing with constant turtle content. To my Grandma Tili, your kindness, support, and willingness to listen are a beacon of light in my life. Jake, whether it was using you as a sounding board for ideas while out for our walks with Piper, editing my work, or bringing me a tea during lengthy writing sessions, you are a continuous source of support and inspiration.

Table of Contents

List of Tables

List of Figures

List of Appendices

CHAPTER 1: General Introduction

1.0 Introduction

1.0.1 Global decline of freshwater biodiversity

Anthropogenic change has driven Earth into a new geological epoch: the 'Anthropocene' (Crutzen 2002). This unprecedented change to the natural environment on global and local scales has led to dramatic losses of biodiversity (Vitousek et al. 1997). Freshwater ecosystems cover less than one percent of Earth's surface but are home to 10 percent of all species (Strayer and Dudgeon 2010; Dijkstra et al. 2014). The rates of decline for freshwater biodiversity have been greater than their marine and terrestrial counterparts since 1970, with one-third of freshwater species threatened with extinction (Collen et al. 2014; Harrison et al. 2018). Migratory fish that use freshwater have had global population declines on average of 76 percent from 1970 to 2016 (Deinet et al. 2020). Despite these declines, populations that are protected and/or managed showed lower rates of decline; thus, proving the importance of programs such as habitat protections, fishery regulations, and dam removals (Deinet et al. 2020).

Dudgeon et al. (2006) grouped anthropogenic effects in freshwater ecosystems into five major stressors: overexploitation, water pollution, flow modification, destruction or degradation of habitat, and invasion by exotic species. Reid et al. (2019) built upon this framework and identified 12 new or intensified threats, including changing climates; e-commerce and invasions; infectious diseases; harmful algal blooms; expanding hydropower; emerging contaminants; engineered nanomaterials; microplastic pollution; light and noise; freshwater salinization; declining calcium; and cumulative stressors. Of the factors mentioned above, habitat degradation and loss have been identified as the leading cause of population declines in freshwater ecosystems (WWF 2018).

1.0.2 Decrease of available spawning habitat as a driver of population decline

To support healthy and productive ecosystems, habitat quantity and quality are prerequisites for fish productivity and ecosystem function (Lapointe et al. 2014). Further, fisheries management is shifting focus from a single target species to an ecosystem-based approach (Pikitch et al. 2004). One area that has experienced anthropogenic stressors and has numerous projects to rehabilitate fish habitat and facilitate fish population recovery are the Laurentian Great Lakes (hereafter referred to as the Great Lakes). Collectively the Great Lakes occupy a surface area of 244,000 km^2 and have a water volume of 23,000 km^3, roughly 18 percent of Earth's surface freshwater (Waples et al. 2008). The main factors leading to a reduction in fish productivity across the Great Lakes are overexploitation, habitat loss, and invasive species introduction (Collingsworth et al. 2017).

One of the greatest threats to freshwater biodiversity is river fragmentation through dam construction (Dudgeon et al. 2006) that compromises migration and habitat for many fishes (Winemiller et al. 2016), and can limit the potential for recovery through the loss of and access to spawning habitat (Dumont et al. 2011; Thiem et al. 2013). Several sturgeon species, as well as Walleye (*Sander vitreus*), spawn over cobble substrates in shallow, fast-flowing waters (Haxton et al. 2016; Bozek et al. 2011), with historical Lake Sturgeon spawning habitats known to have been sites of dam construction for hydroelectric power generation (Haxton 2006). Dams have been constructed throughout the Great Lakes watershed, dramatically fragmenting fish habitat in tributaries (Neeson et al. 2015).

Under the federal *Fisheries Act* in Canada, if installation or construction of a hydroelectric generating station impacts fish and fish habitat, offsetting measures (i.e., actions taken to offset adverse effects) and subsequent monitoring actions can be mandated. Effectiveness monitoring aims to evaluate if offsetting measures result in measurable benefits for fish and fish habitat (DFO 2012). Full effectiveness monitoring programs are time consuming and costly and regulators are looking to streamline this challenging process. A possible solution has been identified through support for functional monitoring, which uses indirect measures of fish productivity (i.e., physical habitat characteristics) that are hypothesized to support fish production (Braun et al. 2019).

While there is much debate surrounding dam removal, migratory fish have been found to recolonize previously accessible habitats when barriers are removed (O'Connor et al. 2015), with positive effects in both the short and long-term (Birnie-Gauvin et al. 2017, 2018). While there are positive impacts for native species with restored connectivity, negative consequences through further dispersion of invasive species need to be considered (McLaughlin et al. 2013). Of particular concern for the Great Lakes and its tributaries are the invasive, parasitic Sea Lamprey (*Petromyzon marinus*) as they do not exhibit site fidelity for spawning (Bergstedt and Seelye 1995). Increases of spawning ability in one tributary could impact parasitic stage abundance throughout the lake afterwards. As noted in Smith and Tibbles (1980) Sea Lamprey predation has also contributed to the decline in populations of top predators [i.e., Lake Trout (*Salvelinus namaycush*) and Burbot (*Lota lota*)] as well as other species key to commercial, recreational, and Indigenous fisheries [i.e., Walleye and Lake Whitefish (*Coregonus clupeaformis*)]. Thus, if a barrier is removed, a replacement barrier or additional budget for lampricide treatments is necessary to control the Sea Lamprey population from further dispersal and establishment (Jensen and Jones 2018).

To date, engineering systems that address human water needs are more of a consideration than ecosystem functioning and biodiversity (Garcia-Moreno et al. 2014). Many regions worldwide are expanding hydropower production (Zarfl et al. 2015), despite impacts on overall ecosystem functioning, whereas dam removal has been shown to increase fish abundance (Watson et al. 2018). Destruction or degradation of habitat and invasion by exotic species (Dudgeon et al. 2006), as well as expanding hydropower (Reid et al. 2019) have been identified as major anthropogenic stressors and threats to freshwater ecosystems. Thus, to aid with rehabilitation of freshwater fish populations, habitat degradation and loss of spawning habitats should be met with management options for habitat restoration as well as increased access in fragmented river systems (Hartig et al. 2018). Thus, data-driven recommendations through evidence-based management must be used in the Great Lakes (Landsman et al. 2011), where research should be conducted collaboratively and inclusively, with respect and engagement from all partners (Cooke et al. 2020).

1.1 Objectives of the present study

The research aims to highlight spawning habitat restoration challenges and management options in the Laurentian Great Lakes through two lenses: a narrative review assessing the feasibility of functional monitoring for sturgeon spawning habitats, and through analysis of Walleye and Lake Sturgeon (*Acipenser fulvescens*) movement downstream of a dam to assess whether dam removal is likely to benefit populations of these species.

In Chapter 2, the use of physical parameters, such as water velocity, depth, and substrate size as a proxy for biological metrics such as egg deposition and larval drift will be synthesized to see if functional monitoring is feasible for assessment of sturgeon spawning habitats. This will be achieved by completing a narrative review and data extraction for studies conducted on natural and artificial spawning habitats for sturgeon species. This narrative review is the first step towards seeing if there is a connection between biological metrics of productivity and physical habitat characteristics for sturgeon species that is well-established and defensible.

In Chapter 3, fish movement data will be analyzed to determine if the recovering Walleye population in Black Bay are river spawners that use the lower Black Sturgeon River for spawning habitat. The extent of Walleye and Lake Sturgeon migration in the lower Black Sturgeon River will also be determined. The Camp 43 Dam is located 17 km from the mouth of the Black Sturgeon River and impedes access to the upper reaches during spawning migrations for Walleye, Lake Sturgeon, and other native fishes (Bobrowicz 2010). The proposed removal of the Camp 43 Dam is a contested and intricate management issue as removal increases access for non-native species, including Sea Lamprey. This research aims to present science-based recommendations regarding the removal of the Camp 43 Dam.

Chapter 2: Can physical parameters be used as a proxy for biological metrics in support of functional monitoring?

2.1 Introduction

2.1.1 Functional monitoring

In Canada, Fisheries and Oceans Canada (DFO) ensures that aquatic ecosystems are protected from negative impacts of humans and invasive species and through the *Fisheries Act* that regulates ecosystems to protect fish and fish habitat. Specifically, the Fish and Fish Habitat Protection Program [FFHPP; formerly the Fisheries Protection Program (FPP)] within the DFO administers the fisheries protection provisions of the *Fisheries Act* and can administer letters of advice or *Fisheries Act* authorizations that require monitoring activities. There are three hierarchical levels of monitoring used by the DFO: compliance, functional, and effectiveness monitoring (DFO 2012). The first, compliance monitoring, is not a science-based approach and focuses on operational activities (i.e., whether terms or conditions that have been mandated are being met). The next level, functional monitoring, is defined "as a science-based, scaled-down version of effectiveness monitoring that relies on surrogate metrics to assess whether management measures provide expected conditions suitable for fish to carry out their life processes" (DFO 2012, 2019). The most in-depth form of monitoring is effectiveness monitoring, "a science-based activity, requiring a standardized, transferable design. The metrics or indicators must measure productive capacity or fish-based surrogates of productive capacity" (Smokorowski et al. 2015).

Braun et al. (2019) conducted a global review of the use of functional monitoring for mitigation, restoration, and offsetting projects and found that standardized monitoring is key to ensure metrics are measured, recorded, analyzed, and reported consistently. With standardized methods in general, a meta-analysis of program results can be used to determine the overall effectiveness of offsetting, mitigation, or restoration measures by assessing effects observed (Braun et al. 2019). Functional monitoring uses indirect measures of fish productivity (i.e., physical habitat characteristics) that are hypothesized to support fish production (Braun et al. 2019). The main objective of functional monitoring programs is to determine if management measures, such as mitigation, restoration, and offsetting that are mandated, are functioning properly (Braun et al. 2019). Functional monitoring aims to be used in systems where impacts are smaller and/or more well-understood, but still uses scientifically defensible monitoring standards. Also, programs that use effectiveness monitoring can later transition to functional monitoring and vice versa, depending on results (Braun et al. 2019). It is important to note that determining decreases or increases in fish productivity, as a result of management action, is the main objective of effectiveness monitoring, not functional monitoring.

As an example, if an offsetting measure such as the creation of spawning habitat for sturgeon species downstream of a hydroelectric generating station is mandated, an effectiveness monitoring program would assess changes in fish productivity (or index thereof; e.g., measure egg or larval fish density, abundance, biomass, growth, etc.), whereas functional monitoring would measure the functioning of the mandated measure (i.e., quantity and suitability of substrate offsets for spawning; Braun et al. 2019). Thus, if a well-established and consistent connection can be made between physical habitat characteristics (i.e., water velocity, depth, and substrate) and biological productivity metrics (i.e., eggs and larvae), in theory functional monitoring may be used in place of a full effectiveness monitoring program. For example, the creation of an artificial spawning shoal that was mandated could be considered functional if water velocity, depth, and substrate composition met species-specific benchmarks that are well-established and tied consistently to productivity.

2.1.2 Current status of sturgeon species

There are 27 sturgeon species on the International Union for Conservation of Nature (IUCN) Red List, listed as critically endangered (16), endangered (3), vulnerable (4), near threatened (2), and least concern (2; IUCN 2020). Primary threats to sturgeon globally include overfishing, harvesting roe for caviar, pollution, and habitat fragmentation and destruction (WWF 2020). Sturgeons have a slow growth rate, late age of maturity, and exhibit spawning periodicity, causing them to have slow recovery potential (IUCN 2020). In North America, there are nine species of sturgeon: Shortnose Sturgeon (*Acipenser brevirostrum*), Lake Sturgeon, Green Sturgeon (*Acipenser medirostris*), Atlantic Sturgeon (*Acipenser oxyrinchus oxyrincus*) and the Gulf Sturgeon (*Acipenser oxyrinchus desotoi;* a sub-species), White Sturgeon (*Acipenser transmontanus*), Pallid Sturgeon (*Scaphirhynchus albus*), Shovelnose Sturgeon (*Scaphirhynchus platorynchus*), and Alabama Sturgeon (*Scaphirhynchus suttkusi*; Haxton et al. 2016). General life history observations for sturgeon species include: spawning in riverine freshwater habitats; that suitable spawning habitat availability is imperative for reproductive success; spawning success and recruitment is highly unpredictable and may not occur if river flows are too high; spawning site fidelity is observed; and spawning sites are often just downstream of major rapids or other barriers to migration (Bemis and Kynard 1997).

Due to their reliance on riverine environments for spawning and recruitment, and the prevalence of barriers in many of these systems, sturgeon species are frequently listed as species at risk. In Canada, Lake Sturgeon are listed under the Ontario *Endangered Species Act* (*ESA*) as Threatened and are listed federally under the *Species at Risk Act* (*SARA*) as Special Concern in northern populations. They are also listed as Endangered and Threatened in western and Great Lakes populations, respectively, by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and recommended for listing under the *SARA* (COSEWIC 2017). The Maritimes and St. Lawrence populations of Atlantic Sturgeon are listed as Threatened by COSEWIC and recommended for listing under the *SARA* (COSEWIC 2011). In the Pacific Region, Green Sturgeon and in New Brunswick and Nova Scotia, Shortnose Sturgeon are listed under COSEWIC and *SARA* as Special Concern (COSEWIC 2004, 2014, 2015). Lastly, in British Columbia, the Kootenay, Nechako, Upper Columbia, and Upper Fraser River populations of White Sturgeon are listed as Endangered under *SARA.* Under COSEWIC, the Upper Columbia, Upper Fraser, and Upper Kootenay River populations are listed as Endangered, and the Lower Fraser River population is listed as Threatened (COSEWIC

2003, 2012). As sturgeon have slow recovery potential (IUCN 2020), and are frequently listed as species at risk, habitat rehabilitation is important for recovery.

2.1.3 Sturgeon spawning habitat requirements

There are consistencies in spawning habitat preferences among sturgeon species in North America, as summarized in Haxton et al. (2016; transcribed in Table 2.1). Data summarized in Table 2.1 were extracted from species specific status reports. Across all sturgeon species, spawning substrate varies from gravel (2-16 mm) to boulders (>256 mm), depths conducive to spawning vary from 0.1 to 12 m, and water velocity ranges vary from 0.1-2.0 m/s. While there are similarities for spawning habitat requirements among sturgeon species, including coarse substrates and high water velocities, the observed ranges are vast. For example, water velocity requirements for sturgeon spawning range from barely flowing (0.1 m/s) to flows where sampling would be extremely difficult (2.0 m/s). While there are spawning habitat ranges for Shortnose Sturgeon, Atlantic Sturgeon, Gulf Sturgeon, and Pallid Sturgeon, peak suitability has not been determined. There are also data gaps in the literature including substrate composition requirements for Shovelnose Sturgeon and Alabama Sturgeon, depth ranges for Green Sturgeon, White Sturgeon, Shovelnose Sturgeon, and Alabama Sturgeon, and velocity ranges for Green Sturgeon, Shovelnose Sturgeon, and Alabama Sturgeon. However, there are limitations for certain species (i.e., White Sturgeon) where spawning occurs in deep and sometimes turbid water, which makes observations difficult (Hildebrand et al. 2016).

Table 2.1: Spawning habitat requirements for North American Sturgeons, adapted from Haxton et al. (2016). NA denotes that specific information was not given in the original article.

Species	Substrate	Depth (m)	Water Velocity (m/s)	Citation
Shortnose Sturgeon	Mixture of rubble and smaller rocks	1–5	0.7 (0.2–1.3)	Kynard et al. 2016
Lake Sturgeon	Coarse substrate interspersed with boulders and large rocks	0.1–6.0 up to 12	0.34–2.0	Bruch et al. 2016
Green Sturgeon	Cobble and gravels with interstices or irregular surfaces	NA	NA	Moser et al. 2016
Atlantic Sturgeon	Hard bottom substrate (rocks and gravel)	2–11	0.46–0.76	Hilton et al. 2016
Gulf Sturgeon	2–10 cm gravel	0.15–9.5	0.1–1.5	Sulak et al. 2016
White Sturgeon	Coarse	NA	>1	Hildebrand et al. 2016
Pallid Sturgeon	Outside revetted bends over or adjacent to gravel, cobble or rock	>3	>1	Jordan et al. 2016
Shovelnose Sturgeon	NA	NA	NA	Phelps et al. 2016
Alabama Sturgeon	NA	NA	NA	Kuhajda and Rider 2016

In the context of a review, it is more useful to determine peak suitability for spawning habitats than to state broad ranges in sturgeon spawning parameters for easier comparability. Baril et al. (2018) conducted a quantitative review of Lake Sturgeon spawning habitat characteristics based on data from 48 sites within their range. Peak suitability for Lake Sturgeon spawning occurred at an average water velocity of 0.6 m/s, depths of 0.55-0.85 m in small rivers (i.e., rivers with <100 m^3/s annual average discharge) and 0.75-5.25 m in large rivers (i.e., rivers with >100 m^3/s annual average discharge), and in areas with cobble substrates (64-256 mm diameter). In contrast, the full ranges reported for each metric were water velocities between 0.03-2.51 m/s, depths between 0.03-25 m, and substrate sizes between 0.01-256.1 mm. Although Lake Sturgeon spawning can occur within these larger ranges, egg density increases as water velocity increases from 0.4 to 0.6 m/s (Johnson et al. 2006) and egg density decreases as water velocity increases from 0.6 to 1.1 m/s (LaHaye et al. 1992). This further supports 0.6 m/s as the peak suitable water velocity for Lake Sturgeon spawning.

2.1.4 Threats to sturgeon spawning habitats and creation of artificial spawning habitats

One of the greatest threats to freshwater biodiversity is river fragmentation through dam construction (Dudgeon et al. 2006), which impacts migration for many fishes (Winemiller et al. 2016). In a global review of obstructions in freshwater ecoregions, almost 50 percent were found to have rivers obstructed by medium and large-sized dams, with approximately 27 percent having more than one obstruction (Liermann et al. 2012). Similarly, Grill et al. (2019) assessed the connectivity of 12 million kilometres of rivers globally to identify rivers that remain entirely free-flowing (i.e., have no dams); only 37 percent of the world's very long rivers (i.e., >1,000 km) remain free-flowing (Grill et al. 2019). Strategic removal of dams, if operation is disproportionately costly or they have become obsolete, or modification to include effective fish passage, is recommended for increasing connectivity (Grill et al. 2019).

Sturgeon species spawn in freshwater riverine habitats, usually downstream of major rapids or other barriers to migration (i.e., dams, waterfalls, etc.; Bemis and Kynard 1997). As there are many threats to sturgeon spawning habitat, following management and rehabilitation strategies is of great importance for the recovery and health of sturgeon populations (Arthington et al. 2016). General conservation measures that apply to several sturgeon species include actions such as: maintaining appropriate spawning habitats, increasing access to river reaches across dams, establishing minimum flow regimes, and decreasing suspended sediments that may lead to sedimentation (Billard and Lecointre 2001). For all North American sturgeon, overexploitation, habitat degradation and loss, and barriers to spawning, rearing, or feeding areas are threats (Haxton et al. 2016).

Using Lake Sturgeon as an example, strategies such as barrier removal and fish passage at dams (Auer 1996), establishing run-of-the-river flow regimes, downstream guidance and diversion structures, water quality improvements (Kerr et al. 2010), enhancement of young-of-the-year (YOY) and juvenile survival (Gross et al. 2002), creation or enhancement of spawning habitats, and maintaining flow during spawning and drift seasons have been identified as ways to mitigate the impacts of dams (Kerr et al. 2010). Recovery programs for Lake Sturgeon have been in place since the late 1970s (Bruch et al. 2016). Past studies have shown that artificial spawning habitats can be successful if the physical characteristics are within the known preferred ranges for Lake Sturgeon spawning, including having sufficient water flow and depth, coarse cobble and rubble substrate with boulders, and are free of fine sediment (Kerr et al. 2010). Artificial spawning habitat creation has been used to help address Lake Sturgeon population declines when recruitment is limiting. However, it is difficult to determine if the installation of artificial spawning substrates leads to population increases as sturgeon are long-lived, late-maturing (i.e., 12-20 years for males and 15-30 years for females; COSEWIC 2017), and exhibit spawning periodicity (Kerr et al 2010; Thiem et al. 2013). For Lake Sturgeon, Fischer et al. (2018) noted that despite having 12 years of data post-construction of artificial spawning habitats for Lake Sturgeon, the data are unable to detect long-term population responses. Thus, continued monitoring is required to appropriately assess the effectiveness of artificial spawning habitats for population recovery (Fischer et al. 2018).

Under the federal *Fisheries Act* in Canada, if a project proposed such as developing or changing operations at a hydroelectric generating station is likely to affect productivity due to 'harmful alteration, disruption or destruction of fish habitat', offsetting measures may be required. For spawning habitat creation to be accepted as an offsetting measure, the effectiveness of the created habitat to increase productivity must be monitored (Smokorowski et al. 2015). An ongoing monitoring project by Fisheries and Oceans Canada aims to determine the efficacy of artificial spawning habitats created specifically for Lake Sturgeon. Initial findings from the 2019 field season are summarized in Appendix A.

2.1.5 Efficacy of artificial spawning habitats

Taylor et al. (2019) completed a review on the efficacy of techniques used to enhance or create spawning habitat for substrate spawning fish. This technique is often employed to increase fish productivity after degradation by development, such as installing and operating hydroelectric generating stations. However, due to low study validity and limited replication, the efficacy of many of these created spawning shoals could not be confirmed. Taylor et al. (2019) classified studies as having low, medium, or high validity, and only high validity studies were included in the review. Internal validity includes study design, replication, control matching, measured outcome, outcome method, intervention application coverage, and confounding factors, and is the measure of how likely a study is to be free from bias. External validity includes how generalizable or relevant the study is and is captured by the reviewer (Bilotta et al. 2014; Collaboration for Environmental Evidence 2018; Taylor et al. 2019).

Through a synthesis of available studies, Taylor et al. (2019) showed that the addition or alteration of rock material was an effective form of spawning habitat enhancement; however, five of the six high validity studies were completed on salmonids (Taylor et al. 2019). While Taylor et al. (2019) only included studies with high validity, Rytwinski et al. (2019) included studies with medium and low validity, resulting is a total of 134 studies and 359 datasets reviewed. Findings from Rytwinski et al. (2019) showed that the addition of rock material (i.e., sediment, gravel, and/or cobble, removal of sediment, and gravel washing) to spawning habitats was the most commonly used offsetting measure that achieved increases in abundance. Results were given for the addition or alteration of rock material, as well as the sub-categories of gravel, cobble, gravel washing, and rock combinations. Specifically, with the addition of cobble, the weighted-mean percent change in effectiveness for increasing abundance was higher for non-salmonid fishes (i.e., 22% CI: -6.69, 51.48; n = 20) than for salmonids (i.e., -1.34% CI -35.54, 32.86; n = 23); of the non-salmonid studies 13/20 were completed on sturgeons (i.e., 11 on Lake Sturgeon and two on White Sturgeon). This indicates that the type of rock addition to the substrate may be species- or genera-specific. In Taylor et al. (2019), using high quality studies, the weighted-mean percent change in effectiveness for increasing abundance with addition of rock material was higher than in Rytwinski et al. (2019), when low, medium, and high quality studies were included (i.e., 90% CI:

75.02, 105.43; n = 6 and 18% CI: 1.32, 35.12; n = 78, respectively). As higher quality studies generally have better study design and execution this may lead to higher effectiveness. In Taylor et al. (2019), five of the six studies were completed on salmonids, which may also explain variation and points to the importance of species- or genera-specific reviews. Although addition of cobble was found to achieve increases in abundance, long-term studies are also needed to assess whether artificial spawning habitats maintain their original construction designs to determine continued effectiveness.

Fischer et al. (2020) completed a long-term evaluation of artificial spawning habitats installed in the St. Clair-Detroit River System for Lake Sturgeon, Walleye, and Lake Whitefish. The installation of the Belle Isle Reefs in the Detroit River was completed in 2004, and in the St. Clair River, the Fighting Island Reefs were constructed in 2008, and the Middle Channel Reefs were constructed in 2012. To quantify substrate changes, annual Sonar surveys started in the St. Clair River in 2012, underwater video surveys began in 2015, and sonar surveys started in the Detroit River in 2016 after the installation of the Grassy Island Reef. Years studied range from 1-6, where the Middle Channel Reef had the largest dataset of six years since baseline (i.e., 2012-2018). After evaluating the substrate changes from 2012 to 2018, sediment accumulation was observed at all reefs. Although accumulation of dreissenid mussels was observed at Hart's Light Reef, Lake Sturgeon eggs were regularly collected at the site, and provides evidence that dreissenid mussel shells may provide spawning substrate or do not degrade substrate (Fischer et al. 2020). Recommendations for long-term monitoring of sediment composition were given as the composition of spawning substrate can change or degrade over time, which is likely to influence the ability of the reef to provide successful spawning habitats (Fischer et al. 2020).

2.2 Objectives

The objectives of Chapter 2 are 1) to collect and synthesize available physical habitat characteristics (i.e., water velocity, depth, and substrate composition) and biological metrics of productivity (i.e., egg deposition and larval drift) for sturgeon spawning habitats to see if there is a consistent link, and 2) to determine if egg and/or larval data differ between natural spawning habitats and artificial spawning habitats for sturgeon. This will be achieved through the completion of a narrative review and data extraction for studies conducted on natural and artificial spawning habitats for sturgeon species.

2.3 Methods

2.3.1 Search terms and publication databases

English language search terms, outlined in Table 2.2, were used for database searches. Initial database searches were completed on March 7, 2020 and additional searches were conducted on December 28, 2020 to capture any records published after the initial search. The search term is made up of three categories (population, productivity outcome, and physical habitat outcome) using the Boolean operator "OR" within them. An asterisk (*) was used to allow for any results with differing end characters. For example, spawn* would capture any words beginning with "spawn" and is open to various ending characters (i.e., spawns, spawning, spawner, etc.). The three categories were combined using the Boolean operator "AND". To increase the potential for capturing all relevant articles, general search terms such as "habitat", "spawn*", and "reproduct*" were added to the physical habitat outcome terms. Searches were completed using two databases, the ISI Web of Science core collection (hereafter Web of Science) and Scopus. Several search strings were tested in both Web of Science and Scopus against a list of 5 relevant articles (i.e., Auer and Baker 2002; Bouckaert et al. 2014; Chiotti et al. 2008; Dumont et al. 2011; Thiem et al. 2013). The set of search terms used is the result of numerous iterations, resulting in a search string with the purpose of increasing the comprehensiveness of the search to include all potentially relevant articles. Using the search string shown in Table 2.2, all of the aforementioned relevant articles were results of both Web of Science and Scopus searches, except Thiem et al. (2013) in the Scopus search, as it is not available in that database. The search term was modified to fit the functionality for each database (i.e., "TS=" in Web of Science and "TITLE-ABS-KEY" in Scopus). Resultant references were imported into MS-Excel, and duplicates were deleted.

Table 2.2: Search string used for database searches.

Component	**Search string**
Population terms	TS = ((sturgeon OR Acipenser) AND
Productivity Outcome Terms	(egg* OR larva*) AND
Physical Habitat Outcome Terms	("water velocity" OR flow* OR depth OR substrate OR habitat OR spawn* OR reproduct*))

2.3.2 Screening of articles

Resultant publications from the two searches were evaluated for relevance to the objective, based on inclusion criteria at three levels: title, abstract, and full-text. If the reviewer was uncertain of inclusion or exclusion at any level, articles were included and reviewed at the next stage. Papers were evaluated based on their titles and all articles deemed to be potentially relevant to the objective were then reviewed at the abstract level. Each publication found to be potentially relevant, based on the abstract, was reviewed at full-text. Studies rejected at the full-text screening stage are provided in Appendix B, with reasons for exclusion. The Interlibrary Loans program at Carleton University was used to acquire digital copies of articles for full-text review that were unavailable through Carleton University Library databases.

2.3.3 Study inclusion criteria

Articles were assessed for inclusion based on the relevancy of the subject, intervention, outcome, study, and language.

Relevant populations

Any population of sturgeon/Acipenser.

Relevant types of interventions

General sturgeon spawning assessments and interventions including the creation or enhancement of spawning habitat were included.

Relevant types of outcome

Quantitative measurements of biological productivity (i.e., eggs and/or larvae) and physical habitat characteristics (i.e., water velocity and/or depth and/or substrate) were included. If a study collected larval drift data at differing distances downstream from a singular spawning location, only data from the closest downstream location were included.

Relevant types of study

Any primary field study was included. Any laboratory-based research or study with metrics that were modelled was excluded unless modelled outcomes were confirmed in the field.

Language

Only English-language articles were included at the extraction stage.

2.3.4 Potential effect modifiers

Potential effect modifiers from articles included at the full-text stage were recorded, including study location, species life history, sampling methodology (including replication), and study duration.

2.3.5 Data extraction strategy

Meta-data from studies deemed to be included after the full-text review stage were extracted and entered in an MS-Excel database with predetermined coding. Information on study characteristics, measured biological outcome data, physical habitat characteristics, and any potential effects modifiers were recorded. Data from figures were extracted using WebPlotDigitizer. For certain articles, where methodologies indicated a specific metric was sampled but not included in the article, corresponding authors were contacted (via email) for access to unpublished data.

2.3.6 Data synthesis and presentation

The results and methodologies of all included articles at full-text were described in a narrative synthesis. The studies included in this narrative review are available in Additional File 1 in table format with the following details: author(s), title, year, country, province/state, name of river, year of data collection (if multiple), spawning habitat location, if the spawning area is natural or artificial, size of spawning area (if given), sturgeon species and common name, biological productivity outcome and methodologies, and physical habitat metric outcomes and methods. Studies included varied in study designs, in duration of data collection, collection methods, reported outcomes, and the quantity of habitat sampled. Densities were calculated if possible based on the information presented in the study (i.e., had known size and amount of egg mats used). Densities could not be calculated for many sites and as such, those results were reported as abundances. Some studies differed based on ecological contexts (i.e., spawning habitat confirmation vs. long-term studies). Thus, a quantitative synthesis at this point is not possible.

2.4 Results

The database search yielded 628 unique studies (364 duplicates removed) where 52 of the 70 studies reviewed at the full-text stage were eliminated for various reasons including outcome metric, invalid collection methods, or missing data or information, among others, leaving 18 studies. The corresponding authors of the 15 studies that were excluded due to lack of data were contacted via email but received no response, or the data sent still did not meet the requirements of the review. This review narratively synthesized 18 studies that collectively had n = 63 datasets from differing years and sampling locations. Additional data were received from the authors for three studies (i.e., Dammerman et al. 2020; Fox et al. 2000; Poytress et al. 2015), which were not explicitly provided in the original published articles. Data from an unpublished 2019 DFO sampling program were also included with an additional n=4 datapoints for Lake

Sturgeon egg data and n=5 datapoints for larval Lake Sturgeon data across five sites (Appendix A). A ROSES (i.e., RepOrting standards for Systematic Evidence Syntheses) flow diagram shows the number of studies included at different stages of the review (Figure 2.1; Haddaway et al. 2018).

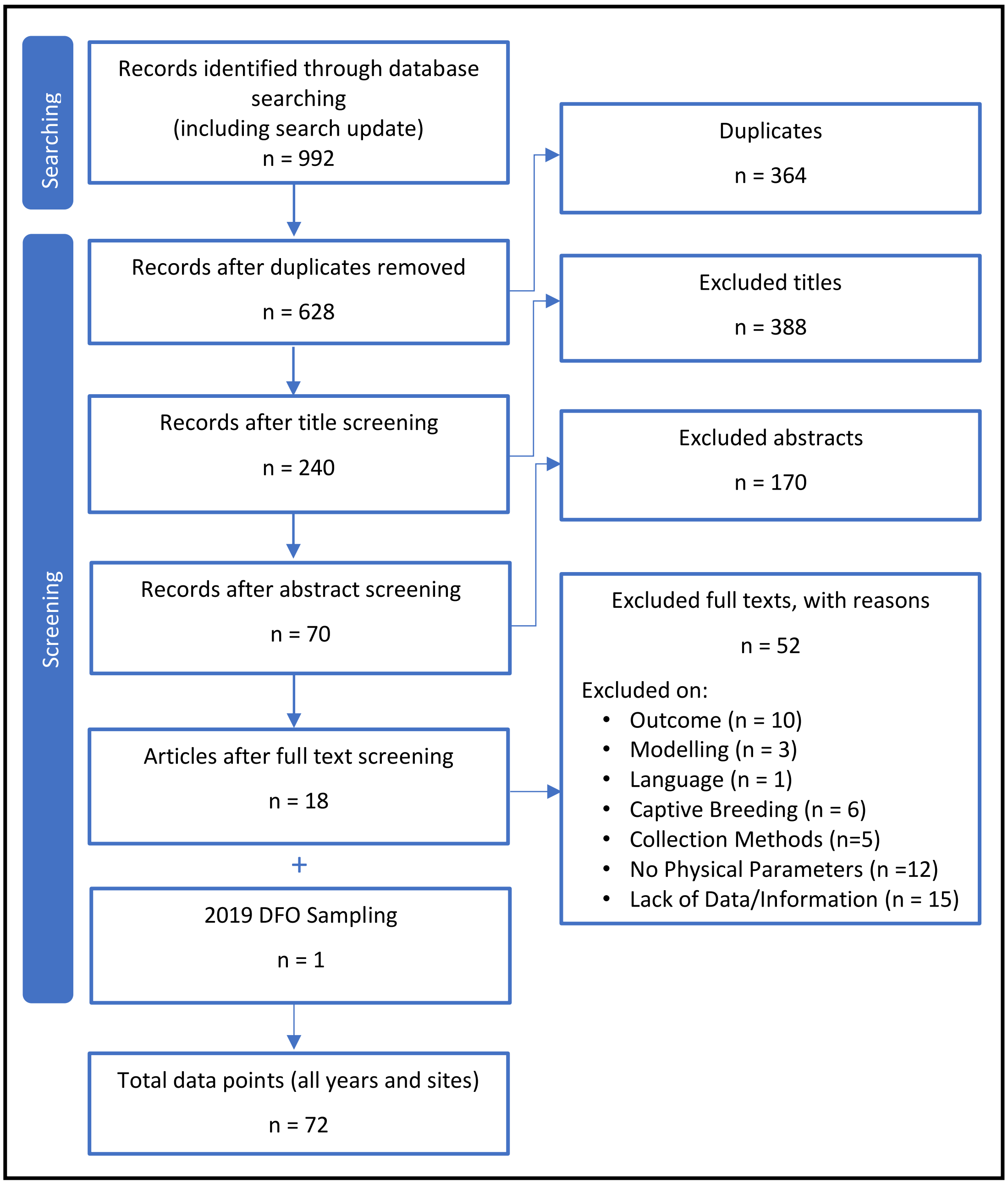

Figure 2.1: ROSES flow diagram showing literature sources as well as the inclusion and exclusion process.

2.4.1 *Characteristics of studies included in the narrative synthesis*

The 18 included studies were published between 1998 and 2020 generally at a rate of 1-2 per year (Figure 2.2). All of the included articles were written in English and were academic peer-reviewed journal articles. While there were no geographical restrictions for study inclusion, all included studies were from the US (13) or Canada (4), with one study that had sites in both countries. Studies conducted on Lake Sturgeon (12) were conducted in the US states of Michigan and New York and the Canadian provinces of Ontario and Quebec (Figure 2.3). Studies conducted on Gulf Sturgeon (3) had study sites in Alabama, Georgia, and Florida. Studies on White Sturgeon (2) were conducted in Idaho and Oregon. Only one study was conducted on Green Sturgeon (in California).

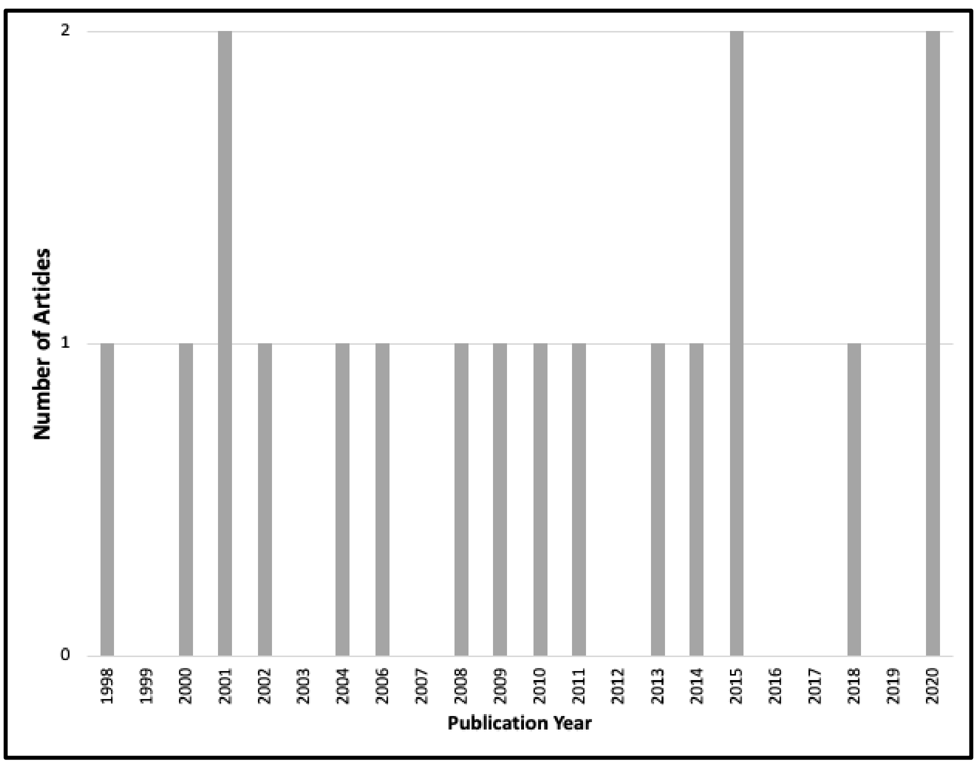

Figure 2.2: Number of articles showing distribution by year of publication.

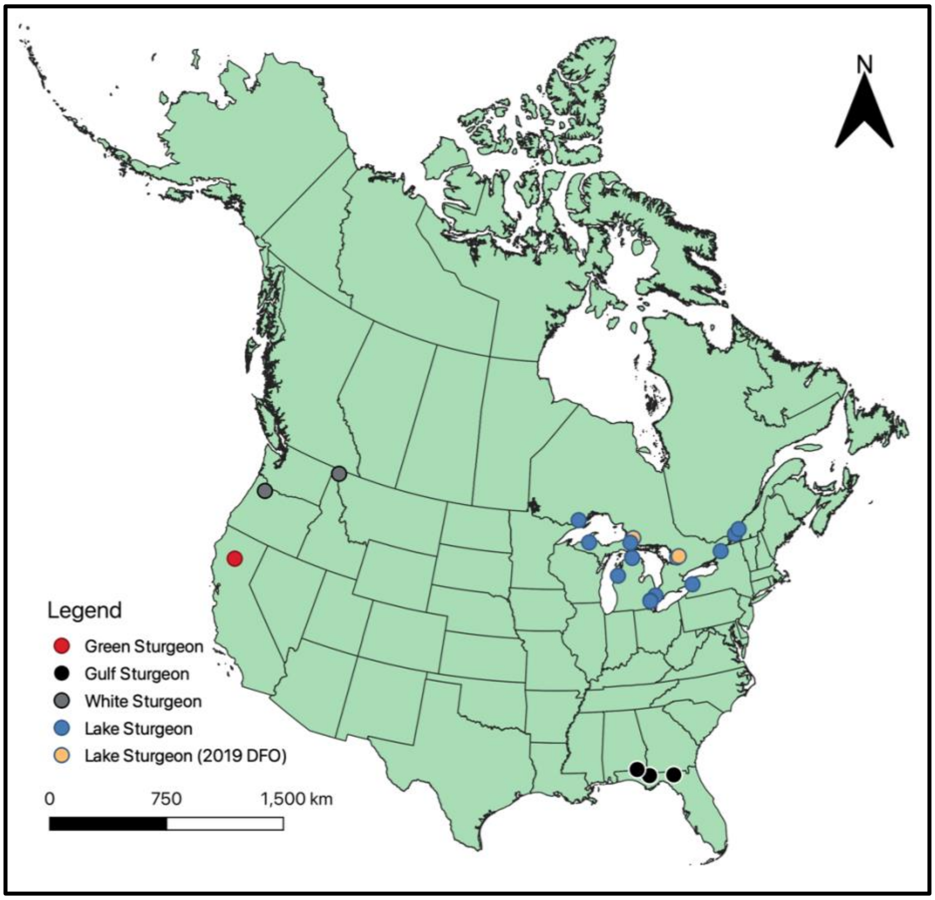

Figure 2.3: Map of Canada and the US with study sites for each of the 18 studies and the unpublished DFO sites that were used in the functional monitoring narrative review (Appendix A). The datapoints for Roseman et al. (2011) and Casewell et al. (2004) appear as one point due to the scale of this figure as the locations are within the same area of the Detroit River. The data points for the 2019 DFO sampling for the Moon and Musquash River sites also appear as one point due to the scale of this figure. Each species of sturgeon is indicated by a coloured circle, where grey is White Sturgeon, red is Green Sturgeon, blue and orange are Lake Sturgeon, and black is Gulf Sturgeon. GIS data for Canada and the US was provided by Statistics Canada and the United States Census Bureau, respectively. Map created in QGIS.

2.4.2 Narrative synthesis

This review included studies that measured a biological metric of productivity (i.e., eggs and/or larvae), as well as at least one physical habitat metric (i.e., water velocity and/or depth and/or substrate composition). If both of these metric categories were not included in the article, it was excluded from the summary.

The methodologies and deployment strategies for all studies that collected sturgeon eggs were compared (Table 2.3). The studies varied greatly based on egg collection technique from a metal rod to dislodge eggs that were subsequently collected by a kick net, to egg mats and egg collectors. Also, when reported, the size of egg mats/collectors differed substantially, as the smallest had a surface area of 0.076 m^2 and the largest had a surface area of 1 m^2. The number of mats used and duration of sampling also varies with some studies only confirming sturgeon spawning (i.e., deploying five mats in one gang for three days; Caswell et al. 2004) and others sampling throughout the entire spawning season (i.e., deployed from mid-May to mid-June and checked every second day; 2019 DFO Sampling).

Egg abundance or density measurements, as well as physical habitat data for each site that collected eggs across all sturgeon species, were summarized (Table 2.4). Reported outcomes varied, with some studies reporting a total abundance of sturgeon eggs collected across the sampling period [ranges of one to 2,277 eggs (natural spawning sites)], while others report average eggs per metre-squared [ranges of 0 to 175.7 for all sites, 0 to 175.7 (artificial spawning habitats), and 0.3 to 171.7 (natural spawning sites)]. For water velocity and depth, some studies reported a general range that was observed during sturgeon spawning, whereas others gave an average across the study period. For substrate composition, outcomes ranged from an average substrate size (in mm) to a general description of the substrate (e.g., fine and coarse gravel). Some notable examples of differences in egg deposition abundances between years at the same spawning sites exist, despite using similar methodologies in each year of sampling. Despite sampling a larger area in 1995, Sulak and Clugston (1998) collected 0.7 eggs/m^2 (abundance of 49) Gulf Sturgeon eggs in 1995 and 368 in 1996 from a natural spawning site at river kilometer (rkm) 215 in the Suwannee River. White Sturgeon egg abundances collected in Paragamian et al. (2001) ranged from two eggs collected in 1993 and 393 eggs collected in 1998 from a spawning location near Bonners Ferry in the Kootenai River. In the Sacramento River, nine Green Sturgeon eggs were collected from rkm 424.5 in 2009 and 93 were collected in 2010 (Poytress et al. 2015). There are also examples of varied distribution and density of eggs throughout the spawning season at both artificial and natural spawning sites. Original densities for each sampling event at an artificial spawning habitat in Johnson et al. (2006) in 1996 show the varied distribution and density of eggs throughout the spawning season (Table 2.5). Similar to Johnson et al. (2006), zero eggs/m^2 were collected during 13 of the 15 sampling events for the Musquash B natural spawning site and 2,532.9 and 42.8 eggs/m^2 were collected during the remaining two (2019 DFO Sampling).

Water velocities where sturgeon eggs were collected ranged from 0.34-1.11 m/s (Lake Sturgeon), 0.2-1.2 m/s (Gulf Sturgeon), 0.19-0.9 m/s (White Sturgeon), and 0.2-1.0 m/s (Green Sturgeon). Depth where sturgeon eggs were collected ranged from 0.79-11 m (Lake Sturgeon), 1.4-7.9 m (Gulf Sturgeon), 3-16 m/s (White Sturgeon), and 3.7-9.2 m (Green Sturgeon). For substrate composition, only Dammerman et al. (2020) measured substrate size with an average of 11.74 mm (± 5.39 SD) and range of 3.58-33.83 mm [i.e., gravel (2-16 mm) and pebble (16-64 mm)]. In Dammerman et al. (2020) substrate size was determined by recording substrate (i.e., capturing a photo) within a rectangular frame along each transect line where eggs were collected.

Table 2.3: Summary of methodologies and deployment strategies for egg deposition sampling of Lake Sturgeon, Gulf Sturgeon, White Sturgeon, and Green Sturgeon.

Study	Year	Location (size of spawning habitat)	Size of Egg Mat/ Collector	Number of Mats Used	Duration of Sampling	Deployment Strategy
Lake Sturgeon						
Caswell et al. 2004	-	Zug Island	0.40x0.40 m = 0.16 m^2 One gang – 0.16*5 = 0.8 m^2	One gang of five mats.	Deployed on May 5 and retrieved after 3 days.	Based on telemetry data from April and May 2001 and water temperatures of 14-15°C.
Chiotti et al. 2008	2003	Tippy Dam rkm 1.4	Furnace filter secured around a 12.3 kg (0.40x0.20x0.10 m) cement cinder block.	Gang of five blocks – unknown how many per site, likely one.	May 5 to 15 – checked every 3 days. May 15 to June 5 – checked at least every 6 days.	Timing based on previous timing of adult arrival from telemetry data and visual observation.
	2004				April 26 to May 27 – retrieved every 6 days.	
	2004	Tippy Dam rkm 8.8				
Dammerman et al. 2020	-	Black Lake	Metal Rod – dislodge eggs. Kick Net – collect dislodged eggs.	Seven transect lines, placed 5 m apart, were sampled at 1 m intervals.	Beginning of May to mid-June.	Timing based on spawning observed in previous years.
Johnson et al. 2006	1995	Ogdensburg: Transect 1	One egg mat – 1 m^2	One gang of three mats per site (1*3 = 3 m^2).	Deployed on June 7 and retrieved June 9.	Unknown
	1995	Ogdensburg: Transect 2				
	1996	Ogdensburg: Transect 1*			All transects sampled June 17, 18, and 19. T2 and T3 also sampled June 21.	
	1996	Ogdensburg: Transect 2*				
	1996	Ogdensburg: Transect 3*				

Study	Year	Location (size of spawning habitat)	Size of Egg Mat/ Collector	Number of Mats Used	Duration of Sampling	Deployment Strategy
Neuenhoff et al. 2018	-	Bird Island Reef	Unknown (potentially 3.47/5 = 0.694 m^2) – egg mats cut from a 6.1x110 m roll of standard furnace filter to fit welded steel frames.	Five gangs of three mats. Weekly trap area noted to be 3.47 m^2.	May 10 to June 13 and retrieved once per week.	Unknown
Roseman et al. 2020	2018	St. Marys Rapids	One egg mat – 0.38x0.24 m = 0.0912 m^2	Gangs of three mats – unknown how many per site.	June 28 to July 26 – retrieved once per week.	Based on water temperatures.
	2019				June 26 to July 16 – retrieved once per week.	
Roseman et al. 2011	2009	Upstream of Fighting Island (3,300 m^2)	Unknown	Twelve gangs of three mats.	Eggs collected on May 5, 12, and 18.	Unknown
	2010				Eggs collected on May 12 and 18.	
Thiem et al. 2013	-	St. Ours Dam	Unknown	Sixty-eight single mats deployed in a 17x4 grid.	May 12 to June 13 and checked every 2-6 days.	Unknown
2019 DFO Sampling	2019	Musquash A	One egg mat – 0.40x0.19 m = 0.076 m^2 One gang of four – 0.076*4 = 0.304 m^2	Five gangs of four mats (0.304*5 = 1.52 m^2).	May 8 to June 13 – retrieved every second day.	Sampling from mid-May to mid-June, depending on temperature. Egg sampling starts prior to when temperatures reach 13°C.
		Musquash B		One to two gangs of four mats (0.304 m^2 or 0.304*2 = 0.608 m^2).	May 24 to June 13 – retrieved every second day.	
		Moon A		Five gangs of four mats (0.304*5 = 1.52 m^2).	May 23 to June 16 – retrieved every second day.	
		Goulais			May 17 to June 10 – retrieved every second day.	

Study	Year	Location (size of spawning habitat)	Size of Egg Mat/ Collector	Number of Mats Used	Duration of Sampling	Deployment Strategy
Gulf Sturgeon						
Flowers et al. 2009	2005	rkm 170.6	One egg collector – 0.508 m diameter circular floor buffer pad (0.203 m^2).	Between 2-67 egg collectors deployed per site. Exact numbers unknown.	Unknown, collectors checked every 48-72 hours.	Sampling began based on timing of entry of telemetry-tagged adults and ended when temperatures reached 25°C.
	2006					
	2006	rkm 160.1				
	2008	rkm 170.6				
	2008	rkm 160.1				
	2008	rkm 161.4				
Fox et al. 2000	1997	Choctawhatchee/ Pea	One egg collector – 0.559 or 0.686 m diameter circular floor buffer pad (0.245 or 0.370 m^2).	Between 15-50 egg collectors deployed per site. Exact numbers unknown.	April 4 to May 13 at six sites – retrieved every 24-72 hours.	Based on telemetry data.
Sulak and Clugston 1998	1995	rkm 215	One egg collector – 0.559 m diameter circular floor buffer pad (0.245 m^2).	80 collectors deployed in 20 fixed transects.	March 28 to April 21 – retrieved every other day.	Unknown.
	1996			48 collectors and 16 two-layer collectors (size unknown).	February 29 to May 7 – sampled every third day but sampled every other day once eggs were collected.	
White Sturgeon						
Chapman and Jones 2010	-	Willamette Falls	One egg mat – 0.76x0.91 m = 0.692 m^2	10 mats were deployed (0.692*10 = 6.92 m^2).	Deployed on May 18 and retrieved on May 20.	Based on peak temper-atures in the nearby lower Columbia River.
Paragamian et al. 2001	1991	Bonners Ferry	Unknown	Arranged in gangs of three or five mats. Between 70-100 mats were deployed each year. Exact numbers unknown.	Exact sampling dates unknown – checked weekly.	Unknown.
	1993				Exact sampling dates unknown – checked every 2-13 days.	
	1994				Exact sampling dates unknown – checked daily.	
	1995					
	1996					
	1997					
	1998					

Study	Year	Location (size of spawning habitat)	Size of Egg Mat/ Collector	Number of Mats Used	Duration of Sampling	Deployment Strategy
Green Sturgeon						
Poytress et al. 2015	2008	rkm 424.5	One egg mat – 0.89x0.61 m = 0.543 m^2	Gangs of two or four per site (0.543*2 = 1.09 m^2 or 0.543*4 = 2.17 m^2). Exact numbers unknown.	2008: rkm 424.5 and 377 - April 22 to August 1. 2009: rkm 424.5 and 407.5 - March 30 to July 30 and rkm 377 - March 31 to July 31. 2010: rkm 424.5 and 407.5 - March 17 to July 23 and rkm 377 and 366.5 - March 23 to July 25. 2011: rkm 332.5 - April 12 to July 15 and rkm 426 - April 12 to July 18. 2012: rkm 424.5 - April 9 to May 27 and rkm 332.5 - April 6 to July 14 and rkm 426 - April 5 to July 10. Egg mats were retrieved every 72-96 hours from all sites.	Based on radiotelemetry data, knowledge of fisherman, and prior egg mat sampling.
	2009					
	2010					
	2012					
	2008	rkm 377				
	2009					
	2010					
	2009	rkm 407.5				
	2010					
	2010	rkm 366.5				
	2011	rkm 332.5				
	2012					
	2011	rkm 426				
	2012					

*Note: *Unknown if Transects 1 and 2 are the same locations in both years. Size of spawning habitat sampled in Johnson et al. (2006) is 1,296 m^2.*

Table 2.4: Summary of egg abundance/density and physical habitat metrics for Lake Sturgeon, Gulf Sturgeon, White Sturgeon, and Green Sturgeon.

Study	Year	Location (size of spawning habitat)	Natural or Artificial	Egg Abundance/ Density (# or eggs/mat)	Water Velocity (m/s)	Depth (m)	Substrate Composition
Lake Sturgeon							
Caswell et al. 2004	-	Zug Island	Natural	7.5 eggs/m^2	0.35	11	
Chiotti et al. 2008	2003	Tippy Dam rkm 1.4	Natural	2277	0.34 - 0.9	1.9-2.2	
	2004			1136	0.9	2.8	
	2004	Tippy Dam rkm 8.8		500	1.11	1.9	
Dammerman et al. 2020	-	Black Lake	Natural	541	1.1	0.79	Mean of 11.74 mm (± 5.39 SD) and range of 3.58 to 33.83 mm.
Johnson et al. 2006	1995	Ogdensburg: Transect 1*	Artificial	175.7 eggs/m^2	0.46		Made up of 76-102 mm crushed limestone (cobble).
	1995	Ogdensburg: Transect 2*		0 eggs/m^2	0.37		
	1996	Ogdensburg: Transect 1*		145 eggs/m^2	0.6		
	1996	Ogdensburg: Transect 2*		22.4 eggs/m^2	0.52		
	1996	Ogdensburg: Transect 3*		3.8 eggs/m^2	0.42		
Neuenhoff et al. 2018	-	Bird Island Reef	Natural	86		2.63	
Roseman et al. 2020	2018	St. Marys Rapids	Natural	11	0.39	4.13	
	2019			45		3.4	
Roseman et al. 2011	2009	Upstream of Fighting Island (3,300 m^2)	Artificial	102 eggs/m^2	≥0.8	5-8	Limestone (10-50cm), 5-10cm diameter limestone, rounded igneous rock (10-25cm), and a mix of all three substrates.
	2010			12 eggs/m^2	≥0.8	5-8	
Thiem et al. 2013	-	St. Ours Dam	Natural	136	0.52-1.27 (0.93±0.02)	4.24-7.78 (6.05 ± 0.14)	Fine and Coarse Gravel

Study	Year	Location (size of spawning habitat)	Natural or Artificial	Egg Abundance/ Density (# or eggs/mat)	Water Velocity (m/s)	Depth (m)	Substrate Composition
2019 DFO Sampling	2019	Musquash A	Artificial	0 (45)**		1.6	Primarily boulder substrates with some cobble and bedrock with areas of gravel/cobble nearshore.
		Musquash B	Natural	171.7 ± 168.7 (15)**		1.4	Bedrock substrates with large boulders.
		Moon A	Artificial	0 (45)**		1.5	Cobble and boulder substrates.
		Goulais	Natural	0.3 ± 0.3 (44)**		3.0	Primarily cobble and boulder substrates with gravel deposits in interstitial spaces.
Gulf Sturgeon							
Flowers et al. 2009	2005	rkm 170.6	Natural	21	0.93	3.5	
	2006			180	0.75	3.9	
	2006	rkm 160.1		11	0.81	3.4	
	2008	rkm 170.6		204	0.68	3.3	
	2008	rkm 160.1		36	0.7	2.5	
	2008	rkm 161.4		42	0.71	3.3	
Fox et al. 2000	1997	Choctawhatchee/ Pea	Natural	42		1.4-7.9	Substrates were limestone (21/24), with some noted to be limestone-gravel (2) and one noted as sand (1/24).
Sulak and Clugston 1998	1995	rkm 215	Natural	0.7 eggs/m^2		2-4	Substrate at 43/49 eggs locations was limestone bedrock overlain by 0-10 cm of a matrix of sand and gravel-cobble.
	1996			368	0.2-1.2	1.8-6.4	
White Sturgeon							
Chapman and Jones 2010	-	Willamette Falls	Natural	3.2 eggs/m^2		6-16	

Study	Year	Location (size of spawning habitat)	Natural or Artificial	Egg Abundance/ Density (# or eggs/mat)	Water Velocity (m/s)	Depth (m)	Substrate Composition
Paragamian et al. 2001	1991	Bonners Ferry	Natural	13	0.9		Gravel-cobble
	1993			2	0.83	3	
	1994			213	0.25	8.5	Sand with pockets of gravel.
	1995			162	0.19	11	
	1996			349	0.22	11.6	
	1997			75	0.67	13.3	
	1998			393	0.61	11.4	
Green Sturgeon							
Poytress et al. 2015	2008	rkm 424.5	Natural	12	0.7	4.2	
	2009			9	0.4	5.5	
	2010			93	0.7	6.8	
	2012			39	0.7	7.1	
	2008	rkm 377		29	1	5.2	
	2009			43	1	4.8	
	2010			9	1	3.9	
	2009	rkm 407.5		2	0.7	8.1	
	2010			1	1	3.7	
	2010	rkm 366.5		1	0.2	6.1	
	2011	rkm 332.5		1	1.7	7	
	2012			3	0.9	7.6	
	2011	rkm 426		6	1	9.2	
	2012			16	0.7	7.4	

*Note: *Size of spawning habitat sampled in Johnson et al. (2006) is 1,296 m^2. **Values are reported as average eggs per m^2 ± Standard Error (n).*

Table 2.5: Summary of densities during each sampling event in 1996 for the artificial spawning site near Ogdensburg, NY, adapted from Johnson et al. (2006).

Citation	Year	Transect	Egg Mat	17-Jun	18-Jun	19-Jun	21-Jun	Average Density per Egg Mat	Average Density per Transect
Johnson et al. 2006	1996	1	1	150	300	-	138	196	145
			2	75	175	-	250	166.7	
			3	85	-	75	57	72.3	
		2	4	0	13	8	2	5.8	22.4
			5	2	68	55	50	43.8	
			6	1	31	35	4	17.8	
		3	7	0	15	8	4	6.7	3.8
			8	0	0	5	0	1.3	
			9	0	8	5	0	3.3	

There were many differences in methodologies among the larval studies (Table 2.6), especially the number of nets used and duration of sampling. Certain studies did not include the number of nets deployed (Auer and Baker 2002; Roseman et al. 2011, 2020), while others had multiple nets spaced across the entire river width (D'Amours et al 2001). Further, all studies appeared to sample through the entire duration when larval Lake Sturgeon would be drifting from the site, with the exception of the 2019 sampling of Roseman et al. (2020) that only collected larvae on one day. When specified, the time that nets were deployed for each sampling event varied between 3-7 hours. Unspecified times were simply described as being set overnight, with no specific time given.

In this review, a summary of larval abundances and physical habitat parameters measured at each site that collected sturgeon larvae was completed (Table 2.7). All studies reported values as abundances (range of 1 to 1,350), except the 2019 DFO sampling program, where values could be reported as an average larvae per net per night value (range of 0 to 7.7) as raw data for each sampling event was available. While all studies that collected larval Lake Sturgeon used drift nets, the shape and size varied among and within studies. Within the four years of data from Auer and Baker (2002) there is a large range of larval Lake Sturgeon abundance with four collected in 1998 and 279 collected in 1997, following the same methodologies. The range of water velocities when larvae were collected in those years was similar with 0.31-0.67 m/s in 1998 and 0.21-0.66 m/s in 1997.

Table 2.6: Summary of methodologies and deployment strategies for larval drift sampling of Lake Sturgeon.

Study	Year	Location (size of spawning habitat)	Size of Drift Net	Number of Nets Used	Duration of Sampling	Deployment Strategy
Auer and Baker 2002	1993	Prickett Dam	Rectangular net – 0.58x0.81 m opening with 950 µm Nitex net (3 m long) and a detachable cod-end.	Exact numbers unknown. It was noted that in 1997 and 1998 2-3 D-frame nets were used exclusively and in 1999 2-3 D-frame nets were used in combination with 1-2 rectangular nets.	Unknown. Duration was given for sampling of all five rkm distances sampled (i.e., rkm 14, 26, 35, 45, and 61), whereas only the first location sampled was of interest in this study.	Drift sampling began 8-10 days after identified spawning dates and continued until no or few larvae were collected.
	1997		D-frame net – 0.76 m base and 0.54 m high with a 1600 µm nylon mesh bag (3.18 m long) and a detachable cod-end.			
	1998					
	1999		Used both rectangular and D-frame nets.			
Bouckaert et al. 2014	2010	North Channel of the lower St. Clair River	D-frame net – 0.76 m base and 0.54 m high with a 1600 µm nylon mesh bag (3.18 m long) and a detachable cod-end.	Two nets were placed both upstream and downstream of the reef.	June 8, 9, and 29 from 21:00-02:00 hrs on a half-hour or hourly basis.	Unknown.
	2011			Two nets were placed upstream and four nets downstream of the reef.	Three nights per week between June 13-July 13 and one night weekly July 13-30 from 20:00-06:00 hrs on a 2-hour cycle.	
	2012			6-7 nets were sampled.	One-two times per week between June 5 to July 2 from 20:00-06:00 hrs on a 2-hour cycle.	
	2012	Middle Channel Reef		Two nets were placed both upstream and downstream of the reef.		
	2012	Fighting Island Reef		Four nets were placed both upstream and downstream of the reef.	Twice weekly from May 15 to June 5 from 20:00-06:00 hrs on a 2-hour cycle.	

Study	Year	Location (size of spawning habitat)	Size of Drift Net	Number of Nets Used	Duration of Sampling	Deployment Strategy
D'Amours et al. 2001	1994	Des Prairies River powerhouse and spillway	Circular nets – 1 m in diameter and 4.2 m long with 800 µm mesh. Square nets – 0.5x0.5 m opening and 2.7 m long with 800 µm mesh.	Two drift nets were set, one 0.5 m from the surface (circular) and another (square) on the river bottom at 4 sites for a total of 8 nets. A fifth site and two more nets were sampled in 1995.	Nets were set between 21:00-04:00 hrs for 45-60 minutes from May 19 to June 19.	Nets were set for 24 hours
	1995				Nets were set between 21:00-24:00 hrs for 45-60 minutes from May 15 to June 22.	Nets were set 15-20 m from both the Montreal and Laval shores with two more nets 1/3 and 2/3 of the distance between the border nets.
Roseman et al. 2020	2018	St. Marys Rapids	D-frame net – 0.76 m base and 0.54 m high with a 1600 µm nylon mesh bag and a detachable cod-end.	Four sites were fished. Unknown how many drift nets at each site.	July 6-28 and fished from 22:00-04:00 hrs.	Sampling determined based on temperature-drift relationships described in Auer and Baker (2002) and Roseman et al. (2011).
	2019			Three sites were fished. Unknown how many drift nets at each site.	July 16 and fished from 22:00-04:00 hrs.	
Roseman et al. 2011	2009	Upstream of Fighting Island	D-frame net – 0.76 m base and 0.54 m high with a 1600 µm nylon mesh bag (3.18 m long) and a detachable cod-end.	Unknown.	May 19-21 and 26-27 and set from 21:00-02:00 hrs.	Sampled 10 days after egg deposition.
Welsh et al. 2015	2005	Kakabeka Falls	D-frame net – 0.76 m base and 0.53 m high with a 1000 µm nylon mesh bag (3.6 m long) and a detachable cod-end.	Twelve nets during each night of sampling (276 net sets).	Nets set from June 2 to 28 overnight (set at dusk and lifted the following day).	Based on movement of radio-tagged adults and optimal temperatures for larval hatch (130-150 cumulative temperature units).
	2006			Twelve nets during each night of sampling (216 net sets).	Nets set from June 1 to 20 overnight (set at dusk and lifted the following day).	

Study	Year	Location (size of spawning habitat)	Size of Drift Net	Number of Nets Used	Duration of Sampling	Deployment Strategy
2019 DFO Sampling	2019	Musquash A	D-frame net – 0.75 m base and 0.5 m high with a 1600 µm nylon mesh bag (3.18 m long) and a detachable cod-end.	2-5 nets set daily.	Set overnight from June 11-26 and retrieved daily.	Sampling conducted 5-10 days after spawning or prior to reaching 150 cumulative temperature units. Sampling ends when larvae have not been observed for several days or at 400 cumulative temperature units.
		Musquash B		3 nets set daily.		
		Moon A		4-5 nets set daily.	Set overnight from June 12-26 and retrieved daily.	
		Moon B		One net set daily	Set overnight from June 12-19 and retrieved daily.	
		Goulais		3-4 nets set daily.	Set overnight from June 11-28 and retrieved daily.	

Table 2.7: Summary of larval abundance/density and physical habitat metrics for Lake Sturgeon.

Study	Year	Location (size of spawning habitat)	Natural or Artificial	Larval Abundance (#)	Water Velocity (m/s)	Depth (m)	Substrate Composition
Auer and Baker 2002	1993	Prickett Dam	Natural	29	0.46-0.73		
	1997			279	0.21-0.66		
	1998			4	0.31-0.67		
	1999			5	0.40-0.66		
Bouckaert et al. 2014	2010	North Channel of the lower St. Clair River	Artificial	11	0.34-0.38	9.4-13.2	Coal clinker (0.25-12cm)
	2011			51	0.21-0.34	9.4-11.3	
	2012			81	0.21-0.41	9.4-13.8	
	2012	Middle Channel Reef		35	0.21-0.41	9.4-13.8	Angular limestone (10.2-20.3cm), rounded igneous rock (10.2-15.3cm), and 1:1 mix of both substrates.
	2012	Fighting Island Reef		34	0.55-0.70	6.5-9.8	Limestone shot rock (10-50cm), sorted limestone (5-10cm), rounded igneous rock (10-25cm), and a mix of all three substrates.
D'Amours et al. 2001	1994	Des Prairies River powerhouse and spillway	Natural	1350	0.74	3.9	
	1995			607	0.86	5.29	
Roseman et al. 2020	2018	St. Marys Rapids	Natural	21	0.33	6.27	
	2019			1		7.1	
Roseman et al. 2011	2009	Upstream of Fighting Island	Artificial	7	≥0.8	5-8	Limestone (10-50cm), 5-10cm diameter limestone, rounded igneous rock (10-25cm), and a mix of all three substrates.
Welsh et al. 2015	2005	Kakabeka Falls	Natural	479	0.32-0.66	0.17-0.74	
	2006			279	0.15-0.67	0.27-0.60	

Study	Year	Location (size of spawning habitat)	Natural or Artificial	Larval Abundance (#)	Water Velocity (m/s)	Depth (m)	Substrate Composition
2019 DFO Sampling	2019	Musquash A	Artificial	1.1 ± 0.6 (32)*		2.7	Primarily boulder substrates with some cobble and bedrock with areas of gravel/cobble nearshore.
		Musquash B	Natural	0.4 ± 0.3 (27)*		3.7	Bedrock substrates with large boulders.
		Moon A	Artificial	7.7 ± 1.7 (59)*		2.4	Cobble (50-250 mm diameter), cobble/boulder (100-400 mm diameter) and boulder (three size classes of >250mm, >500mm, and >800mm diameter) substrates.
		Moon B	Natural	1.3 ± 0.8 (6) *		1.6	
		Goulais	Natural	0 (50)*		0.8	Primarily cobble and boulder substrates with gravel deposits in interstitial spaces.

*Note: *Values are reported as average larvae per net per night ± Standard Error (n).*

2.5 Discussion

Under the federal *Fisheries Act* in Canada, if anthropogenic development impacts fish and fish habitat, offsetting measures (i.e., actions taken to offset adverse effects) and subsequent monitoring actions can be mandated. Effectiveness monitoring aims to evaluate if offsetting measures result in measurable benefits for fish and fish habitat (DFO 2012). Regulators are trying to streamline this challenging process as impacts to fish and fish habitat are sometimes unavoidable (i.e., installation and operation of hydroelectric generating stations) and full effectiveness monitoring programs are time consuming and costly. Functional monitoring has been identified as a possible solution as assessment of key habitat parameters (i.e., depth, water velocity, and substrate composition), which should be able to support all necessary life stages, may be sufficient to determine if offsetting measures are functioning as intended if a consistent and well-established connection can be made.

2.5.1 Challenges in sturgeon biology

There are a number of important factors about sturgeon biology that may explain the interannual variability observed in sturgeon spawning studies. Sturgeon exhibit life history traits such as spawning periodicity and late maturation (i.e., up to 30 years for Lake Sturgeon; Haxton et al. 2016). Thus, population growth potential for sturgeon is lower than early maturing and annually spawning species (Vélez-Espino and Koops 2009). Sturgeon population sizes can also vary widely as they are recovering species with generally low abundances. Timing of sampling is also critical as water temperature and declining discharge are drivers of Lake Sturgeon spawning (Forsythe et al. 2012).

Despite following similar methods of data collection and having similar water velocities and depths reported in each year, differences in egg densities and larval abundances were observed within the same sites, which may be due to spawning periodicity. Further, in years with similar numbers of adult spawners, this variability may be due to varied mortality rates during early life stages. Donofrio et al. (2018) estimated spawning periodicity for Lake Sturgeon based on acoustic telemetry detections at known spawning sites during the spawning season across four rivers (i.e., Menominee and Fox rivers 2015-2016; Oconto River 2014-2016; and Peshtigo River 2013-2016). Spawning periodicity was estimated to be between 1.9-3.4 years (Donofrio et al. 2018). While spawning periodicity was investigated for a maximum of four years in Donofrio et al. (2018), advances in acoustic telemetry technology including increased battery life of acoustic transmitters will allow for increased tracking of individual fish over multiple years, as sturgeons are long-lived. Examples in this review that show variances in abundances and densities across multiple years of sampling at the same site are shown in Auer and Baker (2002) and Roseman et al. (2011). Auer and Baker (2002) collected 279 larval Lake Sturgeon in 1997 and 4 larvae in 1998 at a natural spawning site downstream of Prickett Dam in the Sturgeon River, Michigan. Roseman et al. (2011) collected 102 eggs/m^2 in 2009 and 12 eggs/m^2 in 2010 in the Detroit River, upstream of Fighting Island. Although spawning periodicity may explain year-to-year variation in densities and abundances for eggs and larvae, in systems with similar numbers of spawners in each year, differences in abundances may be due to varied rates of mortality during early life stages. Duong et al. (2013) estimated the annual effective adult breeding number (N_b) by testing genotypes for Lake Sturgeon adults (n=796) captured during ten spawning seasons and offspring (i.e., larvae; n=3925) collected during dispersal over 8 years. Interannual variation in N_b/N ratios, where N is the adult census size, ranged from 0.27-0.86. Variance in reproductive success was noted to be low due to high proportions of breeding adults and the polygamous mating characteristics of Lake Sturgeon. To determine rates of polygamy, annual means of mates were determined (i.e., 1.76-6.76 mates for males and 3.09-15.08 mates for females), where sex ratios were male-biased in all years (i.e., 1.61-3.01 males to females). Despite similar numbers of spawning adults between years, larval production varied 40-fold across years (i.e., 437-16,417 larvae), suggesting varied rates of mortality during the egg and larval life stages (Duong et al. 2013). Thus, variation in egg and larval densities in populations that have similar interannual numbers of spawning adults may be attributed to varied rates of mortality during early life stages.

A major deficiency of developing a functional monitoring program is lack of data on adult spawners or vast differences in spawning populations among sites. Sturgeon are recovering species with generally low abundances; and using Lake Sturgeon in designatable unit 4 (DU4; Great Lakes Upper St. Lawrence) as an example, there are 21 populations that have very low (i.e., >10 and >50) estimates of mature individuals (COSEWIC 2017). Roseman et al. (2011) sampled a population (i.e., Detroit River) with approximately 4,070 mature individuals and Auer and Baker (2002) sampled the

Sturgeon River, where a mark-recapture study was completed and estimated a total adult population of 316 (300-352, 95% CI; Commanda 2018). Similarly, Johnson et al. (2006) sampled a population near Ogdensburg, New York, where the number of mature individuals upstream of the Beauharnois Dam (i.e., Lake St. Francis) in the St. Lawrence River is estimated to be >50 (COSEWIC 2017). Transect sampling conducting in Johnson et al. (2006) also showed variation in egg densities across multiple transects in both years studied.

Egg deposition within a site is not uniform (Johnson et al. 2006; Thiem et al. 2013); and similarly, larval drift is not uniformly distributed across the river channel (Smith and King 2005). Thus, it is important to ensure that sampling effort is adequate to capture accurate rates of egg deposition and larval drift at sturgeon spawning habitats.

2.5.2 Sturgeon spawning habitat requirements

In order for physical parameters (i.e., water velocity, depth, and substrate composition) to be used as a proxy for biological metrics (i.e., egg deposition and larval drift data collection) a well-established and consistent connection must be made. The ranges for habitat parameters for sturgeon are too broad to be used for functional monitoring; however, an approach such as determining peak suitability (i.e., as Baril et al. (2018) did for Lake Sturgeon), could be used to support functional monitoring.

Haxton et al. (2016) summarized habitat requirements for North American sturgeon (Table 2.1) using data from species specific status reports. All studies synthesized in this review were included in the species specific status reports used by

Haxton et al. (2016), with the exception of the unpublished 2019 DFO Sampling and Thiem et al. (2013). Three additional studies were published after the species specific status reported and thus were also not included (i.e., Neuenhoff et al. 2018; Roseman et al. 2020; Dammerman et al. 2020). While general ranges of spawning habitat requirements were presented in Haxton et al. (2016), Baril et al. (2018) determined that peak suitability for Lake Sturgeon spawning habitat occurred at average water velocities of 0.6 m/s, depths of 0.55-0.85 m in small rivers and 0.75-5.25 m in large rivers, with cobble (64-256 mm) substrates. Although Lake Sturgeon spawning can occur within large ranges, it has been shown that egg density increases as water velocity increases from 0.4 to 0.6 m/s (Johnson et al. 2006) and egg density decreases as water velocity increases from 0.6 to 1.1 m/s (LaHaye et al. 1992), which further supports 0.6 m/s as the peak suitable water velocity for Lake Sturgeon spawning. For the full range of water velocity, Lake Sturgeon can spawn in nearly stand-still water (0.03 m/s) to high flows of 2.51 m/s (Baril et al. 2018). Similar to methods in Baril et al. (2018), a midpoint could be used for ranges to make comparisons between studies with different outcome types. Depths and substrate also have large ranges from 0.03 to 25 m and substrate sizes of 0.01 to 256.1 mm or sand/fine materials to boulders, whereas the peak suitability range is much smaller (Baril et al. 2018). Lake Sturgeon data summarized within this review had water velocity ranges of 0.34-1.11 m/s, depths of 0.79-11 m, and average substrate size was given in one study (Dammerman et al. 2020) as 11.74 mm.

2.5.3 Limitations of the current literature base

As expected physical habitat data collected in this review are within the reported ranges in the literature for spawning habitats, with some exceptions. However, the evidence base is limited for determining broad relationships between productivity outcomes and physical habitat characteristics. Also, there were few studies conducted on Gulf Sturgeon (3), White Sturgeon (2), and Green Sturgeon (1) and thus, comparisons within each of those species are not possible.

This study is unique in its summary of biological outcomes of productivity (i.e., eggs and/or larvae) and of physical habitat characteristics (i.e., water velocity and/or depth and/or substrate composition) associated with spawning habitats for four sturgeon species. While the main objective of this review was to scope the feasibility of the use of functional monitoring for sturgeon spawning habitats, it has highlighted gaps in the literature. Despite the purposely broad search terms used in the database searches, only 18 studies were deemed to have reported at least one productivity outcome and at least one physical habitat outcome. This review has shown that standardized methodologies and full reporting of data is necessary to enable future meta-analysis. However, for sturgeon species, functional monitoring may not be possible as life history characteristics (i.e., spawning periodicity) and differences in population sizes could result in large variances in spawning abundances and densities among years.

Using English-language search terms, and excluding studies that did not report egg and larval abundances separately, gave modelled outcomes, or used captive-bred individuals, limited the scope of the review. This review had a search strategy designed to capture studies with sturgeon spawning assessments of two life stages (i.e., eggs and larvae) that also collected physical habitat metrics (i.e., water velocity and/or depth and/or substrate composition). This study did not include modelled outcomes or any outcomes that resulted from captive-bred individuals. Also, only articles in the English language were included. For example, studies published in Chinese, Russian, and French, which would consist of sturgeon species with ranges in China, Russia, and Québec, Canada would not have been captured by the English search terms used. Another limiting factor is different conventions for reporting biological productivity outcomes. For example, two English language studies on Chinese Sturgeon (i.e., Wei et al. 2009; Zhang et al. 2011) reviewed at the full-text stage measured eggs and larvae together as one value [i.e., early life stages (ELS)]. This methodology varied too greatly from how outcomes were reported for other sturgeon species and as such, these studies (2) were excluded at the full-text stage. To broaden results of future reviews, search terms should be selected to capture differences in sampling conventions across the large geographic distribution of sturgeon species.

Outcomes reported in the studies reviewed varied for both biological productivity metrics (i.e., eggs and larvae) and habitat characteristics (i.e., water velocity, depth, and substrate composition). In order for studies to be more easily included in meta-analyses, standardized methodologies are recommended (Braun et al. 2019). Using substrate composition as an example, there are various methods that can achieve the same objective (i.e., determine mean substrate size through various methodologies including sieved substrate samples, visual assessment of dominant substrate size (%), etc.). However, comparisons between methods are difficult as there are varied assumptions and biases (Braun et al. 2019). In this review, differences in reported substrate types identified the need for more standardized and transferable methods. As an example, substrate composition for Gulf Sturgeon was given as general rock types of limestone, limestone-gravel, sand (<2 mm), and gravel-cobble (64-256 mm; Fox et al. 2000; Sulak and Clugston 1998), rather than an average substrate size across the site, which was only reported in Dammerman et al. (2020) for a natural Lake Sturgeon spawning site. The use of side-scan sonar to collect habitat data has been identified as a low-cost and relatively quick option (Kaeser and Litts 2010; Walker and Alford 2016). As specific substrate compositions are frequently cited as requirements for successful spawning habits, studies could use side-scan sonar methodologies to map entire spawning sites and determine mean substrate sizes in a quick and effective way.

Assessing whether artificially created habitats are effective offsetting measures for sturgeon species is important as they are often built to offset destroyed or degraded fish habitats and fish productivity losses below hydroelectric generating stations. Although it appears that artificial spawning habitats have not achieved the same high densities and abundances that some natural spawning sites have, there are numerous confounding factors that prevent directly comparing habitat productivity in this way. An interesting example are the spawning sites sampled on the Musquash River, Ontario. Zero eggs were collected from the artificial spawning site (i.e., Musquash A), despite large sampling effort (i.e., n = 45 egg mat strings) whereas higher densities of deposition (i.e., 171.7 ± 168.7 SE (n=15) average eggs per m^2) were collected from the nearby upstream natural spawning site (i.e., Musquash B). The habitat characteristics were similar at both sites with average depths of 1.6 m at the artificial Musquash A site and 1.4 m at the natural Musquash B site. Substrate composition was also similar with boulder and cobble over bedrock at Musquash A and bedrock and boulders at Musquash B (2019 DFO Sampling). The Musquash A artificial site was initially constructed as low water levels in Georgian Bay impeded Walleye from further upstream migration, with the added benefit of establishing good quality spawning habitat for Lake Sturgeon (Mason 2012). Despite matching of physical habitat conditions at the Musquash A and B sites, there were large differences in the productivity outcomes collected (i.e., no eggs collected at the Musquash A artificial spawning habitat). Further showing that the connection between physical habitat metrics and biological productivity outcomes cannot be made to support functional monitoring.

2.5.4 Further research and implications for policy and management

For a more complete meta-analysis to be conducted on sturgeon spawning habitats and biological productivity in the future, data from grey literature sources should be obtained. In this narrative review, only peer reviewed journal articles available through the Web of Science and Scopus databases were included. As there is likely publication bias (i.e., successful outcomes are more likely to be published), it is likely that evidence of unsuccessful spawning habitats is limited in the literature. Sturgeon spawning and habitat assessments are completed by various groups, including government (both federal and state/provincial), non-governmental organizations, Indigenous groups, industry (i.e., hydroelectric companies), and academic institutions. As such, there are likely grey literature sources with the outcomes presented in this review that have not been published as they may have non-significant results.

Establishing functional monitoring may be better suited to fishes that have consistent annual spawning and exhibit early maturation (Vélez-Espino and Koops 2009). For example, Walleye are known to spawn over cobble substrates at high water velocities (Bozek et al. 2011) and while skipped spawning [i.e., not spawning in a given year due to various potential factors (metabolic stress, deficient diet, poor nutritional condition etc.); Rideout and Tomkiewicz 2011] of female Walleye has been observed (Henderson et al. 1996), it is uncommon. Artificial spawning substrates for Walleye have been installed in various riverine environments, including the St. Clair-Detroit River system (Fischer et al. 2018). If enough data with similar methodologies or raw data to transform outcomes for comparison is sourced, this could aid in the possibility of conducting a meta-analysis to see if there is a consistent link between biological metrics of productivity and physical habitat characteristics for Walleye.

McAdam et al. (2017) explored the question “if you build it, will they come” with respect to created sturgeon spawning habitat. While it was noted that projects focussing on artificial spawning habitats for Lake Sturgeon showed promise, the need for further research to identify remediation measures for consistent long-term effectiveness was identified. At this point, effectiveness monitoring is necessary for any offsetting projects for sturgeon spawning habitat; however, a similar review on other species (i.e., Walleye or salmonids) could yield more promising results for the use of functional monitoring. Also, the ranges for habitat parameters for sturgeon are broad and an approach such as determining peak suitability (i.e., as Baril et al. (2018) did for Lake Sturgeon), could be used to support functional monitoring. As ranges are broad, artificially created spawning habitats could easily fit into habitat requirement ranges, which does not necessarily guarantee increases in productivity (i.e., the objective of many offsetting programs). This review points to the complexity of developing scientifically defensible functional monitoring programs, which should motivate further investment in targeted research as full effectiveness monitoring programs are time-consuming and costly, and a functional monitoring approach, if determined to be feasible for other species, could be of great benefit. As one of the main limitations in this review was variable methodologies, standardized protocols for productivity and habitat data collection are recommended.

As noted by Braun et al. (2019), standardized monitoring methods, where metrics are measured, recorded, analyzed, and reported consistently, allows for easier meta-analysis, which can be used to determine the overall effectiveness of management measures (i.e., offsetting, mitigation, or restoration). To see if physical habitat characteristics (i.e., water velocity, depth, and substrate composition) can be used as a proxy for biological productivity metrics (i.e., egg deposition and larval drift), a well-established and consistent connection must be made to support functional monitoring. This study shows that varied methodologies make comparisons between studies difficult; as such, suggestions for standardized methods for Lake Sturgeon egg deposition, larval drift, and collection of water velocity, depth, and substrate composition data are given below as an example. If standardized methodologies are followed for all spawning studies, results will have comparable outcomes that would allow for easier meta-analysis and may allow for functional monitoring to become a reality. When assessing the efficacy of artificial spawning habitats for Lake Sturgeon studies would ideally follow a Before-After-Control Impact (BACI) design or another appropriate comparator (i.e., Control-Impact design matching the artificial site to a natural spawning shoal as a control; DFO 2012; Smokorowski et al. 2015).

Lake Sturgeon spawning usually occurs in late spring to early summer, depending on location, in the temperature range of 9-18°C (Bruch et al. 2016). Thus, it is suggested that egg mats be deployed prior to river temperatures of 9°C. Egg mats made with a material conducive to egg deposition (i.e., furnace filter) wrapped around a heavy material (i.e., steel plates or cinderblocks) have been used for Lake Sturgeon egg collection in many systems (e.g., Caswell et al. 2004; Bouckaert et al. 2014). To easily compare different studies, the surface area of the egg mat, number of egg mats set, and duration the mats were set for should be recorded. Also, egg deposition is not uniform (Johnson et al. 2006; Thiem et al. 2013), as such it is important to determine that the deployed egg mats give adequate coverage of the spawning habitat. Determining substrate composition using side-scan sonar also allows for relatively quick and cost-effective mapping of spawning habitats that can be used to determine overall spawning habitat size.

Larval Lake Sturgeon will emerge and drift downstream at night, with peak drifting times between 21:00 and 02:00 hours (Bruch et al. 2016), at a minimum water temperature of 16°C (Peterson et al. 2007). However, the metabolic rate of the larvae has been shown to increase with higher temperatures, decreasing incubation and yolk sac absorption times (Duong et al. 2011). Thus, large ranges of larval emergence times have been observed: 3-5 days (Auer and Baker 2002), 6-15 days (Duong et al. 2011), and 11-19 days (Bruch et al. 2016). Friday (2014) outlines the use of cumulative daily water temperature units (CTU) to estimate when larval drift will occur. CTU is calculated using the mean daily water temperature (°C) for a day, minus the constant of 5.8°C for all days from spawning to the end of drift. Smith and King (2005) noted that the mean CTU value from spawning to the start of larval drift in the Upper Black River, Michigan was 151.2. The date of first spawning is arbitrarily set to when the water temperature reaches 13°C, larval drift starts at 150 CTU, and normally ends at 400 CTU (Friday 2014). It is suggested that larval drift nets are deployed prior to reaching 150 CTU or promptly after spawning if temperatures are high to collect early drifting larvae as noted in (Auer and Baker 2002). Larval drift is not uniformly distributed across the river channel (Smith and King 2005). Thus, it is important to ensure that sampling effort is adequate to capture accurate rates of egg deposition and larval drift at sturgeon spawning habitats. To report a clear, comparable outcome and to assess whether sampling efforts were adequate, area (i.e., size of net compared to river size), volume of water fished, and time sampled (i.e., timing of each net deployed and total duration of sampling during the larval drift period) should be included. Also, if different drift nets are used differences in the volume of water fished would also need to be determined for each net type as larger abundances in nets could be attributed to larger volumes of water fished. It is suggested that D-frame drift nets deployed along the substrate and set at night be used for collecting larval Lake Sturgeon. Timing of drift net deployment should follow confirmed egg deposition or prior to reaching 150 CTU.

Suggested methods for sampling that can be standardized include measuring water velocity using a flow meter and wading rod (for shallow water) or sounding weight (for deeper water), and a side-scan sonar unit to measure depth and substrate composition. It is suggested that water velocity be measured roughly at the midpoint of the column. Due to the inability to keep a boat perfectly stationary in high velocity water on a large river, two to three separate measurements of water velocity should be taken at each egg mat for 10 seconds and averaged to determine the mean water velocity. Depth data can be recorded using a side-scan sonar unit, where geographic positioning and river depth data can be set to record at a specific interval (i.e., 3 seconds). In boatable water, substrate characteristics can be recorded using side-scan sonar, where the side imaging system uses a boat-mounted transducer to collect data on spawning habitat (Kaeser and Litts 2010). Walker and Alford (2016) used this methodology as a low-cost and relatively quick option for collecting substrate data for Lake Sturgeon spawning habitat. For best results it is suggested that side imaging is recorded using strait line navigation with speeds of 3.2-9.7 km/h (2-6 m/h). Side beam range for the side imaging is set to a given distance per side (i.e., approximately three times the depth of the river on either side) and the global positioning system (GPS) antenna can be installed with the transducer at the front of the boat to ensure positional accuracy. Overlapping sonar images and coordinate data are recorded into the side imaging system while boating downstream. It is suggested that a standardized analysis be conducted using either IrfanView graphic viewer, ArcView, ArcGIS or a python-based program (PyHum) to create maps of the substrate and categorize areas based on substrate size. When river depth is shallow and does not allow for a side-scan sonar unit to be used, a GoPro can be used to capture video of the river substrate. Analysis of videos can compare the substrate size to a known measurement (i.e., the size of an egg mat) in the video to determine relative substrate size. ImageJ can be used to measure substrate size in the GoPro videos. It is suggested that substrate classification be based on the Wentworth Classification System, modified by Cummins and categorized into diameter class descriptions used in Chiotti et al. (2008) as follows: sand/fine material (<2mm), gravel (2-16mm), pebble (16-64mm), cobble (64-256mm), and boulder (>256mm; Wentworth 1922; Cummings 1962).

This narrative review is the first step towards determining if there is a connection between biological metrics of productivity (i.e., eggs and/or larvae) and physical habitat characteristics (i.e., water velocity and/or depth and/or substrate composition) that is well-established and defensible. There are budgetary and timing advantages to conducting functional monitoring over effectiveness monitoring, as effectiveness monitoring requires significant resources (DFO 2012). If the connection between physical habitat characteristics and biological metrics of productivity can be made, it could impact management options for spawning habitats offsets.

Chapter 3: Movement of Lake Sturgeon (*Acipenser fulvescens*) and Walleye (*Sander vitreus*) in the Lower Black Sturgeon River, Lake Superior

3.1 Introduction

The Great Lakes experience anthropogenic stressors and numerous projects within this system have been developed to rehabilitate fish habitat with the goal of facilitating population recovery. Some Great Lakes Walleye populations have declined from historical levels due to overexploitation, alteration of spawning habitats, and pollution (Schneider and Leach 1977). In Lake Erie, management measures have resulted in Walleye populations becoming one of the largest and most self-sustaining in North America (Matley et al. 2020); however, other populations, including Black Bay Walleye in Lake Superior, have not seen the same levels of recovery (Bobrowicz 2010). Walleye have three different life-history strategies: river resident-river spawner, lake resident-lake spawner, and lake resident-river spawner (Bozek et al. 2011). Black Bay Walleye were initially thought to be a lake resident-lake spawning population; however, an extensive sampling program of the substrate in Black Bay mainly found clay, silt, and mud substrates that are not conducive to Walleye spawning (Biberhofer and Prokopec 2007). Further, a radio telemetry study conducted during spring spawning observed Walleye gathering in the Black Sturgeon River (Furlong et al. 2006). As such, Black Bay Walleye may be a lake resident-river spawning population that utilizes the Black Sturgeon River for spawning habitat.

Another key species that utilizes Black Bay and the Black Sturgeon river are Lake Sturgeon. Lake Sturgeon populations in the Great Lakes declined in the late 1800's due to overfishing and eventually collapsed around 1910 (Harkness and Dymond 1961). A primary factor affecting their recovery is the installation of dams as water control structures and hydroelectric generating stations (Haxton and Findlay 2008; Kerr et al. 2010), as they block access to and alter spawning habitat (Haxton et al. 2014a). Lake Sturgeon are listed under the Ontario *ESA* as Threatened and are listed federally under *SARA* as Special Concern for northern populations (COSEWIC 2017). Black Bay is a critical management area for Lake Sturgeon, and the Black Sturgeon River is listed in the Lake Superior Technical Committee Rehabilitation Plan as one of nine tributaries targeted for rehabilitation (Auer 2003).

3.1.1 History of the Camp 43 Dam

In 1937, log drives commenced on the Black Sturgeon River, the largest tributary to Black Bay, located in northwestern Lake Superior (Figure 3.1). In 1959/1960, the Great Lakes Paper Company built the Camp 43 Dam to facilitate this process (Furlong et al. 2006; Horns et al. 2003). The Camp 43 Dam, also known as the Twin Rapids Dam or Black Sturgeon Dam, was constructed 17 km upstream from the mouth of the Black Sturgeon River (Bobrowicz 2010). Log drives were discontinued in 1965 and ownership of the Camp 43 Dam subsequently passed to the Ontario Provincial Government (Horns et al. 2003). The original dam measures approximately 50 m across the width of the river, and in 1968 a 50 m concrete overflow weir was added to the east side of the dam (Bobrowicz 2010). The Camp 43 Dam is currently operated by the Ontario Ministry of the Environment, Conservation and Parks (OMECP) and Ontario Parks, after responsibility was transitioned from the Ontario Ministry of Natural Resources and Forestry (OMNRF) in 2018. After modification to restrict access to the upper reaches in 1966, it now primarily serves as a barrier to invasive species, primarily Sea Lamprey (*Petromyzon marinus*), but also naturalized Pacific salmonids [i.e., Rainbow Trout (*Onchorhynchus mykiss*), Chinook Salmon (*Onchorhynchus tshawytscha*), Coho Salmon (*Onchorhynchus kisutch*), and Pink Salmon (*Onchorhynchus gorbuscha*)], Rainbow Smelt (*Osmerus mordax*), and Common Carp (*Cyprinus carpio*; Horns et al. 2003; Bobrowicz 2010). Two other dams previously existed on the waterway: the Camp 1 Dam at the outlet of Eskwanonwatin Lake (i.e., approximately 67 km from the mouth of the Black Sturgeon River) and the Split Rapids Dam at the outflow of Black Sturgeon Lake (i.e., approximately 91 km from the mouth of the Black Sturgeon River). In 1999 the Camp 1 Dam was destroyed in a forest fire, and in 2001 the Split Rapids Dam was removed by the OMNRF. There are no additional barriers for Sea Lamprey control purposes or to impede access for native fish movement upstream of the Camp 43 Dam.

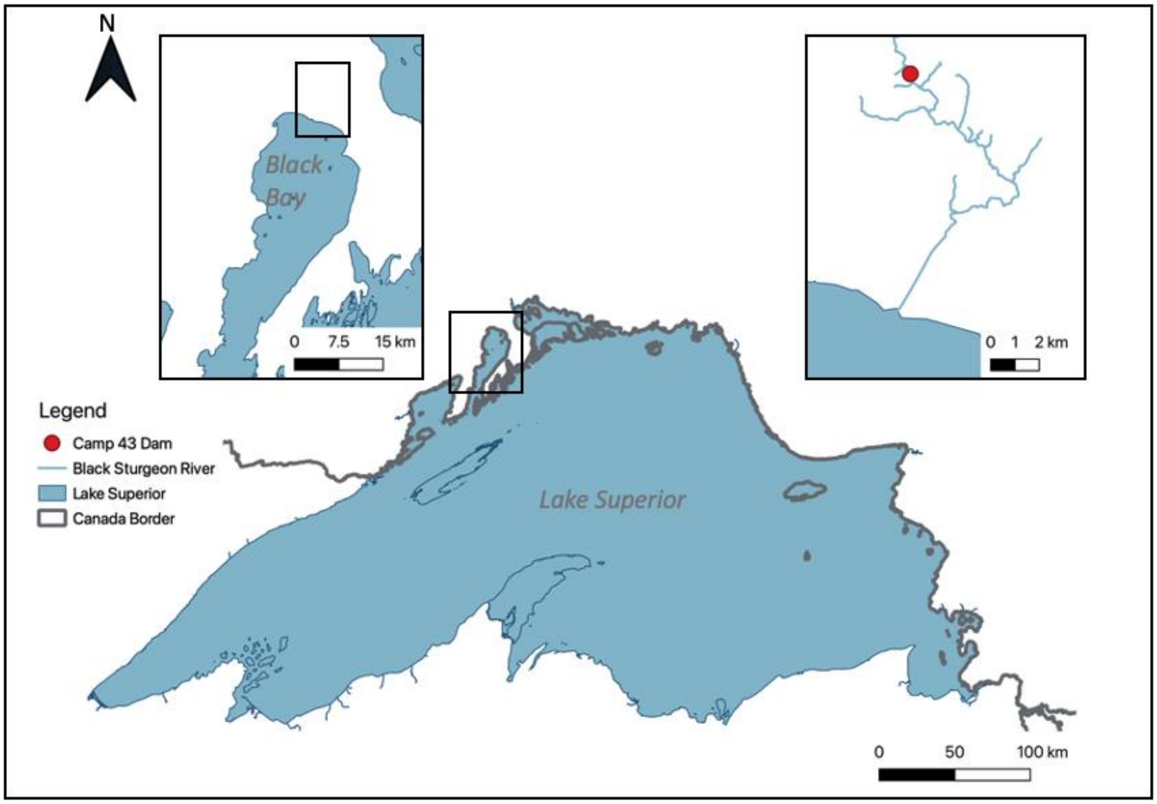

Figure 3.1: Map of Lake Superior with inset map (left) of Black Bay relative to Lake Superior and the right inset map shows the Black Sturgeon River, with the Camp 43 Dam indicated by a red circle. The portion of the Black Sturgeon River downstream of the Camp 43 Dam is referred to as the lower Black Sturgeon River, with the portion upstream referred to as the upper Black Sturgeon River. GIS data for each feature was provided by Statistics Canada (Canada Land Border), NOAA (Lake Superior), and Ontario Hydro Network (Black Sturgeon River and Camp 43 Dam). Map created in QGIS.

3.1.2 The collapse of the Black Bay commercial Walleye fishery

From the late 1800s to the eventual collapse of the Walleye fishery in 1968, Black Bay was home to the largest population of Walleye in Lake Superior and supported a commercial and recreational fishery (Bobrowicz 2010; McLaughlin et al. 2013). Records from 1966, the peak of the commercial Walleye harvest, estimated the pre-collapse adult Walleye population to be between 340,000 and 680,000 individuals; although by this time the population would have been stressed (Furlong et al. 2006; Bobrowicz 2010). In 1966 over 150,000 kg of Walleye were harvested (Furlong et al. 2006), making up approximately 90 percent of the Walleye commercial harvest in Lake

Superior (Schram et al. 1991). Using bathymetric data, 50,000 ha of the bay (approximately 83%) are estimated as potential habitat for adult Walleye (Bobrowicz 2010). An annual Walleye harvest over 1 kg/ha is considered unsustainable; thus, making the maximum yield for Black Bay 50,000 kg, one-third the size of the 1966 harvest (Colby and Baccante 1996). In 1968 the Walleye population in Black Bay collapsed due to several potentially causative factors, including overharvesting, habitat degradation and disruption from logging drives, high non-native Rainbow Smelt abundance and subsequent predation on YOY Walleye, as well as impeded access to historical upstream spawning sites after the construction of the Camp 43 Dam (Bobrowicz 2010). The commercial Walleye fishery in Black Bay was closed in 1971, and despite many rehabilitation efforts, the population has not fully recovered (Garner et al. 2013). However, the OMNRF Fall Walleye Index Netting (FWIN) program has shown consistent recruitment (i.e., more young fish being captured), and a 10-fold increase in relative abundance from 2002 to 2017 (E. Berglund (OMNRF) 2021, unpublished data).

3.1.3 Walleye population rehabilitation efforts

A rehabilitation workshop for Black Bay Walleye was held in 2004 that discussed the economic value of a rehabilitated commercial and recreational Walleye fishery (*Sensu* Petzold 2004; Bobrowicz 2010). In 2005, the Black Bay Walleye Rehabilitation Plan was formed, which recommended stocking of summer fingerlings with an estimated cost of $500,000 (*Sensu* Petzold 2005; Bobrowicz 2010). Although this plan did not come to fruition, stocking events have occurred in Black Bay, including 1,032 adult Walleye from the Current and Pigeon rivers in 1972, 768 adult Walleye from local inland lakes from 1998-2000, the stocking of 1,000,000 Walleye fry from Cloud Lake in 2003, and stocking 260,000 summer fingerlings from the St. Marys River in 2004-2005 (Bobrowicz 2010). In 1999, Walleye angling was banned in the Black Sturgeon River from the first set of rapids downstream to Lake Superior, as well as in Black Bay north of Bent Island (Furlong et al. 2006). In 2008, fisheries management zone (FMZ) 6 Fishing Regulations changed to ban Walleye fishing from the Camp 43 Dam downstream to Lake Superior. First Nations continue to exercise their subsistence fishing rights in Black Bay and the Black Sturgeon River, with many angling in the lower Black Sturgeon River during spring spawning. The Red Rock Indian Band (RRIB) encourages members to exercise their rights in an ecologically responsible manner, while The Métis Nation of Ontario (MNO) has a ban for its members fishing on the Black Sturgeon River. The subsistence harvest is the only remaining fishery on the lower Black Sturgeon River (Bobrowicz 2010).

FWIN assessments completed by the OMNRF and genetic analysis of Black Bay Walleye reveal the current status of Walleye recovery. In 2008, the OMNRF FWIN assessment showed that 30 percent of the 2008 catch was comprised of St. Marys River strain fish that were stocked in 2004 and 2005 (Addison and Bobrowicz 2009). Garner et al. (2013) collected genetic data from Black Bay Walleye between 2007 and 2010, building upon initial genetic analysis completed by Wilson et al. (2007). Results indicated that the 2003 release of 1,000,000 fry (i.e., Cloud Lake stocking event) had no measurable contribution to the population, making up less than two percent of individuals. Further, the 2004 release of fingerlings made up 71 percent of their age class and the 2005 release comprised 45 percent of theirs (i.e., the two St. Marys River fingerling stocking events; Garner et al. 2013). More than half of the sampled fish were assigned as wild-origin fish from the lower Black Sturgeon River, and approximately one-sixth was attributed to the upper Black Sturgeon River. The genetic data collected by Garner et al. (2013) were analyzed along with samples from the 1966 Black Bay commercial fishery, lower Black Sturgeon River, upper Black Sturgeon River, St. Marys River, Cloud Lake, and Black Bay. Garner et al. (2013) noted there was significant differences in the genetics of all reference populations, as well as the 1966 historical sample. The current Walleye population in Black Bay and the Black Sturgeon River are made up of the St. Marys River population from the 2004-2005 stocking events and Walleye native to the Black Sturgeon River. Garner et al. (2013) noted that of the 68 individuals sampled from the 2007-2009 age classes, none had either full or mixed St. Marys River ancestry; thus, it was suspected that hatchery fish were not contributing to natural recruitment (Garner et al. 2013). More recent FWIN data from 2012-2017 have high proportions of younger Walleye being captured, showing signs of natural recruitment in Black Bay. Genetic analysis was not conducted for the recent FWIN assessments; however, frequencies of age-5 Walleye in 2010, age-7 Walleye in 2012, and age-8 Walleye in 2013 are also consistent with the 2004 and 2005 St. Marys River strain stocking events (E. Berglund (OMNRF) 2021, unpublished data).

3.1.4 Lake Sturgeon and Walleye spawning habitat

Walleye

The Walleye in Black Bay were initially thought to be a shoal-spawning population (Furlong et al. 2006); however, substrate sampling in Black Bay did not find habitat conducive to spawning (Biberhofer and Prokopec 2007) and a radio telemetry study observed Walleye gathering in the Black Sturgeon River during the spring spawning period (Furlong et al. 2006). As such, Black Bay Walleye may be a lake resident-river spawner population that is limited by spawning habitat availability, as only 80,000 m^2 of rapids exist downstream of the Camp 43 Dam, 25 percent of which is immediately below the dam. Increased access to spawning habitat was deemed essential for the Black Bay Walleye population recovery as it was estimated that approximately 400,000 m^2 of spawning habitat would be needed to support the pre-collapse population, based on Walleye female fecundity and average rates of egg deposition (Bobrowicz 2010). Historically, it is thought that the nearby Wolf River contributed some spawning habitat for the Black Bay Walleye population; however, a Sea Lamprey barrier was constructed 4.5 km from the river mouth in 1987, impeding access to historical spawning sites (Bobrowicz et al. 2010).

Walleye spawning occurs in early spring when water temperatures are between 6-13°C (Ellis and Giles 1965). Typical depths conducive to Walleye spawning are between 0.5 and 1 m, while water velocities can range from 0-3 m/s (Bozek et al. 2011). Walleye prefer cobble and gravel substrates and are broadcast spawners whose eggs adhere to the substrate and can later settle into interstitial spaces (Scott and Crossman

1973; Ivan et al. 2010; Bozek et al. 2011). Further, Walleye have three different life-history strategies: river resident-river spawner, lake resident-lake spawner, and lake resident-river spawner (Bozek et al. 2011), the latter two being the proposed typologies of the Walleye in Black Bay. In Lake Erie and Lake Huron, Walleye have been observed to spawn on in-lake reefs, while separate stocks also spawn within tributaries (Bozek et al. 2011). Walleye are also known to demonstrate natal homing. For example, adult Walleye captured at the mouth of the Current River were tagged and released roughly 150 km away in the northern portion of Black Bay; within eight months of their release, 18.2 percent of the tagged Walleye were recaptured near the Current River (Colby and Nepszy 1981).

Lake Sturgeon

Lake Sturgeon populations throughout Lake Superior used to be plentiful; however, due to overharvesting, barriers to migration, and habitat degradation, their populations have reduced to critical levels (Auer 2003). Access to additional suitable spawning habitats could be beneficial for the rehabilitation of Lake Sturgeon in Black Bay as long as the habitat is conducive to Lake Sturgeon spawning requirements. A self-sustaining population of Lake Sturgeon is deemed as one with a minimum of 1,500 mature spawning adults, using a common tributary (Auer 2003). Abundances in the Black Sturgeon River are much lower as COSEWIC (2017) estimates the number of mature individuals as less than or equal to 200 (Haxton et al. 2014b). Specifically, Lake Sturgeon in the Black Sturgeon River were surveyed in 2003-2004 as part of an adult Lake Sturgeon spawning study. The abundance of both adults and juveniles was noted to be low (i.e., between 50-500 individuals), with an estimated adult abundance of 89 spawning individuals (CI: 54,138) in 2003 and 96 spawning individuals (CI: 47, 240) in 2004 (Friday 2004; COSEWIC 2017). A population estimate could not be conducted in 2002 based on small sample size (n = 11) and low recaptures (n = 2; Friday 2004). The presence of a dam as a migratory barrier (i.e., the Camp 43 Dam) was noted as an impediment to recovery (COSEWIC 2017).

Lake Sturgeon spawn in late spring to early summer, depending on location, at temperatures in the range of 9-18°C (Bruch et al. 2016). Baril et al. (2018) noted that peak suitability for Lake Sturgeon spawning habitats had water velocities of 0.6 m/s, depths of 0.55-0.85 m in small rivers and 0.75-5.25 m in large rivers, and cobble substrates (64-256 mm in diameter). Kerr et al. (2010) notes that boulders for current breaks and distances of less than 3 km from staging areas (i.e., where Lake Sturgeon will rest during migration) are also preferred. Sexually mature female Lake Sturgeon can each lay between 49,000-667,000 eggs (Peterson et al. 2007) every 3-7 years (Kerr et al. 2010). When prevented from further migration, Lake Sturgeon will spawn at the base of dams (Auer 1996) that pose an impassable barrier and prevent access to historic spawning grounds (Ferguson and Duckworth 1997; Peterson et al. 2007; Dumont et al. 2011; Thiem et al. 2013). Radio telemetry has shown that Lake Sturgeon occupy the Black Sturgeon River during the open water season (*Sensu* Friday, unpublished data; reviewed in Bobrowicz 2010) and that Lake Sturgeon congregate at the Camp 43 Dam during the spring spawning period (Friday 2004), where the Camp 43 Dam is a barrier to further migration.

Spawning Habitat Availability

Upstream of the Camp 43 Dam to the Camp 1 site (i.e., a former dam at the outlet of Eskwanonwatin Lake, 67 km from the mouth of the Black Sturgeon River), Sakamoto (2007) identified an additional 325,000 m^2 of rapids. Addison and Bobrowicz (2009) suggested that the 2007 rapids inventory was a reasonable approximation of potential spawning habitat in the river. The rapids were also partially assessed by the Upper Great Lakes Management Unit (UGLMU); and it noted that 75 percent are likely suitable for spawning and are primarily cobble substrate, with gravel and boulders, ranging in depth from 0.5-2 m (Bobrowicz 2010). Thus, of the 80,000 m^2 of rapids below the Camp 43 Dam and the 325,000 m^2 above the Camp 43, approximately 303,750 m^2 (i.e., 75%) is likely suitable spawning habitat. However, only 20% is available downstream of the Camp 43 Dam. High Falls, located upstream of the Camp 43 Dam, was thought to be a potential natural barrier to Walleye migration. However, in August 2009, the UGLMU concluded that based on observations and literature reviewed, it is unlikely that High Falls is a vertical or velocity barrier to Walleye, with the exception of extreme conditions (Bobrowicz 2010). This was concluded through depth and flow measurements at the site and analysis of water levels during spring runoff and throughout the summer (Bobrowicz 2010). As such, there are no natural barriers to migration upstream of the Camp 43 Dam, and Lake Sturgeon and Walleye may have used historical spawning sites farther upstream. Thus, the Camp 43 Dam restricts access to 80 percent of historically available spawning habitat.

3.1.5 *Sea Lamprey control and potential management options*

In 1946, Sea Lamprey were first reported in Lake Superior (Smith 1972). In 1955, as a response to the threat of Sea Lamprey predation, the Great Lakes Fishery Commission (GLFC) was established with the task of eradicating or managing Sea Lamprey populations in the Great Lakes and to protect the fishery (Miehls et al. 2020). The GLFC and their Sea Lamprey Control Program (SLCP) is a binational partnership between the DFO and the U.S. Fish and Wildlife Service (USFWS). Sea Lamprey populations have been reduced to 10 percent of their peak population size, which has been widely attributed to the use of 3-trifluoromethyl-4-nitrophenol (TFM) treatments in infected streams that started in Lake Superior in 1958 (Lawrie 1970). TFM is a piscicide that has been used to control Sea Lamprey that rarely harms non-target fishes (Birceanu and Wilkie 2018), although lampricides have been found to negatively impact certain fish, including Lake Sturgeon (Boogaard et al. 2003; O'Connor et al. 2017). Niclosamide (a molluscicide) is also used as a lampricide and to coax lamprey from their burrows; also, by adding 1% niclosamide to TFM can reduce the amount needed for treatments (Boogaard et al. 2003; McDonald and Kolar 2007). Currently, control of Sea Lamprey populations is mainly achieved by targeting larval life stages with lampricides (i.e., TFM and niclosamide). Adult life stages are additionally impacted by barriers to further migration as they reduce access to spawning areas (Furlong et al. 2006; Wilkie et al. 2019; Miehls et al. 2020). The Camp 43 Dam is currently used as a barrier to Sea Lamprey migration and is an essential component of the binational SLCP mandated by the GLFC and completed by the DFO (Bobrowicz 2010).

Bobrowicz (2010) presented an options evaluation for the rehabilitation of native fisheries in Black Bay and the Black Sturgeon River, including Walleye and Lake Sturgeon. Of the five main options presented, only two are currently considered to be potentially successful management options. The first option is to modify the Camp 43 Dam to include a trap and sort fishway. The next is to decommission the Camp 43 Dam and install a new barrier at the Camp 1 site (i.e., approximately 67 km from the mouth of the river at the outlet of Eskwanonwatin Lake). Prior to the construction of the Camp 43 Dam, the Black Sturgeon River, as well as three of the four significant tributaries located above the current dam (i.e., Mound, Mouseau, Shillabeer, and Larson) were treated with lampricide (OFAH 2017). In the event the Camp 43 Dam is removed, Sea Lamprey would migrate to inhabit upper reaches of the Black Sturgeon River and lampricide treatments would have to be implemented to control the spread. Also, there is a Northern Brook Lamprey (*Ichthyomyzon fossor*) population upstream of the Camp 43 Dam that would be impacted by lampricide treatments (Bobrowicz 2010).

3.1.6 Fish movement

There has been much debate surrounding the potential removal of the Camp 43 Dam. Understanding the movement ecology and spawning activity of Walleye and Lake Sturgeon within the Black Sturgeon River would help to inform management decisions regarding the Camp 43 Dam. Spatial ecology and the use of telemetry can provide information on important habitats that are necessary to manage and conserve fish populations. The use of biotelemetry (i.e., passive integrated transponders, radiotelemetry, and acoustic telemetry) allows for determining fish movement patterns

(Cooke et al. 2016). For passive tracking, acoustic telemetry uses battery-powered tags that are surgically implanted into fish and autonomous fixed-position receivers (Kessel et al. 2015a). Acoustic tags emit a unique acoustic signal that is recorded when the tag enters the detectable range of a receiver (Stasko and Pincock 1977). Each recording stores information on tag identity, as well as the date and time of each detection, which can be used, along with a receiver array to determine fish movement. When determining migration routes in riverine systems, gated designs are used for the receiver array (Hayden et al. 2014).

3.2 Objectives

The objectives of this chapter are 1) to determine if the recovering Walleye population in Black Bay are river spawners that use the lower Black Sturgeon River for spawning habitat, and 2) to determine the extent of Walleye and Lake Sturgeon migration in the lower Black Sturgeon River. It is hypothesized that Walleye and Lake Sturgeon populations are currently limited by spawning habitat availability in the lower Black Sturgeon River, and as such, they will attempt to migrate up the river during the spawning season until they are prevented from moving further by the Camp 43 Dam. It is hypothesized that Lake Sturgeon and Walleye will migrate to the base of the Camp 43 Dam during the spawning season.

3.3 Methods

3.3.1 Study site

The Black Sturgeon River empties into Black Bay, a 60,000 ha embayment located in northwestern Lake Superior. The Camp 43 Dam is situated approximately 17 km upstream from the mouth of the Black Sturgeon River, creating an impassible barrier. Of the 303,750 m^2 of potential spawning habitat downstream of the Camp 1 site (i.e., approximately 67 km from the mouth of the Black Sturgeon River), approximately 20 percent is downstream of the Camp 43 Dam (Sakamoto 2007; Bobrowicz 2010). The Camp 43 Dam location had been identified as a potential spawning site for Lake Sturgeon and the OMNRF has also identified the Highway 17 Rapids and Unnamed Rapids, as well as the Camp 43 Dam, as potential spawning locations for Walleye (Figure 3.2).

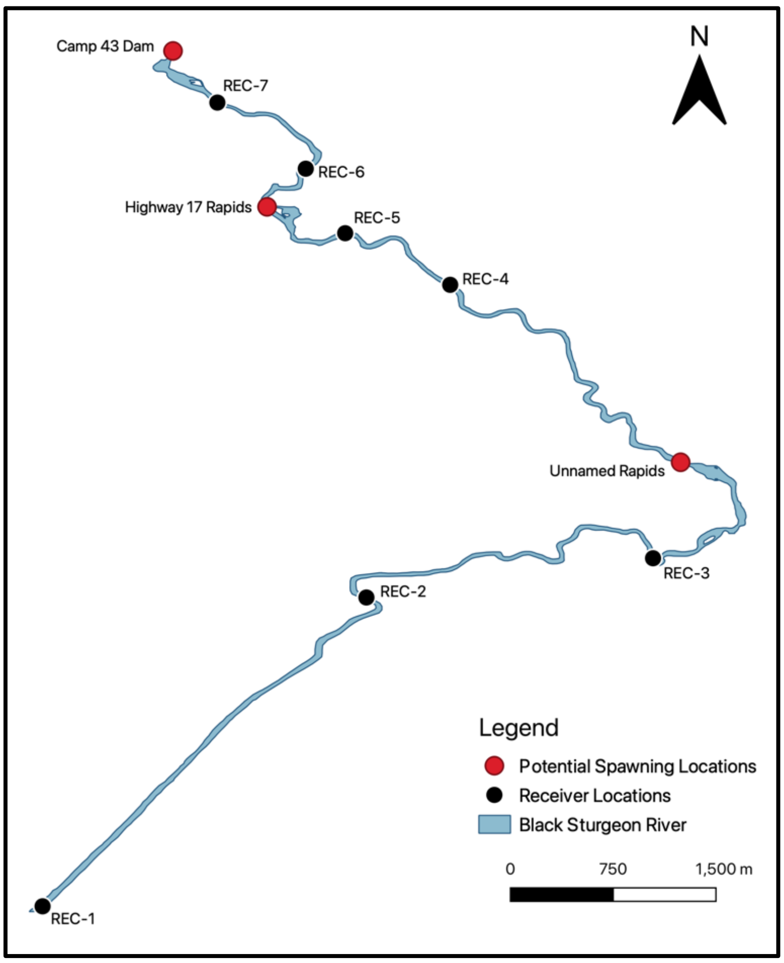

Figure 3.2: Map of the study site in the lower Black Sturgeon River, Ontario. Acoustic telemetry receiver locations are indicated by black circles and labelled (e.g., "REC-1"). Potential spawning locations are indicated by red circles and labelled (e.g., Unnamed Rapids). GIS data was provided by the DFO Sea Lamprey Control Centre (Black Sturgeon River) and the OMNRF (Receiver Locations). Map created in QGIS.

3.3.2 Walleye and Lake Sturgeon acoustic tagging

Walleye

A total of 198 Walleye were tagged in Black Bay and the Black Sturgeon River from 2016-2019 for use in the Black Bay Walleye Acoustic Telemetry (BBWAT) Project. From May to July 2016, 94 Walleye were tagged; in 2017, 61 Walleye were tagged; from May to September 2018, 37 Walleye were tagged; and in October 2019, 6 Walleye were tagged. Adult Walleye were captured using various sampling methods (i.e., trap nets, electrofishing, short set gill nets, and angling; G. McKee (Lakehead University) 2021, unpublished data). All fish were measured for total length, and the second dorsal spine was clipped for age and growth analysis. In 2016, fish were anesthetized with clove oil (60 mg clove oil/L water), and in 2017-2019, electric fish handling gloves (32V-39V, 4mA-25mA; Smith-Root, Vancouver, WA) were used for anesthetization before surgery. Following procedures approved by the Canadian Council on Animal Care (Lakehead University file #1465777), fish were placed in a padded trough for surgery, where a small incision was made on the ventral side posterior to the pelvic girdle. Then, acoustic transmitter tags (Vemco V16-4X (n = 167; 2435 days projected battery life); V16TP-4X (n = 15; 2305 days projected battery life); V13-1X (n = 8; 904 days projected battery life and n = 9; 602 days projected battery life), Halifax, Canada) were surgically implanted into the coelomic cavity while onshore, near capture locations. The incision was closed with three sutures (polydioxanone absorbable monofilament; Ethicon, Somerville, NJ), and all fish were tagged with an external anchor tag (Floy Manufacturing) and released at their capture sites (G. McKee (Lakehead University) 2021, unpublished data; K. Stratton (OMNRF) 2021, personal communication).

Lake Sturgeon

In May 2016, the DFO tagged 20 Lake Sturgeon captured in Black Bay with acoustic transmitters. Sub-adult or adult Lake Sturgeon with total lengths greater than 1000 mm were the preferred size range for capture. Gill nets with mesh sizes of 20.3, 25.4, 30.5, and 35.6 cm (8, 10, 12, and 14 in; stretched bar measure) were set between 4-15 m depth for approximately 24 hours. Lake Sturgeon were processed and handled using DFO animal care protocols. Fish were removed from the live well and placed with their ventral side up, inducing a state of tonic immobility (Cooke et al. 2013). Total length, weight, fork length, and girth measurements were taken, as well as a fin segment from the left pectoral fin ray for age analysis. All individuals were tagged with an external Floy-style spaghetti tag, inserted on the left-hand side below the dorsal fin, and an 11 mm PIT tag, applied below the third dorsal scute. A subcutaneous lidocaine (2 mg/kg) injection at the site of the incision was given to numb the area. Acoustic transmitter tags (Vemco Model V16-4X; 3393 days projected battery life) were surgically implanted by making an approximately 25 mm incision left of the mid-lateral line. The transmitter was inserted, and the incision was closed with 2-4 sutures. Fish were then returned to a live well for recovery and were later released near their capture location (W. Gardner (DFO) 2021, personal communication).

In addition to the Lake Sturgeon tagged by the DFO, from May 2015 to 2017, the OMNRF tagged 11 (3 in 2015, 5 in 2016, and 3 in 2017) Lake Sturgeon in the Black

Sturgeon River at a known staging location (i.e., High Falls), upstream of the Camp 43 Dam, to capture pre-spawn male and female Lake Sturgeon as part of an OMNRF study. These individuals migrated below the Camp 43 Dam after tagging and are included in the analysis. Gill nets with mesh sizes between 20.3 to 30.5 cm (8 to 12 in) were set overnight, at depths less than 4 m as per Haxton et al. (2014c). Lake Sturgeon were processed and handled using OMNRF animal care protocols. Total length, sex, and weight (to the nearest 100 g) were recorded, and a PIT tag was inserted beneath the third dorsal scute for each fish. All individuals of sufficient size for transmitter implementation were placed in a large, covered tank and transported a short distance to an upstream landing for processing. MS-222 (tricaine methanesulfonate) was used to anesthetize the Lake Sturgeon for surgery. A small incision in the body wall proximal to the midline was made, and an acoustic transmitter tag (Vemco Model V16-4X) was implanted. The incision was closed using gut suture material (size 4-0), and individuals were placed in a tank with fresh water and an aerator for recovery. All OMNRF Lake Sturgeon were transported by boat and released in the upper Black Sturgeon River at High Falls (i.e., approximately 5 km upstream of the Camp 43 Dam) after recovery. All fish tagged by the OMNRF had estimated tag battery lives of 2883 days, except two fish tagged in 2017 with an expected tag life of 1549 days (T. Cano (OMNRF) 2021, personal communication).

3.3.3 Acoustic receiver deployment and retrieval

Walleye and Lake Sturgeon in the Black Sturgeon River were detected on omnidirectional acoustic receivers (VR2W, 69 kHz; Vemco) deployed as part of the Great

Lakes Acoustic Telemetry Observation System (GLATOS) BBWAT Project. In the Great Lakes, GLATOS, established by GLFC, uses a network of acoustic telemetry data from various researchers to aid in the understanding of overall fish ecology and to help with management decisions (Krueger et al. 2018). Receivers were deployed in a gated design to determine fish movement along a potential migratory route in the river (Heupel et al. 2006). Receiver names in the original dataset were BSR-001 through BSR-007 but were not in a stepwise order from down- to upstream. Receivers were renamed as REC-1, at the river mouth, through REC-7, at the Camp 43 Dam, with receivers numbered accordingly with lower numbers downstream and higher numbers upstream.

In 2016 and 2017 only two receivers were deployed in the Black Sturgeon River; REC-1 at the river mouth and REC-7 downstream of the Camp 43 Dam. In 2018, three additional receivers were deployed (REC-2, REC-3, and REC-4) to further delineate migration extent in the river. In 2019, REC-6 was added between REC-4 and REC-7, upstream of the Highway 17 Rapids. And in 2020, REC-5 was deployed upstream of REC-4 (see Figure 3.2). Receivers were deployed with an anchor and float and sat on the river bottom or were suspended 1 m from the substrate, at depths between 2 to 6 m. Receivers were initially deployed in 2016; however, there were gaps between recoveries and subsequent deployments until early 2018. Since then, there have been consistent detections as receivers are recovered and deployed on the same day.

3.3.4 Potential mortalities and removed fish

Although Walleye and Lake Sturgeon were tagged over multiple years, some fish are removed (i.e., via natural or human-induced mortality) and as such, it is important to know the total number of fish actively migrating in a given year. If tagged Walleye or Lake Sturgeon were never detected after surgery on a BBWAT receiver, they were removed from the dataset and considered potential mortalities. For each year, additional potential mortalities were determined if a fish had their last detection on a BBWAT receiver prior to the spawning season (i.e., April of a given year for Walleye and May of a given year for Lake Sturgeon) and were not detected again. Although it is important to note that these are not definitive mortalities, rather potential mortalities, as fish may be harvested, or simply occupying habitat where no acoustic receivers are present.

3.3.5 Data analysis

Detection range testing of receivers in the BBWAT array was conducted by the UGLMU near Bent Island in Black Bay in 2017 and at REC-7 in 2018. Detection ranges for REC-7 in the Black Sturgeon River near the Camp 43 Dam were 97% at 150 m and 81% at 185 m (UGLMU 2018, unpublished data). The REC-1 receiver is located at the mouth of the Black Sturgeon River and conditions are likely more similar to receivers within Black Bay. At Bent Island in Black Bay detection range is 91% at 750 m (UGLMU 2017, unpublished data). As such, based on the positioning of REC-1 at the mouth of the river and the detection range of receivers within the BBWAT array, some detections were likely on fish swimming by the mouth of the river and not entering for the purpose of spawning. Thus, fish detected at REC-2 are considered as the total number that entered the river to potentially spawn in each year. REC-2 was not deployed until 2018, and therefore analysis and interpretation of spawning for previous years (i.e., 2016 and

2017) for both Walleye and Lake Sturgeon is limited. For this assessment, a fish was considered as likely spawning at a location if it was not detected at the next upstream receiver during the spawning season (i.e., April to June for Walleye and May to July for Lake Sturgeon).

All analyses were conducted using R (Version 3.6.2; R Core Team, 2019). Before analysis of detection data, false detections were removed from the dataset. This was done following the method of Pincock (2012), who indicated that when using Vemco acoustic transmitter tags with a nominal delay of 120 seconds (i.e., transmits randomly every 60 to 180 seconds), it is recommended that single detections (i.e., not accompanied by another detection on the same receiver) within a one-hour time interval (i.e., 30 times the nominal delay of the tag) be removed. Given the nature of the receiver array (i.e., gated in a river), false detections were manually checked to ensure a true false detection, rather than a delayed detection of a fish moving in a section of the river outside the receiver range. For example, if a fish was detected at REC-3 at a given time, but then was next detected on that receiver more than an hour later, it is flagged as a "false detection" rather than a fish that moved outside of the receiver range; thus, these detections were not removed. Using the criteria above 1,010 (0.18%) of 560,104 detections were flagged as false; after a manual check, only three detections were deemed truly false and removed from the dataset. As mentioned in Section 3.3.4, potential mortalities or removals were identified and were removed from each year's dataset for analysis, and all remaining fish had estimated tag lives that should span the entirety of the analysis.

To complete the first objective of determining if Black Bay Walleye are river spawners, the proportion of individual Walleye detected at each receiver in the river was assessed from 2017 through 2020. Further, residency was determined using fish detection data on the receivers throughout the Black Sturgeon River. Based on the distances between each receiver, there should be no detection range overlap. Thus, based on detections at each receiver, travel direction can be inferred, and movements during migration can be quantified. Residency indices (RI) were completed following methods from Kessel et al. (2015b). To determine RI, the total number of distinct days (T) a fish was detected at any receiver, and the distinct days per each receiver (S) were determined and RI was calculated using the following equation: $RI = S/T$. To complete this for a monthly time frame, the number of days a fish was detected per receiver per month (i.e., the RI value) was calculated and divided by the total number of days within that month (i.e., the mean RI value). For example, if a fish was heard ten times on a receiver in April (i.e., the RI value), which has 30 days, the mean RI would be $Mean\ RI = 10/30 = 0.33$. Each month and year with detections was subset into a data frame, and the mean RI was calculated.

Mean RI values were then transposed onto a shapefile for the lower Black Sturgeon River to create bubble plots, with each circle representing the mean RI for all tagged individuals separated by species. RI for Walleye was calculated for April, May, and June of each year to correspond to the spawning season. For the Lake Sturgeon spawning season, RI was calculated for May, June, and July. The residency indices and proportions of individual Walleye and Lake Sturgeon detected at each receiver in the

Black Sturgeon River from 2017 through 2020 were used to investigate the second objective, specifically to determine the extent of Walleye and Lake Sturgeon migration in the lower Black Sturgeon River.

To compare differences in Lake Sturgeon using the Black Sturgeon River during the spawning season and individuals that have not been detected at the Camp 43 Dam during the spawning season, mean ages were compared. As assumptions (i.e., normal distribution and homogeneity of variance) of a parametric test could not be met, a Kruskal-Wallis test was completed as a non-parametric alternative to an analysis of variance (ANOVA; Whitlock and Schluter 2009). As each Lake Sturgeon's age was determined in the year of capture, age was prorated to age in 2020. The age of fish that were detected at REC-7 were compared to individuals that did not enter the river to spawn (i.e., were not detected at REC-7 during the spawning season).

To determine if there is a difference between Walleye migration during the spawning season in the lower Black Sturgeon River, Chi-squared contingency tables were performed for each year with adequate delineation (i.e., more than two receivers) within the river (i.e., 2018, 2019, and 2020). For this statistical test, the null hypothesis is that Walleye spawning is the same across the three potential spawning sites (i.e., the Unnamed Rapids, the Highway 17 Rapids, and the Camp 43 Dam); thus, a proportion of 0.33 was expected at each site. If results were significant (i.e., $p < 0.05$), post-hoc 2x2 contingency tables were completed to identify the significant differences among sites. The level of significance for the post-hoc tests (alpha) was adjusted to 0.0167 (i.e., 0.05/3), as there were three post-hoc tests. Chi-squared contingency tests have the assumption that no more than 20% of the cells can have expected frequencies less than five and that no cells can have an expected frequency less than one. Sample sizes should also be greater than 20. As such, this test cannot be used for the Lake Sturgeon data; instead, a Fisher's exact test was conducted (McCrum-Gardener 2008; Whitlock and Schluter 2009). However, as Fisher's exact test is a 2x2 contingency table, the null hypothesis is that Lake Sturgeon spawning is distributed equally between spawning at the Camp 43 Dam and elsewhere in the river; thus, an expected proportion of 0.5 was used.

3.4 Results

There were 560,104 Walleye detections and 111,102 Lake Sturgeon detections on receivers in the Black Sturgeon River array from 2016 to 2020. Two Lake Sturgeon tagged in 2017, and 51 Walleye (i.e., 29 in 2016, 9 in 2017, 7 in 2018, and 6 in 2019) were never detected after surgery on a BBWAT receiver and were not included in further analysis. Also, 2 (2017), 5 (2018), 16 (2019), and 18 (2020) Walleye and 2 (2018) Lake Sturgeon were last detected before a given year's spawning season and were also considered as harvested or a potential mortality. Thus, sample sizes of alive Walleye during the spawning season for each year are n = 65 (2016), n = 115 (2017), n = 140 (2018), n = 124 (2019), and n = 106 (2020); and for Lake Sturgeon are n = 28 (2016), n = 29 (2017), and n = 27 (2018, 2019, and 2020).

3.4.1 Walleye movement in the Black Sturgeon River

The proportion of Walleye detected at each receiver was fairly similar among years, given the number of total fish in a year. The proportion of Walleye entering the river to spawn and detected at REC-2 ranged from 56-68%. Nearly all those fish moved upstream to receiver REC-3, located just downstream of a potential spawning area (i.e., Unnamed Rapids), where 54-68% were detected. The second set of rapids (i.e., the Highway 17 Rapids) is located downstream of REC-6, and the proportion of fish detected upstream of REC-4 (in 2018 and 2019) and REC-5 (in 2020) ranged from 40-53%. The REC-7 receiver is located downstream of the Camp 43 Dam. Detections on this receiver, including the year 2017, ranged from less than 1% to 5%. Thus, from 2018 to 2020 (i.e., the years with more than two receivers in the river for delineation), 14-17% were likely spawning at the Unnamed Rapids location, 35-52% were likely spawning at the Highway 17 Rapids, and less than 1% to 5% of Walleye were likely spawning below the Camp 43 Dam (Figure 3.3). Across all years, the proportion of fish likely spawning in the river was the most at the Highway 17 Rapids, followed by the Unnamed Rapids, and with the least number likely spawning at the Camp 43 Dam. Across all years of detections with adequate delineation in the river (i.e., 2018-2020), 49% of Walleye were detected every year in the river during the spawning season, 25% were detected at least one year in the river during spawning season, and 26% were never detected in the river during the spawning season.

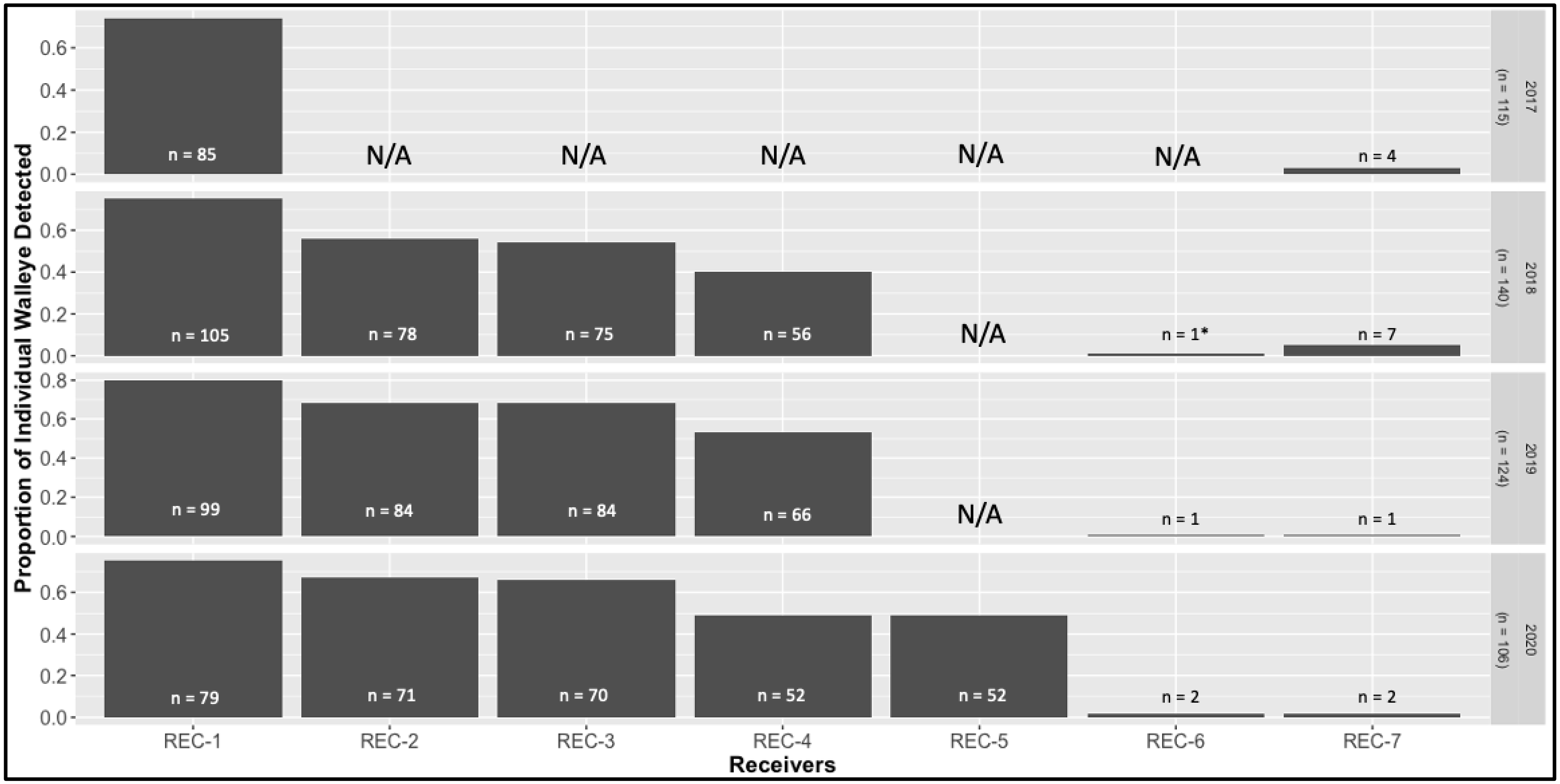

*Figure 3.3: The proportion of Walleye detected at each receiver of the total number for that year. Receivers are ordered by location from downstream (i.e., the river mouth) to upstream (i.e., the Camp 43 Dam). N/A indicates that the receiver was not deployed within the array in that year. * indicates a detection on a receiver deployed after the spawning season in that year.*

There were significant differences between expected proportions of Walleye spawning across all three spawning sites for 2018, 2019, and 2020 (Table 3.1). Post-hoc tests were conducted (i.e., three 2x2 contingency tables) to determine which sites were significantly different within a given year (Table 3.1). In 2019 and 2020, more Walleye potentially spawned at the Unnamed Rapids location than at the Camp 43 Dam. In all three years, more Walleye spawned at the Highway 17 Rapids than at the Camp 43 Dam. In 2019, there were fewer Walleye spawning at the Unnamed Rapids location than at the Highway 17 Rapids.

Table 3.1: Chi-square contingency tables for Walleye spawning across three sites. Significant comparisons are bolded.

Year		**χ^2**	**Df**	**p**
2018	All Sites	18.73	2	<.0001
	Unnamed Rapids X Highways 17 Rapids	5.09	1	.024
	Unnamed Rapids X Camp 43 Dam	2.85	1	.091
	Highway 17 Rapids X Camp 43 Dam	15.89	1	**<.0001**
2019	All Sites	42.03	2	<.0001
	Unnamed Rapids X Highways 17 Rapids	10.86	1	**<.001**
	Unnamed Rapids X Camp 43 Dam	10.16	1	**<.01**
	Highway 17 Rapids X Camp 43 Dam	36.67	1	**<.0001**
2020	All Sites	28.35	2	<.0001
	Unnamed Rapids X Highways 17 Rapids	5.66	1	0.17
	Unnamed Rapids X Camp 43 Dam	7.89	1	**<.01**
	Highway 17 Rapids X Camp 43 Dam	25.09	1	**<.0001**

Mean RI values of all Walleye (n = 115 in 2017, n = 140 in 2018, n = 124 in 2019, and n = 106 in 2020) indicated a preference towards REC-3 (i.e., just downstream of the Unnamed Rapids), which had four of the seven highest RI values (between 0.15-0.33) during the spawning season (i.e., April, May, and June). During the 2017 spawning season, the highest mean RI value was at REC-7 in June (i.e., 0.27). Throughout the 2018 spawning season, mean RI was highest at REC-3 in April (i.e., 0.32). The highest mean RI value during the 2019 spawning season and throughout the entire study period was 0.33 at REC-3 in June. Similarly, the highest mean RI value during the 2020 spawning season was at REC-3 in June. The mean RI values across each receiver through the spawning season are visualized in Figures 3.4, 3.5, and 3.6.

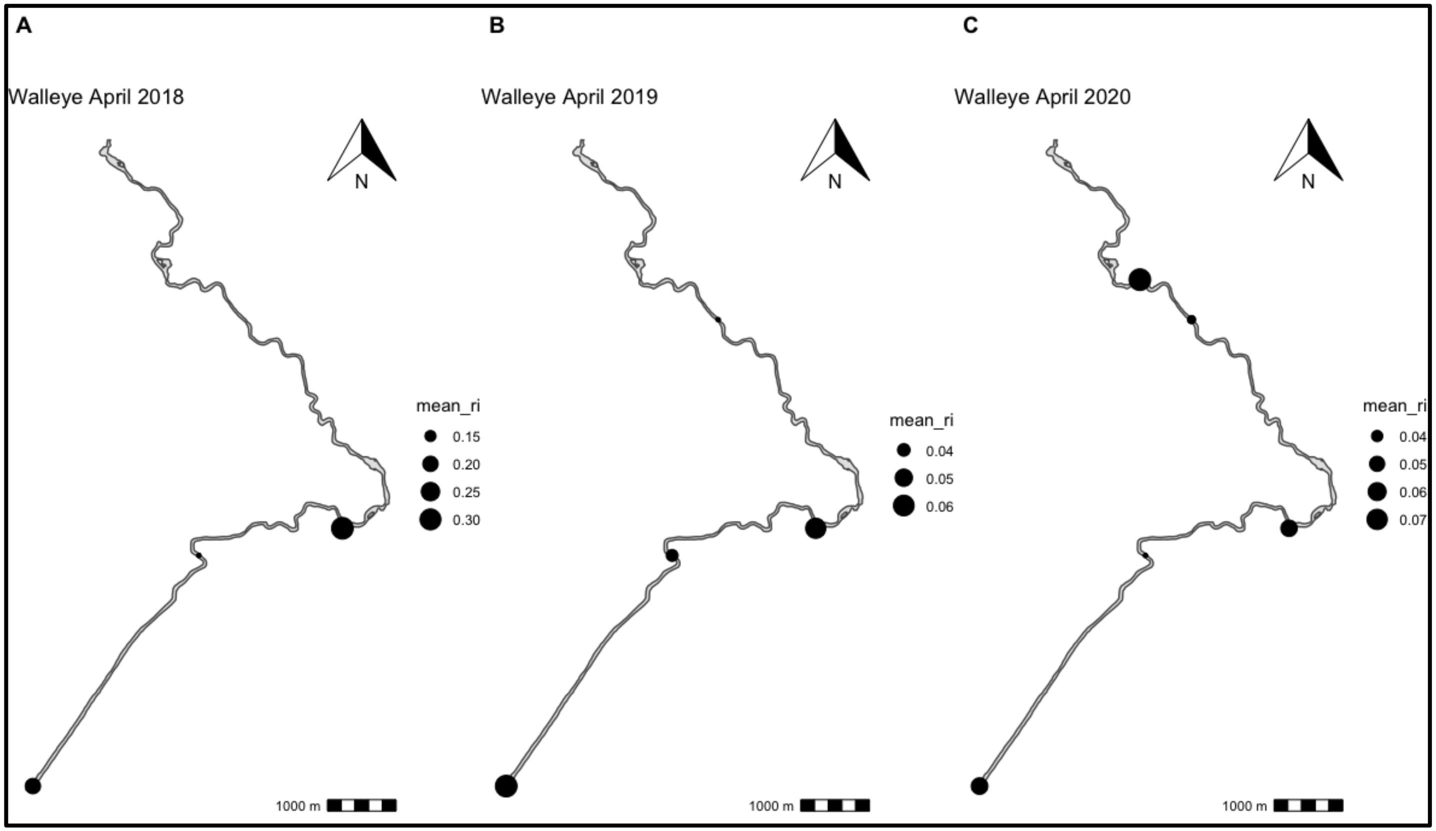

Figure 3.4: Walleye Residence Index by acoustic receiver station. Black circles represent the mean RI for all tagged Walleye across the spawning season, with specific month and year indicated [A) April 2018; B) April 2019; C) April 2020].

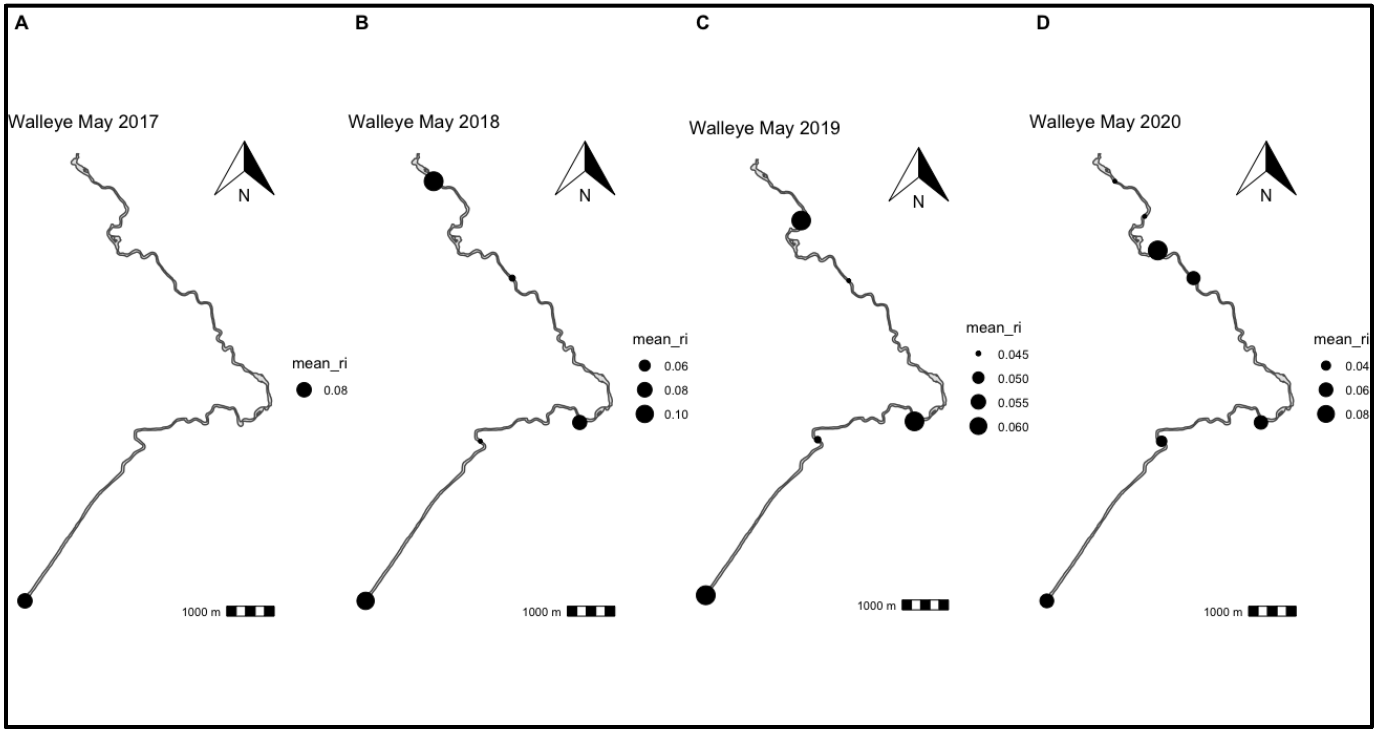

Figure 3.5: Walleye Residence Index by acoustic receiver station. Black circles represent the mean RI for all tagged Walleye across the spawning season, with specific month and year indicated [A) May 2017; B) May 2018; C) May 2019; D) May 2020].

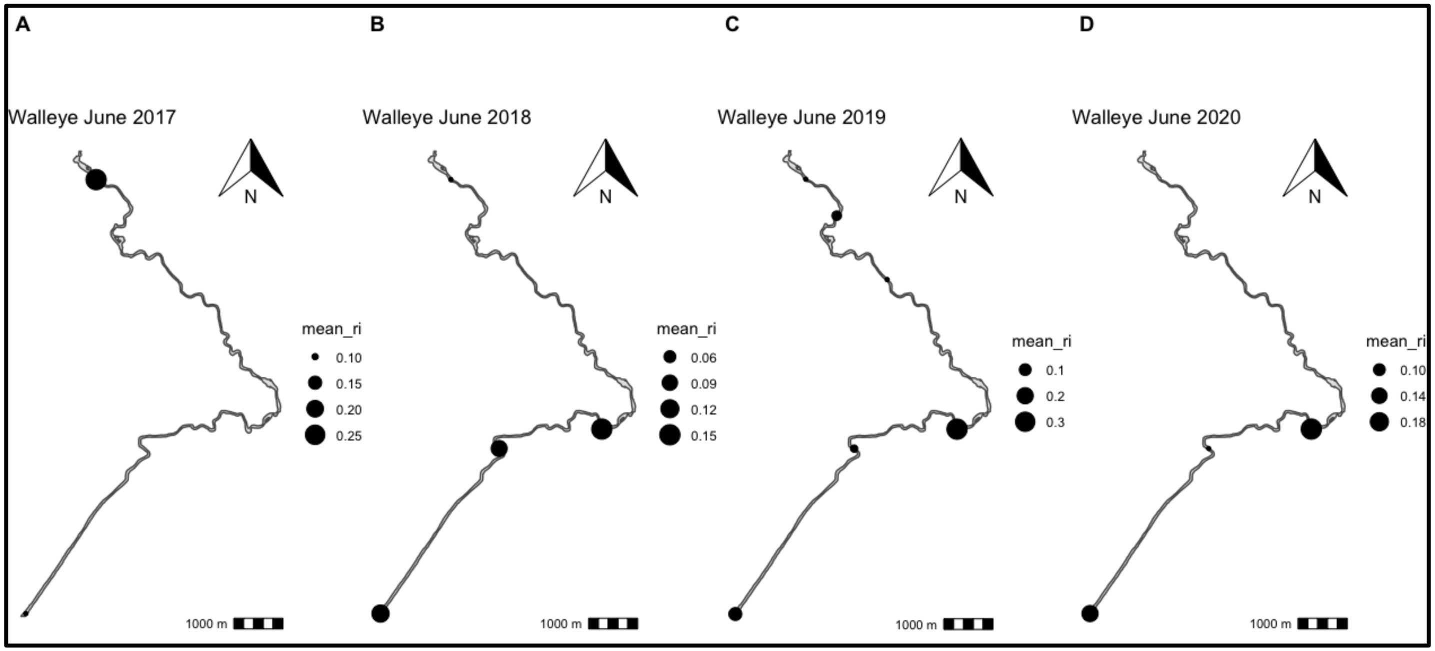

Figure 3.6: Walleye Residence Index by acoustic receiver station. Black circles represent the mean RI for all tagged Walleye across the spawning season, with specific month and year indicated [A) June 2017; B) June 2018; C) June 2019; D) June 2020].

3.4.2 *Lake Sturgeon movement in the Black Sturgeon River*

The proportion of tagged and alive Lake Sturgeon (n = 29 in 2017, and n = 27 in 2018, 2019, and 2020) detected at each receiver was comparable among years. The proportion of Lake Sturgeon detected entering the river during spawning season at REC-2 ranged from 37-52% (i.e., 10-14 individuals) annually. The remainder of the tagged Lake Sturgeon not entering the river to spawn were detected on other BBWAT receivers during the spawning period. The proportion of Lake Sturgeon detected at REC-7 (i.e., the receiver closest to the Camp 43 Dam), including 2017, ranged from 24-41%. Across all years, a consistent trend was observed where fish entering the river (i.e., detected at REC-2) migrate their full extent (i.e., to REC-7) until barred from further upstream movement by the Camp 43 Dam, with few exceptions (i.e., 4 of 14 fish in 2018 and 1 fish in 2019; Figure 3.7). Ten fish were not detected at REC-7 during the spawning season in any year. Lake Sturgeon were significantly more likely to aggregate just below the Camp 43 Dam than elsewhere in the river (Fisher's exact test; all years: $p < 0.05$).

Mean RI values of all Lake Sturgeon indicated a preference towards the upper section of the river during the spawning season (i.e., May, June, and July). The highest mean RI value during the 2017 spawning season was 0.22 at REC-7 in June. During the 2018 and 2019 spawning seasons, the highest mean RI values were at REC-7 in July (i.e., 0.23). The highest mean RI during the 2020 spawning season was at REC-2 (i.e., 0.19) in June, closely followed by REC-7 in June (i.e., 0.16). The mean RI values across each receiver through the spawning season are visualized in Figures 3.8, 3.9, and 3.10.

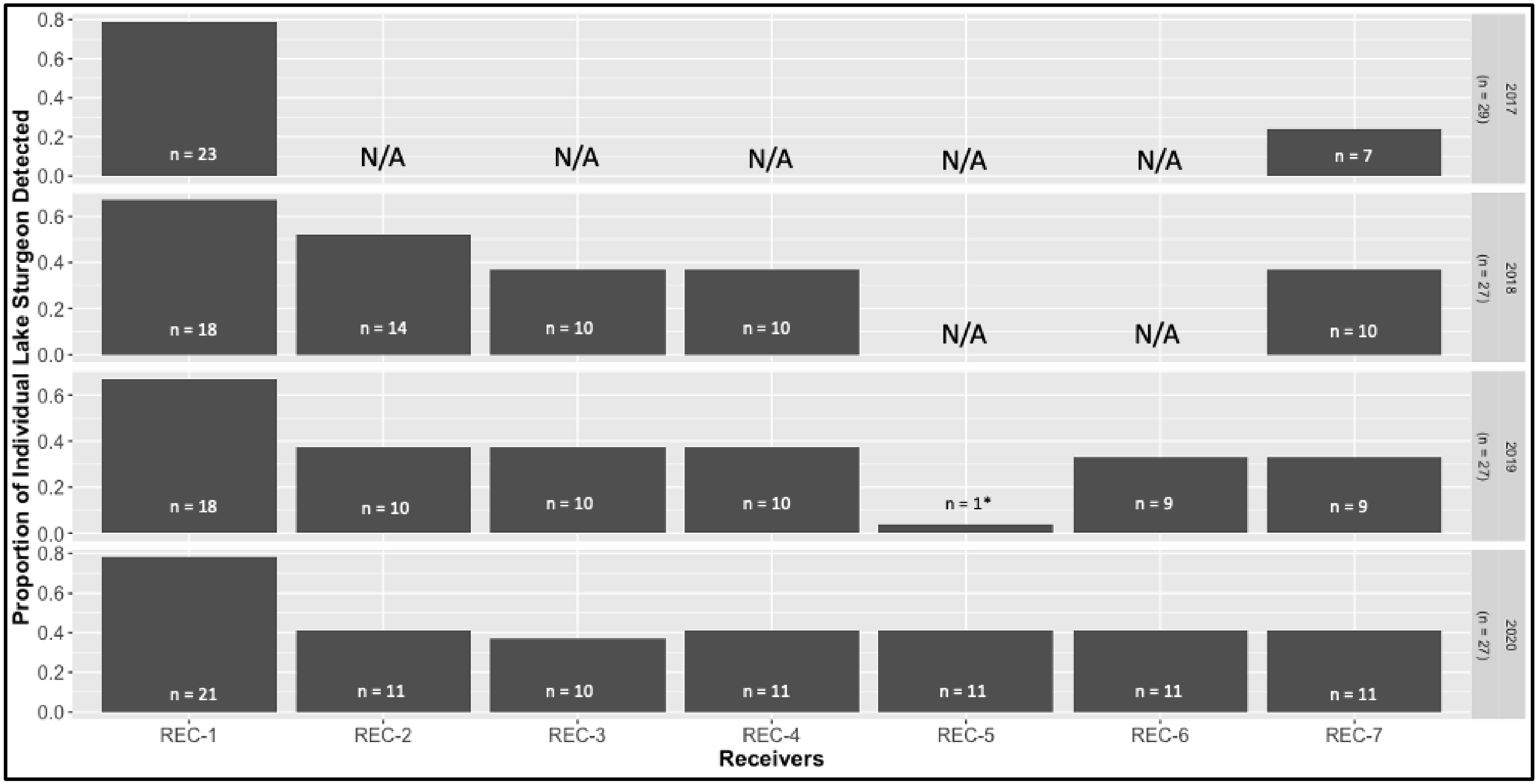

*Figure 3.7: The proportion of Lake Sturgeon detected at each receiver of the total number for that year. Receivers are ordered by location from downstream (i.e., the river mouth) to upstream (i.e., the Camp 43 Dam). N/A indicates that the receiver was not deployed within the array in that year. * indicates a detection on a receiver deployed after the spawning season in that year.*

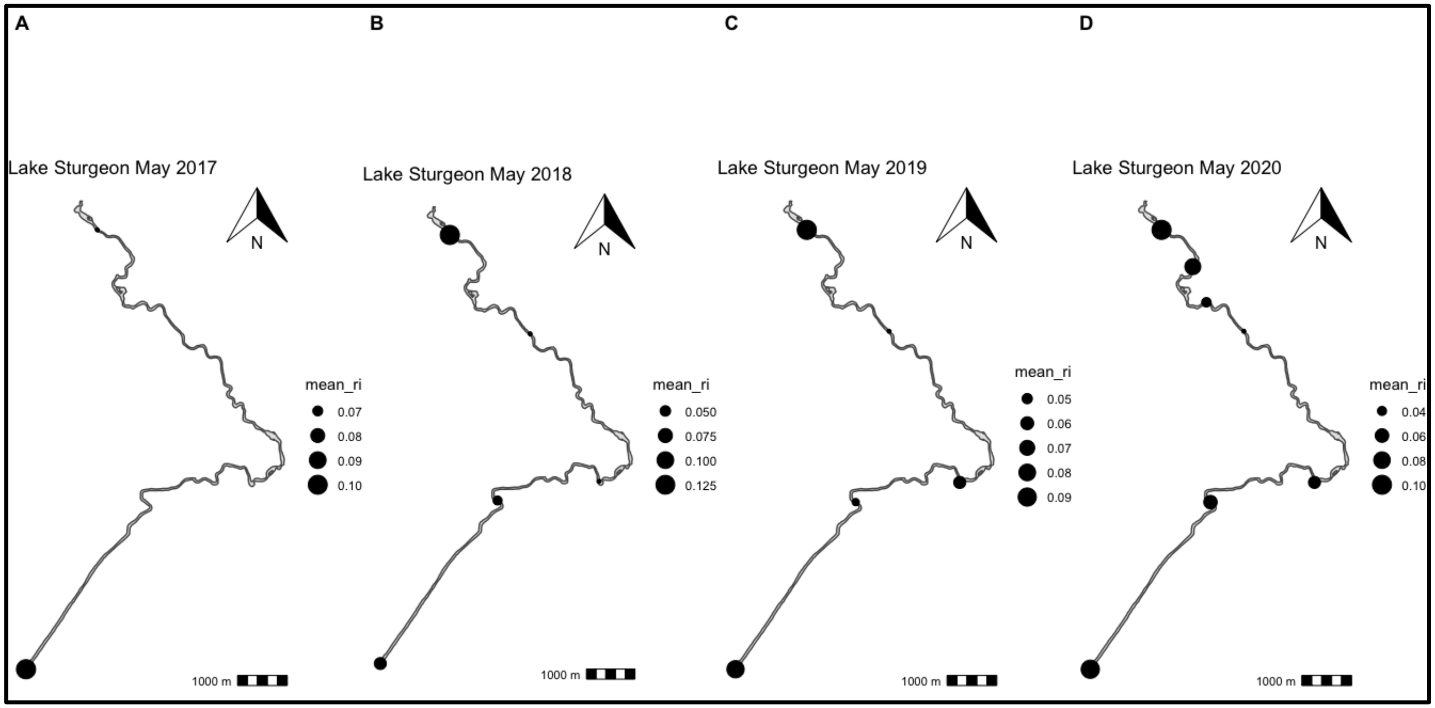

Figure 3.8: Lake Sturgeon Residence Index by acoustic receiver station. Black circles represent the mean RI for all tagged Lake Sturgeon across the spawning season, with specific month and year indicated [A) May 2017; B) May 2018; C) May 2019; D) May 2020].

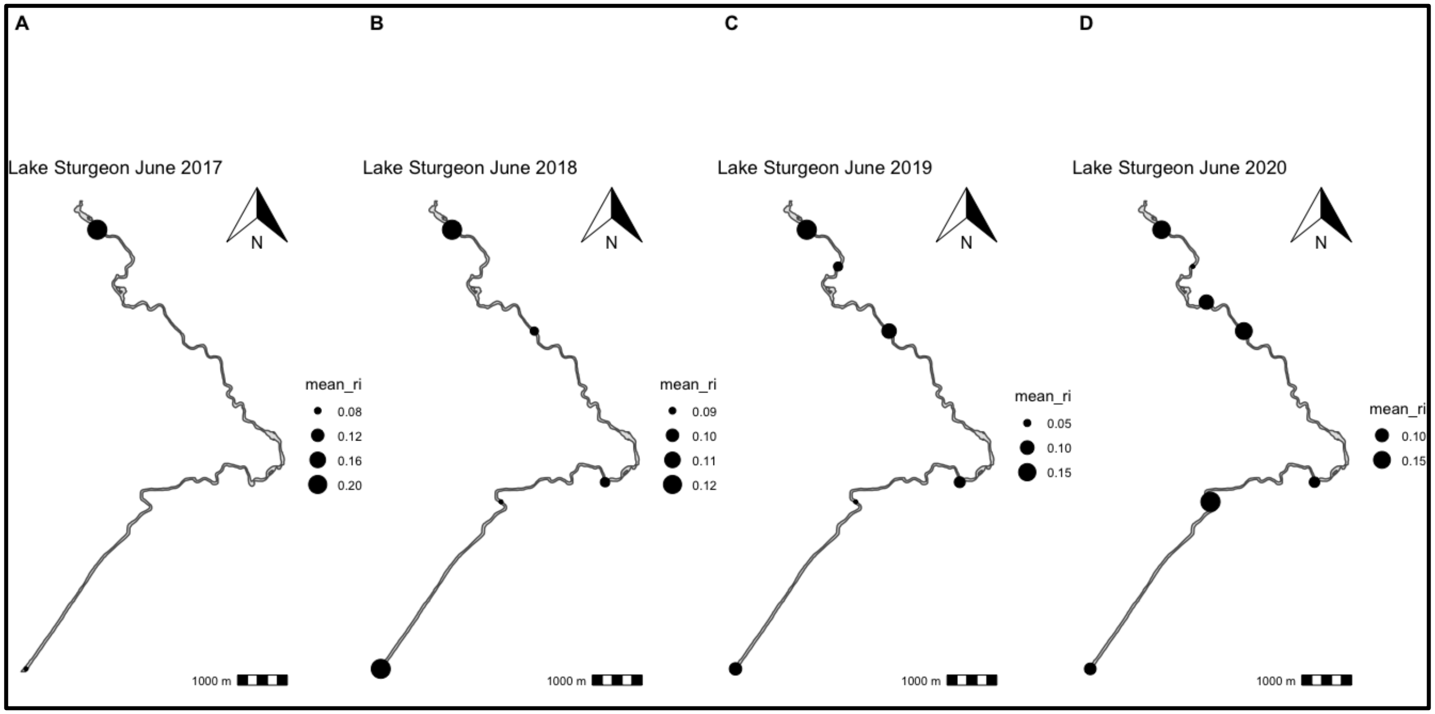

Figure 3.9: Lake Sturgeon Residence Index by acoustic receiver station. Black circles represent the mean RI for all tagged Lake Sturgeon across the spawning season, with specific month and year indicated [A) June 2017; B) June 2018; C) June 2019; D) June 2020].

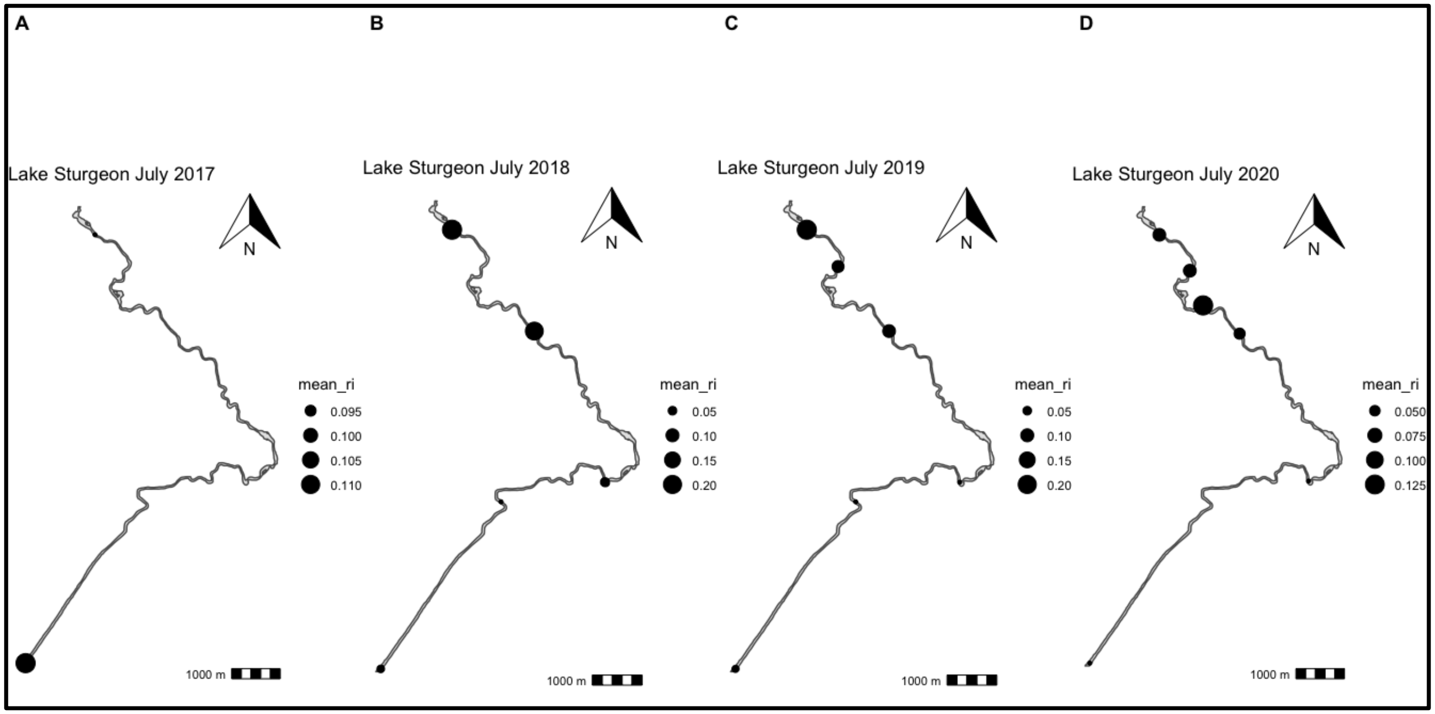

Figure 3.10: Lake Sturgeon Residence Index by acoustic receiver station. Black circles represent the mean RI for all tagged Lake Sturgeon across the spawning season, with specific month and year indicated [A) July 2017; B) July 2018; C) July 2019; D) July 2020].

The mean age of Lake Sturgeon, adjusted from age at capture to age in 2020, was determined to be 19.6 years for Lake Sturgeon that migrated to REC-7 (i.e., the Camp 43 Dam) and 16.8 years for the individuals that did not. A Kruskal-Wallis Test determined that there was not a significant difference between groups ($p > 0.05$).

3.5 Discussion

The purpose of this study was to examine the spatial ecology of Walleye and Lake Sturgeon, in the Black Sturgeon River during their respective spawning seasons, using acoustic telemetry. This study assessed the spawning migration of Black Bay Walleye and Lake Sturgeon populations at a local scale relative to the Camp 43 Dam, a barrier to further migration, to address whether the Camp 43 Dam is limiting access to spawning habitat and ultimately limiting Walleye and Lake Sturgeon recovery in Black Bay. This was achieved by determining if the recovering Walleye population in Black Bay are river spawners that use the lower Black Sturgeon River for spawning habitat, and by determining the extent of Walleye and Lake Sturgeon migration in the lower Black Sturgeon River. It was initially hypothesized that Walleye and Lake Sturgeon would spawn at the base of the Camp 43 Dam when prevented from further migration, as they are currently limited by spawning habitat availability in the lower Black Sturgeon River. However, for Walleye migrating in the river, it was identified that only a small percentage (i.e., 1% to 5% from 2017 to 2020) were potentially spawning at the Camp 43 Dam, and thus restoring river connectivity may not be as crucial for Walleye recovery as previously thought. In contrast, extent of movement upstream in the Black Sturgeon River towards the Camp 43 Dam for Lake Sturgeon indicates that further migration is likely if given access to the upper reaches of the river. Thus, if river connectivity was restored, either through the removal of the Camp 43 Dam or modification to include selective fish passage, this could lead to increased recruitment for Lake Sturgeon.

3.5.1 Walleye findings

Two additional potential spawning sites for Walleye were identified in the Black Sturgeon River (i.e., Unnamed Rapids and Highway 17 Rapids) in addition to the Camp 43 Dam location. It was determined that 49% of the tagged Walleye were detected in the Black Sturgeon River each year studied during the spawning season, whereas 25% were detected in at least one of the three years (i.e., 2018-2020), and 26% were never detected in the river during the spawning season. Walleye have three known life-history strategies: river resident-river spawner, lake resident-lake spawner, and lake resident-river spawner (Bozek et al. 2011). It may be that different life-history strategies are being utilized by Walleye throughout this system as not all tagged Walleye migrate into the Black Sturgeon River during the spawning season. Thus, those entering the Black Sturgeon River during the spawning season may be a lake resident-river spawner population. While the 26% that were never detected in the river during the spawning season may be a lake resident-lake spawner population. Genetic analysis has not been completed for the tagged Walleye, thus another explanation may be that the 26% of fish that were never detected in the river during the spawning season may be from stocking events and do not have the site fidelity towards potential spawning locations in the Black Sturgeon River. For the 25% of fish that were detected in the Black Sturgeon River during at least one of the three years with adequate delineation in the river (i.e., more than two receivers; 2018-2020), skipped spawning may be occurring. Skipped spawning is not unheard of for iteroparous fishes, as there may be advantages for mature fish to skip spawning (i.e., due to metabolic stress, deficient diet, poor nutritional condition, etc.; Rideout et al. 2005; Jørgensen et al. 2006; Rideout and Tomkiewicz 2011). Further, female Walleye and Lake Trout have been found to skip reproduction events, likely due to inadequate lipid reserves (Henderson et al. 1996; Sitar et al. 2014).

Residency Indices were conducted using Walleye detection data to better describe their movements during the spring spawning period. Mean RI values for Walleye during the spawning season indicated a preference towards REC-3, the last receiver before the first set of rapids (i.e., the Unnamed Rapids). This may suggest that Walleye stage in a slower section of the river before spawning at the first available rapids or migrating further to spawn at other upstream locations. Walleye may also be looking to feed post-spawn in a calmer segment of the river.

3.5.2 Lake Sturgeon findings

Lake Sturgeon detection data confirmed the hypothesis that they likely spawn at the base of the Camp 43 Dam as, unlike Walleye, they migrate upstream all the way to the dam during the spawning season. Across all years, a consistent trend was observed where fish entering the river migrated their full extent until barred from further upstream movement by the Camp 43 Dam, with few exceptions. Lake Sturgeon also preferred to spawn at the Camp 43 Dam, and showed a residency preference towards the upper reaches of the river.

Lake Sturgeon, especially females, are known as periodic spawners (i.e., it takes females 3-7 years to fully develop their eggs after spawning; Kerr et al. 2010), and they have a late age of maturity (i.e., 12-20 years for males and 15-30 years for females; COSEWIC 2017), both of which may explain why 10 of the tagged Lake Sturgeon have not been detected at REC-7 during the spawning season in the years studied. In addition, the four years of detection data analysed in this study may not be long enough to have had all females migrate to spawn. Also, any females tagged may still be immature due to their ages as the average age of Lake Sturgeon spawners migrating to the Camp 43 Dam during the spawning season was approximately two years higher than those not migrating to the Camp 43 Dam. It is also important to note that Lake Sturgeon do have non-spawning conspecifics that migrate with the spawners (Peterson et al. 2007). Thus, some migrating Lake Sturgeon of younger ages may not be spawning and simply migrating with the spawning conspecifics. As such, individuals not migrating to the Camp 43 Dam may not have reached sexual maturation or may be experiencing spawning periodicity.

Lake Sturgeon generally have better recruitment with increased access to habitat through unimpeded river segments (Haxton 2008). Subpopulations with the largest numbers of mature individuals seem to be from habitats with longer, unimpeded river segments (i.e., Lake of the Woods – Rainy River; Upper St. Clair River, Southern Lake Huron; St. Clair River, Lake St. Clair; and the St. Lawrence River, downstream of Beauharnois Dam; COSEWIC 2017). Lake Sturgeon migrate up rivers to spawning habitats and thus require watersheds that have diverse and unobstructed habitats

(Beamesderfer and Farr 1997; Earle 2002). To support self-sustaining Lake Sturgeon populations, a barrier-free distance of 250-300 km should be maintained (Auer 1996). Thus, Lake Sturgeon recruitment in the Black Sturgeon River would likely increase with the removal of the Camp 43 Dam. Further, if the Camp 43 Dam was removed, this would increase access in the river for Lake Sturgeon from 17 km of river habitat to 67 km, up to the Camp 1 site; and give access to an additional 80% of spawning habitat (i.e., 243,000 m^2).

3.5.3 Management implications

The fate of the Camp 43 Dam has been at the forefront of debate for nearly two decades. In November 2012, the Northwest Region Planning Unit for the OMNRF put forth a summary report in favour of decommissioning the Camp 43 Dam and the installation of a new barrier at Camp 1 (OMNRF 2012). It was noted that the FMZ 9 Advisory Council reviewed all options for the Camp 43 Dam and also stated removal of the Camp 43 Dam as their preferred option. However, in 2018, responsibility of the Camp 43 Dam transferred from the OMNRF to the OMECP and Ontario Parks. In 2018-2019, an engineering review by KGS Consulting Engineers stressed the need for repairs to the dam to meet mandatory safety requirements in the *Lakes and Rivers Improvement Act*. As a result of this decision, repairs to the dam were completed in 2020.

The dam currently serves as a Sea Lamprey barrier and restricts migration of other invasive species (e.g., Pacific salmon, Rainbow Smelt, and Common Carp; Bobrowicz 2010). Further, the dam protects Northern Brook Lamprey, a species of

Special Concern, from lampricide treatments that would be implemented if the dam was removed and Sea Lampreys migrated to inhabit the upper reaches of the Black Sturgeon River. A trap and sort fishway was noted as an alternative; however, all other options were considered impractical, as were stocking, installation of artificial spawning habitats, and harvest and regulatory controls. For a trap and sort fishway, challenges such as future staffing and funding to ensure its' success were noted. Also, there was uncertainty whether the facility could pass large sturgeon during outmigration (Bobrowicz 2010), as there can be difficulties for large, bottom-oriented fish like adult Lake Sturgeon (McDougall et al. 2013). However, all Lake Sturgeon tagged by the OMNRF and released upstream of the Camp 43 Dam were detected on receivers in the Black Sturgeon River below the dam. Trap and sort fishways require manual sorting and are effective for blocking further Sea Lamprey migration; however, passage of desirable fish varies between 7 and 88% and passed fish can experience migration delays of 5 to 28 days (Pratt et al. 2009).

Conversely, the Ontario Federation of Anglers & Hunters (OFAH) have not expressed support for dam removal due to lack of evidence for success (OFAH 2017). In its decision, the OFAH cited the unknown impact of non-native species (i.e., Rainbow Trout, Pacific salmon, etc.) on native species, the unknown quality of the estimated 325,000 m^2 of potential spawning habitat noted in Sakamoto (2007) for Walleye (i.e., that was assessed by the UGLMU and determined that 75% was likely suitable spawning habitat), as well as the cost to treat the Black Sturgeon River and its main tributaries with lampricides. Although, it is important to note that before the construction of the Camp 43 Dam in 1959/1960, three of the four significant tributaries to the Black Sturgeon River located above the current dam (i.e., Mound, Mouseau, Shillabeer, and Larson) needed to be treated with lampricide (OFAH 2017).

Structural repairs to the dam were completed in 2020 and the OMECP stated they would not be proceeding with the decommissioning of the Camp 43 Dam and the installation of a new multi-purpose barrier at the Camp 1 site (CBC News 2020). In a letter to stakeholders, Ontario Parks also stated that no further steps will be carried out under the related class environmental assessment process, suggesting that further options to restore connectivity for native fishes to access additional potential spawning habitat will not be considered. The MNO opposes this decision stating that the Camp 43 Dam impedes Lake Sturgeon and Walleye from going upstream to spawn. The Thunder Bay Métis council president Kevin Muloin also said their organization feels mislead as there were already plans to remove the Camp 43 Dam to increase spawning capacity to help rehabilitate the Walleye population (Clutchey 2020). Repairs to the dam and OMNRF support for dam removal happened prior to the completion of this study. It was identified that only a small percentage (i.e., 1% to 5% from 2017 to 2020) of Walleye were potentially spawning at the Camp 43 Dam, and thus restoring river connectivity may not be as crucial for Walleye recovery as previously thought. However, the timing and extent of Lake Sturgeon movement upstream in the Black Sturgeon River towards the Camp 43 Dam indicates that further migration is likely if given access to the upper reaches of the river. Thus, if river connectivity was restored, either through the removal of the Camp 43 Dam or modification to include selective fish passage, this could lead to increased recruitment for Lake Sturgeon. As such, there is a need for passage of desirable fishes (i.e., Walleye, Lake Sturgeon, and other native species) while impeding passage of non-native species (Pratt et al. 2009; McLaughlin et al. 2013; Rahel 2013).

The Camp 43 Dam is a “lowermost barrier”, the first structure within a tributary that blocks fish passage (Zielinski et al. 2019). These lowermost barriers are a critical component of the strategy of Sea Lamprey control in the Great Lakes (Zielinski and Freiburger 2020). The use of Sea Lamprey control barriers has created tension among stakeholders that value Sea Lamprey control and connectivity of rivers for native fish passage over one another (McLaughlin et al. 2013). This debate has recently been referred to as the ‘connectivity conundrum’ (Zielinski et al. 2020). In Zielinski et al. (2019), it was noted that the GLFC is leading a project on selective and bi-directional fish passage, titled FishPass, to provide up- and downstream passage of desirable, native fishes, but restrict movement of undesirable fish (i.e., invasive Sea Lamprey). While this project is in its infancy, potential solutions such as incorporating sorting akin to recycling facilities to select for target traits of undesirable fishes to impede passage, but select for passage of desirable fishes has been promising (Zielinski et al. 2020). Although the effectiveness of this system has yet to be tested in the field, findings of this and other FishPass projects may yield potential solutions for selective fish passage at the Camp 43 Dam, where full connectivity (i.e., dam removal) would have unintended consequences (i.e., further dispersal of non-native Sea Lamprey) but selective connectivity could aid in ecosystem restoration (Zielinski et al. 2020). In a recent global review of barrier use to limit the spread of aquatic invasive species, Jones et al. (2021) noted that development of selective passage is promising but still in early stages of development. As such, selective fish passage seems like a promising alternative for the Camp 43 Dam after the decisions by the OMECP and Ontario Parks regarding no further actions regarding dam removal; however, this option may come with the price of time.

3.5.4 Further research

The observed differences in Walleye and Lake Sturgeon potential spawning locations may be due to differed life history characteristics. It is uncertain if there are historical spawning sites upstream of the Camp 43 Dam for Walleye, although it is highly likely. For Walleye, the goal of rehabilitation is to achieve a population size similar to historical values. Walleye experience early maturation and may exhibit site fidelity (Bozek et al. 2011) towards spawning locations downstream of the Camp 43 Dam as there is no imprinting of that spawning location. However, Olson et al. (1978) suggested that site fidelity in Walleye is not natally-imprinted and is learned behaviour as an adult. As the literature is divided on this topic, future research should include passing Walleye and over the Camp 43 Dam to see if there is an increase in YOY recruitment in the Black Sturgeon River from active spawning of passed individuals, spawning at upstream locations. This research could add to the literature of site fidelity in Walleye and would also show if spawning upstream of the Camp 43 Dam is possible for Walleye that enter the lower Black Sturgeon River during the spawning season. In contrast, Lake Sturgeon are long-lived and may experience natal site fidelity (Donofrio et al. 2018) to spawning locations upstream of the Camp 43 Dam, which was constructed in 1959/1960. Specifically, Lake Sturgeon are known to spawn at High Falls, upstream of the Camp 43

Dam (Bobrowicz 2010) and the OMNRF tagged Lake Sturgeon were retrieved from this location. Future research in this system for Lake Sturgeon should include passing individuals over the dam to see if spawning occurs at High Falls or other suitable spawning locations as any abundance increase in Lake Sturgeon is desirable as the Black Sturgeon River is one of only nine self-sustaining populations in Lake Superior (Auer 2003).

Further, differences in drifting of larval life stages of Walleye and Lake Sturgeon may also explain differences in spawning habitat selection. Larval Walleye hatch and drift downstream after a period of incubation (Mion et al. 1998). To maximize survival, it is critical that larval Walleye reach nursery habitat early (Zhao et al. 2009), and as such, larvae will move out of the river faster than Lake Sturgeon. In contrast, Lake Sturgeon eggs hatch and larvae stay within the interstitial spaces of the spawning habitat and absorb the yolk sac (Duong et al. 2011). Larval Lake Sturgeon have been observed to emerge up to 19 days after spawning (Bruch et al. 2016). As such, spawning habitat selection for Lake Sturgeon may require higher quality to ensure survival of larvae that take longer to leave the river than Walleye. Also, to support self-sustaining Lake Sturgeon populations, a barrier-free distance of 250-300 km should be maintained (Auer 1996). As such, Lake Sturgeon may require longer stretches of river for larvae to drift. Thus, species specific differences in life history should be considered when considering future research on Walleye and Lake Sturgeon in this system.

Another suggestion for future research is a more comprehensive assessment of the rapids upstream of the Camp 43 Dam and their suitability as Walleye and Lake

Sturgeon spawning habitat. Also, having increased sex determination in tagged Walleye would allow for more comparisons in potential sex-specific differences in Walleye. If there becomes reliable techniques for determining sex from genetics of already tagged Walleye or if future tagged Walleye can determine sex, this could be beneficial for data analysis in this system, as spawning activity is known to be different between Walleye sexes; specifically, time of arrival at spawning locations (i.e., males arrive earlier and leave later than females; Bade et al. 2019). Expansion of telemetry projects upstream of the Camp 43 Dam, genetic analysis, and gender identification will allow further elucidation of migratory movements of Walleye and Lake Sturgeon in this system, and may aid in further understanding of species specific differences.

4.0 Conclusion and Recommendations

This study has highlighted spawning habitat restoration challenges and management options in the Laurentian Great Lakes through two lenses: a narrative review assessing the feasibility of functional monitoring for sturgeon spawning habitats and through analysis of Walleye and Lake Sturgeon movement around an impassable barrier, to inform restoration planning.

The narrative review in Chapter 2 collected and synthesized studies with data for both physical habitat characteristics (i.e., water velocity and/or depth and/or substrate composition) and data for biological metrics of productivity (i.e., eggs and/or larvae) for sturgeons, including natural and artificial spawning habitats. This review showed that functional monitoring cannot be used in place of a full effectiveness monitoring program for sturgeon at this time, as the available studies vary widely in methodologies, habitat requirements for sturgeon species are highly variable, sturgeon population sizes vary widely, and sturgeon exhibit spawning periodicity making year-to-year comparisons difficult. However, a similar review on other species (i.e., Walleye or salmonids) could yield more promising results for the use of functional monitoring if there is enough data with similar methodologies or raw data to transform outcomes for comparison. If a similar review is conducted on another species, it is also recommended that data from grey literature sources be included, peak suitability of spawning habitats be determined, and estimates of mature spawning populations be included. Further, recommendations for standardized methodologies have been suggested that would aid with future reviews and meta-analyses to see if functional monitoring is feasible.

In Chapter 3, the spatial ecology of Walleye and Lake Sturgeon, two migratory species that use the Black Sturgeon River during their respective spawning seasons, was examined. Acoustic telemetry detection data were analyzed to address whether spawning habitat is limiting Walleye recovery in Black Bay, provide insight on the actual benefits of the potential removal of the Camp 43 Dam, and the rehabilitation of Lake Sturgeon in the Black Sturgeon River. For Walleye migrating in the river, it was identified that only a small percentage (<5% annually from 2017 to 2020) were potentially spawning at the Camp 43 Dam, and thus restoring river connectivity may not be as crucial for Walleye recovery as previously believed. In contrast, the timing and extent of Lake Sturgeon movement upstream in the Black Sturgeon River where all individuals migrated to the Camp 43 Dam indicates that further upstream migration by Lake Sturgeon is likely if given access to the upper reaches of the river. Thus, if river connectivity was restored, either through the removal of the Camp 43 Dam or modification to include selective fish passage, this could lead to increased recruitment for Lake Sturgeon. Research on selective fish passage is in its' infancy; however, technology developed under programs such as FishPass could yield beneficial outcomes for Sea Lamprey control barriers, including the Camp 43 Dam.

Currently, Walleye and Lake Sturgeon only have access to 80,000 m^2 of rapids in the Black Sturgeon River for spawning habitat (Sakamoto 2007). If given access to upper reaches of the river, there is a further 325,000 m^2 of rapids (Sakamoto 2007), 75 percent of which is likely suitable as spawning habitat for Walleye and Lake Sturgeon (Bobrowicz 2010). However, there are management concerns regarding dam removal as the dam currently serves as a Sea Lamprey barrier and restricts migration of other invasive species (e.g., Pacific salmon, Rainbow Smelt, and Common Carp; Bobrowicz 2010). Further, the dam protects Northern Brook Lamprey, a species of Special Concern, from lampricide treatments that would be implemented if the dam was removed and Sea Lampreys migrated to inhabit the upper reaches of the Black Sturgeon River. Currently, a decision on the fate of the Camp 43 Dam has been postponed. It was initially decided that the Camp 43 Dam would be removed; however, structural repairs were completed in 2020 and afterwards the OMECP stated they would not be proceeding with dam removal (CBC News 2020). Findings of this study have shown that dam removal would likely benefit Lake Sturgeon recovery but may not necessarily aid in Walleye recovery to the same extent.

It is uncertain if there are historical spawning sites upstream of the Camp 43 Dam, although it is highly likely. Also, the proportion of Walleye and Lake Sturgeon that would spawn on newly accessible habitat is uncertain. For Walleye, the goal of rehabilitation is to achieve a population size similar to historical values. Whereas for Lake Sturgeon, any abundance increase would be desirable. A more comprehensive assessment of the rapids upstream of the Camp 43 Dam and their suitability as Walleye and Lake Sturgeon spawning habitat is recommended. Suggested further research includes the facilitation of passing Walleye and Lake Sturgeon over the Camp 43 Dam to see if there is an increase in YOY recruitment in the Black Sturgeon River from active spawning of passed individuals. Also, having increased sex determination in tagged Walleye would allow for more comparisons in potential sex-specific differences in

Walleye, as spawning activity is known to be different between Walleye sexes (Bade et al. 2019). Expansion of telemetry projects, genetic analysis, and gender identification will allow further elucidation of migratory movements and population recovery of Walleye and Lake Sturgeon in Black Bay and the Black Sturgeon River.

One of the greatest threats to freshwater biodiversity is river fragmentation through dam construction (Dudgeon et al. 2006) that compromises migration (Winemiller et al. 2016), and can limit the potential for recovery through the loss of spawning habitat (Dumont et al. 2011; Thiem et al. 2013). Despite impacts on overall ecosystem functioning, hydropower production is expanding (Zarfl et al. 2015). Thus, determining the effectiveness of offsetting measures (DFO 2012) mandated for hydroelectric generating stations (in Canada), as well as determining if functional monitoring can be used over full-scale effectiveness monitoring programs is of importance. While there are positive impacts for native species with restored connectivity through dam removal (O'Connor et al. 2015; Birnie-Gauvin et al. 2017, 2018) there are also unintended consequences and trade-offs (i.e., further invasion of non-native species) that need to be considered (McLaughlin et al. 2013). In the Black Sturgeon River, removal of the Camp 43 Dam would lead to increased lampricide treatments to target larval stage Sea Lamprey, which may impact Northern Brook Lamprey, a species of Special Concern. This study has shown that Lake Sturgeon are likely to benefit greatly from the removal of the Camp 43 Dam but benefits for Walleye are less certain. As such, future research in selective fish passage could yield beneficial outcomes for the Camp 43 Dam; however, this option comes with the consequence of time as technology development is ongoing and effectiveness has yet to be verified in the field.

Works Cited

Addison, P.A. and S.M. Bobrowicz. 2009. Fall Walleye Index Netting Black Bay – 2002 and 2008. Upper Great Lakes Management Unit, Ontario Ministry of Natural Resources, Thunder Bay, ON. 37 pp. plus appendices.

Arthington, A.H, N.K. Dulvy, W. Gladstone, and I.J. Winfield. 2016. Fish conservation in freshwater and marine realms: status, threats and management. Aquatic Conservation: Marine and Freshwater Ecosystems 26:838-857.

Auer, N.A. 1996. Importance of habitat and migration to sturgeons with emphasis on lake sturgeon. Canadian Journal of Fisheries and Aquatic Sciences 53(S1):152-160.

Auer, N.A., and E.A. Baker. 2002. Duration and drift of larval lake sturgeon in the Sturgeon River, Michigan. Journal of Applied Ichthyology 18:557-564.

Auer, N.A. 2003. A lake sturgeon rehabilitation plan for Lake Superior. Great Lakes Fishery Commission. Special Publication 2003-02, Ann Arbor, Michigan.

Bade, A.P., T.R. Binder, M.D. Faust, C.S. Vandergoot, T.J. Hartman, R.T. Kraus, C.C. Krueger, and S.A. Ludsin. 2019. Sex-based differences in spawning behavior account for male-biased harvest in Lake Erie walleye (*Sander vitreus*). Canadian Journal of Fisheries and Aquatic Sciences 76:2003-2012.

Baril, A.-M., J.T. Buszkiewicz, P. M. Biron, Q. E. Phelps, and J. W. A. Grant. 2018. Lake sturgeon (*Acipenser fulvescens*) spawning habitat: a quantitative review. Canadian Journal of Fisheries and Aquatic Sciences 75:925-933.

Beamesderfer, R.C.P. and R.A. Farr. 1997. Alternatives for the protections and restoration of sturgeon and their habitat. Environmental Biology of Fishes 48:407-417.

Bemis, W.E. and B. Kynard. 1997. Sturgeon rivers: an introduction to acipenseriform biogeography and life history. Environmental Biology of Fishes 48:167-183.

Berglund, E., OMNRF. 2021. Unpublished data.

Bergstedt, R.A., and J.G. Seelye. 1995. Evidence for Lack of Homing by Sea Lampreys. Transactions of the American Fisheries Society 124:235-239.

Biberhofer, J. and C.M. Prokopec. 2007. Mapping and classification of submerged aquatic substrates in Black Bay, Lake Superior. NWRI Technical Note No. AEMRB-TN07-004. Environment Canada National Water Research Institute, Burlington, ON. 91 pp.

Billard, R. and G. Lecointre. 2001. Biology and conservation of sturgeon and paddlefish. Reviews in Fish Biology and Fisheries 10:355-392.

Bilotta, G.S., A.M. Milner, and I.L. Boyd. 2014. Quality assessment tools for evidence from environmental science. Environmental Evidence 3:14.

Birceanu, O. and M.P. Wilkie. 2018. Post-exposure effects of the piscicide (3-trifluoromethyl-4-nitrophenol (TFM) on the stress response and liver metabolic capacity in rainbow trout (*Oncorhynchus mykiss*). PLOS ONE 13, e0200782.

Birnie-Gauvin, K., M.H. Larsen, J. Nielsen, and K. Aarestrup. 2017. 30 years of data reveal dramatic increase in abundance of brown trout following the removal of a small hydrodam. Journal of Environmental Management 204:467-471.

Birnie-Gauvin, K., M.M. Candee, H. Baktoft, M.H. Larsen, A. Koed, and A. Aarestrup. 2018. River connectivity reestablished: effects and implications of six weir removals on brown trout smolt migration. River Research and Applications 34:548-554.

Bouckaert, E.K., N.A. Auer, E.F. Roseman, and J. Boase. 2014. Verifying success of artificial spawning reefs in the St. Clair-Detroit River System for lake sturgeon (*Acipenser fulvescens* Rafinesque, 1817). Journal of Applied Ichthyology 30:1393-1401.

Bobrowicz, S.M. 2010. Black Bay & Black Sturgeon River Native Fisheries Rehabilitation – Options Evaluation. Upper Great Lakes Management Unit, Ontario Ministry of Natural Resources, Thunder Bay, ON. 120 pp. plus appendices.

Boogaard, M.A., T.D. Bills, D.A., and Johnson, 2003. Acute toxicity of TFM and a TFM/ Niclosamide mixture to selected species of fish, including lake sturgeon (*Acipenser fulvescens*) and mudpuppies (*Necturus maculosus*), in laboratory and field exposures. Journal of Great Lakes Research 29:529–541.

Bozek, M.A., T.J. Haxton, and J.K. Raabre. 2011. Walleye and sauger habitat. Pages 133–197 in B. A. Barton, editor. Biology, management, and culture of walleye and sauger. American Fisheries Society, Bethesda, Maryland, USA.

Braun, D.C., K.E. Smokorowski, M.J. Bradford, and L. Glover. 2019. A review of functional monitoring to assess mitigation, restoration, and offsetting activities in Canada. DFO Canadian Science Advisory Secretariat Research Document 2019/057. vii + 75p.

Bruch, R.M., T.J. Haxton, R. Koenigs, A. Welsh, and S.J. Kerr. 2016. Status of Lake Sturgeon (*Acipenser fulvescens* Rafinesque 1817) in North America. Journal of Applied Ichthyology 32(S1):162-190.

Cano, T., OMNRF. 2021. Personal communication.

Caswell, N.M., D.L. Peterson, B.A. Manny, and G.W. Kennedy. 2004. Spawning by lake sturgeon (*Acipenser fulvescens*) in the Detroit River. Journal of Applied Ichthyology 20:1-6.

CBC News. 2020. 'High risk of failure' of dam on Black Sturgeon River prompts Ontario to proceed with 'critical repairs'. https://www.cbc.ca/news/canada/thunder-bay/black-sturgeon-dam-repairs-1.5513811.

Chapman, C.G., and T.A. Jones. 2010. First Documented Spawning of White Sturgeon in the Lower Willamette River, Oregon. Northwest Science 84(4):327-335.

Chiotti, J.A., J.M. Holtgren, N.A. Auer, and S.A. Ogren. 2008. Lake Sturgeon Spawning Habitat in the Big Manistee River, Michigan. North American Journal of Fisheries Management 28:1009-1019.

Clutchey, C. 2020. Opponents question need for Black Sturgeon dam. Chronicle Journal. https://www.chroniclejournal.com/news/local/opponents-question-need-for-black-sturgeon-dam/article_2d2f33aa-b96b-11ea-9b26-1351a7ed0d60.html.

Colby, P.J., and S.J. Nepszy. 1981. Variation among stocks of walleye (*Sander vitreum vitreum*): management and implications. Canadian Journal of Fisheries and Aquatic Sciences 38:1814-1831.

Colby, P.J. and D.A. Baccante. 1996. Dynamics of an experimentally exploited walleye population: sustainable yield estimate. Annales Zoologici Fennici 33:589-599.

Collaboration for Environmental Evidence. 2018. Guidelines and Standards for Evidence synthesis in Environmental Management. Version 5.0 (AS Pullin, GK Frampton, B Livoreil & G Petrokofsky, Eds) www.environmentalevidence.org/information-for-authors.

Collen, B., F. Whitton, E.E. Dyer, J.E.M. Baillie, N. Cumberlidge, W.R.T Darwall, C. Pollock, N.I. Richman, A.M. Soulsby, and M. Böhm. 2014. Global patterns of freshwater species diversity, threat and endemism. Global Ecology and Biogeography 23:40–51.

Collingsworth, P.D., D.B. Bunnell, M.W. Murray, Y-C. Kao, Z.S. Feiner, R.M. Claramunt, B.M. Lofgren, T.O. Höök, and S.A. Ludsin. 2017. Climate change as a long-term stressor for the fisheries of the Laurentian Great Lakes of North America. Reviews in Fish Biology and Fisheries 27:363-391.

Commanda, N. 2018. Lake Sturgeon (*Acipenser fulvescens*) management and status update for the Nipissing watershed – part of the Great Lakes Upper St. Lawrence Population, Ontario, Canada. M.E.S Thesis, Nipissing University, North Bay, Ontario 128 pp.

Cooke, S.J., V.M. Nguyen, K.J. Murchie, J.D. Thiem, M.R. Donaldson, S.G. Hinch, R.S. Brown, and A. Fisk. 2013. To Tag or not to Tag: Animal Welfare, Conservation, and Stakeholder Considerations in Fish Tracking Studies That Use Electronic Tags. Journal of International Wildlife Law & Policy 16:352-374.

Cooke, S.J., E.G. Martins, D.P. Struthers, L.F.G Gutowsky, M. Power, S.E. Doka, J.M. Dettmers, D.A. Crook, M.C. Lucas, C.M. Holbrook, and C.C. Krueger. 2016. A moving target – incorporating knowledge of the spatial ecology of fish into the assessment and management of freshwater fish populations. Environmental Monitoring and Assessment 188(4):239.

Cooke, S.J., V.M. Nguyen, J.M. Chapman, A.J. Reid, S.J. Landsman, N. Young, S.G. Hinch, S. Scott, N.E. Mandrack, and C.A.D. Semeniuk. 2020. Knowledge co-production: A pathway to effective fisheries management, conservation, and governance. Fisheries. https://doi.org/10.1002/fsh.10512.

COSEWIC. 2003. COSEWIC assessment and update status report on the white sturgeon *Acipenser transmontanus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 51 pp.

COSEWIC. 2004. COSEWIC assessment and update status report on the green sturgeon *Acipenser medirostris* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. Vii + 31 pp.

COSEWIC. 2011. COSEWIC assessment and status report on the Atlantic Sturgeon *Acipenser oxyrinchus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. xiii + 49 pp.

COSEWIC. 2012. COSEWIC assessment and status report on the White Sturgeon *Acipenser transmontanus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. xxvii + 75 pp.

COSEWIC. 2014. COSEWIC Annual Report. Committee on the Status of Endangered Wildlife in Canada. Ottawa. ii + 42 pp.

COSEWIC. 2015. COSEWIC Annual Report. Committee on the Status of Endangered Wildlife in Canada. Ottawa. ii + 12 pp.

COSEWIC. 2017. COSEWIC assessment and status report on the Lake Sturgeon *Acipenser fulvescens*, Western Hudson Bay populations, Saskatchewan-Nelson River populations, Southern Hudson Bay-James Bay populations and Great Lakes-Upper St. Lawrence populations in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. xxx + 153 pp.

Crutzen, P.J. 2002. Geology of mankind. Nature 415:23.

Cummins, K.W. 1962. An evaluation of some techniques for collection and analysis of benthic samples with special emphasis on lotic waters. American Midland Naturalist 67:477-504.

D'Amours, J., S. Thibodeau, and R. Fortin. 2001. Comparison of lake sturgeon (*Acipenser fulvescens*), *Stizostedion* spp., *Catostomus* spp., *Moxostoma* spp., quillback (*Carpoides Cyprinus*), and mooneye (*Hioden tergisus*) larval drift in Des Prairies River, Quebec. Canadian Journal of Zoology 79(8):1472-1489.

Dammerman, K.J., J.P. Steibel, and K.T. Scribner. 2020. Relative influences of microhabitat incubation conditions and genetic parentage effects on lake sturgeon (*Acipenser fulvescens*) offspring traits during early ontogeny. Environmental Biology of Fishes 103:1565-1581.

Deinet, S.L., McRae, and R. Freeman. 2020. The Living Planet Index (LPI) for migratory freshwater fish – Technical Report. World Fish Migration Foundation, The Netherlands.

DFO. 2012. Assessing the Effectiveness of Fish Habitat Compensation Activities in Canada: Monitoring Design and Metrics. DFO Canadian Science Advisory Secretariat Science Advisory Report 2012/060.

DFO. 2019.Science Advice on Operational Guidance on Functional Monitoring: Surrogate Metrics of Fish Productivity to Assess the Effectiveness of Mitigation and Offsetting Measures. DFO Canadian Science Advisory Secretariat Science Advisory Report 2019/042.

Dijkstra, K.D.B., M.T. Monaghan, and S.U. Pauls. 2014. Freshwater biodiversity and aquatic insect diversification. Annual Review of Entomology 59:143-163.

Donforio, M.C., K.T. Scribner, E.A. Baker, J. Kanefsky, I. Tsehaye, and R.F. Elliot. 2018. Telemetry and genetic data characterize lake sturgeon (*Acipenser fulvescens* Rafinesque, 1817) breeding ecology and spawning site fidelity in Green Bay Rivers of Lake Michigan. Journal of Applied Ichthyology 34:302-313.

Dudgeon, D., A.H. Arthington, M.O. Gessner, Z.I. Kawabata, D.J. Knowler, C. Lévêque, R.J. Naiman, A.H. Prieur-Richard, D. Soto, M.L.J. Stiassny, and C.A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81:163–182.

Dumont, P., J. D'Amours, S. Thibodeau, N. Dubuc, R. Verdon, S. Garceau, P. Bilodeau, Y. Mailhot, and R. Fortin. 2011. Effects of the development of a newly created spawning ground in the Des Prairies River (Quebec, Canada) on the reproductive success of lake sturgeon (*Acipenser fulvescens*). Journal of Applied Ichthyology 27:394-404.

Duong, T.Y., K.T. Scribner, J.A. Crossman, P.S. Forsythe, and E.A. Baker. 2011. Environmental and maternal effects on embryonic and larval developmental time until dispersal of lake sturgeon (*Acipenser fulvescens*). Canadian Journal of Fisheries and Aquatic Sciences 68:643-654.

Duong, T.Y., K.T. Scribner, P.S. Forsythe, J.A. Crossman, and E.A. Baker. 2013. Interannual variation in effective number of breeders and estimation of effective population size in long-lived iteroparous lake sturgeon (*Acipenser fulvescens*). Molecular Ecology 22:1282-1294.

Earle, S. 2002. Status of the lake sturgeon (*Acipenser fulvescens*) in Alberta. Alberta Wildlife Status Report No. 46. Alberta Sustainable Resource Development, Edmonton, AB. 30 pp.

Ellis, D.V., and M.A. Giles. 1965. The Spawning Behavior of the Walleye, *Stizostedion vitreum* (Mitchill). Transactions of the American Fisheries Society 94(4): 358-362.

Ferguson, M.M., and G.A. Duckworth. 1997. The status and distribution of lake sturgeon, *Acipenser fulvescens*, in the Canadian provinces of Manitoba, Ontario and Quebec: a genetic perspective. Environmental Biology of Fishes 48:299-309.

Fischer, J.L., J.J. Pritt, E.F. Roseman, C.G. Prichard, J.M. Craig, G.W. Kennedy, and B.A. Manny. 2018. Lake Sturgeon, Lake Whitefish, and Walleye Egg Deposition Patterns with Response to Fish Spawning Substrate Restoration in the St. Clair-Detroit River System. Transactions of the American Fisheries Society 147:79-93.

Fischer, J.L., E.F. Roseman, C. Mayer, and T. Wills. 2020. If you build it and they come, will they stay? Maturation of constructed fish spawning reefs in the St. Clair-Detroit River System. Ecological Engineering 150:105837.

Forsythe, P.S., K.T. Scribner, J.A. Crossman, A. Ragavendran, E.A. Baker, C. Davis, and K.K. Smith. 2012. Environmental and lunar cues are predictive of the timing of river entry and spawning-site arrival in Lake Sturgeon *Acipenser fulvescens*. Journal of Fish Biology 81(1):35-53.

Fox, D.A., J.E. Hightower, and F.M. Parauka. 2000. Gulf Sturgeon Spawning Migration and Habitat in the Choctawhatchee River System, Alabama-Florida. Transactions of the American Fisheries Society 129:811-826.

Flowers, H.J., W.E. Pine III, A.C. Dutterer, K.G. Johnson, J.W. Ziewitz, M.S. Allen, and F.M. Parauka. 2009. Spawning Site Selection and Potential Implications of Modified Flow Regimes on Viability of Gulf Sturgeon Populations. Transactions of the American Fisheries Society 138:1266-1284.

Friday, M., unpublished data.

Friday, M. 2004. Population characteristics of Black Sturgeon River lake sturgeon (*Acipenser fulvescens*). Technical Report 04-01. 25 pp.

Friday, M.J. 2014. Estimating the critical reproductive periods of lake sturgeon (*Acipenser fulvescens*) using daily water temperature units. Ontario Ministry of Natural Resources, Northwest Biodiversity and Monitoring Section (BAMS). Technical Report, TN-48, 7 pp.

Furlong, P., R.F. Foster, P.J. Colby and M. Friday. 2006. Black Sturgeon River dam: A barrier to the rehabilitation of Black Bay walleye. Upper Great Lakes Management Unit – Lake Superior technical report no. 06-03. Ontario Ministry of Natural Resources, Thunder Bay, ON. 27 pp.

Garcia-Moreno, J., I. Harrison, D. Dudgeon, V. Clausnitzer, W. Darwall, T. Farrell, C. Savy, K. Tockner, and N. Tubbs. 2014. Sustaining freshwater biodiversity in the Anthropocene. In The Global Water System in the Anthropocene: Challenges for Science and Governance, J. Bogardi, A. Bhaduri, J. Leentvaar, and S. Marx (eds). Springer International Publishing: Switzerland; 247–270.

Gardner, W., DFO. 2021. Personal communication.

Garner, S.R., S.M. Bobrowicz, and C.C. Wilson. 2013. Genetic and ecological assessment of population rehabilitation: walleye in Lake Superior. Ecological Applications 23(3):594-605.

Grill, G., B. Lehner, M. Thieme, B. Geenen, D. Tickner, F. Antonelli, S. Babu, P. Borrelli, L. Cheng, H. Crochetire, H. Ehalt Macedo, R. Filgueiras, M. Goichot, J. Higgins, Z. Hogan, B. Lip, M.E. McClain, J. Meng, M. Mulligan, C. Nilsson, J.D. Olden, J.J. Opperman, P. Petry, C. Reidy Liermann, L. Sáenz, S. Salinas-Rodríguez, P. Schelle, R.J.P. Schmitt, J. Snider, F. Tan, K. Tockner, P.H. Valdujo, A. van Soesbergen, and C. Zarfl. 2019. Mapping the world's free-flowing rivers. Nature 569:215-221.

Gross, M.R., J. Repka, C.T. Robertson, D.H. Secor, and W. Van Winkle. 2002. Sturgeon Conservation: Insights from Elasticity Analysis. American Fisheries Society Symposium 28:13-30.

Haddaway, N.R., B. Macura, P. Whaley, and A.S. Pullin. 2018. ROSES Flow Diagram for Systematic Reviews. Version 1.0. figshare. https://doi.org/10.6084/m9.figshare.5897389.v3.

Harkness, W.J.K., and J.R. Dymond. 1961. The lake sturgeon: the history of its fishery and problems of conservation. Ontario Department of Lands and Forests, Toronto, Ontario.

Harrison, I., R. Abell, W. Darwall, M.L. Thieme, D. Tickner, and I. Timboe. 2018. The freshwater biodiversity crisis. Science 362(6421):1368-1369.

Hartig, J.H., C. Sanders, R.J.H Wyma, J.C. Boase, and E.F. Roseman. 2018. Habitat rehabilitation in the Detroit Area of Concern. Aquatic Ecosystem Health & Management 21(4):458-469.

Haxton, T.J. 2006. Characteristics of a Lake Sturgeon Spawning Population Sampled a Half Century Apart. Journal of Great Lakes Research 32:124-130.

Haxton, T. 2008. A synoptic review of the history and our knowledge of lake sturgeon in the Ottawa River. Southern Science and Information Technical Report SSI #126, 31p.

Haxton, T.J., and C.S. Findlay. 2008. Variation in lake sturgeon abundance and growth among river reaches in a large regulated river. Canadian Journal of Fisheries and Aquatic Sciences 65:645-657.

Haxton, T., G. Whelan, and R. Bruch. 2014a. Historical biomass and sustainable harvest of Great Lake lake sturgeon (*Acipenser fulvescens* Rafinesque, 1817). Journal of Applied Ichthyology 30:1371-1378.

Haxton, T.J., M. Friday, T. Cano, and C. Hendry. 2014b. Variation in Lake Sturgeon (*Acipenser fulvescens* Rafinesque, 1817) abundance in rivers across Ontario, Canada. Journal of Applied Ichthyology 30:1371-1341.

Haxton, T.J., M. Friday, T. Cano, and C. Hendry. 2014c. Evaluation of a netting protocol to quantitatively assess lake sturgeon in Ontario rivers. Ontario Ministry of Natural Resources. 10pp + appendices.

Haxton, T.J., K. Sulak, and L. Hildebrand. 2016. Status of scientific knowledge of North American sturgeon. Journal of Applied Ichthyology 32(1):5-10.

Hayden, T.A., C.M. Holbrook, D.G. Fielder, C.S. Vandergoot, R.A. Bergstedt, J.M. Dettmers, C.C. Kreuger, and S.J. Cooke. 2014. Acoustic Telemetry Reveals Large-Scale Migration Patterns of Walleye in Lake Huron. PloS One:1-19.

Henderson, B.A., J.L. Wong, and S.J. Nepszy. 1996. Reproduction of walleye in Lake Erie: allocation of energy. Canadian Journal of Fisheries and Aquatic Sciences 53:127-133.

Heupel, M.R., J.M. Semmens, and A.J. Hobday. 2006. Automated acoustic tracking of aquatic animals: scales, design and deployment of listening stations arrays. Marine Freshwater Research 57:1-13.

Hildebrand, L.R., A. Drauch Schreier, K. Lepla, S.O. McAdam, J. McLellan, M.J. Parsley, V.L. Paragamian, and S.P. Young. 2016. Status of White Sturgeon (*Acipenser transmontanus* Richardson, 1863) throughout the species range, threats to survival, and prognosis for the future. Journal of Applied Ichthyology 32(S1):261-312.

Hilton, E.J., B. Kynard, M.T. Balazik, A.Z. Horodysky, and C.B. Dillman. 2016. Review of the biology, fisheries, and conservation status of the Atlantic Sturgeon, (*Acipenser oxyrinchus oxyrinchus* Mitchill, 1815). Journal of Applied Ichthyology 32(S1):30-66.

Horns, W.H., C.R. Bronte, T.R. Busiahn, M.P. Ebener, R.L. Eshenroder, T. Gorenflo, N. Kmiecik, W. Mattes, J.W. Peck, M. Petzold and D.R. Schreiner. 2003. Fish-community objectives for Lake Superior. Great Lakes Fishery Commission Special Publication 03-01. 78 pp.

IUCN. 2020. The IUCN Red List of Threatened Species. Version 2020-3. https://www.iucnredlist.org. Downloaded on [10 December 2020].

Ivan, L. N., E.S. Rutherford, C. Riseng, and J.A. Tyler. 2010. Density, production, and survival of walleye (*Sander vitreus*) eggs in the Muskegon River, Michigan. Journal of Great Lakes Research 36:328-337.

Jensen, A.J., and M.L. Jones. 2018. Forecasting the response of Great Lakes sea lamprey (*Petromyzon marinus*) to barrier removals. Canadian Journal of Fisheries and Aquatic Sciences 75:1415-1426.

Johnson, J.H., S.R. LaPan, R.M. Klindt, and A. Schiavone. 2006. Lake sturgeon spawning on artificial habitat in the St. Lawrence River. Journal of Applied Ichthyology 22:465-470.

Jones, P.E., J.S. Tummer, S.M. Galib, D.J. Woodford, J.B. Hume, L.G.M. Silva, R.R. Braga, C. Garcia de Leaniz, J.R.S. Vitule, J.E. Herder, and M.C. Lucas. 2021. The Use of Barriers to Limit the Spread of Aquatic Invasive Animal Species: A Global Review. Frontiers in Ecology and Evolution 9:1-19.

Jordan, G.R., E.J. Heist, P.J. Bratten, A.J. DeLonay, P. Hartfield, D.P. Herzog, K.M. Kappenman, and M.A.H. Webb. 2016. Status of knowledge of the Pallid Sturgeon (*Scaphirhynchus albus* Forbes and Richardson, 1905). Journal of Applied Ichthyology 32(S1):191-207.

Jørgensen, C., B. Ernande, Ø. Fiksen, and U. Dieckmann. 2006. The logic of skipped spawning in fish. Canadian Journal of Fisheries and Aquatic Sciences 63(1):200-211.

Kaeser, A.J., and T.L. Litts. 2010. An assessment of deadhead logs and large woody debris using side scan sonar and field surveys in streams of southwest Georgia. Fisheries 33:589-597.

Kerr, S.J., M.J. Davidson, and E. Funnel. 2010. A review of lake sturgeon habitat requirements and strategies to protect and enhance sturgeon habitat. Fisheries Policy Section, Biodiversity Branch, Ontario Ministry of Natural Resources, Peterborough, ON. 58 pp + app.

Kessel, S.T., N.E. Hussey, D.M. Webber, S.H. Gruber, J.M. Young, M.J. Smale, and A.T. Fisk. 2015a. Close proximity detection interference with acoustic telemetry: the importance of considering tag power output in low ambient noise environments. Animal Biotelemetry 3(5):1-14.

Kessel, S.T., N.E. Hussey, R.E. Crawford, D.J. Yurkowski, C.V. O'Neill, and A.T. Fisk. 2015b. Distinct patterns of Arctic Cod (*Boreogadus saida*) presence and absence in a shallow high Arctic embayment, revealed across open-water and ice-covered periods through acoustic telemetry. Polar Biology 39:1057-1068.

Kuhajda, B.R., and S.J. Rider. 2016. Status of the imperiled Alabama Sturgeon (*Scaphirhynchus suttkusi* Williams and Clemmer, 1991). Journal of Applied Ichthyology 32(S1):15-29.

Krueger, C.C., C.M. Holbrook, T.R. Binder, C.S. Vandergoot, T.A. Hayden, D.W. Hondorp, N. Nate, K. Paige, S.C. Riley, A.T. Fisk., and S.J. Cooke. 2018. Acoustic telemetry observation systems: challenges encountered and overcome in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 75(10):1755-1763.

Kynard, B., S. Bolden, M. Kieffer, M. Collins, H. Brundage, E.J. Hilton, M. Litvak, M.T. Kinnison, T. King, and D. Peterson. 2016. Life history and status of Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Journal of Applied Ichthyology 32(S1):208-248.

LaHaye, M.A., A. Branchaud, M. Gendron, R. Verdon, and R. Fortin. 1992. Reproduction, early life history, and characteristics of the spawning grounds of Lake Sturgeon (*Acipenser fulvescens*) in Des Prairies and L'Assomption rivers, near Montreal, Québec. Canadian Journal of Zoology 70:1681-1689.

Landsman, S.J., V.M. Nguyen, L.F.G. Gutowsky, J. Gobin, K.V. Cook, T.R. Binder, N. Lower, R.L. McLaughlin and S.J. Cooke. 2011. Fish movement and migration studies in the Laurentian Great Lakes: Research trends and knowledge gaps. Journal of Great Lakes Research 37(2):365–379.

Lapointe, N.W. R., S.J. Cooke, J.G. Imhof, D. Boisclair, J.M. Casselman, R.A. Curry, O.E. Langer, R.L. McLaughlin, C.K. Minns, J.R. Post, M. Power, J.B. Rasmussen, J.D. Reynolds, J.S. Richardson, and W.M. Tonn. 2014. Principles for ensuring healthy and productive freshwater ecosystems that support sustainable fisheries. Environmental Reviews 22(2):110–134.

Lawrie, A.H. 1970. The Sea Lamprey in the Great Lakes. Transactions of the American Fisheries Society 99(4):766-775.

Liermann, C.R., C. Nilsson, J. Robertson, and R.Y. Ng. 2012. Implications of Dam Obstruction for Global Freshwater Fish Diversity. BioScience 62:539-548.

Mason, G., MNR. 2012. Musquash River Spawning Bed Construction Summary Report, 20pp.

Matley, J.K., M.D. Faust, G.D. Raby, Y. Zhao, J. Robinson, T. MacDougall, T.A. Hayden, A.T. Fisk, C.S. Vandergoot, and C.C. Kreuger. 2020. Seasonal habitat-use differences among Lake Erie's walleye stocks. Journal of Great Lakes Research 46:609-621.

McAdam, S.O., J.A. Crossman, C. Williamson, I. St-Onge, R. Dion, B.A. Manny, and J. Gessner. 2017. If you build it, will they come? Spawning habitat remediation for sturgeon. Journal of Applied Ichthyology 34:258-278.

McCrum-Gardner, E. 2008. Which Is the Correct Statistical Test to Use? British Journal or Oral and Maxillofacial Surgery 46:38-41.

McDonald, D.G., and C.S. Kolar. 2007. Research to Guide the Use of Lampricides for Controlling Sea Lamprey. Journal of Great Lakes Research 33:20-34.

McDougall, C.A., C.L. Hrenchuk, W.G. Anderson, and S.J. Peake. 2013. The Rapid Upstream Migration of Pre-Spawn Lake Sturgeon following Trap-and-Transport over a Hydroelectric Generating Station. North American Journal of Fisheries Management 33:1236-1242.

McKee, G., Lakehead University. 2021. Unpublished data.

McLaughlin, R.L., E.R.B. Smyth, T. Castro-Santos, M.L. Jones, M.A. Koops, T.C. Pratt, and L.A. Vélez-Espino. 2013. Unintended consequences and trade-offs of fish passage. Fish and Fisheries 14:580–604.

Miehls, S., P. Sullivan, M. Twohey, J. Barber, and R. McDonald. 2020. The future of barriers and trapping methods in the sea lamprey (*Petromyzon marinus*) control program in the Laurentian Great Lakes. Reviews in Fish Biology and Fisheries 30:1–24.

Mion, J.B., R.A. Stein, and E.A. Marshall. 1998. River discharge drives survival of larval walleye. Ecological Applications 8:88-103.

Moser, M.L., J.A. Isreal, M. Neuman, S.T. Lindley, D.L. Erickson, B.W. McCovery Jr., and A.P. Klimley. 2016. Biology and life history of Green Sturgeon (*Acipenser medirostris* Ayres, 1854): state of the science. Journal of Applied Ichthyology 32(S1):67-86.

Neeson, T.M., M.C. Ferris, M.W. Diebel, P.J. Doran, J.R. O'Hanley, and P.B. McIntyre. 2015. Enhancing ecosystem restoration efficiency through spatial and temporal coordination. Proceedings of the National Academy of Sciences 112:6236-6241.

Neuenhoff, R.D., J.L. Withers, L.A. Davis, N.D. Markley, S. Dowell, M.L. Bartron, and J.A. Sweka. 2018. Discovery of an Undocumented Lake Sturgeon Spawning Site in the Headwaters of the Niagara River. Journal of Fish and Wildlife Management 9:266-273.

O'Connor, J.E., J.J. Duda, and G.E. Grant. 2015. 1000 dams down and counting. Science 348:496-497.

O'Connor, L.M., T.C. Pratt, T.B. Steeves, B. Stephens, M. Boogaard, and C. Kaye. 2017. In situ assessment of lampricide toxicity to age-0 lake sturgeon. Journal of Great Lakes Research 43:189–198.

OFAH. 2017. EBR Registry Number: 012-9197. Black Sturgeon River Camp 43 Dam Project – Public Inspection of Draft Environmental Study Report.

Olson, D.E., D.H. Schupp, and V. Macins. 1978. An hypothesis of homing behavior of walleyes as related to observed patterns of passive and active movements. American Fisheries Society Special Publication 11:52-57.

OMNRF. 2012. Black Bay & Black Sturgeon River Native Fisheries Rehabilitation – Decommissioning of the Camp 43 dam and construction of a multi-purpose sea lamprey barrier at Eskwanonwatin Lake. Northwest Region Planning Unit, Ontario Ministry of Natural Resources, Thunder Bay, ON. 6 pp.

Paragamian, V.L., G. Kruse, and V. Wakkinen. 2001. Spawning Habitat of Kootenai River White Sturgeon, Post-Libby Dam. North American Journal of Fisheries Management 21:22-33.

Peterson, D.L., P. Vecsei, and C.A. Jennings. 2007. Ecology and biology of the lake sturgeon: a synthesis of current knowledge of a threatened North American Acipenseridae. Reviews in Fish Biology and Fisheries 17:59-76.

Petzold, M. 2004. Proceedings of the Black Bay walleye rehabilitation workshop (January 6-8, 2004, Sault Ste. Marie, Ontario). Upper Great Lakes Management Unit, Ontario Ministry of Natural Resources, Thunder Bay, ON. 32 pp. plus appendices.

Petzold, M. 2005. Black Bay walleye rehabilitation plan. Upper Great Lakes Management Unit, Ontario Ministry of Natural Resources, Thunder Bay, ON. 17 pp.

Phelps, Q.E., S.J. Tripp, M.J. Hamel, J.Koch, E.J. Heist, J.E. Garvey, K.M. Kappenman, M.A.H. Webb. 2016. Status of knowledge of the Shovelnose Sturgeon (*Scaphirhynchus platorynchus* Raffinesque, 1820). Journal of Applied Ichthyology 32(S1):249-260.

Pikitch, E.K., C. Santora, E.A. Babcock, A. Bakun, R. Bonfil, D.O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E.D. Houde, J. Link, P.A. Livingston, M. Mangel, M.K. McAllister, J. Pope, and K.J. Sainsbury. 2004. Ecosystem-Based Fishery Management. Science 305:346-347.

Pincock, D.F. 2012. False detections: What they are and how to remove them from detection data (Document #: DOC-004691 Version 03). Vemco, Halifax, Canada.

Poytress, W.R., J.J. Gruber, J.P Van Eenennaam, and M. Gard. 2015. Spatial and Temporal Distribution of Spawning Events and Habitat Characteristics of Sacramento River Green Sturgeon. Transactions of the American Fisheries Society 144(6):1129-1142.

Pratt, T.C., L.M. O'Connor, A.G. Hallett, R.L. McLaughlin, C. Katopodis, D.B. Hayes, and R.A. Bergstedt. 2009. Balancing aquatic habitat fragmentation and control of invasive species: enhancing selective fish passage at sea lamprey control barriers. Transactions of the American Fisheries Society 138:652-665.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rahel, F.J. 2013. Intentional fragmentation as a management strategy in aquatic systems. BioScience 63:362-372.

Reid, A.J., A.K. Carlson, I.F. Creed, E.J. Eliason, P.A. Gell, P.T.J. Johnson, K.A. Kidd, T.J. MacCormack, J.D. Olden, A.J. Ormerod, J.P. Smol, W.W. Taylor, K. Tockner, J.C. Vermaire, D. Dudgeon, and S.J. Cooke. 2019. Emerging threats and persistent conservation challenges for freshwater biology. Biological Reviews 94:849-873.

Rideout, R.M., G.A. Rose, and M.P. Burton. 2005. Skipped spawning in female iteroparous fishes. Fish and Fisheries 6(1):50-72.

Rideout, R.M., and J. Tomkiewicz. 2011. Skipped spawning in fishes: more common than you might think. Marine and Coastal Fisheries 3(1):176-189.

Roseman, E.F., B. Manny, J. Boase, M. Child, G. Kennedy, J. Craig, K. Soper, and R. Drouin. 2011. Lake sturgeon response to a spawning reef constructed in the Detroit river. Journal of Applied Ichthyology 27(2):66-76.

Roseman, E.F., E. Admas, R.L. DeBruyne, J. Gostiaux, H. Harrington, K. Kapuscinski, A. Moerke, and C. Olds. 2020. Lake sturgeon (*Acipenser fulvescens*) spawn in the St. Marys River Rapids, Michigan. Journal of Great Lakes Research 46:1479-1484.

Rytwinski, T., L.K. Elmer, J.J. Taylor, L.A. Donaldson, J.R. Bennet, K.E. Smokorowski, A.K. Winegardner, and S.J. Cooke. 2019. How effective are spawning-habitat creation or enhancement measure for substrate-spawning fish? A synthesis. Canadian Technical Report of Fisheries and Aquatic Sciences 3333: vii +183 p.

Sakamoto, K. 2007. The effects of geologic history and human activity on the morphology of the Black Sturgeon River, northwestern Ontario: Potential effects on fish habitat. B.Sc. thesis. Lakehead University, Thunder Bay, ON. 67 pp. plus appendices.

Schneider, J.C., and J.H. Leach. 1977. Walleye (*Stizostedion vitreum vitreum*) Fluctuations in the Great Lakes and Possible Causes, 1800-1975. Journal of the Fisheries Research Board of Canada 34:1878-1889.

Schram, S.T., J.R. Atkinson, and D.L. Pereira. 1991. Lake Superior walleye stocks: status and management, p. 1-22. In P.J. Colby, C.A. Lewis, and R.L. Eshenroder [ed.]. Status of walleye in the Great Lakes: case studies prepared for the 1989 workshop. Great Lakes Fisheries Commission Publication 91-1.

Scott, W.B., and E.J. Crossman. 1973. Freshwater fishes of Canada. Bulletin 184, Fisheries Research Board of Canada, Ottawa, 966 pp.

Sitar, S.P., A.J. Jasonowicz, C.A. Murphy, and F.W. Goetz. 2014. Estimates of Skipped Spawning in Lean and Siscowet Lake Trout in Southern Lake Superior: Implications for stock assessment. Transactions of the American Fisheries Society 143(3):660-672.

Smith, S.H. 1972. Factors of ecologic succession in oligotrophic fish communities of the Laurentian Great Lakes. Journal of the Fisheries Research Board of Canada 29:717-730.

Smith, B.R., and J.J. Tibbles. 1980. Sea Lamprey (*Petromyzon marinus*) in Lakes Huron, Michigan and Superior: history of invasion and control, 1936-1978. Canadian Journal of Fisheries and Aquatic Sciences 37:1780-1801.

Smith, K.M., and D.K. King. 2005. Dynamics and extent of larval lake sturgeon *Acipenser fulvescens* drift in the Upper Black River Michigan. Journal of Applied Ichthyology 21(3):161-168.

Smokorowski, K.E., M.J. Bradford, K.D. Clarke, M. Clément, R.S. Gregory, and R.G. Randall. 2015. Assessing the effectiveness of habitat offset activities in Canada: Monitoring design and metrics. Canadian Technical Report of Fisheries and Aquatic Sciences 3132: vi + 48 p.

Stasko, A.B., and D.G. Pincock. 1977. Review of Underwater Biotelemetry, with Emphasis on Ultrasonic Techniques. Journal of the Fisheries Research Board of Canada 34(9):1261-1285.

Stratton, K., OMNRF. 2021. Personal Communication.

Strayer, D.L., and D. Dudgeon. 2010. Freshwater biodiversity conservation: recent progress and future challenges. Journal of the North American Benthological Society 29:344-358.

Sulak, K.J., and J.P. Clugston. 1998. Early Life History Stages of Gulf Sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society 127:758-771.

Sulak, K.J., F. Parauka, W.T. Slack, R.T. Ruth, M.T. Randall, K. Luke, M.F. Mettee, and M.E. Price. 2017. Status of scientific knowledge, recovery progress, and future research directions for the Gulf Sturgeon, *Acipenser oxyrinchus desotoi* Vladykov, 1955. Journal of Applied Ichthyology 32(S1):87-161.

Taylor, J.J., T. Rytwinski, J.R. Bennet, K.E. Smokorowski, N.W.R. Lapointe, R. Janusz, K. Clarke, B. Tonn, J. C. Walsh, and S.J. Cooke. 2019. The effectiveness of spawning habitat creation or enhancement for substrate-spawning temperate fish: a systematic review. Environmental Evidence 8:19.

Thiem, J.D., D. Hatin, P. Dumont, G. Van Der Kraak, and S.J. Cooke. 2013. Biology of lake sturgeon (*Acipenser fulvescens*) spawning below a dam on the Richelieu River, Quebec: behaviour, egg deposition, and endocrinology. Canadian Journal of Zoology 91:175-186.

UGLMU. 2017. Unpublished data.

UGLMU. 2018. Unpublished data.

Vélez-Espino, L.A., and M.A. Koops. 2009. Recovery Potential Assessment for Lake Sturgeon in Canadian Designatable Units. North American Journal of Fisheries Management 29:1065-1090.

Vitousek, P.M., H.A. Mooney, J. Lubchenco, and J.M. Melillo. 1997. Human Domination of Earth's Ecosystems. Science 277(5325):494-499.

Walker, D.J., and J.B. Alford. 2016. Mapping Lake Sturgeon Spawning Habitat in the Upper Tennessee River using Side-Scan Sonar. North American Journal of Fisheries Management 36(5):1097-1105.

Waples, J.T., B. Eadie, J.V. Klump, M. Squires, J. Cotner, and G. Mckinley. 2008. The Laurentian Great Lakes. Continental Margins: A synthesis and planning workshop.

Watson, J.M., S.M. Coghlan Jr., J. Zydlewski, D.B. Hayes, and I.A. Kiraly. 2018. Dam Removal and Fish Passage Improvement Influence Fish Assemblages in the Penobscot River, Maine. Transactions of the American Fisheries Society. 147:525-540.

Wei, Q.W., B. Kynard, D.G. Yang, X.H. Chen, H. Du, L. Shen, and H. Zhang. 2009. Using drift nets to capture early life stages and monitor spawning of the Yangtze River Chinese Sturgeon (*Acipenser sinensis*). 25(S2):100-106.

Welsh, A.B., M.R. Baerwald, M. Friday, and B. May. 2015. The effect of multiple spawning events on cohort genetic diversity of lake sturgeon (*Acipenser fulvescens*) in the Kaministiquia River. Environmental Biology of Fishes 98:755-762.

Wentworth, C.K. 1922. A scale of grade and class terms for clastic sediments. Journal of Geology 30:377-392.

Wilkie, M.P., T.D. Hubert, M.A. Boogaard, and O. Birceanu. 2019. Control of invasive sea lampreys using the piscicides TFM and niclosamide: Toxicology, successes and future prospects. Aquatic Toxicology 211:235-252.

Wilson, C.C., M. Lavender, and J. Black. 2007. Genetic assessment of walleye (*Sander vitreus*) restoration efforts and options in Nipigon Bay and Black Bay, Lake Superior. Journal of Great Lakes Research 33(Supp. 1):133-144.

Winemiller, K.O., P.B. McIntyre, L. Castello, E. Fluet-Chouinard, T. Giarrizzo, S. Nam, I.G. Baird, W. Darwall, N.K. Lujan, I. Harrison, M.L.J. Stiassny, R.A.M. Silvano, D.B. Fitzgerald, F.M. Pelicice, A.A. Agostinho, L.C. Gomes, J.S. Albert, E. Baran, M. Petrere Jr., C. Zarfl, M. Mulligan, J.P. Sullivan, C. C. Arantes, L. M. Sousa, A.A. Koning, D. J. Hoeinghaus, M. Sabaj, J. G. Lundberg, J. Armbruster, M.L. Thieme, P. Petry, J. Zuanon, G. Torrente Vilara, J. Snoeks, C. Ou, W. Rainboth, C. S. Pavanelli, A. Akama, A. van Soesbergen, and L. Sáenz. 2016. Balancing hydropower and biodiversity in the Amazon, Congo, and Mekong. Science 351:128-129.

Whitlock, M., and D. Schluter. 2009. The Analysis of Biological Data. Roberts and Co. Publishers, Greenwood Village, CO.

WWF. 2018. Living Planet Report 2018: Aiming Higher (Eds M. Grooten and R. E. A. Almond). WWF International, Gland, Switzerland.

WWF. 2020. Sturgeon: Living Fossils in Danger. https://wwf.panda.org/discover/knowledge_hub/endangered_species/sturgeon/?. [Accessed on 10 December 2020].

Zarfl, C., A.E. Lumsdon, J. Berlekamp, L. Tydecks, and K. Tockner. 2015. A global boom in hydropower dam construction. Aquatic Sciences 77:161-170.

Zhang, H., Q.W. Wei, B.E. Kynard, H. Du, D.G. Yang, and X.H. Chen. 2011. Spatial structure and bottom characteristics of the only remaining spawning area of Chinese sturgeon in the Yangtze River. Journal of Applied Ichthyology 27:251-256.

Zhao, Y., M.L. Jones, B.J. Shuter, and E.F. Roseman. 2009. A biophysical model of Lake Erie walleye (*Sander vitreus*) explains interannual variations in recruitment. Canadian Journal of Fisheries and Aquatic Sciences 66:114-125.

Zielinski, D.P., R. McLaughlin, T. Castro-Santos, B. Paudel, P. Hrodey, and A. Muir. 2019. Alternative Sea Lamprey Barrier Technologies: History as a Control Tool. Reviews in Fisheries Science & Aquaculture 27(4):438-457.

Zielinski, D.P., and C. Freiburger. 2020. Advances in fish passage in the Great Lakes basin. Journal of Great Lakes Research: https://doi.org/10.1016/j.jglr.2020.03.008.

Zielinski, D.P., R.L. McLaughlin, T.C. Pratt, R. A. Goodwin, and A.M. Muir. 2020. Single-Stream Recycling Inspires Selective Fish Passage Solutions for the Connectivity Conundrum in Aquatic Ecosystems. BioScience 70(10):872-886.

Appendix A

A.1 Introduction

Under the federal *Fisheries Act* in Canada, if a hydroelectric generating station is likely to affect productivity due to ‘harmful alteration, disruption or destruction of fish habitat’, offsets to restore degraded fish habitat may be required. For spawning habitat creation to be accepted as an offsetting measure, effectiveness of the created habitat must be monitored, and quantitative targets should be established before monitoring (Smokorowski et al. 2015). Biological productivity measures collected in this study aim to determine the efficacy of artificial spawning habitats created for Lake Sturgeon. The second field season for this project was scheduled for May-June 2020; however, due to the COVID-19 pandemic, sampling has been postponed. Initial planning for the second field season included the additional sampling of three artificial spawning shoals (i.e., two sites on the Ottawa River and one at Port Severn) installed directly downstream of hydroelectric generating stations.

A.2 Objectives

Research conducted during the 2019 field season aimed to evaluate if the creation of artificial spawning habitats are an effective offsetting measure for hydroelectric development. Specifically, egg mats were used to measure egg deposition and larval drift nets to quantify drifting larval Lake Sturgeon on a suite of artificial and natural sturgeon spawning sites. Depth measurements as well as general descriptions of substrate were noted for all sites. During the proposed second field season, the aforementioned parameters will again be collected. To add to findings from Chapter 2, physical parameters, such as water velocity, depth, and substrate size will be collected during the second field season.

It is important to evaluate the efficacy of spawning habitat offsets for substrate spawning fish and to investigate potential solutions for unavoidable impacts that hydroelectric dams and their operations cause. As Lake Sturgeon spawning requirements are well documented, offsetting measures such as the creation of artificial spawning shoals can be designed to match known preferences of water velocities, depths, and substrate composition. Specifically, within Canada, this research is in line with the goals of improving confidence in decisions made by the DFO regarding s.35 authorization/offset measures installed below hydroelectric developments. Findings from this study could help to inform the restoration of imperilled populations throughout the native range of Lake Sturgeon by evaluating the efficacy of artificial spawning habitats as an offsetting measure.

A.3 Methods

A.3.1 Study sites

Initially, the focus of the study was the Ottawa River, as it historically supported abundant Lake Sturgeon populations (Haxton 2006). The Ottawa River has been fragmented by hydroelectric development, which has been noted as a main factor in impeding their recovery (Haxton and Findlay 2008). As a result, artificial spawning shoals have been installed below hydroelectric generating stations on the Ottawa River. However, during the 2019 field season, high flooding on the Ottawa River disqualified any sampling in this system as the Ontario Transport Minister announced a ban from operating a vessel on the Ottawa River on April 27, 2019. This ban was in place until May 30, 2019. As it was unknown when the ban would be lifted, other sites were chosen for sampling. Although the ban was lifted on May 30, 2019, an alternative plan was already in place and egg deposition and larval drift were not collected on the Ottawa River in 2019.

During the 2019 field season, egg mats and larval drift nets were deployed on three natural spawning sites (Batchawana, Goulais, and Musquash rivers) and two created sites (Moon and Musquash rivers). The sites were selected based on habitat enhancements completed, known Lake Sturgeon spawning tributaries, and furthest reachable upstream locations with preferred physical characteristics. Spawning was not confirmed at the Batchawana River site, as no Lake Sturgeon eggs or larvae were collected. In the proposed second field season, five natural sites (Moon River and two sites on the Musquash and Goulais rivers) and five artificial sites [Musquash and Moon rivers (downstream of the natural sites), Port Severn, and two sites on the Ottawa River] will be sampled for eggs. For larval drift, five natural sites (Moon River and two sites on the Goulais and Musquash rivers) and four artificial sites [Moon and Musquash rivers (downstream of the natural sites) and two sites on the Ottawa River] will be sampled. The Musquash and Moon rivers both have enhanced spawning shoals; however, in both cases, there are nearby upstream natural spawning shoals that will also be sampled. Depth and general descriptions of substrate were collected at all sites.

A.3.2 Egg mats

Sampling of Lake Sturgeon eggs is temperature dependent but usually occurs from mid-May to mid-June. Egg mats are checked every other day and all deposited Lake Sturgeon eggs are removed from the mats, recorded, and preserved in ethanol. Identification of Lake Sturgeon eggs can be completed through visual identification (i.e., a distinctive dark coloured egg and large size; Scott and Crossman 1973). During the 2019 field season, 70% ethanol was used for egg preservation. In the proposed second field season, 95% ethanol will be used to account for water dilution from the sample.

In 2019, five strings with four egg mats were deployed on each field site during the spawning season. Each string consisted of four steel plates (measured 40 x 19 x 1cm), all covered with an 80 x 20 cm furnace filter wrapped around the plate. The furnace filter material was secured with five approximately 5 cm (2-inch) binder clips (two on each of the 40 cm length sides and one at the top of the steel plate). The plates were attached, using a stainless-steel quick link, to a 5m long stainless-steel cable with loops created using an oval swage sleeve. A labelled float with organization identification and contact information, as well as a float number, was attached with a stainless-steel quick link and gill net sideline to mark the location of the egg mat string.

For the proposed second field season, the protocol will vary slightly in terms of the number of egg mats per string. It is proposed that each string will only have two egg mats as opposed to the four used in 2019 for manageability. Each site will have five egg mat strings, except for sites with space limitations that will set a minimum of three egg mat strings.

A.3.3 *Larval drift nets*

To capture larval emergence, 5-10 days after spawning, drift nets are deployed at each site. Stainless steel D-frame larval drift nets (75cm across the base and 50cm high) with a knotless 1600µm mesh nylon bag (317.5cm long), and a detachable cod-end, are used to collect drifting Lake Sturgeon larvae. Each drift net is equipped with a flow meter, mounted in the opening of the net to determine the total volume of water sampled per net. Nets collect various drifting larval fish and debris moving downstream in the river. When possible, the net contents are to be sorted in the field; however, samples with larger processing times will be preserved in 95% ethanol for sorting at a later date. Larval Lake Sturgeon, other larval fish, and eggs will be collected and preserved in ethanol.

During the 2019 field season, five drift nets were originally deployed at each site. For the proposed second field season, larval drift is not being captured at all egg mat sites due to personnel limitations and the number of drift nets per site may be reduced. Also, 70% ethanol was used for larvae preservation in 2019 and in the second field season 95% ethanol will be used. The drift nets used in the 2019 field season had two variates of mesh shape. One of the mesh varieties was observed to catch larval Lake Sturgeon within them, rather than have them filtered to the cod-end. A mesh effect was determined, and in the second field season only one variation of mesh shape will be used.

A.3.4 *CTU calculations*

To estimate when larval drift will occur, CTU at each site are calculated. CTU was calculated for the Musquash, Moon, and Goulais rivers using temperature logger data from the 2019 field season. The temperature loggers collected readings at 15 minute intervals; these values were averaged to determine the average daily water temperature. CTU was also calculated for the Ottawa River, using temperature logger data collected at one hour intervals from Hydro Quebec in 2019, these values were also averaged to determine the average daily water temperature.

A.3.5 Statistical analysis

Preserved egg and larvae samples from the 2019 field season were processed and the data was validated. Egg deposition was measured by determining egg density (eggs/m^2). Following Bouckaert et al. (2014), the total number of eggs from each egg mat string was divided by the total surface area of the string (m^2). The total number of egg mat strings set once a water temperature of 10°C was reached were included in the calculation. The total number of eggs counted on a string was divided by the area of an entire egg mat string (0.304 m^2), which was calculated by taking the area of one steel plate (0.4 x 0.19 m) and multiplying the area by 4 to account for the number of plates on the string. These values were then averaged to calculate the average eggs per square metre for each site in 2019. Standard error was also calculated as a measure of variation.

Larval drift calculations are usually represented by a number per volume of water. Due to instrument limitations (i.e., 6 digit flow meters that reset counts during the sampling time), reliable flow data was not collected. Thus, we used number of larvae collected per net. As drift nets were set when larval drift was expected to start, all nets were included in the calculation. The average number of larvae (± SE) collected per drift net was calculated by taking the average of all larvae per net calculations. Drift nets were excluded from the analysis if they were deemed to not be fishing well or if the cod-end detached from the drift net.

Due to the limited number of study sites, based on the nature of Lake Sturgeon and logistics required for sampling spawning habitats, the statistical power needed for further calculations is not possible with our sample size.

A.4 Results

A.4.1 Egg deposition and larval drift data

Spawning was not confirmed at the Batchawana River site in 2019 as no Lake Sturgeon eggs or larvae were collected from that site, thus it will not be sampled during the proposed second field season. Average number of eggs per metre-squared and average number of larvae per net values, as well as average depths and substrate composition for sites with presence in 2019 are summarized in Table A.1.

Table A.1: Average number of Lake Sturgeon eggs per square metre and average larvae collected per net, as well as average depth and substrate composition at sites with presence in 2019.

Site	Site Type	Average Eggs per m^2 ± Standard Error (n)	Depth (m): Egg Sampling	Average Larvae per Net ± Standard Error (n)	Depth (m): Larvae Sampling	Substrate Composition
Musquash A	Artificial	0 ± 0 (45)	1.6	1.1 ± 0.6 (32)	2.7	Primarily boulder substrates with some cobble and bedrock with areas of gravel/cobble nearshore.
Musquash B	Natural	171.7 ± 168.7 (15)	1.4	0.4 ± 0.2 (27)	3.7	Bedrock substrates with large boulders.
Moon A	Artificial	0 ± 0 (45)	1.5	7.7 ± 1.7 (59)	2.4	Cobble and boulder substrates.
Moon B	Natural	NA	NA	1.3 ± 0.8 (6)	1.6	
Goulais	Natural	0.3 ± 0.3 (44)	3.0	0 ± 0 (50)	0.8	Primarily cobble and boulder substrates with gravel deposits in interstitial spaces.

Egg deposition was confirmed at the natural spawning shoal on the Musquash River (i.e., Musquash B) with an average of 171.71 eggs per metre squared (SE: 168.68; n = 15). Spawning was also confirmed on the Goulais River with an average of 0.30 eggs per metre squared (SE: 0.30; n = 44). The spawning shoal on the Goulais River is very large (i.e., >1km length of the river), making sampling efforts at the site difficult. As only a small portion of the spawning shoal was sampled using the egg mat strings, this value is likely an underestimate of egg density.

For larval drift, larval Lake Sturgeon were collected at both the natural and artificial spawning shoals on each of the Moon and Musquash rivers. Although larval Lake Sturgeon were collected from drift nets downstream of the artificial sites on the Musquash and Moon rivers, it is important to note that the natural spawning sites on these rivers are upstream and in close proximity, and thus there is uncertainty of where the larvae were produced. Based on our initial sampling in 2019, Lake Sturgeon spawning was not confirmed at any of the artificial spawning shoals as it cannot be confirmed if the drifting larval Lake Sturgeon captured from the artificial sites on the Moon and Musquash rivers were from the artificial site or the upstream natural site.

Depth at egg and larva sampling locations ranged from 1.4-3.0 m and 0.8-3.7 m, respectively. General descriptions of substrate composition at each site were given; gravel, cobble, boulder, bedrock, and gravel substrates were observed across all sites.

A.4.2 CTU calculations

Spawning was observed on the Musquash River on egg mats set between June 3 and 5, and temperatures reached 13°C by June 6. Similarly, larval drift was captured on

June 19 and 150 CTU was reached on the Musquash River on June 21. Drift was observed sooner than expected on the Moon River, with drift occurring on June 15 (at 88.8 CTU) and 150 CTU reached on June 21. Temperatures reached 13°C on May 29 on the Goulais River and spawning was observed on egg mats set between May 31 and June 2. Overall, CTU calculations using 13°C to represent first spawning and 150 CTU to indicate drift date, expected and observed dates for egg deposition and larval drift, shown in Table A.2 are close to each other. Thus, these data will be used for scheduling purposes in the second field season. Egg mat sampling will start prior to when the river temperature is 13°C. Larval drift will be sampled prior to reaching 150 CTU; this will be discussed to account for early drift that occurred on the Moon River. Larval drift sampling will cease either when larvae have not been observed for a set amount of days, CTU has reached 400, or when personnel limitations require the end of sampling.

Table A.2: CTU calculations representing expected spawning and drift dates compared to observed dates using 2019 temperature logger data.

Site	Date to reach 13°C	Date to achieve 150 CTU	Observed Start of Spawn	Observed Start of Drift
Ottawa*	June 9	June 26	NA*	NA*
Musquash**	June 6	June 21	June 3-5	June 19
Moon**	June 7	June 21	NA	June 15
Goulais	May 29	June 15	May 31-June 2	NA

*Note: NA – No observed spawning or drift dates. * – High flooding on the Ottawa River in Spring 2019 limited access to sites. ** – As two sites were sampled on the Musquash and Moon rivers in 2019, first observed spawning and drift dates were used as the sites are in close proximity to each other.*

Appendix B

Table B.1: Summary of studies rejected at the full-text screening stage with reasons for exclusion.

Authors	Publication Year	Title	Reason for Exclusion
Akbulut B., Zengin M., Çiftçi Y., Ustaoğlu Tiril S., Memiş D., Alkan A., Çakmak E., Kurtoğlu I.Z., Aydin I., Üstündağ E., Eroğlu O., and Serdar S.	2011	Stimulating sturgeon conservation and rehabilitation measures in Turkey: An overview on major projects (2006-2009)	Captive Breeding
Brown K.	2007	Evidence of spawning by green sturgeon, *Acipenser medirostris*, in the upper Sacramento River, California	No Physical Parameters
Bruch R.M. and Binkowski F.P.	2002	Spawning behavior of lake sturgeon (*Acipenser fulvescens*)	Outcome
Buckley J. and Kynard B.	1981	Spawning and rearing of shortnose sturgeon from the Connecticut River	Captive Breeding
Buszkiewicz J.T., Phelps Q.E., Tripp S.J., Herzog D.P., and Scheibe J.S.	2016	Documentation of lake sturgeon (*Acipenser fulvescens* Rafinesque, 1817) recovery and spawning success from a restored population in the Mississippi River, Missouri, USA	Collection Methods
Caroffino D.C., Sutton T.M., and Daugherty D.J.	2009	Assessment of the vertical distribution of larval lake sturgeon drift in the Peshtigo River, Wisconsin, USA	No Physical Parameters
Chapman F.A. and Carr S.H.	1995	Implications of early life stages in the natural history of the Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi*	Captive Breeding
Collins M.R., Cooke D., Post B., Crane J., Bulak J., Smith T.I.J., Greig T.W., and Quattro J.M.	2003	Shortnose Sturgeon in the Santee-Cooper Reservoir System, South Carolina	Collection Methods
Counihan T.D. and Chapman C.G.	2018	Relating river discharge and water temperature to the recruitment of age-0 White Sturgeon (*Acipenser transmontanus* Richardson, 1836) in the Columbia River using over-dispersed catch data	No Physical Parameters
Cox A.L., Thornton C.I., and Eriksen K.W.	2008	Effectiveness of artificial substrate in capturing and retaining sturgeon eggs	Outcome
Crossman J.A. and Hildebrand L.R.	2014	Evaluation of spawning substrate enhancement for white sturgeon in a regulated river: Effects on larval retention and dispersal	Captive Breeding
Crossman J.A., Scribner K.T., Davis C.A., Forsythe P.S., and Baker E.A.	2014	Survival and Growth of Lake Sturgeon during Early Life Stages as a Function of Rearing Environment	No Physical Parameters
Dumont P., D'Amours J., Thibodeau S., Dubuc N., Verdon R., Garceau S.,	2011	Effects of the development of a newly created spawning ground in the Des Prairies River (Quebec, Canada) on the	Lack of Data/ Information

Authors	Publication Year	Title	Reason for Exclusion
Bilodeau P., Mailhot Y., and Fortin R.		reproductive success of lake sturgeon (*Acipenser fulvescens*)	
Duncan M.S., Isely J.J., and Cooke D.W.	2004	Evaluation of shortnose sturgeon spawning in the Pinopolis Dam tailrace, South Carolina	No Physical Parameters
Duong, T.Y.; Scribner, K.T.; Crossman, J.A.; Forsythe, P.S.; and Baker, E.A.	2011	Environmental and maternal effects on embryonic and larval developmental time until dispersal of lake sturgeon (*Acipenser fulvescens*)	No Physical Parameters
Fischer J.L., Pritt J.J., Roseman E.F., Prichard C.G., Craig J.M., Kennedy G.W., and Manny B.A.	2018	Lake Sturgeon, Lake Whitefish, and Walleye Egg Deposition Patterns with Response to Fish Spawning Substrate Restoration in the St. Clair–Detroit River System	Lack of Data/ Information
Fischer, J.L., Roseman, E.F., Mayer, C., and Wills, T.	2020	If you build it and they come, will they stay? Maturation of constructed fish spawning reefs in the St. Clair-Detroit River System	Lack of Data/ Information
Gao X., Lin P., Li M., Duan Z., and Liu H.	2016	Impact of the Three Gorges Dam on the spawning stock and natural reproduction of Chinese sturgeon in Changjiang River, China	Modelling
Gillespie M.A., McDougall C.A., Nelson P.A., Sutton T., and MacDonell D.S.	2020	Observations regarding Lake Sturgeon spawning below a hydroelectric generating station on a large river based on egg deposition studies	Lack of Data/ Information
Hager C.H., Watterson J.C., and Kahn J.E.	2020	Spawning Drivers and Frequency of Endangered Atlantic Sturgeon in the York River System	Outcome
Heise R.J., Slack W.T., Ross S.T., and Dugo M.A.	2004	Spawning and Associated Movement Patterns of Gulf Sturgeon in the Pascagoula River Drainage, Mississippi	Collection Methods
Hunter R.D., Roseman E.F., Sard N.M., DeBruyne R.L., Wang J., and Scribner K.T.	2020	Genetic Family Reconstruction Characterizes Lake Sturgeon Use of Newly Constructed Spawning Habitat and Larval Dispersal	Outcome
Jay K., Crossman J.A., and Scribner K.T.	2014	Estimates of Effective Number of Breeding Adults and Reproductive Success for White Sturgeon	No Physical Parameters
Koenigs R.P., Bruch R.M., Reiter D., and Pyatskowit J.	2019	Restoration of naturally reproducing and resident riverine lake sturgeon populations through capture and transfer	Outcome
Krieger J.R. and Diana J.S.	2017	Development and evaluation of a habitat suitability model for young lake sturgeon (*Acipenser fulvescens*) in the north channel of the St. Clair River, Michigan	Lack of Data/ Information
Krieger J.R., Young R.T., and Diana J.S.	2018	Evaluation and Comparison of a Habitat Suitability Model for Postdrift Larval Lake Sturgeon in the St. Clair and Detroit Rivers	Lack of Data/ Information
Kynard B., Pugh D., Parker T., and Kieffer M.	2011	Using a semi-natural stream to produce young sturgeons for conservation stocking: Maintaining natural selection during spawning and rearing	Outcome

Authors	Publication Year	Title	Reason for Exclusion
Lahaye M., Branchaud A., Gendron M., Verdon R., and Fortin R.	1992	Reproduction, early life history, and characteristics of the spawning grounds of the lake sturgeon (*Acipenser fulvescens*) in Des Prairies and L'Assomption Rivers, near Montreal, Quebec	Lack of Data/ Information
Lawrence D.A., Elliott R.F., Donofrio M.C., and Forsythe P.S.	2020	Larval lake sturgeon production and drift behaviour in the Menominee and Oconto Rivers, Wisconsin	Lack of Data/ Information
Li, Y.H., Kynard, B., Wei, Q.W., Zhang, H., Du, H., and Li, Q.K.	2013	Effects of substrate and water velocity on migration by early-life stages of kaluga, *Huso dauricus* (Georgi, 1775): an artificial stream study	Outcome
Marchant S.R. and Shutters M.K.	1996	Artificial substrates collect gulf sturgeon eggs	Collection Methods
Marranca J.M., Welsh A.B., and Roseman E.	2015	Genetic effects of habitat restoration in the Laurentian Great Lakes: An assessment of lake sturgeon origin and genetic diversity	Outcome
McCabe, G.T. and Tracy, C.A.	1994	Spawning and early-life history of white sturgeon, *Acipenser transmontanus*, in the lower Columbia River	Lack of Data/ Information
Nichols S.J., Kennedy G., Crawford E., Allen J., French III J., Black G., Blouin M., Hickey J., Chernyák S., Haas R., and Thomas M.	2003	Assessment of Lake Sturgeon (*Acipenser fulvescens*) Spawning Efforts in the Lower St. Clair River, Michigan	Lack of Data/ Information
Paragamian V.L. and Wakkinen V.D.	2002	Temporal distribution of Kootenai River white sturgeon spawning events and the effect of flow and temperature	No Physical Parameters
Parsley M.J., Beckman L.G., and McCabe G.T., Jr.	1993	Spawning and rearing habitat use by white sturgeons in the Columbia River downstream from McNary dam	Outcome
Parsley M.J. and Kappenman K.M.	2000	White sturgeon spawning areas in the lower Snake River	Lack of Data/ Information
Perrin C.J., Rempel L.L., and Rosenau M.L.	2003	White sturgeon spawning habitat in an unregulated river: Fraser River, Canada	Lack of Data/ Information
Seesholtz A.M., Manuel M.J., and Van Eenennaam J.P.	2015	First documented spawning and associated habitat conditions for green sturgeon in the Feather River, California	No Physical Parameters
Smith A., Smokorowski K.E., and Power M.	2017	Spawning lake sturgeon (*Acipenser fulvescens* Rafinesque, 1817) and their habitat characteristics in Rainy River, Ontario and Minnesota	Lack of Data/ Information
Smith J.A., Flowers H.J., and Hightower J.E.	2015	Fall Spawning of Atlantic Sturgeon in the Roanoke River, North Carolina	No Physical Parameters
Smith K.M. and King D.K.	2005	Dynamics and extent of larval lake sturgeon *Acipenser fulvescens* drift in the Upper Black River, Michigan	Lack of Data/ Information

Authors	Publication Year	Title	Reason for Exclusion
Taylor A.D. and Litvak M.K.	2017	Timing and Location of Spawning Based on Larval Capture and Ultrasonic Telemetry of Atlantic Sturgeon in the Saint John River, New Brunswick	Collection Methods
Tripp S.J., Phelps Q.E., Colombo R.E., Garvey J.E., Burr B.M., Herzog D.P., and Hrabik R.A.	2009	Maturation and reproduction of shovelnose sturgeon in the middle Mississippi River	Captive Breeding
Usvyatsov, S., Picka, J., Taylor, A., Watmough, J., and Litvak, M.K.	2013	Timing and Extent of Drift of Shortnose Sturgeon Larvae in the Saint John River, New Brunswick, Canada	Lack of Data/ Information
Verdon R., Guay J.C., LaHaye M., Simoneau M., Côté-Bherer A., Ouellet N., and Gendron M.	2013	Assessment of spatio-temporal variation in larval abundance of lake sturgeon (*Acipenser fulvescens*) in the Rupert River (Quebec, Canada), using drift nets	Modelling
Veshchev P.V.	1998	Influence of principal factors on the efficiency of natural reproduction of the Volga stellate sturgeon *Acipenser stellatus*	No Physical Parameters
Wei Q.W., Kynard B., Yang D.G., Chen X.H., Du H., Shen L., and Zhang H.	2009	Using drift nets to capture early life stages and monitor spawning of the Yangtze River chinese sturgeon (*Acipenser sinensis*)	No Physical Parameters
Wu J., Wang C., Zhang S., Zhang H., Du H., Liu Z., and Wei Q.	2017	From continuous to occasional: Small-scale natural reproduction of Chinese sturgeon occured in the Gezhouba spawning ground, Yichang, China	Language
Xiao, H. and Duan, Z.H.	2011	Hydrological and water chemical factors in the Yichang reach of the Yangtze River pre- and post-impoundment of the Three Gorges Reservoir: consequences for the Chinese sturgeon *Acipenser sinensis* spawning population (a perspective)	Modelling
Zhang H., Wei Q.W., Kynard B.E., Du H., Yang D.G., and Chen X.H.	2011	Spatial structure and bottom characteristics of the only remaining spawning area of Chinese sturgeon in the Yangtze River	Outcome
Zhuang P., Kynard B., Zhang L., Zhang T., and Cao W.	2002	Ontogenetic behavior and migration of Chinese sturgeon, *Acipenser sinensis*	Captive Breeding